IMPROVING MULTI-OBJECTIVE SUSTAINABILITY AT SMALL DRINKING WATER SYSTEMS: MODELLING & DECISION-SUPPORT

by

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ABSTRACT

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Improving Multi-objective Sustainability at Small Drinking Water Systems: Modelling & Decision-support

Thesis directed by Assistant Professor Sherri M. Cook

Small systems represent 95% of United States' drinking water systems, but they often face technical, managerial, and financial challenges. These challenges hinder small systems' abilities to make sustainable treatment decisions and these issues are being exacerbated as regulations become more stringent and source water quality degrades due to population growth and anthropogenic pollution. To provide effective decision-support that is relevant to the tens of thousands of small systems, this dissertation aimed to: characterize and then assess the relative environmental impacts of common small system filtration and disinfection treatment processes; construct a decision making tool that navigates trade-offs between the multiple sustainability dimensions to provide a small system stakeholder with custom treatment recommendations; and provide insights on treatment process selection based on source water quality, stakeholder preferences, and regulatory requirements while considering design uncertainty.

Life cycle assessment was used to quantify the environmental impacts resulting from filtration and disinfection treatment processes. For filtration processes, the results showed that conventional filtration was more environmentally preferred for more pristine source waters. Biological filtration was more preferred when the source water was more degraded and treatment requirements were more stringent. For disinfection processes, chlorine was environmentally preferred to ultraviolet disinfection in most cases. Ultraviolet disinfection was preferred when its Cryptosporidium reduction benefits could be realized. Life cycle assessment and multiple-criteria decision analysis methods were used in unison to construct a comprehensive and rigorous decision-making tool for small systems. The tool is universal in its ability to provide small systems with insights for making a treatment decision under distinct source water qualities, stakeholder preferences, and regulatory requirements while considering design uncertainty. The tool was used to evaluate 60,000 diverse, simulated small systems. Results provided specific insights for diverse small systems that have different source water quality and treatment constraints. Also, the selected filtration process has the largest impact on a treatment train recommendation. Ultimately, different stakeholder groups (e.g., consumers, engineers, operators, and regulators) can gain insights, using site specific characteristics, to make sustainable treatment decisions that will improve drinking water quality.

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Chapter 1. Introduction

1.1 Background

Small drinking water systems serve up to 2.5 billion people globally.[1] In the United States (U.S.), 95% of drinking water systems are small drinking water systems (serving less than 10,000 people),[2] and they serve about one fifth of the population.[3] A majority of drinking water systems are being supported by a minority of the population, which often proliferates in small systems having problems with technical, managerial, and financial planning.[4] Therefore, many small systems currently need help improving their existing treatment. Further complicating the problems that small systems face, regulations are becoming more stringent, and source water quality is expected to degrade into the future due to population growth and climate change.[5–8] Source water quality has already necessitated updating treatment or watershed management approaches, even for large systems (e.g., Portland, OR; New York, NY). Regulation is also becoming more stringent and, in some cases, states have the ability to make treatment processes viable by either allowing their use or giving them pollutant reduction credit. This will continue complicating small drinking water systems' ability to remain in compliance with regulations, and subsequently, to protect consumer health.[3]

Drinking water systems of all sizes cope with degrading source water quality by increasing chemical doses, adding new treatment processes, and/or using more energy intensive processes. This additional use of materials, chemicals, and energy provides a positive feedback loop for the very contaminants that contribute to source water degradation (i.e., via climate change). Anthropogenic actions have put the earth beyond allowable thresholds for biodiversity loss, climate change, and nitrogen cycle impacts,[9] so any reduction in those forms of pollution is imperative. Drinking water treatment may not be the largest contributor to pollution, but the industry does have opportunities to reduce pollution. Therefore, small drinking water systems require sustainable decision-making tools to provide insight for treatment decisions while ensuring that their consumers remain protected under changing conditions.

1.2 Problem

There have been previous attempts at providing sustainable decision-making tools to small and large drinking water systems. However, previous studies are limited in evaluating drinking water treatment alternatives for environmental impacts using life cycle assessment (LCA), which is an ISO standardized methodology to quantify environmental impacts.[10] There has been a lack of focus on smaller drinking water systems (less than 1.56 MGD,[3] assuming 157 gal/day/capita[11] for 10,000 people), but some environmental LCA work has proven useful for larger drinking water systems (with a flow range of 3.18[12] to 174[13] MGD[12,13,22–31,14,32–37,15–21]). Many of these studies have large system boundaries that incorporate all pumping to and from the drinking water plant as well as wastewater treatment impacts,[12,13,38,39,14,16,20,21,27,29,30,36] which are location specific parameters and means these studies do not provide sufficient resolution for evaluating drinking water treatment processes. There were also two studies that developed modeling tools for drinking water treatment plants; however, their data cannot be translated to small systems because they focused on optimizing operations at larger drinking water treatment plants and assumed that infrastructure impacts were negligible.[19,31] These models have proven useful for larger systems, but there is still a need for a model that captures small system implications. These studies are better suited to determine the relative environmental impacts of municipal water cycle processes than to determine environmental impact trends specific to drinking water treatment processes for a wide range of technologies.

Small systems use technologies on a different scale from larger systems, which further limits the translation of large system LCA results to small systems; for example, ultraviolet (UV) disinfection at small systems often uses a single lamp reactor instead of a bank of lamps. Common processes that have been assessed in larger drinking water system boundaries include: ozone generation,[15,17,19,22,25,28,34] granular activated carbon generation for tertiary and primary filtration,[17,18,21,29,31,32,37] operational energy for membrane treatment,[15,17,18,31,34] and pumping from the source water.[16,18,19,22–24,34,38] Technologies that small systems may use include chlorine disinfection with multiple contact zones, conventional (coagulation, sedimentation, rapid media filtration) filtration, membrane filtration (ultra and micro), cartridge filtration, bag filtration, biofiltration, and slow sand filtration. Therefore, there is a need for life cycle assessment studies that are relevant to small drinking water system specific treatment processes that each have their own benefits and disadvantages.

Also, previous drinking water LCA work has used case studies,[12,13,23–30,32,33,14,34–37,15–18,20–22] which have limited applicability outside of the case study location because each location has unique considerations with different functional units. Further, most of the studies were conducted in Europe, which provides different embodied impacts and treatment requirements for drinking water treatment plants than the U.S. For example, a chlorine residual is not required in Europe and more emphasis is placed on the biostability of the water before entering distribution system.[40] One previous study did evaluate small drinking water treatment alternatives for reducing disinfection by-products at a small system scale, but has the same limitations of other case study-based LCAs.[41] There are tens of thousands of small systems that require treatment insights and it is infeasible to perform a case study of each one. Therefore, there is a need for small drinking water decision-support results, including environmental impacts, that are applicable to a wide range of systems as opposed to site specific case study results.

Comprehensive environmental factors should be incorporated in the decision-making process using life cycle thinking.[42] Although there is a need for small drinking water system LCA work to characterize environmental impacts, other categories—such as affordability, human health, and system specific challenges—should also be evaluated with life cycle thinking to support a truly sustainable decision.[42] Using environmental as well as economic and social factors while making treatment decisions can prevent unforeseen problems that may occur if only one or a few factors are assessed.

When more decision criteria are considered, it becomes difficult to navigate the trade-offs between them. One strategy that is helpful for navigating multiple decision criteria is multiple-criteria decision analysis (MCDA). However, there has been limited research using MCDA that is relevant to small drinking water systems. Previous drinking water MCDA studies have primarily focused on larger drinking water systems, and subsequently on decision criteria specific to those larger systems (e.g., operational cost, human health risks).[8,43,44] However, small systems are unique in the decision criteria that represent goals of their stakeholders. Large system studies were also mostly focused on optimizing complex treatment processes that were already installed. Small systems are not well represented by the treatment processes being optimized and often have difficulty making an appropriate capital decision before optimization is an option. There has been some research to help small systems make decisions about source water availability,[45] cost modeling,[46] and financial aid options,[47] but each of these studies only focused on one decision criteria. One study did a case study of hydroelectric plant's small drinking water system treatment selection, but its criteria and treatment alternatives were specific to that plant.[48] Existing drinking water MCDA literature has provided useful information for specific systems, but existing MCDA approaches need to be significantly modified to small system needs. Therefore, there is a need for a comprehensive MCDA tool for small drinking water systems that incorporates life cycle thinking, provides small drinking water system stakeholders with easily understandable recommendations, provides stakeholders with decision-making agency, and represents small drinking water system specific treatment alternatives and decision criteria.

1.3 Research Questions

To address the need for small drinking water system LCA comparisons and MCDA decision-making tools, the work presented in this dissertation compared small drinking water treatment alternatives using LCA (Chapters 2-3) and incorporated those results into a broader decision-making tool whose results can be used to augment treatment decisions at small drinking water systems (Chapters 4-5). Research questions were developed to fill the major gaps in small drinking water system LCA and MCDA literature (Table 1.1):

1) Environmental Life Cycle Comparison of Conventional and Biological Filtration Alternatives for Drinking Water Treatment (**Chapter 2)**

<u>Motivation:</u> To meet the challenge of increasing concentrations of organic matter in source waters, coagulation chemical doses can be increased to remove excess organic matter. An alternative to increasing chemical doses is to achieve organic removal with a biologically active filter. However, the potential advantages and disadvantages of this newer biological filtration approach compared to conventional approaches are not well established.

<u>Question:</u> What filtration alternatives, between conventional filtration and biological filtration, are more environmentally sustainable for a variety of source water qualities and regulatory conditions?

Main Conclusion: Conventional filtration was more environmentally preferred for more pristine source waters, and biological filtration was more preferred when the source water was more degraded and treatment requirements were more stringent.

2) Life Cycle Environmental Impacts of Disinfection Technologies Used in Small Drinking Water Systems (**Chapter 3)**

Motivation: Due to concerns about disinfection byproducts (DBPs) and chlorine-resistant pathogens, alternatives to chlorine-intensive disinfection have drawn interest. However, there has been limited research comparing alternative drinking water disinfectants to chlorine, mostly because it is difficult to make an equal comparison, but LCA can be used to ensure comparisons are equal.

Question: What disinfection technologies, between chlorine and UV, are most environmentally sustainable for a variety of regulatory conditions?

Main Conclusion: Chlorine was environmentally preferred to UV in most cases, but UV was preferred when its benefits of *Cryptosporidium* removal could be leveraged.

3) A New Framework for Small Drinking Water Plant Sustainability Support and Decision-making (**Chapter 4)**

Motivation: Small drinking water systems face numerous challenges when making decisions about treatment processes. These challenges will be exacerbated by degrading source water and stringent regulations, and they must manage these changes while considering multiple decision criteria. However, they do not have decision-support resources to navigate complex trade-offs and make more informed treatment decisions.

Question: Can a model-based tool that aggregates multiple decision-criteria scores support small drinking water decision making through relevant engineering information, navigating trade-offs, and decision-making insights?

Main Conclusion: The tool is universal and provides small drinking water systems with insights for making a treatment decision under varied design parameters, source water inputs, and stakeholder preferences. It also has the capacity to be updated easily to remain relevant over time as treatment technologies change.

4) Evaluating Small System Treatment Processes for Different Source Water Qualities, Regulations, and Stakeholder Preferences (**Chapter 5)**

Motivation: Small systems will have to update their treatment with changing conditions and there are many treatment processes that may be used to do so, each has benefits and disadvantages. Previous studies have used case studies to navigate treatment process selection, but this approach is not well suited to providing decision-support for the tens of thousands of diverse small systems that have unique needs.

Question: What are the most sustainable treatment processes for different small systems that have diverse source waters, stakeholder preferences, and regulatory requirements?

Main Conclusion: There was no one treatment process that was the best option for every simulated small system and filter selection was the most important indicator of the performance of the treatment train.

Table 1.1 Summary of motivation of chapters and research findings.

Ch.	Gap in Literature	Question	Methods	Main Conclusion
2	Biological filtration has drawn interest for its potential to reduce chemical use, but there has been no environmental characterization of the filtration process to comprehensively and quantitatively evaluate its potential benefits.	What filtration alternatives, between conventional filtration and biological filtration, are more environmentally sustainable for a variety of source water qualities and regulatory conditions?	Life cycle assessment; process modeling	Conventional filtration was more environmentally preferred for more pristine source waters, and biological filtration was more preferred when the source water was more degraded and treatment requirements were more stringent.
3	Alternatives to chemical disinfection have several benefits, but the environmental impacts of using non-chemical disinfection, such as UV, are not well understood and have not been rigorously quantified, especially in a drinking water treatment context.	What disinfection technologies, between chlorine and UV, are most environmentally sustainable for a variety of regulatory conditions?	Life cycle assessment; process modeling	Chlorine was environmentally preferred to UV in most cases, but UV was preferred when its benefits of Cryptosporidium removal could be leveraged.
4	Small systems do not have access to comprehensive decision-support when managing challenges of more stringent regulations and degrading source water quality.	Can a model-based tool that aggregates multiple decision-criteria scores support small drinking water decision making through relevant engineering information, navigating trade-offs, and decision-making insights?	Multiple-criteria decision analysis; life cycle assessment; process modeling	The tool is universal and provides small drinking water systems with insights for making a treatment decision under varied design parameters, source water inputs, and stakeholder preferences. It also has the capacity to be updated easily to remain relevant over time as treatment technologies change.
5	There are tens of thousands of diverse small drinking water systems that will need to update their treatment, but there are too many to use a case study decision-support approach at all of them.	What are the most sustainable treatment processes for different small systems that have diverse source waters, stakeholder preferences, and regulatory requirements?	Monte Carlo analysis; multiple-criteria decision analysis; life cycle assessment; process modeling	There was no one treatment process that was the best option for every simulated small system and filter selection was the most important indicator of the performance of the treatment train.

Chapter 2. Environmental Life Cycle Comparison of Conventional and Biological Filtration Alternatives for Drinking Water Treatment

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Abstract

Drinking water utilities face challenges with meeting increasingly stringent regulations, often at higher costs and operational complexity, and all are expected to increase, especially with deteriorating source water quality. Biofiltration, which enables organic matter biodegradation, may be used as an alternative to conventional filtration to reduce chemical coagulant requirements while maintaining equivalent treatment. However, the advantages and disadvantages of filtration options depend on many factors, including source water, complex water chemistry interactions, and other site-specific conditions, such as the use of pre-ozonation to improve biofiltration performance and different types of chemicals for coagulation and disinfection. To identify and quantify the environmental and performance trade-offs between conventional and biological filtration, with and without pre-ozonation, a comprehensive modelling and systems approach is needed. To this end, life cycle assessment methodology was used to develop a new model in order to compare the environmental impacts of drinking water treatment trains that used conventional filtration, nonozonated biofiltration, and ozonated biofiltration. All were designed to produce the same water quality, in terms of total organic carbon (TOC), turbidity, *Giardia*, *Cryptosporidium*, and virus reductions. To account for different treatment targets, 4 treatment scenarios (summarized by differences in TOC removal requirements: 30%, 40%, 50%, and enhanced coagulation requirements based on U.S. regulation) were evaluated. The relative environmental impacts of all three treatment train alternatives, under each treatment scenario, were evaluated for 60,000 unique

source waters. Generally, ozonated biofiltration had the worst environmental performance while nonozonated biofiltration had the lowest environmental impacts. However, the comparison of nonozonated biofiltration to conventional filtration depended on the treatment scenario and source water quality. For example, under the 50% TOC removal treatment scenario, nonozonated biofiltration had better relative environmental performance when the source water had high alkalinity (>50 mg $CaCO_3$/L), low SUVA (<2.75 L/mg/m), high pH (>7), and high temperature (especially >20°C). Under the enhanced coagulation treatment scenario, both conventional filtration and nonozonated biofiltration had similar environmental impacts for most source waters. This new model and comprehensive water quality analysis can help utilities decide which filtration alternative best meets their needs, especially by reducing environmental impacts while improving drinking water quality.

2.1 Introduction

Drinking water utilities are facing many challenges associated with source water degradation and increasingly stringent regulations. Contaminant concentrations, especially of organic matter, are expected to increase in source waters as the use of non-traditional source waters and population density increases.[5–8] Utilities are trying to modify their existing treatment processes to address these challenges and improve drinking water quality, but this usually results in higher costs and operational complexity. For example, utilities modify the conventional surface water treatment train, which consists of chemical coagulation, flocculation, sedimentation, filtration and disinfection (with disinfectant addition before the filter).

A common modification is to increase the coagulant dose in order to increase organic matter removal, which is often measured as total organic carbon (TOC) removal and improve the control of disinfection by-product (DBP) formation. While effective, the excessive use of coagulation chemicals adds burdens to the water treatment utility, such as increased costs, chemical handling, and residuals management.[49] Another modification, which is being used increasingly and well established in Europe, is biological filtration (biofiltration). By precluding a disinfectant residual in the filter influent, indigenous microorganisms can form an attached biomass on the filter media and utilize the source water's organic matter as substrate. A transition to biofiltration can be achieved by moving the disinfectant addition until after the filter or allowing any upstream disinfectant to dissipate before the filter, thus facilitating biological acclimation and growth.[50]

There are advantages and disadvantages of biofiltration. It can provide the same final water quality as conventional treatment; biofiltration was found to achieve the same filtered water turbidity regulations[50–53] and pathogen removal, including *Cryptosporidium*.[54] The benefit is that the biofiltration treatment approach can lower the dependence on coagulation, resulting in lower chemical coagulant doses. This is because the coupling of coagulation and biodegradation has a mostly additive organic matter removal benefit since biofiltration preferentially removes aliphatic, smaller molecular weight compounds while coagulation removes aromatic, larger molecular weight compounds.[55–57] Also, biofiltration can remove biodegradable trace organic compounds, such as 2-methylisoborneol and geosmin.[58] Biofiltration can also remove specific organic contaminants such as pesticides,[58,59] pharmaceuticals[60] and personal care products, which can be endocrine disrupting compounds.[61,62]

In contrast, biofiltration may be less robust than conventional filtration. For instance, biofilters require an acclimation period of at least a month[63] before achieving steady-state contaminant removal.[64] Also, seasonal variations in water quality, especially temperature, can affect organic matter removal efficiency.[53,65] One of the most important factors that can limit the benefits and effectiveness of biofiltration is the source water organic matter composition. Source water organic matter is comprised of a suite of organic compounds that exist in natural waters as well as those from upstream treated wastewater discharge,[57,66] but only some of these compounds (humic substances, amino acids, carbohydrates, aldehydes and ketoacids) are biodegradable.[67–71] Further, coagulation can also remove some biodegradable organic matter.[52,72] As a result, biofiltration may only be able to remove a negligible amount of TOC.

Therefore, ozone is used to help avoid this result and to change the composition of organic matter. Specifically, ozone is used to increase the biodegradability of the organic matter in drinking water by oxidizing aromatic, unsaturated organic structures to saturated polycarbonaceous compounds of low molecular weight, which are termed organic ozone by-products.[51,55,73,74] To achieve this and increase the total organic matter removed by biofiltration, ozone is added after coagulation but before the biofilter (pre-ozonation). Other benefits of pre-ozonation include the oxidation of taste and odor causing compounds, anthropogenic trace organic compounds, and inorganic compounds. Some disadvantages include the increased cost and complexity of a pre-ozonation process, and some of the ozone by-products can be a human health concern if not removed.[75–77]

Given the many disadvantages and advantages of the different filtration alternatives, a systems approach is needed to understand and quantitatively evaluate the currently unclear trade-offs between these options. Specifically, life cycle assessment (LCA) can be used to analyze and help improve the efficiency of a treatment process, especially by identifying ways to reduce chemicals or costs. Multiple drinking water LCA studies have compared diverse filtration alternatives, such as using granular activated carbon for tertiary drinking water treatment against other filtration options,[17,18,31] or granular activated carbon within a specific drinking water treatment plant,[21,22,29,32] or comparing membrane to conventional filtration;[17,18,31,78] but they have focused on physicalchemical processes. Assessments of biological drinking water treatment approaches are still needed. Specifically, since previous studies have found that sorptive media production was a significant source of environmental impacts[17,18,32] and costs,[49] a study that evaluates the use of non-sorptive media, such as anthracite over sand, for biological filtration is needed.

Also, previous drinking water LCAs have evaluated the impacts of and identified ways to optimize existing treatment processes, such as the use of granular activated carbon at specific drinking water treatment plants in Romania,[21] Spain,[29] Amsterdam,[22] and the United States (U.S.)[32] While providing very detailed information, case studies are usually limited in their wide application to multiple drinking water treatment utilities. There is also still a need for a systems approach that evaluates a diverse set of source waters and treatment objectives to help utilities understand and navigate the trade-offs between treatment alternatives; this is especially true given the many trade-offs due to water quality, specifically complex water chemistry interactions and organic matter composition. This evaluation approach will help provide insight and improve understanding of how treatment and environmental performance will change with changing regulations and source water quality, both seasonally and long-term.

To help address these needs, this study used LCA methodology to identify and quantify the environmental performance and trade-offs between three filtration alternatives for a diverse set of source waters and treatment objectives. An LCA model was developed to quantify the relative life cycle environmental impacts associated with the different types and amounts of energy, chemicals, and materials used during the operation of the three drinking water treatment trains. Each treatment train used the same common treatment processes of coagulation, filtration, disinfection, and pH adjustment, but with different filtration alternatives (Figure 2.1): 1) conventional filtration; 2) biofiltration without pre-ozonation, termed nonozonated biofiltration; and 3) biofiltration with pre-ozonation, termed ozonated biofiltration. All treatment trains were designed to meet the same functional unit: the treatment of 1 m^3 of a surface water to drinking water standards. To account for different treatment standards, 4 treatment scenarios were defined, in terms of TOC, turbidity, *Giardia*, *Cryptosporidium*, and virus reductions. A total of 60,000 different source waters were evaluated. Uncertainty and sensitivity analyses evaluated the impact of typical design ranges for different parameters and multiple types of pH adjustment approaches and chemicals, coagulant chemicals, and disinfectants on the results. Overall, the comparative LCA model was used to elucidate the treatment standards and source water quality conditions that can result in environmental impact reductions when using either conventional filtration, nonozonated biofiltration, or ozonated biofiltration.

2.2 Methods

2.2.1 Life Cycle Assessment

To compare drinking water treatment trains using either conventional filtration, nonozonated biofiltration, or ozonated biofiltration, comparative LCA methodology following the ISO 14040 framework was used.[10] The functional unit was to treat 1 m^3 of a defined surface water quality to a defined drinking water standard, over 40 years, at a drinking water treatment utility with a flow rate of 1.0 Mm^3/yr (0.7 mgd to represent a medium sized system); to account for different drinking water treatment objectives and conventions, four treatment scenarios were evaluated, which are summarized as having constant TOC removals of 30%, 40%, and 50%, as well as an “enhanced coagulation” TOC removal requirement that was based on U.S. regulations (one of the following: 15% to 50% (based on source water quality, Table A1); 2 mg/L effluent TOC; or 2 mg/L effluent specific ultraviolet absorbance (SUVA)). Regardless of the TOC removal goal, treatment also included the reduction of turbidity (met with minimum allowable alum dose), *Giardia* (3-log reduction), *Cryptosporidium* (2-log reduction), and viruses (4-log reduction). The three filtration options (conventional filtration, nonozonated, and ozonated biofiltration) were considered in the context of a full treatment train, which consisted of coagulation, flocculation, sedimentation, (bio)filtration (with pre-ozonation as applicable), disinfection, and final water pH adjustment.

The LCA model framework and major assumptions were summarized below, with more details in Appendix A. The LCA system boundary for each treatment train included the production of chemicals, infrastructure materials, and electricity as needed for each treatment process and included hauling, which accounted for frequent chemical and waste solids hauling but excluded infrequent materials hauling (Figure 2.1). The hauling of waste solids was included in the system boundary to account for different amounts of settled coagulant and TOC, removed after coagulation, for each treatment train, similar to previous drinking water LCAs.[32] Requirements that were equivalent for all three treatment train alternatives were excluded from the comparative analysis (e.g., mostly water flow dependent requirements, such as sedimentation tank material amount and coagulation-flocculation mixing energy).

For 40 years of drinking water treatment utility operation, the total amount and type of chemicals, energy, hauling, and materials requirements, which accounted for typical replacement rates,[79,80] were calculated for each source water under each treatment scenario (Figure 2.1 and Appendix sections A3 to A5). All equations used to calculate the inventory are in the supporting information, and the full life cycle inventory (LCI) for the typical source water treatment under the enhanced coagulation scenario, normalized to 1 m^3, was provided as an example in Table 2.3. Data and equations for the treatment process modeling was based on conventional engineering design and experimental data from the literature. The inventory values were translated into life cycle environmental emissions using the unit processes available in the ecoinvent[81] and US-EI

2.2[82] LCI databases (Table 2.2). These total emissions were then translated into environmental impacts using the EPA’s Tool for Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI v2.1)[83] method, which includes 10 categories: acidification (kg SO_2 eq.), carcinogenics (comparative toxic unit human toxicity potential, CTUh), ecotoxicity (comparative toxicity unit ecotoxicity potential, CTUe), eutrophication (kg N eq.), fossil fuel depletion (MJ surplus), global warming (kg CO_2 eq.), smog (kg O_3 eq.), non carcinogenics (CTUh), ozone depletion (kg CFC-11 eq.), and respiratory effects (kg $PM_{2.5}$ eq.). The LCI database unit process data and the TRACI characterization factors were accessed via SimaPro software and then imported into the treatment process model, which was a spreadsheet-based model.

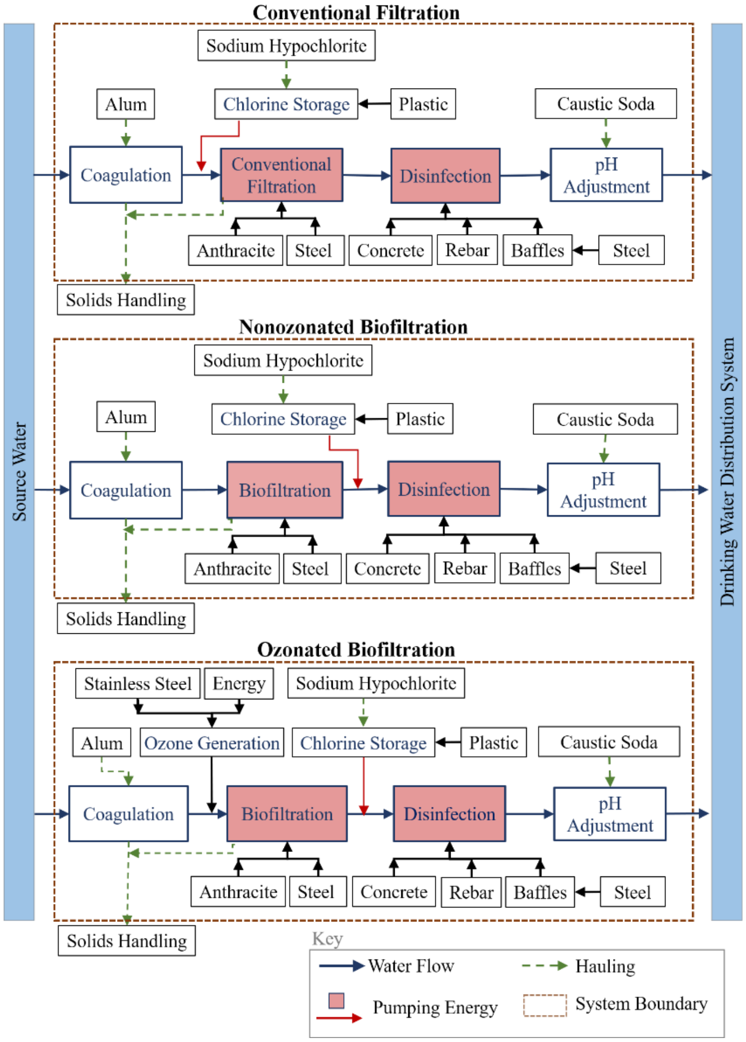

Figure 2.1. The LCA system boundary of the three treatment trains, which included treatment processes (blue text and lines) as well as LCI unit processes for the materials and chemicals (black text and lines), for hauling (dashed green lines), and for energy (red lines or red fill color). Refer to the web version for proper interpretation of references to color in this figure.

2.2.2 Treatment Scenarios

To account for different drinking water treatment objectives, regulations, and conventions, four treatment scenarios were evaluated; each included a specific requirement for TOC but the same requirements for effluent turbidity and reductions of *Cryptosporidium*, *Giardia*, and viruses. The first three treatment scenarios have a constant percent TOC removal, of 30%, 40%, and 50%; these percent removals were independent of source water quality. These scenarios represent regulations and conventions that maximize TOC removal, by adjusting the coagulant dose, such as in Europe, or utilities with high source water TOC that must achieve a percent TOC reduction, while the fourth scenario represents a focus on source water treatability, such as in the U.S. The fourth treatment scenario had an "enhanced coagulation" TOC removal requirement, which was based on U.S. regulations,[84] where the percent TOC removal was between 15% to 50% and specified for each source water quality, based on source water TOC and alkalinity (Table A1). All four treatment scenarios required an effluent turbidity limit,[84] which was met with a minimum allowable aluminum sulfate (alum) dose (Table 2.1), and required 2-log *Cryptosporidium*, 3-log *Giardia*, and 4-log virus reductions, which were met using a combination of credits from the filtration and disinfection processes.

2.2.3 Removal of Biodegradable Organic Carbon

While the details of competition between organic matter removed by coagulation and biofiltration processes are still not well known or documented, there is general trend shown by experimental data. Experimental data from the literature[52,85] and a pilot plant (Appendix section A3.3) show that the sequential removal of biodegradable dissolved organic carbon (BDOC) by coagulation and then biofiltration is mostly additive and linear. This experimental trend is also supported by theory. Specifically, larger molecular weight, aromatic organic compounds are preferentially removed by coagulation than smaller molecular weight, hydrophilic compounds,[52,55,56,86–88] and the opposite is true for biodegradation during biofiltration.[55,57,87] Coagulation and biofiltration processes remove mostly different types of organic matter but both have been found to remove biodegradable TOC, and this LCA model accounts for the additive TOC removal experimental trend up until BDOC availability is limited.

First, the total amount of biodegradable TOC was determined by applying a source water BDOC/TOC ratio, which was based on experimental data (Table 2.).[85] Second, the amount of this BDOC that was first removed by coagulation was determined based on experimental data (Table 2.).[52,72] Specifically, the experimental literature determined that the amount of biodegradable TOC removed during coagulation, which was found to have similar values for multiple source waters within two groupings of SUVA:[52,72] at SUVA values higher than 3 L/mg/m, 9% of the TOC removed by coagulation was biodegradable; at SUVA values less than 3 L/mg/m, 4% of the TOC removed by coagulation was biodegradable. This BDOC removal was assumed to be part of the total TOC removed by coagulation, which was predicted using the Edwards model (ESI† Section 2.5.1).

Third, the amount of the remaining non-coagulated BDOC that was removed by nonozonated biofiltration and by ozonated biofiltration was usually higher than the experimental removal values (Table 2.1 and Table A11);[85] the amount of removal modeled could be less than the experimental value if there was not enough remaining BDOC in the coagulated water (for nonozonated biofiltration) or not enough remaining BDOC in the coagulated and pre-ozonated water (for ozonated biofiltration) to achieve the experimental removal values. For nonozonated biofiltration, there were three temperature ranges that described average experimental BDOC removals, which were reported in the context of the source water's TOC:[85] 7% TOC removal at ≤ 10 °C, 10% TOC removal between 10 °C and 20 °C, and 15% TOC removal at ≥ 20°C.

To account for pre-ozonation, the amount of remaining BDOC after coagulation was adjusted to account for the transformation of some of the remaining organic matter into BDOC. This was done by also including an ozonated water BDOC/TOC ratio, which was based on experimental data (Table 2.1),[85] in the calculations. Pre-ozonation increases the biodegradability of the organic matter remaining in the coagulated water. Specifically, ozone transforms large molecular weight organic compounds into smaller, less aromatic compounds. Some transformations include the formation of hydroxyl, carbonyl and carboxyl groups, which have increased polarity and hydrophilicity,[51,55,74] and are more biodegradable than large organic compounds. Therefore, higher TOC removals were seen in the ozonated biofilter. For ozonated biofiltration, three temperature ranges described average experimental BDOC removals, which were reported in the context of the source water's TOC:[85] 11% TOC removal at ≤ 10 °C, 13% TOC removal between 10 °C and 20 °C, and 20% TOC removal at ≥ 20°C. Fourth, the amount of TOC removal needed by coagulation was reduced based on the amount of TOC (as BDOC) removal that was achieved by biodegradation, in the nonozonated or ozonated biofilter.

2.2.4 Source Waters and Quality

The parameters that defined a source water included TOC, pH, SUVA, alkalinity (measured as calcium carbonate, $CaCO_3$), and temperature. The typical source water was based on the average values from 271 water treatment utilities that used conventional treatment:[89] 3.2 mg/L TOC, 7.6 pH, 3.1 L/mg/m SUVA, 77 mg/L $CaCO_3$, and 15˚C (These average values are similar to the median values of all source waters leading to conventional treatment).[89] An additional 20,000 source waters, based on unique combinations of alkalinity, pH, TOC, and SUVA, for all four treatment scenarios, were also evaluated due to the dependence of treatment performance on varied source water qualities and treatment requirements. Each source water was generated using Latin Hypercube sampling of four continuous uniform distributions, which were based on common source water qualities: alkalinity of 20 to 125 mg/L $CaCO_3$, which included common values for surface waters (77 mg/L $CaCO_3$ average); pH of 6 to 8.5, which also included common values for surface waters (7.6 average); SUVA of 2 to 5 L/mg/m, which was within the range of common values (3.1 L/mg/m average); TOC of 2 to 5 mg/L, which was within the range of common values (0.35 to 5.3 mg/L, with a 3.2 mg/L average) but adjusted to represent only a range valid for the process modeling assumptions (e.g., based only on the range of TOC values where a linear, additive range of TOC removal by coagulation and then biodegradation was observed, as reported in the experimental literature[52,85]).[89] All 20,000 generated source waters were then assigned three different temperature ranges, less than 10°C, between 10°C and 20°C, and above 20°C, for a total of 60,000 unique source waters evaluated.

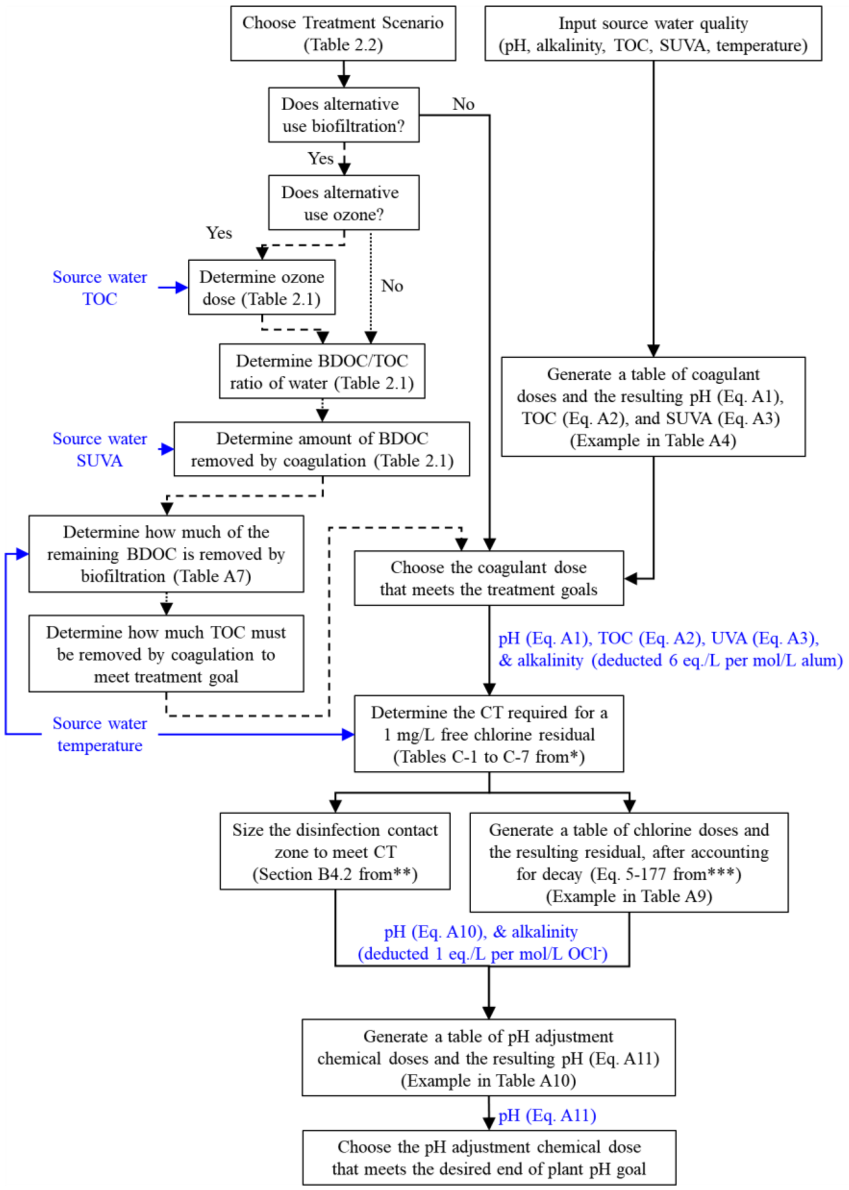

Figure 2.2. Summary of water treatment process modeling calculation steps. Note: * refers to U.S. EPA CT requirements,[90] ** refers to a previous disinfection LCA,[91] and *** refers to the water treatment plant model.[92]

2.2.5 Individual Treatment Process Inventory

The three treatment trains included the conventional drinking water treatment processes of coagulation, filtration, disinfection, and pH adjustment; the filtration processes was either conventional filtration, nonozonated biofiltration, or ozonated biofiltration (Figure 2.1). The nonozonated biofiltration treatment train was configured identical to the conventional filtration treatment train with one exception: disinfectant injection occurred after the filter (instead of before the filter) to support biological growth and organic matter biodegradation. The ozonated biofiltration train was configured identical to the nonozonated biofiltration treatment train with one exception: the addition of ozone prior to the filter to increase the biodegradability of the organic matter.

