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Evaluating Direct Filtration as an Alternative to Conventional Carbon-Based Advanced Treatment for Indirect Potable Reuse

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EVALUATING DIRECT FILTRATION AS AN ALTERNATIVE TO CONVENTIONAL

CARBON-BASED ADVANCED TREATMENT FOR INDIRECT POTABLE REUSE

by

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ABSTRACT

EVALUATING DIRECT FILTRATION AS AN ALTERNATIVE TO CONVENTIONAL
CARBON-BASED ADVANCED TREATMENT FOR INDIRECT POTABLE REUSE

Savannah M. Flemmer
Old Dominion University, 2023
Director: Dr. Gary Schafran

Hampton Roads Sanitation District (HRSD) is recharging purified wastewater into the Potomac Aquifer via the Sustainable Water Initiative for Tomorrow (SWIFT) project. Conventional SWIFT treatment applies coagulation, flocculation, sedimentation, ozonation, biofiltration, granular activated carbon (GAC) adsorption, and ultraviolet disinfection to secondary effluent to produce water that meets drinking water standards for managed aquifer recharge. HRSD is considering implementing direct filtration as an alternative to conventional treatment for two additional SWIFT facilities. Direct filtration presents an opportunity for significant cost savings by eliminating sedimentation, shortening flocculation detention time, and reducing coagulant usage. Without upstream removal of solids and organics, however, biofilters may have difficulty meeting turbidity requirements, and downstream GAC contactors may require more frequent media replacement, potentially increasing operating costs more than estimated savings. Additionally, a lower coagulant dose may not be sufficient to meet permitted phosphorus discharge requirements for when treated water is diverted to a receiving surface water. As a result, a pilot study was developed comparing conventional and direct filtration under variable operating conditions to determine the feasibility of direct filtration for HRSD's York River Treatment Plant (YRTP) and Nansemond Treatment Plant (NTP).

Conventional and direct filtration pilot operations were evaluated for both YRTP tertiary denitrification filter effluent and NTP secondary clarifier effluent. At a 3.8 gpm/sf loading rate and 10-minute empty bed contact time (EBCT), direct filtration achieved filter effluent turbidity

less than 0.15 NTU with a mean filter runtime of 35 hours for YRTP, while a mean runtime of 19 hours was achieved for NTP, a more turbid source water. In comparison, conventional treatment for YRTP and NTP achieved considerably longer mean filter runtimes of 105 and 65 hours, respectively. Mean total organic carbon (TOC) removal efficiency through direct filtration was comparable for both source waters, 35% for YRTP and 34% for NTP. Conventional treatment demonstrated greater TOC removal, 41% for YRTP and 44% for NTP. At a 2.5 gpm/sf loading rate and 15-minute EBCT, NTP direct filtration achieved longer filter runs (43 hours) and enhanced TOC removal (39%). Under similar operating conditions for NTP conventional treatment, mean filter runtime increased to 128 hours and mean TOC removal increased to 48%. Direct filtration with 0.8 mg-Al/L of aluminum chlorohydrate (ACH) addition achieved sufficient phosphorus removal for YRTP but not for NTP. Aluminum sulfate achieved more phosphorus removal per unit aluminum than ACH but resulted in shorter filter runs and less efficient TOC removal for both conventional and direct filtration. Direct filtration managed spikes in total suspended solids and turbidity up to 49 mg/L and 8.7 NTU, respectively. The pilot study results demonstrated direct filtration is a feasible alternative to conventional treatment, while its implementation is dependent on a cost-benefit analysis.

To the serendipitous encounter

that led me down the path of water research.

ACKNOWLEDGMENTS

Writing a thesis is hard. But it is easy to recognize how valuable an experience working on this project has been. And that value primarily comes from the people I met along the way.

I would like to give special thanks to Charles Bott for the job opportunity. Thank you for fostering such an important, exciting, and successful applied research program and for demonstrating just how productive one human can be. I also want to thank Charles and my other committee members, Dr. Gary Schafran and Dr. Mujde Erten-Unal, for the thoughtful review, guidance, and encouragement throughout this thesis.

It takes a village to raise a pilot plant. From research planning to pilot operations to sampling and analysis, so many individuals contributed to the success of this project. A special shoutout goes to Mack Pearce for his time building the pilot plant, leading the York River pilot study, and playing an integral role in the direction of this research. I also want to thank him and Samantha Hogard for their brilliant problem-solving skills and for coming to the rescue for all things pilot troubleshooting.

A big thank you to the rest of my colleagues-turned-friends at the SWIFT Research Center not only for their help on this project but especially for the fond memories and for making work feel like such a special place to be. I also want to recognize the rest of the HRSD team, laboratory staff, and supporting consultants for their invaluable contribution to this project. None of this would be possible without the combined effort of these individuals, and I feel fortunate to have had the opportunity to work alongside each and every one of them.

Last but certainly not least, I would like to thank my wonderful family for their endless love and support.

NOMENCLATURE

ACH	Aluminum chlorohydrate
Al	Aluminum
Alum	Aluminum sulfate
AOC	Assimilable organic carbon
BAF	Biofilter
BDOC	Biodegradable dissolved organic carbon
°C	degrees Celsius
$CaCO_3$	Calcium carbonate
CF	Conventional filter effluent
DF	Direct filter effluent
DFE	Denitrification filter effluent
DOC	Dissolved organic carbon
EBCT	Empty-bed contact time
EPA	Environmental Protection Agency
FE	Flocculation effluent
FSE	Flocculation/sedimentation effluent
ft	Foot
G	Velocity gradient
GAC	Granular activated carbon
gal	Gallon
gpm	Gallons per minute
H_2O_2	Hydrogen peroxide
H_3PO_4	Phosphoric acid
hr	Hour
HRSD	Hampton Roads Sanitation District
HRT	Hydraulic retention time
MCL	Maximum contaminant level
mg/L	Milligrams per Liter
MGD	Million gallons per day

min	Minute
mm	millimeter
NDMA	N-Nitrosodimethylamine
NPDES	National Pollutant Discharge Elimination System
NTP	Nansemond Treatment Plant
NTU	Nephelometric Turbidity Unit
O_3	Ozone
OP	Orthophosphate
Ops	Operations
P	Phosphorus
s	Second
SCE	Secondary clarifier effluent
scfm	Standard cubic feet per minute
sf	Square foot
SWIFT	Sustainable Water Initiative for Tomorrow
SWTR	Surface Water Treatment Rule
TOC	Total organic carbon
TP	Total phosphorus
TSS	Total suspended solids
UFRV	Unit filter run volume
UIC	Underground Injection Control
μm	micrometer
UV	Ultraviolet
YRTP	York River Treatment Plant

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CHAPTER I

INTRODUCTION

Background and Motivation

In a world facing water scarcity and population growth, potable reuse has emerged as a potential solution to water supply challenges. Advanced treatment technologies continue to evolve to meet water quality standards at lower costs for diverse applications. Membrane-based treatment with reverse osmosis and advanced oxidation has proven to be effective at producing high-quality drinking water from treated wastewater. Alternatively, carbon-based treatment with ozone and biofiltration offers a less energy-intensive process that produces comparable final effluent and eliminates the need for brine disposal that is required with reverse osmosis. While membrane-based treatment achieves more efficient total organic carbon (TOC) removal, carbon-based treatment may be preferred for applications where treated water is injected into the subsurface because the final effluent may be more compatible with groundwater chemistry (Vaidya et al., 2019). Ozone-biofiltration systems traditionally follow conventional drinking water design, including an upstream clarification step to avoid overloading filters and to achieve additional contaminant and pathogen removal.

Unlike conventional treatment, direct filtration does not include a clarification step prior to filtration (Figure 1). In direct filtration, coagulation is used to form filterable pin-flocs instead of larger settleable flocs, therefore requiring lower coagulant doses and shorter detention times than conventional treatment. With fewer and smaller concrete tanks and less coagulant usage, direct filtration can significantly reduce capital costs, chemical costs, and site footprint. In the context

of reuse, this potential cost-savings alternative could eliminate the need for a separate solids handling facility and operations. However, compared to conventional filters, direct filters may require more frequent backwashing, give lower unit filter run volumes, and remove less organic matter. Without a sedimentation step, direct filtration may only be suitable for low turbidity source waters. Direct filtration has been researched and implemented in the drinking water sector for several decades (Culp, 1977). There is, however, a gap in literature on direct filtration for carbon-based advanced treatment for potable reuse applications.

Conventional Treatment

Flocculation/Sedimentation → Ozonation → Biofiltration → GAC Adsorption → UV Disinfection

Direct Filtration

Ozonation → Flocculation → Biofiltration → GAC Adsorption → UV Disinfection
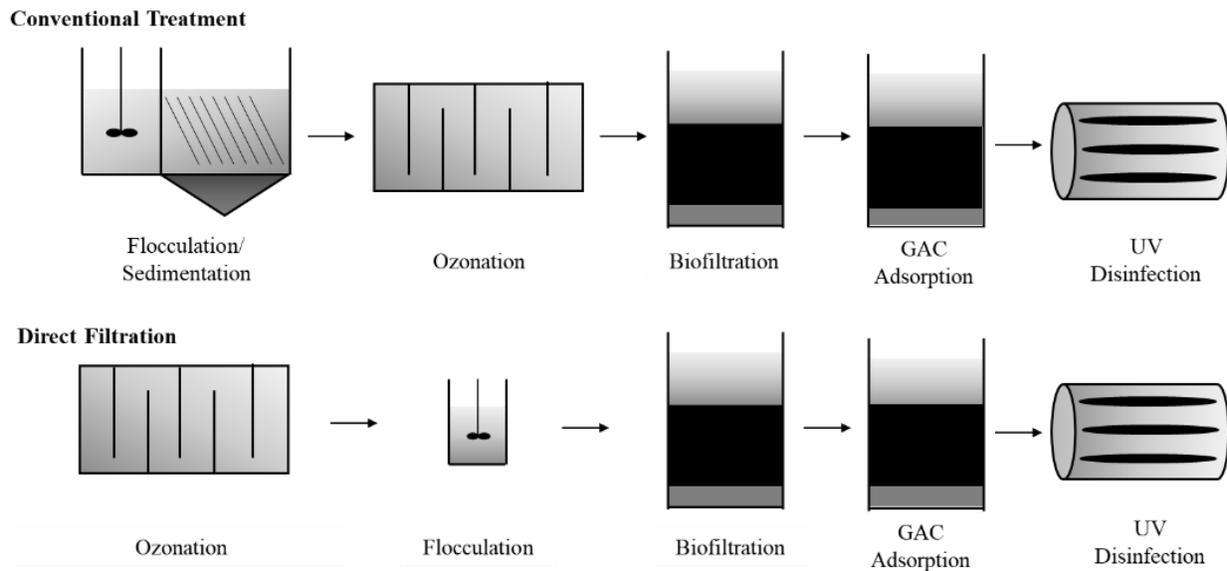
Figure 1: Process Flow Diagrams for Conventional Treatment and Direct Filtration

Hampton Roads Sanitation District (HRSD) is considering direct filtration as an alternative to conventional carbon-based advanced treatment for their indirect potable reuse project, the Sustainable Water Initiative for Tomorrow (SWIFT). HRSD is a wastewater utility in southeastern

Virginia operating 14 water reclamation facilities with a total average capacity of 150 million gallons per day (MGD). Through SWIFT, HRSD will apply advanced treatment technologies to treated wastewater to produce high-quality drinking water for recharge into the Potomac Aquifer. This effort will reduce nutrient loading into the Chesapeake Bay, replenish the groundwater supply, protect against saltwater intrusion, and slow or reverse land subsidence (HRSD, n.d.). Since 2018, HRSD has demonstrated successful treatment and recharge operations at the 1-MGD SWIFT Research Center (Hogard et al., 2021). The demonstration facility receives secondary effluent and applies coagulation, flocculation, sedimentation, ozonation, biofiltration, granular activated carbon (GAC) adsorption, and ultraviolet (UV) disinfection for managed aquifer recharge (Figure 1). The first full-scale SWIFT facility will be located at James River Treatment Plant and will implement a similar conventional carbon-based advanced treatment process. The second full-scale SWIFT facility will be located at Nansemond Treatment Plant, with the possibility of additional SWIFT facilities at some of HRSD's other major treatment plants. Current research efforts include optimizing the existing conventional carbon-based system for treating different wastewater plant effluents to meet all drinking water maximum contaminant limits and health advisory limits.

As a wastewater utility, HRSD complies with the Clean Water Act's National Pollutant Discharge Elimination System[1] (NPDES) and the Environmental Protection Agency's (EPA) Chesapeake Bay Total Maximum Daily Load[2], governed by the Virginia Department of Environmental Quality[3]. Managed aquifer recharge through SWIFT is regulated under the Safe Drinking Water Act's Underground Injection Control (UIC) program[4], implemented by EPA

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[1] 33 U.S. Code § 1342
[2] CWA § 303
[3] 33 U.S. Code § 1313, 9VAC25-720, 9VAC25-820
[4] 40 CFR §144

Region 3[5]. Regulations referenced in this study include UIC filter effluent turbidity and TOC limits and NPDES phosphorus limits. Individual filter effluent turbidity must be below 0.15 NTU 95% of the time and never greater than 0.30 NTU in two consecutive 15-minute measurements, and TOC must meet a 4 mg/L monthly average or 5 mg/L maximum instantaneous value[6]. When recharge operations are offline, SWIFT treatment processes may be used as tertiary wastewater treatment, and to maintain compliance with nutrient discharge regulations, filter effluent total phosphorus (TP) must be below 0.3 mg/L-P as an annual average[7]. Conventional SWIFT treatment achieves sufficient turbidity, TOC, and TP removal, as demonstrated at the 1-MGD SWIFT Research Center. While eliminating the sedimentation step from the treatment process has the potential to provide significant cost-savings, it is uncertain whether direct filtration can reliably achieve treatment goals.

Project Objectives

A pilot study was developed to determine whether direct filtration could be a viable alternative to conventional treatment for HRSD's SWIFT project. A pilot plant with two parallel treatment trains was configured to compare conventional treatment and direct filtration for both secondary and tertiary effluents. The objectives of this study were to assess whether direct filtration could achieve (1) filter effluent turbidity requirements without excessive backwashing, (2) sufficient TOC removal without driving up the cost of GAC regeneration, and (3) phosphorus

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[5] 40 CFR §145
[6] VAS5B170028617
[7] VA Code § 62.1-44.19:14

removal requirements despite lower coagulant doses. This study also evaluated the range of influent water quality suitable for direct filtration.

The two source waters tested on the pilot were tertiary denitrification filter effluent from HRSD's York River Treatment Plant (YRTP) and secondary clarifier effluent from HRSD's Nansemond Treatment Plant (NTP). Conventional treatment and direct filtration pilot operations were compared for each source water with respect to three key parameters (filter runtimes, TOC removal, and TP removal).

CHAPTER II

LITERATURE REVIEW

Direct Filtration

Direct filtration emerged in the mid-1900s as a potential cost-savings alternative to conventional drinking water treatment and has been extensively studied and applied in the drinking water sector (Culp, 1977). Unlike conventional treatment, direct filtration does not have a clarification step prior to filtration, therefore reducing capital costs, operating costs, and site footprint associated with sedimentation and solids handling. Compared to the larger settleable flocs formed in conventional treatment, filterable pin-flocs formed in direct filtration require lower coagulant doses and shorter mixing times. Culp (1977) estimated direct filtration can provide up to 30% capital cost savings from flocculation, sedimentation, and solids handling structures and 10-30% chemical cost savings. However, without upstream removal of solids, filters may need more frequent backwashing, reducing final effluent production. Also, the lower detention time across the treatment process increases the risk of treatment upsets from rapid changes in source water quality or improper coagulant doses, emphasizing the need for a robust control system (Bryant & Brailey, 1980). Numerous drinking water plants employ direct filtration at a range of capacities, such as the 3-MGD City of Charlevois Water Treatment Facility in Michigan, the 70-MGD Griswold Purification Facility in Colorado, and the 475-MGD Seymour-Capilano Filtration Plant in Canada (Charlevoix, n.d.; Aurora Colorado, n.d.; Water Technology, n.d.). Applications in potable reuse, however, are limited.

Existing research in direct filtration for drinking water treatment provides insight for its potential use in the potable reuse sector. Direct filtration for drinking water treatment gained

popularity in the late 1900s and research efforts apparently slowed down in the 2000s, based on the quantity of relevant published literature during these time periods. A lack of recent publications could be explained in part by a well-developed understanding of direct filtration for drinking water applications.

Direct filtration is generally suitable only for source waters low in turbidity and particulate matter so that filters can achieve sufficiently low effluent turbidity and operate without excessive backwashing. However, the range of acceptable source water quality is unique to each application. While successful reductions of raw water turbidity greater than 40 NTU have been reported, many of these plants produced final effluent turbidity greater than 0.15 NTU due to varying national regulations (Wagner & Hudson, 1982). Therefore, acceptable source water quality is dependent on effluent turbidity requirements, and direct filtration of these higher turbidity source waters may not produce effluent compliant with UIC standards. An early survey of existing plants in North America suggested direct filtration may not be feasible for source waters with color and turbidity greater than 25 units but may be possible for source waters with turbidity greater than 100 NTU if color is significantly low (Culp, 1977). A later study found that direct filtration may only work for source waters with turbidity less than 10 NTU but can handle more turbid waters if color and algal content are sufficiently low (McCormick & King, 1982). In drinking water treatment, coagulants are typically dosed based on organic content, and coagulation alters particle characteristics in the filter influent. Pernitsky & Edzwald (2006) concluded that direct filtration may not be feasible for low turbidity source waters if TOC is greater than 3 mg/L after observing rapid head loss development in a direct filtration test. Overall, direct filtration is generally limited to low turbidity source waters, but there is not a well-defined range of acceptable water quality. Pilot testing is

frequently used to determine whether direct filtration is practical for specific applications, and there is limited research on direct filtration of treated wastewater for potable reuse.

Coagulation and Flocculation

Direct filtration typically involves coagulation and flocculation to aggregate primary particles for better removal through filtration. Coagulation in direct filtration is used to form small filterable flocs instead of the larger settleable flocs removed in conventional treatment. Floc size is dependent on particle characteristics and the types of coagulants, coagulant doses, mixing intensities, and detention times. Direct filtration is designed to achieve comparable turbidity removal to conventional treatment at significantly lower coagulant doses. Compared to a full-scale conventional treatment plant, a direct filtration pilot plant receiving the same source water achieved similar turbidity removal at less than one-third of the coagulant dose (Amirsardari et al., 1998).

When treating low turbidity source waters, coagulant selection may be the most important factor for particle removal (Tanaka & Pirbazari, 1986). Coagulant types include metal-salts, cationic polymers, or a combination of both. In studies by Weng et al. (1986) and Eyvaz et al. (2010), an increase in metal-salt coagulant doses resulted in greater turbidity removal and faster head loss accumulation, which led to shorter filter runs. Hutchison (1976) found that aluminum sulfate (alum) doses from 0.35-1.8 mg-Al/L produced acceptable filter effluent turbidity and suggested that direct filtration might only be practical for water with a coagulant demand of less than 1.4 mg-Al/L alum due to rapid head loss accumulation observed at higher doses. Pernitsky & Edzwald (2006) observed that for a source water with low TOC (3.1 mg/L), low turbidity (0.8

NTU), and low alkalinity (<30 mg/L-CaCO$_3$), an alum dose of 1.5 mg-Al/L reduced filter effluent turbidity to 0.1 NTU but with rapid head loss development of 0.9 ft/hr.

Due to head loss development concerns, metal-salts are not always effective as sole coagulants for direct filtration. Polymers are preferred over metal-salts for these applications because polymers can neutralize charge without forming metal hydroxide precipitates and can lead to longer filter runs with less backwash solids (Bolto & Gregory, 2007). Cationic polymers allow deeper penetration into the direct filter bed and are commonly used at doses ranging from 0.1-5.0 mg/L (Culp, 1977). In a pilot study by Burns et al. (1984), alum as a sole coagulant (0.8 mg-Al/L) reduced turbidity from 8.1 NTU to 1.1 NTU, while the same dose of alum with polymer addition (1.5 mg/L) reduced turbidity to 0.16 NTU. Hutchison (1976) found that a small dose of alum (0.2 mg-Al/L) was required to supplement polymer to reduce source water turbidities less than 5 NTU to below 1 NTU. However, for source waters with turbidity greater than 5 NTU, cationic polymer as a sole coagulant was optimal. Research by Edzwald et al. (1986) showed polymer demand was significantly influenced by TOC concentrations ranging from 2.7 to 15 mg/L and not significantly affected by turbidities ranging from 1.2 to 6.5 NTU. For source waters with moderate to high organic content, optimal coagulant dose is determined by the concentration of dissolved organic carbon (DOC) (Becker & O'Melia, 2001). For high DOC source waters, high polymer demand may result in rapid head loss development, making direct filtration costly and impractical. In such cases, a low dose of metal-salt coagulant can be used in addition to the cationic polymer (Bolto & Gregory, 2007; Rebhun et al., 1984). Polymer as a sole coagulant may also not be effective in applications requiring phosphorus removal, where metal-salt addition would be necessary to achieve precipitation of phosphates to be removed in subsequent filtration.

Numerous studies investigated the effects of polymer dose and properties on direct filter performance. Filter effluent water quality improved with increasing polymer doses ranging from 1 to 5 mg/L, although performance varied for different types of polymers (Hutchison, 1976). Anionic floc-aid polymers are not recommended for direct filtration (Stump et al, 1979). Researchers observed a correlation between filter performance and polymer molecular weight, with turbidity removal and head loss accumulation increasing with increasing molecular weight (Stump et al., 1979; Yeh & Gosh, 1981). An optimal filter run attains simultaneous turbidity and head loss breakthrough, which may be achieved with a low to medium weight polymer. Rebhun et al. (1984) suggested using a high molecular weight polymer such as polyDADMAC for direct filtration to neutralize charge without forming large bridges. However, Bolto & Gregory (2007) found polymer molecular weight had no significant influence on direct filter performance. In addition to floc-aid polymers, filter aid polymers can enhance direct filter performance. Culp (1977) recommended using a nonionic or slightly anionic filter aid polymer at a low dose (0.05-0.50 mg/L). In a study by Logsdon et al. (1993), filter aid polymer significantly influenced filter runs, where a low dose resulted in early turbidity breakthrough and a higher dose caused rapid head loss.

Flocculation mixing speeds and detention times also influence downstream filter operations. Flocculation enhances turbidity removal and reduces head loss accumulation of direct filtration through pin-floc formation (Edzwald et al., 1986). Recommended mixing speeds and detention times, however, vary significantly in literature. Culp (1977) reported several successful direct filtration plants that operated either with only rapid mix basins or with rapid mix and flocculation basins, with detention times ranging from 15 seconds to 5 minutes. Yeh & Gosh (1981) recommended a flocculation detention time of 3 to 8 minutes, with velocity gradients from

300 to 650 s^{-1}, to avoid potential polymer shearing from longer mixing exposure. For source waters with high concentrations of suspended solids, rapid mix alone may be sufficient. Stump et al. (1979) also recommended rapid mix velocity gradients from 300 to 600 s^{-1}, while Hutchison (1976) suggested lower rapid mix velocity gradients of 20 to 100 s^{-1} because 300 s^{-1} formed smaller flocs that passed through the filter. In that study, flocculation detention times ranging from 4.5 to 28 minutes showed comparable head loss development, while early turbidity breakthrough occurred for detention times greater than 4.5 minutes (Hutchison, 1976). Alternatively, Treweek (1979) proposed that the minimum flocculation detention time for sufficient floc formation is 7 minutes. Monscvitz et al. (1978) reported that a flocculation detention time of 20 to 30 minutes enhanced turbidity removal and slowed head loss accumulation. Tate et al. (1977) found no significant difference between 13- and 26-minute flocculation detention times. With variable recommendations offered in literature, the effects of flocculation design criteria on filter operations ultimately depend on source water characteristics and other operating parameters and can be determined through pilot testing.

Ozonation

Ozone is a powerful oxidant and disinfectant used in advanced treatment to eliminate harmful compounds and improve color, taste, and odor of treated effluents. The highly reactive, unstable ozone molecule consists of three oxygen atoms and reacts with contaminants through direct oxidation or decomposition into hydroxyl radicals. Upon injection into the water, ozone readily inactivates viruses and bacteria, breaks down trace organic compounds, and oxidizes dissolved metals. Ozonation also enhances downstream biofiltration by promoting biological growth through increased dissolved oxygen content and by transforming dissolved organic matter

into more biodegradable forms. As a result, ozonation is an important factor in conventional treatment and direct filtration operations.

Pre-ozonation influences filter runtimes by improving turbidity removal and potentially affecting head loss development. Tobiason et al. (1992) evaluated the effect of ozone on direct filter operations, and ozonation increased filter runtimes more than 30% by improving turbidity removal and slowing head loss development compared to direct filtration without ozonation. In that study, the applied ozone dose was low (0.50-0.75 mg/L) and source water TOC was low (3.0 mg/L). In another study on low TOC source water, Yüksel et al. (2002) found ozonation, without coagulation, significantly enhanced turbidity removal across direct filtration but shortened filter runtimes due to increased head loss development. In a pilot study comparing filtration with pre-ozonation to filtration with pre-aeration, Eyvaz et al. (2010) observed ozonation consistently improved direct filter effluent turbidity but did not significantly affect head loss development or filter runtimes. Rather, the type and amount of coagulant dosed had significant influences on filter effluent turbidity and head loss development. In that experiment, a low ozone dose was applied (2 mg/L) for a moderate TOC source water (7.0 mg/L). Eyvaz et al. (2010) also compared ozone-biofiltration with and without intermediate coagulation, and the filter with coagulant addition exhibited faster head loss development but achieved significantly lower effluent turbidity. In that study, filter effluent turbidity remained greater than 2.0 NTU for two testing conditions (ozonation without coagulation and coagulation without ozonation) highlighting the importance of both ozonation and coagulation for turbidity removal in direct filtration.

