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EVALUATION OF A SIDE-BY-SIDE, FULL-SCALE CONVERSION TO BIOLOGICAL FILTRATION

OCENA STOPNIOWEJ KONWERSJI DO FILTRACJI BIOLOGICZNEJ

W ostatnich kilku dekadach filtracja biologiczna, lub biofiltracja okazały się obiecującą praktyką w przemyśle uzdatniania wody. Chociaż instalacje biofiltracji stają się coraz powszechniejsze, praktyki operacyjne wymagane do konwersji i utrzymania biologicznie aktywnych filtrów nadal nie są dobrze poznane. Dlatego celem tego badania było dokonanie oceny jakości wody i trendów operacyjnych w konwersji biofiltracji na pełną skalę i określenie wpływu eliminacji chlorowania na wydajność filtrów.Cztery z dwunastu filtrów w stacji uzdatniania wody Quail Creek, zlokalizowanej w Hurricane w stanie Utah, wykorzystano do przetestowania zdolności filtrów do pracy w trybie biologicznym. Jeden filtr działał jako kontrola i działał podobnie do pozostałych ośmiu filtrów w stacji wody. Pozostałe trzy zostały przekształcone w biofiltry poprzez eliminacje resztkowego chloru tiosiarczanem. Spośród trzech biofiltrów, jeden był standardowym biofiltrem (tj. bez wstępnego chlorowania, niechlorowane płukanie), drugi miał chlorowane płukanie, a trzeci miał niechlorowane płukanie i uzupełnianie z azotem i fosforem.

Eksperymentalna konwersja trwała rok, czego rezultatem było zalecenie zamiany wszystkich filtrów na biologicznie aktywne, bez wzbogacania składnikami odżywczymi. Stwierdzono, że w wodzie wpływającej do stacji jest mało węgla organicznego (całkowity węgiel organiczny (TOC) <2 mg/L i kwasy karboksylowe (CBXA) <50 µg-C/L), co spowodowało niewielkie różnice między filtrami w trójfosforanie adenozyny (ATP) lub stężenia komórkowych substancji polimerowych. Biofiltry miały nieco wyższe stężenia heterotroficznej liczby płytek (HPC), ATP i EPS niż filtr kontrolny. Produkty uboczne dezynfekcji (DBP) były niższe w biofiltrach w stosunku do kontroli,pomimo braku mierzalnych różnic w usuwaniu węgla organicznego. Konwersja biologiczna spowodowała nieco bardziej zmienne (~ 0,013 NTU) wartości mętności w porównaniu z filtrami niebiologicznymi; jednak plukanie filtow było nadal wywolywane przez wzrost ciśnienia, a nie przez przełom zmętnienia. Jednostki objętości filtrów (UFRV) nie uległy zmianie pod wpływem konwersji, z wyjątkiem biofiltra wzbogaconego w składniki odżywcze.

Przedawkowanie składników odżywczych powodowało zbijanie się mediów w tym biofiltrze, podczas gdy filtr doświadczył wczesnego przełomu zmętnienia, powodując dłuższy czas płukania.Większość badań o biofiltracji pokazuje wyniki przed i po konwersji biofiltrów.

To badanie jest wyjątkowe, ponieważ porównanie biologicznie i nie biologicznie aktywnych filtrów przebiegalo w tym samym czasie; w ten sposób usuwając różnice w jakości wody surowej obserwowane z miesiąca na miesiąc. Ponieważ tylko trzy wybrane filtry zostały przekształcone w biofiltry, operatorzy mogliby powrócić do poprzedniej operacji gdyby wydajność uzdatniania wody nie była satysfakcjonująca. Po zakończeniu tej częściowej konwersji postanowiono zamienić wszystkie filtry na biofiltry, bez wzbogacania w składniki odżywcze. Pełna konwersja rozpoczęła się w styczniu 2018 r. i będzie monitorowana i oceniana przez jeden rok.

In the last few decades, biological filtration, or biofiltration, has proven to be a promising practice within the water industry. Though biofiltration treatment plants are becoming more prevalent, the operational practices required to convert to and maintain biologically active filters are still not well understood, especially in carbon-limited environments. Therefore, the purpose of this study was to evaluate the water quality and operational trends of a side-by-side full-scale biofiltration conversion and to determine the impact of pre-chlorination elimination on filter performance.

Four of twelve filters at the Quail Creek Water Treatment Plant, located in Hurricane, Utah, were used to test the plant's ability to operate in biological mode. One acted as a control and ran similar to the other eight filters in the treatment plant. The other three were converted to biofilters by quenching the influent chlorine residual with thiosulfate. Of the three biofilters, one was a standard biofilter (i.e. no pre-chlorination, non-chlorinated backwash), the second had chlorinated backwash, and the third had non-chlorinated backwash and nitrogen and phosphorus supplementation.

The experimental conversion lasted one year, resulting in a recommendation to convert all the filters in the plant to biologically active, without nutrient enhancement. The influent water was found to be low in organic carbon (total organic carbon (TOC) < 2 mg/L and carboxylic acids (CBXAs) < 50 µg-C/L), which resulted in small differences among filters in adenosine triphosphate (ATP) or extra cellular polymeric substances (EPS) concentrations (median differences ranging from 175–2,300 ng ATP/cm^3 and 0.00 to 0.08 mg glucose/g TS, respectively). Biofilters had slightly higher concentrations of heterotrophic plate count (HPC), ATP, and EPS than the control filter. Disinfection by-products (DBPs) were lower in the biofilters relative to the control (~11.3 and 22.9 µg/L median difference for haloacetic acids (HAA5s) and total trihalomethanes (TTHMs), respectively), despite finding no measureable differences in organic carbon removal. Biological conversion resulted in slightly more variable (~0.013 NTUs) effluent turbidity values compared to the non-biological filters; however, filter backwashes were still triggered by headloss rather than turbidity breakthrough. The unit filter run volumes (UFRVs) were unaffected by the conversion, except in the nutrient-enhanced biofilter. Overdosing nutrients caused the media in the enhanced biofilter to clump together, while the filter experienced early turbidity breakthrough, resulting in longer backwash times and rates.

The majority of biofiltration studies show results before and after biofilter conversion. This study is unique, for a comparison of biologically active filters and non-biologically active filters was compared in real time; thus, removing differences in raw water quality typically observed from month to month. Since only selected filters were converted to biofilters, operators could revert back in case the performance was not satisfactory. At the conclusion of this fractional conversion, it was decided to convert all the filters to in the plant to biofilters but without nutrient enhancement. The full conversion started in January 2018 and will be monitored and evaluated for one year.

1. Introduction and objectives

Drinking water utilities are required to consistently produce water that meets regulations that are becoming more and more stringent. Therefore, finding new operational practices to improve drinking water quality is always on the horizon. In the last few decades, biological filtration, or biofiltration, has proven to be a promising practice within the water industry. Biological filtration is an operational practice of managing and maintaining biological activity within the rapid rate (2-10 gpm/ft^2) on granular, aerobic drinking water filters [50]. Biofiltration is similar to conventional treatment, in that water is filtered at the rapid rate, but, unlike in conventional treatment, biological activity is not suppressed on filters so that the microbial removal of organic and inorganic constituents is enhanced [50]. The benefits of biofiltration over conventional treatment can include: a decrease in bacterial regrowth in the distribution system by the reduction of more easily oxidized organic matter, a reduction of disinfection by-product (DBP) formation by reducing the content of DBP precursors, a decrease in oxidant demand in the clearwell, and improved water aesthetics such as taste and odor [14];[51]. As the benefits of biofiltration are becoming better known, more utilities are converting their filters to biological mode [57]. However, the implications of converting a conventional filtration plant to a biofiltration plant are still not well understood.

1.1 Problem statement

The potential benefits of biological filtration have led many drinking water treatment plants to convert their conventional filtration system to a biologically active system [50]. This is most commonly accomplished by eliminating pre-chlorination in the treatment process. However, system upsets (i.e., turbidity breakthrough, manganese release, water quality deterioration, etc.) are sometimes experienced by plants that convert to biological mode [50]. In order to mitigate these problems, a better understanding of possible operational challenges would greatly benefit water utilities.

1.2 Objectives

In this study, three out of twelve filters at a conventional surface water filtration plant in Hurricane, Utah were converted to biofilters and their hydraulic performance and water quality were evaluated against the non-biological filters for one year. The objectives of this study were to:

1. Evaluate the impact of conversion to biofiltration on water quality and hydraulic performance across three selected types of biofilters (i.e., standard biofilter, biofilter with chlorinated backwash, and nutrient-supplemented biofilter) and compare to a simultaneously operated control filter at the Quail Creek Water Treatment Plant (QCWTP) in Hurricane, UT.
2. Determine the impact of pre-chlorination elimination on filter performance.
3. Provide operational and monitoring guidance to water treatment operators at QCWTP.

The QCWTP began evaluating biological filtration at full-scale in August 2016 after a one month transition period following equipment modification. Four of twelve filters were used to test the plant's ability to operate in biological mode. One filter was used as a control and the other three operated as the different types of biofilters described above. Under normal operation the plant applies chlorine at the intake of the source water from two open reservoirs and a river; therefore, the flow to all three biofilters required de-chlorination with thiosulfate upstream of the test filters to encourage biological growth. Of the three biofilters, one had no pre-chlorination with chlorinated backwash, a second was a standard biofilter: no pre-chlorination with de-chlorinated backwash; and the third was an engineered biofilter: no pre-chlorination, de-chlorinated backwash, with nitrogen and phosphorus supplementation. Each test filter was monitored for organic carbon concentration, biological activity, and a variety of typical water quality parameters. Different forms of carbon, nitrogen, and phosphorus were monitored across the filters to establish C:N:P ratios and evaluate nutrient abundance. Samples of the biofilter media were analyzed at the end of each filter run to determine biological activity via ATP activity assays. Hydraulic performance (effluent turbidity, run time, unit filter run volume (UFRV), backwash time) of the test filters was also evaluated against the conventional (non-biological) filters at the plant.

The majority of biofiltration studies show results before and after conversion of a filter from conventional to biofiltration operation [48];[50]. This study is unique, for biologically active filters and non-biologically active filters were compared in parallel and in real time; thus, removing differences in raw water quality typically observed from month to month. Since only selected filters were converted to biofilters, finished water could be blended with conventionally treated water before distribution, and operators could revert back to conventional filtration in case the performance was not satisfactory.

2. Background

2.1 Historical background

Rapid rate biological filtration, or biofiltration, did not begin to surface until after the recognition of disinfection by-products and the regrowth of microorganisms in distribution pipe. Biofiltration involves filtration through traditional granular media (sand, anthracite, or granular activated carbon (GAC)) and the elimination of chlorine residual on the filter bed [14]. Diminished chlorine residual allows heterotrophic bacteria to colonize on the surface of the media and create a biofilm that is capable of degrading various organic contaminants and micropollutants [14]. The implementation of biological filtration for the removal of organic material started in Europe in the 1970's but did not begin appearing in North America until the late 1980's. Biofiltration has not been well received in North America because of the practice to eliminate bacterial growth in drinking water treatment systems, but that mentality is slowly shifting [14];[57].

As biofiltration gained wider acceptance in the water treatment industry, new improvement strategies began to surface to increase efficiency without impacting operational practices or costs. Lauderdale et al. [27] were among the first to implement "engineered biofiltration", which changes biofiltration from a passive process to a purposeful operation designed to target multiple water quality objectives without compromising hydraulic performance. The study found that the addition of phosphorus decreased head loss by 15% and increased DOC removal by 75% at the John F. Kubala Water Treatment Plant in Arlington, Texas. Several biofiltration plants are now moving toward engineered filtration in order to optimize their filters and improve water quality.