2.2.5.1 Coagulation

Alum was used to represent an effective and common coagulant. To achieve the required TOC removal needed by coagulation, the alum dose was calculated using the Edwards Langmuir-based semi-empirical model,[93] which uses source water SUVA and the pH after coagulation, which was based on the USEPA's water treatment plant model equations[92] (Eq. A), for each source water under each treatment scenario (Figure 2.2 and Appendix section A3.1). For the three constant TOC removal treatment scenarios, the alum dose needed to achieve the stated percent TOC removal (30%, 40%, or 50%) was used. For the enhanced coagulation TOC removal scenario, three alum doses had to be calculated, one for each of the U.S. regulated enhanced coagulation criteria: (i) the bin's percent TOC removal (Table A1), (ii) 2 mg/L effluent TOC, or (iii) 2 L/mg/m effluent SUVA.[94] Since only one of the three criteria needs to be met, the smallest of the three alum doses was used.

The calculated alum dose was then compared to the minimum allowable alum dose, which was the dose required for turbidity control; the final, modeled alum dose had to be the same as or larger than the minimum allowable alum dose. Since a quantitative relationship between coagulant dose and turbidity removal does not exist, this minimum allowable alum dose was assumed to be 10 mg/L (Table 2.1), which was based on alum doses used for turbidity removal only (i.e., typical alum doses used before enhanced coagulation was common) for 193 utilities.[95] The impact of this assumed value on the results was tested in the uncertainty and sensitivity analyses. Ferric chloride was also evaluated to determine any major environmental impact differences between coagulant chemicals.

2.2.5.2 Filtration

All filtration options used a rapid, dual media filter, anthracite over sand, with a stainless-steel housing and square cross-section. The filter area was based on a typical hydraulic loading rate of 15 m/h.[96] The filter height was based on the depth needed during operation, which was based on typical packed media depths and typical depths needed for water height above the media and freeboard.[96] In addition to the filter dimensions and configuration, typical steel thickness[97] and media densities[51] were assumed to calculate the total mass of anthracite and sand media and stainless steel mass for the filter housing. The filter energy requirement included pumping energy for operation and backwashing. It was assumed that pumping was required to overcome all head losses throughout the utility (e.g., there was no excess head), so operational pumping energy was based on the total filter height. Backwashing energy was based on a typical pumping pressure of 9 m, flow of 50 $m^3/h/m^2$, and 10 minute duration once per day.[98] Since previous studies have shown that biofiltration achieved similar effluent turbidity as conventional filtration under identical operating conditions,[50–53,99] it was assumed that nonozonated biofiltration and ozonated biofiltration achieved the same pathogen removal credits, as they are both high-rate granular media filters.[84] Therefore, the filtration process, for all treatment trains, was assumed to achieve a turbidity of 0.3 NTU; this earns credit for a minimum of 2-log *Giardia* (up to 2.5-log), 2-log *Cryptosporidium*, and 2-log virus removal.[84,100] More details are in the Appendix Section A4.

2.2.5.3 Disinfection

Chlorine disinfection was modeled as using sodium hypochlorite for all treatment trains. For conventional filtration, chlorine was added before the filter, to minimize any biological activity during filtration, and as seen in practice, the chlorine contact time credit for disinfection was only achieved after filtration. For the ozonated biofiltration treatment train, any disinfection credit from pre-ozonation was assumed to be negligible due to the high ozone demand and the low ozone doses. The chlorine contactor was a reinforced concrete basin with two stainless steel baffles. The disinfection calculations were based on a previously developed LCA drinking water disinfection model.[91] In summary, the concentration times time (CT) was determined based on the inactivation goals and the water's pH and temperature. The required dose was then calculated by assuming a required 1.0 mg/L free chlorine residual[91] leaving the utility and accounting for the water's chlorine demand. The chemical hauling and chlorine storage tank material requirements were based on this dose and weekly chemical deliveries. For every treatment train, disinfection achieved 1-log *Giardia* and 2-log virus inactivation in order to, in combination with the removal credit given for filtration, meet common disinfection requirements (e.g., the Long Term 2 Enhanced Surface Water Treatment Rule). Contact zone materials' masses were based on the required contact time. Pumping requirements were based on calculated head loss in the contact zone. Also, ultraviolet (UV) disinfection for the ozonated biofiltration treatment train was also considered (Figure A5); the life cycle environmental impacts of UV disinfection were quantified using an existing LCA model.[91]

2.2.5.4 pH Adjustment

Corrosion in the distribution system is a major concern and can be associated with low pH, which can result due to the pH dropping during treatment (e.g., during coagulation). Therefore, caustic soda (sodium hydroxide) was used as a common pH adjustment chemical. The final water's pH was adjusted to be a pH of 8.2 for common corrosion control. To determine the amount of pH adjustment chemical needed, carbonate chemistry equilibrium relationships[92] were used to first determine the pH after coagulation (Eq. A) and then subsequently after chlorination (Eq. A10). Next, the amount of caustic soda needed to achieve the final pH was determined (Eq. A11). To represent a wide array of utilities, since corrosion control is very site specific, two other pH adjustment (i.e., final pH) options were also considered: (i) no pH adjustment and (ii) effluent pH adjusted to source water pH. In addition, lime was also considered to determine any environmental impact differences between different pH adjustment chemicals.

2.2.5.5 Ozone Generation

An air-fed ozone generator was used for on-site ozone generation. This treatment process was only included in the ozonated biofiltration treatment train, where ozone was applied as an oxidant immediately before filtration. The typical pre-oxidation ozone dose in the literature per mass of TOC was 0.5,[85] and the literature specific energy use per mass ozone generated value was 0.020 kWh/g O_3[101–103] (Table 2.1). The amount of stainless steel needed for the generation unit was based on the ozone dose and commercially available unit sizes.[104,105]

2.2.6 Uncertainty and Sensitivity Assessments

The impact of 31 major assumptions (Table 2.1 and Table A11), which accounted for chemical and filter design variables (uncertainty parameters), was assessed using a Monte Carlo analysis for the enhanced coagulation treatment goal. Each parameter’s range of values were assigned a distribution that matched the available data; i.e., triangular distributions were used if the maximum, minimum, and medium values were known, and uniform distributions were used if only maximum and minimum values were known. Uncertainty results were the output of 10,000 Monte Carlo simulations (i.e., trials), which was sufficient to generate reproducible results, and the spreadsheet model was simulated using Crystal Ball™ software. The conventional filtration, nonozonated biofiltration, and ozonated biofiltration results were evaluated on a trial basis. The sensitivity of each TRACI category to an uncertainty parameter was determined from the Monte Carlo simulation data by evaluating the resulting Spearman's rank correlation coefficients; a category was defined as highly sensitive to an uncertainty parameter if the corresponding correlation coefficient's magnitude was greater than 0.8 ($|\rho| > 0.8$) and defined as moderately sensitive if the magnitude was between 0.3 to 0.8 ($0.3 < |\rho| < 0.8$). Sensitive design variables were also assessed for all four treatment goals over 20,000 Latin Hypercube generated source water qualities.

Table 2.1. The low (L), high (H), and typical (T) values for 9 of the 31 uncertainty parameters (all parameters are in Table A11). Uncertainty parameters 1 to 8 used a triangular distribution with the known L, H, and T values; the rest had uniform distributions based on the known L and H values.

#	Uncertainty Parameter	Low Value	High Value	Typical Value	Basis and Citations
1	Minimum allowable alum dose (mg alum/L)	6.0	17	10	L=25th percentile value[95], H=75th percentile value[95], T=median[95]
2	Source water BDOC/TOC ratio	14%	27%	20%	L=min[85], H=max[85], T=median[85]
3	Ozonated water BDOC/TOC ratio	20%	38%	30%	L=min[85], H=max[85], T=median[85]
4	Biodegradable fraction of TOC removed by coagulation, when source water SUVA<3 L/mg/m	2.0%	5.0%	4.0%	L=25th percentile[52,72], H=75th percentile[52,72], T=average[52,72]
5	Biodegradable fraction of TOC removed by coagulation, when source water SUVA>3 L/mg/m	7.5%	14%	9.0%	L=25th percentile[52,72], H=75th percentile[52,72], T=average[52,72]
6	Nonozonated biofilter percent TOC removal (of the available biodegradable fraction of TOC) for 10°C to 20°C	5.0%	22%	10%	L=min[85], H=max[85], T=median[85]
7	Ozonated biofilter TOC removal (of the available biodegradable fraction of TOC) for 10°C to 20°C	3.0%	47%	13%	L=min[85], H=max[85], T=median[85]
8	Pre-ozonation dose (g O_3/g TOC)	0.25	1.6	0.50	L=min[85], H=max[85], T=median[85]
9	Air-fed ozone specific energy use (kWh/g O_3 generated)	0.018	0.022	0.020	L[101], H[102,103], T=estimated as average of L&H

2.3 Results and Discussion

2.3.1 Typical Source Water Analysis

When a typical source water was treated to meet the enhanced coagulation scenario's requirements, nonozonated biofiltration had the best environmental performance while ozonated biofiltration had the worst (Figure 2.3). Specifically, across all 10 TRACI environmental impact categories, nonozonated biofiltration had the same as or up to 4% lower impacts than conventional filtration. However, when considering the full range of possible values for all 31 uncertainty parameters (Table 2.1 and Table A11), the uncertainty results show that nonozonated biofiltration had up to 30% lower impacts than conventional filtration (Figure 2.3). The uncertainty analysis showed the potential of nonozonated biofiltration to further reduce environmental impacts because the current typical design values for biofiltration are conservative, as expected since biofiltration is not as wide-spread and data, especially for nonozonated biofiltration, is limited.

The nonozonated biofiltration results were not highly sensitive to any of the 31 uncertainty parameters, but some were moderately sensitive (with $-0.30>\rho>-0.43$) to three parameters: minimum allowable alum dose, source water BDOC/TOC ratio, and nonozonated biofilter percent TOC (as BDOC) removal. Sensitivity to the minimum allowable alum dose helped to show that the benefits of a certain technology are not solely based on TOC removal, specifically, that there are limits to the amount of alum that can be reduced. Since both TOC and turbidity need to be at least partially removed by coagulation, there is a lower limit to the alum dose. Even if biofiltration can remove a majority of the TOC, and therefore minimize the amount of alum needed to coagulate TOC, extra alum may be required to assure that effluent turbidity requirements are also met. Sensitivity to the source water's BDOC/TOC ratio also helped to show limits to biofiltration's performance since only the biodegradable fraction of a source water's TOC, remaining after coagulation, can be removed biologically. Every source water is different, which is why the available BDOC in each must be considered to determine the biofilter's capabilities. Sensitivity to the third parameter, the nonozonated biofiltration percent TOC (as BDOC) removal, highlights that a biofilter's performance will vary based on site specific conditions, such as filter operation and source water characteristics. Overall, all three uncertainty parameters had large impacts on the nonozonated biofilter's required alum dose, and the production of alum had one of the largest contributions to environmental impacts (Figure A7).

Chemical production, specifically of alum and caustic, had the largest contribution to impacts for all three filtration alternatives (hashed portion of bars in Figure 2.3). Alum and caustic accounted for about 55% to 80% of the impacts, in any given category, for both conventional filtration and nonozonated biofiltration. Both had similar doses; conventional filtration required 13 mg/L alum and 12.8 mg/L caustic, and nonozonated biofiltration required 12 mg/L alum and 12.2 mg/L caustic (Table 2.2). The alum and caustic doses for each treatment train were linked because coagulant addition effected pH and alkalinity, and these water quality parameters influenced the amount of caustic needed. Other studies have also found that the operational impacts of drinking water treatment were the most significant contributors to environmental impacts.[12,16,18–20,34,78] In particular, coagulant and pH adjustment chemicals were one of the most significant sources of environmental impacts.[17,18,22,26,31,35]

Table 2.2. Summary of chemical doses and biofiltration performance for all four treatment scenarios under typical source water conditions (3.2 mg/L TOC, 7.6 pH, 3.1 L/mg/m SUVA, 77 mg/L $CaCO_3$, and 15˚C). See Table A1 for a more detailed inventory of the enhanced coagulation treatment scenario.

		Conventional		**Nonozonated Biofiltration**			**Ozonated Biofiltration**			
Treatment Scenario	**TOC Reduction**	Alum Dose (mg/L)	Caustic Dose (mg/L)	Alum Dose (mg/L)	Biofiltration TOC Removal	Caustic Dose (mg/L)	Alum Dose (mg/L)	Biofiltration TOC Removal	Ozone Dose (mg O_3/mg TOC)	Caustic Dose (mg/L)
30% TOC Removal	30%	23	20.0	15	10%	14.1	13	13%	0.5	12.8
40% TOC Removal	40%	33	26.0	23	10%	20.0	21	13%	0.5	19.0
50% TOC Removal	50%	48	36.0	33	10%	26.0	30	13%	0.5	24.0
Enhanced Coagulation	one of the following: 25% TOC (Table A1); 2 mg/L effluent TOC; or 2 L/mg/m effluent SUVA	13 *(meets goal due to: effluent SUVA)*	12.8	12 *(meets goal due to: 25% TOC reduction)*	10%	12.2	10 *(meets goal due to: 25% TOC reduction)*	13%	0.5	10.8

In addition to operational impacts from chemical production, filter operational energy also had a large contribution to overall impacts. Across all categories, the filter energy accounted for up to 24% of impacts for conventional filtration and nonozonated biofiltration and up to 16% for ozonated biofiltration (Figure A7). The largest contribution to environmental impacts from infrastructure was due to steel production, for the filter housing, which contributed up to 27% of impacts across all categories for all three treatment trains (Figure A7). This is consistent with other studies that found that steel dominated infrastructure-specific impacts.[27,31,35,78,91] The environmental impact contribution from the production of filter media (i.e., anthracite and sand), concrete for the disinfection contact zone, and steel for the contact zone baffles were found to be negligible (less than 13% contribution in any impact category).

Since chemicals were a dominant source of impacts, substitute chemicals were considered for coagulation (ferric chloride instead of alum), pH adjustment (lime instead of caustic), and disinfection (UV instead of chlorine). The trends of relative environmental impacts between the three filtration alternatives stayed the same when ferric chloride was used instead of alum (Figure A8). However, there were several small changes; for all three treatment trains, the impacts in four categories (smog, acidification, carcinogenics, and fossil fuel depletion) were reduced by 12% to 40%; the impacts in five categories (global warming, eutrophication, non carcinogenics, respiratory effects, ecotoxicity) had similar impacts with less than a 10% change; and in one category (ozone depletion), impacts increased by up to 16%. There may be some potential to reduce environmental impacts associated with coagulation by using polymers or coagulant aids, but data on those chemicals is too limited to be able to assess their environmental impacts. When lime was used instead of caustic, the relative environmental impact trends still did not change between the filtration alternatives (Figure A9) even though there was a total absolute reduction in impacts for all alternatives. When UV was used instead of chlorine, the change in impacts was minor, with less than a 15% change, up or down, when compared to using chlorine (Figure A6). Since disinfection was found to have minor contribution to overall impacts, other disinfectants were not considered. On an absolute basis, utilities can use alternative chemicals such as ferric chloride and lime to reduce environmental impacts. Overall, though, changing the different chemicals didn’t change any of the results’ relative trends. Reducing chemical use through biofiltration was an effective approach to reduce relative impacts and further reduce absolute environmental impacts of conventional filtration by using biological activity instead of chemicals to remove BDOC.

For ozonated biofiltration, though, there is a trade-off between chemicals. The use of ozone can increase the amount of organic matter removed by a biofilter in order to reduce the amount of coagulant required to achieve the desired TOC removal. However, ozonated biofiltration had the same as or up to 145% more impacts than conventional filtration in all impact categories (Figure 2.3). When considering the full range of possible values for all uncertainty parameters, ozonated biofiltration's impacts were, depending on the category, as low as 79% or as high as 282% of the conventional filtration's impacts (Figure 2.3). The larger environmental impacts were because the environmental impacts associated with ozone energy outweighed the benefits associated with reducing the alum dose (Figure 2.4). The results were moderately sensitive to the minimum allowable alum dose ($\rho \leq 0.59$) and highly sensitivity to the ozone dose requirement for pre-ozonation ($\rho > 0.82$). The ozonated biofiltration results' uncertainty ranges were large because the possible values for ozone generation energy had a large range of 14 to 111 kWh/m^3, which was similar to the range of 20 to 150 kWh/m^3 reported in the literature for pre-ozonation energy requirements,[15,22,34,106] was calculated from ozone dose and specific energy values reported in the literature (Table 2.1).

Overall, adding pre-ozonation did not improve the environmental performance of biofiltration (Figure 2.4). Pre-ozonation resulted in an alum dose that was 2 mg/L lower than the nonozonated biofiltration's dose, and this reduction led to a reduction in environmental impacts, specifically by 0.005 kg CO_2 eq./m^3 for the global warming impact category. However, the additional energy for ozone generation increased environmental impacts so much that it offset the benefits of the alum reduction; specifically, ozone generation energy increased global warming impacts by 0.09 kg CO_2 eq./m^3. While these results are for a typical source water, this trend of ozonated biofiltration having larger environmental impacts than nonozonated biofiltration was found to be the same regardless of source water quality (Figure A10). Other decision criteria must also be considered when choosing to pre-ozonate or not. For instance, using ozone could increase the biodegradability of the organic carbon in the source water so much that even the biofilter effluent will have relatively high BDOC levels, which can decrease biological stability in distribution system, and some of the remaining BDOC could include ozonated by-products, which are regulated and unregulated DBPs. On the other hand, some other benefits to pre-ozonation include improved control of taste and odor causing compounds, anthropogenic trace organic compounds, and inorganic compounds. These other objectives should be carefully considered with site-specific conditions.

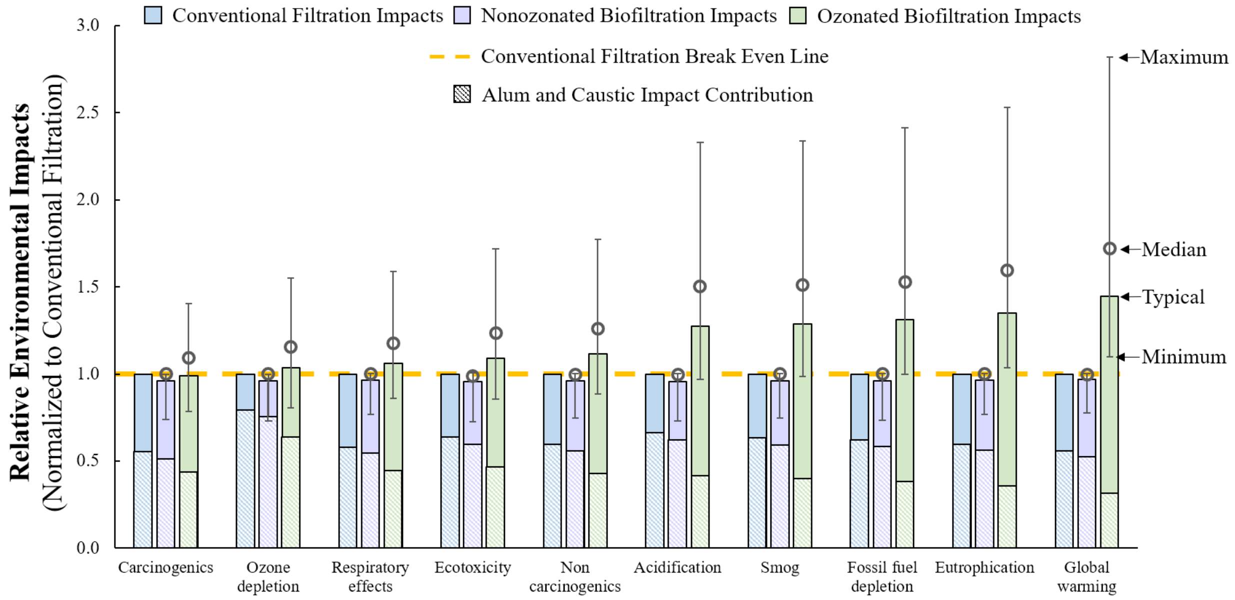

Figure 2.3. Compared to conventional filtration (blue column), the relative environmental impacts of nonozonated (purple column) and ozonated (green column) biofiltration when a typical source water (3.2 mg/L TOC, 7.6 pH, 3.1 L/mg/m SUVA, 77 mg/L $CaCO_3$, and 15˚C) was treated under the enhanced coagulation treatment scenario and typical uncertainty parameter values were used. The results' uncertainty ranges, from the Monte Carlo analysis, are shown as error bars (minimum and maximum) and the open circles represent median values. Values below the dashed line indicate better environmental performance than conventional filtration.

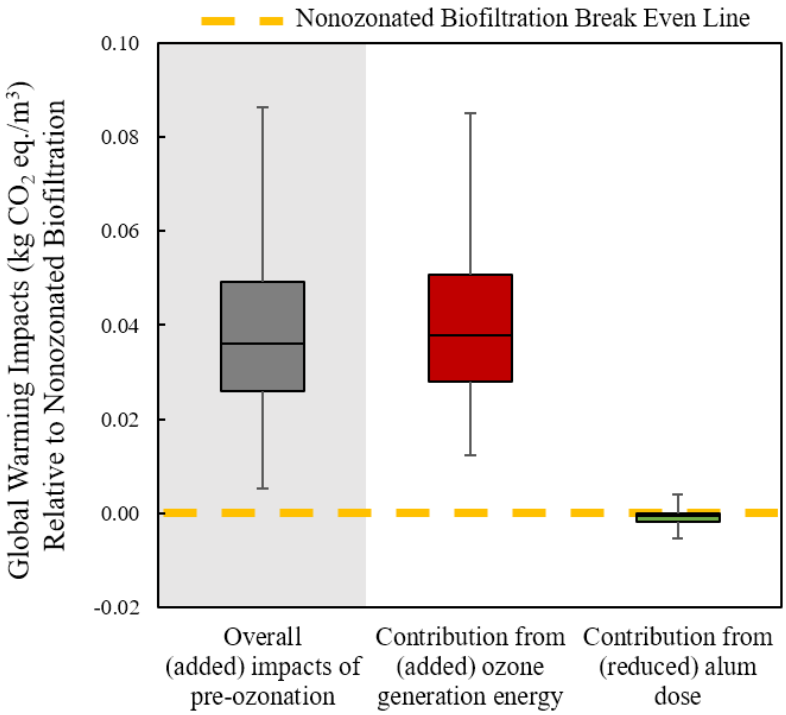

Figure 2.4. Global warming impact (kg CO_2 eq./m^3) uncertainty ranges for ozonated biofiltration compared to nonozonated biofiltration (values below the dashed line indicate that ozonated biofiltration had better environmental performance than nonozonated biofiltration) when a typical source water (3.2 mg/L TOC, 7.6 pH, 3.1 L/mg/m SUVA, 77 mg/L $CaCO_3$, and 15˚C) was treated under the enhanced coagulation treatment scenario. The global warming category was most representative of all of the 10 TRACI categories (Figure A7).

2.3.2 Comprehensive Source Waters Analysis

While nonozonated biofiltration had better environmental performance than the other two filtration alternatives, when a typical source water was treated under the enhanced coagulation treatment scenario (Figure 2.3), the relative environmental performance between it and conventional filtration changed based on source water quality and treatment scenario (Figure 2.5). Environmental impact trends for the three constant percent TOC removal scenarios were similar to each other, with the trends becoming more pronounced with increasing removal requirements (Figure 2.5). When comparing the 30% to 50%, the 50% TOC removal scenario required higher chemical doses, which resulted in larger benefits of nonozonated biofiltration relative to conventional filtration. This increase of 20% in TOC removal had an 8% increase in the number of source water samples that had lower environmental impacts when treated with nonozonated biofiltration (i.e., the global warming impacts of nonozonated biofiltration were 25% less than conventional filtration's). Generally, with a constant percent TOC removal requirement, nonozonated biofiltration had the best environmental performance (i.e., lower environmental impacts than conventional filtration) for source waters at higher temperatures (Figure A11) and with higher alkalinity, higher pH, and lower SUVA (Figure 2.5) (Table 2.3).

For these treatment scenarios, there were environmental impact trends associated with the source water's SUVA. The environmental benefits of nonozonated biofiltration were most pronounced at SUVA values below 2.75 L/mg/m, because at lower SUVA values TOC is more difficult to coagulate,[107] and conventional filtration only relied on coagulation chemical use for TOC removal. These results show that there is an environmental benefit associated with the experimentally determined SUVA trends. There was not a major trend when looking at source water TOC in isolation. Even when evaluating a range of 5 to 8 mg/L TOC, which is outside of experimentally validated biofiltration performance, results did not change significantly (Figure A12).

The relative environmental impact trend associated with alkalinity, better nonozonated biofiltration environmental performance with higher source water alkalinity, was mostly based on water chemistry interactions that relied on buffering capacity. For low alkalinities, between 20 and 50 mg $CaCO_3$/L, the conventional filtration alum dose was so high for the 50% TOC removal scenario that most of the alkalinity was consumed; since the caustic dose is a function of both pH and alkalinity, when all alkalinity was consumed, only a relatively small amount of caustic was needed to adjust the final water to the target pH. At low alkalinities, the lowered buffering capacity allowed a lower caustic dose despite the larger change in pH needed. On the other hand, the nonozonated biofiltration alum dose was not high enough to consume all alkalinity, so the caustic dose was up to 5,900% higher (47 mg/L caustic compared to 0.8 mg/L for conventional filtration) in the most extreme case, to account for the remaining buffering capacity when adjusting the final water pH (Figure A13). Therefore, conventional filtration had better environmental performance for most source waters with low alkalinity (Figure 2.5). At higher alkalinity, the caustic doses were more similar, with biofiltration usually having much lower doses.

For those source waters with higher buffering capacity, 50 to 125 mg $CaCO_3$/L, the use of nonozonated biofiltration could reduce the caustic dose by 50%. Nonozonated biofiltration was also able to meet the lower percent TOC removal requirements mostly through biodegradation and so had considerably smaller alum doses (e.g., 45 mg/L compared to 80 mg/L for conventional filtration). Therefore, nonozonated biofiltration had lower alum and lower caustic doses, which resulted in up to a 40% reduction of environmental impacts, compared to conventional filtration.

For low SUVA source waters, the largest difference in environmental performance was seen at high pH coupled with high alkalinity and at low pH coupled with low alkalinity. At low source water pH (<6.5) and alkalinity, alum doses for nonozonated biofiltration and conventional filtration were similar enough that the caustic dose needed to bring the pH back up to 8.2 was similar between the filtration alternatives and provided diminishing returns of nonozonated biofiltration. Therefore, conventional filtration had up to 54% reduction in impacts compared to nonozonated biofiltration for the 50% TOC removal scenario. At high source water pH (>7) and alkalinity, coagulation was more difficult to achieve, so higher alum and caustic doses were needed for conventional filtration compared to nonozonated biofiltration. Nonozonated biofiltration had up to 36% reduction in impacts compared to conventional filtration for the 50% TOC removal scenario.

The caustic dose required to bring the final water pH up to 8.2 had a large contribution to the relative environmental impacts for each filtration alternative. Therefore, two other pH adjustment goals were also considered: (i) no final water pH adjustment, and (ii) final water pH adjusted back to the source water pH value. For the three pH adjustment approaches, all environmental impact trends were similar (Figure A14). The only slight difference was that the trends based on pH were slightly adjusted, such that conventional filtration had better environmental performance with high pH source waters and nonozonated biofiltration had better higher environmental performance with low pH source waters. This is because the final water pH change needed for the two additional pH adjustment goals was smaller than for the adjustment to a final pH of 8.2, therefore smaller caustic doses were required for every source water; for example, no caustic for option (i) and 9.2 mg/L for option (ii), compared to 12.8 mg/L needed for a final pH of 8.2 for a typical source water. When the final pH was adjusted to match the source water's pH there were fewer diminishing relative returns of pH adjustment chemical reduction (for adjustment to 8.2) and coagulation chemical reduction (no pH adjustment), which is why this pH adjustment option had the lowest nonozonated biofiltration relative impacts compared to conventional filtration. There are many possible variations for coagulation (e.g., polymer addition as a flocculation aid, pH adjustment before coagulation), which should be evaluated further to better understand the impact of utility specific operations.

Conventional filtration was found to be environmentally preferred to nonozonated biofiltration both on a relative (Figure 2.5) and absolute (Figure 2.6) basis, and vice versa, depending on source water quality. When considering absolute global warming impact differences (Figure 2.6), nonozonated biofiltration's better environmental performance was most influenced by a source water with high alkalinity and low SUVA; specifically, nonozonated biofiltration could achieve up to a 0.06 kg CO_2 eq./m^3 reduction of global warming impacts, compared to conventional filtration, when the source water's alkalinity was between 50 to 125 mg $CaCO_3$/L and SUVA was below 2.75 L/mg/m (Figure 2.6). For conventional filtration, it could have 0.02 kg CO_2 eq./m^3 fewer annual global warming impacts than nonozonated biofiltration when the source water had a low alkalinity as well as low SUVA and low pH values. Overall, while the source water quality greatly influences the comparison of filtration options, the average source water quality advantages biofiltration (Table 2.3).

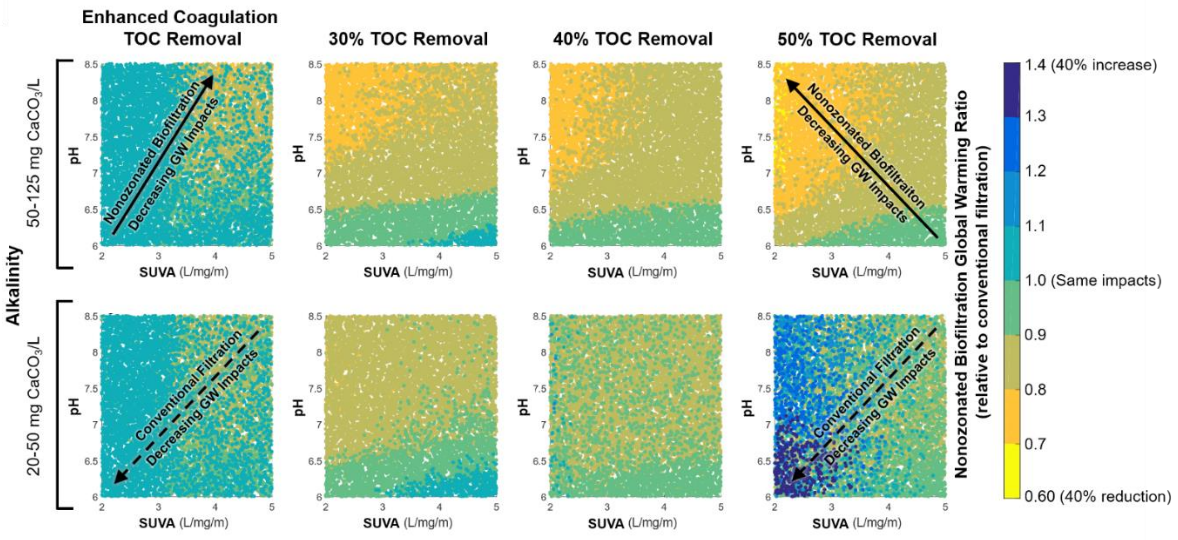

Figure 2.5. Global warming impact of nonozonated biofiltration, relative to conventional filtration, for the treatment of 20,000 unique source waters under four different treatment scenarios (columns): enhanced coagulation (15 to 50% TOC removal), 30% TOC removal, 40% TOC removal, and 50% TOC removal. Each source water had a 15°C temperature and a TOC between 2 and 5 mg/L, pH between 6 and 8.5, SUVA between 2 and 5 L/mg/m, and alkalinity between 20 and 125 mg/L CaCO$_3$ (each alkalinity bin includes half of the source waters). Results are for the typical values for each uncertainty parameter since none of the conventional filtration and nonozonated biofiltration typical source water results were highly sensitive to any uncertainty parameter.

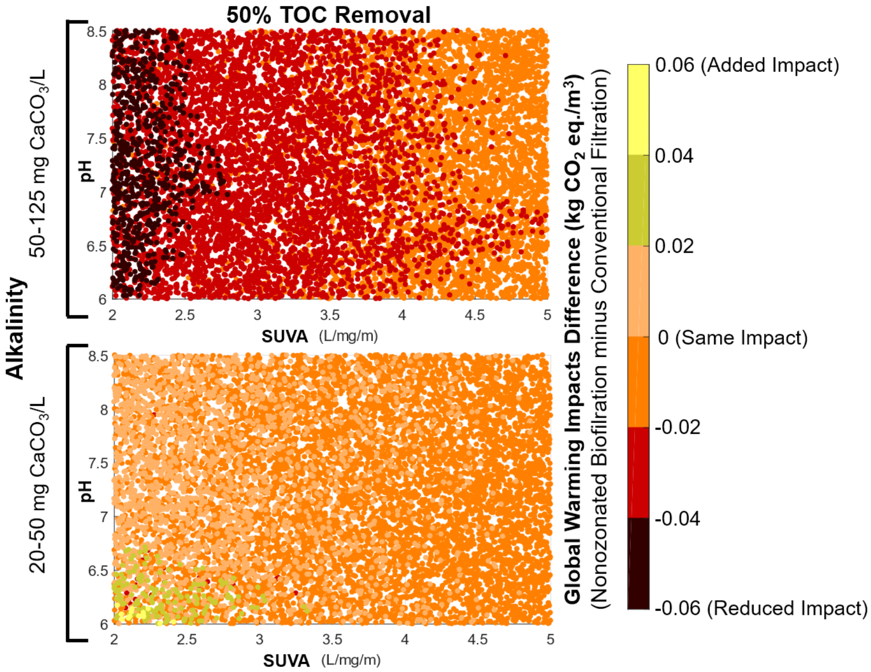

Figure 2.6. Absolute differences in global warming impacts (kg CO_2 eq./m^3) of nonozonated biofiltration, relative to conventional filtration, for 20,000 unique source waters at a constant 50% TOC removal scenario. Each source water had a 15°C temperature and a TOC between 2 and 5 mg/L, pH between 6 and 8.5, SUVA between 2 and 5 L/mg/m, and alkalinity between 20 and 125 mg/L $CaCO_3$ (each alkalinity bin includes half of the source waters). Results are for the typical values for each uncertainty parameter since none of the conventional filtration and nonozonated biofiltration typical source water results were highly sensitive to any uncertainty parameter.

Table 2.3. Summary of source water quality results of 20,000 Latin Hypercube trials for nonozonated biofiltration and conventional filtration. Average conventional treatment source water parameters of 271 drinking water utilities are also presented for context.[89]

Source Water Parameter	Source Water Range Considered	Average Conventional Source Water	Nonozonated Biofiltration Best Performance*	Conventional Filtration Best Performance*
TOC (mg/L)	2 – 5	3.2	N/A	N/A
Alkalinity (mg $CaCO_3$/L)	20 – 125	77	50-125	20 – 50
SUVA (L/mg/m)	2 – 5	3.1	< 2.75	< 3.5
pH	6 – 8.5	7.6	> 7	< 7
Temperature (°C)	<10, 10 – 20, >20	15	Higher	Lower
TOC Goal	4 Scenarios	Varies by Country	50% TOC Removal	Enhanced Coagulation

* Parameters should be considered in combination to find the best environmental performance for each filtration alternative compared to the other filtration alternatives.

Since the typical source water analysis showed that the nonozonated biofiltration results were moderately sensitive to three uncertainty parameters, a one-at-time sensitivity analysis was used to evaluate each. The results did not change when using the maximum or minimum value for the minimum allowable alum dose (Figure A15), or when the maximum BDOC/TOC ratio was used (Table 2.1) (Figure A16). Biofiltration had about a 10% relative increase of nonozonated biofiltration environmental impacts compared to conventional filtration when BDOC/TOC was below 19%, for source waters with SUVA >3 L/mg/m (Figure A16), because so little BDOC was available for biodegradation that the TOC removal by biodegradation could not be fully realized. This was also seen at higher temperatures (Figure A11) due to higher TOC expected removals,[85] and overall, the environmental benefits of biofiltration decreased with decreasing temperatures (Figure A11). Nonozonated biofiltration percent TOC removal performance was also variable within temperature ranges. The median TOC removal performance in a biofilter operating between 10°C and 20°C is 10%, but the full range is 5% to 22%; this large range is mainly due to the range of empty bed contact times (EBCTs) used in 21 different experiments.[85] Nonozonated biofiltration’s percent TOC removal results for a high removal of 22% had the same trends as increasing temperatures (Figure A17). However, experimental data has shown that the biomass acclimate to new conditions rapidly, so biofiltration TOC removal can be kept consistent, despite seasonal temperature swings, by adjusting the EBCT. Increasing the EBCT may increase environmental impacts, though; specifically, the impacts associated with the production of the filter media and steel housing would have the relative impacts of nonozonated biofiltration increase up to 18% for treatment of a typical source water under the enhanced coagulation treatment target. Regardless, utilities that are using biofiltration should consistently measure the DOC removed by the filter to inform the amount of coagulant that is needed to meet TOC removal goals.

In contrast to the constant percent TOC removal treatment scenarios, the enhanced coagulation scenario’s treatment requirements were based on source water treatability and could be met with either percent TOC removal or absolute effluent values for TOC or SUVA. This resulted in more similar environmental impacts between conventional filtration and nonozonated biofiltration for most source water qualities (Figure 2.5). The enhanced coagulation scenario had sharp transitions between equivalent impacts and better environmental performance of nonozonated biofiltration. The sudden transitions to equivalent impacts occurred because the minimum allowable alum dose was enough to meet the treatment requirements of the enhanced coagulation scenario, specifically to meet the absolute effluent values for TOC and SUVA. One transition was seen at source water SUVA values around 3.5 L/mg/m SUVA. There were equivalent environmental impacts for source waters with a low SUVA because the alum dose was similar between the filtration alternatives as it was based on meeting the treatment requirement of an effluent SUVA of 2 L/mg/m; since these waters are harder to coagulate, this treatment approach allows for a lower alum dose to be used. The dose for the SUVA criteria was much lower than for TOC removal or for the effluent TOC criteria, so both alternatives had the same alum dose, as biofiltration was not modeled to reduce SUVA. There was another transition around a source water pH of 6.5. For lower pH waters, which are easier to coagulate, the alum dose was based on the absolute effluent TOC goal, which resulted in equivalent impacts between conventional filtration and nonozonated biofiltration. This sensitivity to the minimum alum dose meant that impacts between conventional filtration and nonozonated biofiltration became more similar with increasing minimum alum doses (Figure A18). When using the enhanced coagulation treatment target, this dose needs to be determined experimentally for each source water to more accurately determine the relative environmental impacts because no current models can accurately predict the alum dose required for turbidity.