Ozone transforms natural organic matter into more oxygenated compounds that may alter coagulant demand in the direct filtration treatment train. A study by Edwards et al. (1994) investigated the relationship between ozonation and coagulant demand for different coagulants.

When polymer was used as a sole coagulant, ozonation reduced both coagulant demand and head loss development. When polymer was used in addition to alum or ferric chloride, ozonation had no significant effect on coagulant demand but reduced both filter effluent turbidity and head loss development. When ferric chloride was used as a sole coagulant, ozonation increased coagulant demand, filter effluent turbidity, and head loss development. In that study, filter effluent turbidity was reduced to 0.1 NTU with a metal-salt/polymer combination while it was 1.0 NTU with polymer alone. In a direct filtration pilot study that dosed alum and a cationic polymer, coagulant demand was not reduced for ozonated waters (Tobiason et al., 1992). Becker & O'Melia (2001) observed ozonation increased coagulant demand for a source water high in organic content and decreased coagulant demand for a source water low in organic content. In that study, increasing ozone doses from 0.0 to 2.0 O_3/TOC ratios increased DOC removal at low alum doses (<2.5 mg-Al/L) and, in contrast, increasing ozone doses decreased DOC removal at higher coagulant doses (>6.8 mg-Al/L alum). The effects of ozonation on coagulation ultimately depend on the type and amount of coagulant used and source water characteristics.

Ozonation also transforms bulk organics into more biodegradable molecules, allowing for increased removal of organic matter through biofiltration (Weng et al., 1986; Pruden et al., 2020). Ozone itself has a marginal effect on the total concentration of organic matter present, but the newly converted biodegradable dissolved organic carbon (BDOC) is more readily removed through subsequent biofiltration. Blair (2023) observed that the fraction of BDOC present in ozone effluent was positively correlated with the ozone dose when dosed at an O_3/TOC ratio. Aquino (2017) reported up to 20% more TOC removal across filtration with pre-ozonation than in filtration without ozonation. However, Tobiason et al. (1992) observed slightly greater DOC concentrations in ozone-biofiltration effluent compared to filtration without ozonation (Tobiason et al., 1992). In

that study, however, the applied ozone dose was less than 1.0 mg/L for a source water with 3.3 mg/L TOC. A meta-analysis by Peterson & Summers (2021) found that pre-ozonation increased TOC removal by an average of 10-20% for ozone doses from 0.5 to 1.1 O_3/TOC compared to filtration without ozonation, and marginal gains in TOC removal were observed for increasing the applied ozone dose. In a pilot study by Gifford et al. (2018), increasing the ozone dose from 0.35 to 0.62 to 1.12 O_3/TOC ratios resulted in increased TOC removal from 17-19% to 21-24% to 26-30%, respectively.

However, like other disinfectants, ozone risks disinfection byproduct formation when precursors are present in the source water. Bromate formation is of particular concern for reuse applications on source waters that contain bromide, which reacts with ozone to form bromate, a potential carcinogen regulated at 10 micrograms per liter[8]. Various bromate control mechanisms such as pH suppression, monochloramine addition, and hydrogen peroxide addition have been studied and applied to minimize disinfection byproduct formation while maintaining the benefits of ozonation (von Gunten & Pinkernell, 2000; Pearce et al., 2022).

In summary, studies demonstrated that ozone generally improves downstream filter turbidity removal, but the effects of ozone on head loss development are inconclusive and head loss development is more affected by coagulant type and dose. Studies observed ozone can influence coagulant demand, but the extent of the impact depends on coagulant type and dose and source water organic content. Ozone also transforms organic matter into more biodegradable forms, enhancing TOC removal through downstream biofiltration, and TOC removal generally increases with ozone dose. Ozone operations should be balanced to achieve sufficient removal of

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[8] 40 CFR §141

pathogens, organics, turbidity, and other contaminants across the treatment train while minimizing disinfection byproduct formation.

Biofiltration

Filter effluent water quality is also influenced by factors such as temperature, loading rate, empty bed contact time (EBCT), media type, and filter aid polymer usage. In a study on conventional treatment processes, Beniwal et al. (2018) found that removal of organic compounds was less efficient at lower temperatures. Pharand et al. (2015) also observed that assimilable organic carbon (AOC) removal was less efficient at lower temperatures, even when longer EBCTs were used in the winter months. In that study, filters achieved 56% AOC removal during temperatures greater than 10°C and only 40% removal during temperatures less than 10°C. In another study, it was noted that sedimentation performance declined with colder temperatures, whereas direct filtration was less affected by temperature (Pernitsky & Edzwald, 2006).

TOC removal is influenced by EBCT, where longer contact times allow for additional biodegradation of contaminants. LeChevallier et al. (1992) found that longer EBCTs led to more efficient AOC and TOC removals. In that study, effective AOC removal occurred within 5 to 10 minutes of filtration while effective TOC removal required a 20-minute EBCT. Many drinking water plants have reported diminishing returns in TOC removal as EBCT increases from 5 to 15 minutes (Peterson & Summers, 2021). In reuse applications, increasing TOC removal efficiencies were observed for EBCTs up to 30 minutes due to greater and more recalcitrant biological organic carbon present in wastewater effluents (Peterson & Summers, 2021). However, in some reuse applications, TOC removal began to plateau with EBCTs greater than 10 minutes (Aquino, 2017). Reungoat et al. (2012) observed a similar nonlinear relationship between EBCT and DOC removal.

Gifford et al. (2018) observed a positive linear relationship between ozone dose (0.35 to 1.1 O_3/TOC) and optimal EBCT (2 to 10 minutes) for biologically activated carbon and anthracite filters, indicating that a longer EBCT was needed to remove additional transformed BDOC and acknowledging that the linear relationship may not hold at higher ozone doses and EBCTs. At high ozone doses (2.0-4.0 O_3/TOC), Hozalski (1996) found that increasing EBCT from 4 to 20 minutes had no significant effect on TOC removal because the BDOC was rapidly removed in the first four minutes, indicating that an increased ozone dose may decrease the optimal EBCT. Compared to conventional treatment, direct filtration involves a higher organic loading to the filters, and a longer EBCT may be required to provide sufficient contact time for biodegradation.

In addition, a lower hydraulic loading rate may be needed in direct filtration to slow head loss development and reduce contaminant breakthrough from shear forces. Hutchison (1976) reported that direct filters operating at filtration rates from 2.4 to 7.2 gpm/sf produced comparable filter effluent turbidity. Williams et al. (2007), however, observed filter effluent turbidity increased with increasing loading rates from 5 gpm/sf to 10 gpm/sf on coagulated secondary effluent for tertiary filtration. A minimum hydraulic loading rate may be necessary to distribute particles across the depth of the filter if a low loading rate would cause particles to aggregate at the top of and consequentially clog the filter. A higher loading rate, however, also increases the solids loading rate and may promote more rapid head loss development and turbidity breakthrough, decreasing filter runtimes. A study by Logsdon et al. (1993) found that increasing the hydraulic loading rate resulted in shorter filter runs, with runtimes of 15-25 hours at 6 gpm/sf and 11-13 hours at 9 gpm/sf (Logsdon et al., 1993).

Filter performance is also affected by media type and effective particle size. In direct filtration applications, dual media demonstrated superior performance to single media filters with

the same filter depth (Weng et al., 1986; Zouboulis et al., 2007). In a direct filtration study on high turbidity (20-30 NTU) source water, a deep-bed monomedium filter performed better than a shallow dual media filter (Logsdon et al., 1993). In a study where ozone and biofiltration were applied to membrane bioreactor effluent, exhausted GAC media outperformed anthracite media in terms of TOC removal (Gifford et al., 2018). In advanced treatment applications with similar source water quality and operating conditions, such as ozone dose and EBCT, exhausted GAC media removed more TOC than sand or anthracite media by an average of 7 percentage points (Peterson & Summers, 2021). Smaller effective size media are more tightly packed with smaller void spaces and larger total media surface, providing greater solids removal capacity. On the other hand, larger effective size media provide larger void spaces for greater filter depth penetration but less solids removal capacity for a given depth. In direct filtration applications, media size did not have a significant effect on turbidity removal, but smaller effective size media exhibited more rapid head loss accumulation (Tate et al., 1977).

Turbidity removal is an indicator of contaminant and pathogen removal. A direct filtration pilot study showed a linear correlation between aerobic spores and turbidity removals (Ndiongue et al., 2000). The Surface Water Treatment Rule (SWTR), revised in 2006, outlined pathogen removal credits for achieving target filter effluent turbidities for conventional and direct filtration[9]. Full-scale and pilot-scale studies reported comparable removal of Giardia and Cryptosporidium for conventional treatment and direct filtration and demonstrated a relationship between turbidity and cyst and oocyst removal (Nieminski & Ongerth, 1995; Ndiongue et al., 2000). On the other hand, Patania et al. (1995) found that direct filtration achieved less pathogen removal than conventional treatment and detected pathogen breakthrough as filter effluent turbidity increased

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[9] 40 CFR §141

from 0.1 to 0.2 NTU. The results of that study contributed to the SWTR decision to grant less log removal credit for direct filtration than conventional treatment and to grant additional log removal for filter effluent turbidity less than 0.15 NTU[10]. Also in Patania et al. (1995), the quantity of pathogens in the source water affected its removal, and sedimentation achieved significantly less log removal of turbidity and pathogens for source waters low in turbidity (1 NTU) compared to source waters with higher turbidity (10 NTU). Results suggested that for low turbidity source waters, direct filtration may achieve sufficient pathogen removal and sedimentation may not provide substantial additional removal. Mukherjee (1999) observed indicator virus breakthrough after about a week of filter operations despite low effluent turbidity, indicating that turbidity removal does not guarantee sufficient pathogen removal and pathogen removal should be validated. That study also demonstrated enhanced indicator virus removal with upstream flocculation compared to in-line filtration; and virus, turbidity, and particle removal efficiencies declined for influent DOC greater than 5 mg/L. Overall, studies demonstrated a relationship between turbidity and pathogen removal, and the SWTR established turbidity limits and pathogen removal credits for conventional and direct filtration drinking water treatment plants. To extend this framework to potable reuse applications, the relationship between pathogen removal and turbidity should be further validated.

Applications

Pilot-scale and full-scale tests have demonstrated that direct filtration can be a practical alternative to conventional drinking water treatment. In a pilot study comparing conventional

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[10] 40 CFR §141

treatment and direct filtration, direct filtration achieved comparable water quality to conventional with an estimated 15% cost savings (Bryant & Brailey, 1980). In that study, direct filtration produced effluent turbidity less than 0.20 NTU with 1.4-1.8 mg-Al/L alum, 0.2-1.0 mg/L polymer, 47-75 seconds rapid mix, and 2.5-4.0 gpm/sf filter loading rate. A different pilot study showed direct filtration can reduce source water turbidity from 13.6 NTU to 0.16 NTU with 0.75 mg-Al/L alum, 1.5 mg/L polymer, and a 4.0 gpm/sf filter loading rate (Burns et al., 1984). Burns et al. (1984) also compared full-scale conventional and direct filtration plants with similar source water and found comparable turbidity removal. In another pilot study, direct filtration removed 99% of particles from the water and achieved filter effluent turbidities of 0.04-0.18 NTU (Tate et al., 1977). Westerhoff et al. (1980) found that direct filtration and conventional treatment achieved comparable final effluents, reducing source water turbidity of 1-100 NTU to 0.1-0.3 NTU at 2.0-6.0 gpm/sf filter loading rates. Based on those results, Westerhoff et al. (1989) estimated that direct filtration at a 34-MGD facility can achieve $50,000 annual operating cost-savings, not including solids handling operations. While several direct filtration plants successfully operate worldwide, water quality standards vary, and filter effluent turbidities vary (Melo et al., 2021). The feasibility of direct filtration ultimately depends on regulatory requirements. However, there is limited research on direct filtration for potable reuse applications. Pilot studies may be used to assess the viability of direct filtration, and Knowles et al. (2012) demonstrated that pilot performance is a reasonable estimate of full-scale direct filtration treatment.

CHAPTER III

METHODOLOGY

Source Water

A pilot plant was used to compare direct filtration and conventional treatment for secondary and tertiary effluents. Source water included (1) denitrification filter effluent from HRSD's York River Treatment Plant (YRTP) in Seaford, Virginia and (2) secondary clarifier effluent from HRSD's Nansemond Treatment Plant (NTP) in Suffolk, Virginia. YRTP is a 15-MGD water reclamation facility employing tertiary treatment through denitrification filters. NTP is a 30-MGD water reclamation facility applying secondary treatment through five-stage Bardenpho and clarification. Source water quality data during pilot testing are summarized in Table 1.

Table 1: Source Water Quality

Parameter	YRTP	NTP	Units
Turbidity	1.46 ± 0.80 (n=116)	1.67 ± 1.75 (n=699)	NTU
TSS	2.11 ± 0.78 (n=19)	4.49 ± 3.22 (n=113)	mg/L
TOC	6.86 ± 0.59 (n=198)	9.27 ± 1.61 (n=329)	mg/L
DOC	6.71 ± 0.57 (n=38)	8.92 ± 1.16 (n=75)	mg/L
TP	0.27 ± 0.11 (n=37)	0.70 ± 0.61 (n=88)	mg/L-P
OP	0.21 ± 0.12 (n=34)	0.44 ± 0.51 (n=79)	mg/L-P
Alkalinity	125 ± 14.0 (n=47)	159 ± 19.5 (n=96)	mg/L-CaCO$_3$
pH	7.2 ± 0.11 (n=335)	7.2 ± 0.22 (n=697)	pH Units

YRTP: Data from 2/22/2020-5/11/2020, 6/8/2020-1/22/2021
NTP: Data from 2/1/2021-12/31/2022

HRSD plans to construct SWIFT advanced treatment facilities at YRTP and NTP. YRTP was identified as a potential candidate for direct filtration due to its consistently low turbidity, total suspended solids (TSS), and total phosphorus (TP) in the final effluent. Based on the success of the YRTP pilot study, direct filtration was also considered for NTP, a facility with slightly higher and more variable effluent turbidity, TSS, and TP.

Pilot Setup

Intuitech coagulation/flocculation/sedimentation, ozonation, and granular media filtration pilot plants were configured for two parallel treatment trains comparing conventional treatment and direct filtration (Figure 2).   The conventional treatment train consisted of (1) coagulation/flocculation/sedimentation; (2) ozonation; and (3) biofiltration. The direct filtration train consisted of (1) ozonation; (2) coagulation/flocculation; and (3) biofiltration. Design criteria are listed in Table 2. Photographs of the pilot plant are in Appendix A. The pilot plant was located inside the SWIFT Research Center at HRSD's Nansemond Treatment Plant in Suffolk, Virginia.

The pilot feed tank was piped with options to receive either NTP secondary clarifier effluent or water stored outside in a 20,000-gallon Adler tank. For the YRTP study, denitrification filter effluent from YRTP was transported to the pilot by a water truck, refilling the Adler tank a few times a week. Pilot operations were automated and monitored through Intuitech operating systems. Aluminum chlorohydrate (ACH) and aluminum sulfate (alum) were tested as coagulants for both conventional and direct filtration.   Polyacrylamide polymer Clarifloc C-6220 and polyDADMAC polymer Clarifloc C-308P from Polydyne, Inc. were used as flocculant aids for
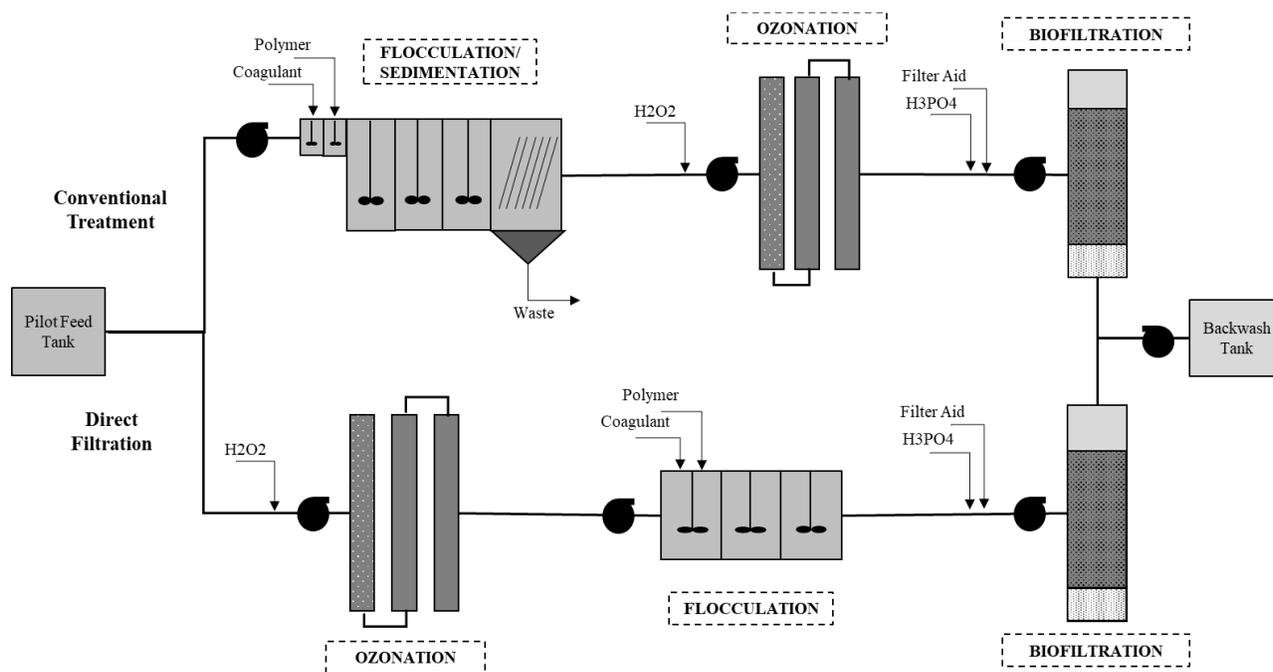
Figure 2: Pilot Plant Process Flow Diagram

Table 2: Pilot Plant Design Criteria

Flocculation/Sedimentation (CF)			Flocculation (DF)			Ozonation		
Flowrate	2.5-4.0*	gpm	Flowrate	0.9	gpm	Flowrate	1.0	gpm
Rapid Mix Stages	2		Rapid Mix Stages	1		Ozone Dose	0.8-1.0	O_3/TOC
Rapid Mix HRT	0.6-1.0	min/stage	Rapid Mix HRT	6.8	min/stage	Contact Time	9 (?)	min
Rapid Mix G	1000	s^{-1}	Rapid Mix G	1000	s^{-1}	H_2O_2 Dose	1.0	H_2O_2/O_3
Floc Stages	3		Floc Stages	2		Biofiltration		
Floc HRT	21-33	min/stage	Floc HRT	6.8	min/stage	Flowrate	0.49-0.71	gpm
Floc G	40/20/10	s^{-1}	Floc G	100,60	s^{-1}	Loading Rate	2.5-3.8	gpm/sf
Plate Loading Rate	0.2-0.4	gpm/sf	Coagulant Dose	0.8-1.5	mg/L-Al	GAC Depth	5	ft
Coagulant Dose	3.8-6.0	mg/L-Al	Polymer Dose	0.5-5.0	mg/L	Sand Depth	1	ft
Polymer Dose	0.75-1.0	mg/L				EBCT	10-15	min
*For YRTP, 1.23 gpm and one-stage floc at 15 s^{-1}, 67 min						H_3PO_4 Dose	0.05	mg/L-P
						Filter Aid Polymer	0.05-0.10	mg/L

conventional treatment and direct filtration, respectively. Coagulants and polymers were dosed to rapid mix chambers. Prior to ozonation, hydrogen peroxide (H_2O_2) was added at a 1.0 H_2O_2/ozone ratio for bromate control and advanced oxidation. Ozone was injected via fine bubble diffusion and the ozone dose was manually adjusted daily based on ozone influent TOC. The two identical biofilters, each with a column diameter of six inches, consisted of five feet of exhausted granular activated carbon (Calgon F816, effective size = 1.4 mm, uniformity coefficient = 1.4) above one foot of sand (effective size = 0.7 mm, uniformity coefficient = 1.4). Each filter received low doses of phosphoric acid (H_3PO_4) to enhance biological activity and Clarifloc N-3300P (Polydyne, Inc.) as a nonionic filter aid polymer. Filters were programmed to backwash with filtrate at 10 feet of head loss or when effluent turbidity exceeded 0.15 NTU. The filter backwash sequence consisted of draining the column; air scour at 1.8 scfm for 300 seconds; a mix of air scour and backwash to refill the column; followed by a mix of low-rate and high-rate backwashing of 3.0 gpm for 120 seconds, 4.5 gpm for 240 seconds, and 1.0 gpm for 120 seconds. The mix of air scour and backwash flow rates were different for conventional and direct filters to provide a more intense backwash for the direct filter. The conventional filter received air flow of 1.0 scfm and backwash flow of 2.0 gpm and the direct filter received air flow of 1.8 scfm and backwash flow of 1.0 gpm. Filter effluent was diverted to waste for at least 30 minutes and until effluent turbidity was below 0.15 NTU for at least 120 seconds before returning to service.

Coagulation occurred prior to ozonation for conventional treatment and after ozonation for direct filtration. An alternative configuration for direct filtration, with coagulation prior to ozonation, was considered. Placing coagulation prior to ozonation has the potential to provide additional capital cost savings by utilizing a single rapid mix stage with coagulant addition and using the ozone contactor and fine bubble diffusion for flocculation. However, upstream chemical

addition could contribute to ozone demand, and the polyDADMAC polymer is a potential precursor for N-Nitrosodimethylamine (NDMA) formation by ozone, particularly with bromide present (Padhye et al., 2011). Furthermore, flocculation through ozonation could lead to solids build up in the ozone contactor or other operational issues. The alternative setup was tested on the direct filtration pilot and promptly rejected due to turbidity removal issues. Additional details on this experiment are found in Appendix B.

Pilot Operations

The pilot plants were operated and evaluated over a range of operating conditions and temperatures. Filter runtimes were evaluated to determine if direct filtration can achieve filter effluent turbidity requirements with a reasonable backwashing frequency (runtimes >24 hours). TOC removal was used to investigate whether direct filtration can achieve sufficient organics removal without increasing downstream GAC replacement frequency. Phosphorus removal was measured to see whether direct filtration could meet effluent total phosphorus requirements (<0.3 mg-P/L). TSS challenge testing was performed to further assess the range of influent water quality viable for direct filtration. During NTP testing, an alternative loading rate, EBCT, and coagulant were evaluated. The testing schedule is outlined in Table 3.

"Normal operations" denotes the YRTP and NTP testing periods when both pilot filters were operated at a 10-minute EBCT, 3.8 gpm/sf loading rate, with ACH as a coagulant, and over a range of winter to summer temperatures.

Table 3: Pilot Testing Plan

Operations	Dates
YRTP	
Normal operations	2/21/20-5/11/20, 6/8/20-1/22/21
TSS Excursions	5/25/20-6/4/20
NTP	
Normal operations	2/1/21-7/1/21, 12/25/21-1/7/22
Lower loading rate/longer EBCT	1/7/22-3/1/22 (ACH), 3/1/22-8/2/22 (Alum)
Alum as coagulant	3/1/22-8/2/22
TSS Challenge Testing	1/17/23-1/31/23

YRTP direct filtration pilot testing began in February 2020 and influent temperature was controlled to 20°C. Jar testing was conducted to determine initial coagulant and polymer doses based on visual pin-floc formation for the direct filtration treatment train and based on turbidity removal for the conventional treatment train. For direct filtration, coagulant and polymer doses were further adjusted on the pilot plant to improve filter turbidity removal and head loss development. The coagulant doses that most consistently achieved filter runtimes greater than 24 hours were 0.8 mg-Al/L ACH and 1.0 mg/L polymer. For conventional treatment, 3.8 mg-Al/L ACH and 0.75 mg/L polymer were used. In the next phase of YRTP testing, secondary clarifier effluent from YRTP was tested on the pilot to evaluate direct filtration of more turbid source water. Returning to study denitrification filter effluent, the final phase of YRTP testing covered a range of temperatures from 14°C to 26°C with phosphoric acid (0.05 mg-P/L), filter aid polymer (0.10 mg/L to the conventional filter, 0.05 mg/L to the direct filter), and hydrogen peroxide (1.0 H_2O_2/O_3) additions, and ozone was dosed at a constant ratio (0.8 O_3/TOC). Filter runtimes from the last two phases of testing and TOC removal from all periods of normal operations were evaluated in this study.