2.2 Contaminants of concern

The list of known drinking water contaminants is long and daunting. Drinking water utilities are required to monitor and remove multiple pollutants simultaneously, while trying to optimize plant performance. In comparison to conventional filtration, biofiltration is known to produce equal or better quality water with minimal chemical requirements; biofilters remove and reduce contaminants by converting them to safer, less toxic, or easily separable compounds [50].

The following contaminants are of special concern for utilities implementing biological filtration:

- Natural organic matter (NOM): Numerous studies have shown significant NOM removal across biologically active filters [5];[34];[29];[51]. NOM is composed of carbon, hydrogen, oxygen, and nitrogen of varying elemental fractions but its composition and concentration varies widely between sources and seasons, making it difficult to characterize [15]. Because of its complexity, NOM is quantified through surrogate measurements. The most common NOM indicators in the water industry are total organic carbon (TOC), dissolved organic carbon (DOC), biodegradable dissolved organic carbon (BDOC), assimilable organic carbon (AOC), UV254 absorbance (UVA), specific UV absorbance (SUVA), and carboxylic acids (CBXAs) [14]. CBXAs (acetate, formate, and oxalate) can be used as a surrogate to estimate the amount of AOC, since AOC analysis is an expensive and time consuming [14].

- Disinfection by-products (DBPs): The discovery of trihalomethanes (THMs) and other halogenated DBPs in the early 1970's led to new regulations, which placed some water utilities in a position of having to either reduce chlorine to meet the DBP regulations or face non-compliance with the disinfection requirements and thus potentially expose the public to waterborne diseases [45]. This dilemma led to the development of strategies designed to reduce DBPs. Currently, total trihalomethanes (TTHMs) and five haloacetic acids (HAA5s) are the chlorinated byproducts that are regulated under the Disinfectants/Disinfection By-products (D/DBP) Rule [13]. The Maximum Contaminant Levels (MCLs) for TTHMs and HAA5s are 80 and 60 μg/L, respectively [13].

- Manganese (Mn) release: Manganese can accumulate on the filter media over time from the source water. The secondary MCL from the EPA is effluent manganese concentrations below 0.05 mg/L [20]. The removal of a pre-oxidant can result in the release of manganese into the distribution system causing color, turbidity, and taste problems, resulting in customer complaints. Media replacement or chemical washing are two common solutions to manganese release from filter media; however, these strategies can be costly. WCWCD replaced their filter media at the QCWTP shortly before the full-scale evaluation began. Therefore, it was determined that manganese release would be minor due to the short manganese oxide accumulation period.

2.3 Design and operations

There are many design criteria that need consideration while constructing and operating a full-scale water treatment plant; these criteria also apply when a plant is considering conversion to biofiltration. Proper operational practices are essential in order to obtain optimal filter performance. The following are the most important design and operational consideration:

- Filter media type: The typical media configuration for most conventional gravity filters and biofilters is a dual media system with sand as the base layer and anthracite or GAC as the top layer. The purpose of granular media is to remove particles through adsorption, biodegradation, interception, and screening mechanisms. For a biofilter to perform efficiently, the media should provide a suitable surface for biological activity, including high surface texture to foster growth and protect biomass from shear stresses. Multiple studies have been conducted to decipher differences in performance between GAC and anthracite media. Typically GAC is a better choice, for GAC has higher contaminant removal efficiencies at low temperatures [10];[29];[34];[51], can hold more biomass due to its larger and irregular surface and interior pore structure [29];[51];[54], provides better protection for microbes against shear stresses and chlorinated backwash [34];[54], can better remove biodegradable substances [26];[34], has a faster recovery and acclimation period [50];[51], and has the ability to adsorb dissolved contaminants [50]. During biofiltration conversion a plant might consider media replacement.

- Empty bed contact time (EBCT): Contact time, expressed as empty bed contact time (EBCT), is a key variable influencing organic matter removal through filtration. EBCT was introduced by Zhang and Huck [56] and can be adjusted by changing either the loading rate or filter bed depth. Previous studies have shown that 90% removal of BDOC could be achieved with an EBCT of 10-20 minutes [41] but this general assumption should be evaluated on case-by-case bases. Studies have consistently shown that increasing EBCT removes more biodegradable compounds [5];[23];[29].Moll et al. [36] suggest that in order to meet treatment objectives at cold temperatures a higher empty bed contact time (EBCT) may be necessary.

- Backwash procedure: Proper backwashing techniques help to maintain a healthy microbial community and obtain optimal hydraulic performance from a biofilter. In order to attain sufficient biological growth on a filter several backwashing conditions should be considered, including frequency, rate, duration, bed expansion, air scour, pulsing, and addition of chlorine [26]. A backwash is normally triggered by one of the three different criteria: head loss, run times, or turbidity breakthrough. The backwashing frequency is highly dependent on plant operation and can range from less than 12 hours to more than 48 hours [26]. Multiple studies concluded that air scour had no significant impact on biofilter performance [1];[10];[34]. Liu et al. [34] as well as others [56];[6];[54] found that chlorinated backwash adversely affected biomass concentration as well as BOM removal, especially at low temperatures.

- Acclimation period: An important consideration is the time required to reach steady-state biological activity, or the acclimation period. Wang et al. [54] demonstrated that this time could vary greatly with different parameters; steady state periods ranged from 2 to 99 days for non-chlorinated filters. In natural surface waters, an approximate 3-month period is needed to reach the maximum amount of biomass on GAC filters. Stoddart et al. [48] found that biomass reached steady state after about 7 months of full-scale operation on anthracite media. Factors such as chlorinated backwash, media type, and temperature can all affect acclimation time, thus making the acclimation period highly variable and site specific.

2.4 Microbial growth

Maintaining a healthy microbial community on a biofilter is a common challenge for drinking water utilities. A healthy and diverse community can increase biodegradation of contaminants and promote biologically stable water (i.e., decrease regrowth in the distribution system), whereas an uncontrolled population can lead to filter clogging and the release of toxins and other harmful substances. The following parameters require careful consideration in operation of biological filters:

- Biofilm formation: In a biofiltration system, dissolved organics are removed mostly through biodegradation on filter media. Microorganisms gradually attach and grow on filter media, creating a slime known as a biofilm. A biofilm is composed of microorganisms, extracellular polymeric substances (EPS), water, as well as other sorbed particles. A biofilm's thickness can range from tens of micrometers to more than 1 cm and is influenced by flow rate, bedding material, and substrate concentrations (i.e., nutrients and organic substances) [6].

- Extracellular polymeric substances (EPS): The EPS matrix is mostly produced by the microorganisms themselves and is composed of polysaccharides and proteins, accompanied by nucleic acids, lipids or humic substances. The typical composition of biofilms is less than 10% microorganisms by dry weight and more than 90% extracellular matrix [17]. Studies show that nutrient limitation leads to higher EPS secretion [26];[46]. Flemming [17] suggested that the EPS is the main contributor to biofouling and that nutrients are the most important fouling factors.

- Biological stability: Once water leaves the treatment plant, it still undergoes multiple physical, chemical, and biological processes. Biologically stable water is produced when all nutrients in the finished water that could promote bacterial growth have been sufficiently removed [30]. A known practice to mitigate water instability is the use of chlorine. The next most important factors, after disinfectant residual, were water age, corrosion rate, DOC, and AOC. LeChevallier et al. [30] suggested that if chlorine dose cannot be increased then it is recommended that the corrosion rate be reduced by phosphate addition or pH/alkalinity adjustment, or that DOC and AOC removals are increased through improved coagulation or filtration processes.

2.5 Nutrients

The utilization of nutrients is essential to the life and productivity of aquatic organisms. An over-abundance of nutrients in the natural environment can result in eutrophication and water quality problems, whereas their absence leads to diminished primary production. According to Liebig's Law of Minimum [7], growth and productivity are controlled by the rate of the slowest sub-process. Meaning that one or more of the essential nutrients can limit the growth of organisms. Studies have shown that nitrogen and phosphorus are the most common limiting nutrients in terrestrial, freshwater, and marine waters [9]. Sardans et al. [44] has indicated that other nutrients, such as iron and potassium, can play an important role in aquatic and terrestrial environments. However, Vahala et al. [52] investigated phosphorus and inorganic nutrient (i.e., nitrogen, calcium, potassium, magnesium, molybdenum, zinc, copper, cobalt, and sodium) addition and found that phosphorus was likely the limiting nutrient. Utilization of nutrients in biofilters should be considered as follows:

- Nitrogen (N): Total nitrogen (TN) is calculated by summing inorganic and organic nitrogen. The two most biologically important forms of dissolved inorganic nitrogen in water are ammonium (NH_4+) and nitrate (NO_3-) [26]. Nitrite (NO_2-) is a less common form of inorganic nitrogen (normally rapidly converted to NO_3-). Organic nitrogen comes from multiple sources, e.g., amino acids, proteins, or nucleic acids [7]. Seasonal variations play a crucial role in which forms of nitrogen are present in water. The forms of nitrogen can vary depending on if water conditions are oxic or anoxic.

- Phosphorus (P) is thought to be the main limiting nutrient in aquatic habitats [7];[9] because it occurs at low levels in the environment; and within natural waters only a small portion is carried in soluble (available) forms. The main dissolved form of phosphorus is orthophosphate (PO_4); it is the most bioavailable form of phosphate. Within drinking water treatment plants phosphorus can be reduced to below detection limits through coagulation and sedimentation processes [39], as both iron and aluminum phosphates are insoluble at neutral pH.

- C:N:P ratio: The balance of C, N, and P plays a vital role in the life of all organisms, but the relative ratio of these nutrients to achieve balanced growth is still a debated topic. The most well-known ratio is the Redfield ratio, which gives a C:N:P molar ratio of 106:16:1 for oceanic phytoplankton [43]. Sterner et al. [47] and Sardans et al. [44] demonstrated that C:N:P ratios differ between environments and species and the optimal balance is dependent on multiple different factors. The recommended molar ratio for drinking water bacteria is approximately 100:10:1 (C:N:P), which converts to a concentration ratio of 1 mg-C/L:0.117 mg-N/L:0.026 mg-P/L of bioavailable carbon, ammonia-nitrogen, and orthophosphate-phosphorus [26];[29];[31] . Few studies have demonstrated the impact of overdosing N and P, especially at full-scale drinking water treatment plants [47];[1].

- Impact of nutrients on contaminant removal: Since the work of Lauderdale et al., promoting the idea of "engineered" biofiltration [27], several studies have been conducted to confirm their findings. Multiple studies have concluded that nutrient addition is beneficial [18];[20];[32];[55], whereas others found no significant impact [3];[35];[39];[52]. The contradicting results between these studies emphasize that more information is needed to better understand the potential benefit of N and P augmentation.

- Impact of nutrients on EPS production: Li et al. [32] investigated the effect of phosphorus on biofilm formation in terms of growth, EPS production, and microbial community function. Lauderdale and Brown's [26] experiment was carried out by dosing biologically active filters with phosphorus (to fulfill the 100:10:1 C:N:P molar ratio) and monitoring organic removal, hydraulic performance, and biological activity. The results suggest that when the filter was no longer "nutrient limited", the microbes responsible for EPS production were reduced. All studies showed that creating an environment that is no longer phosphorus-limited could increase biological activity, decrease EPS, and change the overall microbial community and structure.

3. Material and methods

3.1 Hypothesis

The research approach was built upon the following hypothesis: the removal of pre-chlorination at a filtration plant can improve water quality without compromising filter performance (i.e., effluent turbidity, run time, UFRV, water quality, etc.) in selected types of biofilters (i.e., standard, chlorinated backwash, engineered). The hypothesis was tested by observing the performance of the biofilters against the non-biological filters throughout a full study year.