The enhanced coagulation scenario was based on U.S. regulations, which were intentionally designed to allow for lower alum doses when the source water was hard to coagulate, regardless of technology, so that the requirements would be economically efficient.[108] When the treatment scenario was enhanced coagulation, alum doses were lower, and more feasible to meet, so conventional filtration and nonozonated biofiltration had similar environmental impacts for a majority of source waters. While a constant TOC removal treatment goal showed the potential environmental benefits of nonozonated biofiltration, this treatment approach could have diminishing returns, with alum doses that increase much faster than the improvements in water quality.

The decision to transition from conventional filtration to nonozonated biofiltration, by moving the chlorine addition to after the filter, requires several important considerations, including cost, water quality, and operational complexity. In general, nonozonated biofiltration had better environmental performance than conventional filtration when the source water quality was more difficult to treat. If water quality continues to degrade, which is expected due to the use of alternative source waters and climate change,[5–8] then it will be even more important to consider biofiltration. However, if treatment requirements account for source water treatability, then nonozonated biofiltration and conventional filtration will have similar environmental performances. When these filtration alternatives have similar impacts, conventional filtration may be more advantageous because biofiltration is currently less common and is expected to have additional operator training, increased operational complexity, lack of technology specific regulations, and decreased reliability. On the other hand, many full-scale and pilot-scale studies have shown multiple drinking water quality advantages of biofiltration, including the ability to reduce distribution maintenance, improve the control of DBP precursors and DBP formation, and remove organic contaminants.

2.4 Conclusions

The environmental impacts of conventional filtration, nonozonated biofiltration, and ozonated biofiltration were compared on a life cycle basis with the treatment goals of TOC, pathogen, and turbidity reduction. Results indicated that nonozonated biofiltration outperformed or equaled the impacts associated with ozonated biofiltration in all 10 impact categories for all source waters. Nonozonated biofiltration was also shown to perform equivalently or better than conventional filtration under the enhanced coagulation TOC removal scenario for the typical source water (3.2 mg/L TOC, 7.6 pH, 3.1 L/mg/m SUVA, 77 mg/L $CaCO_3$, and 15˚C); this was still true when considering design uncertainty. The environmental impact trends for the filtration alternatives were primarily due to the production of coagulation and pH adjustment chemicals.

When comparing conventional filtration and biofiltration over 60,000 unique source waters and 4 treatment scenarios, results were dependent on the TOC removal scenario (enhanced coagulation, 30%, 40%, and 50% TOC removal) and source water quality. Generally, nonozonated biofiltration had less environmental impacts than conventional filtration for the majority of the source waters evaluated, with the greatest offset of environmental impacts seen under the 50% TOC removal scenario and source waters with high alkalinity (50-125 mg CaCO3/L), low SUVA (less than 2.75 L/mg/m), high pH (higher than 7), and higher temperature (increasing from less than 10°C to above 20°C). Nonozonated biofiltration had worse environmental performance than conventional filtration at lower alkalinities (20 to 50 mg $CaCO_3$/L), where water chemistry relationships advantaged conventional filtration for pH adjustment chemical use. Nonozonated biofiltration can be a more environmentally sustainable option for numerous source water qualities and treatment scenarios when it can significantly reduce chemical use. These environmental impact results should be considered in unison with other water treatment utility goals when deciding whether to use conventional filtration, nonozonated biofiltration, or ozonated biofiltration.

Chapter 3. Life Cycle Environmental Impacts of Disinfection Technologies Used in Small Drinking Water Systems

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Abstract

Small drinking water systems serve a fifth of the U.S. population and rely heavily on disinfection. While chlorine disinfection is common, there is interest in minimizing chemical addition, especially due to carcinogenic disinfection byproducts and chlorine-resistant pathogens, by using ultraviolet technologies; however, the relative, broader environmental impacts of these technologies are not well established, especially in the context of small (<10,000 people) water systems. The objective of this study was to identify environmental trade-offs between chlorine and ultraviolet disinfection via comparative life cycle assessment. The functional unit was the production of 1 m^3 of drinking water to U.S. standards. Treatment included cartridge filtration followed by either chlorine disinfection or ultraviolet disinfection with chlorine residual addition. Environmental performance was evaluated for various chlorine contact zone materials (plastic, concrete, steel), ultraviolet validation factors (1.2 to 4.4), and electricity sources (renewable; U.S. average, high, and low impact grids). Performance was also evaluated when filtration and chlorine residual were not required. From an LCA perspective, replacing chlorine with UV was preferred only in a limited number of cases (i.e., high pumping pressure but filtration is not required). In all others, chlorine was environmentally preferred, although some contact zone materials and energy sources had an impact on the comparison. Utilities can use these data to inform their disinfection technology selection and operation to minimize environmental and human health impacts.

3.1 Introduction

Small drinking water systems account for about 90% of United States (U.S.) systems and serve about 53 million people (20% of the population).[3] Due to size and lack of resources, small systems struggle more to meet regulations (e.g., maximum contaminant limits, monitoring and reporting) than larger utilities.[3] Small systems rely heavily on disinfection.[3] Due to concerns about disinfection byproducts (DBPs) and chlorine-resistant pathogens, alternatives to chlorine-intensive disinfection have drawn interest; however, most small systems have not been able to comprehensively evaluate alternatives, which limits their ability to consistently achieve current regulations, adapt to future regulations, and improve water quality.

Chlorine disinfection and residuals have significantly reduced the risk of drinking water associated illnesses[109] and prevented biological growth in distribution systems.[110,111] Many small systems already have infrastructure and training related to chlorine disinfection, with liquid sodium hypochlorite being a common chlorine source. For consumers, in addition to chlorine taste and odor issues,[40] there are concerns about health risks from DBPs, which are formed when chlorine is exposed to organic matter during water treatment.[112,113] The U.S. Environmental Protection Agency (EPA) has identified DBPs as carcinogenic and primary drinking water pollutants.[113,114]

UV can reduce DBP formation by reducing chlorine doses.[115] Low pressure UV (LPUV) lamp systems are some of the most commonly used for drinking water treatment.[116] While UV disinfection can be highly automated, it still has the perception of increased workloads, especially associated with training, operation, and maintenance. UV is most commonly used for disinfection and is effective at inactivating *Giardia* and *Cryptosporidium* (*Crypto*) at very low doses.[116,117] For virus inactivation, though, LPUV lamps require very high doses.[118] Validation factors are used to adjust the regulated dose for operation and ensure that the regulated UV dose is met; they account for any uncertainties inherent in the UV reactor validation process, such that even during fluctuating operational conditions, the regulated dose is being met by the UV system at all times. Energy and dose are related, with higher doses requiring more energy. High UV doses can also be used to degrade contaminants that chlorine cannot, such as antibiotics,[119] THMs,[115] NDMA,[120,121] and TCE.[122] Despite these capabilities, UV must still be coupled with chlorine to provide a distribution system residual.

Given the complicated tradeoffs between these disinfection options, including direct and indirect risks, there is a need for a comprehensive, quantitative comparison. Multicriteria decision making assessment methodologies have been used to evaluate[48] and identify[123] different sustainability objectives for water treatment, and life cycle assessment (LCA) has been used to elucidate environmentally sustainable drinking water treatment practices. Including comparing conventional and newer treatment processes,[17,18] and operational decisions for treatment plants.[34,35] However, literature thus far has focused on UV applications other than drinking water primary disinfection. There are studies that assessed wastewater disinfection,[124–127] but this uses different treatment targets and UV reactor systems than drinking water. There are multiple drinking water studies that looked at life cycle impacts of entire treatment trains for large utilities; for example, previous work has evaluated UV applications during conventional treatment processes modeled for a large (5 Mm^3/yr) UV system compared to gaseous chlorine,[33] but most of those processes are not common at small systems. So far, there is a limited small system focus and direct comparison of UV or chlorine disinfection in the literature.

Therefore, the objective of this study is to compare chlorine and UV primary disinfection using LCA methodology in order to inform the design and operation of these technologies and water systems. The functional unit was the treatment of a specified source water to meet U.S. drinking water regulations using typical small system treatment processes. Three distinct water quality and regulatory scenarios were evaluated (Figure 3.1): (i) typical source water scenario, which represents systems that use filtration and a chlorine residual; (ii) chlorine residual exemption scenario, which represents a residual regulatory exemption, possible for various small systems throughout the U.S.;[128] and (iii) filtration exemption scenario, which represents an unfiltered system allowance option for the multiple small systems that could only disinfect their source water.[3] The life cycle environmental impacts associated with the material, chemical, and energy use of each small system treatment train were quantified for each scenario. The analysis also included multiple types of chlorine contact zone materials, UV validation factors, local energy sources, and operational conditions. Overall, this study will provide the information needed for small drinking water system stakeholders to make design and operational decisions about disinfection technologies in order to minimize environmental impacts without compromising public health.

3.2 Methods

The production and use of chlorine or UV disinfection systems at small drinking water systems were evaluated using comparative LCA methodology following the ISO 14040 framework (Figure 3.2).[10] The end of life stage was not included because water treatment LCAs have found that the operational life cycle stage impacts were dominant compared to end of life impacts.[18,78,124] For example, UV lamp recycling is recommended and commonly available,[116,129] so negligible impacts from the mercury in used lamps were expected.[33] The functional unit was the treatment of 1 m^3 of a scenario-defined source water over 40 years to produce an effluent that met U.S. regulation; the reference flow of 273 m^3/day was used to represent a small system. The chlorine disinfection alternatives included different chlorine contact zones, including a concrete basin with steel, plastic, or no baffles; steel or plastic tank; and plastic pipe. The UV disinfection alternatives included different UV validation factors (from 1.2 to 4.4).

3.2.1 Scenarios

The three distinct drinking water treatment scenarios evaluated different regulatory considerations, which dictated the treatment processes required to meet the functional unit (Figure 3.); alternatives between scenarios are not comparable, but the alternatives within a scenario are comparable. The three scenarios represented typical source water treatment, water treatment when there is a chlorine residual regulatory exemption, and water treatment when the water has very low turbidity but a high risk of *Crypto* such that filtration may not be required. Regulations were based on the U.S. EPA's Surface Water Treatment Rule (SWTR)[130] and Long Term 2 Enhanced SWTR (LT2SWTR).[84] All final drinking water qualities were equivalent according to U.S. regulation; equivalence was based on a regulatory perspective since many small systems struggle to meet regulation, which dictates their technology selection and operational decisions, but these decisions should consider that water equivalent from a regulatory perspective is not necessarily equivalent from a risk perspective. Generally, treatment achieved the following log reductions: 2-log *Crypto*, 3-log *Giardia*, and 4-log virus. The source water (i) represented national average surface water quality: a pH of 7.5, 15°C temperature, 82% ultraviolet transmission (UVT_{254}), and 2.8 mg/L total organic carbon;[89] (ii) had less than 0.075 *Crypto* oocysts/L, which required the typical 2-log *Crypto* reduction, with one exception discussed for scenario 3 below; (iii) had less than 1 NTU turbidity[131] (a higher turbidity would add the same pre-filtration process to all alternatives, so it did not need to be evaluated); and (iv) a Crypto concentration less than 0.075 oocysts/mL, and greater than 0.01 oocysts/mL when unfiltered (Figure 3.1).[84]

Scenario 1, the typical source water scenario, evaluated the treatment of surface water using typical small system technologies, consisting of filtration followed by either chlorine disinfection or ultraviolet disinfection with chlorine residual addition (Figure 3.1 and Figure 3.2). It was assumed that a 1 mg/L free chlorine residual leaving the plant was needed to assure the minimum residual in a distribution system of 0.2 mg/L Cl_2.[132] For chlorine disinfection and residuals, the sodium hypochlorite dose was calculated using the U.S. EPA's water treatment plant model v2.[133] For UV disinfection, since the amount of free chlorine needed for the residual was more than that needed for virus inactivation, UV was not used for virus inactivation. Therefore, the LT2SWTR regulated UV dose for 1-log *Giardia*, 2.1 mJ/cm^2,[84] was used.

For scenario 2, the chlorine residual exemption scenario, the source water quality, regulations, and treatment trains were the same as scenario 1 except chlorine addition was excluded from the UV disinfection treatment train (Figure 3.1 and Figure 3.2). Therefore, the disinfection requirements of 1-log *Giardia* and 4-log virus inactivation were achieved solely by UV; the LT2SWTR regulated UV dose for 4-log virus inactivation, 186 mJ/cm^2, was used.[84] The chlorine disinfection treatment train was the same as in scenario 1; specifically, it kept a 1 mg/L residual to represent realistic chlorine dosing decisions and avoid unrealistically large contact times and basins. European drinking water guidelines do not require a chlorine residual,[134] and as a result, several countries do not use chlorine for secondary disinfection (e.g., Austria, Germany, Switzerland, and the Netherlands), which is also an option for some small systems in the U.S.[3]

Scenario 3, the filtration exemption scenario, evaluated source water that resulted in different regulatory requirements and treatment processes between chlorine and UV disinfection.[84,130,135] This scenario is exemplified by current operations at large water systems (e.g., New York City, Seattle). The source water was similar to scenario 1 except that it had a specified fecal coliform count (less than 20/100 mL).[135] Since chlorine *Crypto* inactivation requires very large doses and contact times,[136] the chlorine disinfection treatment train used filtration to achieve the required 2-

log *Crypto* removal, which consequently also achieves 2-log *Giardia* reduction;[84,137] the remaining 1-log *Giardia* and 4-log virus inactivation requirements and chlorine residual was achieved by chlorination. Since UV is effective at inactivating *Crypto* at relatively low doses,[116,117,138] the UV disinfection treatment train did not include filtration and used UV instead of filtration to achieve the *Crypto* log reduction. Therefore, the largest of the 3-log *Giardia* and 3-log *Crypto* LT2SWTR regulated UV doses, 12 mJ/cm^2,[84] was used. Chlorine was still used for virus inactivation and chlorine residual in this UV disinfection treatment train. The application of this unfiltered treatment train also requires six site-specific conditions (e.g., source water protection), as outlined by U.S. regulation.[135]

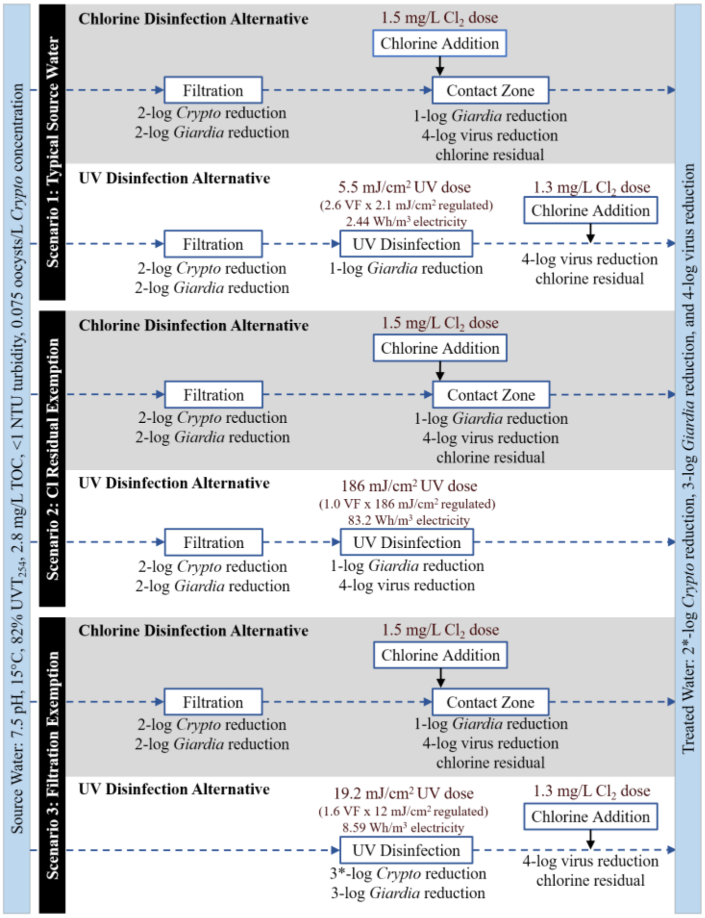


Figure 3.1. The log reduction credit achieved by each treatment process (filtration, chlorination, UV disinfection) within a treatment train to produce equivalent treated water quality, from a regulatory perspective, within all 3 scenarios. The source water quality parameters were based on national average values,[89] and the regulated log reduction requirements were based on U.S. regulation.[84,130,135] The actual UV dose was the product of the validation factor ("VF") and the minimum UV dose required by regulation ("regulated"). Giardia and virus reduction requirements are independent of initial concentrations; Crypto reduction requirements are also independent when the initial concentration is less than 0.075 oocysts/L, but regulation requires higher removal when a system is unfiltered*. The LCI unit processes considered for each disinfection alternative are shown in Figure 3.2. * For the treated water to be equivalent, from a regulatory perspective, between the treatment trains in Scenario 3, the unfiltered treatment train was required to achieve a 3-log instead of 2-log Crypto reduction.[84,116]

3.2.2 Life Cycle Inventory and Impact Methods

Full-scale designs for all treatment trains were developed and assessed in a spreadsheet application (Excel^{TM}). Inventory data associated with each unit process were acquired using data from the literature, manufacturers, and ecoinvent[81] life cycle inventory (LCI) unit processes (accessed via SimaPro® software) (Table A1). The amount of each material, chemical, and energy was calculated for 40 years of operation (Table A1 to Table A4). These calculations accounted for typical material lifetimes and replacements[79,80,116] and assumed U.S. average electricity emissions. The impacts of different energy sources and individual U.S. grids were also evaluated. Life cycle inventory emissions were converted into 10 impact categories by using characterization factors from the U.S. EPA's Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI)[83] (accessed via SimaPro® software) and were compiled into the spreadsheet-based LCA model. DBPs are currently outside of LCA scopes[33,83,139] and LCIA methods but should be considered in the decision process. Crystal Ball® software was used to simulate the spreadsheet-based LCA model during the Monte Carlo analysis.

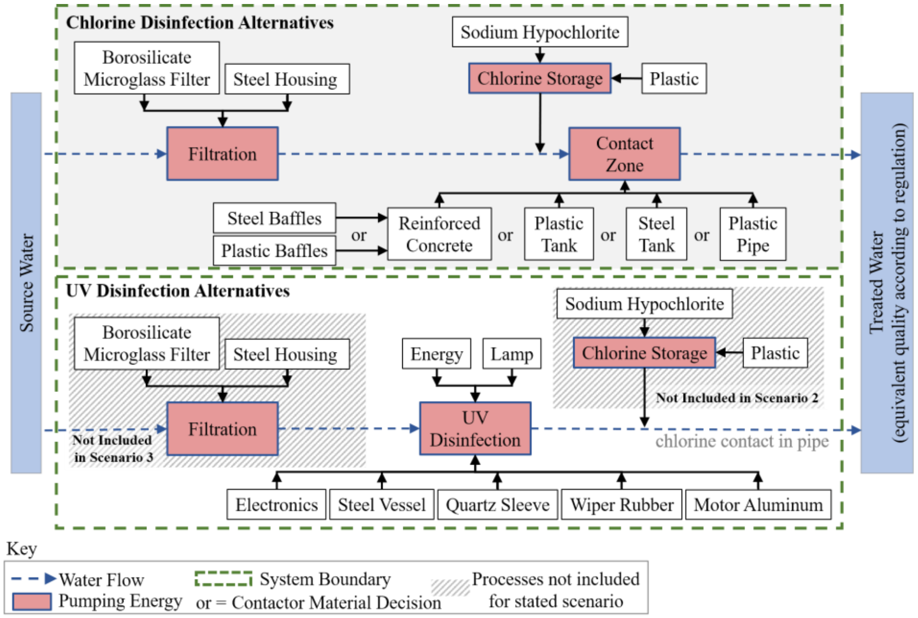


Figure 3.2. LCA system boundary and LCI unit processes included in the chlorine and UV disinfection treatment trains. The disinfection alternatives were compared under three scenarios (see Figure 3.1). For UV disinfection, all scenarios had the same treatment processes except scenario 2 did not use chlorine and scenario 3 did not use filtration. Chlorine and UV disinfection alternatives included different contact zones, which impacted materials, and UV validation factors, which impacted energy.

3.2.3 Filtration

Cartridge filtration was used to represent typical small systems[140] and included the LCI unit processes for production of stainless steel for the filter housing; borosilicate microglass for the filters; and energy for pumping. Manufacturer data was used to characterize the pumping energy, materials, and quantities of the micropore cartridge filtration systems.[141–147] The filter head loss was 3.5 meters water column (mWc) for all alternatives in Scenarios 1 and 2. The head loss in Scenario 3 only impacted the chlorine disinfection alternative and was evaluated between 0.7 mWc to 11 mWc to represent a wide range of location-specific conditions. Pump materials were expected to have negligible impacts,[18,33,34] so they were not included. More details in Appendix section B3.

3.2.4 Chlorine Disinfection

The chlorination process included the LCI unit processes for production of liquid sodium hypochlorite; polyethylene for the chlorine storage tank; energy for chlorine dosing and contact zone pumping; and materials for the contact zone, which included three common materials: reinforced concrete basin with stainless steel, polypropylene plastic, or no baffles; stainless steel or polyethylene plastic cylindrical tank; and polyvinyl chloride (PVC) plastic serpentine piping[97] (Figure 3.2). The sodium hypochlorite dose was a function of the required concentration times time (CT) value,[148] chlorine demand,[92] and chlorine residual. The contact zone size was based on the baffling factor,[97] CT value, and chlorine residual. Chlorine storage and pumping were based on pump efficiency and pressure[149] and the sodium hypochlorite amount. Contact zone pumping energy was based on major and minor head losses.[150] More details in Appendix section B4.

3.2.5 Ultraviolet Disinfection

The UV disinfection process included the LCI unit processes for production of energy to operate the LPUV lamp and for pumping water through the vessel; materials for the stainless steel vessel, quartz sleeve, and lamp; rubber and aluminum for the cleaning system; and control electronics (Figure 3.2). Pumping energy was based on the head loss in existing UV vessels (0.15 to 1.52 mWc).[116] Energy use and material types and amounts were based on manufacturer data for existing LPUV systems, which were small, commercially available, and validated by U.S. EPA approved protocols for drinking water. Flow normalized lamp energy for a 40 mJ/cm^2 dose ranged from 7.9 to 25 Wh/m^3, which was assumed to scale linearly with dose.[151] These values match energy values in the literature; a range of 10 to 50 Wh/m^3 has been reported for drinking water UV disinfection[33,38] (and 25,[127] 44 to 52,[124] and 74[126] Wh/m^3 for wastewater disinfection). Each scenario's regulated UV dose was multiplied by a validation factor, obtained via personal communication with the two major U.S. validation centers.[152,153] Validation factors ensure that the regulated dose is being met by the UV system at all times. Validation factors for scenario 1 were 1.4 (minimum), 2.6 (average), and 4.4 (maximum). Similarly, validation factors of 1.2, 1.6, and 1.9 were used for scenario 3. A default value of 1.0 was assumed for scenario 2's because virus inactivation validation factors are not yet publicly available; this assumption's impact was discussed.

Material compositions and masses were for a single UV lamp system (typical of existing small systems) (Table A8). Masses were assumed to be flow independent since minimum requirements would need to be met regardless of flow (e.g., at least one UV sensor would be needed). The quartz sleeve mass was based on typical sleeve thicknesses and diameters[116] and lamp lengths. The main materials associated with the cleaning system were an aluminum wiper motor and rubber wipers (torus shape). The rubber amount was based on typical quartz sleeve dimensions[116] and aluminum amount based on existing motors.[154] The largest electronics were the UVT analyzer, ballast, and UV sensor. To account for different small system operations (e.g., some small systems may not use a UVT analyzer[155]), the electronics total mass was an uncertainty parameter. When chlorine was added after UV (scenarios 1 and 3), the associated impacts were calculated using the same system boundary and calculations described in the chlorine disinfection section above; the one exception was that a contact zone was not included because the contact time for chlorine virus inactivation was so short.[148]

3.2.6 Uncertainty and Sensitivity Analyses.

A Monte Carlo analysis was conducted to propagate input uncertainty for 29 parameters and quantify life cycle environmental impacts for each alternative. All parameters were independent of one another (no correlations were expected) and included major assumptions, such as UV lamp energy use, baffling factors, and material lifetime (Table A6) but not potential uncertainty in ecoinvent data. Each parameter's data justified a uniform probability distribution and was assigned plausible maximum and minimum values based on literature values, manufacturer data, and small drinking water treatment plant operations. The sensitivity of each TRACI category to each input parameter was determined by comparing Spearman's rank correlation coefficients;[156,157] a category was defined as sensitive to a parameter if $|\rho|$ was greater than 0.8. Results are the median and uncertainty ranges are the 25th to 75th percentiles of the TRACI output values' distributions from 100,000 Monte Carlo simulations; the analysis was repeated to confirm that 100,000 simulations were more than sufficient to generate reproducible results.

3.2.7 Results Presentation

Parallel coordinate plots were used (Figure 3.3 and Figure 3.6); for these types of plots, each alternative's impact in a given category (e.g., ozone depletion) is normalized between 0 and 1 using a linear scale. Briefly, for each impact category, the alternative with the minimum impact is set to1, and all other alternatives are scaled linearly between the minimum and maximum. These plots do not imply weighting of any kind; categories should be considered independent of one another. Figure 3.4 and Figure 3.5 show results normalized to the total impact, in each category, for the chlorine alternative that used plastic piping for its contact zone.

3.3 Results and Discussion

3.3.1 Typical Source Water Scenario

Overall, the lowest impacts were observed with chlorine disinfection, but this depended on contact zone material (Figure 3.3). The four chlorine contactor alternatives with the least amount of steel (plastic pipe, plastic tank, concrete basin, and concrete basin with plastic baffles) had the smallest impacts in all categories. The best contactor material was plastic. The plastic tank and plastic pipe had the lowest impacts and were therefore most closely representative of small systems that don't have dedicated contactor infrastructure (i.e., they use water storage tanks to achieve CT). The use of a concrete contactor resulted in relatively higher emissions than using plastic baffles, but the use of steel baffles had the largest impacts compared to the other chlorine alternatives and the largest impacts in 2 out of 10 categories when compared with the UV alternatives. The production of steel for the baffles (0.96 g steel/m^3) accounted for up to 80% of impacts in each category (79% to 97% of infrastructure impacts) while the concrete basin accounted for up to 4% (Figure B1); previous studies also found that steel was the most significant contributor to infrastructure impacts.[35,78] The UV alternatives had the largest impacts in the majority of categories (Figure 3.3), mostly due to energy requirements (4.74 Wh/m^3 for validated dose of 2.1 mJ/cm^3), and electronics production impacts (0.023 g electronics/m^3), with minor impacts due to steel (0.007 g steel/m^3) (Figure B1).

Infrastructure impacts for all alternatives were lower compared to operational impacts (i.e., UV lamp energy, filter pumping energy, chlorine production), which accounted for up to 85% of total impacts in every category for all UV alternatives, and up to 99% for all chlorine alternatives (Figure B1). Other published studies show that infrastructure impacts become increasingly negligible as the energy use increases[18] or as the complexity of a water treatment train (i.e., number of processes, amount of chemicals and energy) increases.[33,34] Filter pumping energy accounted for up to 79% of impacts, for all impact categories, across all alternatives in this scenario, and was the same for all UV and chlorine alternatives; so the filter head loss assumption does not change the alternatives' comparison. UV Electronics production was up to 45% of impacts in all categories, with the largest contribution in the carcinogenic category (Figure B1). The only difference between UV alternatives was due to different validation factors, which impacted the operational dose and therefore energy requirement. Energy for the UV lamp and vessel pumping had large contributions, up to 30% in all categories, with the largest in the global warming category (Figure B1). For this scenario's regulated UV dose (2.1 mJ/cm^2), the UV vessel pumping energy was 3.8 Wh/m^3 versus 0.94 Wh/m^3 for the lamp.

There have been several wastewater treatment LCA studies that compared UV and chlorine disinfection. One found that UV and chlorine options were comparable on a cost basis.[125] Other studies found that UV disinfection had larger environmental impacts than chlorine disinfection, mostly due to the energy demands of UV lamps.[126,127] However, the results from these wastewater studies have limited transferability to drinking water; specifically, wastewater treatment requires dechlorination, which has been found to have larger environmental impacts than other disinfection processes,[126,127] and water quality is very different between wastewater and drinking water, which impacts the chlorine demand and UV dose. There are limited drinking water disinfection LCAs that directly compare UV and chlorine disinfection; there is one that compared UV to gaseous chlorine and also found that UV had worse environmental performance.[33]

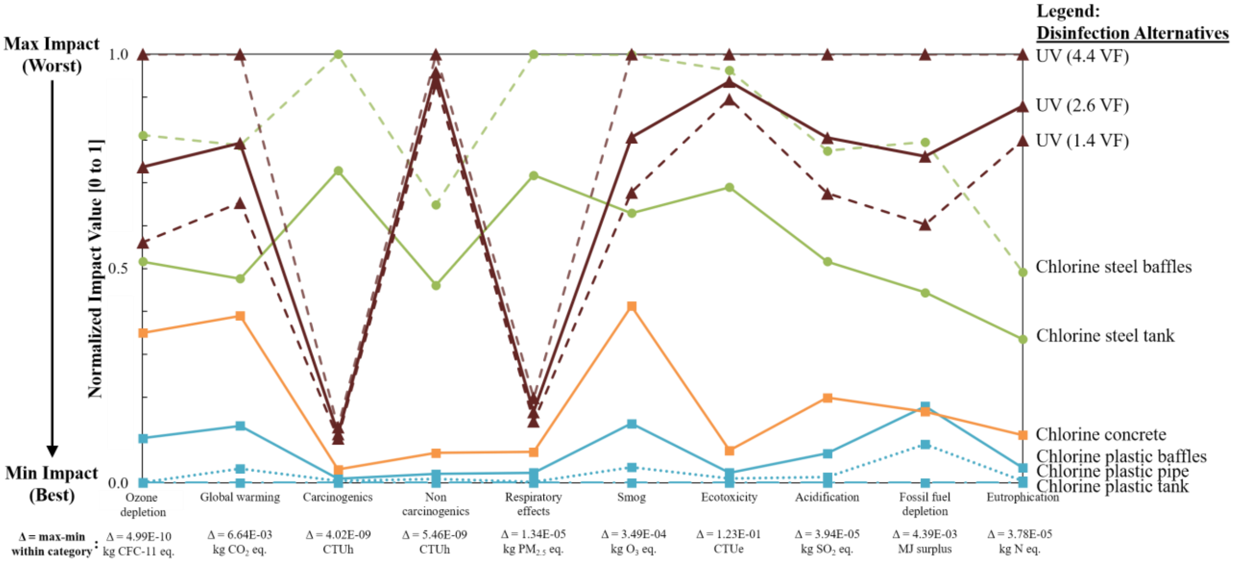


Figure 3.3. Parallel coordinate plot showing the relative environmental impacts in all 10 TRACI categories for the first typical source water scenario's 6 chlorine disinfection alternatives with different contact zone materials and 3 UV disinfection alternatives with different validation factors. This graph does not imply weighting of any kind, and categories should be considered independent of one another. Symbols: rectangles represent chlorine disinfection alternatives with limited steel infrastructure (concrete basin, concrete basin with plastic baffles, plastic tank, plastic pipe); circles represent chlorine disinfection alternatives using steel infrastructure (concrete with steel baffles, steel tank); and triangles represent UV disinfection alternatives with different validation factors (VF) of 1.4, 2.6, and 4.4, which dictated the operational UV dose and energy requirement.

3.3.2 Uncertainty and Sensitivity

The aggregate impact of technology and design assumptions on the results was evaluated by conducting a Monte Carlo analysis. Overall, the chlorine alternatives using steel infrastructure and the UV alternatives had the largest uncertainty ranges (Figure B3). the non-steel chlorine alternatives' uncertainty range values were plus or minus the median value by less than 4%, steel chlorine alternatives by less than 25%, and UV alternatives by less than 15% (Figure B3). There was some overlap of uncertainty ranges between the steel chlorine, concrete chlorine, and UV alternatives; but there was no overlap between the plastic chlorine and UV alternatives. Overall, this minimal overlap of uncertainty ranges signifies that the results were robust to input uncertainty.

The chlorine disinfection plastic tank and plastic pipe alternatives were sensitive ($|\rho|>0.8$) to plastic thicknesses and filter replacement rate. The chlorine with steel tank alternative was sensitive to steel thickness. Functionally thinner materials would reduce environmental impacts. The UV alternatives were sensitive to the electronics mass and UV vessel head loss uncertainty parameters. Reducing both will improve the environmental performance of UV systems. Minimizing UV vessel head loss is an active research area.[158,159] As water systems increase automation, there may be a tradeoff to evaluate between the environmental cost of electronics production, water quality, and consistent monitoring and automation, which provides greater opportunity for chemical and energy optimization.

To determine the impact of individual assumptions on the results, a sensitivity analysis was performed. The values of major assumptions were changed (one at a time) while keeping all other assumptions at their expected value. Assumptions on energy source, disinfectant dose, regulatory requirements for a chlorine residual (scenario 2), and regulatory requirements for filtration (scenario 3) had the largest impact on the results.

3.3.3 Energy Source

The typical source water scenario’s trends were found to be sensitive to energy source (e.g., coal versus. solar). Overall, chlorine alternatives had increasingly better environmental performance than UV alternatives as energy-related impacts increased (Figure B6 and Figure B7); for example, when the highest-impact U.S. electricity grid (MRO) was used, the moderate chlorine alternative (the chlorine alternative with an unbaffled concrete basin, which had moderate impacts compared to the other alternatives; Figure 3.3) was better than the average UV (2.6 VF) alternative in all TRACI categories (Figure 3.4). However, if a utility was powered solely by an energy source with negligible life cycle emissions (i.e., solar power), then the average UV alternative would have better environmental performance in 4 categories, similar in 3 categories, and worse in 3 categories, compared to the chlorine concrete alternative (Figure 3.4). For UV alternatives, the benefits of switching to a renewable energy source increased with UV dose (Figure B6 and Figure B7). Other studies have also identified environmental benefits of using renewable energy sources for drinking water treatment.[160] Since no current renewable energy source has completely negligible emissions,[161] the relative impacts were quantified for the lowest-impact U.S. electricity grid (NPCC), which resulted in the UV alternative having worse impacts than the chlorine concrete alternative in 9 categories.

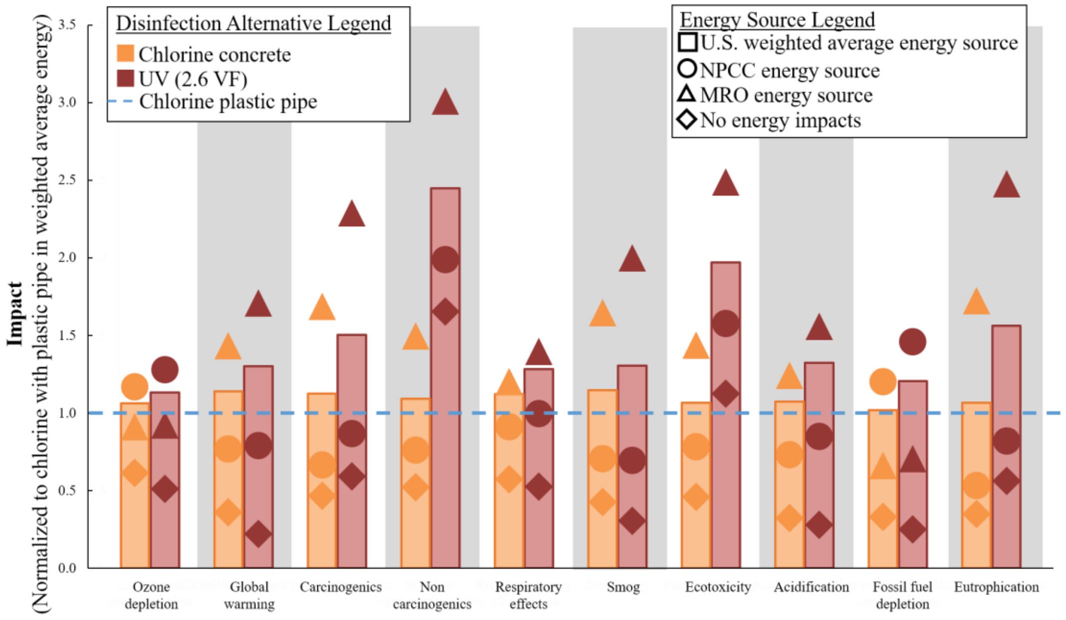


Figure 3.4. Impact comparison between chlorine and UV disinfection alternatives (typical source water scenario) when considering different energy sources: (i) U.S. weighted average energy source (bars); (ii) Northeast Power Coordinating Council (NPCC) grid, a relatively low impact U.S. energy grid (circles); (iii) Midwest Reliability Organization (MRO), a relatively high impact U.S. energy grid (triangles); (iv) "Zero-impact Energy," a case where there are no impacts associated with energy (diamonds). VF is validation factor.

3.3.4 Disinfectant Dose

To represent potential small system operation of "overdosing" or use of safety factors, the primary disinfectant doses were increased. The chlorine concrete alternative's chlorine dose was doubled from 1.5 to 3.0 mg/L. The higher dose increased the environmental impacts by about 150% in all categories but decreased the contact zone volume required to meet the CT value, so relative infrastructure impacts were reduced, from up to 15% contribution to less than 9% in each category (Figure B9). For the UV alternative, the UV validated dose was set to 10 mJ/cm^2, which was recommended due to sensor limitations and UV dose variability.[153] This increase only slightly increased environmental impacts (by about 1%) from the UV maximum validation factor alternative, which had an operational dose of 9.2 mJ/cm^2. With these dose assumptions, UV disinfection had better performance in 3 categories and worse in 7 categories than the chlorine concrete alternative (Figure B9). Overall, UV was worse than chlorine in all 10 categories when doses were optimized (Figure 3.3). While providing a chlorine safety factor has historically been emphasized due to acute pathogen risk,[162] the tradeoffs associated with increased chemical use, and associated environmental impacts and costs, as well as the increased DBP risks should be considered.

3.3.5 Chlorine Residual Exemption Regulatory Scenario

For this scenario, the UV alternative (186 mJ/cm^2 pre-validation dose and no chlorine addition) had the largest impacts, when compared to the non-steel chlorine alternatives, in all categories (Figure 3.5 and Figure B4). Similar to scenario 1, filtration impacts were equivalent for all alternatives. In addition to filtration impacts, the largest contributions to UV impacts were due to electronics (22% contribution to non-carcinogenic impacts) and energy required to operate the UV lamp, which accounted for 60% to 80% of impacts in every category. The amount of energy required to treat 1 m^3 of water with the 186 mJ/cm^2 (pre-validation) dose was 83 Wh/m^3, which is similar to the amount of energy used to produce potable water from surface water (72 to 230 Wh/m^3).[163] In addition, the inclusion of a LPUV virus inactivation validation factor, as future research could allow, is expected to increase lamp energy requirements. Overall, the UV lamp energy's negative impacts far outweighed the benefits from not producing chlorine and the use of chlorine, or at least the partial use of chlorine, is environmentally preferred.