Appendix C contains a graph of direct filter runs across the entire YRTP direct filtration pilot test, including testing periods beyond the scope of this paper. NTP direct filtration pilot testing began in February 2021 and during normal operations, influent temperature ranged from 15°C to 27°C. The pilot filters received a constant dose of phosphoric acid (0.05 mg-P/L), filter aid polymer (0.10 mg/L), and hydrogen peroxide (1.0 H_2O_2/ozone). Ozone was dosed at approximately a 0.8 O_3/TOC ratio. Coagulants for direct filtration were initially dosed at 0.8 mg-Al/L ACH and 1.0 mg/L polymer, doses that were used in the YRTP pilot study. For conventional treatment, coagulants were dosed at 6.25 mg-Al/L ACH and 1.0 mg/L polymer based on jar testing results for turbidity removal. During startup, NTP secondary clarifier effluent turbidity and TSS were above average, and temperatures were low due to winter conditions. For direct filtration, coagulant doses were incrementally increased to improve filter turbidity removal and extend filter runtimes (0.8-2.0 mg-Al/L ACH; 1.0-5.0 mg/L polymer). The coagulant doses that most consistently achieved direct filter runtimes greater than 24 hours during this testing period were 1.0 mg-Al/L ACH and 5.0 mg/L polymer.

A lower loading rate and a higher EBCT were evaluated during NTP pilot testing. The loading rate and EBCT were changed simultaneously by adjusting the filter influent flow rate and keeping filter dimensions constant. Flow through the upstream processes (coagulation, ozone) remained constant. Other loading rates and EBCTs were evaluated in a separate experiment and results are included in Appendix G. A 15-minute EBCT and 2.5 gpm/sf loading rate were selected for further testing. For this testing period, coagulant doses on the direct filtration process train were 1.0 mg-Al/L ACH and 3.0 mg/L polymer and on the conventional treatment train were 6.25 mg-Al/L ACH and 1.0 mg/L polymer. For both treatment trains, other chemical doses included

ozone at a 1.0 O_3/TOC ratio, hydrogen peroxide at a 1.0 H_2O_2/ozone ratio, 0.05 mg-P/L phosphoric acid, and 0.10 mg/L filter aid polymer.

Alum was evaluated as an alternative coagulant during NTP pilot testing. Both filters remained at a 15-minute EBCT and 2.5 gpm/sf loading rate. Alum was dosed at an equivalent dose of aluminum as ACH to both treatment trains (6.25 mg-Al/L for conventional treatment and 1.0 mg-Al/L for direct filtration). Polymer was initially dosed at 1.0 mg/L for conventional treatment and 3.0 mg/L for direct filtration. In response to elevated direct filter effluent turbidity, a range of alum doses (1.0–4.0 mg-Al/L) and polymer doses (3.0 – 5.0 mg/L) were tested on the direct filtration treatment train. Increasing the polymer dose was found to be more effective at reducing filter effluent turbidity than increasing the coagulant dose within these ranges. The coagulant doses that most consistently achieved direct filter runtimes greater than 24 hours during this testing period were 1.0 mg-Al/L alum and 4.0 mg/L polymer.

NTP secondary clarifier effluent turbidity and TSS were consistently lower during the EBCT/loading rate and alum testing periods compared to the first phase of testing, normal operations. To revisit periods of elevated influent turbidity at a lower loading rate, TSS challenge testing was conducted and consisted of two phases. In Phase 1, NTP secondary clarifier effluent was stored in the 20,000-gallon Adler tank and continuously mixed with two 70 gpm submersible pumps. Return activated sludge collected from the Nansemond Treatment Plant was spiked into the tank. TSS of the return activated sludge was measured with Royce Model 711 Portable Suspended Solids Analyzer. Influent tank volume was estimated, and sludge was spiked daily to target an increasing pilot influent TSS from 3 to 15 mg/L over time. Pilot influent temperature was controlled to 15°C. Coagulant doses remained unchanged. Sludge was spiked in the evenings and TSS and turbidity were measured the next morning to allow sufficient time for mixing. Measured

TSS and turbidity were lower than expected for the first few days. The mass of solids spiked to the tank was unaccounted for in the pilot influent, and a general trend of decreasing turbidity over time after spiking was observed. Therefore, incomplete mixing in the tank likely led to temporary slugs of solids that went undetected in the TSS and turbidity data records. Improved mixing was achieved as the tank level lowered, indicated by higher measured TSS and more constant turbidity. In Phase 2, diluted return activated sludge was continuously fed to ozone influent using a peristaltic pump to target a constant TSS greater than 10 mg/L for a few days. Turbidity and TSS samples were collected daily.

During the YRTP and NTP pilot studies, the pilot plant experienced occasional mechanical and operational problems such as chemical feed issues, ozone generation issues, flow constraints, and power outages that affected filter performance and resulted in premature turbidity breakthrough and short filter runtimes. To eliminate bias from operational issues, filter runs less than 10 hours in length were excluded from the dataset. The one exception is during NTP pilot startup under normal operations, legitimate filter runs around 7 to 9 hours were still included.

Analytical Methods

TOC samples were collected daily (5x/week) and DOC, TSS, TP, and OP samples were collected once per week. During NTP testing with alum as a coagulant, TP and OP sampling frequency increased to 3x/week. DOC samples were filtered using a syringe and Whatman Puradisc polyethersulfone membrane syringe filters (25 mm, 0.45 μm). OP samples were filtered using a vacuum and Millipore mixed cellulose esters filters (47 mm, 0.45 μm). Routine samples were collected, stored on site in a laboratory refrigerator at 4°C, then transported on ice in coolers to HRSD's Central Environmental Laboratory in Virginia Beach, Virginia for analysis. Turbidity

was measured in the pilot lab on site. YRTP denitrification filter effluent (DFE) turbidity was measured with each new truck load, a few times a week. Turbidity for pilot conventional filter effluent (CF), direct filter effluent (DF), and NTP secondary clarifier effluent (SCE) were monitored continuously via online analyzers. Other analyses conducted in the pilot lab include TOC samples for daily ozone dosing and TSS and turbidity samples for NTP TSS challenge testing. For NTP challenge testing, TSS samples were filtered on Whatman Grade 934-AH glass microfiber filters (47 mm, 1.5 μm).

Table 4 summarizes analytical methods used in this pilot study.

Table 4: Analytical Methods

Continuous Monitoring		
Parameter	**Sample Location**	**Analyzer**
Turbidity	NTP SCE	Hach TU5300sc Low Range Laser Turbidimeter, EPA Version; with Hach Automatic Cleaning
Turbidity	CF, DF	HF Scientific – MTOL+ Online Process Turbidimeter; Compliant with EPA 180.1 and ISO 7027
Grab Samples		
Parameter	**Sample Location**	**Method**
Turbidity	YRTP DFE	Hach TU5200 Laboratory Laser Turbidimeter, EPA Version
TOC, (DOC)	YRTP DFE, NTP SCE, FSE, CF, DF	Shimadzu TOC-4200; Standard Method 5310B; (filtered sample)
TSS	YRTP DFE, NTP SCE	Standard Method 2540D
TP	YRTP DFE, NTP SCE, FSE, CF, DF	QuikChem Method 10-115-01-1-E
OP	YRTP DFE, NTP SCE, FSE, CF, DF	QuikChem Method 10-115-01-1-A

CHAPTER IV

MANUSCRIPT #1: PILOT-SCALE EVALUATION OF DIRECT FILTRATION FOR CARBON-BASED ADVANCED TREATMENT OF SECONDARY AND TERTIARY EFFLUENTS

Abstract

Pilot testing was performed to evaluate the feasibility of direct filtration for indirect potable reuse applications. A pilot plant compared conventional and direct filtration under variable operating conditions for secondary and tertiary wastewater effluents. Results determined whether direct filtration could achieve sufficient turbidity, total organic carbon, and phosphorus removal at a lower cost than conventional treatment and remain compliant with potable reuse and wastewater discharge regulations. At a 3.8 gpm/sf loading rate and 10-minute EBCT, direct filtration of tertiary denitrification filter effluent achieved filter effluent turbidity <0.15 NTU with 35-hour runs and of secondary clarifier effluent, 19-hour runs. Conventional treatment of tertiary and secondary effluents achieved 105 and 65 hours, respectively. TOC removal was comparable for both source waters with 34-35% removal through direct filtration and 41-44% removal through conventional treatment. At a 2.5 gpm/sf loading rate and 15-minute EBCT, direct filtration of secondary effluent achieved 43-hour filter runs and 39% TOC removal. Direct filtration with 0.8 mg-Al/L ACH achieved sufficient phosphorus removal on tertiary effluent but not on secondary effluent. For an equivalent dose of aluminum, alum achieved more efficient phosphorus removal than ACH but resulted in shorter filter runs and less efficient TOC removal. TSS challenge testing demonstrated direct filtration could manage spikes in TSS and turbidity up to 49 mg/L and 8.7 NTU, respectively. Pilot study results demonstrated direct filtration is a feasible alternative to conventional treatment, with implementation dependent on a cost-benefit analysis.

Introduction

Advanced treatment technologies continuously evolve to meet water quality standards at lower costs for diverse applications. Carbon-based advanced treatment employing ozone and biofiltration can produce final effluent compliant with drinking water standards and comparable to alternative membrane-based treatment at lower costs. Ozone-biofiltration systems typically follow conventional drinking water design, including an upstream clarification step to avoid overloading filters and to achieve additional contaminant and pathogen removal. Alternatively, direct filtration systems do not have a clarification step prior to filtration, potentially reducing capital costs, operating costs, and site footprint. In addition, lower coagulant doses and shorter mixing times are needed to form filterable floc compared to settleable floc. Culp (1977) estimated direct filtration can provide up to 30% capital cost savings from flocculation, sedimentation, and solids handling structures and 10-30% chemical cost savings. Without sedimentation, however, filters may experience shorter runs and require more backwash water, reducing final effluent production. Several drinking water treatment plants demonstrate the success of direct filtration around the globe, while applications in potable reuse are limited.

Hampton Roads Sanitation District (HRSD) is considering direct filtration as an alternative to conventional treatment for their indirect potable reuse project, the Sustainable Water Initiative for Tomorrow (SWIFT). SWIFT will purify up to 100-MGD of HRSD's treated wastewater for recharge into the Potomac Aquifer to reduce nutrient loading into the Chesapeake Bay, replenish the groundwater supply, protect against saltwater intrusion, and slow or reverse land subsidence. Conventional carbon-based advanced treatment with sedimentation achieves water quality goals, as demonstrated at the 1-MGD SWIFT Research Center (Hogard et al., 2021). HRSD is

considering direct filtration as an alternative to conventional treatment for future advanced treatment facilities. Eliminating the sedimentation step may provide significant cost-savings, but it is uncertain whether direct filtration can reliably achieve treatment goals. Water quality regulations include, but are not limited to, filter effluent turbidity (<0.15 NTU) and final effluent TOC (<4 mg/L)[11] under the Safe Drinking Water Act's Underground Injection Control program[12] for managed aquifer recharge, and filter effluent phosphorus (<0.3 mg/L) under Virginia Department of Environmental Quality's waste load allocations[13] for discharge.

The study of direct filtration in the drinking water sector offers insight for potable reuse applications. With increased solids loading to filters, direct filtration may only be suitable for source waters low in turbidity and particulates. Based on an early survey from existing plants, direct filtration may not be feasible for source waters with color and turbidity greater than 25 units but may be feasible for source waters with turbidity greater than 100 NTU if color is significantly low (Culp, 1977). A later study found that direct filtration may only work for source water turbidity less than 10 NTU but agreed direct filtration can handle more turbid waters if color and algal content are low (McCormick & King, 1982).

Direct filtration typically involves coagulation and flocculation to aggregate primary particles for better removal through filtration, and coagulant selection and dosage have significant impacts on downstream filter performance. In studies by Weng et al. (1986) and Eyvaz et al. (2010), increasing metal-salt coagulant dose reduced turbidity and increased headloss, resulting in shorter filter runs. Hutchison (1976) found that alum doses ranging from 0.35-1.8 mg-Al/L produced acceptable filter effluent turbidity and proposed that direct filtration may only be feasible

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[11] Permit VAS5B170028617
[12] 40 CFR §144
[13] VA Code § 62.1-44.19:14

for water with coagulant demand less than 1.4 mg-Al/L alum due to rapid head loss development. Polymers are preferred over metal salts for coagulation in direct filtration because polymers can neutralize charge without forming metal hydroxide precipitates, increasing filter runtimes and decreasing backwash solids production (Bolto & Gregory, 2007). Cationic polymers allow deeper penetration into the filter bed with dosages ranging from 0.1 – 5 mg/L (Culp, 1977). Becker & O'Melia (2001) observed that the optimum coagulant dose for source waters with moderate to high organic content was determined by the DOC. For source waters with TOC greater than 5 mg/L, polymer demand could exceed 6 mg/L; therefore, for high organic content source waters, the polymer dose required could be costly and impractical. In such cases, metal salt coagulants could be used at low doses as a flocculant aid (Rebhun et al., 1984; Bolto & Gregory, 2007). In addition, polymer as a sole coagulant could be ineffective for applications requiring phosphorus removal, where metal-salt addition would be necessary to achieve precipitation of phosphates to be removed in subsequent filtration. Hutchison (1976) found that direct filter effluent water quality improved with increasing polymer doses from 1 to 5 mg/L, but filter performance varied for different types of polymers. Rebhun et al. (1984) recommended a high molecular weight polymer such as polyDADMAC for direct filtration. Bolto & Gregory (2007), however, found no significant effect of polymer molecular weight on direct filter performance, rather flocculation detention time and mixing speed had greater influences. Flocculation enhances turbidity removal and slows head loss development through pin-floc formation (Edzwald et al., 1986). Studies reported flocculation detention times that varied from 15 seconds to 30 minutes depending on source water characteristics and other operating parameters (Hutchison, 1976; Culp, 1977; Tate et al., 1977; Monscvitz et al., 1978; Treweek, 1979; Yeh & Gosh, 1981). Recommended rapid mix velocity

gradients varied from 20 to 650 s^{-1} for sufficient floc formation without shearing (Hutchison, 1976; Stump et al., 1979; Yeh & Gosh, 1981).

Ozonation is not only applied for disinfection but also affects direct filter performance by extending filter runtimes and enhancing TOC removal. Tobiason et al. (1992) observed that ozonation increased filter runtimes by more than 30% through increased turbidity removal and decreased head loss development. In two pilot studies comparing the effects of a filter receiving pre-ozonated water to an identical filter receiving pre-aerated water, ozonation significantly improved filter effluent turbidity (Yüksel et al., 2002; Eyvaz et al., 2010). Ozonation also transforms bulk organics into more biodegradable molecules, allowing for increased removal of organic matter through biofiltration (Weng et al., 1986; Pruden et al., 2020). While ozonation marginally affects TOC concentrations, newly converted BDOC is more readily removed through downstream biofiltration. A meta-analysis reported that pre-ozonation increased TOC removal 10-20% on average for ozone doses from 0.5-1.1 O$_3$/TOC (Peterson & Summers, 2021). In a pilot study by Gifford et al. (2018), TOC removal increased from 17-19% to 21-24% to 26-30% by increasing the ozone dose from 0.35 to 0.62 to 1.12 O$_3$/TOC ratios.

Filter effluent water quality is also influenced by factors such as temperature, loading rate, empty bed contact time (EBCT), media type, and filter aid polymer usage. In studies on conventional treatment, lower temperatures have been correlated with less efficient organics removal (Pharand et al., 2015; Beniwal et al., 2018). In a study comparing conventional and direct filtration, sedimentation performance declined with colder temperatures, while direct filtration was less affected by temperature (Pernitsky & Edzwald, 2006). A lower hydraulic loading rate may be needed in direct filtration compared to conventional treatment to slow head loss development and reduce contaminant breakthrough from shear forces.   A direct filtration study showed filter

runtimes ranging from 15-25 hours at 6 gpm/sf and 11-13 hours at 9 gpm/sf (Logsdon et al., 1993). Hutchison (1976), however, observed that direct filters operating at 2.4 – 7.2 gpm/sf produced comparable effluent turbidity. TOC removal is influenced by EBCT, where longer EBCTs extend contact time for biodegradation. A study by LeChevallier et al. (1992) on conventional drinking water treatment demonstrated a positive relationship between EBCT and TOC removal. Drinking water plants reported diminishing returns in TOC removal as EBCT increased from 5 to 15 minutes, while in reuse applications, increasing TOC removal efficiencies were observed for EBCTs up to 30 minutes due to greater and more recalcitrant biological organic carbon present in wastewater effluents (Peterson & Summers, 2021). Other reuse studies found diminishing returns on TOC removal with EBCTs greater than 10 minutes (Reungoat et al., 2012; Aquino, 2017). Gifford et al. (2018) observed a positive linear relationship between ozone dose (0.35 to 1.1 O_3/TOC) and optimal EBCT (2 to 10 minutes) for biologically activated carbon and anthracite filters, indicating that a longer EBCT was needed to remove additional transformed BDOC and acknowledging that the linear relationship may not hold at higher ozone doses and EBCTs. At high ozone doses (2.0-4.0 O_3/TOC), Hozalski (1996) found increasing EBCT from 4 to 20 minutes had no significant effect on TOC removal as the BDOC was rapidly removed in the first four minutes, implying an increased ozone dose may decrease the optimal EBCT. Compared to conventional treatment, direct filtration involves a higher organic loading to the filters, and a longer EBCT and lower loading rate may be required to provide sufficient contact time for biodegradation and reduce contaminant breakthrough from shear forces.

In addition, turbidity removal is an indicator of contaminant and pathogen removal. Full-scale and pilot-scale results demonstrated comparable removal of Giardia and Cryptosporidium for conventional and direct filtration and demonstrated a relationship between turbidity and cyst

and oocyst removal (Nieminski & Ongerth, 1995; Ndiongue et al., 2000). Patania et al. (1995) observed that direct filtration achieved less pathogen removal than conventional treatment, guiding the SWTR to grant less log removal credit for direct filtration[14]. In that study, sedimentation removed significantly less turbidity and pathogens for low turbidity source waters (1 NTU) compared to higher turbidity source waters (10 NTU), suggesting that direct filtration may achieve sufficient pathogen removal and sedimentation may not provide substantial additional removal for low turbidity source waters. Mukherjee (1999) observed indicator breakthrough after about a week of filter operations despite low turbidity, indicating that turbidity does not guarantee sufficient pathogen removal, and pathogen removal should be validated.

Several pilot-scale and full-scale studies demonstrated direct filtration can achieve comparable performance to conventional treatment (Tate et al., 1977; Bryant & Brailey, 1980; Westerhoff et al., 1980; Burns et al., 1984; Melo et al., 2021). In addition, Knowles et al. (2012) concluded that direct filtration pilot plants could produce statistically equivalent water quality to their full-scale counterparts. There is, however, a lack of research on direct filtration for potable reuse applications.

The objectives of this project were to determine through pilot testing whether direct filtration could achieve (1) filter effluent turbidity requirements without excessive backwashing, (2) comparable TOC removal without driving up the cost of GAC regeneration, and (3) sufficient phosphorus removal despite lower coagulant doses. Secondary and tertiary effluents were tested, offering insight on the range of influent water quality viable for direct filtration.

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[14] 40 CFR § 141

Methodology

A pilot plant was developed to compare direct filtration and conventional treatment for secondary and tertiary effluents. Source water included (1) denitrification filter effluent from HRSD's York River Treatment Plant (YRTP) in Seaford, Virginia and (2) secondary clarifier effluent from HRSD's Nansemond Treatment Plant (NTP) in Suffolk, Virginia. Source water quality data are summarized in Table 5.

Table 5: Source Water Quality

Parameter	YRTP	NTP	Units
Turbidity	1.46 ± 0.80 (n=116)	1.67 ± 1.75 (n=699)	NTU
TSS	2.11 ± 0.78 (n=19)	4.49 ± 3.22 (n=113)	mg/L
TOC	6.86 ± 0.59 (n=198)	9.27 ± 1.61 (n=329)	mg/L
DOC	6.71 ± 0.57 (n=38)	8.92 ± 1.16 (n=75)	mg/L
TP	0.27 ± 0.11 (n=37)	0.70 ± 0.61 (n=88)	mg/L-P
OP	0.21 ± 0.12 (n=34)	0.44 ± 0.51 (n=79)	mg/L-P
Alkalinity	125 ± 14.0 (n=47)	159 ± 19.5 (n=96)	mg/L-CaCO$_3$
pH	7.2 ± 0.11 (n=335)	7.2 ± 0.22 (n=697)	pH Units

YRTP: Data from 2/22/2020-5/11/2020, 6/8/2020-1/22/2021
NTP: Data from 2/1/2021-12/31/2022

Intuitech coagulation/flocculation/sedimentation, ozonation, and granular media filtration pilot plants were configured for two parallel treatment trains comparing conventional treatment and direct filtration (Figure 3). The conventional treatment train consisted of (1) coagulation/ flocculation/sedimentation; (2) ozonation; and (3) biofiltration. The direct filtration train consisted of (1) ozonation; (2) coagulation/flocculation; and (3) biofiltration. Design criteria are listed in Table 6.
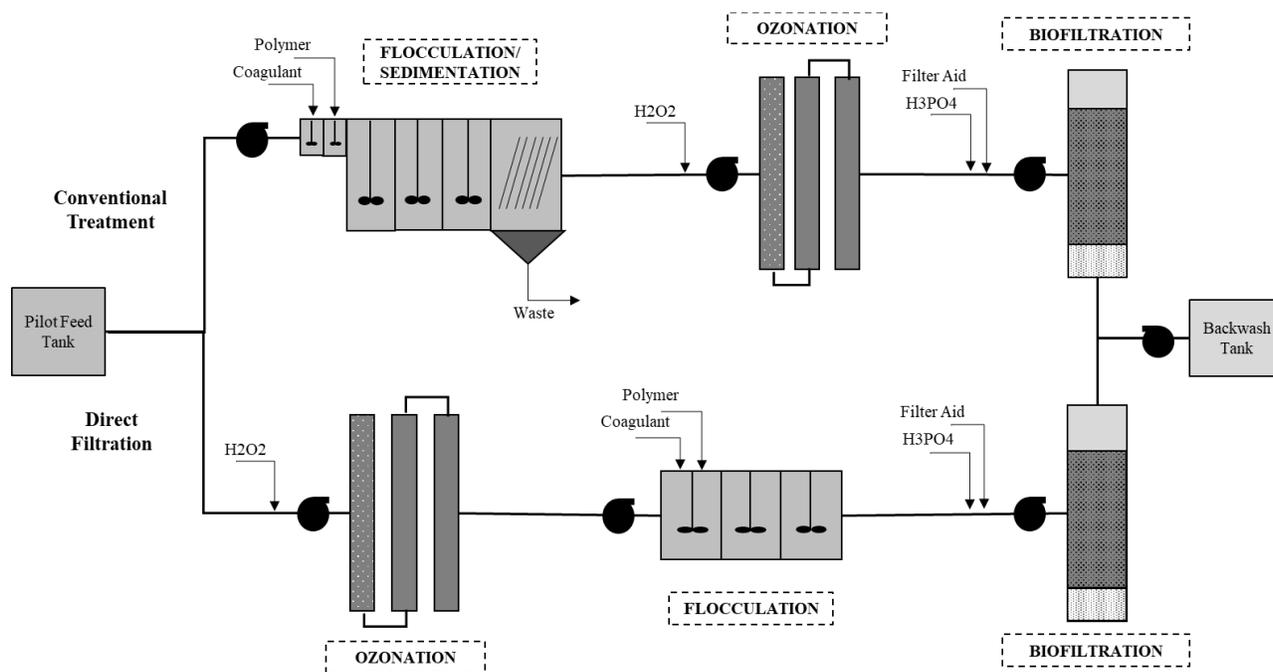
Figure 3: Pilot Plant Process Flow Diagram

Table 6: Pilot Plant Design Criteria

Flocculation/Sedimentation (CF)			Flocculation (DF)			Ozonation		
Flowrate	2.5-4.0*	gpm	Flowrate	0.9	gpm	Flowrate	1.0	gpm
Rapid Mix Stages	2		Rapid Mix Stages	1		Ozone Dose	0.8-1.0	O_3/TOC
Rapid Mix HRT	0.6-1.0	min/stage	Rapid Mix HRT	6.8	min/stage	Contact Time	10	min
Rapid Mix G	1000	s^{-1}	Rapid Mix G	1000	s^{-1}	H_2O_2 Dose	1.0	H_2O_2/O_3
Floc Stages	3		Floc Stages	2		Biofiltration		
Floc HRT	21-33	min/stage	Floc HRT	6.8	min/stage	Flowrate	0.49-0.71	gpm
Floc G	40/20/10	s^{-1}	Floc G	100,60	s^{-1}	Loading Rate	2.5-3.8	gpm/sf
Plate Loading Rate	0.2-0.4	gpm/sf	Coagulant Dose	0.8-1.5	mg/L-Al	GAC Depth	5	ft
Coagulant Dose	3.8-6.0	mg/L-Al	Polymer Dose	0.5-5.0	mg/L	Sand Depth	1	ft
Polymer Dose	0.75-1.0	mg/L				EBCT	10-15	min
*For YRTP, low 1.23 gpm, one-stage floc at, 15 s^{-1}, 67 min						H_3PO_4 Dose	0.05	mg/L-P
						Filter Aid Polymer	0.05-0.10	mg/L

The pilot feed tank was piped to receive either NTP secondary clarifier effluent or water stored outside in a 20,000-gallon Adler tank. For the YRTP study, denitrification filter effluent was transported by a water truck, filling the Adler tank a few times a week. Pilot operations were automated and monitored through Intuitech operating systems. Aluminum chlorohydrate (ACH) and aluminum sulfate (alum) were used as coagulants for conventional and direct filtration. Polyacrylamide polymer Clarifloc C-6220 and polyDADMAC polymer Clarifloc C-308P from Polydyne, Inc. were used as flocculant aids for conventional and direct filtration, respectively. All coagulants were dosed to rapid mix chambers. Prior to ozonation, hydrogen peroxide was added for bromate control and advanced oxidation. Ozone was injected via fine bubble diffusion and ozone dose was manually adjusted based on daily ozone influent TOC. Two identical biofilters consisted of five feet of exhausted granular activated carbon (Calgon F816, effective size = 1.4 mm, uniformity coefficient = 1.4) above one foot of sand (effective size = 0.7 mm, uniformity coefficient = 1.4). Each filter received low doses of phosphoric acid to enhance biological activity and Clarifloc N-3300P (Polydyne, Inc.) as a nonionic filter aid polymer. Filters were programmed to backwash with filtrate at 10 feet of head loss or when effluent turbidity exceeded 0.15 NTU.