3.2 Quail Creek water treatment plant

The Quail Creek Water Treatment Plant located in Hurricane, Utah was constructed in 1986. The plant originally had four filters with a 10 million gallon per day (MGD) capacity. Over the last three decades it has received a series of upgrades and expansions to meet increased water demands. In 1997, four additional filters were constructed, expanding the plant to a 20 MGD capacity. In 2005, the final four filters were added, expanding the plant to a 40 MGD capacity. All of the media were replaced in 2015, allowing the plant to produce up to 60 MGD. QCWTP receives ~98% water from Quail Creek Reservoir, with the rest from Sand Hollow Reservoir and the Virgin River. The plant treatment train is shown in Figure 1. The QCWTP was originally built as a conventional treatment plant (i.e., coagulation, flocculation, sedimentation, filtration, and disinfection), with powdered activated carbon (PAC) for taste and odor (T&O) control, but within the last decade has added a dissolved air floatation (DAF) train to flocculate and separate liquids and solids more rapidly and reliably. The sedimentation and flocculation basins are now only put in operation in periods of high water demand during summer months.

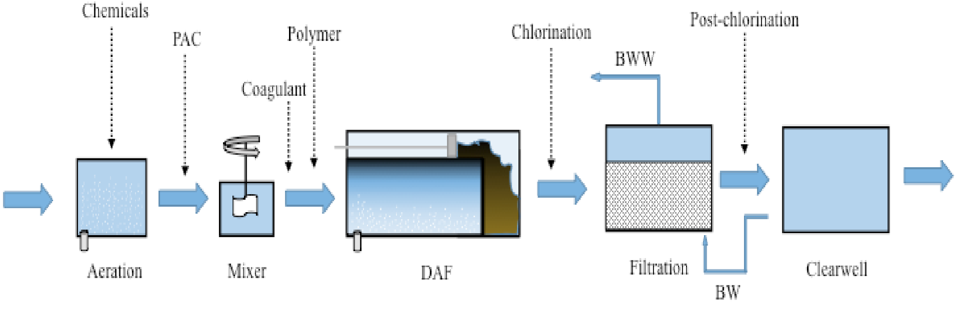

Fig. 1. Quail Creek Water Treatment Plant process schematic
Rys. 1. Schemat procesów uzdatniania wody w stacji Quail Creek

Influent water is characterized as low-organic carbon (TOC < 2 mg/L), low-turbidity (< 1 NTU), and high-pH (> 8 Standard Units (SUs)). During the study, alum doses ranged from 10 to 20 ppm (with higher average doses in the winter months) and chlorine from 1.5 to 2.25 ppm. Daily demand for the plant ranged from < 1 MGD in the winter to 38.3 MGD in the summer. The plant was in operation for < 1 hour per day in the winter and up to 19.7 hours per day in the summer. Filters are not run for 24 hours a day, but are shut off after the demand is met and storage reservoirs are filled. The filters have 27 inches of anthracite over 12 inches of sand, with an average loading rate of 5.7 gpm/ft^2, which equates to an EBCT of ~4.2 minutes. The filters are composed of two cells measuring 22 ft by 12 ft, for a total surface area of 528 ft^2. The backwashing mechanisms involves a low wash at 1,800 gpm for 48 seconds concurrent with air scour at 1,000 SCFM followed by a high wash at 6,000 gpm for 8 to 10 minutes.

WCWCD is planning to add an ozone system to the QCWTP within the next five years. The hope is that as the plant converts to biological filtration, the ozone application will improve water quality and biostability. The main driver for biofiltration conversion was the reduction of DBP precursors and increased savings through reduced chlorine use. The QCWTP, in conjunction with the Utah Division of Drinking Water and Utah State University, began evaluating biofiltration at plant scale in August 2016.

3.3 Experimental design

This research studied, at full scale, the potential for biofiltration under otherwise normal operating conditions at the QCWTP. The research plan consisted of analyzing the performance of three filters operated in biological mode in contrast to a control filter, operated traditionally. Performance was assessed using a combination of routine monitoring of plant operations (finished water turbidity, filter run times, backwash run times/volumes, head loss, etc.) and special periodic sampling for biological activity (ATP, HPC, EPS), organic carbon removal (TOC/DOC, UV_{245}, CBXAs) and nutrient availability (TP, TN, NH_4, PO_4). A filter core study was conducted where sampling and analysis of the filter media was performed to track the development and control of biological activity in the filters over time and filter depth.

The study involved the analysis of four different full-scale filters at the QCWTP; three of which were operated in biological mode. The first filter acted as a control and ran similarly to all the other filters in the treatment plant. The water in the other three biofilters was de-chlorinated with thiosulfate prior to filtration to quench any chlorine that would inhibit biological activity. Of the three biofilters, one was a standard biofilter (no pre-chlorination with de-chlorinated backwash), another had no pre-chlorination but had chlorinated backwash, and a third was an engineered biofilter (no pre-chlorination, de-chlorinated backwash, and nutrient supplementation) (Figure 2).

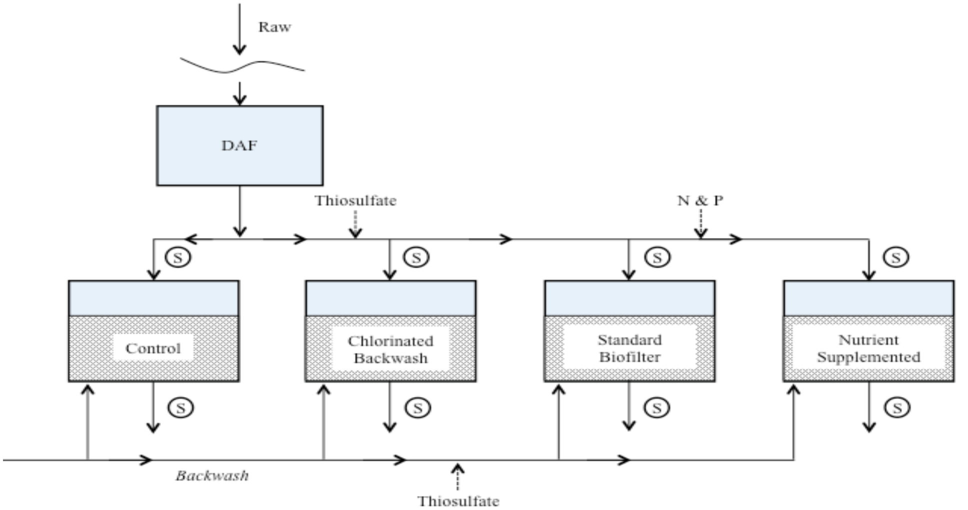

Fig. 2. Schematic of the control and biological filters. The S in a circle around signifies sample location.
Rys. 2. Schemat filtrów biologicznych i konrolnych. Litera S w kółku oznacza pobieranie próbek wody.

The engineered biofilter was supplemented with 75% phosphoric acid at a rate of 120 mL/hr and 1.2% ammonium chloride at a rate of 1,100 mL/hr. The solutions of phosphoric acid and ammonium chloride came in 55-gallon drums, where two pumps were set up to add low concentrations of P and N. The nutrients were added immediately before the water entered into the filter cell.

The biofilters were operated under similar conditions as the other filters at the Quail Creek plant; including loading rate (~5.7 gpm/ft^2) and media configuration (27 inches of anthracite over 12 inches of sand). However, as a precaution, backwashing was triggered at a head loss of 7.5 ft, instead of 9 ft.

Concentrations of carbon, nitrogen, and phosphorus were monitored to determine the availability of nutrients and C:N:P ratios. The C:N:P ratios were calculated from DOC removal, ammonia-nitrogen (NH_4-N), and orthophosphate (PO_4-P), as described by Lauderdale et al. [6]. Forms of carbon (TOC/DOC, UV_{254}, and CBXAs), nitrogen (total nitrogen (TN), nitrate plus nitrite, and ammonia), and phosphorus (TP and orthophosphate) were measured to determine the availability of nutrients to each filter. Samples of the biofilter media were analyzed to estimate the biological activity on each filter to determine if a certain treatment strategy (e.g., chlorinated backwash or nutrient augmentation) had an impact on microbial growth. Hydraulic performance (i.e., effluent turbidity, run time, UFRV, backwash time, etc.) of the test filters was evaluated against an additional filter in the plant. In the filter core study, three media cores were collected the summer of 2017 and analyzed to observe changes across depth and filter run time for ATP, CBXAs, TOC/DOC, nutrients, and other water quality parameters.

The experiment ran for one year (August 2016 – August 2017) so biofilter performance could be analyzed in all seasons. The intent was to extrapolate the results beyond this study to benefit other water utilities that are considering biological conversion.

3.4 Sample collection and water quality analysis

The parameters used to determine filter performance were separated into four categories: organic carbon concentration, biological activity, water quality, and operational data, as shown in Table 1.

Water samples were collected from the sampling locations indicated in Figure 2. Filter samples were collected from a tap installed at the influent and effluent of every filter.

Table 1. Summary of monitoring parameters and analyses performed at project laboratories.
Tabela 1. Podsumowanie parametrów i analiz wykonanych w laboratoriach dla projektu.

Washington County Water Conservancy District				
Variable	Units	Category	Method/Instrument	Range
TOC	mg/L	Organic Carbon	EPA Method 415.3[a]	0.0004 – 100 mg/L
DOC	mg/L	Organic Carbon	EPA Method 415.3[a]	0.0004 – 100 mg/L
UV_{254}	1/cm	Organic Carbon	Hach Method 10054[b]	0.005 – 0.900 cm^{-1}
Manganese	mg/L	Water Quality	Hach Method 8149[c]	0.006 – 0.700 mg/L
SUVA	L/mg-m	Organic Carbon	EPA Method 415.3[a]	-
Dissolved Oxygen	mg/L	Water Quality	Myron Ultrapen PT5	-
Chlorine residual	mg/L	Water Quality	Hach Pocket Colorimeter II	>0.01 mg/L
ATP	ng ATP/cm^3	Biological Activity	Standard Methods 10200I[d]	>5,000 $RLU_{Luminase}$
HPC	MPN	Biological Activity	IDEXX SimPlate Method[e]	1 – 738 wells
Utah Public Health Laboratory				
Variable	Units	Category	Method	Detection Limit
Carboxylic acids	mg/L	Organic Carbon	Acetate plus formate & oxalate	-
Acetate	mg/L	Organic Carbon	Modified EPA Method 300.1[f] – Carboxylates by IC	8 µg/L
Formate	mg/L	Organic Carbon	Modified EPA Method 300.1[f] – Carboxylates by IC	7 µg/L
Oxalate	mg/L	Organic Carbon	Modified EPA Method 300.1[f] – Carboxylates by IC	25 µg/L
Total Phosphorus	mg/L	Water Quality	EPA Method 365.1[g]	0.003 mg/L
Total Nitrogen	mg/L	Water Quality	Standard Methods 4500-N B[d]	0.2 mg/L
HAA5s	mg/L	Water Quality	Standard Methods 6251 B[d]	Varies between compounds
TTHMs	mg/L	Water Quality	EPA Method 524.2[h]	Varies between compounds
University of Texas				
Variable	Units	Category	Method	Range
EPS	mg glucose/g media	Biological Activity	Phenol-sulfuric acid assay[i]	0.04-2.64 mg glucose/g media[k]
Utah Water Research Laboratory				
Variable	Units	Category	Method	Detection Limit
Orthophosphate	mg/L	Water Quality	Standard Method 4500-P E[d]	0.005 mg/L
Nitrate/Nitrite	mg/L	Water Quality	EPA Method 353.2[j]	0.003 mg/L
Ammonia	mg/L	Water Quality	EPA Method 350.1[k]	0.004 mg/L

[a] [40]; [b] [21]; [c] [22]; [d] [2]; [e] [25]; [f] [14]; [g] [38]; [h] [37]; [i] [8] [j];[12]; [k] [11]

Turbidity, chlorine residual, temperature, DO, TDS, and pH, head loss, filter run length, filtration loading rates, unit filter run volumes (UFRV), empty bed contact times (EBCT), and backwashing and flow parameters were collected in-situ by the various online instruments within the plant and fed to the Supervisory Control And Data Acquisition (SCADA) system. During peak water demand periods (spring/summer) grab samples were conducted weekly, whereas, during low water demands (winter) samples were collected monthly. ATP, TOC/DOC, UV_{254}, SUVA, Mn, and HPC analyses were conducted at the QCWTP in the Washington County Water Conservancy District's laboratory, located at the QCWTP, on grab samples collected at the end of each filter's run length. The ATP samples were collected from the top 2 inches of the filter media. Carboxylic acids, total phosphorus (TP), total nitrogen (TN), and DBP samples (HAA5s and TTHMs), collected by WCWCD staff at the end of each filter's run length, were sent to the Utah Public Health Laboratory (UPHL) for analysis. The samples were preserved and shipped within a week to UPHL in a cooler on ice. A 7-day simulated distribution system was set up to analyze DBPs (HAA5s and TTHMs) from June to September 2017.