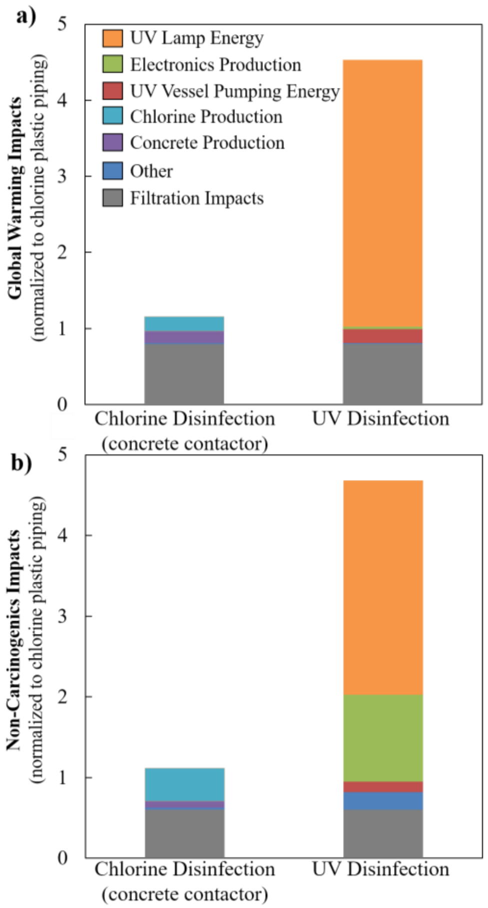


Figure 3.5. Contribution of all LCI unit processes to the global warming (a) and non-carcinogenics (b) TRACI impact categories for the chlorine residual exemption scenario. The global warming category results had similar trends as the acidification, fossil fuel depletion, and smog categories; the non-carcinogenics category results had similar trends as the carcinogenics, ecotoxicity, eutrophication, ozone depletion, and respiratory effects categories. The two alternatives shown are chlorine disinfection with a concrete basin and UV disinfection, which used a 186 mJ/cm^2 UV dose to achieve 4 log-virus inactivation and did not use any chlorine.

3.3.5 Filtration Exemption Regulatory Scenario

For this scenario, UV disinfection was used to achieve *Crypto* log reduction instead of filtration (i.e., an unfiltered system); chlorine disinfection still used filtration and the differences between chlorine alternatives were contact zone material and filtration energy requirement (Figure 3.1). Due to the filtration difference between UV and chlorine alternatives, the environmentally preferred choice depended on the chlorine alternative's filtration energy requirements (Figure 3.6).

The uncertainty of the filtration energy requirements is mostly a function of a combination of location specific considerations, such as head coming into the plant and turbidity of the raw water, that are unique to a given water system, and should be considered when interpreting comparative results. For example, the chlorine concrete basin and plastic pipe alternatives had better environmental performances than UV disinfection, in all categories, if the average filter pressure was less than 0.7 mWc and 1.4 mWc, respectively (Figure 3.6). UV disinfection was better in all categories when the chlorine alternatives' average filter pressure was greater than 11 m. At the most likely filter pressure of 3.5 mWc,[147] the UV alternatives had smaller impacts in six categories, larger impacts in two categories, and similar impacts in two categories, compared to the non-steel chlorine alternatives. When assuming equal weights between TRACI categories, UV was environmentally preferred when protozoan inactivation was needed, and filtration was not required for turbidity removal.

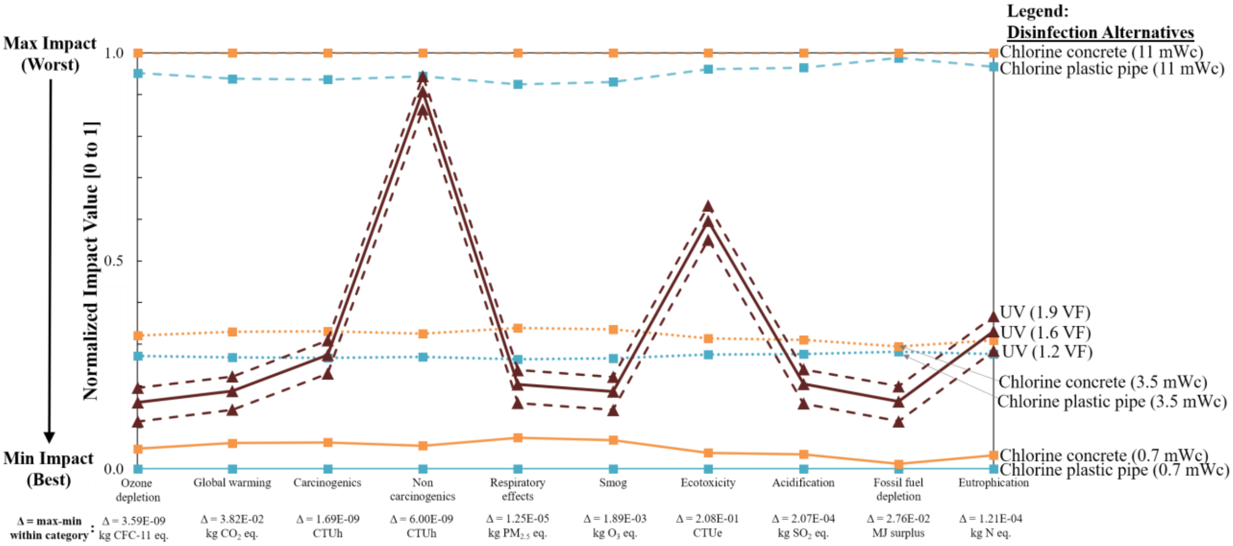


Figure 3.6. Parallel coordinate plot showing the 10 TRACI categories results for six chlorine disinfection alternatives, including 2 contact basin materials (concrete basin and plastic pipe) and 3 pumping energy requirements (filter pressures of 0.7 mWc, 3.5 mWc, 11 mWc), and for 3 UV disinfection alternatives with validation factors (VF) of 1.2, 1.6, and 1.9.

3.3.6 Water System Decision-Making

This study can help a utility improve environmental performance given their specific treatment objectives and resource limitations. From an LCA point of view, replacing chlorine with UV disinfection was preferred only in a limited number of cases (i.e., high pumping pressure but filtration is not required). In all other cases, chlorine was environmentally preferred, although some contact zone materials and energy sources had an impact on the comparison. For example, using plastic materials or storage tanks for contact zones can improve environmental performance. Systems struggling with DBP formation could use UV to reduce the chlorine dose and DBP formation.[115,164] For systems at risk of protozoan parasites, a utility could replace high-energy filtration with UV and become classified as an unfiltered system, as a number of large utilities have done (e.g., New York City, Seattle, Vancouver), to reduce life cycle environmental impacts. For any small system, optimizing disinfectant doses and using renewable energy sources should be considered to reduce environmental impacts. With this environmental data, utilities can comprehensively consider tradeoffs, such as land requirements (e.g., a UV reactor can be compact; using existing infrastructure for chlorine contact can minimize space) and costs (e.g., chemical versus energy costs). Future research on assessment frameworks that integrate the various drinking water decision metrics (e.g., cost, maintenance, environmental objectives, public health) can ensure decision making is comprehensive.

Chapter 4: A New Framework for Small Drinking Water Plant Sustainability Support and Decision-making

The following manuscript and appendix are in preparation for submission for publication: Jones, C. H.; Meyer, J.; Seidel, C. J.; Cornejo, P. K.; Hogrewe, W.; Cook, S. M. A New Framework for Small Drinking Water Plant Sustainability Support and Decision-making.

Abstract

Public drinking water system decisions about treatment processes are becoming more challenging, especially as regulations become more stringent and source water quality degrades. For small systems that serve less than 10,000 people, treatment decisions are particularly difficult due to limited resources and because they do not currently have tools to help them make informed and sustainable decisions. Therefore, a user-friendly decision-making tool, which compares treatment processes relevant to a wide variety of small drinking water systems, was constructed. In summary, the tool uses life cycle assessment and multiple-criteria decision analysis to comprehensively evaluate twelve decision criteria, developed specific to small drinking water systems; the tool then uses an aggregation approach to identify and navigate multiple trade-offs, and make a final recommendation based on stakeholder values. Four hypothetical scenarios were examined to show the tool's applicability to diverse small systems, ability to help stakeholders navigate trade-offs, and engineering relevance. The tool is universal in its capacity to evaluate systems with different design parameters, source waters, treatment criteria, and stakeholder preferences.

4.1 Introduction

Small drinking water systems (SDWSs) serve up to 2.5 billion people globally.[1] They account for 97% of the public water systems in the United States (U.S.).[2] Despite their importance, SDWSs face a wide range of issues that challenge their ability to provide high quality drinking water.[4] These challenges are being exacerbated due to increasingly stringent regulatory requirements and deteriorating source water quality.[6,8] Typically, SDWSs have relied on cost to make treatment technology decisions, but there are many issues not captured in those traditional assessments. Since SDWSs are facing multiple diverse challenges, they need comprehensive decision-making processes that include criteria from all three sustainability pillars of economic, environmental, and social.[165] To further ensure a comprehensive evaluation that includes long-term objectives, life cycle thinking is also needed.[166]

Economic, environmental, and social considerations, over a SDWS's lifetime, need to be represented in decision criteria to avoid overlooking or exacerbating problems. For example, full costs should be considered by using life cycle costing (LCC) to evaluate the affordability of a treatment technology beyond its initial construction. This long-term thinking also needs to include environmental considerations, which are particularly important because of the positive feedback loop between source water degradation and global pollution (i.e., treating degraded water requires more resources which results in more pollution and thus further water degradation). Previous drinking water studies have successfully used life cycle assessment (LCA) as a standardized approach to identify treatment approaches that can reduce global pollution.[34,91,167] Treatment decisions should also incorporate social and health considerations. For example, choosing treatment technologies that can provide higher quality water than is required by regulation increases flexibility, to deal with source water quality changes or contaminant spikes, and capacity to minimize health risks, such as those from non-zero (but below regulatory limit) concentrations of carcinogenic disinfection byproducts.[168] Of course, there may be trade-offs to over-treating drinking water, so there should be multiple social considerations. For example, a SDWS must ensure that a treatment alternative is feasible for plant operations, especially operator training and maintenance. Inclusion of comprehensive decision criteria will best support SDWS sustainability by identifying and explaining tradeoffs.

There has been a recent focus on using multiple-criteria decision analysis (MCDA) for informing decisions on engineered water systems,[165,169–172] but most SDWS studies have only focused on one criterion, such as source water availability,[45] cost modeling,[46] or financial aid options.[47] There are two main decision frameworks used for SDWSs that have evaluated multiple-criteria: the technical, managerial, financial approach developed by the U.S. Environmental Protection Agency (EPA)[4] and the ELECTRE method,[173] which one study used to assess economic and social criteria specific to hydroelectric plant treatment alternatives.[48] Large drinking water treatment system MCDA studies have better assessed multiple-criteria, such as cost and environmental impacts[43] and health risk, cost, and technical feasibility.[174] However, they have limited applicability to small systems given the complexity of issues facing SDWSs (e.g., lack of relevant and comprehensive criteria) and differences due to size (e.g., technology options, personnel). Overall, existing MCDA frameworks can be leveraged to provide a comprehensive decision-making framework but require substantial modifications, such as incorporating life cycle thinking and better accounting for environmental and social criteria, to be applicable to SDWS challenges and stakeholders.

As a result, SDWS stakeholders do not currently have the decision-making support they need to find solutions to complex SDWS challenges. Therefore, a decision-making framework, called

Small drInking water Plant Sustainability support (SIPS), was constructed. SIPS includes small system specific criteria, representing all sustainability pillars, that are evaluated with a combination of LCA and MCDA methodologies to ensure equivalent and comprehensive technology comparisons. The methods selected and developed to form the SIPS framework are described, including the drinking water process models and criteria scoring methods. Four unique, hypothetical SDWS scenarios were evaluated to assess SIPS's ability to identify and navigate trade-offs, between criteria and between alternatives. SIPS recommendations for each scenario, based on different stakeholder values, were used to help assess SIPS's engineering relevance. SIPS can be used by SDWS stakeholders (e.g., community members, engineers, operators, regulators, financiers) for a wide range of conditions to help them make sustainable treatment choices that improve drinking water quality.

4.2 Methods

Decision criteria and goals for the usability and interface of the SIPS tool were collaboratively determined with input from multiple small system stakeholders, which included attendees of the U.S. EPA small system annual meeting and Rural Community Assistance Partnership (RCAP) annual national training conference, RCAP technical providers, and Design of Risk-reducing, Innovative-implementable Small-system Knowledge (DeRISK) SDWS Research Center technical advisers. The goals were for the tool to be transparent and simple, so it can be modified by users with minimal training; accessible, so a wide range of SDWS stakeholders can use it (with existing, common software, i.e., Excel) and can understand its outputs; and comprehensive, so it can be applied to any SDWS. To meet these goals, SIPS was designed to combine life cycle thinking, using the ISO 14040 LCA framework,[10] criteria scoring methods, and MCDA methods (the multiple existing methods considered are detailed in Section C2); each was modified as needed for SDWSs. SIPS recommends treatment train alternatives by following 7 main steps (Figure 4.1).

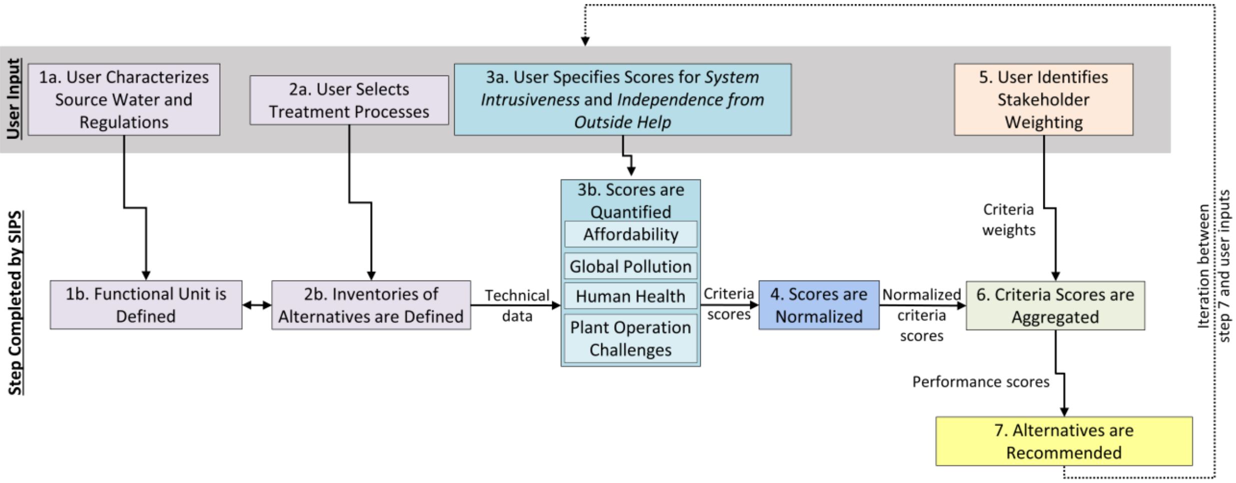


Figure 4.1. The seven main steps that SIPS uses to provide SDWS stakeholders with a recommendation of alternatives being considered.

4.2.1 Functional Unit (Steps 1a and 1b)

The user defines the SDWS's treatment requirements, source water quality, system size (i.e., flow rate), and system location (to identify electricity sector specific information). Treatment requirements must include total organic carbon (TOC) and turbidity reduction goals. SIPS assumes all treatment trains must achieve 2-log *Cryptosporidium* reduction, 3-log *Giardia* reduction, and 4-log virus reduction, based on U.S. drinking water regulation,[84] but the user can modify this if other pathogen goals are desired. The user then inputs source water quality information on TOC, turbidity, ultraviolet transmittance (UVT), pH, temperature, alkalinity, and bromide; SIPS can accommodate all likely values for each of these parameters (Table C1). These user inputs add the required detail to the functional unit skeleton of: the treatment of the user-specified source water and flow to the user-specified drinking water quality (based on treatment requirements) over 40 years of operation, which is a common period for a drinking water LCA.[79,80]

4.2.2 Treatment Train Alternatives (Steps 2a and 2b)

The user selects each treatment train alternative they want to compare. Each treatment train is a combination of treatment processes that can meet the functional unit. The individual treatment process options in SIPS represent the most common SDWS options and include two coagulation chemical options, seven filtration options, two disinfectant options, four chlorine contact zone options, and two pH adjustment chemical options (Table C2). For each treatment train alternative, an inventory of the types and amounts of materials, chemicals, energy, and labor required over the life cycle functional unit was compiled by following LCA methodology. Multiple drinking water treatment process models and design algorithms were selected or developed as needed and then linked (Figure C1) to calculate the inventory data. The LCA system boundaries for each treatment train alternative were chosen to ensure consistency between the alternatives and accurate life cycle comparisons (Figure C2).

All alternatives are compared for the same system; since the flowrate is the same, the inventory of piping, fittings, and pumps and costs of engineering, surveying, building construction, and construction labor were excluded. To minimize user input requirements, SIPS has default values for model and design parameters, which include five operational preferences (i.e., chlorine dose, coagulant dose, ozone feed type, filter concentration times time (CT) credit, final water pH) (Table C3) and 59 design parameters (e.g., media heights, hydraulic residence times, etc.) (Table C4), that can be modified by the user.

The inventory for both coagulation options included the amount of coagulant chemical (alum or ferric chloride) as well as chemical and solids disposal hauling. The life cycle amount of chemical needed was based on the coagulant doses required to achieve the user-defined TOC and turbidity removal requirements. The coagulant dose needed for TOC removal was calculated using the Edwards Model.[93] No model exists to determine the coagulant dose needed for turbidity removal, so this dose was determined using historical data from 193 systems.[95] The user can also enter coagulant doses from jar tests. The hauling requirements for coagulant chemicals and resulting solids production (i.e., coagulated TOC) were included using a previously developed LCA model.[167]

There are seven filtration options. Due to lack of existing models, new LCA models were constructed to calculate the inventory data for the bag, membrane, and slow sand filtration options (Section C7). Bag filtration removes solids and pathogens by pumping water through a bag contained in a filter housing. The inventory included pumping energy and bag filter and housing materials (Section C7.2). Membrane filtration also operates on a solids/pathogen exclusion principal by pumping water through skid-mounted membrane modules. The inventory included pumping energy, membrane filter, and skid materials (Section C7.3). Slow sand filtration operates by pumping water through layered sand and gravel that are contained in a reinforced concrete basin. The inventory included pumping energy, media, and concrete with rebar materials (Section C7.4). The amount of biodegradable dissolved organic carbon (BDOC) removal achieved was determined using experimental data[175] to calculate a reduced coagulant dose for TOC removal.

The inventories of the other four filtration options (cartridge filtration, conventional filtration, nonozonated biofiltration, and ozonated biofiltration), two disinfection options (chlorine with four contact zones and UV), and two pH adjustment chemical options (caustic and lime) were calculated using previously developed LCA models.[91,167] In summary, each filtration option included filter operational energy, housings, vessels, skids, and media, as applicable. The inventory for ozonated biofiltration also included the ozone generator energy and steel. For nonozonated biofiltration and ozonated biofiltration, the amount of BDOC removed during biofiltration was incorporated. The inventory for chlorine included pumping energy for the contact zone, materials for chlorine storage tank, contact zone, and sodium hypochlorite chemical, and chemical hauling. For UV, the inventory included lamp energy, pumping energy to overcome head loss in the UV vessel, and the major materials for a UV disinfection system. The inventory for pH adjustment chemical options included the pH adjustment chemicals and chemical hauling.

4.2.3 Decision Criteria (Step 3)

First, an unabridged list of possible SDWS-specific decision criteria were developed (Table C5), in collaboration with RCAP technical service providers who have significant SDWS experience. Next, to get input from broader groups of SDWS stakeholders, the list was discussed with workshop attendees at an Annual EPA Drinking Water Workshop on Small Systems Challenges and Solutions Conference, RCAP National Conference, and American Water Works Association's Water Quality and Technology Conference. Then, the literature was consulted to help standardize criteria so they could be applicable to any SDWS (Table C6). From this input, twelve SDWS-specific decision criteria were identified to encompass the wide range of SDWS issues (Table 4.1).

Table 4.1. Summary of SIPS's twelve SDWS decision criteria and their scoring methods.

Decision Criteria	Description	Criteria Scoring Method	Units
Affordability Category			
Capital Cost	Initial costs, including materials for infrastructure.	Sum of Initial Material Costs (Section C3) [176,177]	$
Operation & Maintenance Cost	Continuous costs required by operation (energy and chemicals), material replacement, and maintenance.	Sum of Recurring Costs (Section C3) [176,177]	$/yr
Global Pollution Category			
Global Climate Change	Changes that affect the global climate and living conditions of humans and animals (e.g., global warming, sea level rise).	IMPACT 2002+ [178]	kg CO_2 eq.
Global Ecosystem Quality	Reduction of biodiversity and the livability of traditional habitats for animals.	IMPACT 2002+ [178]	PDF*m^2*yr
Global Resources	Consumption and depletion of non-renewable resources (e.g., coal, oil, natural gas).	IMPACT 2002+ [178]	MJ primary
Human Health Category			
Global Human Health	Pollution that results in the loss of human life and reduction of the quality of life.	IMPACT 2002+ [178]	DALY
Local Long-term Health Risk	Chronic health impacts associated with consuming SDWS water.	RHI Model [168,179]	RHI
Resilience to Regulation and Source Water	A treatment train's long-term ability to remove non-regulated contaminants and to provide multiple barriers to contaminant breakthrough.	Ranking Table (Table C10)	Unitless value of 1 to 5
Plant Operation Challenges Category			
Operator Training Requirement	Minimum amount of training required to operate any process in a treatment train.	Ranking Table (Table C12)	Unitless value of 1 to 5
Maintenance Requirements	Total number of hours of maintenance over one year.	Sum of Annual Maintenance Hours (Section C5) [177]	hrs/yr
System Intrusiveness	Abundance and severity of community nuisances.	Ranking Table (Table C14)	Unitless value of 1 to 5
Independence from Outside Help	A treatment train's potential for operation without help from third party entities.	Ranking Table (Table C15)	Unitless value of 1 to 5

Each criterion's scoring method (Table 4.1) uses a treatment train's inventory data. *Capital cost* and *operation and maintenance cost* criteria calculations followed an LCC approach.[176] Relative differences in alternatives' total costs were calculated since only differences between alternatives need to be evaluated (i.e., alternatives are evaluated for the same SDWS, so many costs between alternatives are the same). An alternative's *capital cost* was the sum of construction unit costs (Table C7) multiplied by the relevant inventory amounts. An alternative's *operation and* *maintenance cost* was the net present value of the sum of labor and non-capital inventory items unit costs (Table C7) multiplied by the total maintenance hours and amounts of non-capital inventory items, respectively. Since operation and maintenance costs occur at different times over the 40 year functional unit, all costs were brought to present value using appropriate discount and escalation rates (Figure C3).

For global pollution and global human health impacts, all of an alternative's inventory items were translated to environmental emissions using life cycle inventory (LCI) unit processes from ecoinvent[81] and US-EI 2.2[82] LCI databases (Table C8). Those emissions were then translated into 4 endpoint LCA indicators, which provide a higher level of emissions consolidation than midpoint indicators to improve relevance and understandability,[180] using IMPACT 2002+:[178] *climate change*, *ecosystem quality*, *resources*, and *human health*. The SIPS decision criteria include *global* before each LCA indicator to emphasize that impacts are not necessarily local, as they are happening along a global supply chain, and to differentiate the health criteria. For SIPS calculations, the LCI unit process information was first translated into the LCA indicators (e.g., kg CO_2 equivalent per kg stainless steel) using SimaPro software and next imported into SIPS. This data was then multiplied by each alternative's inventory quantities during criteria scoring.

The *local long-term health risk* criterion represents the health risks associated with consuming a SDWS's water. The risks were quantified using a drinking water RHI model.[168,179] Briefly, the RHI model calculates the risk exposure from disinfection byproducts in the drinking water leaving the SDWS (i.e., before the distribution system) by using contaminant exposure, toxicity, and non-cancer/cancer severity.

The *maintenance requirements* criterion score was estimated based on the total number of hours of maintenance over one year. The labor hours for engineering, surveying, and construction were excluded since the differences were negligible compared to the operational differences. Yearly maintenance hours were calculated by translating the number of instruments, number of vessels, plant area, and number of chemicals to total maintenance time by using the EPA work breakdown structure model's time per task relationships (Table C9).[181]

The following four decision criteria were unique to SDWSs and did not have existing criteria scoring methods, so score ranking tables were developed for each. The criterion *resilience to regulation and source water* accounts for SDWS challenges related to deteriorating source water quality and increasingly stringent regulations. Resilience was defined as a treatment train's ability to provide multiple barriers to contaminant breakthrough and to remove non-regulated contaminants (i.e., to exceed expected drinking water quality). The score ranking table developed for this criterion used integer values from one to five (Table C10), based on the World Health Organization's five-tiered risk scoring approach;[182] a score of one represents a treatment train that only requires a small change in operation (i.e., change in chemical dose) to mitigate any effects of a contaminant spike and that can remove unregulated contaminants of concern (e.g., pharmaceuticals). SIPS has a default *resilience to regulation and source water* score for each treatment process (Table C11) that the user can modify. The treatment train is assigned the best score of all of its individual treatment processes' scores (i.e., it only takes one individual process to provide these resilience capabilities to the entire treatment train).

The *operator training requirement* criterion accounts for a common SDWS problem of recruiting and retaining highly qualified operators (i.e., so SDWS may prefer to have low training requirements). The developed score ranking table was based on multiple states' operator level requirements,[183] which were assigned integer values between one and five (Table C12); a score of one represents a treatment train with the lowest amount of operator certification required. SIPS has an *operator training requirement* score for each treatment process (Table C13) that users can modify. The treatment train is assigned the worst score of all of its individual treatment processes (i.e., an operator must have the highest level of certification required by any process in a treatment train).

The *system intrusiveness* criterion indicates if a treatment train is a nuisance to the community. The score ranking table was developed based on the abundance and severity of community nuisances (e.g., taste and odor complaints, truck traffic complaints, and other complaints from the community) using integer values from one to five (Table C14); a score of one represents a treatment train that is not considered intrusive (i.e., no complaints). When a community nuisance is severe (e.g., taste and odor of drinking water is so bad that consumers avoid using their tap water), it is counted as being worse than if a consumer just notices a nuisance (e.g., taste and odor is noticeable but does not change water use behavior). The scores will be specific to the SDWS, so the user must select scores for each alternative.

The *independence from outside help* criterion accounts for a common SDWS goal of independence (i.e., not having to rely on other drinking water systems for source water or on third-party entities for treatment process parts or chemicals). The score ranking table was developed to identify how reliant a SDWS is on a third-party entity by using integer score values from one to five (Table C15); a score of one represents a treatment train that does not rely on a third-party entity. Scoring may be reversed (i.e., if reliance on other drinking water systems is desirable). The scores will be specific to the SDWS, so the user must select scores for each alternative.

4.2.4 Score Normalization (Step 4)

Within a criterion, all alternatives' scores are normalized by the difference between the minimum and maximum scores (Eq. C1). This approach is an effective method for normalizing values regardless of differences in score magnitudes, within criterion or across criteria.[184] If the differences between the scores within a criterion are expected to be insignificant, the user can define a utopia point (i.e., define the best possible score),[184] which will assure that any score that is at or below that utopia point value is also regarded as having the best score. To help the user evaluate possible trade-offs between alternatives and criteria, the user may visualize the normalized scores for all criteria via a parallel coordinate plot.[185]

4.2.5 Stakeholder Weighting (Step 5) and Alternative Recommendation (Steps 6 & 7)

Individual stakeholder preferences are incorporated by using weights (0% to 100%) for each of the 12 decision criteria. All 12 weights add up to 100%. Each alternative's 12 normalized criteria scores are summed and weighted by the criteria weights to calculate a performance score (Eq. C2). This weighted sum method is a well-established, effective, and transparent single-score aggregation method.[184,186]

There are three options to specify weights in SIPS. One is to take the SIPS AHP survey, which uses pair-wise comparisons to rank the relative importance of each criteria.[187] To reduce the number of comparisons and improve consistency, pair-wise comparisons are first completed between the four criteria categories (affordability, global pollution, human health, and plant operation challenges) and then between each criterion within a category. This criteria weighting option is the most rigorous and provides the most consistent weights. The second option is to select a preset weighting scheme, which include: equal criteria, equal category, cost-focused, health-focused, operation-focused, and pollution-focused weights (Table C16). The third option is to manually input weights for each criterion. If there are negligible differences between the best and worst scores within a criterion, weights can be adjusted while being informed of raw criteria scores (i.e., to mitigate potential overestimations of differences between scores after normalization).[188]

4.3 SIPS Application & Recommendation Relevance

To demonstrate the application of SIPS, four hypothetical scenarios were evaluated. This evaluation included eight treatment train alternatives and three distinct SDWSs (Table 4.2). The functional unit was to treat 0.1 Mm^3/yr (0.07 mgd) source water to meet (SDWS#1 and SDWS#2) or surpass (SDWS#3) U.S. regulation[84] over 40 years of operation. For the two criteria users must score (*system intrusiveness* and *independence from outside help*), hypothetical scores were assigned for each alternative (Section C9). The equal criteria weighting scheme was used initially; then two other weighting schemes were used to evaluate the effects that stakeholder values could have on results.

4.3.1 Hypothetical SIPS Examples

Table 4.2. Summary of SIPS inputs of source water quality and regulations for each hypothetical SDWS. Hypothetical source waters were based on.[189]

Parameter	SDWS#1	SDWS#2	SDWS#3
pH	7.5	7.5	7.5
UVT (%)	95	80	90
Temperature (°C)	10	20	15
TOC (mg/L)	1	4	1.5
Alkalinity (mg $CaCO_3$/L)	50	100	60
Bromide (mg/L)	.02	.08	.02
Turbidity (NTU)	<1	>1	<1
TOC Removal Goal	U.S. Regulation	U.S. Regulation	50% Removal
Reason for Making Treatment Decision	Reclassified as a surface water	Had to change water source	Wants more biostable water

4.3.1.1 Filtration Selection Examples

First, four filtration-focused alternatives, all with the same disinfection process (chlorine using a concrete contact zone), evaluated filtration options: (1) conventional filtration, (2) cartridge filtration, (3) membrane filtration, and (4) nonozonated biofiltration. Hypothetical SDWS#1 was reclassified as a surface water system, so a filter is being added to manage potential pathogen risk. It has a pristine high alpine source water (Table 4.2). SIPS results show that each alternative had trade-offs between the twelve decision criteria, and the most preferred filtration-focused alternative was the cartridge filtration treatment train (Figure 4.2).

The cartridge filtration alternative performed the best in all affordability and global pollution criteria and all but one operational challenges criterion because it used the fewest resources (i.e., materials, energy, chemicals) and had the fewest operational demands (e.g., low operator training and maintenance requirements). This alternative did not perform as well in human health criteria because it has higher disinfection byproduct formation risks (because organic matter is not reduced by the cartridge filter) and is not resilient to deteriorating source water (because cartridge filters are prone to clogging quickly). The membrane filtration alternative had the same operational benefits, with similar trade-offs as the cartridge filtration alternative. The major difference was that the membrane filtration alternative used more operational energy, which made it the worst alternative for *operation and maintenance costs* and for global pollution criteria.

In contrast, the conventional filtration alternative performed best in the *local long-term health risk* and *resilience to regulation and source water* criteria because it included coagulation, which provided disinfection byproduct precursor removal and an additional barrier to contaminant breakthrough. However, this came with a trade-off in being the worst alternative in three of the four operational challenges criteria due to the complexity of operating that additional chemical process. The nonozonated biofiltration alternative also performed best in *local long-term health risk* for the same reasons as the conventional filtration alternative and in *system intrusiveness* because the biological treatment best removes taste and odor nuisance compounds. Overall, for each alternative, the multiple benefits and disadvantages across all the decision criteria were navigated using the SIPS aggregation approach. SIPS recommended the cartridge filtration alternative because its advantages of low chemical, energy, and operational burdens outweighed its health risk disadvantages better than the other alternatives under an equal criteria weighting scheme.

Hypothetical SDWS#2 had to switch source waters due to saltwater intrusion, so new treatment alternatives are being considered to meet the more stringent TOC reduction standards. Its new source water is a tropical water with high turbidity and TOC (Table 4.2). In this scenario, SIPS showed that the nonozonated biofiltration train performed best in all criteria except for *capital cost* (Figure 4.2). SIPS's recommended nonozonated biofiltration, which is different from the first scenario's recommendation of cartridge filtration. The recommendation changed because SDWS#2's had a more contaminated source water that required coagulation addition before cartridge and membrane filtration options, increasing the expense, emissions, and operational complexity of those alternatives in this scenario.

Hypothetical SDWS#3 is concerned about biological growth in the distribution system, so they are increasing their TOC reduction goal to 50% removal to improve the water's biostability. Its source water was pristine, similar to SDWS#1 (Table 4.2). SIPS found that the nonozonated biofiltration alternative performed best in most criteria (Figure 4.2). Although this SDWS's source water was similar to SDWS#1's, the higher treatment standard required for biostable water resulted in more treatment resources needed before the cartridge and membrane filtration options (i.e., coagulation had to be added). As a result, SIPS recommended the same alternative for SDWS#3 as for SDWS#2.

Also, for both scenarios, conventional filtration performed better than the cartridge or membrane filtration alternatives and only slightly worse than nonozonated biofiltration (Figure 4.2). The performance scores are always relative to the alternatives being considered for a given SDWS; small differences in performance scores indicate more similar alternatives. Conventional filtration was worse than nonozonated biofiltration for *local long-term health risk* and *system intrusiveness* but performed the same otherwise. The differences in performance scores were only 0.17 and 0.13 (versus 0.75 and 0.58 difference between the recommended and worst alternative) for SDWS#2 and SDWS#3, respectively, so these two alternatives could be seen as relatively similar choices for these scenarios.

Overall, SIPS's recommendations for these three scenarios aligned well with common engineering knowledge of treatment technologies. For SDWS#1, SIPS's recommendation of cartridge filtration aligns well with what is seen in the field. For example, many existing Colorado SDWSs, with pristine source waters like SDWS#1, use cartridge filtration.[91] For SDWS#2 and SDWS#3, SIPS recommended nonozonated biofiltration, which is becoming a common drinking water filtration process due to its ability to reduce disinfection byproduct health risks and minimize coagulant chemical use.[190] These benefits are realized most with higher levels of organic matter removal.[167] Therefore, SIPS's biofiltration recommendation for the SDWS with the more contaminated source water (SDWS#2) and for the SDWS with the highest treatment goals (SDWS#3) is consistent with current research results and with changing technology implementation trends.

Nonozonated biofiltration, though, is more common at larger drinking water systems. For SDWS#2 and SDWS#3, SIPS also gave high performance scores to conventional filtration, which is also less common at SDWSs but widely implemented at larger systems because it is the default treatment train when treatment goals and source waters require multiple barriers to contaminant breakthrough.[84] Although SIPS recommended treatment processes that are more common at larger systems, many SDWSs will have to start considering more complicated treatment technologies due to more stringent regulations and deteriorating source waters. Also, SIPS's recommendations are supported by numerous decision criteria and quantitative comparisons. This can help SDWSs understand their options beyond traditional small system technologies by illuminating commonly unknown or overlooked benefits and disadvantages of multiple treatment alternatives.

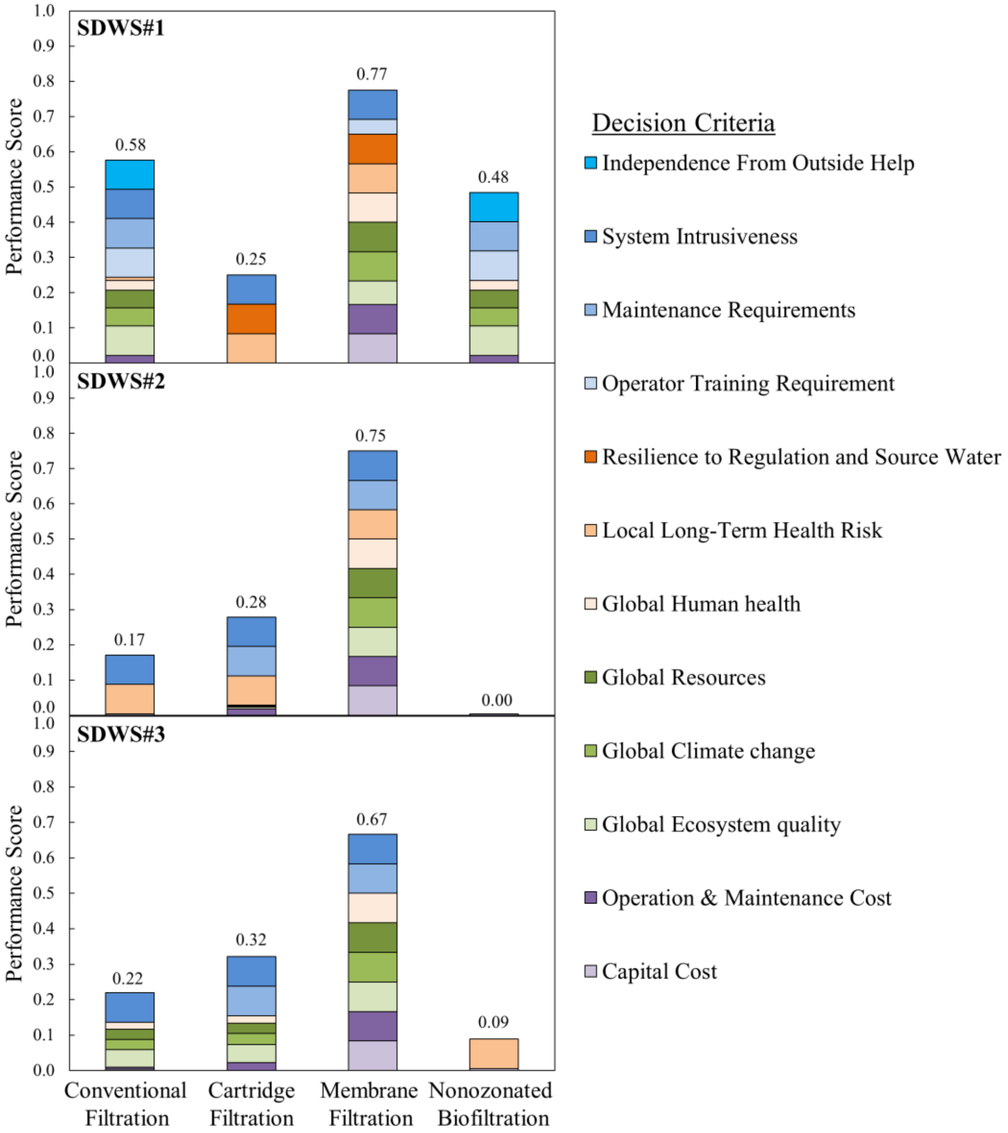


Figure 4.2. SIPS performance scores for the four filtration-focused treatment train alternatives where each plot (a, b, c) represents a different SDWS (Table 4.2). The stacked columns show each criterion's score's contribution to the performance score under an equal criteria weighting scheme.