NTP SCE turbidity was continuously monitored via Hach TU5300sc Low Range Laser Turbidimeter and filter effluent turbidity was continuously monitored via HF Scientific MTOL+ Online Process Turbidimeter. Turbidity grab samples for YRTP DFE and for online instrument verification were measured with Hach TU5200 Laboratory Laser Turbidimeter with RFID. TOC and DOC were analyzed by Standard Method 5310B on the Shimadzu TOC-4200. Total phosphorus and orthophosphate were determined by QuikChem Methods 10-115-01-1-E and 10-115-01-1-A.

The pilot plant was optimized for both source waters and operated over a range of temperatures. Filter turbidity and head loss data were evaluated to determine whether direct filtration could achieve filter effluent turbidity requirements with a reasonable backwashing frequency (24+ hours per filter run). TOC removal data were compared for conventional and direct filtration and used to investigate whether direct filtration could achieve sufficient organics removal without driving up the cost of downstream GAC operations. Phosphorus removal data were assessed to see if direct filtration could meet phosphorus effluent requirements despite a lower coagulant dose. TSS challenge testing was performed to better understand the range of influent water quality viable for direct filtration. For NTP, an alternative hydraulic loading rate, EBCT, and coagulant were evaluated. The pilot testing schedule is displayed in Table 7.

Table 7: Pilot Testing Schedule

Year	2020												2021							2022									2023	
Month	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	⋋ 12	1	2	3	4	5	6	7	8 ⋋	1	2
YRTP		Normal Ops				TSS		Normal Ops																					
NTP														Normal Ops					↑EBCT	↑EBCT+Alum							TSS		

Results and Discussion

Table 8 summarizes pilot operating conditions after parameters were adjusted to best achieve the research objectives. ACH was used as a metal-salt coagulant for the YRTP study, and ACH and alum were both tested for the NTP study. In both pilot studies, direct filtration required only 17-21% of the coagulant dose used for conventional filtration. The chemical dose reduction was even more significant than proposed by Amirsardari et al. (1998) where a drinking water direct

filtration pilot needed 25-33% of the coagulant dose used at a full-scale conventional plant treating the same source water. The coagulant dose used for YRTP and NTP direct filtration (0.8-1.0 mg/L-Al) was comparable to the feasibility range (0.9 – 1.4 mg/L-Al) proposed by Hutchison (1976). NTP secondary clarifier effluent, however, with elevated solids and organic content, required a significantly higher polymer dose (5 mg/L) than both conventional treatment trains (0.75-1.0 mg/L) and YRTP direct filtration (1.0 mg/L). The higher polymer dose increases operating costs but is still within the range of expected doses (1 to 5 mg/L) indicated by Culp (1977) when a cationic polymer is used as a primary coagulant for direct filtration. For YRTP and NTP direct filtration, the total detention time for rapid mix and flocculation was 21 minutes, within the range of proposed 20- to 30-minute mixing period for minimum coagulant usage (Monscvitz et al., 1978).

Table 8: Pilot Plant Operating Parameters

	YRTP		NTP		
Criteria	**Direct**	**Conventional**	**Direct**	**Conventional**	**Unit**
Coagulant dose (ACH or alum)	0.8	3.8	1.0	6.0	mg/L -Al
Polymer dose	1.0	0.75	5.0	1.0	mg/L
Ozone Dose (0.8 O_3/TOC)	4.5-6.5	3.5-5.5	8-12	5-8	mg/L
Filter Aid Polymer	0.05	0.10	0.10	0.10	mg/L
Phosphoric Acid	0.05	0.05	0.05	0.05	mg/L-P
BAF EBCT	10	10	15	10	min
BAF Loading Rate	3.8	3.8	2.5	3.8	gpm/sf

Filter Operations

Filters were initially operated at a loading rate of 3.8 gpm/sf and an EBCT of 10 minutes for both the YRTP and NTP pilot studies. Direct filter operational data for the YRTP study is displayed in Figure 4. Direct filtration achieved an average filter runtime of 35±11 hours (n=59) for a unit filter run volume (UFRV) of 7,600±2,400 gal/sf. Filter head loss development was 0.23±0.05 ft/hr (n=59). Influent water temperature ranged from 14°C to 26°C with a mean of 19°C. Average pilot influent turbidity was 1.1±0.7 (n=32). Filter backwashes were triggered by a mix of turbidity and head loss limits. No significant relationship was observed between temperature and filter runtime. In comparison to direct filtration, conventional treatment achieved a longer average filter runtime of 105±68 hours for a UFRV of 24,000±15,400 gal/sf and a slower head loss development of 0.06±0.02 ft/hr (n=22).
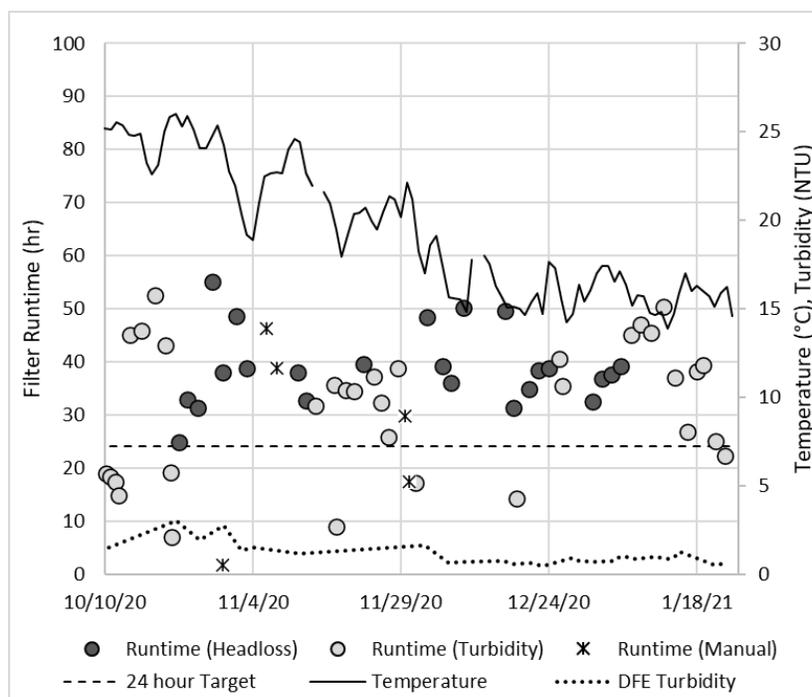
Figure 4: YRTP Pilot Direct Filter Operations – Runtimes (3.8 gpm/sf)

During the NTP pilot study, direct filtration achieved an average filter runtime of 19±9 hours (n=132) for a UFRV of 4,300±2,100 gal/sf (Figure 5). Influent water temperature ranged from 16°C to 27°C with a mean of 20°C. Head loss development was 0.35±0.16 ft/hr (n=132). Average pilot influent turbidity for this testing period was 2.8±1.4 NTU (n=149). During the first month of testing, influent turbidity was above average (4.0±1.5 NTU), temperature was at a minimum (16°C), and direct filter runtimes were below average (13±7 hours). Filter backwashes were primarily driven by turbidity breakthrough. While shorter filter runs were observed during lower temperatures, the relationship between temperature and filter runs was unclear as influent turbidity was higher and chemical doses were adjusted during this testing period. Coagulant and polymer doses were increased on the direct filtration treatment train which improved filter turbidity removal, indicated by the increased length of filter runs and frequency of head loss-triggered backwashes. In comparison to direct filtration, conventional filter runtimes were 65±45 hours for a UFRV of 15,000±10,000 gal/sf and a head loss development of 0.20±0.14 ft/hr (n=40).

YRTP direct filtration consistently met the 24-hour runtime goal, indicating that direct filtration of denitrification filter effluent could achieve filter effluent turbidity requirements without excessive backwashing for HRSD's SWIFT application. At the same hydraulic loading rate (3.8 gpm/sf), NTP direct filtration had significantly shorter filter runtimes, particularly during low temperature, high turbidity testing conditions and due to early turbidity breakthrough. Backwashes consistently triggered by turbidity or head loss alone may indicate potential for process optimization, where optimal filter performance typically occurs with simultaneous turbidity and head loss breakthroughs (Stump et al., 1979). In the NTP direct filtration test,
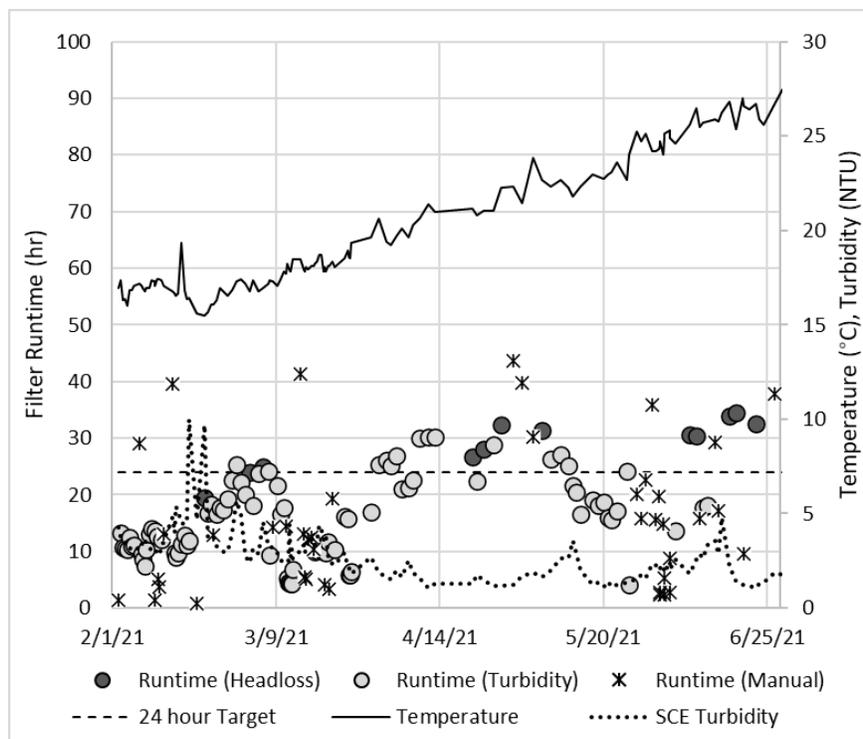
Figure 5: NTP Pilot Direct Filter Operations – Runtimes (3.8 gpm/sf)

increasing the coagulant doses resulted in increased filter runtimes and some head loss triggered backwashes. Coagulation affects turbidity removal as floc formation changes the turbidity and particle size distribution in the filter influent. As floc size and strength increase, particles are better removed through filtration and less likely to pass through the filter. Too much coagulant addition, however, could lead to shorter filter runs from chemical breakthrough or rapid head loss development. Overall, more particulate matter present in NTP secondary clarifier effluent than YRTP denitrification filter effluent resulted in shorter filter runtimes and the need for modifications to direct filtration operating parameters.

Based on these results, direct filtration for reuse applications may be suitable only for lower turbidity source waters and at lower hydraulic loading rates than used in drinking water

applications. In a drinking water treatment study by Logsdon et al. (1993), direct anthracite filters loaded at 6.0 gpm/sf reduced high source water turbidity (20-30 NTU) to below 0.10 NTU with 15 to 25 hour filter runtimes. McCormick & King (1982) found that direct coal/sand filters operated at 5.0 gpm/sf reduced influent turbidity as high as 10 NTU to less than 0.10 NTU with 15 to 30 hour filter runtimes, even at temperatures as low as 4-6 °C. In this pilot study, direct filtration of NTP secondary clarifier effluent achieved comparable effluent turbidity and runtimes, however, with a lower loading rate (3.8 gpm/sf) and a significantly lower source water turbidity (2.8 NTU). In Logsdon et al. (1993), source water TOC was relatively low (<2.0 mg/L) and coagulant doses were low (<0.7 mg-Al/L alum and <0.3 mg/L cationic polymer). Treated wastewater effluents generally contain greater concentrations of organics at a lower turbidity than typical groundwater or surface water sources, and the difference in particle characterization affects direct filtration operations.

The impact of hydraulic loading rate on direct filter runtimes was evaluated in a separate experiment, and filter runtimes increased with decreasing loading rates from 3.8 to 1.8 gpm/sf (Appendix G). A 2.5 gpm/sf loading rate was selected for further NTP testing. Lowering the loading rate from 3.8 to 2.5 gpm/sf, simultaneously increasing the EBCT from 10 to 15 minutes, resulted in increased direct filter runtimes from 19±9 hours (n=132) to 36±10 hours (n=74) (Figure 6). Average pilot influent turbidity during this testing period was 1.4±0.5 NTU (n=88), which was significantly lower than the previous testing period (2.8±1.4 NTU, n=149). Mean UFRV increased from 4,300 to 5,400 gal/sf. Head loss development decreased from 0.35±0.16 ft/hr (n=132) to 0.24±0.08 ft/hr (n=74). There was no clear relationship observed between temperature and filter runtime. Conventional filter runtimes were 86±67 hours (n=21) for a UFRV of 13,000±10,000 gal/sf and head loss development of 0.08±0.05 ft/hr (n=20) at the lower loading rate.
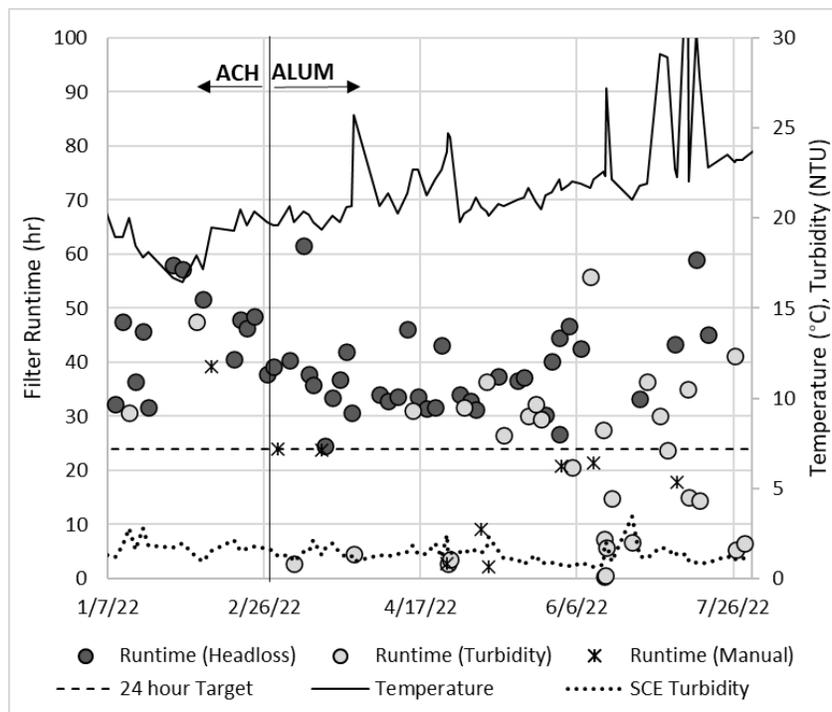
Figure 6: NTP Pilot Direct Filter Operations – Runtimes (2.5 gpm/sf)

In the NTP direct filtration study, a lower loading rate increased filter runtimes to achieve a reasonable backwashing frequency (24+ hour filter runs). A higher loading rate corresponds to more rapid particle collection and declining porosity that can lead to early head loss and turbidity breakthroughs. In a study by Westerhoff et al. (1980), direct filtration reduced turbidity from 1-100 NTU to 0.1-0.3 NTU at hydraulic loading rates of 2-6 gpm/sf. Variable influent TSS resulted in variable filter runs, and filter runtimes were most affected by the hydraulic loading rate. In the NTP pilot study, head loss accumulated faster at the higher loading rate, even with lower coagulant doses, compared to the lower loading rate. Head loss development for YRTP direct filtration at a 3.8 gpm/sf loading rate was comparable to NTP direct filtration at a 2.5 gpm/sf loading rate due to lower solids loading. Both head loss rates were faster than reported in drinking water studies. On a source water with comparable turbidity (1.6-2.0 NTU), direct filtration at a higher loading rate

of 6 gpm/sf produced finished water turbidities 0.04-0.15 NTU with slower head loss development of 0.1-0.2 ft/hr (Tate et al., 1977). This drinking water study, however, applied significantly lower alum and polymer doses (0.3 mg/L-Al and 0.25 mg/L polymer) and head loss development increased with increasing alum dose.

A higher loading rate could negatively impact turbidity removal as stronger shear forces may cause particle detachment, while a lower loading rate and a longer EBCT could improve turbidity removal due to reduced shear forces and extended contact time for particle collection. At the higher loading rate (3.8 gpm/sf), NTP direct filtration demonstrated less efficient turbidity removal, shown by more frequent turbidity breakthroughs and shorter filter runs than at the lower loading rate (2.5 gpm/sf). In a study on coagulated secondary effluent for tertiary filtration, filter effluent turbidity also increased with increasing loading rates from 5 gpm/sf to 10 gpm/sf (Williams et al., 2007). Alternatively, for conventional filtration in the NTP pilot study, while a lower loading rate decreased the head loss rate from 0.20±0.14 ft/hr to 0.08±0.05 ft/hr, no significant effect on turbidity removal was observed as backwashes were primarily triggered by head loss for both loading rates. Harrington (2003) also found no significant effect of filter loading rates from 2-8 gpm/sf on turbidity removal for conventional filtration.

Filter runs with alum as an upstream coagulant were shorter than with ACH for both conventional and direct filtration. Average filter runtime decreased from 43±8 hours with ACH (n=17) to 34±10 hours with alum (n=57). Average UFRV decreased from 6,500±1,200 gal/sf with ACH to 5,100±1,500 gal/sf with alum. Head loss development increased from 0.20±0.09 ft/hr (n=17) with ACH to 0.25±0.07 ft/hr (n=57) with alum. Conventional filter runtimes with ACH were 128±107 hours (n=4) for a UFRV of 19,000±16,000 gal/sf and head loss development of 0.10±0.10 ft/hr and with alum were 77±54 hours (n=17) for a UFRV of 12,000±8,100 gal/sf and a

head loss development of 0.07±0.03 ft/hr (n=16). Table 9 contains a summary of filter operations under the different testing conditions.

Table 9: Summary of Mean Filter Runtime, UFRV, and Head Loss Development

	Influent Turbidity (NTU)	Loading Rate (gpm/sf)	Coagulant Type	Treatment Train	Filter Runtime (hr)	UFRV (gal/sf)	Head Loss (ft/hr)
YRTP Denitrification Filter Effluent	1.1	3.8	ACH	Direct	35	7,600	0.23
				Conventional	105	24,000	0.06
NTP Secondary Clarifier Effluent	2.8	3.8	ACH	Direct	19	4,300	0.35
				Conventional	65	15,000	0.20
	1.4	2.5	ACH	Direct	43	6,500	0.20
				Conventional	128	19,000	0.10
			Alum	Direct	34	5,100	0.25
				Conventional	77	12,000	0.07

Switching coagulants from ACH to alum resulted in shorter filter runtimes and increased head loss development rates for both conventional and direct filtration. Despite shorter filter runs with alum, direct filtration was able to consistently meet the 24-hour runtime target with both coagulants. Filter effluent turbidity, on average, was higher with alum than with ACH but still below the 0.15 NTU limit for at least 24 hours. Flocs formed with ACH were smaller and

apparently denser than the flocs formed with alum. Larger alum floc may lead to a more rapid decline in filter bed porosity, increasing head loss development. Stronger, denser ACH floc may be more resistant to shear forces, decreasing effluent turbidity. Pilot conventional treatment displayed similar trends. Conventional coagulation and flocculation with alum formed larger and apparently less dense floc resulting in higher settled water turbidity (0.4-0.8 NTU) than with ACH (0.2-0.3 NTU) at an equivalent dose of aluminum. In jar tests for conventional coagulation, Yonge (2012) also observed that alum achieved less efficient turbidity removal than ACH.

UFRV measures the total volume of filtrate produced in a filter run and normalizes loading rate and runtime for a better filter performance comparison. For high-rate filters, a minimum UFRV of 5,000 gal/sf is recommended (Trussell et al., 1980). Direct filtration of YRTP denitrification filter effluent demonstrated acceptable performance with an average UFRV of 7,900 gal/sf. At 3.8 gpm/sf, direct filtration of NTP secondary clarifier effluent was below the proposed minimum with an UFRV of 4,300 gal/sf. At 2.5 gpm/sf, direct filtration achieved average UFRVs of 6,500 gal/sf with ACH and 5,100 gal/sf with alum, indicating greater and sufficient productivity per filter run.

While the average conventional filter runtime for YRTP was 105 hours, the maximum filter run was 224 hours (n=22). For NTP, average conventional filter runtime was 65 hours (n=40) at 3.8 gpm/sf and 86 hours (n=21) at 2.5 gpm/sf. Sporadic operational and mechanical issues caused premature backwashing and contributed to high variability in conventional filter runtimes for both YRTP and NTP, but overall, the data demonstrated conventional filters could remain in service for several days. Maximum filter runs for these loading rates were 178 hours and 202 hours, respectively. In full-scale practice, filters are backwashed after a time limit, in addition to effluent

turbidity and head loss limits. In this application, the conventional filter design runtime was 60 hours and direct filter runtime was 24 hours.

Assuming conventional filters are backwashed after 60 hours in service, approximately 1% of filtrate would be used for backwashing. Assuming direct filters are backwashed after 24 hours in service, approximately 4% and 6% of filtrate would be used for backwashing at 3.8 gpm/sf and 2.5 gpm/sf loading rates, respectively. Calculations are found in Appendix F. Results are comparable to drinking water applications, where direct filtration utilizes 4-8% of total water for backwash water and conventional utilizes 1-3% (Culp, 1977; Bryant & Brailey, 1980; McCormick & King, 1982). UFRV at a 3.8 gpm/sf loading rate would be 13,680 gal/ft for 60-hour conventional filters and 5,472 gal/ft for 24-hour direct filters. At a 2.5 gpm/sf loading rate, UFRV would be 3,600 gal/hr, which was less than recommended by Trussell et al. (1989), for a 24-hour direct filter run.

TOC Removal

Table 10 summarizes TOC removal data for when all filters were operated at a 10-minute EBCT (3.8 gpm/sf loading rate). In the YRTP study, average pilot influent (denitrification filter effluent) TOC was 6.9±0.6 mg/L (n=198); settled water TOC was 5.8±1.1 mg/L (n=200); conventional filter effluent TOC was 4.1±0.5 mg/L (n=193); and direct filter effluent TOC was 4.5±0.6 mg/L (n=193). For the conventional filtration process train, 16.0% TOC was removed through coagulation, flocculation, and sedimentation and an additional 24.5% TOC was removed through biofiltration for a total removal efficiency of 40.5%. The conventional filter removed 29.2% of filter influent (settled water) TOC. Direct filtration achieved 34.7% TOC removal.

Average pilot influent DOC was 6.7±0.6 mg/L (n=38) and settled water DOC was 5.5±0.5 mg/L (n=35). No significant relationship between temperature and TOC removal was observed (Appendix E).

In the NTP study, average pilot influent (secondary clarifier effluent) TOC was 9.9±1.9 mg/L (n=100); settled water TOC was 7.5±1.0 mg/L (n=88); conventional filter effluent TOC was 5.5±0.7 mg/L (n=82); and direct filter effluent TOC was 6.6±1.2 mg/L (n=98). For conventional filtration, 24.8% TOC was removed through coagulation, flocculation, and sedimentation and 19.4% TOC was removed through biofiltration for a total removal efficiency of 44.2%. The conventional filter removed 25.8% of filter influent (settled water) TOC. Direct filtration achieved 33.9% TOC removal. Average pilot influent DOC was 9.4±1.5 mg/L (n=22) and settled water DOC was 7.4±1.4 mg/L (n=20). No significant relationship between temperature and TOC removal was observed (Appendix E).