Phosphorus (orthophosphate) and nitrogen (nitrate/nitrite and ammonia) samples were sent in a cooler to the Utah Water Research Laboratory (UWRL). Two separate plastic bottles were collected per site: one unfiltered with no preservation for orthophosphate (60-mL) and one filtered with sulfuric acid preservation for nitrate/nitrite and ammonia (125-mL). Eighteen sample bottles were collected per sampling event, plus an additional two randomly selected samples, totaling 20 samples per week.

EPS samples, collected from the top 2 inches of the filter media bed, were sent to the University of Texas at Austin. At least 2 grams of media were collected and shipped within 3 days on ice to Austin.

3.5 Data analysis

The monitoring data were collected, checked, and entered into a Microsoft SQL Server database, hosted at the UWRL, using the Observations Data Model (ODM) 1.1 protocol of Tarboton et al. [49]. Statistical analysis of the data was carried out using the statistical software program R [42]. R was used to determine statistically significant differences between (1) the influent and effluent concentrations of each filter, (2) the different types of test filters (i.e., control filter, chlorinated backwash biofilter, standard biofilter, and nutrient enhanced biofilter), and (3) to conduct correlation and regression analyses to examine relationships among the measurements.

All data underwent quality-control procedures. Outliers were removed and summary statistics for censored data (<MDL) were estimated with the Robust Ordered Statistics (ROS) method in order to reduce bias [24]. A modified sign paired test [24] was used to determine differences between the influent and effluent concentrations for parameters with values less than the MDL. The test determines differences (positive, negative, or tied) between paired observations. The test incorporates values less than the MDL and lowers the magnitude of outliers, thus making it a more robust test than other, parametric methods. A Wilcoxon signed-rank test, a non-parametric test that can assess the average of paired differences over time, was conducted for all other parameters [4]. If there were sufficient degrees of freedom and the p-value was less than 0.05 for either test, it was assumed that there was a statistically significant difference between the influent and effluent samples.

The non-parametric Wilcoxon score test [24] was used to determine differences among the effluent concentration or removal efficiency of each filter for parameters with values less than the MDL. The Kruskal-Wallis test was conducted for all other parameters. The Kruskal-Wallis test is similar to the ANOVA test, in that it can determine differences in two or more groups, but values are ranked and the distributions are compared to determine if each group is the same. If a difference among groups was found (p-value <0.05), a Pairwise Mann-Whitney U-test was used to determine which groups were different, based on rank sums, from each other. If the variances of the different groups were similar and the p-value was less than 0.05 for either test, it was assumed that there was a statistically significant difference among the filters' removal or effluent concentrations.

4. Results and discussion

The main objective of the Quail Creek full-scale study was to determine if the removal of pre-chlorination had a positive or negative impact on hydraulic performance and water quality and to determine if the benefits of biofiltration commonly seen in waters with more organic matter will be realized in the typically more pristine waters in the mountain west. The following sections describe filter characteristics in regard to organic carbon, biological activity, water quality, operational parameters, and nutrient availability. Variables and their comparisons are shown as boxplots or bar charts and time series plots. The percentage above or to the right of each plot represents median percent removal from the influent concentration. Error bars represent the median absolute deviation (MAD), a robust measure of variability. Different letters (i.e., a, b, c, etc.) denote statistically significant differences among filters for effluent or removal concentrations, as determined by the Mann Whitney U-test or Wilcoxon score test. An asterisk denotes statistically significant differences between the influent and effluent, as determined by the Wilcoxon rank-sum test or modified sign test.

4.1 Full-scale filter study at Quail Creek

Typical water quality and operational parameters during each season, monitored throughout the study, are shown in Table 2, where the MAD represents the median absolute deviation.

Table 2. Operational parameters and influent water quality in each season at the QCWTP.
Tabela 2. Warunki operowania filtrów i jakość wody surowej w czterech porach roku w QCWTP.

Variable	Unit	Season	Median	MAD
EBCT	min	Fall	4.7	0.28
EBCT	min	Winter	4.7	0.47
EBCT	min	Spring	4.8	0.30
EBCT	min	Summer	4.2	0.48
Loading Rate	gpm/ft^2	Fall	5.19	0.37
Loading Rate	gpm/ft^2	Winter	5.18	0.57
Loading Rate	gpm/ft^2	Spring	5.10	0.33
Loading Rate	gpm/ft^2	Summer	5.81	0.67
pH	SU	Fall	8.20	0.15
pH	SU	Winter	8.40	0.15
pH	SU	Spring	8.10	0.30
pH	SU	Summer	8.00	0.15
Temperature	°C	Fall	19.1	4.3
Temperature	°C	Winter	8.1	1.2
Temperature	°C	Spring	11.9	3.3
Temperature	°C	Summer	24.9	3.6
Turbidity	NTU	Fall	0.49	0.13
Turbidity	NTU	Winter	0.90	0.15
Turbidity	NTU	Spring	0.89	0.16
Turbidity	NTU	Summer	0.50	0.15

MAD – Median absolute deviation, EBCT – Empty bed contact time

During the summer months water demands were higher; therefore, higher loading rates and lower EBCTs were observed (5.8 gpm/ft^2 and 4.2 minutes compared to ~5.2 gpm/ft^2 and 4.7 minutes). EBCTs were similar, but low, compared to other full-scale biofiltration plants, where EBCTs ranged from 2.5 to 18 minutes [27]. Higher median turbidity was observed at lower temperatures (~0.9 NTU) compared to higher temperatures (~0.5 NTU).

4.1.1 Biological activity

Biological activity (bioactivity) can be estimated through a variety of methods, which can indicate different types and functions of microorganisms. Velten et al. [53] yet this biological component remains poorly characterized. In the present study we followed biofilm formation and development in a granular activated carbon (GAChas shown that ATP on media can be used to assess if filters have become biologically active. EPS can be used to indicate biofouling [16]. HPC is a less reliable method [14]; however, it has frequently been used historically in drinking water to quantify culturable heterotrophic bacteria. ATP concentrations, measured over time are shown in Fig. 3.

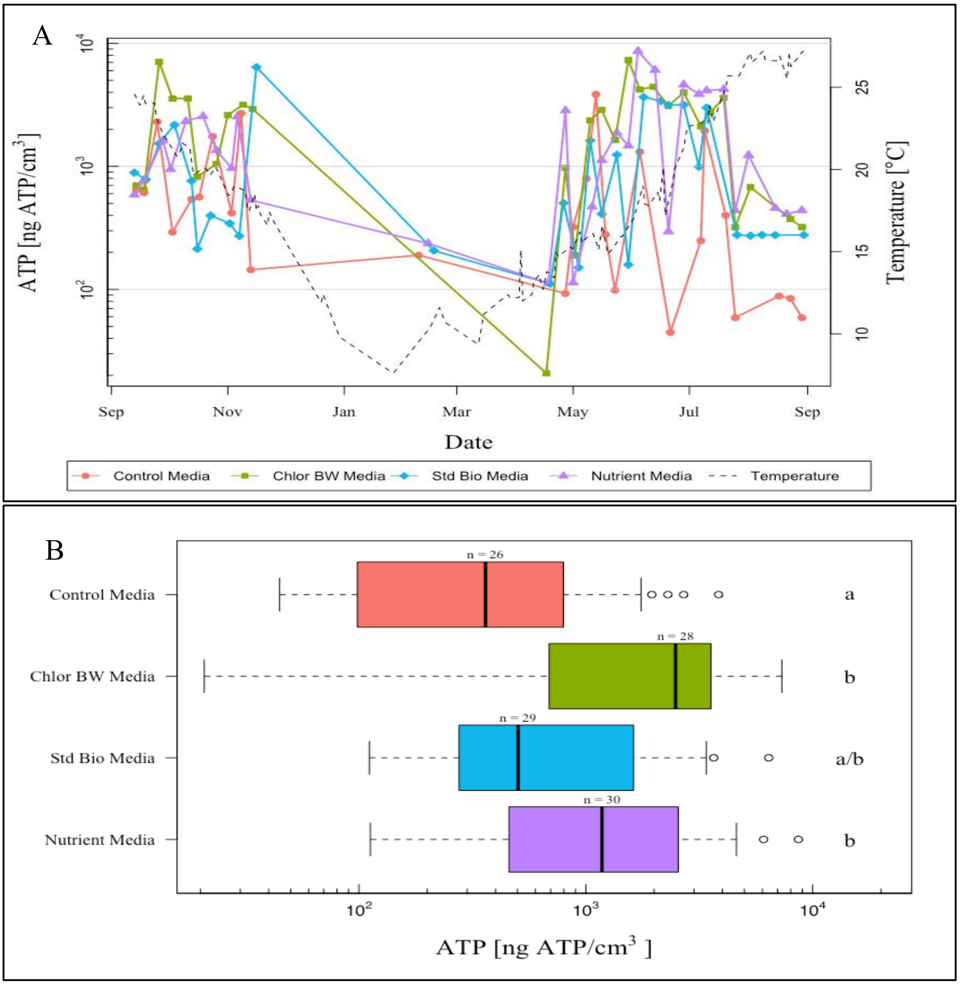

Fig. 3. (A) Time series and (B) box plot comparison of ATP concentrations on the media of biological and non-biological filters. The temperature represents the average influent water temperature. Letters (i.e., a, b, c, etc.) denote statistically significant differences among filters by the Mann-Whitney U-test.

Rys. 3. (A) Wykres ATP w czasie i (B) wykresy statystyczne ATP w filtrach biologicznych i kontrolnych.

ATP concentrations remained fairly stable (~100 - 4,000 ng ATP/cm^3 or ~1x10^5 - 3x10^6 pg ATP/g) throughout the majority of the study (Fig. 3-B). During the winter months and early spring, concentration dropped by over an order of magnitude (Fig. 3-A). Evans et al. [14] suggested that an order of magnitude of difference over time indicates significant change in a biological community. Over the entire study, temperature and ATP were poorly correlated ($r = 0.0067$, p-value = 0.93). Therefore, the decrease in ATP concentrations was likely due to factors other than water temperature. During the study, the chlorinated backwashed biofilter was being repaired from December to March. Water was left stagnant in the filter cells, and without an influx of nutrients, the growth could be limited and the microbial population would decrease due to organism decay. Therefore, the diminished biological growth in the winter months was more likely due to operational practices rather than colder water temperatures.

No measurable acclimation period was observed during the study (Fig. 3-A), which is contrary to other bioconversion studies. The lack of rapid increase of biological activity on the QCWTP's filters could again indicate the limited supply of available carbon for the microbes to grow and thrive.