4.3.1.2 Disinfection Selection Example

SDWS#1 was also used to compare additional disinfection-focused alternatives with the same filtration process (cartridge filtration) but different disinfection options: (1) chlorine disinfection using a concrete contact zone, (2) chlorine disinfection using a plastic pipe contact zone, (3) chlorine disinfection using a plastic baffled concrete zone, and (4) UV disinfection (which included chlorine addition for a distribution system chlorine residual). For this fourth scenario, SIPS recommended using chlorine disinfection (Figure 4.3). The plastic pipe contact zone alternative was most preferred in all criteria except for *local long-term health risk* and *system intrusiveness*. The alternatives that used concrete contact zones had the worst performance in *capital cost* and global pollution criteria. The use of plastic baffles reduced contact zone concrete use, so that alternative performed slightly better than the fully concrete contact zone but still worse than the plastic pipe alternative. The UV disinfection alternative used a minimal amount of chlorine and therefore performed best in *local long-term health risk* (because it had the lowest disinfection byproduct formation) and *system intrusiveness* (because it could improve the drinking water's taste and odor). Since this alternative required two (UV and chlorine) disinfectants and high energy and material requirements (for the UV operational energy and stainless-steel vessel), this alternative had greater maintenance, complexity, and costs that resulted in the worst performance in *operation & maintenance cost, maintenance requirements,* and all global pollution criteria.

Similar to the filtration-focused scenarios, SIPS's recommendations for this disinfection-focused scenario also had engineering relevance. SIPS's recommendation of chlorine disinfection matches U.S. treatment conventions, where chlorine is used ubiquitously for primary and secondary disinfection [189]. Regulations have historically benefitted the selection of chlorine over UV because chlorine is much more effective at achieving virus reduction[191] and because a chlorine residual is required.[132] Additionally, SIPS recommended using a plastic pipe contact zone, which is seen at SDWSs[192] mostly because it is easier to implement and less expensive than common large system contact zones (e.g., concrete). Therefore, SIPS's recommendations were consistent with current SDWS disinfection use and the scenario's selected regulatory structure.

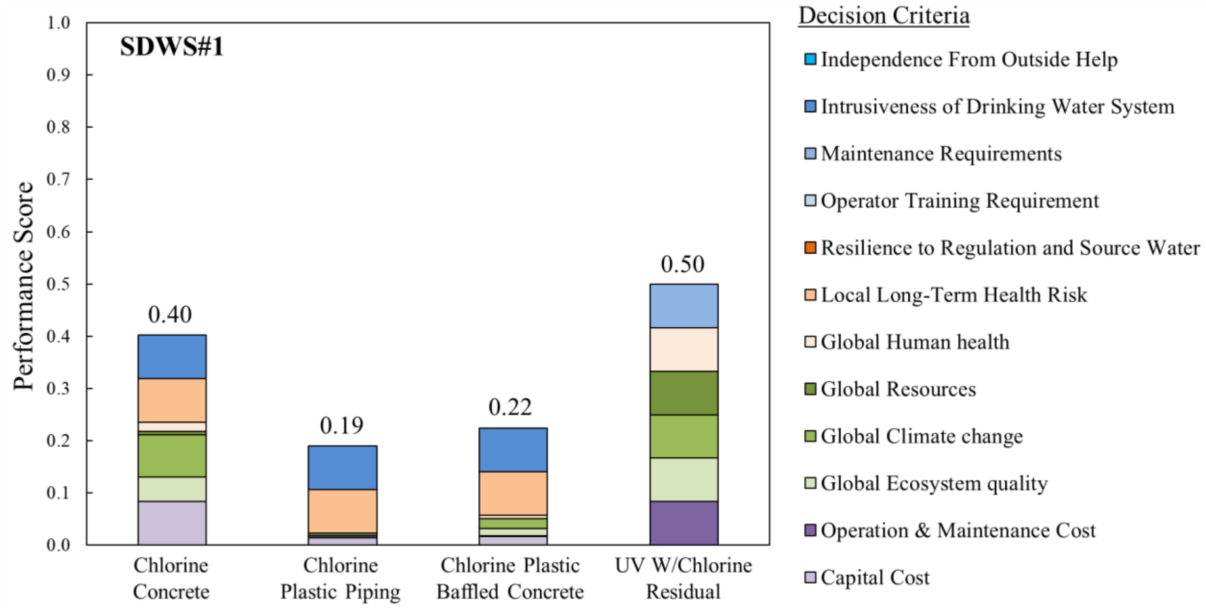


Figure 4.3. SIPS performance scores for the four disinfection-focused treatment train alternatives compared for SDWS#1 (Table 4.2). The stacked columns show each criterion's score's contribution to the performance score under an equal criteria weighting scheme.

4.3.1.3 Additional SIPS Applications

In addition to these scenarios, SIPS has applicability to an even wider variety of scenarios and SDWSs. For example, SIPS can represent any country's treatment objectives because the user can manually specify disinfection doses to make comparisons on a case-by-case basis, such as when a groundwater SDWS can use UV as the sole disinfectant; and can evaluate filtration alternatives without disinfection in cases where solids removal is desired but pathogen reduction is not a concern. Since SIPS uses multiple decision criteria, the potential benefits of less-common treatment processes can be better realized. Overall, SIPS can evaluate any combination of coagulation, filtration, disinfection, and pH adjustment processes for a wide variety of source waters, specific design parameters, and regulatory requirements.

4.3.2 Navigating Trade-offs with SIPS

Each of the 12 SIPS decision criteria, identified by SDWS stakeholders, are needed to fully understand the benefits, disadvantages, and trade-offs of different treatment processes. Every criterion contributed to the final treatment recommendations and provided insights into why one alternative may be preferred over another in each scenario. Also, many criteria and treatment processes would not have been considered without the SIPS approach. For example, there were trade-offs for cartridge filtration between the *resilience to regulation and source water* criterion and the affordability and global pollution criteria; this allows SDWSs the ability to consider long-term versus short-term criteria that may not have been compared otherwise. In another example, most SDWSs would not consider nonozonated biofiltration, but SIPS identified important benefits of this more complicated treatment process. Since there will always be trade-offs, they should be captured and specific to each SDWS. Some SDWS stakeholders, though, may not know how to resolve the differences between treatment alternatives. The SIPS aggregation of results helps users navigate those trade-offs, between and within alternatives, in a systematic and quantitative manor. In particular, the SIPS performance single-scores include stakeholder preferences that are crucial for recommending alternatives that are most appropriate for a given SDWS and its stakeholders.

SIPS updates its recommendations, as needed, based on the SDWS stakeholders and preferences. To exemplify this, three different weighting schemes were applied to the three hypothetical filtration-focused scenarios (Figure 4.4): equal criteria weights, health-focused weights, and pollution-focused weights (Table C16). For SDWS#1, the SIPS recommendation changed from cartridge filtration under equal weights to nonozonated biofiltration under health-focused weights; this was mostly because the nonozonated biofiltration alternative has a high ability to reduce disinfection byproducts and performed best in *system intrusiveness* in that scenario. For SDWS#2, the recommendation between equal and health-focused weights did not change because it was already the best for human health criteria. For SDWS#3, the recommended alternative changed from nonozonated biofiltration under equal weights to conventional filtration under health-focused weights because the conventional filtration alternative outperformed other alternatives in human health criteria in that scenario. When pollution-focused weights were used, the original recommendations under equal criteria weights were maintained for all three SDWSs. This was because the global pollution criteria were not the major factors keeping one alternative from being better than another in these scenarios.

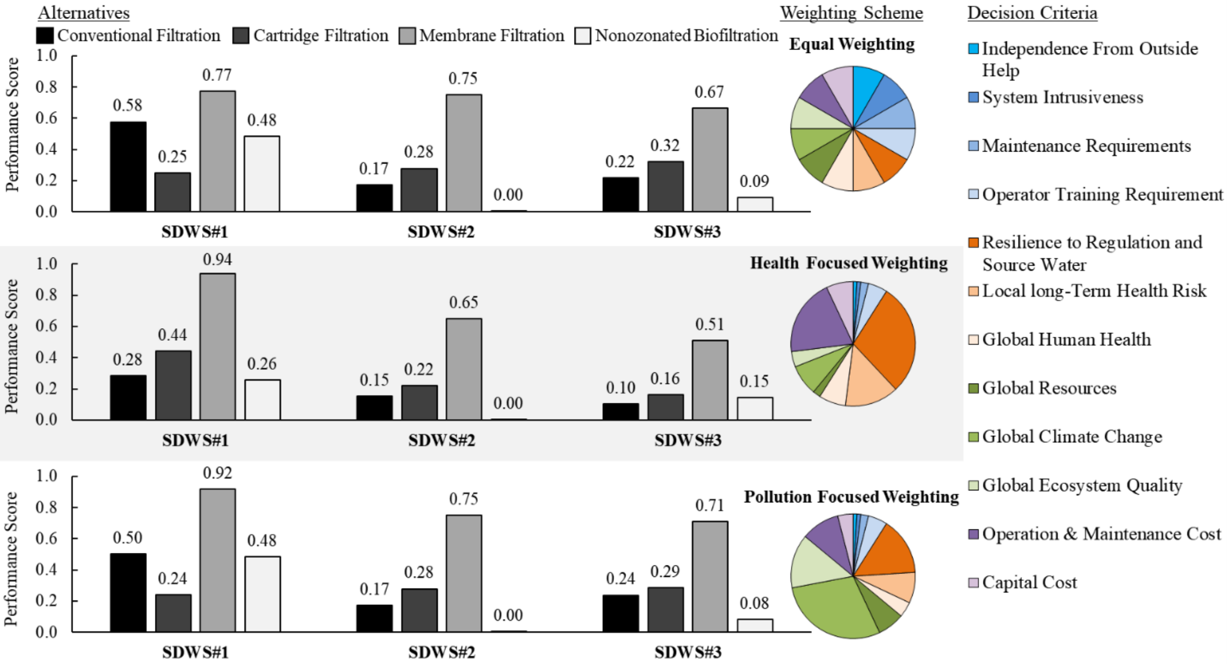


Figure 4.4. Performance scores for each filtration-focused alternative under different criteria weighting schemes (vertical groupings) for all three SDWSs (horizontal groupings) (Table 4.2).

4.4 Conclusions

SIPS was developed to be a universal framework that can be used by any SDWS. It can be used for a large range of source water qualities, treatment standards, and stakeholders (e.g., community members, engineers, operators, regulators, financiers). The SIPS framework can also be helpful in developing frameworks specific to larger systems. The framework can be used over time, even with changing treatment conditions and technology development. SDWS stakeholders, such as community members, engineers, operators, regulators, and financiers can use SIPS to make better, more informed treatment decisions that will improve drinking water quality and system sustainability for the millions of people globally who are served by drinking water systems with limited resources.

Chapter 5. Evaluating Small System Treatment Processes for Different Source Water Qualities, Regulations, and Stakeholder Preferences

The following manuscript and appendix are in preparation for submission for publication: Jones, C. H.; Cook, S. M. Evaluating Small System Treatment Processes for Different Source Water Qualities, Regulations, and Stakeholder Preferences.

5.1 Introduction

Drinking water systems are tasked with providing safe drinking water to customers even as they face increasingly stringent regulations and deteriorating source water quality.[5–8] To manage these challenges, many will be required to update their treatment practices. Drinking water decision-support has mostly been provided through case studies on larger systems. However, small drinking water systems, those serving less than 10,000 people, have different challenges than large systems due to resource limitations. In particular, they struggle with choosing the most appropriate treatment process to improve drinking water quality. Small systems will also require treatment updates to ensure they can provide high quality water to their consumers, which account for up to 2.5 billion people globally,[1] and they require decision-support to do so.

Previous decision-support efforts have not been comprehensive in the decision criteria that have been evaluated. One study used life cycle assessment (LCA) to evaluate small system treatment alternatives for reducing disinfection by-products,[41] and two others compared filtration and disinfection options while considering multiple characteristics of potential small systems.[91,167] However, small systems lack the technical, managerial, and financial resources that are needed to make effective decisions that meet their individual needs so other criteria, economic and social, need to be considered beyond environmental impacts.[4] Another study did evaluate multiple decision criteria, but did not include environmental criteria.[48] So, there is a need for data that will help small systems make a comprehensively sustainable decision. A previously developed small system decision-support tool: Small DrInking Water Plant Sustainability Support (SIPS), can provide a wide range of small systems with decision-support using 12 small system specific decision criteria that span all three pillars of sustainability (Chapter 4). SIPS also employs life cycle thinking and multiple-criteria decision analysis (MCDA) to ensure treatment alternative comparisons are being made on an equal basis.

Case studies have been effective for providing decision-making insights at drinking water systems for reducing environmental impacts using LCA,[12,13,23–30,32,33,14,34–36,15–18,20–22] optimizing treatment processes,[8] and evaluating resilience.[193] However, previous case studies have mostly generated site-specific results and only focused on larger systems, so their results do not provide insights or support for small system decisions. Also, a case study approach is infeasible to provide insights for the tens of thousands of small drinking water systems because small drinking water systems are diverse,[3] each with distinct treatment processes, source water quality, regulatory requirements, and stakeholder preferences. However, insights can be gained using modeling/simulation approach, especially one that incorporates small system treatment alternatives and system specific characteristics. This approach has proven effective at large drinking water systems for gathering simulated data to identify optimization opportunities under future conditions.[8]

Small systems use a wide range of treatment processes to achieve different treatment goals; some only require disinfection while others must use more complex processes such as membrane filtration. They have limited resources to update their treatment processes with changing conditions, which is why many small systems often use simple filtration options, such as cartridge filtration.[91] Complications stemming from unique source water quality and regulatory requirement characteristics makes selecting treatment processes unclear because what works for one system may not work for another. Small system literature has shown that treatment process selection can depend on system specific regulatory scenarios,[91,167] and source waters,[167] but needs to be expanded to incorporate more decision criteria and stakeholder preferences. SIPS has the capacity to quantitatively assess treatment processes commonly used at small systems, using combinations of filtration and disinfection options, for diverse source water quality, regulatory scenario, and stakeholder preference characteristics.

This study was designed to simulate decision-making results for multiple treatment alternatives and a wide range of small drinking water systems with different source water quality, stakeholder preferences, and regulatory requirements while also considering uncertainty. Differences between small systems' specific characteristics were used to simulate diverse small systems. Characteristic thresholds were identified for each treatment alternative to specify when specific treatment processes were most recommended. Ultimately, a holistic understanding of why some treatment alternatives were preferred for small systems with different characteristics was uncovered. The tens of thousands of small systems that cannot afford to perform case studies can use the findings of this study to inform sustainable decision-making while improving drinking water quality.

5.2 Methods

5.2.1 SIPS Decision-making Tool

SIPS uses a quantitative scoring system (i.e., identifying the most preferred to least preferred option) to evaluate drinking water treatment trains using LCA and MCDA methods. SIPS has 7 steps (Figure 4.1): (1) user inputs source water quality and regulation data, then SIPS sets the functional unit (i.e., the basis for an equal life cycle comparison); (2) user chooses treatment train alternatives by selecting a combination of treatment processes, then SIPS conducts a life cycle inventory analysis where the types and amounts of materials, chemicals, and energy required by each alternative are determined using drinking water process models and design algorithms; (3) SIPS calculates criteria scores from each alternative's inventory data by applying a quantitative evaluation method for each criterion; (4) SIPS normalizes criteria scores; (5) user defines criteria weights; (6) SIPS calculates aggregated single-scores for each alternative; and (7) SIPS recommends a treatment train alternative based on the single-scores, and the user can view the score breakdowns to determine if iteration or re-evaluation is required.

SIPS can evaluate 112 unique treatment trains using combinations of the following treatment processes: 2 coagulation chemical options, 7 filtration options, 4 disinfection options, and 2 pH adjustment chemical options. There are 7 source water quality inputs, including: alkalinity, bromide, pH, temperature, TOC, turbidity, and ultraviolet transmittance (UVT). There are 12 decision criteria; each must be assigned a weight between 0% to 100% (all summing to 100%). Specific regulatory goals may also be set for turbidity (<1 NTU or >1 NTU), TOC removal (none, flat percentage removal, or enhanced coagulation rule[194]), and pathogen reduction (a specific dose can be chosen to meet specific pathogen reduction or U.S. requirements[84]).

5.2.2 Treatment Train Selection

The main treatment decisions made at small systems (e.g., most permanent investment) are for filtration and disinfection. More modifiable decisions can be made for chemical selection such as what type of coagulant and pH adjustment chemicals are used. This study evaluated the filtration and disinfection treatment options available in SIPS to represent the most permanent investment decisions for 28 treatment trains. The coagulation and pH adjustment chemicals were set as alum and caustic, respectively; the impact of choosing different chemicals for either are mostly known[22,167] and will be discussed qualitatively in the results section. Of the 12 SIPS decision criteria, there are two (*system intrusiveness* and *independence from outside help*) that require the user to score each treatment train. Each treatment train was assigned the same *system intrusiveness* criterion score because scores are expected to change based on a system by system basis even when the same treatment processes are considered. The same approach was used for the *independence from outside help* criterion scores; each treatment train was assigned the same score.

5.2.3 Main Characteristics of Hypothetical Small Systems

Monte Carlo simulations were used to evaluate treatment alternatives while determining the impact of three major site-specific small system characteristics: regulatory requirements, source water quality, and decision criteria weights (i.e., stakeholder preferences). For each simulation, the resulting SIPS performance scores were used to assign each treatment alternative a rank; for a given simulation, a rank of 1 was assigned to the highest recommended (best performing) alternative, and a rank of 28 was assigned to the least recommended (i.e., worst performing) alternative. Details for each simulation were used to define SIPS's functional unit: to treat *0.1* Mm^3/yr (0.07 mgd) of a *specified source water* to meet a *specified regulatory scenario*, while ensuring a distribution system pH of 8.2,[167] over a *40-year timeframe*. This assured that each simulation resulted in an accurate comparison of the 28 treatment train alternatives.

There are numerous regulatory requirements that systems fall under, and this study considered three common scenarios to evaluate what treatment processes are recommended for a small system that has to update their treatment. The first regulatory scenario is for a case where a system has to add filtration, turbidity treatability is easy, and there are no TOC removal requirements. The second regulatory scenario represents a system that has difficult to treat turbidity and it has to reduce its source water TOC by 30% to prevent disinfection by-product precursors no matter the difficulty. The third regulatory scenario represents a system that has difficult to treat turbidity but is now beholden to U.S. TOC removal requirements (Table A1), which does account for the treatability of the source water.

First, 10,000 different source waters were evaluated within each regulatory scenario. A source water was defined by 6 water quality parameters: alkalinity, bromide, pH, temperature, TOC, and

UVT. To generate each source water, Monte Carlo sampling of uniform distributions of each source water quality parameter was conducted. While SIPS also includes turbidity as a source water quality parameter, turbidity treatability was set based on the regulatory scenario since turbidity removal cannot currently be predicted with high resolution. The range of each parameter's uniform distribution was based on the valid range of the SIPS model (Table C1). Most of these ranges were based on real utility data,[89] so it was expected that the source waters being generated represent all possible source waters that could be treated at a small system. Also, every combination of minimum and maximum source water parameters' values was generated first to check that those "extreme" source waters resulted in feasible treatment design (e.g., coagulant doses did not exceed 122 mg/L, which was close to the highest dose at any system[95]). To minimize unrealistic source waters, waters with a combined low pH (<7) and high alkalinity (>100 mg/L)[189] and waters in Regulatory Scenario#1 with a TOC greater than 2 mg/L[108] were removed from the dataset During this source water analysis, the criteria weighting scheme was set to equal criteria weights.

Next, 10,000 different weighting schemes were evaluated within each regulatory scenario. A criteria weighting scheme was defined as a set of weights (between 0% and 100%) for each of the 12 decision criteria. To generate different weighting schemes, Monte Carlo sampling of uniform distributions from integer values of 0 to 10 for each criterion's weight was first conducted and then each was divided by the sum of all 12 values to generate criterion weights as a percentage (0% to 100%) out of 100%. During this criteria weighting scheme analysis, the source water represented national average source water quality values of 93 mg/L as $CaCO_3$ alkalinity, 0.069 mg/L bromide, 7.5 pH, 15°C temperature, 2.8 mg/L TOC, and 82% UVT. In addition to these simulated weighting schemes, small system stakeholder preferences were also identified by using an analytical hierarchy process (AHP) survey developed for SIPS. Data collection for stakeholder surveys followed the protocol approved by The University of Colorado Boulder's Institutional Review Board # 17-0333.

5.2.4 Uncertainty

To help evaluate the impact of uncertainty on the results, 10,000 different sets of design parameters were evaluated within each regulatory scenario. A set of design parameters was defined by 59 uncertainty parameters (e.g., filter operational head losses, minimum coagulant doses for turbidity, etc.). To generate each set of design parameters, Monte Carlo sampling was conducted using uniform distributions for parameters with minimum and maximum data and triangular distributions for parameters that had minimum, most common, and maximum data (Table C4). Uncertainty was considered to determine the impact of the full range of model values instead of assuming that default design parameter values are representative of all small systems. During this uncertainty assessment, the source water represented national average source water quality decision criteria and criteria weighting scheme was set to equal criteria weights.

5.2.5 Correlating Results

Crystal Ball^{TM} software, which is compatible with the spreadsheet-based SIPS tool, was used to conduct the Monte Carlo simulations. The main output of each simulation were treatment alternative ranks. Spearman's rank correlation coefficients were also quantified. These correlation coefficients were used to identify which input parameters had the greatest effect on each treatment alternatives' rank. The 10,000 Monte Carlo simulations for each analysis (source water, weighting schemes, and uncertainty) was enough to ensure reproducible results and was beyond what is recommended for the Spearman's rank correlation coefficient analysis.[195] A correlation coefficient indicated lack of sensitivity when it was below 0.3 ($\rho < |0.3|$); moderate sensitivity when it was between 0.3 and 0.8 ($|0.3| < \rho < |0.8|$); and high sensitivity when it was above 0.8 ($\rho > |0.8|$).[167]

Since these correlation coefficients indicate monotonic relationships between inputs and outputs, results were also evaluated visually (each output vs. each input) to verify monotonic relationships and identify non-monotonic relationships.[195]

5.3 Results and Discussion

5.3.1 Design Parameter Uncertainty

Regulatory Scenario#1. For a system with national average source water quality and equal criteria weights, under Regulatory Scenario#1 (turbidity treatability was easy, no TOC removal requirements), the most recommended alternative was slow sand filtration coupled with chlorine disinfection with plastic piping (Figure D.1). This treatment train, compared to the 27 others, had the lowest operational energy use and treatment complexity. Each treatment alternative's rank was most dependent on its filtration process (Section D1.1). The rank order of filtration options, from best to worst, was slow sand filtration and cartridge filtration (both first), nonozonated biofiltration, conventional filtration, bag filtration, membrane filtration, and ozonated biofiltration. A treatment alternative's rank was also dependent on its disinfection process. The rank order of disinfection options, from best to worst, were chlorine using plastic piping, chlorine using a concrete tank, UV with a chlorine residual, and chlorine using a concrete tank with steel baffles.

Some of these ranks, though, were sensitive to uncertainty (Section D1.3). For the filtration options, slow sand's rank was moderately sensitive to hydraulic loading rate because it impacts the amount of materials required to construct the filter. Slow sand filtration was usually the most preferred option (rank = 1) but had its worst rank (of 5 out of 7) when the hydraulic loading rate was greater than 0.25 m/hr. The ranks of cartridge, and membrane filtration options were both highly sensitive to filter head loss, which determined operational energy use. Cartridge filtration could have the best rank of 1 when its head loss was below 10 m. Membrane filtration could have its best possible rank of 4 when its head loss was below 9 m. Bag filtration's rank was moderately sensitive to filter head loss. Bag filtration could have the best rank of 1 when head loss was below 9 m for the first stage and below 8 m for the second stage.

The ranks of nonozonated biofiltration and conventional filtration were both moderately sensitive to the minimum turbidity coagulant dose. Higher doses increased cost and pollution, due to chemical production. Both could have their best possible ranks, of 1 and 2, respectively, at doses below 14 mg/L. Ozonated biofiltration's rank was moderately sensitive to ozone dose, which determined energy use for ozone generation. Ozonated biofiltration could have its best possible rank of 5 for doses below 0.6 mg O_3/mg TOC. The ranks of the chlorine disinfection options were not sensitive to uncertainty. The UV disinfection's rank was moderately sensitive to UV vessel head loss (ρ=0.30), and UV could have its best possible rank of 2 when the head loss was below 1 m. Also, several treatment options were so similar to each other that a change in one's rank would usually correspond with a change in another (Table D.1); since these correlations are not based on treatment mechanisms or uncertainty, these relationships will be evaluated during the source water and criteria weighting scheme analyses.

Regulatory Scenario#2. For the same system under Regulatory Scenario#2 (turbidity treatability was difficult, 30% TOC removal was required), the most recommended alternative was nonozonated biofiltration coupled with UV disinfection with a chlorine residual (Figure D.2). The recommended filtration option changed in this scenario because coagulation had to be added to the slow sand, cartridge, bag, and membrane filtration options to meet TOC removal goals and remove turbidity that would clog those filters. As a result, each treatment alternative's rank was still found to be most dependent on its filtration process, but the new rank order, from best to worst, was nonozonated biofiltration, slow sand filtration, conventional filtration, cartridge filtration, ozonated biofiltration, bag filtration, and membrane filtration. The rank order of disinfection options were the same as for Regulatory Scenario#1.

For all but one uncertainty parameter, each filtration and disinfection process was sensitive to the same uncertainty parameters as in Regulatory Scenario#1. The one exception was that no filtration option was sensitive to the minimum turbidity coagulant dose uncertainty parameter in this scenario; this is because that dose was never high enough to achieve this scenario's TOC removal requirement. The parameter values at which different filtration options were preferred changed slightly, but the values were the same for disinfection options between this scenario and Regulatory Scenario #1 (Table 5.1).

Regulatory Scenario #3. For the same system under Regulatory Scenario#3 (turbidity treatability was difficult, U.S. regulatory TOC removal was required), the most recommended alternative was nonozonated biofiltration coupled with chlorine disinfection with plastic piping (Figure D.3). The mechanisms behind rank ranges were the same as in Regulatory Scenario#2. For all but one uncertainty parameter, each filtration and disinfection process was sensitive to the same uncertainty parameters and had similar parameter values at which different options were preferred between this scenario and Regulatory Scenario #2 (Table 5.1). The one exception was that the ranks of disinfection options were moderately sensitive to the minimum turbidity coagulant dose. Specifically, chlorine using plastic piping, chlorine using a concrete tank, and UV with a chlorine residual were moderately sensitive to this dose. An increased coagulant dose resulted in a lower pH before disinfection; this resulted in smaller chlorine concentration times time (CT) values, so fewer materials were needed for the contact zones in each option.

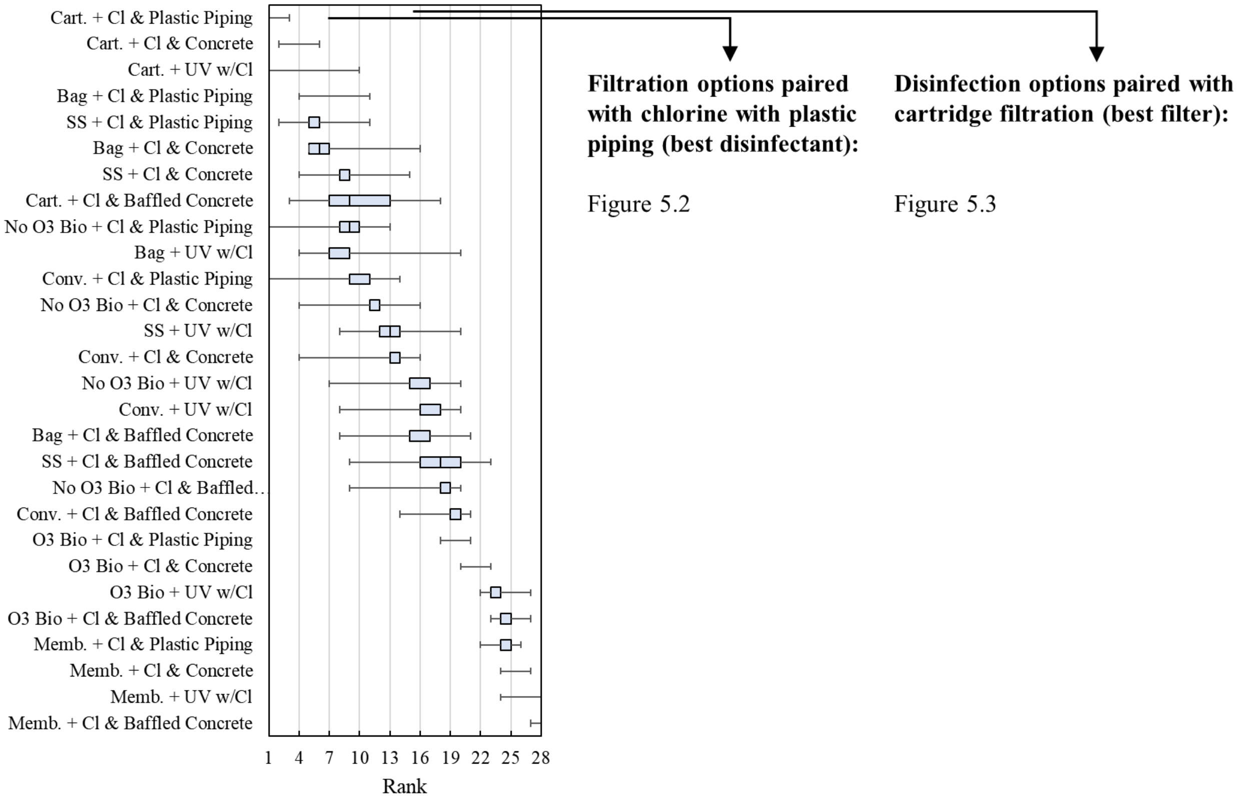
Key Takeaways. In summary, ranks of treatment alternatives changed across regulatory scenarios because some filtration options required coagulant addition when turbidity became more difficult to treat and TOC removal was required. It is likely that TOC and turbidity will increase or decrease together.[189] Also, treatment alternatives' ranks were most dependent on the filtration process and most sensitive to filter parameter uncertainty. The selection, design, and operation of a treatment train should be informed by these results, especially by considering the parameter values at which treatment options were found to be the most preferred (i.e., sustainable). Since those values are representative of a system with national average source water and equal criteria weights, each small system may want to run simulations to provide data that is more representative of their characteristics. The following sections evaluated the impact of source water and criteria weighting schemes on treatment train recommendations. Those analyses used the default values for each uncertainty parameter. Default values were chosen based on the literature and engineering knowledge and they fall in the range of values that enable the best possible ranks for each technology.

Table 5.1. For each regulatory scenario, comparison of best possible ranks and most preferred design thresholds for filtration and disinfection processes for the uncertainty analysis. Values corresponding with the best possible ranks are representative of a system with national average source water quality and equal criteria weights. Filtration and disinfection options were isolated by using the same disinfection (chlorine with plastic piping) or filtration (cartridge filtration) option, respectively, so these results show a subset of all treatment trains to serve as an example of rank range mechanisms. N/A is not applicable.

	Regulatory Scenario#1		**Regulatory Scenario#2**		**Regulatory Scenario#3**	
Treatment Process	Best Possible Rank	Corresponding Values	Best Possible Rank	Corresponding Values	Best Possible Rank	Corresponding Values
Filtration Options (Chlorine W/Plastic Piping was the Disinfectant after each Filter)						
Bag Filtration	1	Operational head loss < 9 m stage 1, < 8 m stage 2	4	Operational head loss < 15 m stage 1, < 14 m stage 2	4	Operational head loss < 15 m stage 1, < 12 m stage 2
Cartridge Filtration	1	Operational head loss < 10 m	2	Operational head loss < 2.3 m	2	Operational head loss < 2.3 m
Conventional Filtration	2	Minimum turbidity coagulant dose < 14 mg/L	2	N/A	2	N/A
Membrane Filtration	4	Operational head loss < 9 m	4	Operational head loss < 10 m	4	Operational head loss < 8 m
Nonozonated Biofiltration	1	Minimum turbidity coagulant dose < 14 mg/L	1	N/A	1	N/A
Ozonated Biofiltration	5	Ozone dose < 0.6 mg O_3/mg TOC	4	Ozone dose < 1.2 mg O_3/mg TOC	4	Ozone dose < 1.4 mg O_3/mg TOC
Slow Sand Filtration	1	N/A	1	Hydraulic loading rate > 0.2 m/hr	1	Hydraulic loading rate > 0.15 m/hr
Disinfection Options (Cartridge Filtration was the Filter used Before each Disinfectant)						
Chlorine W/Concrete	2	N/A	2	N/A	2	N/A
Chlorine W/Concrete & Steel Baffles	3	N/A	3	N/A	3	N/A
Chlorine W/Plastic Piping	1	N/A	1	N/A	1	N/A
UV W/Chlorine Residual	2	Vessel head loss < 1 m	1	Vessel head loss < 1 m	1	N/A

5.3.2 Source Water Analysis

Regulatory Scenario#1. Under Regulatory Scenario#1, 10,000 different source waters were considered, using equal criteria weights. The most often recommended treatment train was cartridge filtration coupled with chlorine with plastic piping (Figure 5.1). To show how a small system could use this information filtration and disinfection options were isolated to better see how source water impacts treatment ranks. The rank order of filtration options, from best to worst, were: cartridge filtration, bag filtration, slow sand filtration, nonozonated biofiltration, conventional filtration, ozonated biofiltration, and membrane filtration. The rank order of disinfection alternatives in this scenario was: chlorine with plastic piping, chlorine with concrete. This scenario's filtration rank order, based on default uncertainty parameter values, was different from the uncertainty assessment rank order because the expected head loss is significantly less than the worst possible head loss for cartridge filtration, and bag filtration (3.5 m versus 21.1m).



Filtration options paired with chlorine with plastic piping (best disinfectant):

Figure 5.2

Disinfection options paired with cartridge filtration (best filter):

Figure 5.3

Figure 5.1. Rank ranges for all 28 treatment alternatives. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Based on these results filtration and disinfection options were isolated to identify rank range mechanisms. Note: when the min max or quartiles are not visible one or many of those are the same value.

The rank ranges of the filtration options were mostly due to source water pH (Figure 5.2 and Table D.4). Cartridge filtration's rank was insensitive to source water quality; it was the top ranked option for all source waters considered. The ranks of both bag and slow sand filtration options were moderately sensitive to pH because the relative difference between chlorine doses coupled with each filter option became slightly smaller as absolute doses became higher. Slow sand filtration was more preferred for pH < 7, which is when the benefits of reducing chlorine demand were realized. Ozonated biofiltration and membrane filtration ranks were insensitive to source water.

Nonozonated biofiltration was impacted by source water pH and got its best possible rank of 1 when pH < 6.25 because there were benefits to having a smaller contact zone after nonozonated biofiltration (due to coagulation lowering the pH), and more relative benefits to reducing chlorine demand compared to other treatment alternatives. Also, when pH > 8.75, using nonozonated biofiltration did not require as much relative pH adjustment chemical because the coagulant dose did not bring the pH down far below the pH adjustment target of 8.2. Conventional filtration also had its best possible rank of 1 in the same ranges and for the same reasons as nonozonated biofiltration.

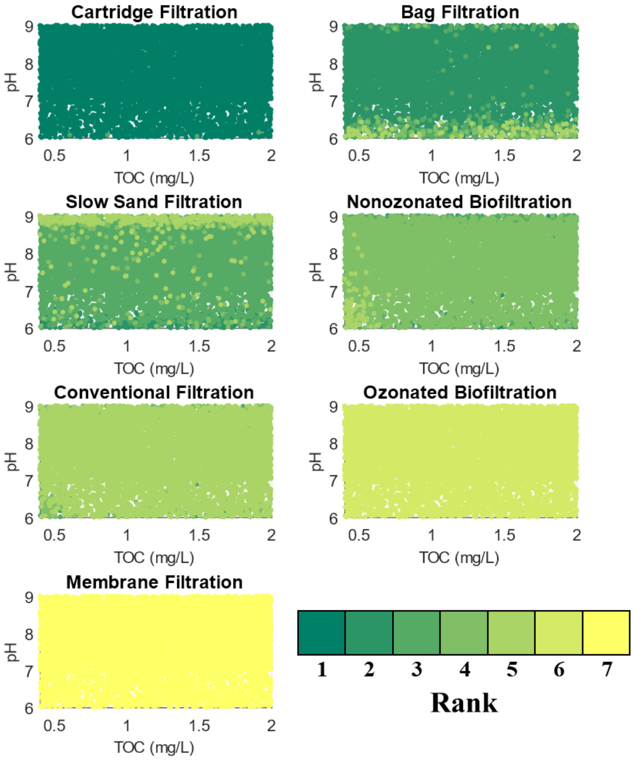


Figure 5.2. As representative examples, the source water effects on filtration ranks were evaluated for all filtration options in Regulatory Scenario#1 when coupled with chlorine with plastic piping, which was the top ranked disinfection option (Figure 5.1). While each point represents a combination of six source water parameters, the multidimensional data is being shown as only a function of two parameters, one of which the ranks were most sensitive to, pH.