Additional TOC graphs can be found in Appendix D. YRTP and NTP pilot studies demonstrated comparable TOC removal efficiency for conventional treatment (41-44%) and comparable TOC removal for direct filtration (34-35%) at the same EBCT. In both studies, conventional treatment removed more TOC than direct filtration. Conventional filtration TOC removal in this study was similar to a pilot study by Vaidya (2019), where conventional ozone/biofiltration with a 10-minute EBCT achieved 44.8% TOC removal. NTP secondary clarifier effluent was also treated conventionally at the 1-MGD SWIFT Research Center and comparable TOC removal was observed. Coagulation/flocculation/sedimentation achieved 26% TOC removal and ozone/biofiltration achieved 30% TOC removal for a total TOC removal efficiency of 48% under similar operating conditions (6.25 mg-Al/L ACH, 0.8 O_3/TOC, 12-minute EBCT) (Hogard et al., 2021). An ozone/biofiltration pilot plant receiving sand filter effluent (no

Table 10: Comparison of TOC Removal for Conventional and Direct Filtration

Treatment Process	YRTP	NTP
Mean TOC Concentrations (mg/L)		
Pilot Influent	6.86	9.93
Settled Water	5.76	7.47
Conventional Filter Effluent	4.08	5.54
Direct Filter Effluent	4.48	6.56
Mean TOC Removal (%)		
Conventional Treatment	40.5	44.2
Flocculation/Sedimentation	16.0	24.8
Biofiltration	24.5	19.4
Direct Filtration	34.7	33.9

sedimentation) with a TOC of 8-11 mg/L, ozone dose of 0.9-2.0 O_3/TOC, and a 10-minute EBCT achieved 16-25% TOC removal (Sundaram & Pagilla, 2019). Another ozone/ biofiltration pilot study on treating membrane bioreactor effluent with an average TOC of 7.9±0.4 mg/L, ozone dose of 1.12 O_3/TOC, and a 10-minute EBCT achieved 25% TOC removal (Aquino, 2017; Gifford et al., 2018). At a 20-minute EBCT, ozone/biofiltration achieved nearly 30% TOC removal. In comparison, in the YRTP and NTP pilot studies, direct filtration at a 10-minute EBCT achieved greater TOC removal (34-35%), where coagulation contributes to additional physical removal during filtration. In filtration, organic matter is removed physically by the media or biologically by the attached biomass. A reuse study by Blair (2023) found that biofiltration primarily removed biodegradable organic carbon and had a marginal impact on non-biodegradable organic carbon, while upstream coagulation/flocculation/sedimentation primarily removed non-biodegradable organic carbon and a moderate amount of biodegradable organic carbon. The high removal

efficiency observed across the YRTP and NTP direct filters suggested that the organic matter present was biodegradable.

The impact of EBCT on TOC removal was tested in a separate experiment, and TOC removal increased with increasing EBCTs from 8 to 20 minutes for conventional and direct filtration (Appendix G). A 15-minute EBCT was selected for further testing. At a 15-minute EBCT with ACH as the upstream coagulant, NTP achieved 48.1% (n=20) TOC removal through conventional filtration and 39.2% (n=20) TOC removal through direct filtration. The average influent TOC was 9.1±1.1 mg/L, settled water TOC was 7.4±1.1 mg/L, conventional filter effluent TOC was 4.7±0.4 mg/L, and direct filter effluent TOC was 5.5±0.6 mg/L. For conventional filtration, 19.1% TOC was removed through coagulation/flocculation/ sedimentation and 28.9% TOC was removed through biofiltration. The conventional filter removed 35.8% of filter influent (settled water) TOC. Table 11 summarizes these findings.

Table 11: Comparison of Mean TOC Removal for Different EBCTs and Coagulant Types

Treatment Process	Mean TOC Removal		
	10-min EBCT, ACH	15-min EBCT, ACH	15-min EBCT, Alum
Conventional Treatment	44.2%	48.1%	39.6%
Settled Water	24.8%	19.1%	17.0%
Conventional Filter	19.4%	28.9%	22.6%
Direct Filtration	33.9%	39.2%	36.7%

Increasing the EBCT from 10 to 15 minutes on the pilot increased TOC removal for conventional and direct filtration, as expected. Greater gains in removal efficiency were observed for direct filtration compared to conventional treatment. Other studies also demonstrated that longer EBCTs, or more contact time with biomass, increased organics removal across a filter. In a pilot study by Vaidya (2019), a 5-minute EBCT biofilter removed less TOC (37.0%) than a 10-minute EBCT biofilter (44.8%). Blair (2023) similarly observed increased BDOC removal during biofiltration with a longer EBCT (10-minutes) than a shorter EBCT (5-minutes). Biological activity decreases along filter depth, and most removal of organics occurs near the top of a filter (Wang et al., 1995). Appendix G shows TOC removal down the filter column for NTP pilot conventional and direct filtration, and while TOC removal increased with media depth, most of the TOC removal occurred in the top portion of the filter. In a study comparing three full-scale ozone/biofiltration advanced treatment facilities, biofilter DOC removal increased, nonlinearly, with EBCT (Reungoat et al., 2012). In that study, with 9-, 18-, and 45-minute EBCTs, biofiltration achieved 17±2%, 25±6%, and 48±10% TOC removal, respectively. Readily biodegradable organic carbon was removed within the first few minutes of filtration, and with longer EBCTs, smaller gains in efficiency were observed due to less biodegradable organic matter being present.

At a 15-minute EBCT and switching coagulants from ACH to alum, NTP achieved 39.6% TOC removal through conventional treatment and 36.7% TOC removal through direct filtration (Table 11). The average influent TOC was 8.1±1.6 mg/L, conventional filter effluent was 5.0±0.7 mg/L and direct filter effluent was 5.2±0.6 mg/L. For conventional treatment, 17.0% TOC was removed through coagulation/flocculation/sedimentation and 22.6% TOC was removed through biofiltration. The conventional filter removed 27.2% of filter influent (settled water) TOC. Note

that there were significantly fewer samples collected for conventional filtration with alum because the coagulation/flocculation/sedimentation pilot was shut down due to mechanical issues.

A decrease in TOC removal efficiency was observed in both conventional and direct filtration treatment processes after switching coagulants from ACH to alum. Coagulant type affects TOC removal by altering the degree of particle destabilization. Yong (2012) observed ACH was more efficient at DOC removal than alum. Superior contaminant removal efficiencies with ACH could be due to its higher degree of polymerization and charge density (Zaman et al., 2021). This pilot study observed that ACH formed stronger, denser floc that could be more resistant to shear forces and less likely to pass through the filter compared to weaker, less dense floc formed with alum. Furthermore, variable influent TOC concentrations could influence average removal efficiencies, where larger influent concentrations are often associated with greater removal efficiencies. During NTP testing, average pilot plant influent (secondary clarifier effluent) TOC decreased over time. During pilot testing with ACH, influent TOC was 9.1±1.1 mg/L, and direct filtration removed 39.2% TOC for an effluent concentration of 5.5±0.6 mg/L (at a 15-minute EBCT). With alum, influent TOC was lower (8.2±1.6 mg/L), and direct filtration removed 36.7% TOC for an effluent concentration of 5.2±0.6 mg/L. In terms of TOC removal efficiency, ACH was superior to alum for organics removal for direct filtration; however, a comparison of coagulant performance with similar influent TOC concentrations is needed to reliably quantify the difference in removal efficiency. Additional TOC graphs for each testing condition can be found in Appendix D.

Overall, direct filtration achieved less TOC removal than conventional treatment. Differences in removal efficiency could lead to significant differences in operating costs. At the SWIFT treatment facility, GAC contactors that are downstream of biofiltration are used to remove

additional TOC to meet the 4 mg/L limit in the final effluent. TOC removal across coagulation and biofiltration can be optimized to reduce the frequency of GAC replacement. The 1-MGD SWIFT Research Center achieved 45% removal through conventional treatment, producing a biofilter effluent TOC of 5.1 mg/L[15].   In comparison, direct filter effluent TOC was lower in the YRTP pilot study (4.5 mg/L), and higher in the NTP pilot study (6.6 mg/L) at a 10-minute EBCT. At a 15-minute EBCT, direct filter effluent TOC was slightly higher (5.2-5.5 mg/L) than at the 1-MGD facility. Increased GAC replacement in response to higher biofilter effluent TOC could offset capital cost savings of direct filtration over time, and a life cycle cost estimate is necessary for a final recommendation.

Phosphorus Removal

Figure 7 and Figure 8 show pilot influent and effluent total phosphorus (TP) concentrations over time for YRTP and NTP, respectively. In the YRTP study, pilot influent (denitrification filter effluent) TP was 0.27±0.11 mg/L-P (n=39), with 0.22±0.13 mg-P/L (n=37) in the form of orthophosphate (OP). Conventional filter effluent TP was 0.06±0.03 mg/L-P (n=38) and direct filter effluent TP was 0.11±0.06 mg/L-P (n=38). In the NTP study, pilot influent (secondary clarifier effluent) TP was 0.71±0.58 mg/L-P (n=52), with 0.39±0.41 mg/L-P (n=52) in the form of OP. Direct filter effluent TP was 0.27±0.26 mg/L-P, with 0.27±0.28 mg/L-P (n=52) in the form of OP. The conventional filter pilot was out of service during this testing period. For a comparison, TP at the 1-MGD SWIFT Research Center (receiving the same NTP secondary clarifier effluent)

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[15] SWIFT Research Center monitoring data from 2/1/21 to 12/31/22

was sufficiently removed to 0.07 mg/L-P through conventional filtration (n=43)[16]. Average settled

water TP was 0.16 mg-P/L and OP was 0.03 mg/L-P (n=25).
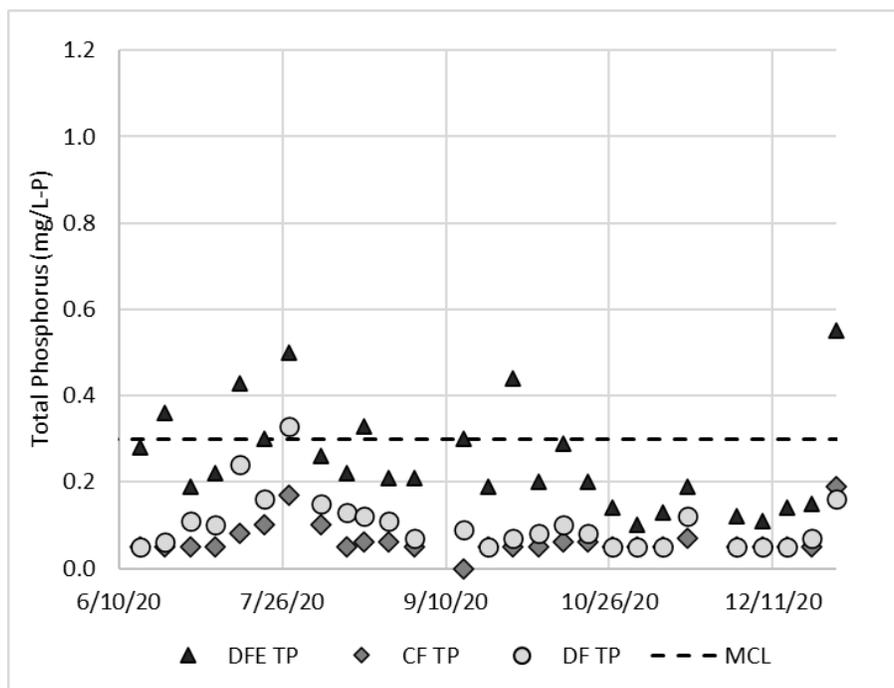
Figure 7: YRTP Pilot Total Phosphorus Concentrations for Conventional Treatment and Direct Filtration

During the YRTP study, pilot influent (denitrification filter effluent) total phosphorus

concentrations were consistently low, and the ACH dose (0.8 mg/L-Al) was sufficient to meet the

0.3 mg/L-P limit in the filter effluent. In the NTP study, pilot influent (secondary clarifier effluent)

total phosphorus concentrations were higher and more variable, and the low ACH dose (1.0 mg/L-

Al) was insufficient to consistently meet the 0.3 mg/L-P limit. Additional phosphorus could be

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[16] SWIFT Research Center monitoring data from 7/11/21 to 1/27/22

removed by increasing the ACH dose. A higher coagulant dose, however, could form large flocs that increase head loss development and shorten filter runs beyond practicality. An alternative coagulant type with greater phosphorus removal potential could be a preferred option.
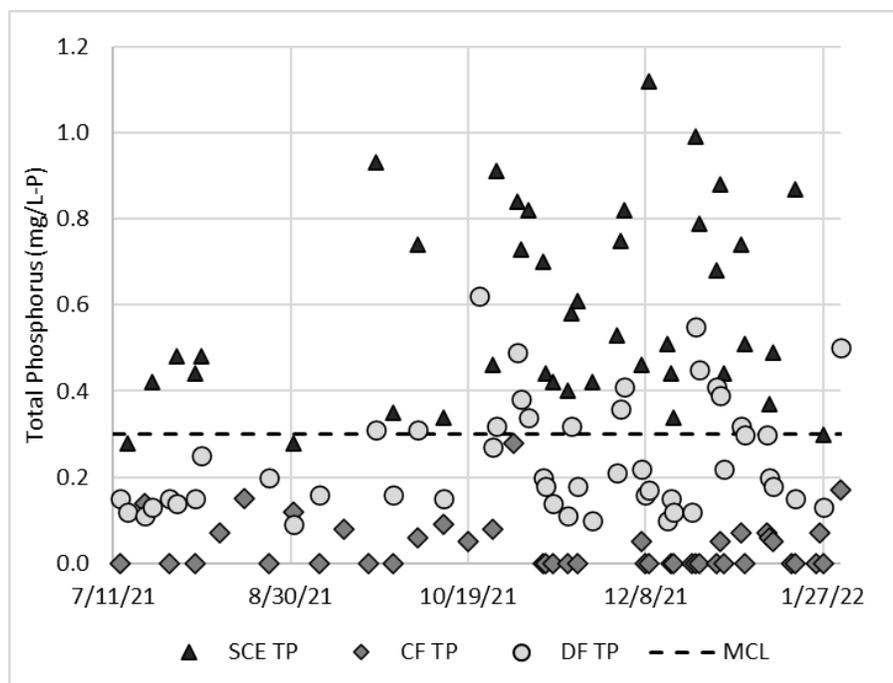
Figure 8: NTP Pilot Total Phosphorus Concentrations for Conventional Treatment and Direct Filtration

Jar testing determined alum could provide significantly more phosphorus removal than ACH for an equivalent dose of aluminum (Appendix H). Alum was subsequently tested on the pilot plant, and total phosphorus removal efficiencies of ACH and alum are compared in Figure 9. For NTP, 0.24±0.17 mol/L-P total phosphorus per mol/L-Al ACH was removed at the 10-minute EBCT (n=25) and 0.24±0.15 mol/L-P per mol/L-Al ACH was removed at the 15-minute EBCT (n=11). With alum and a 15-minute EBCT, 0.33±0.24 mol/L-P total phosphorus was removed per

mol/L-Al alum added (n=34). On a per mol-Al basis, alum removed 0.09 ± 0.29 mol/L-P more than ACH.
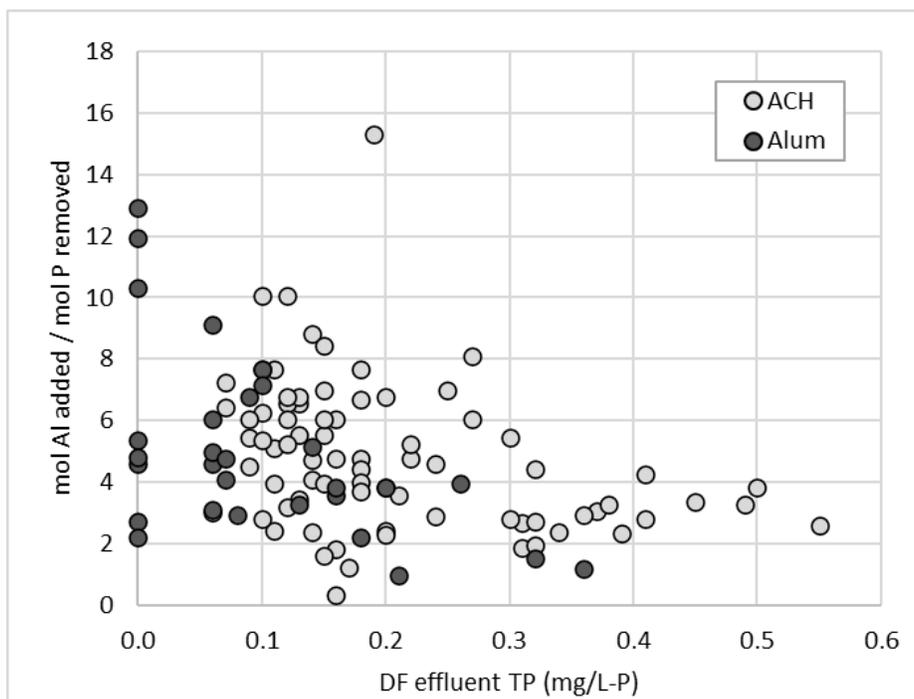
Figure 9: NTP Direct Filter Pilot – Total Phosphorus Removed Per Dose of Aluminum for ACH and Alum

In this study, at an equivalent dose of aluminum, ACH was superior to alum in TOC removal and alum was superior to ACH in phosphorus removal. The differences in removal efficiencies could be explained by their differing degrees of polymerization. The high polymerization of ACH generally leads to greater contaminant adsorptive capacity, as demonstrated with enhanced TOC removal efficiency (Zaman et al., 2021). The lower polymerization and larger monomeric fraction in alum, however, could contribute to better phosphorus removal than ACH. Trinh & Kang (2015) observed a relationship between phosphorus

removal efficiency and aluminum speciation, where greater phosphorus removal occurred with more monomeric aluminum species present. In addition, hydrolyzed aluminum coagulants may demonstrate less efficient phosphorus removal due to competition between hydroxyl ions and phosphate ions. In one study, aluminum sulfate without pre-hydrolysis and aluminum chloride with varying degrees of pre-hydrolysis of 0.0, 0.5, and 1.0 were compared for phosphorus removal, and phosphorus removal significantly decreased with increasing degrees of pre-hydrolysis (Diamadopoulos & Vlachoes, 1996). In the NTP study, alum removed significantly more phosphorus than ACH for an equivalent dose of aluminum. The low alum dose (1.0 mg/L-Al), however, was still insufficient when influent phosphorus was particularly high (>0.6 mg/L-P).

Additional testing was performed to determine the alum dose required to sufficiently remove phosphorus during these slug events (Appendix I). Pilot study results demonstrated that direct filters could manage temporary spikes in alum doses up to 4 mg/L-Al to accommodate phosphorus slugs. To optimize filter runs and chemical costs, a sophisticated control system is desired to automate coagulant dose increases in response to elevated orthophosphate. While alum is generally less expensive than ACH and is more effective at phosphorus removal, alum consumes alkalinity which may necessitate downstream caustic addition for pH adjustment. The low doses applied for direct filtration in this study, however, had a minimal effect on pH and alkalinity. On the other hand, the higher coagulant doses required of the conventional treatment process would result in significant pH drop and alkalinity consumption and would likely require caustic addition.

TSS Challenge Testing

TSS challenge testing was used to evaluate direct filter performance during plant upsets or elevated influent solids loading. During YRTP testing, I pilot plant received YRTP secondary

clarifier effluent to simulate worst-case scenario upstream treatment (Figure 10). During this testing period, the average influent temperature was 17°C. Influent turbidity ranged from 1.4-8.7 NTU with a mean of 3.6 NTU (n=9) and influent TSS ranged from 2.0-49.0 mg/L with a mean of 10.4 mg/L (n=12). The average direct filter runtime was 34±8 hours with a head loss development of 0.30±0.07 ft/hr. The minimum direct filter runtime of 21 hours occurred with an above average TSS of 11.6 mg/L and turbidity of 7.0 NTU (n=2).
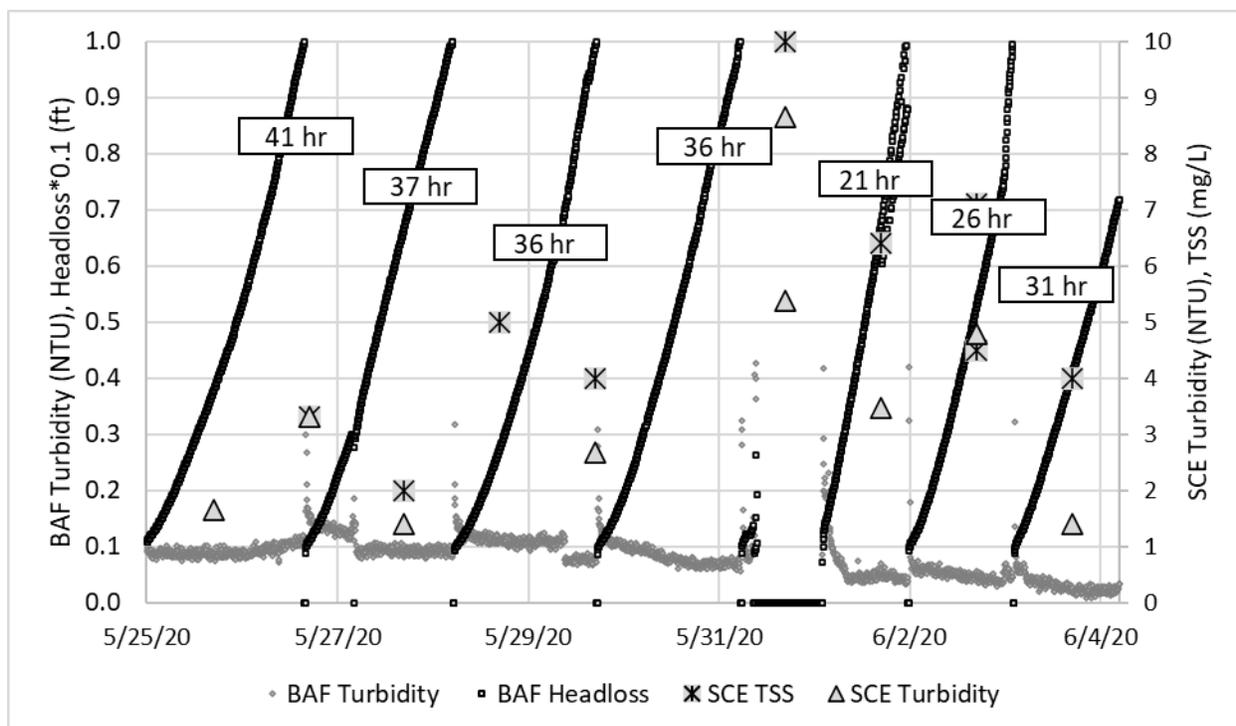
Figure 10: YRTP Direct Filter Pilot Operations During TSS Challenge Testing

The results of the NTP TSS challenge testing are displayed in Figure 11. After solids were spiked, TSS in the pilot influent ranged from 1.8-10.9 mg/L with an average of 5.3 mg/L (n=14) and turbidity ranged from 1.4-4.4 NTU with an average of 2.5 NTU (n=13) during this testing

period. Direct filter runtimes averaged 35±7 hours (n=4). In Phase 2, the direct filter achieved a

21-hour runtime with influent TSS ranging from 12.8-24.3 mg/L and turbidity from 3.9-8.0 NTU

and achieved a 25-hour runtime with TSS ranging from 7.7-9.2 mg/L and turbidity of 3.3 NTU.

Filter effluent turbidity was consistently higher in Phase 2 than Phase 1.
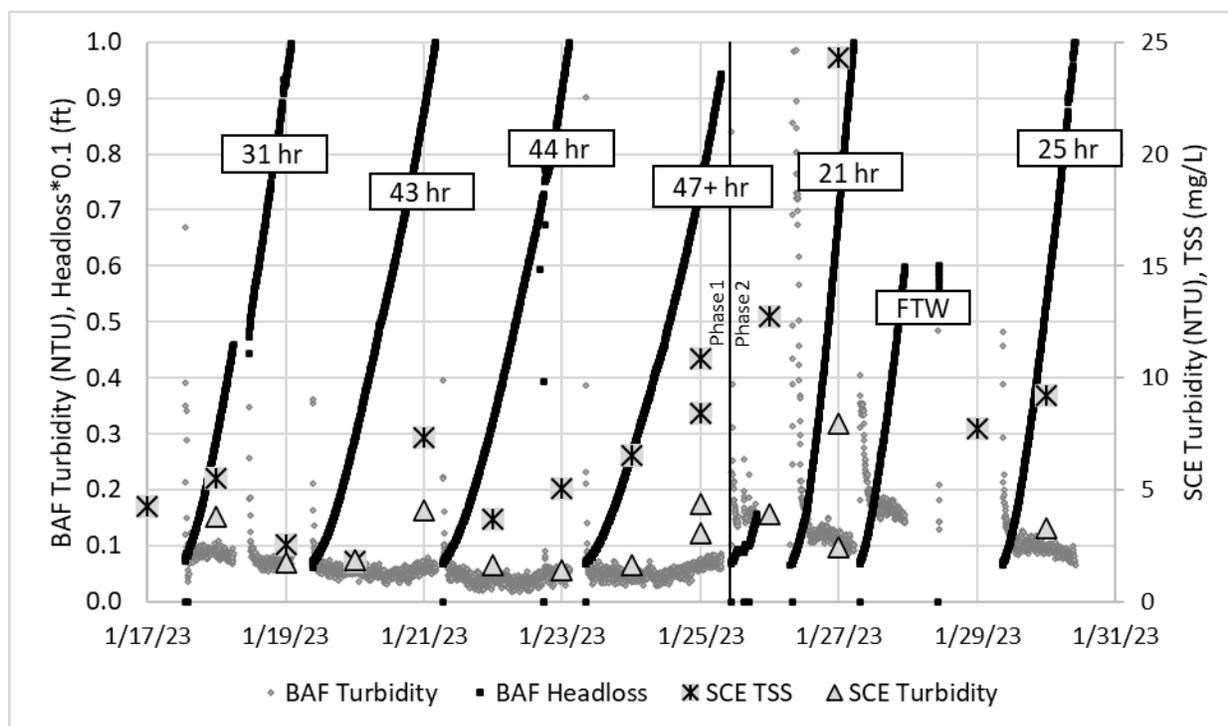
Figure 11: NTP Direct Filter Pilot Operations During TSS Challenge Testing

Direct filtration at YRTP managed spikes in influent turbidity up to 8.7 NTU and TSS up

to 49.0 mg/L and at NTP up to 8.0 NTU and 24.3 mg/L. Both tests demonstrated that direct

filtration could accommodate several days of elevated influent turbidity and TSS. Based on these

results, direct filtration could maintain reliable treatment in the event of comparable upstream

wastewater plant upsets or extreme weather events. Previously, at the higher loading rate (3.8

gpm/sf) and during high influent turbidity, low temperature events, the NTP direct filter was unable to consistently achieve 24-hour filter runs. At the lower loading rate (2.5 gpm/sf) and under similar influent turbidity and temperature conditions, the direct filter achieved target minimum filter runtimes.