Only 8 samples (7%) had concentrations below 100 ng ATP/cm^3 media: 7 out of the 8 samples were from the control filter. This suggests that all of the filters were at least slightly biologically active. There were only small differences between the control and biofilters (median value differences ranged from 175-2,300 ng ATP/cm^3). The ATP in control was statistically lower than in the chlorinated backwash and nutrient enhanced biofilters (Fig. 3-B). The small differences in biological growth could be from the following factors: low carbon supply, dual media type, possible trace of chlorine residual, and low EBCT.

EPS is an important indicator of bioactivity, especially important to monitor if biofouling was a concern. EPS concentration (free- and bound proteins and polysaccharides) is shown in Figure 4.

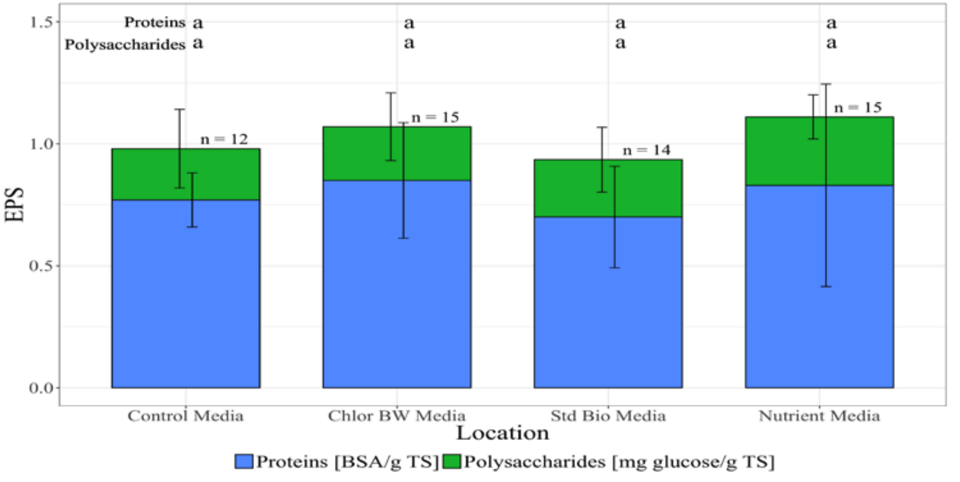

Fig. 4. Bar chart comparison of biological and non-biological filters for EPS (proteins and polysaccharides). Error bars represent one MAD. Letters (i.e., a, b) denote statistically significant differences among filters for proteins and polysaccharides by the Mann--Whitney U-test

Rys. 4. Wykres statystyczny EPS (białka i polisacharydy) w filtrach biologicznych i kontrolnych.

EPS concentrations were statistically the same throughout the sampling period (Figure 4.). Only small differences existed among the EPS concentrations (median differences of 0.00 to 0.08 for proteins + polysaccharides). The results were similar to the filter media ATP concentrations, in that slightly higher median concentrations of EPS were observed on the chlorinated backwash and nutrient enhanced biofilters, but, in this case, the differences were not statistically significant. Studies have shown that nutrient enhancement could decrease EPS production [26];[46], so lower EPS concentrations were expected on the nutrient enhanced biofilter. Since the influent water matrix was carbon-limited, the lack of available carbon in the system may not have supplied the necessary resources for EPS production. To date, no studies were published that investigated EPS concentrations on drinking water biofilters in a carbon-limited environment. More information is required in this subject.

The removal of pre-chlorination did not have a large impact on microbial growth at the QCWTP. A limited influent carbon supply is likely the main contributor to the low growth on the biofilters.

4.1.2. Organic carbon

Figure 5 shows the total and dissolved organic carbon removal in the control filter compared to the different biofilters.

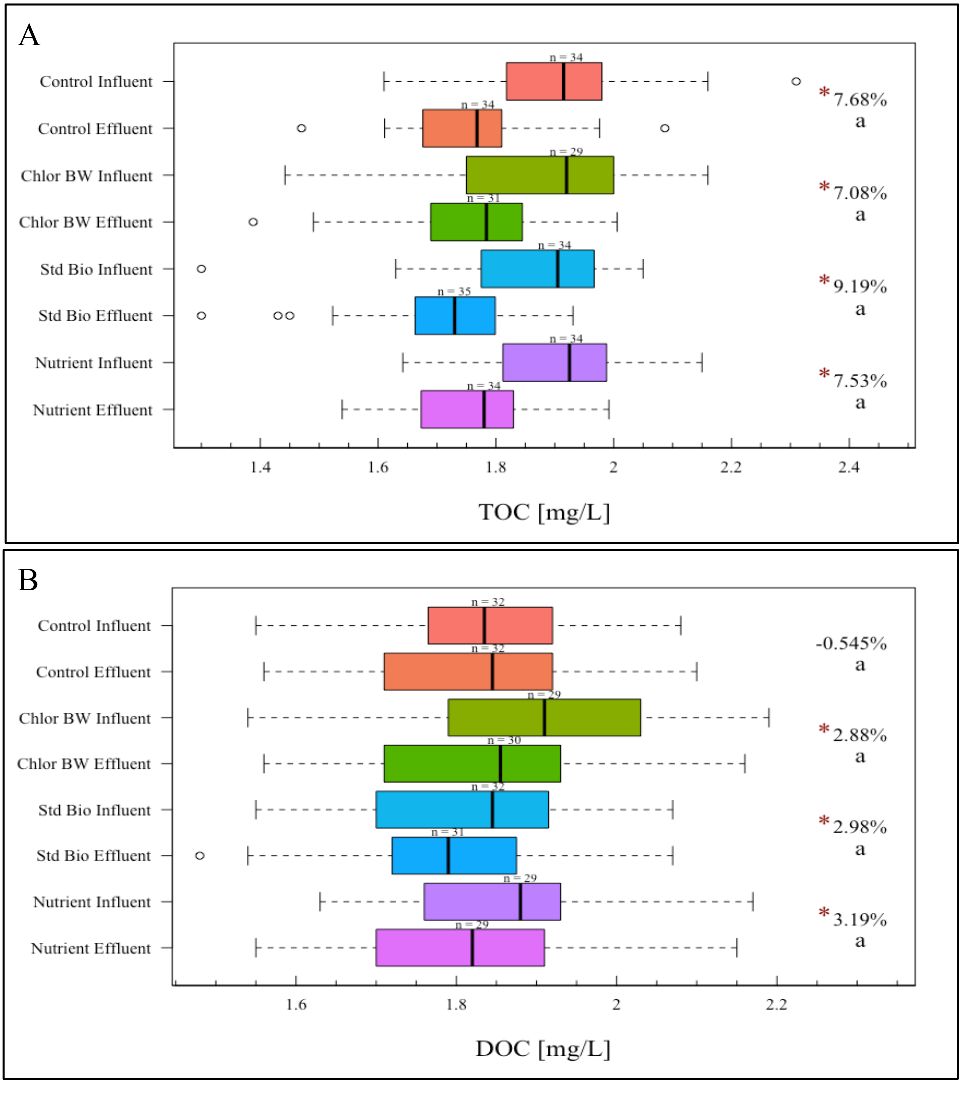

Fig. 5. Box plot comparison of biological and non-biological filters for (A) TOC and (B) DOC.Red asterisks indicate the differences between the filter influent and filter effluent as statistically significant by the Wilcoxon signed-rank test. Percentages represent median percent removal from the filter influent to the filter effluent. Letters (i.e., a, b) denote statistically significant differences among the filter's removal efficiencies by the Mann-Whitney U-test.

Rys. 5. Wykresy statystyczne stężenia TOC (A) i DOC (B) w filtrach biologicznych i kontrolnych

Organic carbon removal is the most common driver for biological conversion [50]. An abundance of NOM can impact filter performance and increase DBPs, while excess available carbon in the distribution system can increase biological regrowth in pipes. Therefore, the investigation of influent carbon concentrations and their removal is important.

The filter influent TOC concentrations were low, where 82% of all samples had concentrations less than 2 mg/L (Figure 5-A). Similar influent DOC concentrations were observed, indicating that the majority of the organic carbon was in the dissolved form, which is common [10]. All filters were removing a statistically significant amount of TOC; however, there were no statistically significant differences among the filters' removal efficiency. All were removing approximately 7 to 9% of TOC (0.14 – 0.17 mg/L). For DOC, only the biofilters had statistically significant removals (Figure 5-B), however, the removals across the biofilters were small (~3% or 0.06 mg/L) and there was no statistically significant difference among the filters' removal efficiency, similarly to the TOC findings. The DOC removal was much lower than other biofiltration studies, where DOC removals ranged from 10 to 30% with average influent DOC ranging from 1.1 to 3.2 [19];[33];[53]. Low influent carbon likely limited the growth of microbes, which would reduce microbial biodegradation. To investigate this further, a removal comparison of carboxylic acids (acetate, formate, and oxalate), a more available form of carbon, is shown in Figure 6.

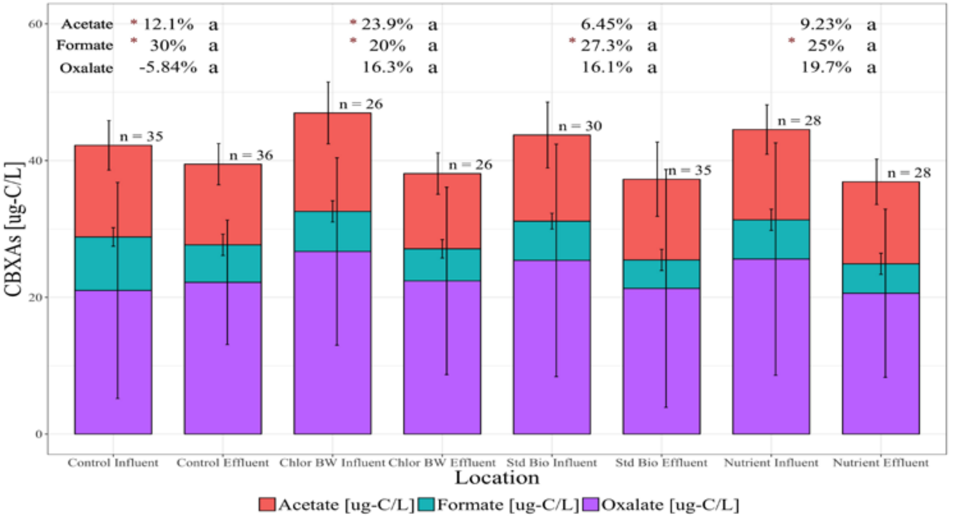

Fig. 6. Bar chart comparison of biological and non-biological filters for CBXAs as C (acetate-C, formate-C, and oxalate-C). Red asterisks indicate if the differences between the filter influent and filter effluent were statistically significant by the Wilcoxon signed--rank test. Percentages represent median percent removal from the filter influent to the filter effluent. Letters (i.e., a, b, c, etc.) denote statistically significant differences among the filter's removal efficiencies by the Mann-Whitney U-test for acetate and the Wilcoxon score test for formate and oxalate.

Rys. 6. Stężenie kwasów karboksylowych (octan, mrówczan, i szczawian) w filtrach biologicznych i kontrolnych

The high variability in the oxalate data, denoted by the large error bars, was likely due to sulfate interference in the water samples; thus the interpretation of the oxalate data is limited. The oxalate data were not removed from the graphs, so overall trends and comparisons to other studies could be made. Acetate and formate had a strong correlation ($r = 0.48$, $p = 3.84 \times 10^{-10}$), whereas oxalate correlated weakly with acetate and formate ($r = -0.014$, $p = 0.86$ and $r = -0.078$, $p = 0.30$, respectively). Therefore, the oxalate results were excluded from the remainder of the discussion.