The rank ranges of the disinfection options mostly depended on source water pH and temperature (Figure 5.3 and Table D.4). These parameters had the largest influence on the size of the chlorine contact zone (due to CT requirements),[148] chlorine dose, and pH adjustment dose. The chlorine with plastic piping option had its best possible rank of 1 when the pH < 8 and temperature>5°C because its contact zone was the least expensive option with the fewest environmental impacts when higher CT values were required. The chlorine using concrete option had its best rank of 2 when pH < 6.5 and temperature > 15°C because the contact zone could be small enough to reduce associated costs and environmental impacts to make this option better than the UV with a chlorine residual option. UV with a chlorine residual had its best rank of 1 when pH > 8 and temperature < 5°C because contact zones had to be designed large enough to achieve CT, which is not required when using UV. Chlorine disinfection using concrete and steel baffles had its best rank of 3 when pH < 6.25 and temperature > 20°C because the contact zone materials were minimized due to low CT requirements.

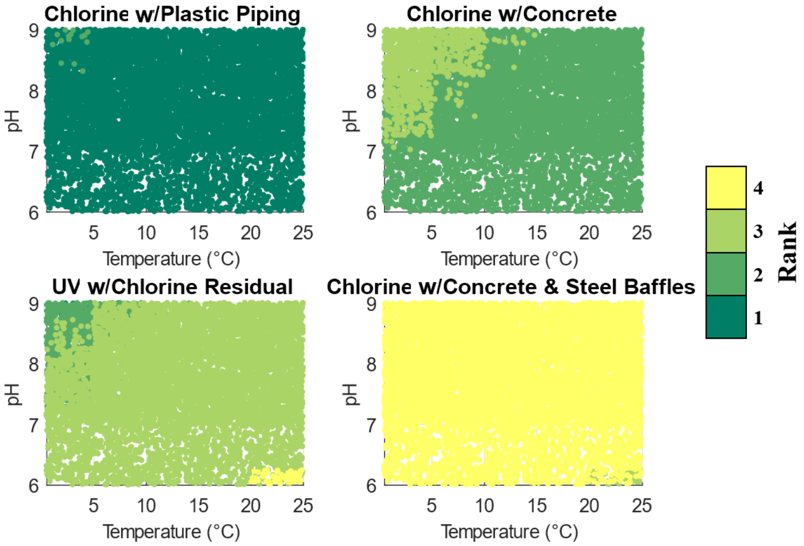


Figure 5.3. As representative examples, the source water effects on disinfection ranks were evaluated for all disinfection options in Regulatory Scenario#1 when coupled with cartridge filtration, which was the top ranked disinfection option (Figure 5.1). While each point represents a combination of six source water parameters, the multidimensional data is being shown as only a function of the two parameters to which the ranks were most sensitive to, pH and temperature.

Table 5.2. For each regulatory scenario, comparison of best possible ranks and most preferred design thresholds for filtration and disinfection processes for the source water quality analysis. Values corresponding with the best possible ranks are representative of a system with expected design parameters and equal criteria weights. Filtration and disinfection options were isolated by using the same disinfection (chlorine with plastic piping) or filtration (cartridge filtration) option, respectively, so these results show a subset of all treatment trains to serve as an example of rank range mechanisms. N/A is not applicable.

Treatment Process	**Regulatory Scenario#1** Best Possible Rank	Corresponding Values	**Regulatory Scenario#2** Best Possible Rank	Corresponding Values	**Regulatory Scenario#3** Best Possible Rank	Corresponding Values
Filtration Options (Chlorine W/Plastic Piping was the Disinfectant after each Filter)						
Bag Filtration	2	pH > 7	5	TOC > 2 mg/L	5	TOC > 2 mg/L
Cartridge Filtration	1	N/A	4	TOC > 1 mg/L	4	TOC > 1 mg/L
Conventional Filtration	1	pH < 6.25, or pH > 8.75	1	Alkalinity < 50 mg/L as $CaCO_3$	1	N/A
Membrane Filtration	7	N/A	6	N/A	6	N/A
Nonozonated Biofiltration	1	pH < 6.25, or pH > 8.75	1	Temp > 10°C	1	Temp < 10°C, TOC < 2.5 mg/L
Ozonated Biofiltration	6	N/A	4	TOC < 1 mg/L	4	TOC < 1 mg/L
Slow Sand Filtration	2	pH < 7	1	Temp < 10°C	1	Temp < 10°C, TOC < 2.5 mg/L
Disinfection Options (Cartridge Filtration was the Filter used Before each Disinfectant)						
Chlorine W/Concrete	2	pH < 7, Temp > 10°C	2	Temp > 10°C	2	Temp > 15°C
Chlorine W/Concrete & Steel Baffles	3	pH < 6.25, Temp > 20°C	3	pH < 6.25, Temp > 20°C	3	pH < 6.25, Temp > 20°C
Chlorine W/Plastic Piping	1	pH < 8, Temp > 5°C	1	N/A	1	Temp > 20°C
UV W/Chlorine Residual	1	pH > 8, Temp < 5°C	1	Temp < 10°C	1	pH > 7, Temp < 10°C

Regulatory Scenarios#2 and #3. Under Regulatory Scenarios #1 and #2, 10,000 different source waters were considered, using equal criteria weights. The most often recommended treatment train was nonozonated biofiltration coupled with chlorine with UV with chlorine residual for Regulatory Scenario#2 (Figure D.5) and nonozonated biofiltration coupled with chlorine with plastic piping for Regulatory Scenario#3 (Table D.6). UV with a chlorine residual was slightly preferred to chlorine with plastic piping in Regulatory Scenario#2 because in this scenario UV's benefit of having the lower disinfection by-product formation than chlorine options, when combined with nonozonated biofiltration, was realized. Filtration and disinfection rank results were isolated to identify how source water impacts treatment ranks. The rank order of filtration options, from best to worst, were: nonozonated biofiltration, slow sand filtration, conventional filtration, cartridge filtration, bag filtration, ozonated biofiltration, and membrane filtration. The rank order of disinfection options changed slightly based on what filtration options were paired with them, but the underlying mechanisms behind rank ranges due to source water quality was the same.

The rank ranges of filtration options for Regulatory Scenario#2 were mostly due to source water temperature and TOC, which was different than for Regulatory Scenario#1 (Figure 5.4, Table D.5, and Table D.6). Nonozonated biofiltration had its best rank of 1 when temperature > 10°C because at higher temperatures nonozonated biofiltration was more effective at removing TOC. Slow sand filtration had its best rank of 1 at temperatures < 10°C because slow sand filtration data's low resolution meant its TOC removal was better than nonozonated biofiltration at that temperature. Conventional filtration had its best rank of 1 at lower alkalinities because as alkalinity gets low enough, all of it is consumed by coagulant addition and less end of plant pH adjustment chemical was needed compared to if there was still buffering capacity in the water (when using nonozonated biofiltration and slow sand filtration).[167]

Cartridge filtration was indirectly affected by TOC because for TOC > 1 mg/L, ozonated biofiltration ozone dose was high enough to make its rank worse than cartridge filtration. Bag filtration was also indirectly affected by TOC because for TOC > 2 mg/L, ozonated biofiltration ozone dose was high enough to make its rank worse than bag filtration. Ozonated biofiltration had its best rank of 4 when TOC < 1 mg/L because less ozone dose energy was required. Membrane filtration was insensitive to different system source water qualities and its rank did not change significantly as a result. Filtration rank trends for Regulatory Scenario#3 were the same as for Regulatory Scenario#2 with a few minor differences. First, conventional filtration was no longer sensitive to alkalinity. Second, conventional filtration had a better rank visually than biofiltration and slow sand filtration when TOC > 2.5 mg/L and the coagulant dose needed to meet U.S. regulation lowered the alkalinity enough that less pH adjustment chemical needed to be added. This was based on the same mechanism of the conventional filtration alkalinity trend in Regulatory Scenario#2.

Disinfection option ranks for Regulatory Scenario#2 were affected by system source water quality in the same way as for Regulatory Scenario#1 with slightly different values when options were preferred because they were no longer as sensitive to pH because every alternative had to use coagulation, which made pH before disinfection more consistently low. However, adding coagulation process did not affect temperature before disinfection, which remained a more sensitive parameter. Disinfection rank trends for Regulatory Scenario#3 were the same as for Regulatory Scenario#2 except options were more sensitive to pH because coagulant doses were not as high under U.S. regulations as when 30% TOC removal was required.[167]

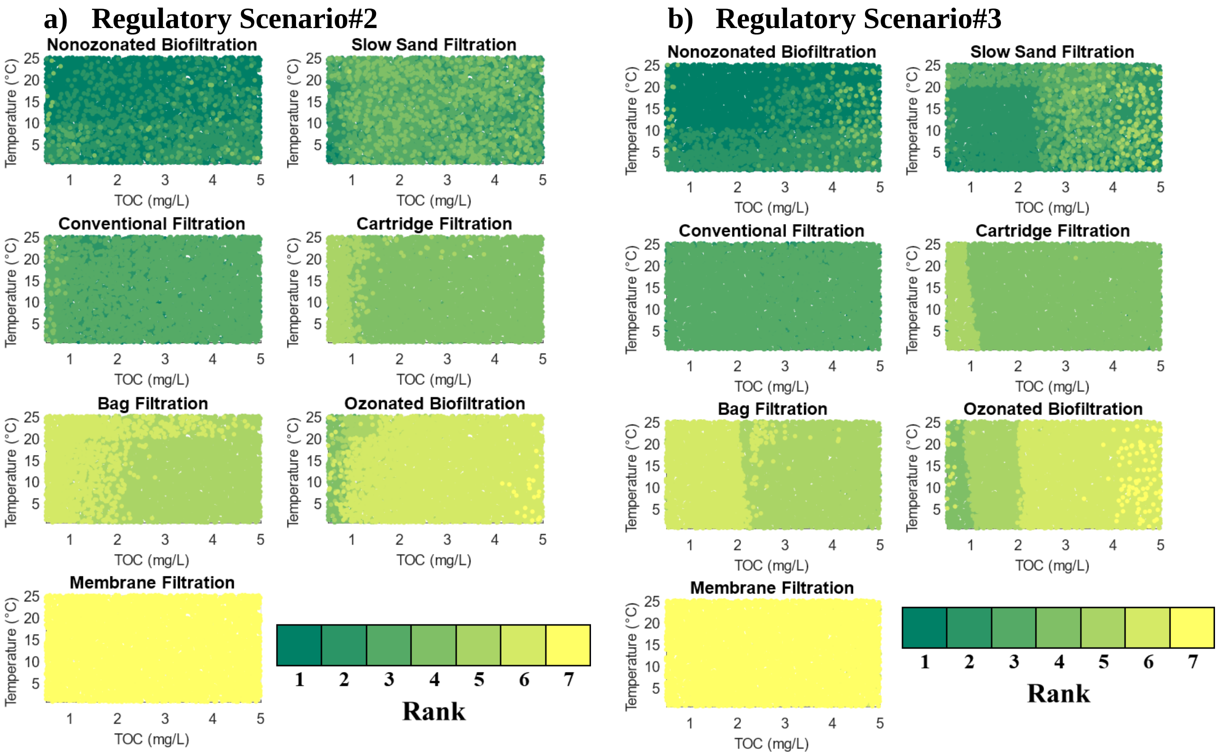


Figure 5.4. As representative examples, the source water effects on filtration ranks were evaluated for all filtration options in Regulatory Scenario#2 (**a**) and #3 (**b**) when coupled with chlorine with plastic piping, which was the top ranked disinfection option (Figure D.5 and Figure D.6). While each point represents a combination of six source water parameters, the multidimensional data is being shown as only a function of the two parameters to which the ranks were most sensitive to, temperature and TOC.

Key takeaways. Treatment process rank data for different system source water quality exhibited similar orders to when uncertainty was considered, and rank trends changed across regulatory scenarios because some filtration options required coagulant addition when turbidity became more difficult to treat and TOC removal was required. In this source water analysis filter selection was the most determining factor in assigning ranks. Filter ranks were most sensitive to pH, TOC, and temperature, all of which benefit different filter processes across their ranges (Table 5.2). Furthermore, the most important uncertainty parameters (e.g., filter operational head loss) should be considered in combination with system source water quality. Treatment train alternatives should be selected to take advantage of system source water quality and pair the most recommended filtration and disinfection alternatives.

5.3.3 Criteria Weighting Scheme Analysis

Ten thousand different stakeholders and stakeholder preferences (i.e., criteria weighting schemes) were considered for a small system with national average source water quality. The criteria whose weights impacted the treatment alternative's ranks and recommendations the most showed up in all three regulatory scenarios (Table D.7, Table D.8, and Table D.9). The most often recommended treatment trains of the 28 alternatives were the same as for the source water quality analysis: cartridge filtration with chlorine with plastic piping was for Regulatory Scenario#1 (Figure D.7), nonozonated biofiltration with UV with chlorine residual for Regulatory Scenario#2 (Figure D.8), and nonozonated biofiltration with chlorine with plastic piping for Regulatory Scenario #3 (Figure D.9).

For the main filtration options, cartridge filtration's rank was moderately sensitive to the *operator training requirement* and *local long-term health risk* weights because cartridge filtration had relatively low training requirements and because it formed more disinfection by-products than the filtration options that used coagulation or biological processes, respectively. Slow sand filtration's rank was moderately sensitive to the *capital cost* and *maintenance requirements* weights because the amount of filter media and concrete needed to house the filter media are expensive, and because the upkeep required for slow sand filtration is less than other similarly ranked filter options, respectively. Bag filtration's rank was moderately sensitive to the *capital cost*, *local long-term health risk*, and *operator training requirements* weights because bag filtration was cheaper than the similarly ranked slow sand filtration, because it formed relatively more disinfection by-products, and because its simple operation had relatively low training requirements, respectively. The ranks of nonozonated biofiltration and conventional filtration were moderately sensitive to the *local long-term health risk* and *operator training requirements* weights; they were sensitive for the opposite reasons as bag filtration. The ranks of ozonated biofiltration and membrane filtration were both moderately sensitive to the *local long-term health risk* and *resilience to regulation and source water* weights because membrane filtration did not remove disinfection byproduct precursors as ozonated biofiltration did, and because ozonated biofiltration is more robust to changes to source water quality and regulations, respectively.

For the main disinfection options, the ranks of chlorine using plastic piping and chlorine using a concrete tank were insensitive to different stakeholder weight schemes because neither had many significant trade-offs with other disinfection options for this analysis' source water. UV with a chlorine residual's rank was moderately sensitive to the *local long-term health risk* and *operation & maintenance costs* weights because UV disinfection formed relatively fewer disinfection by-products and because the cost of electricity and parts to operate and maintain the UV system were relatively high, respectively. The rank of the chlorine using a concrete tank with steel baffles option was moderately sensitive to the *global ecosystem quality* weight because the production of stainless steel for baffles resulted in relatively high pollution.

This criteria weighting scheme analysis helped to highlight each treatment process's benefits and disadvantages, for a national average source water, and provide guidance for what treatment processes are most effected by stakeholder weights for other source waters. Of the 12 decision criteria, 6 effected rank results significantly and should be carefully considered when evaluating real stakeholder preferences. For example, stakeholder data indicated that small systems stakeholders valued human health criteria the most (59% of the total weight) (Figure 5.5). For these stakeholders, alternatives that performed better for human health criteria (e.g., nonozonated biofiltration, conventional filtration, and UV disinfection) but used more resources and were more complicated had a slight decision-making edge compared to simpler treatment alternatives (e.g., cartridge filtration, and chlorine disinfection options) that did not have as many benefits in human health criteria.

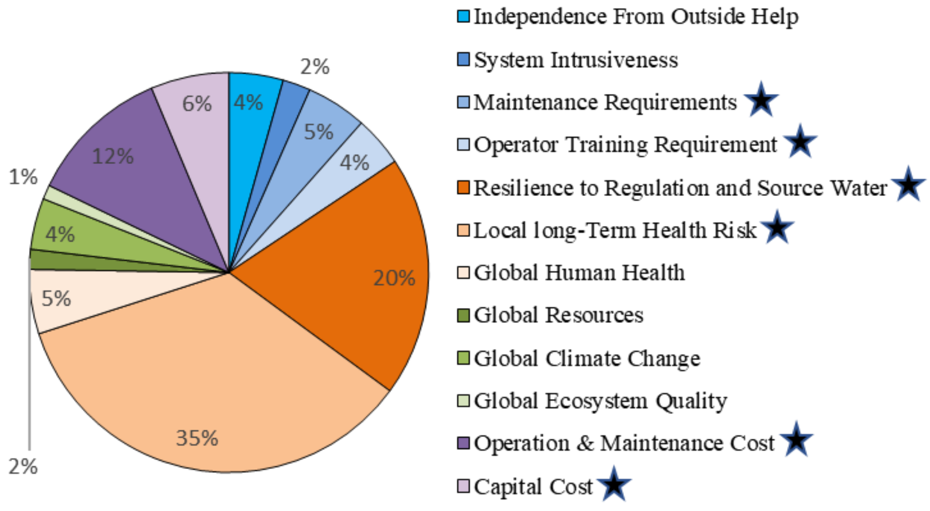


Figure 5.5. Criteria weighting percentages as specified by three small drinking water system stakeholders that had consistent AHP results. The three separate stakeholder's AHP results were averaged geometrically and then processed using AHP methods.[187,196] Note: stars indicate criteria that significantly affected rank results.

5.3.4 Guidelines to Improve Small System Decisions

This analysis focused on the most likely current source water qualities, treatment objective, and treatment trains to help identify some generalizable, useful decision-making insights for small systems. While this analysis highlighted some common technology choices, it also helped to highlight technologies that are commonly overlooked by small systems despite their potential benefits and ability to be the most sustainable choice. For example, some small systems have historically chosen a treatment process based on its simple operation. However, this study helps to show, by using SIPS and multiple decision criteria, that those simple technologies may not be enough to meet all the needs of a small system, especially in cases where a stakeholder is risk averse, source water quality is degrading, or treatment objectives are made more stringent. This study outlined general guidelines for when specific treatment processes were most preferred, which can serve as a good starting point for making a treatment decision at a system where resources are limited and multiple treatment options should be considered. For example, since the ranks of treatment alternatives within a regulatory scenario were similar when considering different source waters and stakeholder preferences, the same treatment processes were generally recommended in each regulatory scenario. Also, slow sand filtration performed well, even if it was not most recommended, regardless of the regulatory scenario. In all simulations, filter selection was the driver for a treatment train's rank, so selecting a filter that performs well at a specific small system should be prioritized.

This study also provides an approach for a small system to evaluate its system-specific characteristics, both those that are static and those that may change over time, such as source water quality. Results of this study were generated by evaluating 28 treatment train alternatives. There are even more treatment trains, over 112, though, that the SIPS tool can evaluate. For example, the impact of chemical selection can also be evaluated; while those impacts are generally known,[22,167] small systems still have the ability to use SIPS to general site-specific information on any possible treatment train. For example, systems that have seasonal source water quality changes can input their most common source water quality and increase the weighting of the *resilience to regulation and source water* criterion, which accounts for the robustness of a treatment train, in order to select the most robust treatment train. Also, since technologies are always changing–there are continuous efforts to improve existing technologies and create new technologies–so there is still a benefit to small systems using SIPS to evaluate more options as they become available.

Chapter 6: Conclusions

This dissertation undertook the challenge of improving the sustainability of small drinking water systems in order to improve drinking water quality. The key findings and contributions of this dissertation are discussed for each chapter (Table 6.1), where: the environmental impacts of conventional filtration and biological filtration (Chapter 2) as well as chlorine and ultraviolet (UV) disinfection (Chapter 3) were quantified using life cycle assessment (LCA); a small drinking water system decision-support tool, SIPS, was constructed (Chapter 4); and a sustainability assessment and comparison of treatment processes at different simulated small drinking water systems was conducted (Chapter 5). Results from Chapters 2 and 3 can be used by technology developers to improve a technology (i.e., reduce the use of and/or change the types of chemicals, energy, and materials) and by operators to operate their system more effectively (e.g., optimize coagulant doses for TOC removal) in order to minimize environmental impacts. While the contribution of global pollution from small drinking water systems may be very small relative to other industries, all industries should leverage opportunities to reduce environmental impacts, especially since these strategies usually provide multiple benefits, such as reducing financial expenses. Chapter 4 combines and expands on the LCA models from Chapters 2 and 3 to ultimately provide a comprehensive sustainability decision-making tool that will help small systems leverage environmentally sustainable opportunities without compromising their ability to consistently provide affordable, high quality drinking water. The SIPS tool uses multiple decision criteria, to avoid unforeseen negative consequences of an alternative, and helps with trade-off navigation, specific to a small system (i.e., source water, technology options, stakeholders, regulations). Chapter 5's results provide general decision-making recommendations to help small systems narrow down their best options. Overall, each small drinking water system can use the SIPS tool to identify customized recommendations.

Table 6.1. Summary of main research results and contributions.

Chapter/Citation	Main Results	Contributions
Chapter 2: Jones, C.H.; Terry, L.G.; Summers, R.S.; Cook, S.M. Environmental life cycle comparison of conventional and biological filtration alternatives for drinking water treatment. *Environ. Sci. Water Res. Technol.* 2018, 4 (10), 1464–1479. (https://pubs.rsc.org/en/content/articlelanding/2018/ew/c8ew00272j#!divAbstract)	Conventional filtration was environmentally preferred for more pristine source waters; biological filtration was environmentally preferred when the source water was more degraded or when treatment requirements were more stringent.	Provided information for stakeholders to select which filtration is most environmentally sustainable based on source water quality and treatment objective; first evaluation and quantification of the environmental impacts of biological filtration; 60,000 source waters were simulated to represent all common water qualities.
Chapter 3: Jones, C.H.; Shilling, E.G.; Linden, K.G.; Cook, S.M. Life Cycle Environmental Impacts of Disinfection Technologies Used in Small Drinking Water Systems. *Environ. Sci. Technol.* 2018, 52 (5), 2998–3007. (https://pubs.acs.org/doi/abs/10.1021/acs.est.7b04448)	Chlorine was environmentally preferred to UV disinfection in most cases; UV was preferred when its benefits of Cryptosporidium removal could be leveraged.	Provided the first environmental comparison of common chemical and non-chemical disinfection options for drinking water treatment; specified a functional unit that made the comparison of disinfection processes equal; showed that the common LCA assumption of negligible infrastructure impacts may not be valid for small systems.
Chapter 4: Jones, C.H.; Meyer, J.; Seidel, C.J.; Cornejo, P.K.; Hogrewe, W.; Cook, S.M. A New Framework for Small Drinking Water Plant Sustainability Support and Decision-making. In preparation for submission to peer-reviewed journal.	SIPS tool provides relevant engineering treatment recommendations and is universal in its capacity to represent diverse small systems; SIPS helps stakeholders navigate trade-offs; SIPS has the capacity to be updated easily to remain relevant over time as treatment technologies change.	Developed a decision-support framework that can be applied to other engineered systems to improve sustainability; provided decision criteria, with decision criteria scoring approaches, and a score aggregation approach that systematically supports trade-off identification and navigation; the tool is user friendly and will be freely accessible; SIPS can be used by multiple stakeholders to inform decisions surrounding small systems.
Chapter 5: Jones, C.H. and Cook, S.M. Evaluating Small System Treatment Processes for Different Source Water Qualities, Regulations, and Stakeholder Preferences. In preparation for submission to peer-reviewed journal.	There was no one treatment process that was the best option for every simulated small system; conditions and small system characteristics that resulted in certain treatment processes being most preferred (i.e., sustainable) were identified; filtration options had the largest impact on an entire treatment train's sustainability.	General recommendations were provided that considered multiple criteria during treatment process selection; treatment selection guidelines were determined that indicate when a treatment process should be recommended.

6.1 Chapter 2: Filtration LCA

Biological filtration has drawn interest for its potential to reduce chemical use, but there has been no environmental characterization of the filtration process to comprehensively and quantitatively evaluate its potential benefits. LCA methodology was leveraged to quantitatively compare conventional filtration and biological filtration treatment options. Conventional filtration was preferred when the TOC removal requirement was based on the enhanced coagulation rule and the source water was more pristine. Also, the benefits of adding pre-ozonation before biological filtration were not enough to outweigh the added environmental impacts of ozone generation energy. Nonozonated biological filtration performed best when the TOC removal requirement was highest and the source water was more degraded.

Contributions. This was the first study to quantify the environmental impacts of biological filtration compared to conventional filtration for different source waters and regulatory scenarios. This study used 60,000 simulated scenarios instead of a case study, which was found to be beneficial because it provided more comprehensive data for decision-making; specifically, because the best filtration option was dependent on regulatory and source water conditions. As a result, the study provides material, energy, and chemical use inventories, which may be applicable to other research. Also, the results of this study could be used to augment how treatment plants operate their coagulation process (e.g., the results show the benefit of optimizing a coagulant dose based on TOC instead of turbidity, which is not currently common practice). Overall, this research can be used by systems that are considering switching to biological filtration; for example, a system looking to reduce chemical use may use their specific source water quality to determine if switching to biological filtration can have environmental benefits and the extent of those potential benefits.

6.2 Chapter 3: Disinfection LCA

Alternatives to chemical disinfection have several benefits, but the environmental impacts of using non-chemical disinfection, such as UV, are not well understood and have not been rigorously quantified, especially in a drinking water treatment context. LCA methodology was leveraged to quantitatively compare chlorine and UV disinfection options. Chlorine disinfection was more environmentally preferred if a filter was required, whether or not a chlorine residual was required. UV disinfection was more preferred when a filter was not required and a chlorine residual was because its *Cryptosporidium* reduction capabilities could be realized. UV disinfection also became more environmentally competitive with chlorine disinfection when chlorine disinfection used stainless steel in its contact zone and when the environmental impacts of energy production were reduced by switching to less impactful energy sources (e.g., renewables).

Contributions. This was the first study to quantify the environmental impacts of UV disinfection compared to chlorine disinfection in a drinking water context. A functional unit was defined that provided an equal comparison and results were gathered for three distinct scenarios that represent a wide range of potential disinfection operational configurations and energy grids. The study provides material, energy, and chemical use inventories, which may be applicable to other research. Also, results showed that at small scales infrastructure contributes significantly to total environmental impacts. Ultimately, systems that are considering using chemical free disinfection may use this study's results to determine if there could be environmental benefits of using UV disinfection.

6.3 Chapter 4: Decision-making Tool

Small systems do not have access to comprehensive decision-support when managing challenges of more stringent regulations and degrading source water quality. LCA and MCDA methods were used to construct a comprehensive decision-making framework that can evaluate 112 different treatment options for a wide range of design parameters, source waters, treatment standards, and stakeholder preferences. Decision-making criteria were developed collaboratively with diverse small system stakeholders, and the framework provides data on trade-offs between criteria. It also aggregates the data to navigate trade-offs and provides a decision recommendation.

Contributions. The tool can provide recommendations for specific small systems because it uses small system decision criteria and accounts for stakeholder preferences. It also provides stakeholders with data to navigate trade-offs. Also, it makes relevant engineering recommendations. The input flexibility, accessibility, and user friendliness of the tool make results applicable for a wide range of small systems that have different technical requirements and preferences for decision criteria. In addition to the benefit to multiple small systems, this work also benefits multiple stakeholders. Different stakeholder groups can gain unique insights from using the tool; for example, consumers can use it to determine if the treatment alternatives being considered and selected are appropriate for their needs; engineers can use it to navigate short-term vs. long-term objectives; operators can use it to optimize their treatment using system specific characteristics; and regulators can use it to better facilitate treatment options that satisfy regulations and identify areas where small systems need more support to comply with regulations. A decision-support framework was developed that could be applied to large drinking water systems or other engineered systems.

6.4 Chapter 5: Treatment Recommendations

There are tens of thousands of diverse small drinking water systems that will need to update their treatment, but there are too many to use a case study decision-support approach at all of them. Therefore, a modelling approach is needed to identify when treatment processes should be selected. Monte Carlo analysis methods were used with the model constructed in Chapter 4 to generate results representing 60,000 unique small drinking water systems for a range of regulatory requirements, source waters, and stakeholder preferences. The effects of uncertainty (i.e., main assumptions) were also considered. It was found that filtration option selection had the largest impact on an entire treatment train's sustainability. Disinfection selection effected the treatment recommendation most when regulatory requirements did not require TOC removal, but disinfection still had a smaller impact on the final recommendation than filtration. Treatment process recommendations were also dependent on regulatory requirements and on the pH, temperature, and TOC of the specified source water. Stakeholder preference results indicated that there could be changes to recommendations when a stakeholder is highly risk averse or risk-taking.

Contributions. Small systems have limited resources to devote to making treatment decisions, and the results of this study provide guidance on how decision-makers should prioritize their time and effort. Specifically, the results show that efforts should be focused on selection of the filtration process. Treatment selection guidelines of regulatory requirements, source water quality, and uncertainty were also specified that indicate when treatment processes are most preferred. These generalizable findings will help the tens of thousands of small drinking water systems needing to improve their drinking water quality that cannot afford to undergo a full decision-making case study analysis.

6.5 Future Research

Although this work provides comprehensive insights into the sustainability of small drinking water systems, there are still areas where this research can be expanded. For instance, the process models in the SIPS tool are applicable for small systems and some are applicable regardless of system size (e.g., coagulation TOC reduction), but there are some algorithms that work discretely rather than linearly (e.g., UV disinfection systems), which would require additional modeling to make results applicable for systems with larger flow rates. Also, additional treatment goals could be included in the tool as the environmental engineering community reaches a consensus on the effectiveness of treatment processes on treating emerging contaminants such as micropollutants. Finally, getting small systems stakeholders comfortable with using the tool will take time. To smoothen this process, further dissemination of this dissertation's work will be conducted.

6.6 Dissemination

The contents of this dissertation will be disseminated in 4 peer-reviewed publications (two are published[91,167] and two in preparation) and 19 professional presentations at various conferences. The SIPS tool (Chapter 4) will be freely available (at: https://www.colorado.edu/lab/cook/), along with a user's guide and a developer's guide, to any small system stakeholder. The user's guide summarizes the tool's capacity and user interface to ensure that the tool is accessible to non-technical stakeholders. The developer's guide will support the longevity of the tool by providing information on how to add treatment processes or updating existing ones as technology development provides new opportunities to small systems.

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Appendix A: Supplementary Information for: "Environmental Life Cycle Comparison of Conventional and Biological Filtration Alternatives for Drinking Water Treatment"

Reproduced from reference (Jones, C. H.; Terry, L. G.; Summers, R. S.; Cook, S. M. Environmental life cycle comparison of conventional and biological filtration alternatives for drinking water treatment. *Environ. Sci. Water Res. Technol.* **2018**, *4* (10), 1464–1479.) with permission from Royal Society of Chemistry

A1. Water Quality Regulations

Table A1. Total organic carbon (TOC) percent removal requirements, as a function of source water TOC and alkalinity, as defined by the enhanced coagulation requirement in the Stage 1 DBP Rule.[89,94]

Source Water TOC (mg/L C)	Source Water Alkalinity (mg/L $CaCO_3$)		
	<60	**60 to 120**	**>120**
>2 to 4	35%	25%	15%
>4 to 8	45%	35%	25%
>8	50%	40%	30%

A2. Life Cycle Inventory and Impact Assessment Categories

Table A2. Life cycle unit process data and descriptions. Data were from the ecoinvent v3 database[81] except for unit process data on anthracite coal, which was from US-EI 2.2 database.[82] The relative amount of US electricity produced by each electrical grid is stated under application (percent contribution);[197] n/a is not available.

Description	Calc. Section #	Unit Process Name	Application
Alum	A3.1	Aluminium sulfate, powder {RoW}| production | Alloc Def, U	Coagulation
Anthracite	A4.1	Anthracite coal, at mine NREL/RNA U	Filter media
Caustic Soda	A5	Sodium hydroxide, without water, in 50% solution state {RoW}| chlor-alkali electrolysis, membrane cell | Alloc Def, U	pH adjustment
Chlorine	[91]	Sodium hypochlorite, without water, in 15% solution state {RoW}| sodium hypochlorite production, product in 15% solution state | Alloc Def, U	Disinfection (free chlorine from NaOCl and was adjusted as such)
Concrete	[91]	Concrete, 20MPa {RoW}| concrete production 20MPa, RNA only | Alloc Def, U	Chlorine contact basin
Electricity	A4.2, [91]	Electricity, medium voltage {ASCC}| market for | Alloc Def, U	ASCC grid (n/a)
		Electricity, medium voltage {FRCC}| market for | Alloc Def, U	FRCC grid (6% US electricity)
		Electricity, medium voltage {NPCC, US only}| market for | Alloc Def, U	NPCC grid (7% US electricity)
		Electricity, medium voltage {MRO, US only}| market for | Alloc Def, U	MRO grid (17% US electricity)
		Electricity, medium voltage {RFC}| market for | Alloc Def, U	RFC grid (20% US electricity)
		Electricity, medium voltage {SERC}| market for | Alloc Def, U	SERC grid (17% US electricity)
		Electricity, medium voltage {SPP}| market for | Alloc Def, U	SPP grid (6% US electricity)
		Electricity, medium voltage {TRE}| market for | Alloc Def, U	TRE grid (9% US electricity)
		Electricity, medium voltage {WECC, US only}| market for | Alloc Def, U	WECC grid (18% US electricity)
		Electricity, medium voltage {HICC}| market for | Alloc Def, U	HICC grid (n/a)
Ferric Chloride	A3.1	Iron (III) chloride, without water, in 40% solution state {RoW}| iron (III) chloride production, product in 40% solution state | Alloc Def, U	Coagulation
Hauling	A4.3	Trasport freight, lorry 3.5-7.5 metric ton, EURO6 {RoW}| transport, freight, lorry 3.5-7.5 metric ton, EURO6 | Alloc Def, U	Solids and chemical hauling
Lime	A5	Lime, hydrated, loose weight {RoW}| production | Alloc Def, U	pH adjustment
Reinforcing Steel	[91]	Reinforcing steel {RoW}| market for | Alloc Def, U	Chlorine contact basin
Sand	A4.1	Sand {GLO}| market for | Alloc Def, U	Sand (filter media)
Soft Plastic	[91]	Polyethylene, high density, granulate {RoW}| production | Alloc Def, U	Plastic cylindrical tank (contact basin)
Stainless Steel	A4.1, [91]	Steel, chromium steel 18/8, hot rolled {RoW}| production | Alloc Def, U	Filter housing, steel baffles, ozone generator
Tap Water	[91]	Tap water {RoW}| tap water production, conventional treatment | Alloc Def, U	Dilution water (chlorine solution)

Table A3. Material, energy, and chemical quantities for the 3 filtration alternatives. Values are for the entire functional unit (i.e., treatment of 2,700 m^3/day over 40 years) normalized to 1 m^3 of treated water. These results are for the treatment of the typical source water under the enhanced coagulation treatment scenario, assuming typical values for each uncertainty parameter (Table A11). Chlorine mass is kg free chlorine from NaOCl.

Inventory Unit Process (Units/m^3 water treated)	**Filtration Alternatives**		
	Conventional Filtration	Nonozonated Biofiltration	Ozonated Biofiltration
Alum (kg)	1.30E-02	1.20E-02	1.00E-02
Anthracite (kg)	1.37E-04	1.37E-04	1.37E-04
Backwash Energy (kWh)	9.46E-04	9.46E-04	9.46E-04
Baffle Steel (kg)	3.61E-04	3.61E-04	3.61E-04
Caustic (kg)	1.28E-02	1.22E-02	1.08E-02
Chemicals Hauling (tkm)	5.75E-04	5.43E-04	5.43E-04
Chlorine (kg)	1.40E-03	1.40E-03	1.40E-03
Chlorine Dose (mg free Cl_2/L)	1.4	1.4	1.4
Chlorine Pump Energy (kWh)	6.26E-06	6.26E-06	6.26E-06
Polyethylene Chlorine Storage Tank (kg)	9.38E-07	9.38E-07	9.38E-07
Concrete (m^3)	1.72E-06	1.72E-06	1.72E-06
Contactor Pump Energy (kWh)	9.71E-09	9.71E-09	9.71E-09
Stainless Steel Filter Housing (kg)	7.45E-04	7.45E-04	7.45E-04
Filter Operational Energy (kWh)	1.37E-02	1.39E-02	1.39E-02
Rebar (kg)	7.78E-06	7.78E-06	7.78E-06
Sand (kg)	1.71E-04	1.71E-04	1.71E-04
Solids Hauling (tkm)	2.61E-04	2.41E-04	2.41E-04
Ozone Energy (kWh)	N/A	N/A	3.22E-02
Stainless Steel Ozone Generator (kg)	N/A	N/A	3.79E-05

A3. TOC Removal Design Calculations

A3.1 Coagulation

Alum and Ferric Chloride coagulation is affected most by pH and specific ultraviolet absorbance (SUVA), both of which are affected by alkalinity and TOC. Also, alum and Ferric Chloride lower the water's pH and alkalinity. Due to these complex interactions, the following approach was used to determine the alum dose needed for a specific percent TOC removal target. An alum dose and TOC removal table (Table A4) was generated for every source water scenario to determine the proper alum dose for coagulation using 6 main steps. First, the source water quality was defined in terms of TOC, alkalinity, pH, SUVA, and temperature. Second, a comprehensive range of possible alum doses was generated (from 0 to 122 mg/L, in 1 mg/L increments). Third, the pH of the coagulated water was calculated by iteratively solving Eq. A from the U.S. EPA's Water Treatment Plant Model v2.[198] Fourth, the coagulated water TOC was calculated using Eq. A2 with values for the coagulated water pH and alkalinity input as well as variables from the Edwards Model.[93] Fifth, the SUVA of coagulated water was determined (Eq. A3). Sixth, the percent TOC removal was calculated based on the source water TOC and final TOC. Table A4 shows example alum doses and the corresponding percent TOC removals for an example source water. The same strategy was used for Ferric Chloride. Overall, the required alum dose for a specified source water quality and TOC removal target was found from these tables. For the enhanced coagulation TOC removal scenario, the selected alum dose was the smallest dose associated with any of the follow situations, as long as that value was above the minimum allowable value (10 mg/L alum, uncertainty parameter, Table A11): (i) percent TOC removal target, (ii) 2 mg/L coagulated water TOC, or (iii) 2 L/mg/m coagulated water SUVA. Table A5 shows the alum doses needed for all of these conditions for an example source water. Note that coagulant ecoinvent data is for dry weights of coagulant chemicals without water as verified through correspondence with ecoinvent.

Table A4. Example alum dose and TOC removal table. Values were calculated for the national average source water scenario (77 mg/L $CaCO_3$, 3.2 mg/L TOC, 15 °C) and enhanced coagulation TOC removal.