Conclusions

Direct filtration, a potential cost-savings alternative to conventional drinking water treatment, can also be feasible for certain potable reuse applications. The pilot studies demonstrated that direct filtration of secondary and tertiary effluents could achieve HRSD's indirect potable reuse treatment goals associated with turbidity, TOC, and phosphorus removal.

- In the YRTP study, direct filtration reduced influent turbidity from 1.09±0.67 NTU to less than 0.15 NTU for 35-hour runtimes at a 3.8 gpm/sf loading rate. In the NTP study, direct filtration reduced influent turbidity from 2.8±1.4 NTU to less than 0.15 NTU for 19-hour runtimes at 3.8 gpm/sf loading rate and from 1.4±0.5 NTU for 36-hour runtimes at 2.5 gpm/sf loading rate. For the source water higher in turbidity and particulate matter, a lower loading rate was used to extend filter runs and avoid excessive backwashing.

- In both YRTP and NTP studies, at 10-minute EBCT, conventional filtration achieved 41-44% TOC removal and direct filtration achieved 34-35% TOC removal. For high TOC source waters, the difference of 6-10% between conventional and direct may be costly. A longer EBCT was used to increase TOC removal. At a 15-minute EBCT, NTP direct filtration achieved 39% TOC removal.

- Longer runtimes and more TOC removal were observed with ACH compared to with alum at an equivalent dose of aluminum. Switching from ACH to alum as a coagulant, NTP

direct filter runtimes decreased from 43 to 34 hours and TOC removal decreased from 39% to 37%.

- In the YRTP study, because influent phosphorus was consistently low (0.27±0.11 mg/L), the ACH dose of 0.8 mg/L-Al was sufficient to meet the 0.3 mg/L-P limit. In the NTP study, with influent phosphorus of 0.71±0.58 mg/L, the ACH dose of 1.0 mg/L-Al was insufficient.

- Alum demonstrated greater phosphorus removal than ACH for an equivalent dose of aluminum. Alum doses up to 4.0 mg/L-Al maintained direct filter runtimes greater than 24 hours.

- Direct filtration managed periods of elevated turbidity and TSS. In the YRTP study, direct filtration maintained 24-hour runs with turbidity spikes up to 8.7 NTU and TSS up to 49.0 mg/L. In the NTP study, direct filtration achieved target filter runtimes with turbidity up to 8.0 NTU and TSS up to 24.3 mg/L.

While the pilot studies demonstrated that direct filtration could meet water quality goals, direct filtration is not necessarily the best alternative. The capital cost savings provided by removing sedimentation from the treatment process could be offset by potential increased operating costs of GAC replacement and, if needed, increased capital costs from larger biofilters. Furthermore, the direct filtration treatment train may require more polymer usage and ozone generation. A recommendation requires a more extensive analysis, including important factors that were excluded from this study such as solids handling. If the costs of conventional treatment and direct filtration are comparable, then conventional treatment may be the preferred alternative, as direct filtration inherently encompasses more risk. Conventional treatment provides an additional barrier in the event of changing source water characteristics or an evolving regulatory

environment. The feasibility of direct filtration is ultimately dependent on source water quality and regulatory requirements; and implementation is dependent on a cost analysis.

CHAPTER V

ENGINEERING SIGNIFICANCE

Direct filtration is a potential cost-savings alternative to conventional treatment and is commonly employed at drinking water plants with low turbidity source waters. Based on results from this pilot study, direct filtration is also feasible for certain potable reuse applications. Potable reuse has gained attention as a solution to water supply challenges, where water scarcity poses a major threat to populations. While several proven advanced treatment technologies are available, implementation is costly, and resources are limited. Investigating cost-savings alternatives like direct filtration is important to overcome economic constraints and advance potable reuse. Facilities that plan on using carbon-based advanced treatment technologies can consider direct filtration as an alternative to conventional treatment. Ultimately, the feasibility of direct filtration is dependent on source water quality and regulatory requirements; and implementation is dependent on a cost-benefit analysis.

In this study, direct filtration was determined to be feasible for SWIFT advanced treatment of tertiary denitrification filter effluent from HRSD's York River Treatment Plant and secondary clarifier effluent from HRSD's Nansemond Treatment Plant. On the pilot plant, direct filtration of tertiary effluent with an average turbidity of 1.46±0.80 NTU, TSS of 2.11±0.78 mg/L, and total phosphorus of 0.27±0.11 mg/L-P met treatment goals at a 3.8 gpm/sf loading rate, 10-minute EBCT, and 0.8 mg-Al/L ACH dose. Direct filtration of secondary effluent with an average turbidity of 1.67±1.75 NTU, TSS of 4.49±3.22 mg/L, and total phosphorus of 0.70±0.61 mg/L-P met treatment goals at a 2.5 gpm/sf loading rate, 15-minute EBCT, and 1.0+ mg-Al/L alum dose. Results were based on a filter effluent turbidity limit of 0.15 NTU, a final effluent TOC limit of 4

mg/L, and a filter effluent phosphorus limit of 0.3 mg/L. While advanced treatment processes must comply with Safe Drinking Water Act requirements, states and local entities have authority to implement additional regulations. Utilities may operate filters for different turbidity targets, states may propose different effluent TOC standards, and potable reuse facilities may not need to meet phosphorus discharge requirements. In addition, following SWTR guidelines, direct filtration is awarded less pathogen log removal credit than conventional, and sedimentation may be required for a utility to meet log removal requirements. Furthermore, as new water quality regulations develop, GAC replacement may be driven by alternative compounds such as per- and poly-fluoroalkyl substances, and cost estimates should be adjusted accordingly. Consequently, the feasibility and costs of direct filtration change with regulatory environment.

The main objective of direct filtration is to meet treatment goals at a lower cost than conventional. While direct filtration met treatment goals for both the YRTP and NTP pilot studies, optimal design parameters differed, therefore costs differed. At YRTP, direct filtration achieved final effluent quality comparable to conventional treatment at similar design parameters; at NTP, direct filtration required larger biofilters to achieve final effluent quality comparable to conventional treatment. If larger filters are needed to accommodate higher solids loading, capital costs associated with biofiltration increase. Removing sedimentation and solids handling provides considerable capital and chemical cost savings, but reduced TOC removal efficiency may drive up the cost of downstream operations, and an increase in GAC replacement frequency could offset capital cost savings over time. A life cycle cost analysis is essential to determine if direct filtration is the best alternative.

Other important considerations for direct filtration not detailed in this study include solids handling, plant operations, and system controls. Conventional treatment may require a solids

handling facility for residuals management that involves capital costs, operating costs, and site footprint. Direct filtration likely eliminates the need for a separate solids handing facility, but residuals from filter backwash water must be properly managed. In addition, direct filtration may require more sophisticated instrumentation and controls than conventional. Since there is less time between coagulant addition and filter effluent in direct filtration, coagulant dose optimization or feed issues require a faster response time. To achieve phosphorus discharge requirements, facilities must plan for coagulant dose changes in response to influent phosphorus levels.

This study highlights the importance of pilot testing to inform full-scale design decisions. Pilot testing evaluated the range of influent water quality viable for direct filtration and investigated the effects of filter loading rate, EBCT, and coagulant type on direct filtration and conventional treatment. Ongoing research includes validating pathogen removal for conventional and direct filtration of secondary and tertiary effluents. Avenues for further research include optimizing flocculation mixing speeds and detention time, as flocculation design criteria was found to have a significant influence on filter operations in drinking water literature. Other parameters that can be further optimized include ozone dose, filter aid polymer, and filter backwashing sequence; and the effects of other types of coagulants and polymers may be of interest. Furthermore, a comparative analysis on the carbon footprint of conventional treatment and direct filtration could contribute to cost/benefit discussions.

REFERENCES

Amirsardari, Y., Yu, Q., & Williams, P. (1998). Effects of Pre-Ozonation of High Humic Content Waters on Direct Filtration in a Pilot Plant. *Environmental Technology, 19*(11), 1103-1110, https://doi.org/10.1080/09593331908616768

Aquino, M. (2017). *Impacts of Ozone Dose and Empty Bed Contact Time on Bulk Organic Removal and Disinfection Byproduct Mitigation in Ozone-Biofiltration Systems.* UNLV Theses, Dissertations, Professional Papers, and Capstones. 2937. http://dx.doi.org/10.34917/10985743

Aurora Colorado. (n.d.). *Drinking Water.* https://www.auroragov.org/residents/water/water_system/water_treatment/drinking_water

Becker, W.C. & O'Melia, C.R. (2001). Ozone: its effect on coagulation and filtration. *Water Supply, 1*(4), 81-88. https://doi.org/10.2166/ws.2001.0070

Beniwal, D., Taylor-Edmonds, L., Armour, J., & Andrews, R.C. (2018). Ozone/peroxide advanced oxidation in combination with biofiltration for taste and odour control and organics removal. *Chemosphere, 212*, 272-281. https://doi.org/10.1016/j.chemosphere.2018.08.015

Blair, M.F. (2023). *Profiling of Microbial Communities, Antibiotic Resistance, Functional Genes, and Biodegradable Dissolved Organic Carbon in a Carbon-Based Potable Water Reuse System.* Virginia Tech Doctoral Dissertations. 16743. http://hdl.handle.net/10919/ 114126

Bolto, B. & Gregory, J. (2007). Organic polyelectrolytes in water treatment. *Water Research, 41*(11), 2301-2324. https://doi.org/10.1016/j.watres.2007.03.012

Bryant, E.A., & Brailey, D. (1980). Large-scale ozone-DE filtration: an industry first. *Journal (American Water Works Association), 72*(11), 604-611. http://www.jstor.org/stable/41269942

Burns, S.K., Adams, V.D., & Maloney, S.B. (1984). *Direct Filtration Versus Conventional Water Treatment in the Intermountain Region.* Utah Water Research Laboratory Reports, Paper 580. https://digitalcommons.usu.edu/water_rep/580

Charlevoix. (n.d.). *Water Treatment Plant.* http://www.charlevoixmi.gov/191/Water-Treatment-Plant

Culp, R. L. (1977). Direct Filtration. *Journal (American Water Works Association), 69*(7), 375-378. https://doi.org/10.1002/j.1551-8833.1977.tb06769.x

Diamadopoulos, E., & Vlachoes, C. (1996). Coagulation-filtration of a secondary effluent by means of pre-hydrolyzed coagulants. *Water Science and Technology, 33*(10-11), 193-201. https://doi.org/10.1016/0273-1223(96)00420-9

Edwards, M., Benjamin, M. M., & Tobiason, J. E. (1994). Effects of ozonation on coagulation of NOM using polymer alone and polymer/metal salt mixtures. *Journal (American Water Works Association), 86*(1), 105–116. http://www.jstor.org/stable/41294487

Edzwald, J., Becker, W.C., & Tambini, S.J. (1986). Organics, Polymers, and Performance in Direct Filtration. *Journal of Environmental Engineering, 113*(1). https://doi.org/10.1061/(ASCE)0733-9372(1987)113:1(167)

Eyvaz, M., Deniz, H., Aktas, T.S., Yuksel, E., & Saatci, A.M. (2010). Pre-Ozonation-Coagulant Interactions in Direct Filtration. Water Quality Research Journal, 45(3), 317-326. https://doi.org/10.2166/wqrj.2010.034

Gifford, M., Selvy, A., & Gerrity, D. (2018). Optimizing Ozone-Biofiltration Systems for Organic Carbon Removal in Potable Reuse Applications. *Ozone: Science & Engineering, 40*(6), 427-440. https://doi.org/10.1080/01919512.2018.1509203

Harrington, G.W., Xagoraraki, I., Assavasilavasukul, P. & Standridge, J.H. (2003). Effect of Filtration Conditions On Removal of Emerging waterborne pathogens. *Journal (American Water Works Association), 95,* 95-104. https://doi.org/10.1002/j.1551-8833.2003.tb10514.x

Hogard, S., Salazar-Benites, G., Pearce, R., Nading, T., Schimmoller, L., Wilson, C., Heisig-Mitchell, J., & Bott, C. (2021). Demonstration-scale evaluation of ozone-biofiltration-granular activated carbon advanced water treatment for managed aquifer recharge. *Water Environment Research, 93*(8), 1157-1172. https://doi.org/10.1002/wer.1525

Hozalski, R. M. (1996). *Removal of biodegradable organic matter in drinking water biofilters: Experimental studies and model development* (Order No. 9629439). Available from ProQuest Dissertations & Theses Global. (304312092). http://proxy.lib.odu.edu/login?url=https://www.proquest.com/dissertations-theses/removal-biodegradable-organic-matter-drinking/docview/304312092/se-2

HRSD. (n.d.). *SWIFT*. https://www.hrsd.com/swift

Hutchison, W. R. (1976). High-Rate Direct Filtration. *Journal (American Water Works Association), 68*(6), 292–298. http://www.jstor.org/stable/41268463

LeChevallier, M. W., Becker, W. C., Schorr, P., & Lee, R. G. (1992). Evaluating the Performance of Biologically Active Rapid Filters. *Journal (American Water Works Association), 84*(4), 136–146. http://www.jstor.org/stable/41294277

Logsdon, G. S., Neden, D. G., Ferguson, A. M. D., & LaBonde, S. D. (1993). Testing Direct Filtration for the Treatment of High-Turbidity Water. *Journal (American Water Works Association), 85*(12), 39–46. http://www.jstor.org/stable/41294458

McCormick, R. F., & King, P. H. (1982). Factors that affect use of direct filtration in treating surface waters. *Journal (American Water Works Association), 74*(5), 234–242. http://www.jstor.org/stable/41271322

Melo, L.D.V, Barroso, G.R., Figueiredo, R.S, Costa, E.P., & Oliveira, S.C. (2021). Applicability of statistical analysis for performance and reliability evaluation of large-scale water treatment plants with direct filtration systems. *Environmental Science and Pollution Research, 28*, 22427-22438. https://doi.org/10.1007/s11356-020-12288-5

Monscvitz, J. T., Rexing, D. J., Williams, R. G., & Heckler, J. (1978). Some Practical Experience in Direct Filtration. *Journal (American Water Works Association), 70*(10), 584–588. http://www.jstor.org/stable/41269339

Mukherjee, J. (1999). *Evaluation of virus removal during direct filtration* (Order No. 9933641). Available from ProQuest Dissertations & Theses Global. (304499044). http://proxy.lib.odu.edu/login?url=https://www.proquest.com/dissertations-theses/evaluation-virus-removal-during-direct-filtration/docview/304499044/se-2

Ndiongue, S., Desjardins, R., Prévost, M. (2000). Relationships between total particle count, aerobic spore-forming bacteria and turbidity in direct filtration. *Journal of Water Supply: Research and Technology, 49*(2): 75–88. https://doi.org/10.2166/aqua.2000.0007

Nieminski, E.C., & Ongerth, J.E. (1995). Removing Giardia and Cryptosporidium by conventional treatment and direct filtration. *Journal (American Water Works Association), 87*(9), 96-106. https://doi.org/10.1002/j.1551-8833.1995.tb06426.x

Padhye, L., Luzinova, Y., Cho, M., Mizaikoff, B., Kim, J.-H., & Huang, C.-H. (2011). PolyDADMAC and Dimethylamine as Precursors of N-Nitrosodimethylamine during Ozonation: Reaction Kinetics and Mechanisms. *Environmental Science and Technology, 45*(10), 4353-4359. https://doi.org/10.1021/es104255e

Patania, N.L., Jacangelo, J.G., Cummings, L., Wilczak, A., Riley, K., & Oppenheimer, J. (1995). *Optimization of Filtration for Cyst Removal.* AWWA Research Foundation, USA.

Pernitsky, D.J., & Edzwald, J.K. (2006). Selection of alum and polyaluminum coagulants: principles and applications. *Journal of Water Supply: Research and Technology- Aqua, 55*(2), 121-141. https://doi.org/10.2166/aqua.2006.062

Peterson, E.S., & Summers, R.S. (2021). Removal of effluent organic matter with biofiltration for potable reuse: A review and meta-analysis. *Water Research, 199*, 117180. https://doi.org/10.1016/j.watres.2021.117180

Pharand, L., Van Dyke, M.I., Anderson, W.B., Yohannes, Y., & Huck, P.M. (2015). Full-Scale Ozone–Biofiltration: Seasonally Related Effects on NOM Removal. *Journal (American Water Works Association), 107*(8), E425–E435. https://www.jstor.org/stable/jamewatworass.107.8.e425

Pruden, A., Bott, C., Blair, M., Miller, J., & Vaidya, R. (2020). *Characterization of Organic Carbon and Microbial Communities for the Optimization of Biologically Active Carbon (BAC) Filtration for Potable Reuse.* (Project #4872). Water Research Foundation. https://www.waterrf.org/research/projects/characterization-organic-carbon-and-microbial-communities-optimization

Rebhun, M., Fuhrer, Z., & Adin, A. (1984). Contact flocculation-filtration of humic substances. *Water Research, 18*(8), 963-970. https://doi.org/10.1016/0043-1354(84)90246-X

Reungoat, J., Escher, B.I., Macova, M., Argaud, F.X., Gernjak, W., Keller, J. (2012). Ozonation and biological activated carbon filtration of wastewater treatment plant effluents. *Water Research, 46*(3), 863-872. https://doi.org/10.1016/j.watres.2011.11.064

Stump, V.L., & Novak, J.T. (1979). Polyelectrolyte Selection for Direct Filtration. *Journal (American Water Works Association), 71*(6), 338-342. https://doi.org/10.1002/j.1551-8833.1979.tb04364.x

Sundaram, V. & Pagilla, K. (2019). Trace and bulk organics removal during ozone-biofiltration treatment for potable reuse applications. *Water Environment Research, 92*(3), 430-440. https://doi.org/10.1002/wer.1202

Tanaka, T. S., & Pirbazari, M. (1986). Effects of Cationic Polyelectrolytes on the Removal of Suspended Particulates During Direct Filtration. *Journal (American Water Works Association), 78*(12), 57–65. http://www.jstor.org/stable/41290470

Tate, C. H., Lang, J. S., & Hutchinson, H. L. (1977). Pilot Plant Tests of Direct Filtration. *Journal (American Water Works Association), 69*(7), 379–384. http://www.jstor.org/stable/41268993

Tobiason, J. E., Edzwald, J. K., Schneider, O. D., Fox, M. B., & Dunn, H. J. (1992). Pilot Study of the Effects of Ozone and PEROXONE on In-line Direct Filtration. *Journal (American Water Works Association), 84*(12), 72–84. http://www.jstor.org/stable/41293945

Treweek, G. P. (1979). Optimization of Flocculation Time Prior to Direct Filtration. *Journal (American Water Works Association), 71*(2), 96–101. http://www.jstor.org/stable/41270231

Trinh, T.K., & Kang, L.S. (2015). Coagulation of phosphorus: effects of Al(III) species $(Al_a, Al_b,$ and $Al_c)$. *Desalination and Water Treatment, 53*(2), 485-492. https://doi.org/10.1080/19443994.2013.841106

Trussell, R. R., Trussell, A. R., Lang, J. S., & Tate, C. H. (1980). Recent developments in filtration system design. *Journal (American Water Works Association), 72(*12), 705–710. http://www.jstor.org/stable/41270507

Vaidya, R., Buehlmann, P., Salazar-Benites, G., Schimmoller, L., Nading, T., Wilson, C.A., Bott, C., Gonzalez, R., & Novak, J.T. (2019). *Pilot Plant Performance Comparing Carbon-Based and Membrane-Based Potable Reuse Schemes. Environmental Engineering Science, 36*(11), 1369-1378. https://doi.org/10.1089/ees.2018.0559

von Gunten, U. & Pinkernell, U. (2000). Ozonation of bromide-containing drinking waters: a delicate balance between disinfection and bromate formation. *Water Science and Technology, 41*(7), 53-59. https://doi.org/10.2166/wst.2000.0115

Wagner, E. G., & Hudson, H. E. (1982). Low-dosage high-rate direct filtration. *Journal (American Water Works Association), 74*(5), 256–261. http://www.jstor.org/stable/ 41271325

Wang, J.Z., Summers, R.S. & Miltner, R.J. (1995), Biofiltration performance: part 1, relationship to biomass. *Journal – American Water Works Association, 87*, 55-63. https://doi.org/10.1002/j.1551-8833.1995.tb06465.x

Water Technology. (n.d.). Seymour-Capilano Water Filtration Plant. https://www.water-technology.net/projects/72eymour-capilano/

Weng, C., Hoven, D. L., & Schwartz, B. J. (1986). Ozonation: An Economic Choice for Water Treatment. *Journal (American Water Works Association), 78*(11), 83–89. http://www.jstor.org/stable/41290450

Westerhoff, G. P., Hess, A. F., & Barnes, M. J. (1980). Plant-Scale Comparison of Direct Filtration Versus Conventional Treatment of a Lake Erie Water. *Journal (American Water Works Association), 72*(3), 148–155. http://www.jstor.org/stable/41270421

Yeh, H.-H., & Ghosh, M. M. (1981). Selecting polymers for direct filtration. *Journal (American Water Works Association), 73*(4), 211–218. http://www.jstor.org/stable/41271156

Yüksel, E., Akgiray, Ö., Saatçı, A.M., Sarıkaya, H.Z., & Koyuncu, I. (2002). Effect of ozone injection location on filter performance in direct filtration. *Water Science and Technology, 46*(9), 171-177. PMID: 12448466.

Zaman, N.K., Rohani, R., Yusoff, I.I., Kamsol, M.A., Basiron, S.A., & Rashid, A.I.A. (2021). Eco-Friendly Coagulant versus Industrially Used Coagulants: Identification of Their Coagulation Performance, Mechanism, and Optimization in Water Treatment Process. *Environmental Research and Public Health, 18,* 9164. https://doi.org/10.3390/ijerph 18179164

Zouboulis, A., Traskas, G., & Samaras, P. (2007) Comparison of single and dual media filtration in a full-scale drinking water treatment plant. *Desalination, 213*(1), 334-342. https://doi.org/10.1016/j.desal.2006.02.102.

APPENDICES

Appendix A – Pilot Plant Photos

Figure 12: Intuitech Flocculation Sedimentation Pilot

Figure 13: Rapid Mix and Flocculation Basins for Direct Filtration Pilot

Figure 14: Intuitech Ozone Pilots

Figure 15: Intuitech Biofiltration Pilots

Appendix B – Intermediate Ozonation Experiment

The pilot plant was reconfigured to compare direct filtration with pre-ozonation to direct filtration with intermediate ozonation. The original direct filtration treatment train remained unchanged. In the other treatment train, a single-stage rapid-mix tank with coagulant addition was placed upstream ozonation, and the ozone contactors and fine bubble diffusion were used for flocculation. The rapid-mix tank volume was 0.3 cubic feet with an HRT of 2.1 minutes and a G-value of 1000 s^{-1}. Both treatment trains received NTP secondary clarifier effluent. Temperature was controlled to 20°C, ozone was dosed at a 1.0 O$_3$/TOC ratio, and 0.05 mg-P/L phosphoric acid was added to both filter influents. Both treatment trains received alum and Clarifloc C-308P polymer. In the pre-ozonation treatment train, coagulant doses were 1.0-1.5 mg-Al/L alum and 5 mg/L polymer. In the intermediate ozonation treatment train, coagulant doses ranged from 1.0-4.0 mg-Al/L alum and 0.0-6.0 mg/L polymer. Filter-aid polymer doses ranged from 0.05-0.50 mg/L. In the treatment train with intermediate ozonation, filter effluent turbidity remained above 0.15 NTU across all chemical doses. Filter effluent turbidity reached a minimum (0.15 NTU) at the highest alum and polymer doses and with rapid head loss development. Furthermore, large floc formation was observed across ozonation and a blanket of solids (> 1 inch) formed on top of the filter media. In comparison, in the treatment train with pre-ozonation, filter effluent turbidity remained below 0.15 NTU and achieved filter runtimes greater than 24 hours. Intermediate ozonation was not a feasible alternative for NTP direct filtration.

The polyDADMAC polymer is a potential precursor for NDMA formation by ozone, particularly with bromide present (Padhye et al., 2011). NDMA samples were collected from the pilot influent, two ozone effluents, and two filter effluents (Table 12). NDMA samples were stored in the SRC laboratory fridge at 4°C and transported in coolers to HRSD's Central Environmental

Laboratory for analysis using EPA method 521. No significant difference in NDMA formation was observed between the two treatment trains. While the intermediate ozonation configuration had no significant effect on NDMA formation, the alternative configuration was ultimately rejected due to insufficient turbidity removal and concerns of solids buildup.