All filters removed statistically significant amounts of formate, with removals ranging from 20 to 30% (1.2 to 2.3 µg-C/L). Only the control and chlorinated backwash filters had statistically significant removals of acetate, ranging from 6.5 to 24% (0.8 to 3.5 µg-C/L). The differences in removals among filters were small (< 3 µg-C/L). The carboxylic data indicate a carbon limited system (less than 50 µg-C/L available for microorganisms in the form of acetate, formate, or oxalate). The average net reduction of carboxylic acids across the biofilters in Evans et al.›s [14] study was 43 µg-C/L. The reason for the low influent carboxylic acids concentrations and consequently the low removals was likely due to the lack of ozone before filtration. Without ozone to break up the already limited source of organic carbon, the microorganisms present in the water had only a small supply of available carbon. It's probable that the biofilters degraded all the assimilable organic carbon available, similar to the Azzeh et al. [3] study. This was likely the reason why larger differences of ATP or EPS were not observed in the biofilters (Fig. 3 and Figure 4). It is expected that the addition of ozone before filtration would increase carboxylic acids and in turn increase biological growth.

Similar trends were observed for TOC, DOC, and UVA ($r = 0.22$- 0.40 with p-values < 0.003), with concentrations gradually increasing over the study period. Increased natural organic matter removal is a common driver for biological conversion. The influent water at the QCWTP has been characterized as being low in carbon (TOC < 2 mg/L), especially available carbon (CBXAs < 50 µg-C/L). Slight differences of carbon removal were detected for DOC and CBXAs, but overall, similar removals were observed across all types of filters. The removal of pre-chlorination did not impact organic carbon removal at the QCWTP.

4.1.3 Water quality

Improving water quality by reducing the formation of DBPs is another common driver for biological conversion. The adaptation to minimize or remove chlorine at the head of drinking water facilities is becoming more prevalent to control chlorinated DBPs. However, the release of manganese (Mn) from filter media after the removal of chlorine can lead to unintended consequences. Therefore, Mn and DBP concentrations at the filter effluent were both evaluated to determine the impact of removing pre-chlorination on water quality. TTHM and HAA5 concentrations are shown in Figure 7.

A decrease in TTHM and HAA5 concentration was observed in the biofilters compared to the control filter (Figure 7-B) (~60% (11.3 µg/L) and ~30% (22.9 µg/L) median difference for HAA5s and TTHMs, respectively). For TTHMs, this trend was observed throughout the entire sampling period, where all biofilters had statistically lower TTHMs than the control filter. The sample size was small ($n < 10$), but the rather consistent results do suggest that the trend would continue.

No differences in HAA5 concentrations were observed at the beginning of the sampling period, but after September, HAA5 concentrations in the biofilters were consistently lower than the control filter. McKie et al. [35] and Stoddart and Gagnon›s [48] studies showed similar results - that biofiltration reduced THMs and HAAs in the finished water without necessarily improving TOC or DOC removal.

HAA5s were well below the MCL of 60 μg/L, but TTHMs were near or above the MCL of 80 μg/L in some cases. The samples came from a 7-day simulated distribution system (SDS), and they represent the worst-case scenario. However, these findings are important for the QCWTP to consider, especially if DBP standards would be lowered in the future.

Manganese (Mn) concentration in the effluent increased slightly during the first few months of the study after chlorine began to be quenched; however, after a few months the effluent Mn stabilized. There were no statistically significant differences among any of the filters after this period. This temporary release of Mn was common at other full-scale biofiltration plants [28]. During the release period, concentrations remained low, where no effluent Mn concentrations ever exceeded the EPA secondary MCL guideline of 0.05 mg/L. The high pH conditions and fresh filter media could explain the limited release of Mn at the QCWTP.

The removal of pre-chlorination at the QCWTP reduced DBPs in the finished water without initiating a large manganese release into the distribution system. Since the THMs from the SDS were near the MCL, the full implementation of biofiltration could be beneficial for the plant.

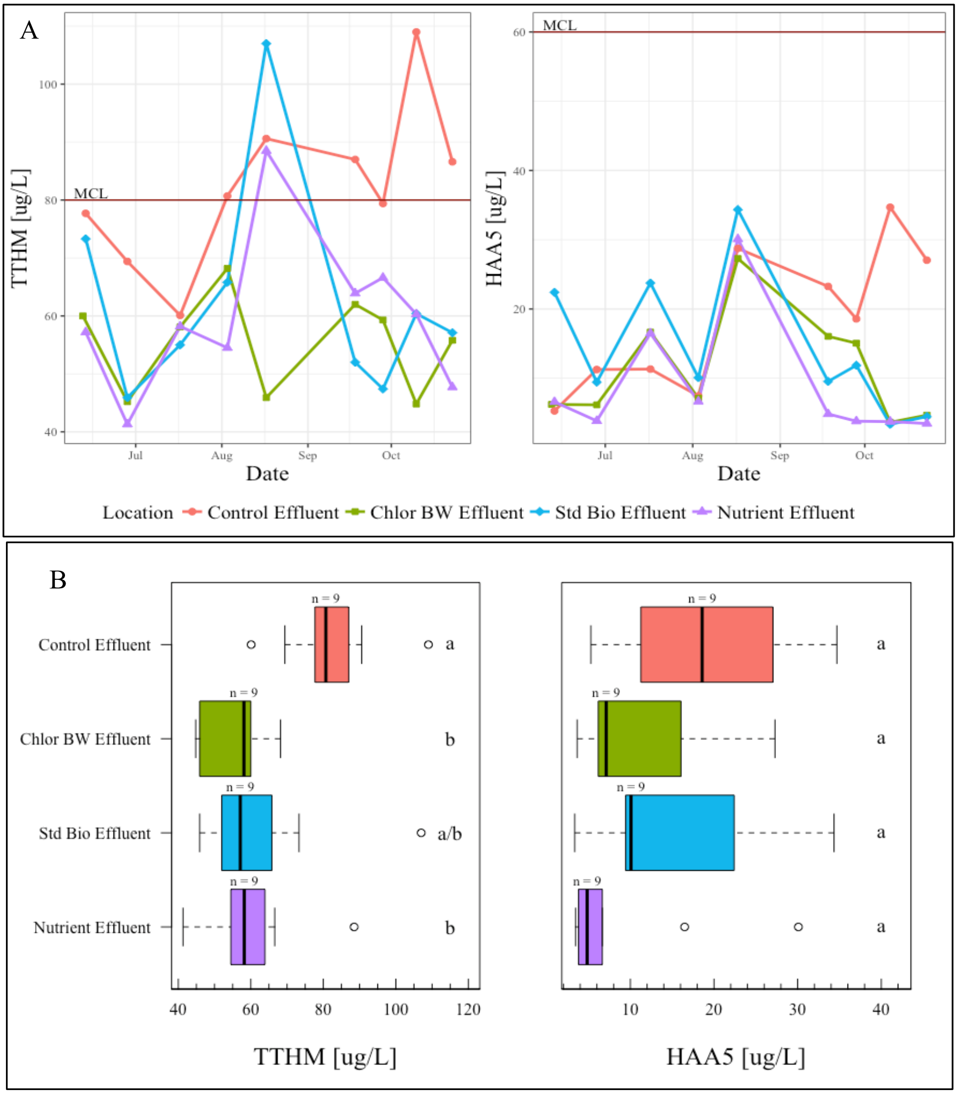

Fig. 7. (A) Time series plot and (B) box plot comparison of biological and non-biological filters for TTHMs and HAA5s. Letters (i.e., a, b, etc.) denote statistically significant differences by the Mann-Whitney U-test.

Rys. 7. (A) Wykres stężenia THM i HAA w czasie i (B) wykresy statystyczne THM i HAA w filtrach biologicznych i kontrolnych.

4.1.4 Operational parameters

Operational parameters (head loss, effluent turbidity, backwashing procedures, filter run length) were closely monitored to determine if the removal of pre-chlorination had detrimental impacts on hydraulic performance. It was paramount that water quality and filter performance were not compromised, since the finished water from the biofilters was distributed to customers. The project was termed a "Do No Harm" study, where if anything went awry, the filters would have been returned to the routine procedure. Few studies have provided extensive operational data of a full-scale side-by-side bioconversion experiment similar to this study. An evaluation of the unit filter run volumes (UFRVs) among the different types of filters is shown in Figure 8.

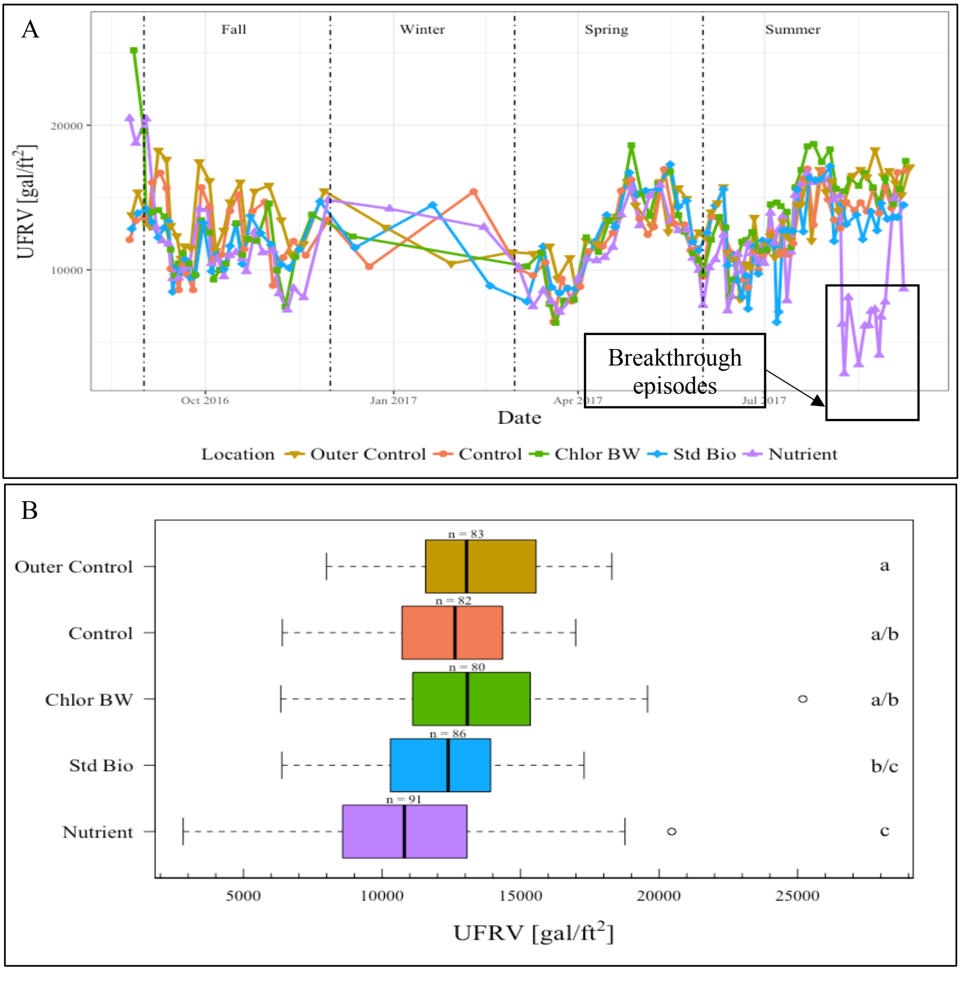

Fig. 8. (A) Time series and (B) box plot comparison of biological and non-biological filters for unit filter run volumes (UFRVs). Letters (a, b,..) denote statistically significant differences by the Mann-Whitney U-test.

Rys. 8. (A) Wydatnosci filtrów w czasie i (B) wykresy statystyczne w filtrach biologicznych i kontrolnych.

The "Outer Control" filter in Figure 8 represents an outlying filter that was not originally part of the study. The filter was run exactly the same as the other filters at the plant not involved with the study.