Alum Dose (mg/L)	Coagulated Water pH	Coagulated Water TOC (mg/L)	Coagulated Water SUVA	TOC Removal (%)
0	7.50	3.20	3.13	0%
1	7.43	3.16	2.53	1.2%
2	7.37	3.12	2.40	2.4%
…	…	…	…	…
13	6.94	2.64	2.01	17%

Table A5. The four alum dose options for the national average source water scenario (77 mg/L $CaCO_3$, 3.2 mg/L TOC, 15 °C) and enhanced coagulation TOC removal. The smallest, above the minimum allowable alum dose, was chosen as the modeled alum dose (green shade).

Purpose	Alum Dose (mg/L)	Coagulated Water TOC (mg/L)	Coagulated Water SUVA	TOC Removal (%)
Turbidity removal (minimum allowable dose)	10	2.78	2.06	13%
SUVA (≤ 2 L/mg/m)	14	2.60	2.00	19%
TOC (≤ 2 mg/L)	30	1.99	1.81	38%
%TOC removal	19	2.38	1.94	25%

$$\begin{aligned}(\alpha_1 + (2 * \alpha_2)) * [Ct_{CO3}] + \left[\frac{k_w}{[H^+]}\right] - [H^+] \\ = [\alpha_1 * C_{TCO3}] + 2 * [\alpha_2 * C_{TCO3}] + \left[\frac{k_w}{[H^+]}\right] - [H^+] - 6 * [Alum]\end{aligned} \tag{Eq. A1a}$$

$$\alpha_1 = \frac{k_{1CO3} * [H^+]}{[H^+]^2 + k_{1CO3} * [H^+] + k_{1CO3} * k_{2CO3}} \tag{Eq. A1b}$$

$$\alpha_2 = \frac{k_{1CO3} * k_{2CO3}}{[H^+]^2 + k_{1CO3} * [H^+] + k_{1CO3} * k_{2CO3}} \tag{Eq. A1c}$$

$$C_{TCO3} = \frac{[Alk] + [H] - [OH]}{\alpha_1 + (2 * \alpha_2)} \tag{Eq. A1d}$$

$$\mathbf{k_{1CO3} = exp\left\{\left[\left(\frac{7700\,\frac{J}{mole}}{8.314\,\frac{J}{K*mole}}\right)\left(\frac{1}{298.15\ K}-\frac{1}{T}\right)\right]-14.5\right\}} \qquad \textbf{Eq. A1e}$$

$$\mathbf{k_{2CO3} = exp\left\{\left[\left(\frac{14900\,\frac{J}{mole}}{8.314\,\frac{J}{K*mole}}\right)\left(\frac{1}{298.15\ K}-\frac{1}{T}\right)\right]-14.5\right\}} \qquad \textbf{Eq. A1f}$$

Where:

$[OH^-]$ = Concentration of hydroxide (M)
$[H^+]$ = Concentration of hydrogen (M)
$[CO_3^{2-}]$ = Concentration of carbonate (M)
$[HCO_3^-]$ = Concentration of bicarbonate (M)
[Alum] = Concentration of dry alum added (M)
α_1 = Water chemistry equilibrium value for the second hydrogen state (Eq. Ab)
α_2 = Water chemistry equilibrium value for the third hydrogen state (Eq. Ac)
C_{TCO3} = Total concentration of all carbonate species (M)
k_{1CO3} = Carbonate equilibrium constant for second hydrogen[199]
k_{2CO3} = Carbonate equilibrium constant for second hydrogen[199]
k_w = Water equilibrium constant (4.52E-15)
[Alk] = Concentration of influent alkalinity eq/L
T = Influent water temperature (K)

$$\mathbf{DOC_i = (1-(SUVA*K_1)-K_2)*TOC} \qquad \textbf{Eq. A2a}$$

$$\mathbf{Al^{3+} = \frac{Dry\ Alum*\left(\frac{2\ mmol\ Al^{3+}}{1\ mmol\ Alum}\right)}{\left(\frac{342.15\ mg\ dry\ Alum}{1\ mmol\ Alum}\right)}} \qquad \textbf{Eq. A2b}$$

$$\mathbf{a = (x_1*pH^3+x_2*pH^2+x_3*pH)} \qquad \textbf{Eq. A2c}$$

$$\mathbf{\frac{DOC_i-[C]_{eq}}{Al^{3+}} = \frac{a*b*[C]_{eq}}{1+b*[C]_{eq}}} \qquad \textbf{Eq. A2d}$$

$$\mathbf{[C]_{eq} = \frac{\sqrt{\left(b^2*(DOC_i-a*Al^{3+})^2+(2b*(DOC_i+a*Al^{3+})+1)\right)}+\left(\boldsymbol{b}*\left(DOC_i-a*Al^{3+}\right)\right)-1}{2b}} \qquad \textbf{Eq. A2e}$$

$$\mathbf{TOC_{coagulated} = [C]_{eq}+(TOC-DOC_i)} \qquad \textbf{Eq. A2f}$$

$$\mathbf{TOC\ Removal = \frac{TOC-TOC_{coagulated}}{TOC}} \qquad \textbf{Eq. A2g}$$

Where:

DOC_i = Sorbable DOC of coagulation influent water (mg/L)

SUVA = Specific ultraviolet absorbance of influent water (L/mg/m)
K_1 = Constant: (-0.075)[93]
K_2 = Constant: (0.56)[93]
TOC = Influent TOC (mg/L)
Alum = Alum dose added (mg/L)
Al^{3+} = Aluminum ions present (mM)
a = Maximum TOC sorption per mM of Al^{3+} added
x_1 = Constant: (284)[93]
x_2 = Constant: (-74.2)[93]
x_3 = Constant: (4.91)[93]
$[C]_{eq}$ = Amount of sorbable TOC remaining after coagulation (mg/L)
b = Constant: (0.147)[93]
TOC = Influent TOC (mg/L)
$TOC_{coagulated}$ = Coagulated water TOC concentration (mg/L)
TOC Removal = Amount of TOC removed from source water (%)

$$\mathbf{SUVA_{coagulated}} = \frac{\left(\mathbf{5.716 * (UVA)^{1.0894} * (3 * Al^{3+})^{0.306} * (pH)^{-0.9513}}\right)}{\mathbf{TOC_{coagulated}}} * \frac{\mathbf{100cm}}{\mathbf{m}} \qquad \textbf{Eq. A3}$$

Where:

$SUVA_{coagulated}$ = Specific ultraviolet absorbance of coagulated water (L/mg/m)
UVA = ultraviolet absorbance at 254 nm of influent water (1/cm)

A3.2 Coagulated and Biological TOC Removal

Table A6 and Table A7 shows experimental data from the literature on biodegradable TOC removal using Table A6 for coagulation and Table A7 for nonozonated biofiltration and ozonated biofiltration, for systems that match the treatment process configuration in this LCA).

Table A6. Experimental data from the published literature that shows coagulated biodegradable TOC removal efficacy at different SUVAs.[52,72]

SUVA	**Coagulation percent TOC removal**		
	25th percentile	75th percentile	Average
< 3 (L/mg/m)	2%	5%	4%
≥ 3 (L/mg/m)	7.5%	13.5%	9%

Table A7. Experimental data from the published literature that shows nonozonated biofiltration and ozonated biofiltration TOC removal efficacy at different temperatures. Table data was adapted from Terry and Summers, 2018.[85]

Temperature	**Nonozonated Biofilter percent TOC removal**			**Ozonated Biofilter percent TOC removal**		
	Min	Max	Median	Min	Max	Median
≤10 °C	2%	14%	7%	3%	24%	11%
10 - 20 °C	5%	22%	10%	3%	47%	13%
≥20 °C	10%	22%	15%	6%	45%	20%

A3.3 City of Boulder Betasso Drinking Water Pilot Plant

A3.3.1 Methods

Pilot filters were set up at the City of Boulder's (CO) Betasso Water Treatment Plant (Betasso WTP) and source water, a combination of Barker Reservoir and Lakewood Reservoir was sent to the raw water tank, from which water was pumped to the treatment train. The Betasso WTP pilot was composed of a pilot treatment train operated at a flow rate of 2 gal/min. The train consists of rapid mix, three stage tapered flocculation, sedimentation, and filtration. The pilot plant schematic is represented by Figure A1. Pilot Plant Schematic. and a detailed schematic of the biofilters can be seen in Figure A2. The water passed through the static mixer, into three flocculation basins with tapered paddles, into a sedimentation basin with plate settlers, then split into one of two filters. For coagulation, aluminum sulfate was added at the front of the pilot plant. Chlorine was added at the front of the second filter to allow comparison of a biofilter (BF) and a conventional filter (RMF).

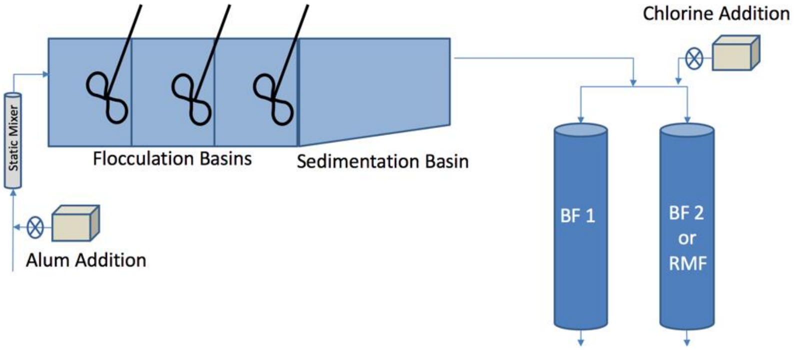

Figure A1. Pilot Plant Schematic.

Biologically active anthracite media from a full-scale filter, which was in operation for over seven years at the City of Longmont (CO) Nelson Flanders Drinking Water Treatment Plant, was used to pack the biofilter column. Inert anthracite was used to pack the conventional filter. The anthracite media had an effective size of 1.0 mm and an approximate uniformity coefficient of 1.3. The pilot system was modified and two columns were fabricated with depth taps to achieve filter depth samples, as seen in Table A8. The placement of the sampling ports allowed for measurement of only removal associated with the biological media and not the feed system. The filters were backwashed once per week with chlorinated water. Flow rates were monitored online via a flow analyzer. The analyzer installed was a Blue White F-400N Inline Rotameter with a range of 0.025 – 0.25 gpm. The flow was changed by adjusting the ball valve at the end of each filter. The flow rate was measured after each filter using in-line flow meters and averaged at 0.04 gpm, and the hydraulic loading rate averaged 2 m/hr.

Table A8. Pilot Filter Design Parameters

Filters	*Media Type*	*Experiment*	*Target EBCTs (min)*	*Media Height (cm)*	*Inner Diameter (mm)*	*Support Media Height (cm)*	*Flow Rate (gpm)*
Fresh (conventional) and Acclimated (biofilter)	Anthracite	Pilot with taps	5 15 30	17 50 100	76	8	0.04

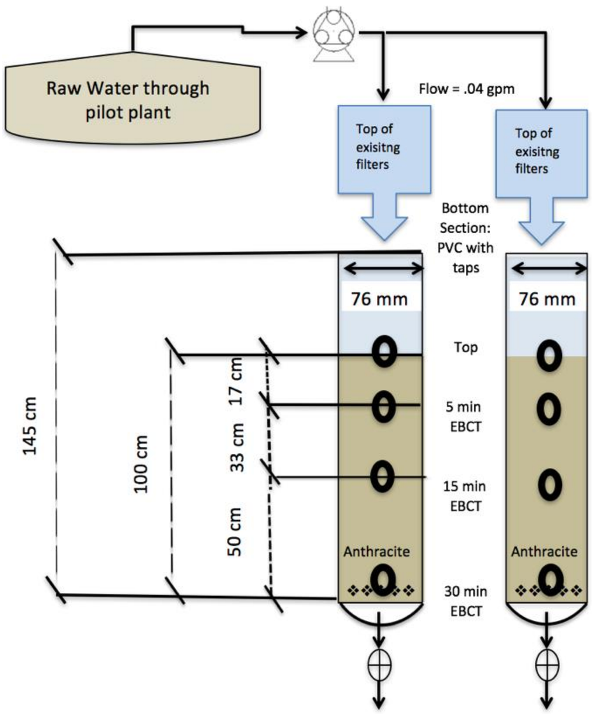

Figure A2. Biofilters Pilot Schematic

Influent feed water samples were collected at the tap directly between the sedimentation basin and the filters. Subsequent EBCT samples were taken at the corresponding EBCT tap. Liquid samples were collected in amber glassware that had been previously cleaned with deionized water and muffled at 550 ˚C for 3 hours. Samples were then transported to the University of Colorado, Boulder to be analyzed.

DOC concentrations were measured at the University of Colorado, Boulder on a Sievers M5310 C Laboratory Organic Carbon Analyzer using the ultraviolet irradiation/persulfate oxidation method (SM 5310C). The samples were collected and immediately filtered through a 0.45 mm membrane filter (Pall Life Sciences). Filters were first rinsed with 250 mL of reverse osmosis water to ensure that carbon leaching from the filters did not occur. After filtration, the samples were stored at 4°C until DOC analysis. All DOC analysis was performed within the hold time of

2 weeks of sample collection. Samples were taken in duplicate and analyzed in groups of four with a blank in between as a quality control measure to ensure stable operations and no organic carbon carryover from previous samples. The instrument was calibrated in accordance with the Operations and Maintenance manual. Quality assurance and quality control tests were performed monthly to ensure instrument accuracy.

A3.3.2 Results

Drinking water treatment utilities are required to remove a certain percentage of influent TOC (15 – 50%) based on source water TOC and alkalinity per the Stage 1 DBP Rule (termed the enhanced coagulation requirement). Utilities have multiple ways of meeting this regulation. The pilot plant was used to determine the trade-offs between DOC removal from coagulant addition and biofiltration. Figure A3 demonstrates the best optimization of biofiltration and coagulant addition to meet the Stage 1 DBP Rule requirements. If the utility is required to remove 30% of the influent TOC, the utility can either dose at 20 mg/L alum to achieve 33% DOC removal or dose at 15mg/L alum and run a biofilter with an EBCT of 30 minutes to achieve 32%. If the source water changes and the utility needs to remove 20% of the influent TOC, the utility can dose 15 mg/L alum to achieve 25% DOC removal or dose 10 mg/L alum and run a biofilter with an EBCT of 30 minutes to achieve 23% DOC removal.

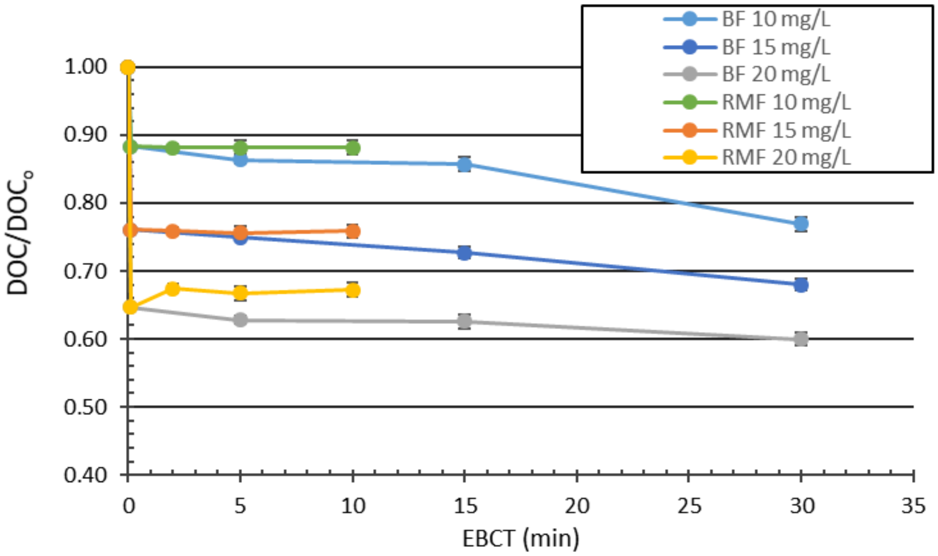

Figure A3. DOC removal throughout the coagulation and filtration process as a function of EBCT for a biological filter (BF) and a conventional rapid media filter (RMF) at 3 different aluminum sulfate doses: 10, 15 and 20 mg/L.

A4. Filter Design Calculations

Major materials and energy requirements to operate each filter were accounted for over the functional unit timeframe (40 years). A dual media filter of anthracite over sand was chosen for the rapid media filter design due to its prevalence in practice.

A4.1 Filter Materials

For each filter, the filter area was calculated using Eq. A4. The mass of media was calculated using Eq. A5. These equations assumed values for hydraulic loading rate and media depth, respectively, which were uncertainty parameters based on typical values for each type of filter (Table A11).[96] The total filter depth included the (packed) media depth and freeboard (0.3 m).[96] Filter volume was calculated using this total depth and Eq. A6. Then, the mass of steel needed for the filter housing was calculated using Eq. A7; this equation assumed a square cross section and typical steel thickness,[97] which was an uncertainty parameter (A6. Uncertainty and Sensitivity Analysis).

$$\mathbf{A_T = \frac{Q}{HLR}} \qquad \textbf{Eq. A4}$$

Where:

A_T = Total filter area requirement (m^2)
Q = Plant capacity flow rate (m3/hr)
HLR = Filter design hydraulic loading rate (m/hr)

$$\mathbf{M_{media} = A_T * D_{media} * \rho_{media}} \qquad \textbf{Eq. A5}$$

Where:

M_{media} = Mass of filter media (kg)
D_{media} = Media Depth (m) (Table A11)
ρ_{media} = Media density (kg/m^3): (1,500 kg/m^3 sand, and 800 kg/m^3 anthracite)[51]

$$\mathbf{V_{Filter} = A_T * \left(D_{media} + H_{fb} + D_{expansion}\right)} \qquad \textbf{Eq. A6}$$

Where:

V_{Filter} = Required filter volume (m^3)
H_{fb} = freeboard (m)
$D_{expansion}$ = Backwash filter expansion depth (m): Assumed 50% bed expansion[200]

$$\mathbf{M_{steel}} = \Bigg\{\Bigg(\Bigg(\bigg(\sqrt{\frac{\mathbf{V_{filter}}}{\mathbf{D_{total}}}}\bigg) * \mathbf{t_b}\Bigg) + \Bigg(\mathbf{4t_w}^{2} + \mathbf{4}\bigg(\sqrt{\frac{\mathbf{V_{filter}}}{\mathbf{D_{total}}}}\bigg) * \mathbf{t_w}\Bigg) * (\mathbf{D_{total}} + \mathbf{t_B})\Bigg\} * \boldsymbol{\rho}_{\mathbf{steel}} \qquad \textbf{Eq. A7}$$

Where:

D_{total} = Sum of media depth and filter head requirement (m)
t_b = Thickness of filter base (m)
t_w = Thickness of filter walls (m)
ρ_{steel} = Density of steel (kg/m^3): (7,500 kg/m^3)[150]

S4.2 Filter Energy Requirements

Pumping energy (for operation and backwash) was determined using Eq. A8. Filter operational head loss uncertainty was accounted for because media depth and water height above media were uncertainty parameters (Table A11).[96] Typical values were used to estimate the backwash flowrate and pressure[98] and ultimately to determine head loss during backwashing; both were uncertainty parameters (Table A11). Other than the 10 minutes of backwash every day, constant filtration was assumed.

$$\mathbf{P} = \frac{(\mathbf{Q} * \boldsymbol{\rho} * \mathbf{g} * \mathbf{H})}{\left(\mathbf{1000}\,\frac{\mathbf{W}}{\mathbf{kW}} * \boldsymbol{\eta}\right)} \qquad \textbf{Eq. A8}$$

Where:

P = Power (kW)
Q = Flow rate (m^3/s): water treatment plant flow rate or backwash flow rate
ρ = Density of liquid solution (kg/m^3): 1000 kg/m^3 for water
g = Gravity (9.81 m/s^2)
H = Head loss (m): filter operational head loss or backwash pressure
η = Efficiency (60%)

A4.3 Solids and Chemical Hauling Requirements

The hauling requirements, in tonne kilometers, were determined for solid coagulation waste and all chemicals (Eq. A9). The masses of alum, caustic soda, and chlorine were based on their treatment doses. The hauling distance was assumed to be the same for all chemicals and was an uncertainty parameter (Table A11). The solid waste generated from coagulation and sedimentation was conservatively estimated as the alum mass plus the mass of TOC removed. This waste was hauled to a landfill; the distance was an uncertainty parameter (Table A11).

$$\mathbf{tkm} = \mathbf{M_T} * \mathbf{L_T} \qquad \textbf{Eq. A9}$$

Where:

tkm = tonne kilometers (tkm)
M_T = Mass of chemicals or solids (tonne)
L_T = Transport Distance (km)

A5. pH Adjustment

The final water's pH was raised to 8.2 at the end of the plant with caustic soda (sodium hydroxide). Other cases were considered where there was no pH adjustment as well as pH adjustment to the source water pH. Similar to the alum dose calculations, the caustic soda and lime doses were determined by generating a caustic soda (Table A10) or lime dose and final pH table for every source water scenario using three main steps. First, the pH after chlorination was calculated using Eq. A10. Second, a comprehensive range of possible caustic and lime doses was generated (from 0 to 2,000 mg/L in 0.1 mg/L increments until 15 mg/L, then 1.0 mg/L increments until 50 mg/L, then 5.0 mg/L increments until 300 mg/L, and then 100 mg/L increments until 2,000 mg/L). Third, the pH of the final adjusted water was calculated by iteratively solving Eq. A11, based on the U.S. EPA's Water Treatment Plant Model v2.[198] Overall, the required caustic or lime dose for the pH adjustment needed was found from the table. Figure A4 shows an example of how pH changed throughout the treatment train and displays the input and output pH at the point of each chemical addition. Table A10 shows example caustic doses and the corresponding final water pH (when the starting pH was 7.5).

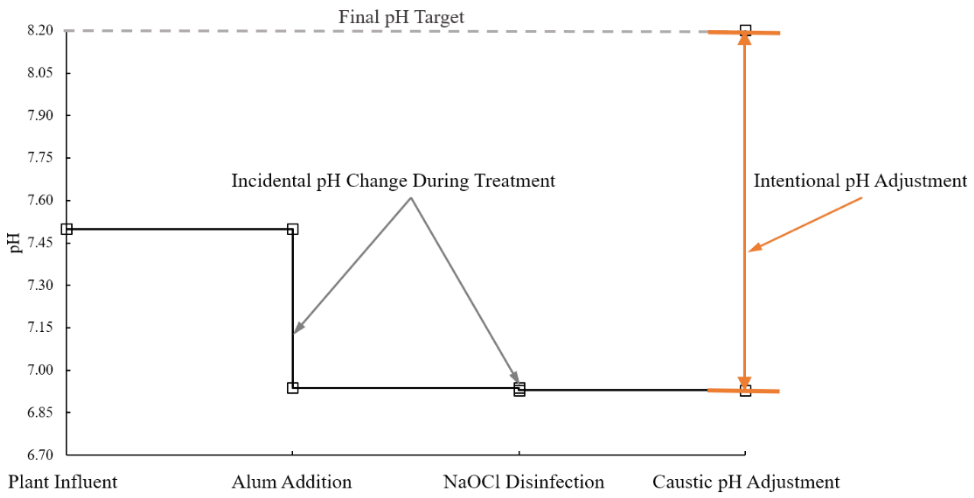

Figure A4. Example pH changes throughout the treatment process. Values were calculated for an example source water (77 mg/L $CaCO_3$, 3.2 mg/L TOC, 7.6 pH, 3.1 SUVA, 15 °C) and enhanced coagulation TOC removal.

Table A9. Example chlorine dose and free chlorine before the distribution system. Values were calculated for an example source water representing national averages (77 mg/L $CaCO_3$, 3.2 mg/L TOC, 7.6 pH, 3.1 SUVA, 15 °C) for enhanced coagulation TOC removal.

Initial Chlorine Added (mg/L as free Cl_2)	Final Chlorine Before Distribution (mg/L as free Cl_2)
1	0.66
1.1	0.75
…	…
1.4	1.01

Table A10. Example caustic dose and final pH table. Values were calculated for an example source water representing national averages (77 mg/L $CaCO_3$, 3.2 mg/L TOC, 7.6 pH, 3.1 SUVA, 15 °C) for enhanced coagulation TOC removal.

Caustic Added (mg/L)	Final pH
0	6.93
0.2	6.94
0.5	6.95
0.8	6.96
…	…
12.8	8.21

$$\left((\alpha_1 + 2) * \alpha_2 * \left[CO_3{}^{2-}\right]\right) + [OH^-] - [H^+] = [HCO_3{}^-] + 2 * \left[CO_3{}^{2-}\right] + [OH^-] - [H^+] - \left(\frac{[Ct_{OCl-}]}{1 + \frac{[H^+]}{k_{ocl-}}}\right) \qquad \textbf{Eq. A10a}$$

$$[Ct_{OCl^-}] = [Ct_{NaOCl}] = \frac{Cl_2\ Dose}{1000\,\frac{mg}{g} * \left(\frac{(2 * 35)g\ Cl_2}{1\ mol\ Cl_2}\right) * \left(\frac{1\ mol\ Cl_2}{2\ mol\ NaOCl}\right)} \qquad \textbf{Eq. A10b}$$

$$k_{OCl^-} = \exp\left\{\left[\left(\frac{13800\,\frac{J}{mol}}{8.314\,\frac{J}{K * mol}}\right)\left(\frac{1}{298.15\ K} - \frac{1}{T}\right)\right] - 17.5\right\} \qquad \textbf{Eq. A10c}$$

$$\left((\alpha_1 + 2) * \alpha_2 * \left[CO_3{}^{2-}\right]\right) + [OH^-] - [H^+] = [HCO_3{}^-] + 2 * \left[CO_3{}^{2-}\right] + [OH^-] - [H^+] + [Caustic] \qquad \textbf{Eq. A11}$$

Where:

$[OH^-]$ = Concentration of hydroxide (M)
$[H^+]$ = Concentration of hydrogen (M): Known target
$[CO_3{}^{2-}]$ = Concentration of carbonate (M)
$[HCO_3{}^-]$ = Concentration of bicarbonate (M)
$[Ct_{NaOCl}]$ = Concentration of sodium hypochlorite added (M) (Eq. A10b)
$[Ct_{OCl}$-] = Concentration of hypochlorite chlorine (M) (Eq. A10b)
k_{ocl}- = Hypochlorite equilibrium constant[199]
α_1 = Water chemistry equilibrium value for the second hydrogen state (Eq. Ab)
α_2 = Water chemistry equilibrium value for the third hydrogen state (Eq. Ac)
[Caustic] = Amount of caustic added (M)
T = Influent water temperature (K)
[Cl_2 Dose] = Required residual chlorine dose (mg Cl_2/L): Using Jones et. al.[91]

A6. Uncertainty and Sensitivity Analysis

Table A11. The low (L) and high (H) values of each uncertainty parameter. The typical (T) value represents the most likely value expected based on the range. Uncertainty parameters 1 to 8 used a triangular distribution with the known L, H, and T values; the rest had uniform distributions based on the known L and H values.

#	Uncertainty Parameter	Low Value	High Value	Typical Value	Basis and Citations
1	Minimum allowable alum dose (mg alum/L)	6.0	17	10	L=25th percentile value[95], H=75th percentile value[95], T=median[95]
2	Source water BDOC/TOC ratio	14%	27%	20%	L=min[85], H=max[85], T=median[85]
3	Ozonated water BDOC/TOC ratio	20%	38%	30%	L=min[85], H=max[85], T=median[48]
4	Biodegradable fraction of TOC removed by coagulation, when source water SUVA<3 L/mg/m	2.0%	5.0%	4.0%	L=25th percentile[52,72], H=75th percentile[52,72], T=average[52,72]
5	Biodegradable fraction of TOC removed by coagulation, when source water SUVA>3 L/mg/m	7.5%	14%	9.0%	L=25th percentile[52,72], H=75th percentile[52,72], T=average[52,72]
6	Nonozonated biofilter percent TOC removal (of the available biodegradable fraction of TOC) for 10°C to 20°C	5.0%	22%	10%	L=min[85], H=max[85], T=median[85]
7	Ozonated biofilter TOC removal (of the available biodegradable fraction of TOC) for 10°C to 20°C	3.0%	47%	13%	L=min[85], H=max[85], T=median[85]
8	Pre-ozonation dose (g O_3/g TOC)	0.25	1.6	0.50	L=min[85], H=max[85], T=median[85]
9	Air-fed ozone specific energy use (kWh/g O_3 generated)	0.018	0.022	0.020	L[101], H[102,103], T=average of L&H
10	Steel Tank Thickness (m)	0.14	0.55	0.27	L/H[97], T=Average
11	Steel Life Expectancy (yr)	30	60	45	L/H[79], T[80]
12	Backwash Flowrate ($m^3/h/m^2$)	30	60	50	L/H[98], T=expert judgment
13	Backwash Pressure (m)	8	10	9	L/H[98], T=average
14	Water Height Above Media (m)	1.5	2.5	2	L/H[96], T=average
15	Media Lifetime (yr)	15	25	20	Expert judgment
16	Hydraulic Loading Rate (m/h)	10	25	15	L/H[96], T=expert judgment
17	Anthracite Depth (m)	0.405	0.5	0.45	B= 0.45[96], L/H = ±10% of T
18	Sand Depth (m)	0.27	0.33	0.3	B= 0.3[96], L/H = ±10% of T
19	Chlorine Storage Tank Lifetime (yr)	30	35	30	L/H[79], T[80]
20	Chlorine Delivery Rate (trips/week)	0.5	2	1	L=every other week, H=twice a week, T=weekly
21	Chlorine Pump Head (m)	1.22	70.3	70.3	L/H[149], T=conservative
22	Concrete Basin Base Thickness (m)	0.30	0.61	0.46	L/H[201], T=average
23	Concrete Basin Wall Thickness (m)	0.23	0.46	0.46	L/H[201], T=same as base thickness
24	Baffling Factor for Tank with 2 Baffles	0.3	0.5	0.4	L/H[97], T=average
25	Baffle Thickness (cm)	3.8	4.5	4.5	L[202], H=[203](Fig. 1), T=conservative
26	Steel Baffle Life Expectancy (yr)	30	60	45	L/H[79], T[80]
27	Concrete Life Expectancy (yr)	30	60	30	L/H[79], T[80]
28	Chlorine Storage Tank Thickness (cm)	1.3	5.1	3.2	L/H[97], T=average
29	Landfill Hauling Distance (km)	20	100	20	L[204], H=expert judgement, T=most reliable estimate[204]
30	Chemical Hauling Distance (km)	20	100	20	L[204], H=expert judgement, T=most reliable estimate[204]
31	Ozone generator mass (kg/(g/hr))	0.8	18.7	9.8	L[105], H[104], T=average

A7. Typical Source Water Analysis

A7.1 UV Disinfection Compared to Chlorine Disinfection

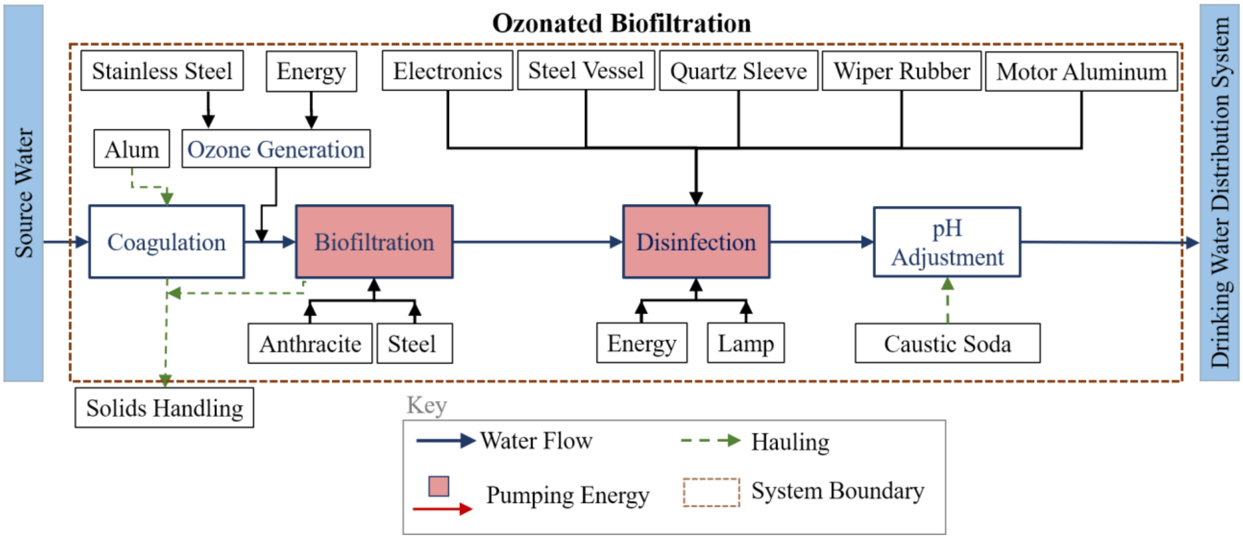

Figure A5. The LCA system boundary of the UV disinfection alternative treatment train, which included treatment processes (blue text and lines) as well as LCI unit processes for the materials and chemicals (black text and lines), for hauling (dashed green lines), and for energy (red lines or red fill color). Refer to the web version for proper interpretation of references to color in this figure.

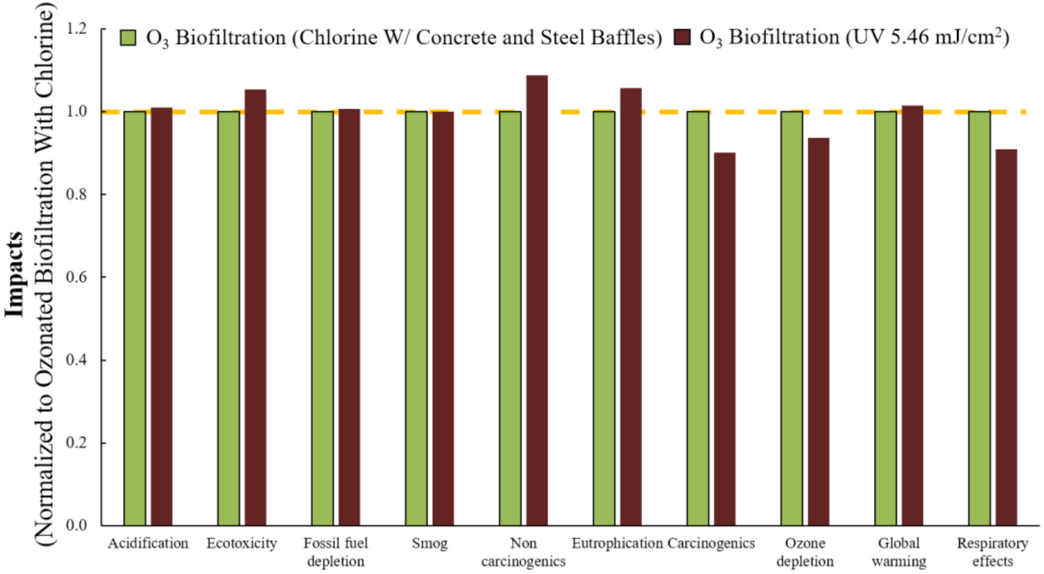

Figure A6. Ozonated biofiltration impacts using chlorine and ultraviolet (UV) disinfection using the model developed by Jones et. al. 2018, where the results are consistent for a comparison of chlorine with a concrete contact zone and steel baffles vs. UV (5.46 mJ/cm^2 dose) without chlorination.[91] Infrastructure and operations were scaled to the flow rate used in this manuscript (2730 m^3/day), and virus removal was expected to be achieved by ozonation and biofiltration. Ultimately, disinfection plays a minor role to total impacts compared to alum and caustic chemical dosing, and results were not significantly different across all impact categories. For detailed analysis of these trends see Jones et. al. 2018.[91]

S7.2 Typical Source Water Scenario Process Contribution

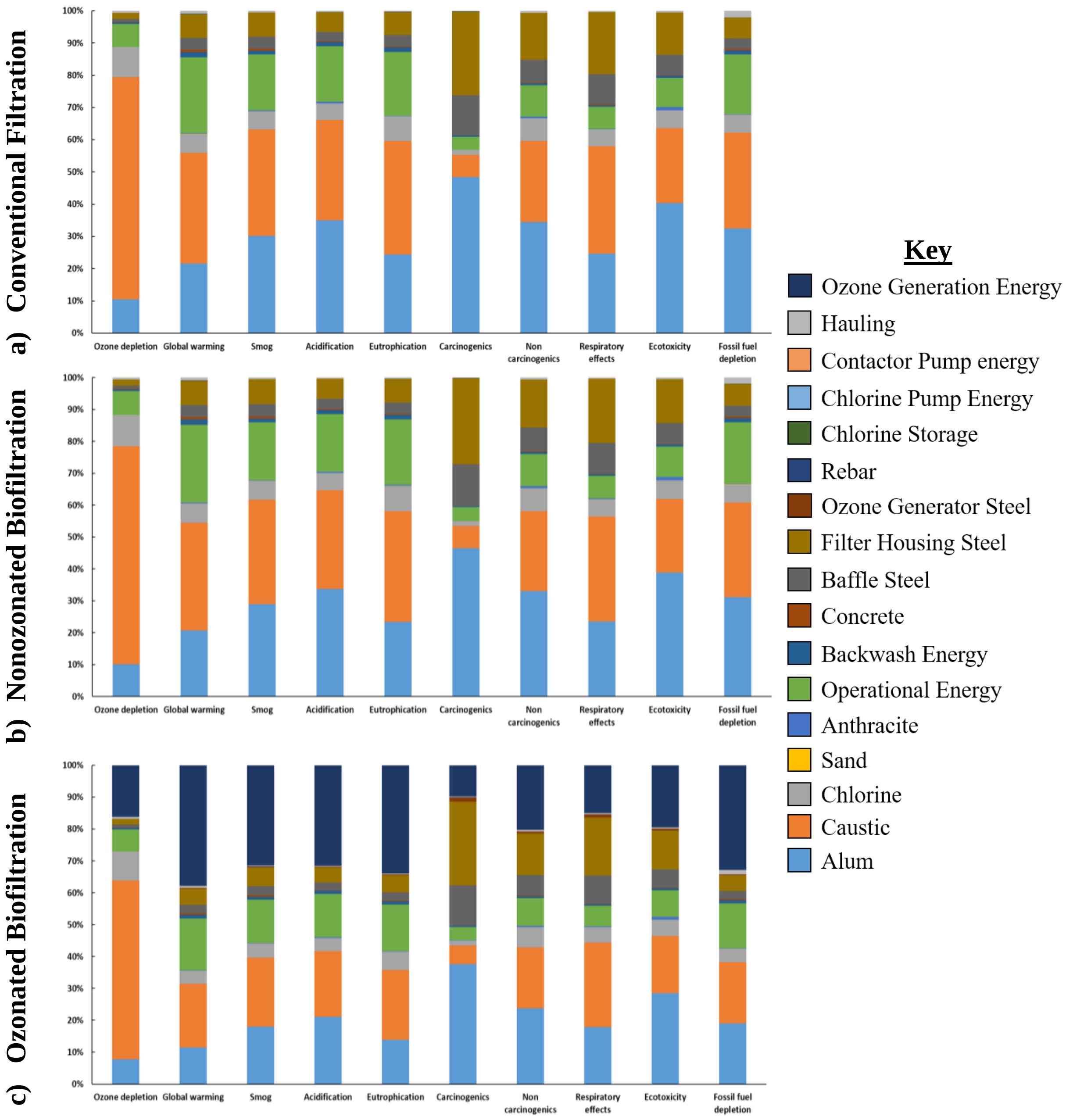

Figure A7. Impact breakdown in relation to total impacts for (a) conventional filtration, (b) nonozonated biofiltration, and (c) ozonated biofiltration for the typical source water scenario (from Figure 2.3).