Table 12: NDMA Formation with Pre-Ozonation and Intermediate Ozonation

NDMA (ng/L)					
Date	Pilot Influent (SCE)	Ozone Effluent (Intermediate)	Ozone Effluent (Pre-)	Direct Filter Effluent (Intermediate)	Direct Filter Effluent (Pre-)
6/27/2022	<2.00	51.8	55.9	<2.00	<2.00
6/28/2022	3.56	59.5	56.1	<2.00	<2.00

Appendix C – Summary of all YRTP Pilot Direct Filtration Testing

YRTP direct filtration pilot testing was conducted from February 2020 through January 2021 and all of the direct filter runs are summarized in Figure 16. At startup, influent ("PF") temperature was controlled to 20°C. Jar testing was conducted to determine initial coagulant and polymer doses based on visual pin-floc formation for the direct filtration treatment train. Coagulant and polymer doses were further adjusted on the pilot plant to improve filter turbidity removal and head loss development. The coagulant doses that most consistently achieved filter runtimes greater than 24 hours were 0.8 mg-Al/L ACH and 1.0 mg/L polymer. For conventional treatment, 3.8 mg-Al/L ACH and 0.75 mg/L polymer were used. In the next phase of YRTP testing, secondary clarifier effluent from YRTP was studied on the pilot to evaluate direct filtration of more turbid source water. Afterwards, the pilot plant returned to study YRTP's denitrification filter effluent. Then, phosphoric acid (0.05 mg-P/L) was added to filter influent to enhance biological activity. After that, monochloramine was added prior to ozonation for bromate control. Then, hydrogen peroxide (1.0 H_2O_2/O_3) was evaluated as an alternative bromate control mechanism. Next, a nonionic filter aid polymer (0.05 mg/L for conventional, 0.10 mg/L for direct) was added to aid turbidity removal and slow head loss development. During a period of elevated influent turbidity and lower temperatures, the direct filter experienced several short runtimes as a result of turbidity breakthrough. In response, a series of backwashes was conducted on the direct filter to deep clean the media. The final phase of YRTP testing covered a range of temperatures from 14°C to 26°C with phosphoric acid, filter aid polymer, and hydrogen peroxide additions and ozone was dosed at a constant 0.8 O_3/TOC ratio. This testing period is referred to as "normal operations."
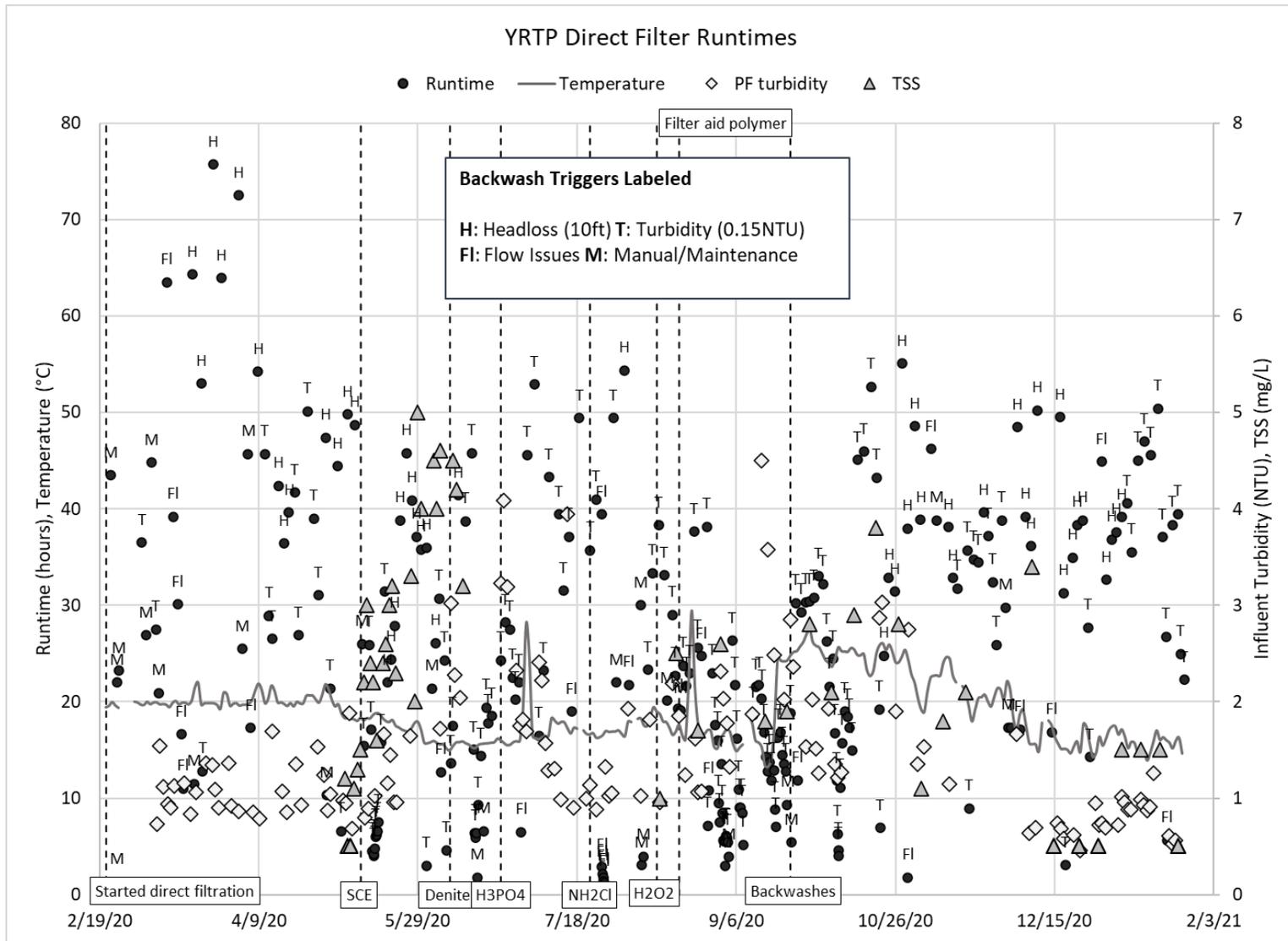
Figure 16: YRTP Direct Filter Operations Across the Entire Pilot Testing Period

Appendix D – TOC Removal Box Plots

Figure 17and Figure 18 display TOC concentrations for each treatment step in the YRTP and NTP pilot tests, respectively, under normal operating conditions (10-minute EBCT, 3.8 gpm/sf loading rate, and ACH).
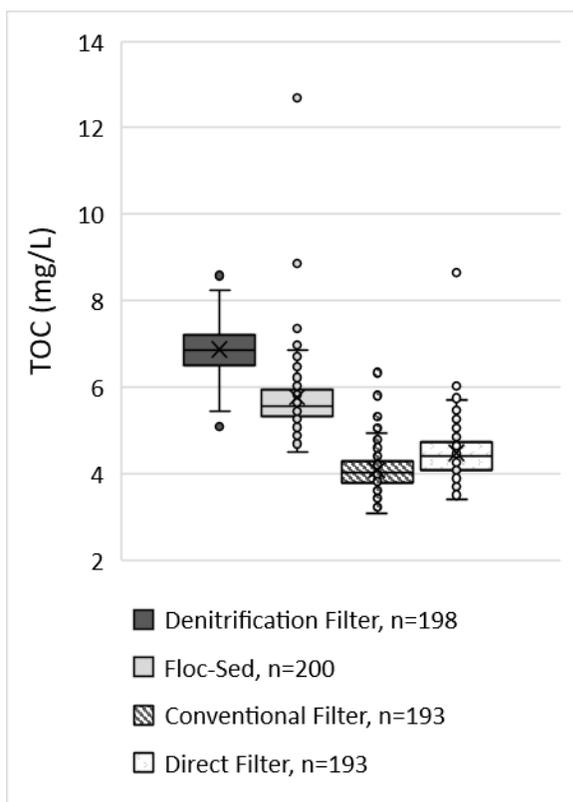
Figure 17: YRTP Pilot TOC Concentrations (10-min EBCT)
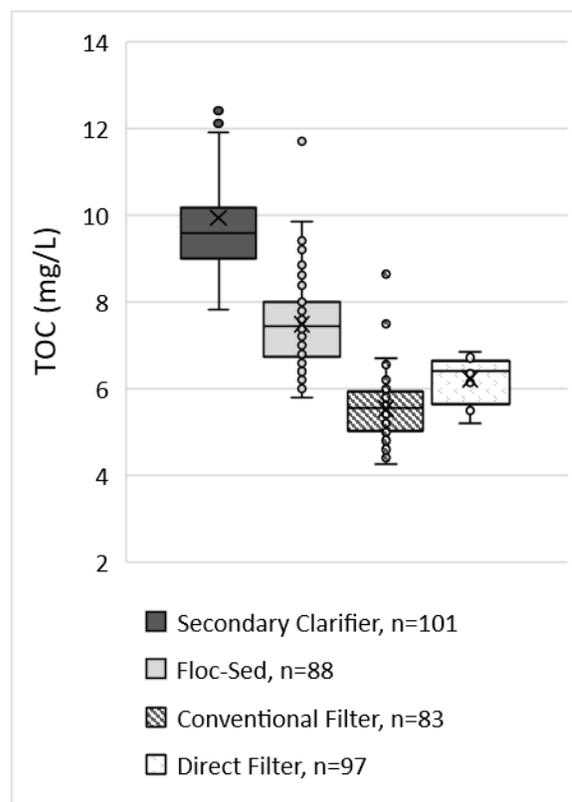
Figure 18: NTP Pilot TOC Concentrations (10-min EBCT)

Figure 19 and Figure 20 compare percent TOC removal across the entire pilot processes for conventional treatment and direct filtration for the YRTP and NTP pilot tests, respectively, under normal operating conditions. Figure 21 and Figure 22 show percent TOC removal for NTP

testing with a longer EBCT and with alum as an alternative coagulant, respectively. For all four of the graphs, conventional treatment TOC removal includes removal across coagulation, flocculation, sedimentation.
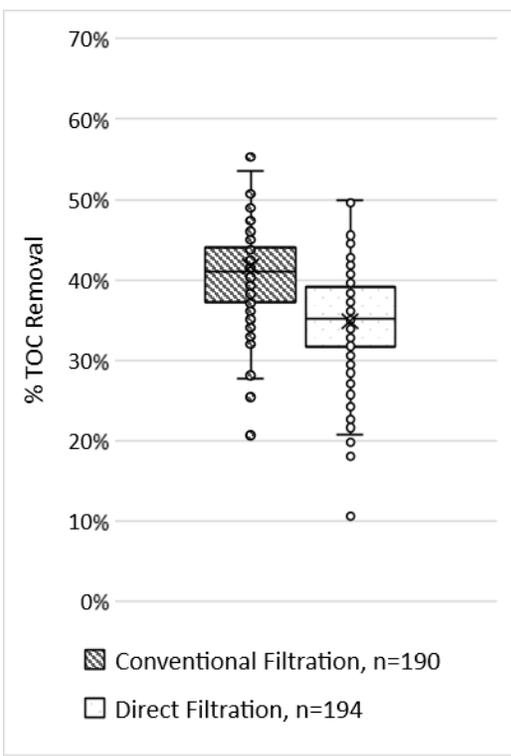
Figure 19: YRTP Pilot Percent TOC Removal (10-minute EBCT)
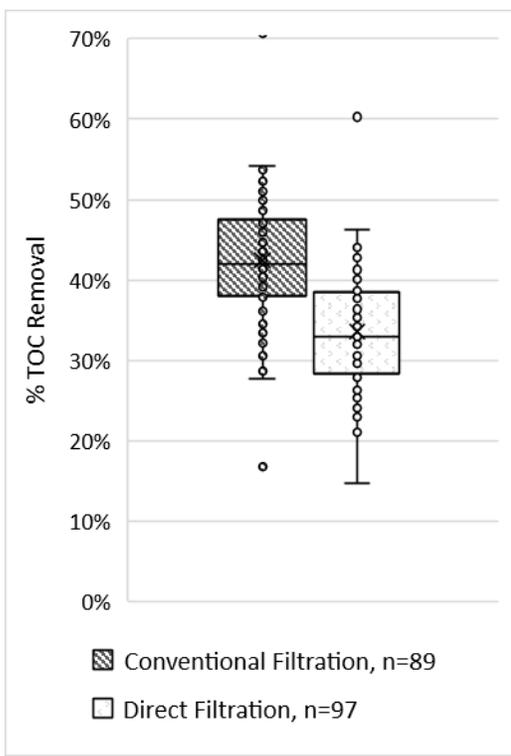
Figure 20: NTP Pilot Percent TOC Removal (10-minute EBCT)
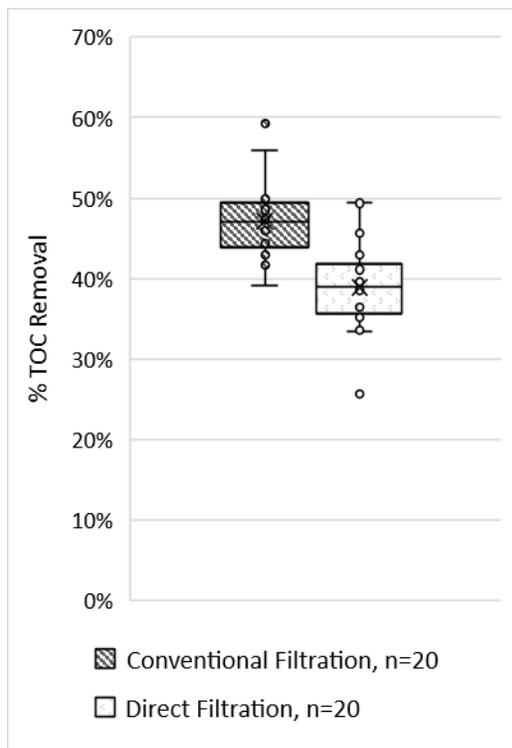
Figure 21: NTP Pilot Percent TOC Removal (ACH, 15-minute EBCT)
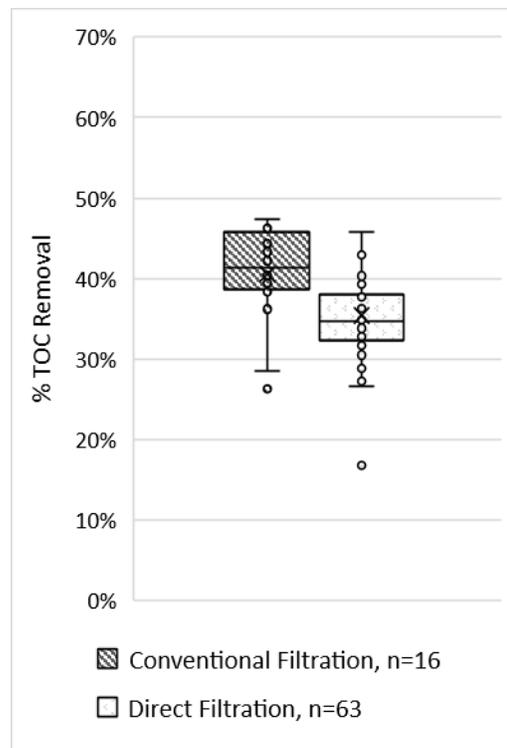
Figure 22: NTP Pilot Percent TOC Removal (Alum, 15-min EBCT)

Appendix E – Analysis of Temperature and TOC Removal

The pilot was operated at a constant filter loading rate (3.8 gpm/sf) and EBCT (10-minutes) for several months over a range of temperatures to simulate cold and warm weather conditions for YRTP and NTP. Temperature and TOC removal are plotted in Figure 23 and Figure 24 for YRTP and NTP, respectively.   No significant relationship between temperature and TOC removal was observed for either source water. It is important to note that the testing environment was not controlled to isolate the effect of temperature on TOC removal. Coagulant and polymer doses varied, and further testing is required to draw conclusions.
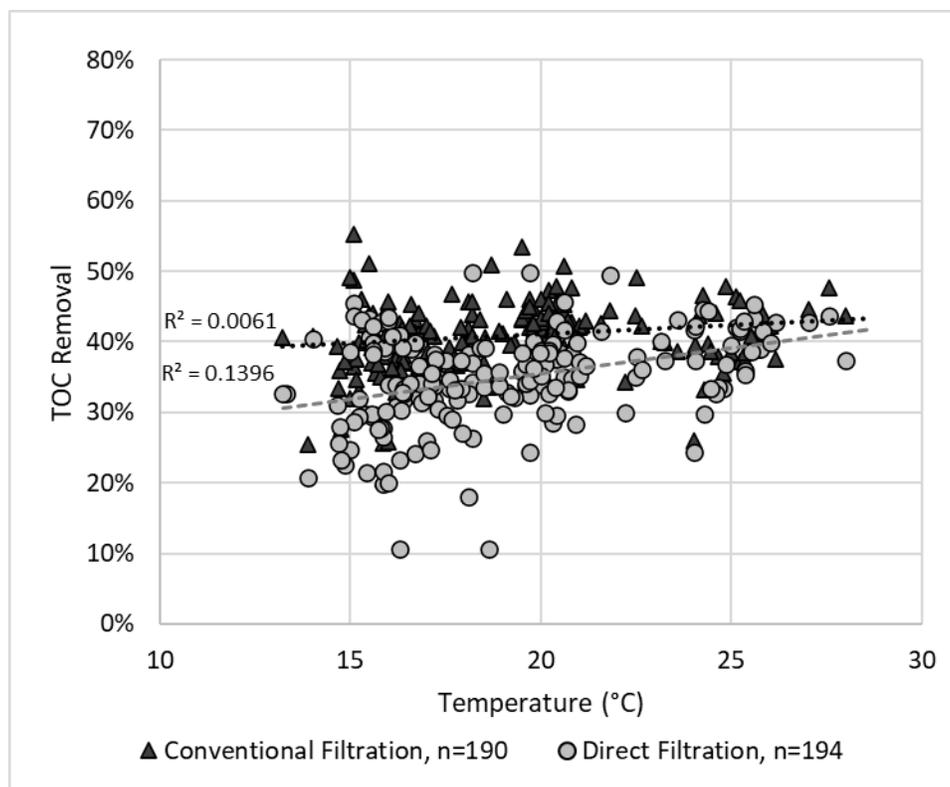
Figure 23: YRTP Pilot Temperature and TOC Removal
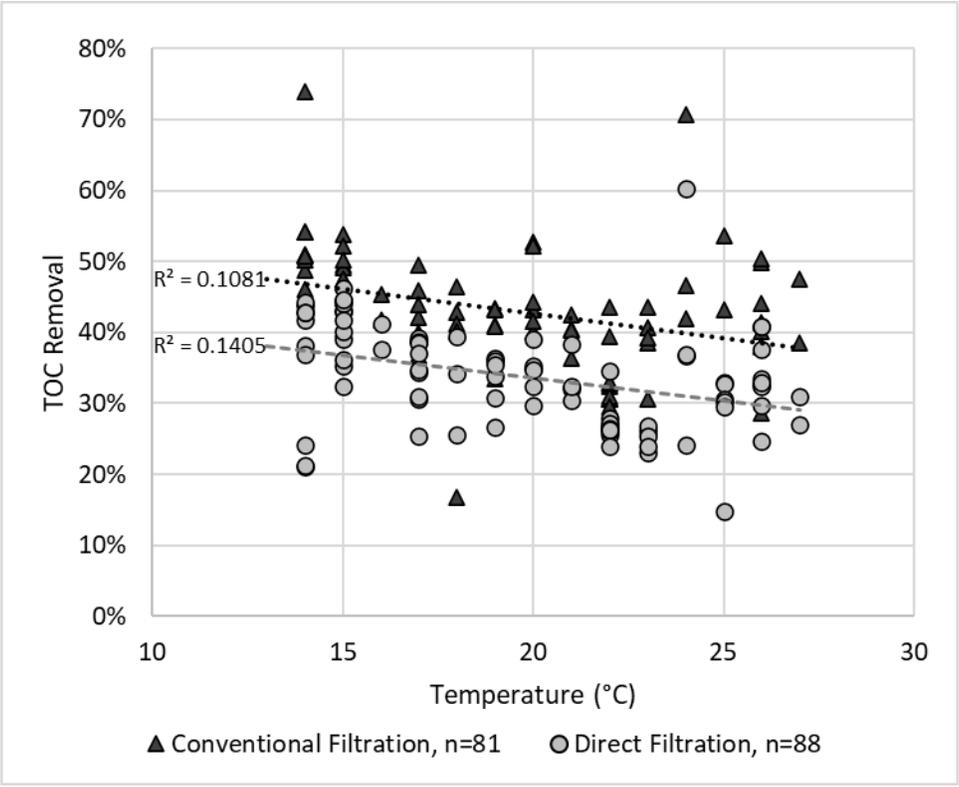
Figure 24: NTP Pilot Temperature and TOC Removal

Appendix F – Backwash Water Utilization Calculations

Table 13 displays values for calculating the pilot backwash water utilization.

Table 13: Backwash Water Utilization

	Conventional Filtration	Direct Filtration	
Loading Rate (gpm/sf)	3.8	3.8	2.5
Influent Flow Rate (gpm)	0.71	0.71	0.49
Runtime (hr)	60	24	24
Runtime (min)	3600	1440	1440
Volume (gal)	2556	1022	706
BW Volume (gal)	30	45*	45*
% BW Water	1.2%	4.4%	6.4%

*The same volume of water was used to backwash the direct filter at both loading rates because the filter dimensions were constant. In full-scale, a larger filter may require more backwash water. Note: filter to waste time (15-30 minutes) was not subtracted from filter runtimes.