UFRVs were similar among all filters until near the end of the study, where the nutrient enhanced biofilter began to experience early turbidity breakthrough (Figure 8-A). Until that time, there appeared to be no difference in filter performance (as indicated by UFRV) among any of the filters. There was a slight difference between the standard biofilter and the filter not involved in the study (Figure 8-B); this was likely because the outer control was backwashed at a head loss of 9 ft instead of 7.5 ft like the test filters.

Near the end of August, the nutrient-enhanced biofilter began to experience frequent turbidity breakthrough within 10 to 20 hours of operation. After a few weeks of early breakthrough, the nutrients being supplied to the filter were shut off and UFRVs returned to normal. However, during this period of higher turbidity, head loss was unaffected and remained low. Because the nutrient-enhanced filter was overdosed, it is possible that the excess nutrients promoted the growth of autotrophic organisms that were loosely attached and sloughed off the filter, triggering turbidity episodes. However, no large differences in ATP or EPS were observed between the nutrient-enhanced biofilter and the other filters, indicating that this could only be a partial contributor. Also, the excess phosphate in the water could act as a surfactant, detaching loose biofilms. It's also possible that increased loading rates during this period removed loose biofilms from the media.

The plant operators consistently reported that the nutrient-enhanced biofilter experienced a more difficult time during the backwashing process, even before the turbidity breakthrough episodes occurred. They indicated that the nutrient-enhanced biofilter media tended to clump together, requiring longer backwashing times and, in some cases, a second backwash. This was likely as a consequence of the buildup of excess nutrients on the filter. The chlorinated backwashed biofilter was also statistically higher than most other filters for backwash water levels and time. The chlorinated backwash consistently had slightly higher ATP and EPS levels (Fig. 3-A and Figure 4), so this discovery was not entirely unexpected. However, since the operators manually choose the wash time and rate, it is possible that they unintentionally had a bias toward some biofilters and chose to backwash them longer as a precaution.

The use of chlorine in the backwash water had no significant impact on ATP concentrations or TOC/DOC removal (Fig. 3 and 5) at the QCWTP. These findings were contrary to most studies which found that chlorinated backwash adversely impacted biomass buildup and NOM removal, especially at low temperatures with anthracite media [1];[34];[6];[54]. However, Upadhyaya et al. [50] proposed that the presence of low levels of chlorine (1.5 to 2 mg/L) may not have an impact on mature biofilters; this might suggest that the microorganisms present on the media were well established. It is also possible that the limited nutrient environment encouraged more robust microorganisms, which were less likely to be impacted by chlorinated backwashing.

A comparison of the effluent turbidity of the different types of filters is shown in Figure 9-A and 9-B. These values represent effluent turbidity values taken immediately before backwash and do not represent average turbidity over a filter run. Average turbidity concentrations were much lower and consistently below the 0.1 NTU standard.

All filters effectively removed turbidity; however, differences were observed between the final effluent turbidity of the biological and non-biological filters (Figure 9). Both the standard biofilter and nutrient-enhanced biofilter had higher effluent turbidity levels ($p < 0.05$) than the other filters (average effluent turbidity ~0.057 NTU compared to 0.038 NTU). The chlorinated backwash biofilter had very similar effluent concentrations to the non-biological filters.

All the biofilters had slightly more variable effluent turbidity (Figure 9-B) than the non -biological filters ($\sigma = 0.023$ NTU compared to $\sigma = 0.010$ NTU). Despite the fluctuations in turbidity, head loss accumulation remained similar in all filters and was not impacted by biological conversion.

Turbidity breakthrough was more common in the biofilters than the non-biological filters in the spring and summer months. Backwashing was triggered more often by head loss than turbidity for the non-biological filters. Despite this finding, run times and UFRVs were unaffected, except in the nutrient-enhanced biofilter (Figure 8-B). There was a transition of the backwash trigger mechanism from head loss to turbidity for the biofilters. Stoddart and Gagnon [48] had a similar experience while converting to biofiltration, where they had higher effluent turbidity with no real differences in UFRVs or filter run times. Initial turbidity breakthrough is a common problem while converting to a biofiltration plant, but turbidity values typically return to normal effluent values [50]. However, for the QCWTP, slightly increased turbidity breakthrough was not experienced directly after conversion, but was more dependent on the season (higher in winter/spring and early summer).

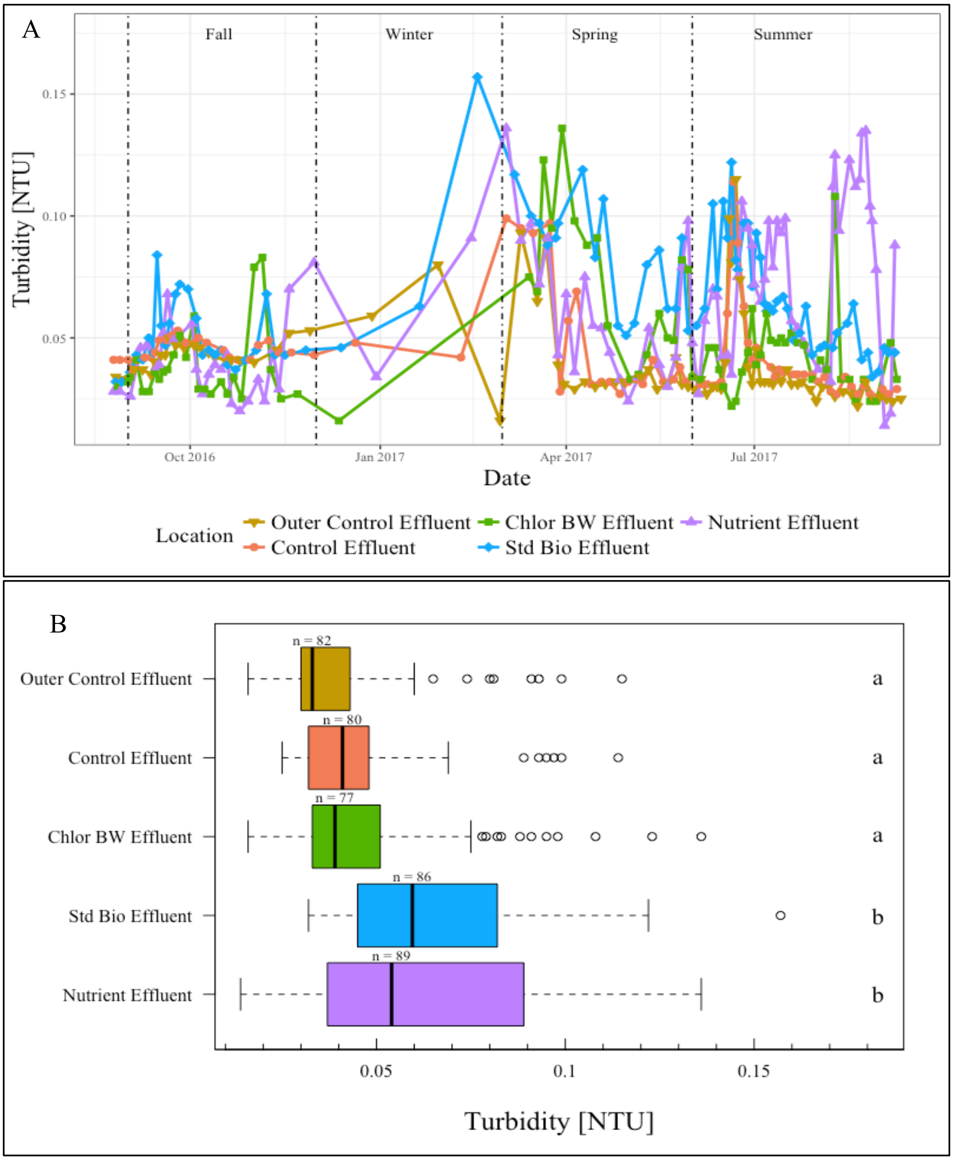

Fig. 9. (A) Time series and (B) Box plot comparison of effluent turbidity for biological and non-biological filters. Letters (i.e., a, b, c, etc.) denote statistically significant differences by the Mann-Whitney U-test.

Rys. 9. Wykresy mętnosci wody w filtrach biologicznych i kontrolnych

The biofilters received the same polymer and alum dosages as all the other filters in the plant. The polymer and alum doses were not optimized for the biofilters, so it is likely they the biofilters weren't performing as efficiently as possible. More information regarding polymer and coagulant dose differences between biological and non-biological filters is necessary to optimize biofiltration at QCWTP.

The removal of pre-chlorination at the QCWTP had a larger impact on operational parameters than the other variables already investigated (i.e., organic carbon removal, water quality, etc.). Longer backwash times and higher backwash rates were also required for the nutrient-enhanced and chlorinated backwash biofilters. Excess nutrient buildup likely caused the media to clump together, which resulted in a change of backwashing procedure for that filter. Backwashing for the biofilters was triggered more often by turbidity than head loss compared to the non-biological filters. However, head loss accumulation, UFRVs, and run times were unaffected. Overall, overdosing nutrients had a detrimental impact on filter performance, but the removal of pre-chlorination only caused small changes in effluent turbidity.

4.1.5 Nutrients

Multiple studies have shown that nutrient supplementation could be beneficial in optimizing the performance of biofilters [18];[20];[27], but others have found no significant benefit [3];[35]. The contradictory results from the literature suggest that nutrient supplementation in biofiltration is still not well understood or justified, especially at full-scale. Different forms of carbon (DOC and CBXAs), nitrogen (TN, NH_4, and NOx), and phosphorus (TP and orthophosphate) were tracked at the influent and effluent of each filter to determine the impact of nutrients at the QCWTP. A comparison of the median nutrient concentrations being supplied to the enhanced and non-enhanced biofilters is shown in Figure 10.

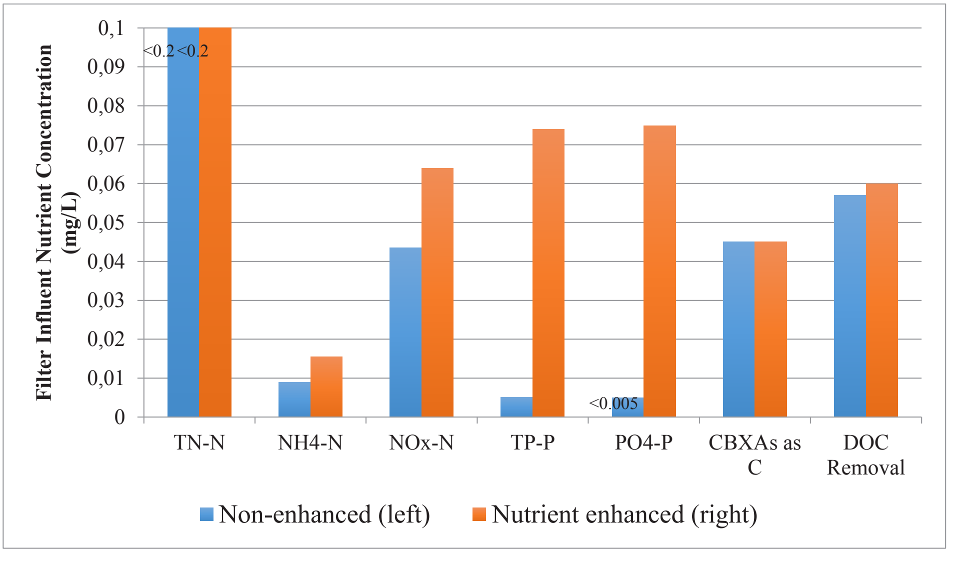

Fig. 10. Median filter influent nutrient concentrations for the nutrient enhanced and non-enhanced biofilters

Rys. 10. Stężenie składników odżywczych w biofiltrach wzbogaconych i nie wzbogaconych w składniki odżywcze

The nutrient-enhanced biofilter was supplied with nitrogen and phosphorus at a rate of 120 and 1,100 mL/hr as phosphoric acid and ammonium chloride, respectively. These were the lowest settings for the nutrient pumps; so lower enhancement concentrations could not be achieved. The operators chose to avoid dilution of nutrients, since adding water to a concentrated acid can be hazardous. Higher concentrations of nitrogen species (NH_4-N and NOx-N) and phosphorus species (TP-P and PO_4-P) were observed in the nutrient-enhanced biofilter, relative to the other filters (Figure 10). This was especially apparent with phosphorus. Similar bioavailable carbon (CBXAs-C and removed DOC) concentrations were observed across all filters, as expected. Evans et al. [14] indicated that DOC removals lower than 0.2 mg/L don't produce practical results; however, DOC removal was used to calculate the C:N:P ratio so comparisons could be made with other studies. Since median DOC removal was similar to CBXAs removal (Figures 5 and 6), it was assumed that DOC removal would be fairly representative of available carbon. The median influent concentrations of the non-enhanced filter for DOC removal, NH_4-N, and PO_4-P were 0.045 mg/L, 0.009 mg-N/L, and 0.005 mg-P/L (PO_4-P MDL), respectively, which equates to a C:N:P ratio of 100:14:3.4. The majority of PO_4-P samples were below the detection limit of 0.005 mg-P/L, so it's likely that the actual concentration was lower. However, similar concentrations of TP and PO_4-P were observed throughout the study, indicating that TP was mostly comprised of PO_4-P. The median non-enhanced filter influent for TP was 0.052 mg-P/L, which would suggest that the median concentration for PO_4-P was likely near the detection limit. The recommended optimum ratio for drinking water filter microorganisms is 100:10:1 [31]. This suggested that the system was carbon-limited and not nitrogen- or phosphorus-limited, since the ratio was being met. Despite the 100:10:1 ratio already being met, WCWCD chose to continue to dose nitrogen and phosphorus to the filter to determine if excess nutrients could further increase growth and organic carbon removal. The median influent concentrations of the enhanced biofilter for DOC removal, NH_4-N, and PO_4-P were 0.06 mg/L, 0.016 mg-N/L, and 0.075 mg-P/L, which equates to a C:N:P ratio of 100:23:49. This suggested that the filter was being overdosed two times the recommended N amount and almost 50 times the recommended P amount.

Azzeh et al. [3] found that overdosing pilot filters to a C:N:P (DOC removal:NH_4:PO_4) ratio of 100:40:20 resulted in a decrease of biopolymer by 25% relative to the control biofilter. To date, no full-scale biofiltration plants studies were found that investigated the impact of nutrient overdosing. The QCWTP full-scale experiment has shown that overdosing can have detrimental impacts on backwash time/volume and turbidity breakthrough (Figures 8-9). Therefore, it is recommended that plant managers properly investigate the water matrix before incorporating nutrient enhancement.

5. Conclusions and engineering significance

The potential benefits of biological filtration (i.e., higher organics and DBP precursor removal) have led many drinking water treatment plants to convert their conventional filtration system to a biologically active system. This is often accomplished by removing chlorination before filtration. However, after the removal of pre-chlorination problems such as turbidity breakthrough, manganese release, and finished water quality deterioration frequently occur. Experiments have been conducted that compare water quality and hydraulic performance before and after biological conversion [48], but little research has been done to evaluate the side-by-side conversion process with other non-biological filters, especially in a carbon limited environment. The overall objective of this study was to evaluate how the biological conversion process impacted water quality and filter performance by comparing operational and water quality parameters against other non--biological filters at the QCWTP. The following are the key findings from the study:

- The source water was found to be low in organic carbon (TOC < 2 mg/L and CBXAs < 50 µg-C/L). There was a statistically significant reduction across the biofilters, but the differences among the filters' performances were small (< 0.06 mg/L-DOC and < 3 µg-C/L-CBXAs) and considered practically insignificant. No relationship was found between filter run time and organic carbon removals.
- Small differences in biological activity (as indicated by ATP) were observed between the biological and non-biological filters (median differences ranging from 175-2,300 ng ATP/cm^3). EPS media concentration was also found to be low on the biofilters. ATP concentration was also found to decrease with media depth. The low biological activity was likely due to the limited carbon supply, where it's probable that the biofilters degraded all the assimilable organic carbon available.
- Biofilters improved finished water quality by reducing DBP concentrations, compared to the non-biological filter. This was observed despite there being no differences between organic carbon removals among filters. Only a slight increase of effluent manganese was observed, compared to the non-biological filter after pre-chlorination removal, but manganese stabilized after a few months.
- Biological conversion had no impact on filter performance, as indicated by UFRVs, head loss accumulation and filter run times. However, it did have a small impact on effluent turbidity, where slightly higher (~0.014 NTU) and more variable final effluent turbidity values (though still within EPA drinking water standards and even within the plant's operational goals, established by the Partnership for Safe Water and the Utah Water Quality Alliance) were observed at the biofilters compared to the non-biological filters. The trigger for biofilters backwash shifted from being driven by head loss to more often being driven by turbidity breakthrough.
- Overdosing nutrients (C:N:P-100:20:50) had an negative impact on filter performance. Longer backwash times and rates were required for the nutrient enhanced biofilter. Near the end of the study, the nutrient-enhanced biofilter experienced frequent early turbidity breakthrough (filter run time of 10- 20 hours) until the extra nutrients were shut off. Excess nutrients can build up on filters, which possibly caused media clumping and turbidity breakthrough.

- Chlorinated backwashing appeared to have no significant impact on the biofilter's performance, where only small differences in ATP concentration and hydraulic performance were observed between the chlorinated backwashed biofilter and standard biofilter.

In a carbon-limited system, the removal of pre-chlorination improved water quality by reducing DBP formation, but slightly increased variability of final effluent turbidity, shifting the backwash triggering mechanisms from being head loss-driven to more turbidity-driven. After the addition of ozone, which is expected within the next 5 years, improvements in organic carbon removal and microbial growth would likely be observed, thus making biological conversion a viable option for the QCWTP.

The majority of biofiltration studies show results before and after biofilter conversion. This study shows a comparison of biologically active filters and non-biologically active filters in real time; thus, removing differences in raw water quality typically observed from month to month. Many conventional drinking water treatment plant managers are converting their filters to biological mode to improve water quality. However, unintended consequences may arise during the conversion process (e.g., manganese release, turbidity breakthrough). In order to mitigate these problems, a greater knowledge base on this subject is required. Resources are especially limited for conventional plants converting to biological mode with a source water low in nutrients (carbon, nitrogen, and phosphorus).

The insights gained from this study can provide guidance to plant managers who convert non-ozonated, anthracite media plants with low influent nutrients to biofiltration plants. The following insights from the study can be applied to other utilities considering biofiltration conversion:

- A proper investigation of the water matrix is important to determine which, if any, nutrient is limited. Overdosing nutrients may be unwise for a system low in carbon.
- Despite carbon limitation, lower formation of DBPs can still be achieved in biofilters compared to non-biological filters.
- Increased NOM removal would likely be small; therefore, this should not be the only driver for biological conversion.
- Slightly more variable final effluent turbidity might occur. This would likely have no impact on a system whose backwashing mechanism is typically triggered by head loss. However, a system whose backwash mechanism is typically triggered by turbidity breakthrough should carefully monitor filter run times and UFRVs after biofilter conversion.

6. Future research

To date, limited research has been conducted at full-scale biological drinking water treatment plants in low-nutrient, low-turbidity environments. Therefore, further research should be conducted to investigate the impact of a low carbon environment on:

- EPS production and its potential impact on filter head loss;
- Impact of different carbon, nitrogen, and phosphorus sources on EPS production and ATP concentration;
- The composition of NOM and its use by microorganisms;
- The reduction of DBPs formation compared to the removal of different forms of carbon;
- The impact of chlorinated backwash water on concentration and composition of microorganisms in filter media.

Extensive research has been conducted investigating proper coagulant and polymer doses for conventional treatment plants. However, conversion to biological filtration could have an impact on coagulant and polymer doses (even though filtration follows coagulation and flocculation). A greater knowledge base in this subject could help plants managers optimize the biological conversion process. A guidance manual (currently under development, sponsored by the Water Research Foundation) specifically catered to plant operations on diagnosing potential problems that could arise during the biofiltration conversion process will be beneficial.

6.1 Guidance for the QCWTP

The following is a summary of recommendations and monitoring guidance for the Quail Creek Water Treatment Plant managers and operators, once the filters are operated in biologically-active mode.

6.1.1 Further investigation

The following list highlights work that could be conducted to close some gaps, indicated throughout the study:

- Determining if a filter is operating in biologically-active mode;
  - Scanning Electron Microscope (SEM) and microbial characterization and abundance could be evaluated to compare differences between non-chlorinated and chlorinated filters.
- Establishing the impact of chlorinated backwash on a biological filter;
  - If full-scale biofiltration is implemented throughout the entire plant, half of the biofilters could be cleaned with chlorinated backwash to compare differences in performance.
  - Microbial characterization should be conducted to determine if chlorinated backwash promotes growth of pathogenic or chlorine-resistant bacteria.
- Evaluating effluent turbidity during biological conversion;
  - Coagulant and polymer dose adjustments should be made.
- Determining the impact of nutrient addition;
  - It's recommended that nutrient addition should not be implemented at full-scale; however, pilot studies could be conducted to determine why nutrient overdosing resulted in turbidity breakthrough.
- Establishing the relationship between organic carbon removal and biological activity;
  - Another filter core study should be conducted to determine which parameter, if not biological activity, has the largest impact on organic carbon removal.

It is recommended that ozone be implemented at the plant to increase the availability of carbon in the water. However, after ozone installation, more testing should be conducted to re-evaluate filter performance. If increased carbon leads to limiting phosphorus and nitrogen conditions, nutrient addition might also be re-evaluated.

6.1.2 Monitoring guidance

Table 3 (after [14];[50]) provides potential monitoring parameters and frequencies for the plant during the conversion process. After the biofilters are fully acclimated and parameter results are remaining stable, sampling frequency may be decreased.

Table 3. Potential monitoring parameters and frequency during biofiltration conversion
Tabela 3. Potencjalne parametry i częstotliwość monitorowania podczas konwersji do biofiltracji

Category	Parameter	Location	Frequency	Comments
Water Quality Parameters	Temperature	Biofilter influent	Continuous	
	pH	Biofilter influent & effluent	Continuous	
	DO	Biofilter influent & effluent	As needed	Consider if in-line DO probe is installed
	Turbidity	Biofilter influent & effluent	Continuous	Biofilter influent must be measured by grab samples
	TOC / DOC	Biofilter influent & effluent	1-2/week	
	UV_{254}	Biofilter influent & effluent	1-2/week	
	BDOC	Biofilter influent & effluent	1-2/month	Use to check DOC and CBXAs removal
	CBXAs	Biofilter influent & effluent	1-2/week	
	Ammonia	Biofilter influent	1/week	For C:N:P ratio
	Orthophosphate	Biofilter influent	1/week	For C:N:P ratio
Water Quality Parameters	Manganese	Biofilter influent & effluent	1/week	More frequent sampling during Mn release evaluation
	DBP formation potential	Finished water	1/month	
	Chlorine residual	Biofilter influent & backwash	1-2/day	
Hydraulic Parameters	Hydraulic loading rate		Continuous	
	Filter run time		Continuous	
	UFRV		1/run	
	Terminal head loss		Continuous	
	Clean bed head loss		1/run	
	Headloss accumulation		Continuous	
	Backwash flow rate		At backwash	
	Air scour rate		At backwash	
	Underdrain pressure		Continuous	
Microbial Parameters	ATP	Biofilter media	1/week	Especially important to monitor during bioacclimation period
	HPC	Finished water	1/week	
	EPS	Biofilter media	As needed	Test if hydraulic performance deteriorates

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