Appendix G – Additional Loading Rate and EBCT Pilot Testing

Additional loading rate and EBCT pilot testing was performed on NTP secondary clarifier effluent. The impacts of loading rate on filter operations are shown in Figure 25 and Figure 26. Both pilot filters were operated as direct filters receiving the same ozone effluent and influent temperature was controlled to 20°C. Both filters initially operated at a 20-minute EBCT (1.8 gpm/sf loading rate) to confirm similar operations and validate the testing method. Then, one filter remained at a 20-minute EBCT as a control and the second filter tested a range of loading rates (2.4, 2.9, 3.3, and 3.8 gpm/sf) by adjusting the flow rate. Runtime and UFRV data are displayed in Table 14. Filter runtimes generally decreased nonlinearly with increasing loading rate, with no significant difference between 3.3 and 3.8 gpm/sf. The largest UFRV was at the highest loading rate and the lowest UFRV was at the lowest loading rate. No significant trend, however, was observed in UFRV for loading rates in between.
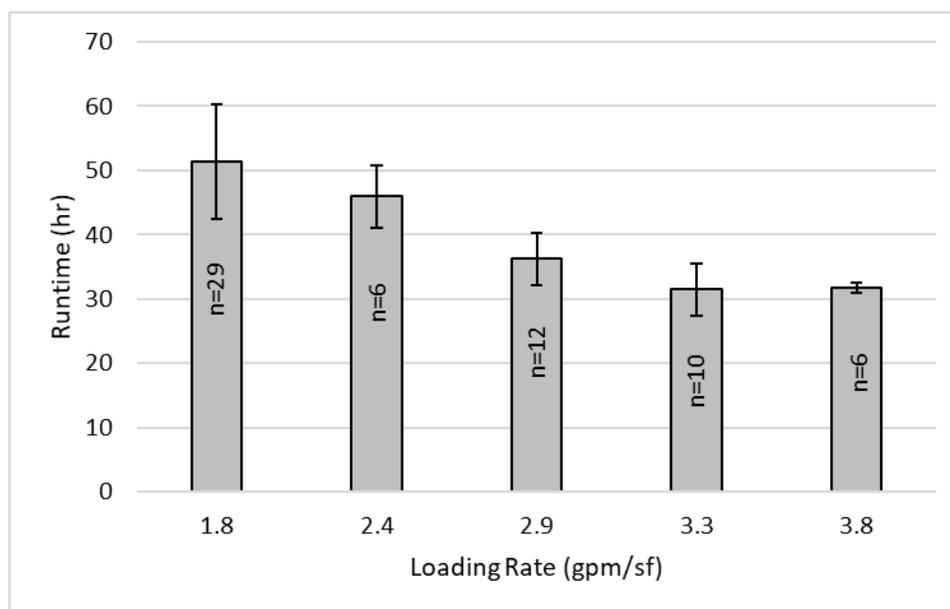
Figure 25: NTP Direct Filter Loading Rate and Filter Runtime
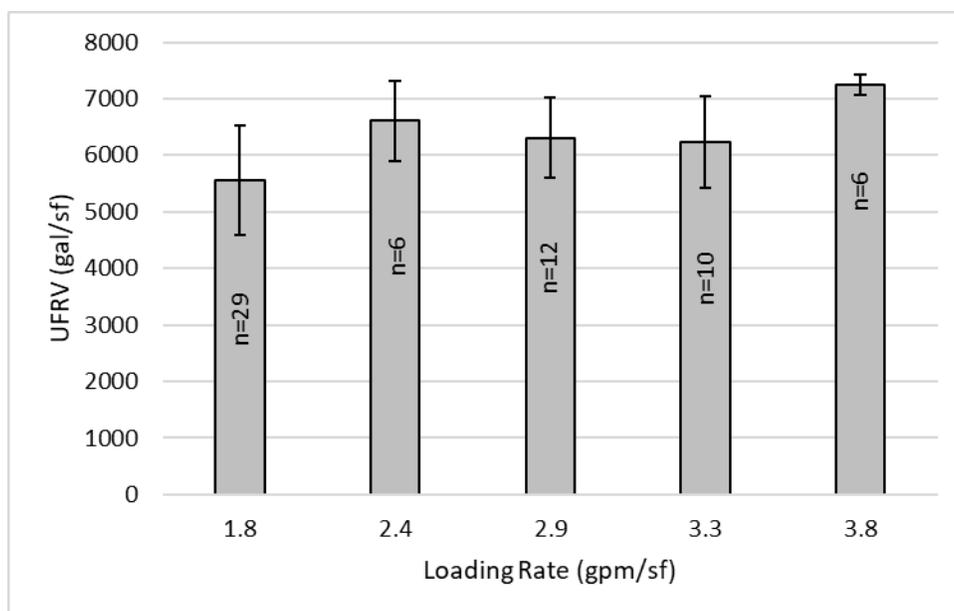
Figure 26: NTP Direct Filter Loading Rate and Unit Filter Run Volume

Table 14: Direct Filter Loading Rate, Filter Runtime, and UFRV

Loading Rate (gpm/sf)	Runtime (hr)	UFRV (gal/sf)
1.8	51 ± 9	5551 ± 970
2.4	46 ± 5	6612 ± 709
2.9	36 ± 4	6307 ± 715
3.3	31 ± 4	6235 ± 809
3.8	32 ± 1	7240 ± 175

To evaluate the effect of EBCT on TOC removal and model TOC removal across the filter column, one conventional filter and one direct filter were operated at 20-minute EBCTs (1.8 gpm/sf loading rate), and TOC samples were collected along each filter column to represent 8-, 12-, 16-, and 20-minute EBCTs. TOC removal data are displayed in Figure 27, Figure 28, and

Table 15. TOC removal increased along each filter column as EBCT increased. Most of the TOC was removed in the top portion of each filter column, and additional gains in TOC removal efficiency were observed for EBCTs up through 20 minutes for conventional and direct filtration. While greater TOC removal efficiency was observed along the direct filter compared to the conventional filter alone, overall conventional treatment removed more TOC than direct filtration.
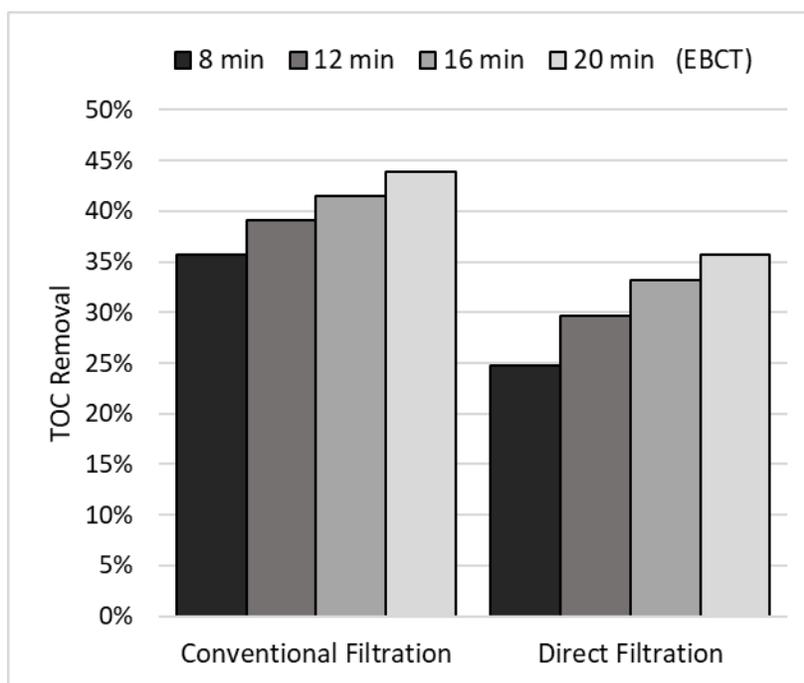
Figure 27: TOC Removal Along the Filter Columns (Including Upstream Treatment)
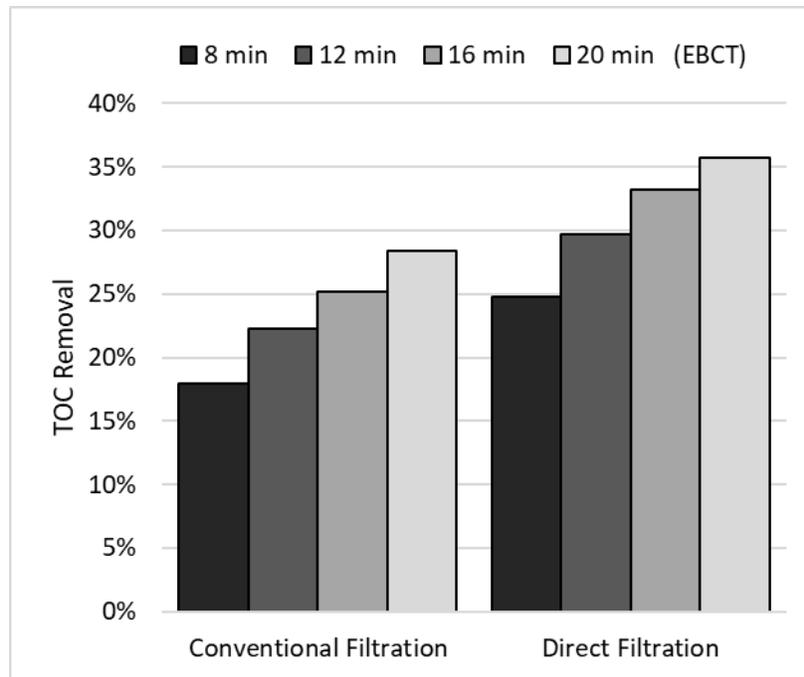
Figure 28: TOC Removal Along the Filter Columns (Filtration Only)

Table 15: Summary of TOC Removal Along the Filter Columns

(n = 8)	Conventional Filtration				Direct Filtration	
EBCT (min)	Total Mean (%)	Std. Dev. (%)	Filter Only Mean (%)	Std. Dev. (%)	Total Mean (%)	Std. Dev. (%)
8	35.8	3.9	18.0	3.4	24.8	5.2
12	39.1	2.6	22.3	2.5	29.7	3.5
16	41.5	2.5	25.2	2.6	33.2	2.1
20	43.9	1.8	28.4	1.9	35.7	1.6

Appendix H – Jar Testing for Phosphorus Removal

Jar testing was performed to compare the performance of ACH and alum as coagulants for direct filtration. Three test conditions were studied – ACH, alum, and alum with caustic addition. In the caustic addition condition, sodium hydroxide was dosed to isolate the effects of pH and alum on flocculation. Three jar tests were performed for each testing condition and results are displayed in Figures 29-34. Orthophosphate was analyzed with Hach TNT 843 for low range reactive phosphorus on the Hach DR3900. pH was measured using Thermo Scientific Orion Star pH probe and benchtop meter. Alkalinity was analyzed using low range alkalinity chemkeys on Hach's SL1000 Portable Parallel Analyzer. Zeta potential was measured on the Malvern Zetasizer Nano ZS. UV254 absorbance was measured on the Thermo Scientific Genesys 150 UV-Visible Spectrophotometer. DOC was analyzed on the Shimadzu 4200. Samples for OP, DOC, and UV absorbance analyses were filtered with a syringe and Environmental Express polyvinylidene fluoride membrane syringe filters (25 mm, 0.45 μm).
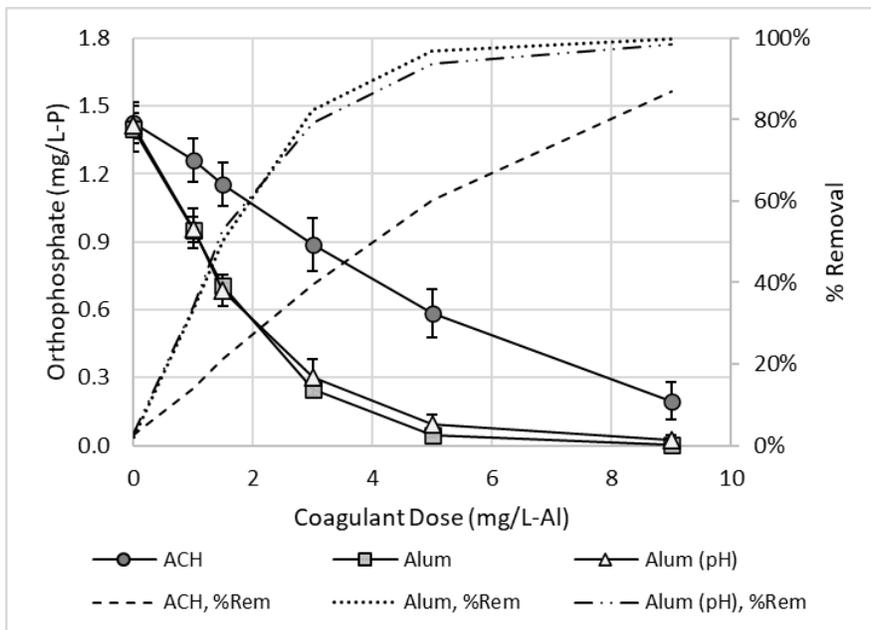
Figure 29: Jar Testing Results – Orthophosphate Removal
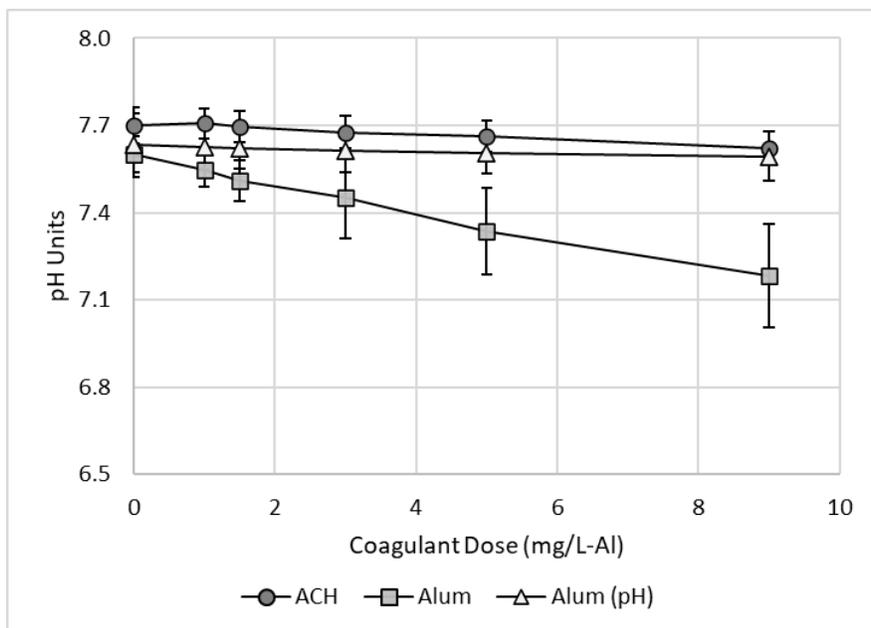
Figure 30: Jar Testing Results – pH Drop
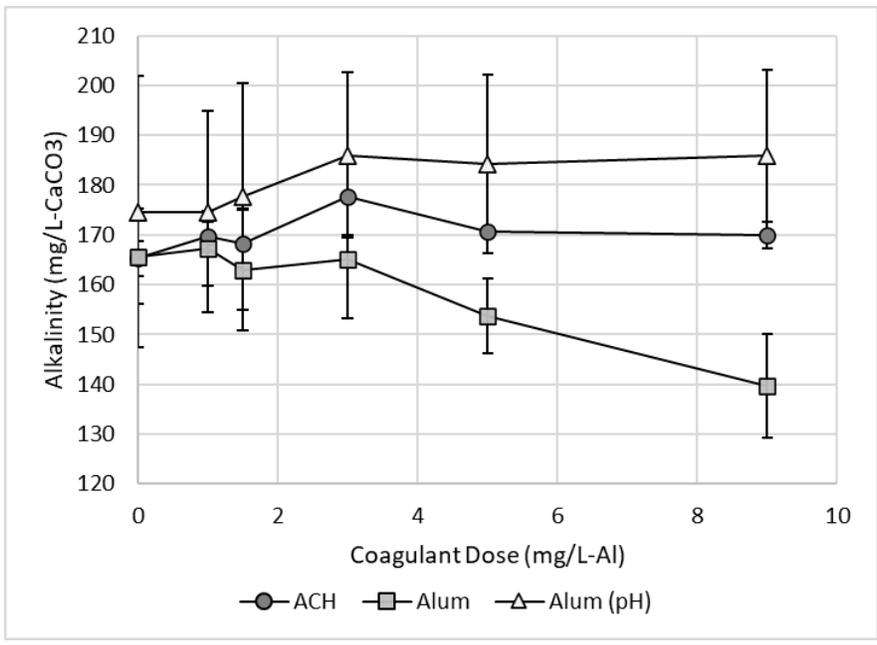
Figure 31: Jar Testing Results – Alkalinity Consumption
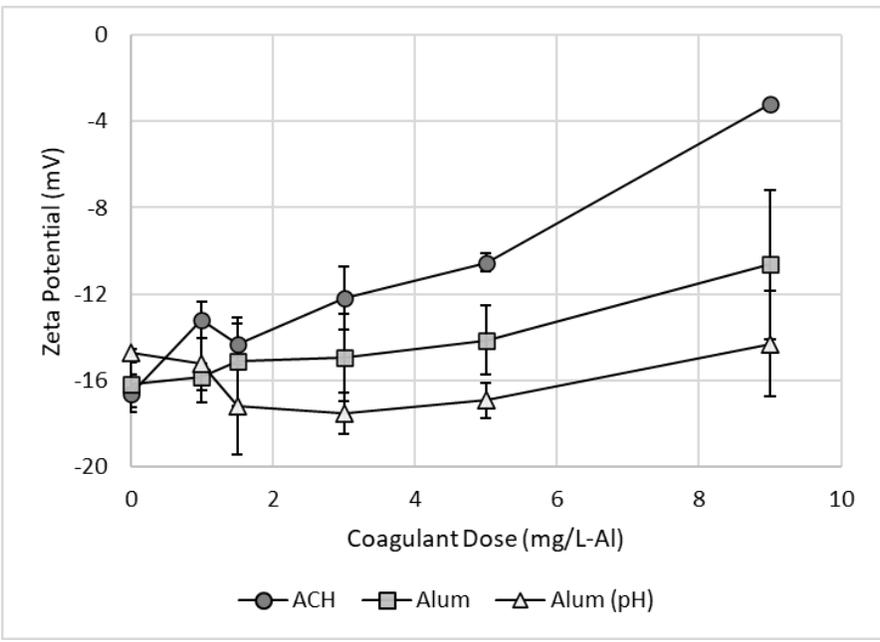
Figure 32: Jar Testing Results – Zeta Potential
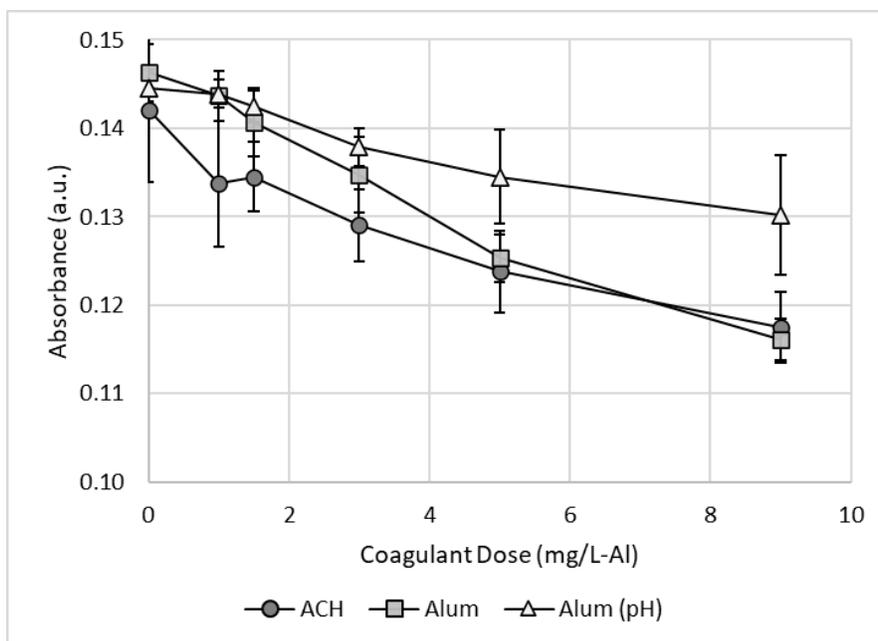
Figure 33: Jar Testing Results – UV254 Absorbance
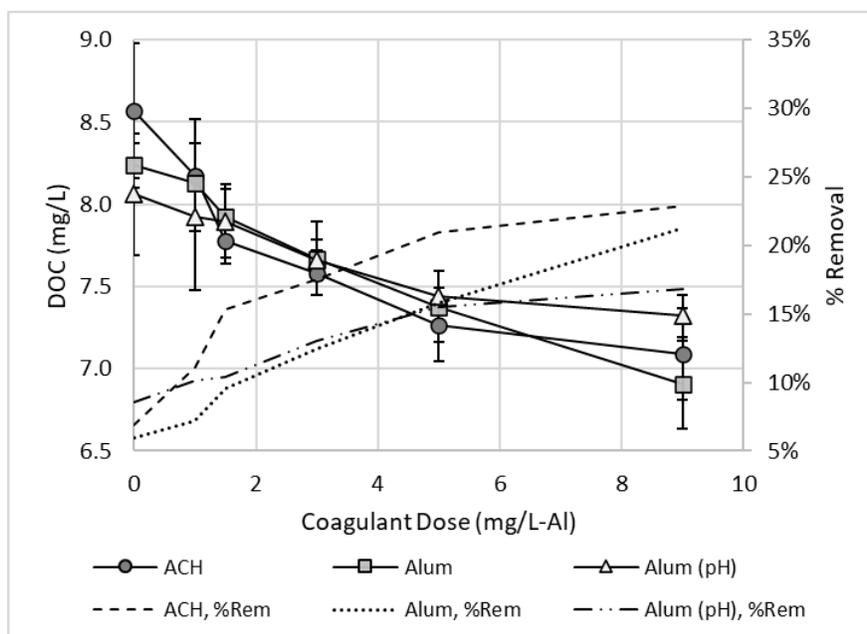
Figure 34: Jar Testing Results – DOC Removal

Appendix I – Direct Filtration Pilot Testing with High Alum Doses

A test was conducted on the pilot plant to estimate the minimum alum dose required for direct filtration to meet the phosphorus requirement during periods of elevated influent phosphorus. Phosphorus was spiked to the pilot influent (NTP secondary clarifier effluent), OP samples were collected, and alum dose was increased until sufficient phosphorus removal was observed. Phosphoric acid was dosed to achieve a minimum background concentration of 1.0 mg/L-P orthophosphate. The low dose of acid had no significant effect on pH. Orthophosphate samples were collected from the pilot influent (SCE), direct filter influent (FLOC), and direct filter effluent (DF). OP samples were filtered with a syringe and Environmental Express polyvinylidene fluoride membrane syringe filters (25 mm, 0.45 µm) and were analyzed with Hach TNT 843 for low range reactive phosphorus on the Hach DR3900.

Results are displayed in Figure 35. For an influent orthophosphate concentration greater than 1.0 mg/L-P, an alum dose greater than 2.7 mg/L-Al was needed for filter effluent to meet the 0.3 mg/L-P limit. Floc effluent samples were collected 20 minutes after pilot influent samples and biofilter effluent samples were collected 15 minutes after floc effluent samples. The delay allowed approximately one detention time to pass after chemical addition, while multiple detention times are necessary for complete mixing. At steady state, it is reasonable to assume greater OP removal would be observed at these doses.
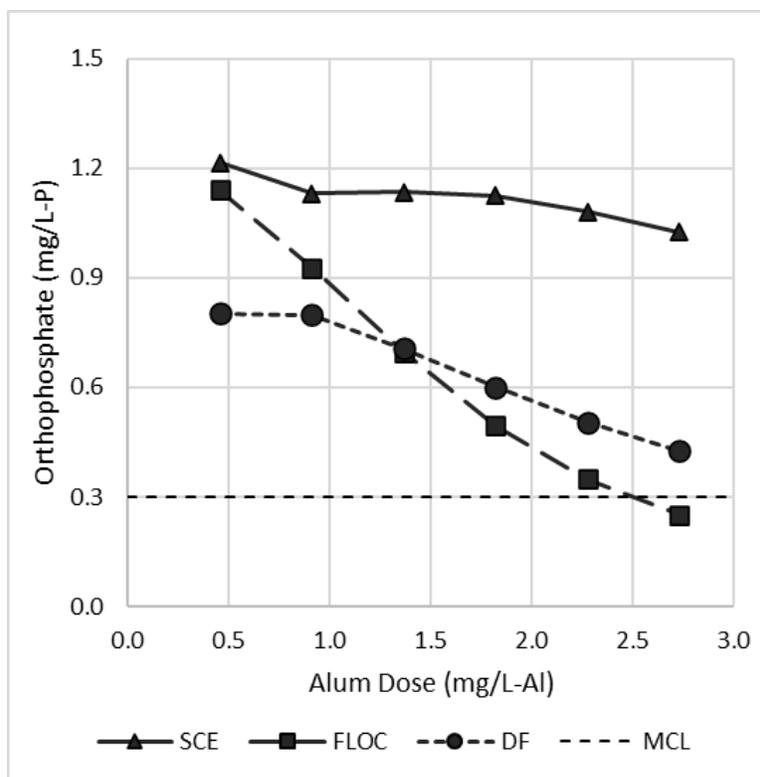
Figure 35: NTP Direct Filtration OP Removal with Increasing Alum Doses

In a separate test, the impact of higher alum doses on direct filter turbidity removal and head loss development was evaluated. Results are shown in Figure 36. Direct filtration achieved 24+ hour runs for alum doses up to 4 mg/L-Al. The 21-hour run was manually backwashed at turbidity <0.15 NTU and head loss less than 7 ft. The filter effluent turbidimeter failed during this testing, but grab samples were collected to ensure filter effluent turbidity was within specifications. It is possible, however, that filter effluent turbidity was greater than 0.15 NTU during some of these runs. A repeat of this experiment with the filter effluent turbidimeter in service should be performed to gain confidence in these conclusions.
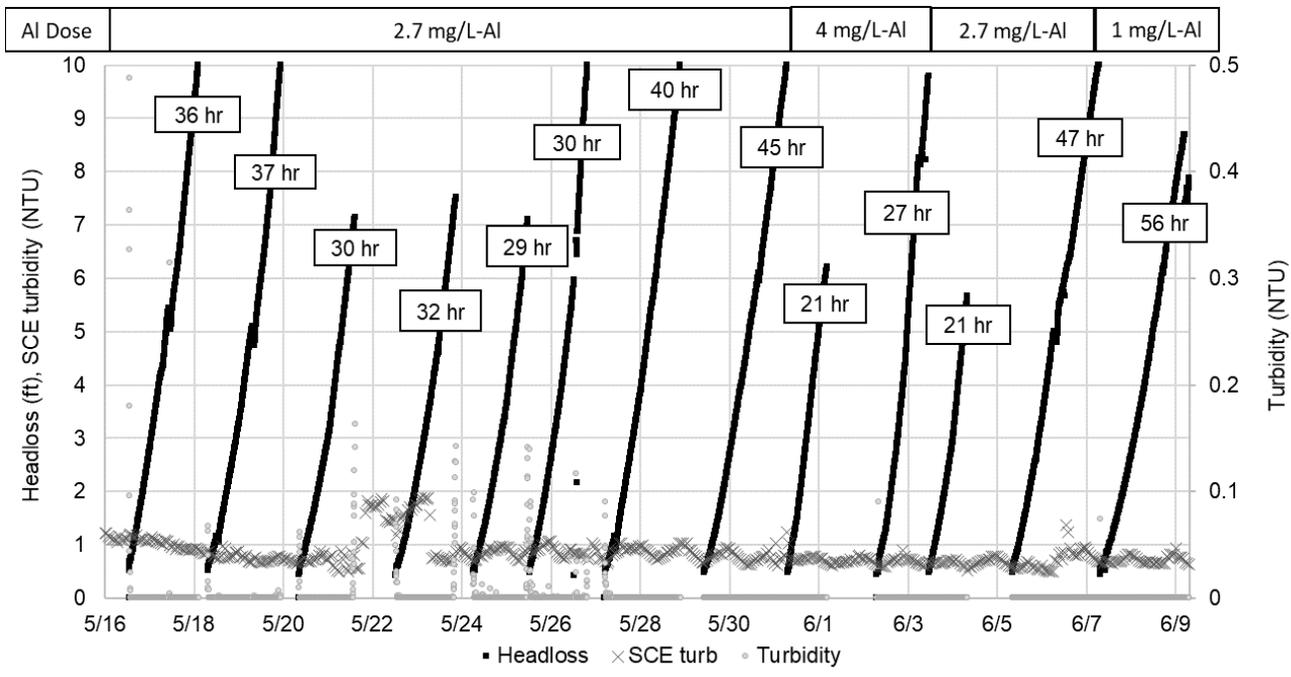
Figure 36: NTP Direct Filter Operations with High Alum Dose

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