A7.3 Alternative Chemical Analysis

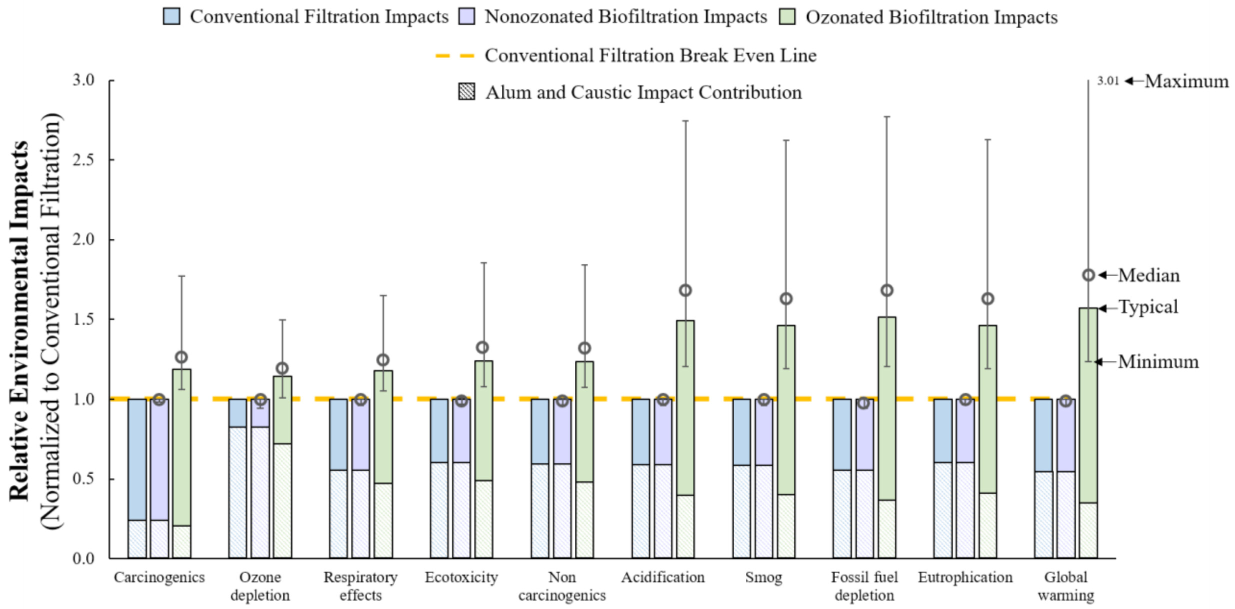

Figure A8. Same as Figure 2.3 in the main paper, except the coagulant used was ferric chloride. The distribution selections had no notable changes in results.

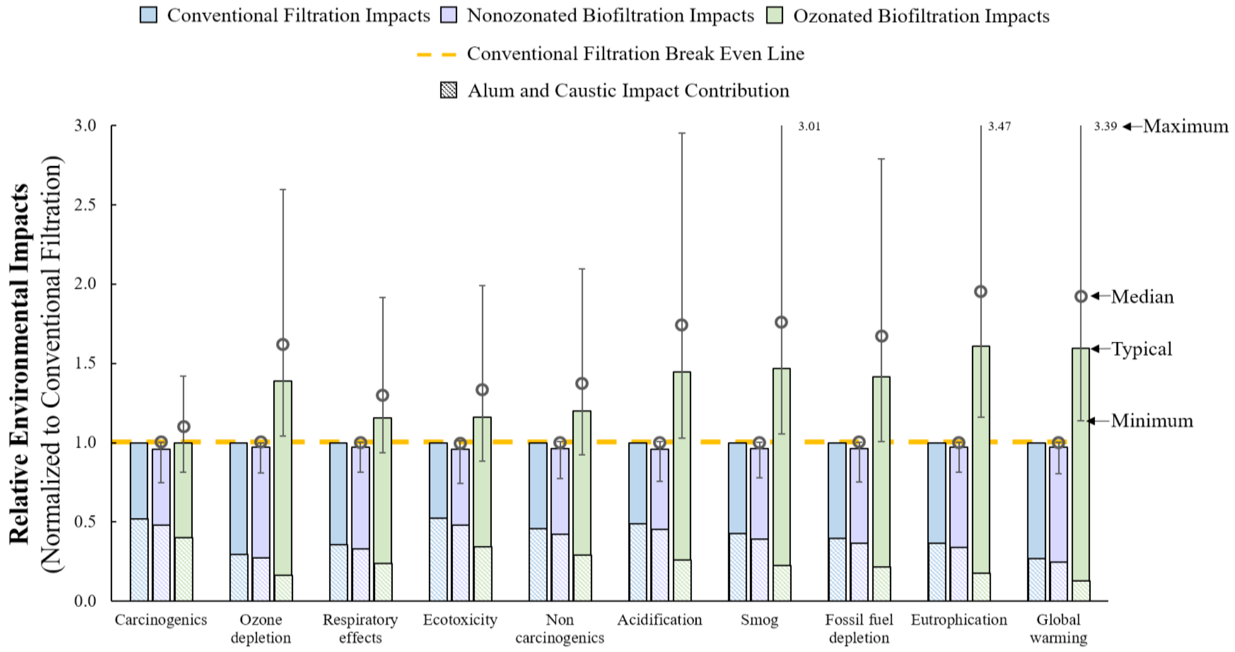

Figure A9. Same as Figure 2.3 in the main paper, except the pH adjustment chemical used was lime. The distribution selections had no notable changes in results.

A7.4 Ozone Uncertainty due to Source Water

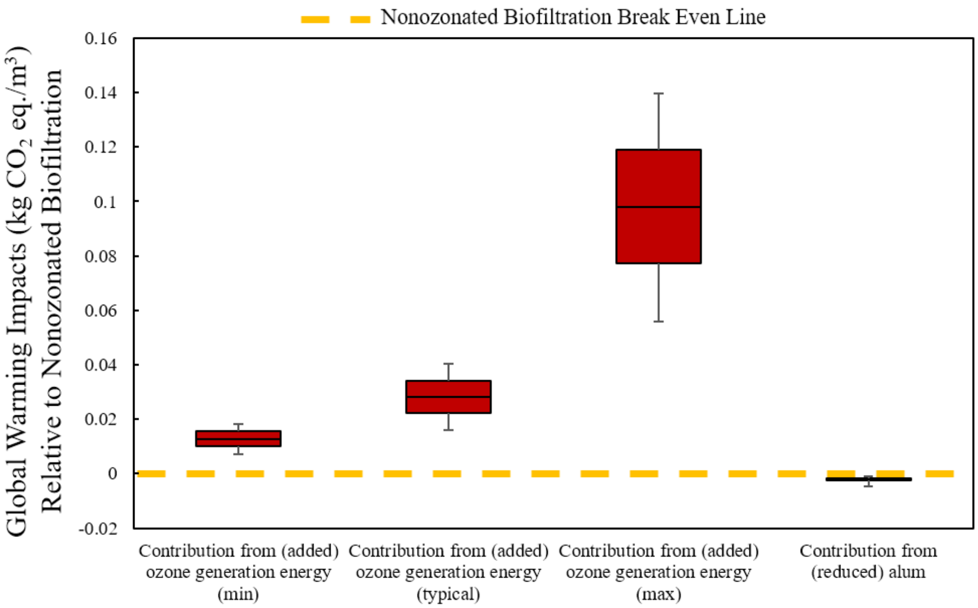

Figure A10. Results for ozone energy and alum offset impacts from using ozonation where error is due to source water quality as opposed to design uncertainty. Ozone energy results were discretely separated into low typical and high ozone energy impact scenarios. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the enhanced coagulation treatment scenario.

A8. Comprehensive Source Waters Analysis

A8.1 Additional Source Water Analysis

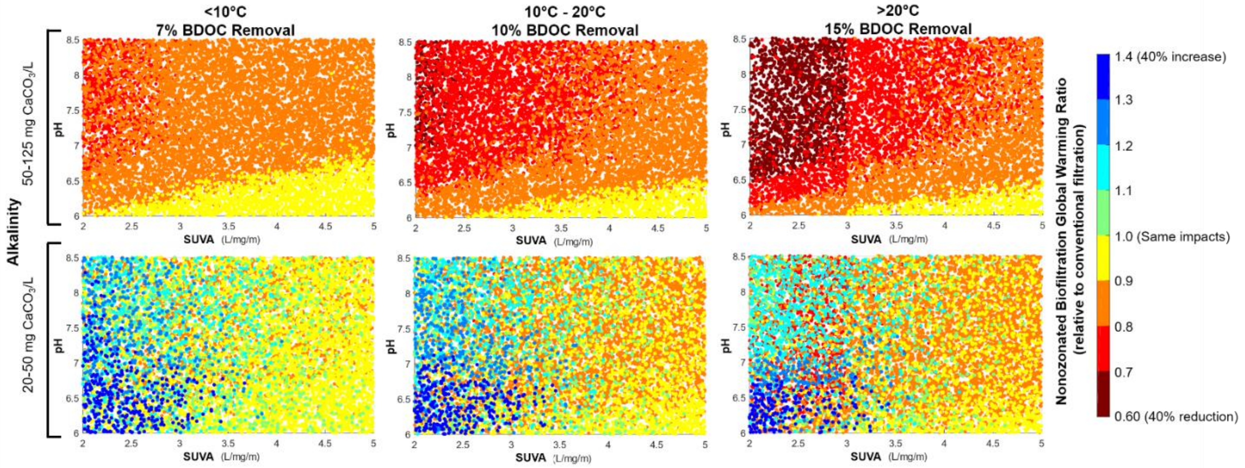

Figure A11. 15% biofilter TOC removal corresponded to performance expected above 20°C or above, 10% between 10°C and 20°C (typical), and 7% biofilter TOC removal corresponded to performance expected below 10°C. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

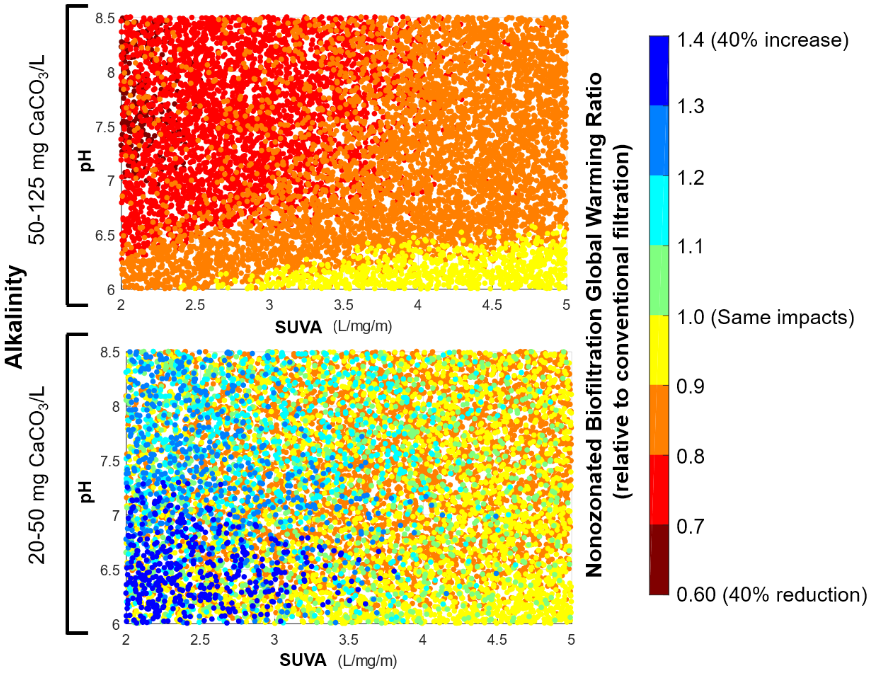

Figure A12. High TOC range (5 to 8 mg/L TOC) bins (excluded from Figure 2.5). Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

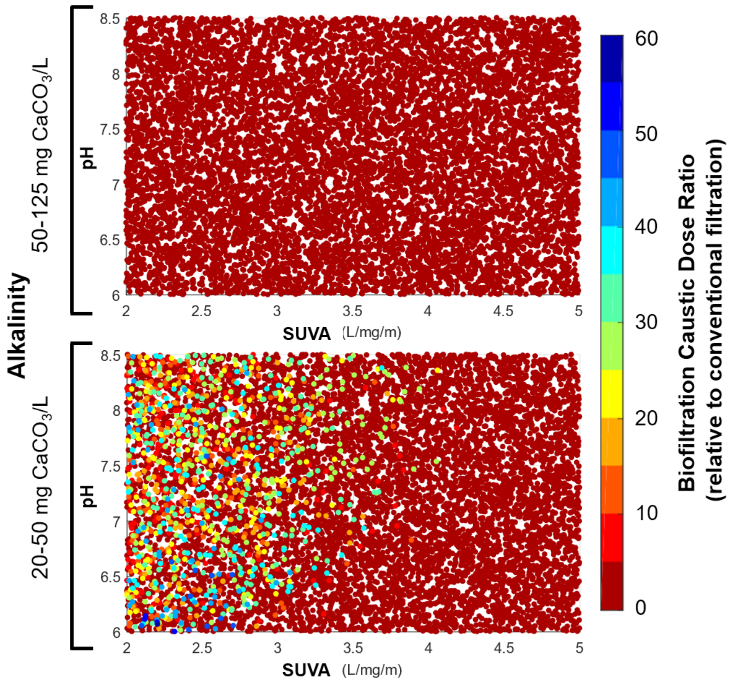

Figure A13. Biofiltration caustic soda dose compared to conventional filtration caustic dose for all 15°C scenarios. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

A8.2 pH Trend Analysis

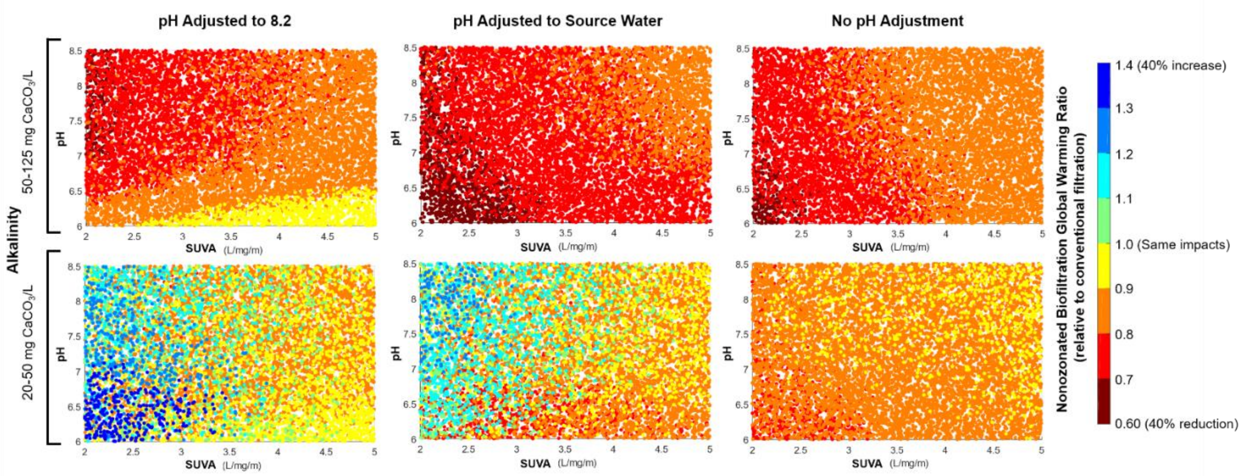

Figure A14. Trends when the caustic pH adjustment at the end of the plant is bringing the pH up to 8.2, pH back to the source water pH, and no pH adjustment. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

A8.3 Sensitive Parameters

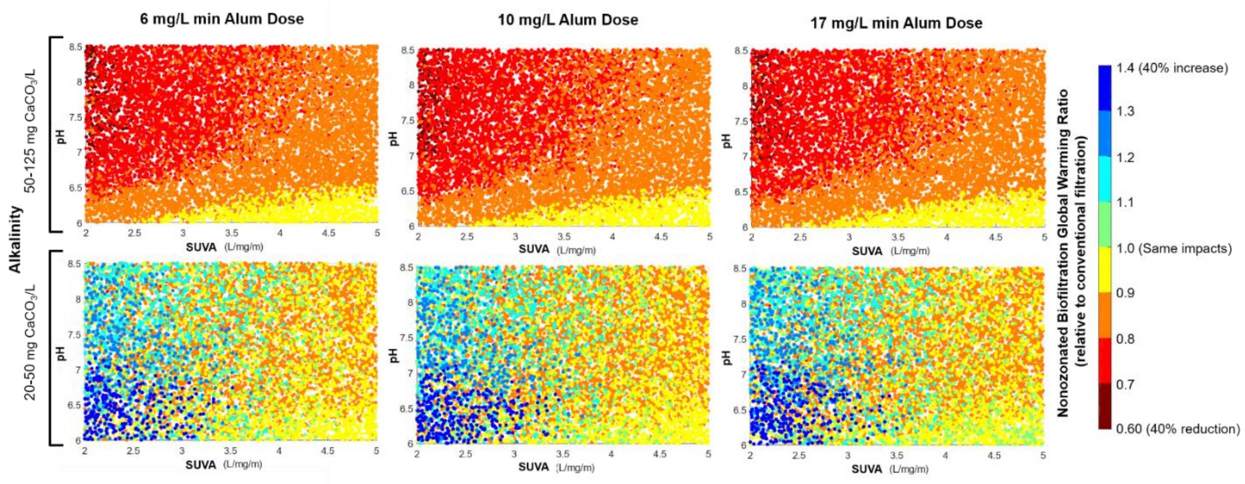

Figure A15. Figure 2.5 from main paper with a 6 mg/L, 10mg/L (typical), and 17 mg/L minimum allowable alum dose. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

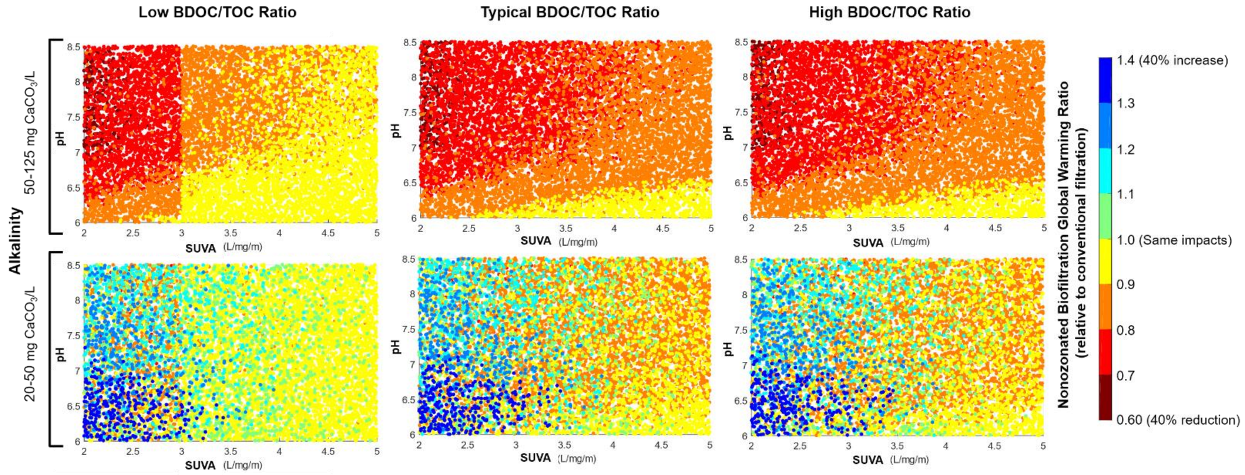

Figure A16. Figure 2.5 from the main paper with a source water BDOC/TOC ratio of 0.14, 0.20 (typical), and 0.27. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

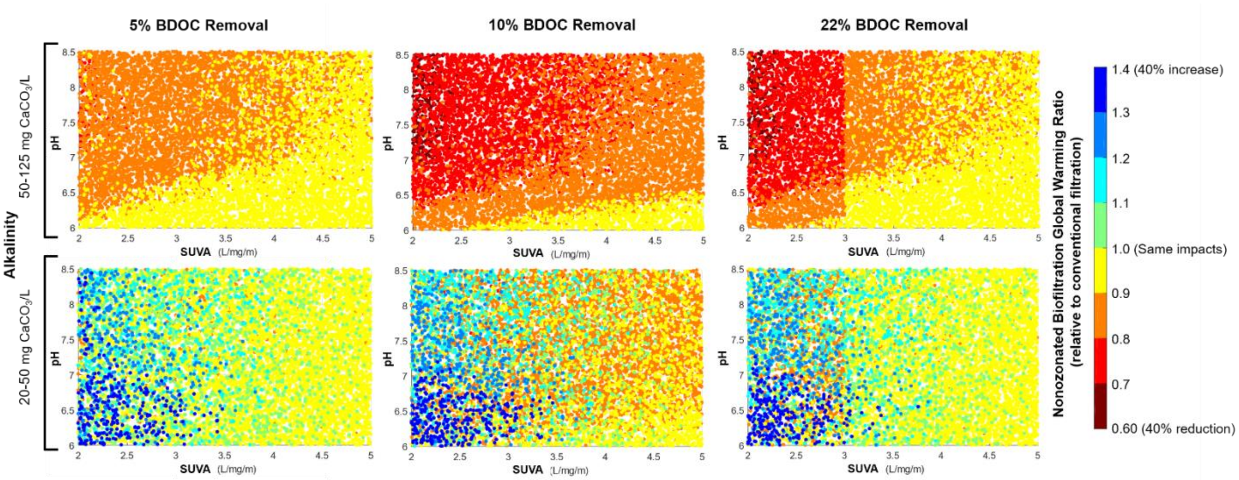

Figure A17. Figure 2.5 from the main paper with 5%, 10% (typical), and 22% BDOC removal through nonozonated biofiltration. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the 50% treatment scenario.

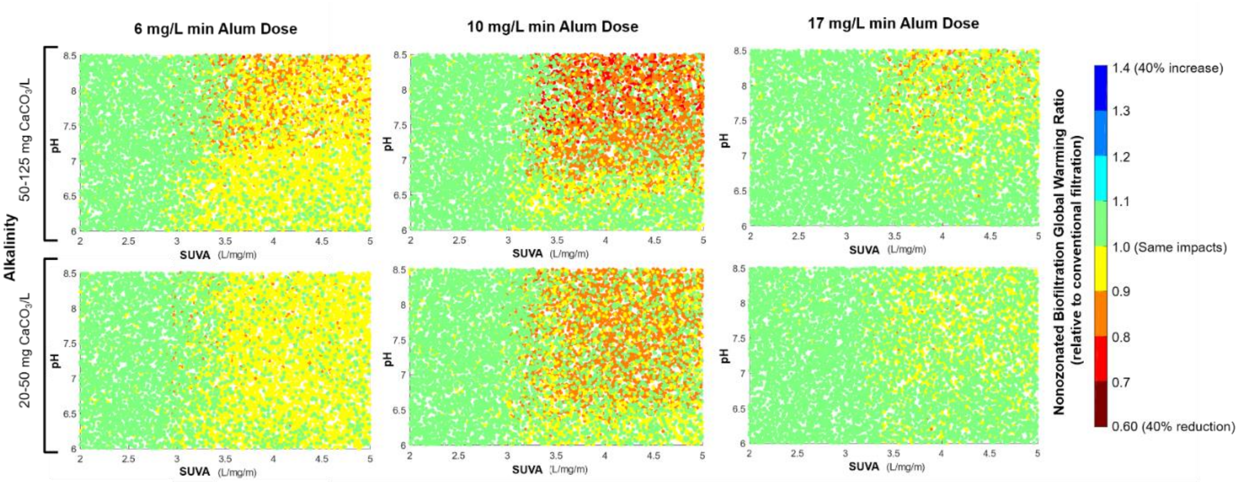

Figure A18. Figure 2.5 from main paper with a 6 mg/L, 10 mg/L (typical), and 17 mg/L minimum allowable alum dose. Comprehensive source water analysis (20,000 source waters at 15°C) evaluated under the enhanced coagulation treatment scenario.

Appendix B: Supporting Information for: "Life Cycle Environmental Impacts of Disinfection Technologies Used in Small Drinking Water Systems"

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B1. Life Cycle Inventory and Impact Assessment

Table B1. Ecoinvent life cycle inventory unit process names. (Note: all were Alloc Def U.)

Unit Process	Section	Ecoinvent Process Name	Application
Aluminum	B5.2	Aluminium, wrought alloy {GLO}| aluminium ingot, primary, to market	UV wiper motor
Chlorine (Sodium hypochlorite)	B4.1, B5.3	Sodium hypochlorite, without water, in 15% solution state {RoW}| sodium hypochlorite production, product in 15% solution state	Free chlorine
Concrete	B4.2	Concrete, 20MPa{RoW}| concrete production 20 MPa, RNA only	Concrete (chlorine contact basin)
Electricity	B3, B4.3, B4.4, B5.1	Electricity, medium voltage {ASCC}| market for	ASCC grid (grid contribution data not available[197])
		Electricity, medium voltage {FRCC}| market for	FRCC grid (6% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {NPCC, US only}| market for	NPCC grid (7% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {MRO, US only}| market for	MRO grid (17% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {RFC}| market for	RFC grid (20% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {SERC}| market for	SERC grid (17% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {SPP}| market for	SPP grid (6% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {TRE}| market for	TRE grid (9% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {WECC, US only}| market for	WECC grid (18% total US electricity, EIA 2014 energy use data per grid[197])
		Electricity, medium voltage {HICC}| market for	HICC grid (grid contribution data not available[197])
Electronics	B5.2	Electronics, for control units {RoW}| production	UV electronics (wires and housing)
Glass	B3, B5.2	Glass tube, borosilicate {RoW}| production	UV quartz sleeve and cartridge filter (portion of total composition)
Hard Plastic	B4.2	Polypropylene, granulate {RoW}| production	Plastic baffles (chlorine contact basin)
Lamp	B5.2	Ultraviolet lamp {GLO}| ultraviolet lamp production, for water disinfection	UV lamp (Note: lamp equivalent units were converted to mass by summing the individual component masses in this unit process.)
PVC	B4.2	Polyvinylchloride, suspension polymerised {RoW}| polyvinylchloride production, suspension polymerisation |	PVC pipe (chlorine contact basin)
Reinforcing Steel	B4.2	Reinforcing steel {RoW}| market for	rebar (chlorine concrete basin)
Rubber	B5.2	Synthetic rubber {RoW}| production	UV sleeve wiper
Soft Plastic	B4.2, B4.3	Polyethylene, high density, granulate {RoW}| production	Plastic cylindrical tank (chlorine contact and chlorine storage basin)
Stainless Steel	B3, B4.2, B5.2	Steel, chromium steel 18/8, hot rolled {RoW}| production	UV vessel, steel baffles (cl contactor), tank (cl contactor), filter housing
Tap Water	B4.3	Tap water{RoW} | tap water production, conventional treatment |	Dilution water (chlorine solution)

Table B2. Material, energy, and chemical quantities for the 9 disinfection alternatives in the typical source water scenario. Values are for the functional unit of 1 m^3 over 40 years (for a reference flow of 273 m^3/day) and are the median output from the Monte Carlo analysis. Unit process details are in Figure B1. Chlorine mass is kg 15% sodium hypochlorite solution. Acronyms: N/A is not applicable; VF is validation factor; eq. equivalents; cl is chlorine; UV is ultraviolet.

Inventory Unit Process (Units)	**Disinfection Alternatives**								
	Cl Plastic Tank	Cl Pipe	Cl Plastic Baffles	Cl Concrete	Cl Steel Tank	Cl Steel Baffles	UV 1.4 VF	UV 2.6 VF	UV 4.4 VF
Aluminum (kg)	N/A	N/A	N/A	N/A	N/A	N/A	6.22E-06	6.22E-06	6.22E-06
Chlorine (kg)	1.50E-03	1.50E-03	1.50E-03	1.50E-03	1.50E-03	1.50E-03	1.30E-03	1.30E-03	1.30E-03
Cl dose (mg Cl_2/L)	1.5	1.5	1.5	1.5	1.5	1.5	1.3	1.3	1.3
Concrete (m^3)	N/A	N/A	3.44E-06	1.14E-05	N/A	3.44E-06	N/A	N/A	N/A
Electricity (UV lamp) (kWh)	N/A	N/A	N/A	N/A	N/A	N/A	1.31E-03	2.44E-03	4.11E-03
Electricity (filtration pumping) (kWh)	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02	1.60E-02
Electricity (contactor pumping, both Cl and UV) (kWh)	7.88E-12	7.95E-06	2.07E-09	2.66E-09	6.92E-12	2.07E-09	3.79E-03	3.79E-03	3.79E-03
Electricity (Cl dosing) (kWh)	3.41E-06	3.41E-06	3.41E-06	3.41E-06	3.41E-06	3.41E-06	2.96E-06	2.96E-06	2.96E-06
Electronics (kg)	N/A	N/A	N/A	N/A	N/A	N/A	2.32E-05	2.32E-05	2.32E-05
Glass(sleeve)(kg)	N/A	N/A	N/A	N/A	N/A	N/A	2.61E-07	2.61E-07	2.61E-07
Glass (filter) (kg)	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04
Hard Plastic (kg)	N/A	N/A	1.17E-04	N/A	N/A	N/A	N/A	N/A	N/A
Lamp (kg)	N/A	N/A	N/A	N/A	N/A	N/A	6.52E-06	6.52E-06	6.52E-06
PVC (kg)	N/A	82.7	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Reinforcing Steel (kg)	N/A	N/A	2.11E-05	6.67E-05	N/A	2.11E-05	N/A	N/A	N/A
Rubber (kg)	N/A	N/A	N/A	N/A	N/A	N/A	8.10E-10	8.10E-10	8.10E-10
Soft Plastic (cl storage) (kg)	6.60E-06	6.60E-06	6.60E-06	6.60E-06	6.60E-06	6.60E-06	6.22E-06	6.22E-06	6.22E-06
Soft Plastic (contactor) (kg)	9.03E-05	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Stainless Steel (baffles) (kg)	N/A	N/A	N/A	N/A	N/A	9.56E-04	N/A	N/A	N/A
Stainless Steel (contactor) (kg)	N/A	N/A	N/A	N/A	7.02E-04	N/A	N/A	N/A	N/A
Stainless Steel (filter housing) (kg)	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05
Stainless Steel (UV vessel) (kg)	N/A	N/A	N/A	N/A	N/A	N/A	7.35E-06	7.35E-06	7.35E-06
Tap Water (kg)	1.95E-02	1.95E-02	1.95E-02	1.95E-02	1.95E-02	1.95E-02	1.69E-02	1.69E-02	1.69E-02

Table B3. Material, energy, and chemical quantities for the 3 disinfection alternatives in the chlorine residual exemption scenario. Values are for the functional unit of 1 m^3 over 40 years (for a reference flow of 273 m^3/day) and are the median output from the Monte Carlo analysis. Unit process details are in Figure B1. Chlorine mass is kg 15% sodium hypochlorite solution. Acronyms: N/A is not applicable; VF is validation factor; eq. equivalents; cl is chlorine; UV is ultraviolet.

Inventory Unit Process (Units)	**Disinfection Alternatives**		
	Chlorine Concrete	UV (5.46 mJ/cm^2)	UV (186 mJ/cm^2)
Aluminum (kg)	N/A	6.22E-06	6.22E-06
Chlorine (kg)	1.50E-03	1.30E-03	N/A
Cl dose (mg Cl_2/L)	1.5	1.3	N/A
Concrete (m^3)	1.14E-05	N/A	N/A
Electricity (UV lamp) (kWh)	N/A	2.44E-03	8.30E-02
Electricity (filtration pumping) (kWh)	1.60E-02	1.60E-02	1.60E-02
Electricity (contactor pumping, both Cl and UV) (kWh)	2.66E-09	3.79E-03	3.79E-03
Electricity (Cl dosing) (kWh)	3.41E-06	2.96E-06	N/A
Electronics (kg)	N/A	2.32E-05	2.32E-05
Glass(sleeve)(kg)	N/A	2.61E-07	2.61E-07
Glass (filter) (kg)	1.03E-04	1.03E-04	1.03E-04
Hard Plastic (kg)	N/A	N/A	N/A
Lamp (kg)	N/A	6.52E-06	6.52E-06
PVC (kg)	N/A	N/A	N/A
Reinforcing Steel (kg)	6.67E-05	N/A	N/A
Rubber (kg)	N/A	8.10E-10	8.10E-10
Soft Plastic (cl storage) (kg)	6.60E-06	6.22E-06	N/A
Soft Plastic (contactor) (kg)	N/A	N/A	N/A
Stainless Steel (baffles) (kg)	N/A	N/A	N/A
Stainless Steel (contactor) (kg)	N/A	N/A	N/A
Stainless Steel (filter housing) (kg)	1.52E-05	1.52E-05	1.52E-05
Stainless Steel (UV vessel) (kg)	N/A	7.35E-06	7.35E-06
Tap Water (kg)	1.95E-02	1.69E-02	N/A

Table B4. Material, energy, and chemical quantities for the 9 disinfection alternatives in the filtration exemption scenario. The filter pump head loss values for the chlorine alternatives are noted for each alternative. Values are for the functional unit of 1 m^3 over 40 years (for a reference flow of 273 m^3/day) and are the median output from the Monte Carlo analysis. Unit process details are in Figure B1. Chlorine mass is kg 15% sodium hypochlorite solution. Acronyms: N/A is not applicable; VF is validation factor; eq. equivalents; cl is chlorine; UV is ultraviolet.

Inventory Unit Process (Units)	**Disinfection Alternatives**								
	Cl Pipe (0.7 m)	Cl Pipe (3.5 m)	Cl Pipe (11 m)	Cl Concrete (0.7 m)	Cl Concrete (3.5 m)	Cl Concrete (11 m)	UV 1.2 VF	UV 1.6 VF	UV 1.9 VF
Aluminum (kg)	N/A	N/A	N/A	N/A	N/A	N/A	6.22E-06	6.22E-06	6.22E-06
Chlorine (kg)	1.50E-03	1.50E-03	1.50E-03	1.50E-03	1.50E-03	1.50E-03	1.30E-03	1.30E-03	1.30E-03
Cl dose (mg Cl_2/L)	1.5	1.5	1.5	1.5	1.5	1.5	1.3	1.3	1.3
Concrete (m^3)	N/A	N/A	N/A	1.14E-05	1.14E-05	1.14E-05	N/A	N/A	N/A
Electricity (UV lamp) (kWh)	N/A	N/A	N/A	N/A	N/A	N/A	6.42E-03	8.58E-03	1.02E-02
Electricity (filtration pumping) (kWh)	3.19E-03	1.60E-02	4.79E-02	3.19E-03	1.60E-02	4.79E-02	N/A	N/A	N/A
Electricity (contactor pumping, Cl and UV) (kWh)	7.95E-06	7.95E-06	7.95E-06	2.66E-09	2.66E-09	2.66E-09	3.79E-03	3.79E-03	3.79E-03
Electricity (Cl dosing) (kWh)	3.41E-06	3.41E-06	3.41E-06	3.41E-06	3.41E-06	3.41E-06	2.96E-06	2.96E-06	2.96E-06
Electronics (kg)	N/A	N/A	N/A	N/A	N/A	N/A	2.32E-05	2.32E-05	2.32E-05
Glass(sleeve)(kg)	N/A	N/A	N/A	N/A	N/A	N/A	2.61E-07	2.61E-07	2.61E-07
Glass (filter) (kg)	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04	1.03E-04	N/A	N/A	N/A
Hard Plastic (kg)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Lamp (kg)	N/A	N/A	N/A	N/A	N/A	N/A	6.52E-06	6.52E-06	6.52E-06
PVC (kg)	2.07E-05	2.07E-05	2.07E-05	N/A	N/A	N/A	N/A	N/A	N/A
Reinforcing Steel (kg)	N/A	N/A	N/A	6.67E-05	6.67E-05	6.67E-05	N/A	N/A	N/A
Rubber (kg)	N/A	N/A	N/A	N/A	N/A	N/A	8.10E-10	8.10E-10	8.10E-10
Soft Plastic (cl storage) (kg)	6.60E-06	6.60E-06	6.60E-06	6.60E-06	6.60E-06	6.60E-06	6.22E-06	6.22E-06	6.22E-06
Soft Plastic (contactor) (kg)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Stainless Steel (baffles) (kg)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Stainless Steel (contactor) (kg)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Stainless Steel (filter housing) (kg)	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05	1.52E-05	N/A	N/A	N/A
Stainless Steel (UV vessel) (kg)	N/A	N/A	N/A	N/A	N/A	N/A	7.35E-06	7.35E-06	7.35E-06
Tap Water (kg)	1.95E-02	1.95E-02	1.95E-02	1.95E-02	1.95E-02	1.95E-02	1.69E-02	1.69E-02	1.69E-02

Table B5. TRACI environmental impact category descriptions.[83]

Impact Category	Unit	Description
Ozone depletion	kg CFC-11 eq	Ozone provides protection from radiation. Emissions of substances known as chlorofluorocarbons (CFCs) reduce stratospheric ozone levels.
Carcinogenics	CTUh	Substances are chemicals of concern that can cause cancer in humans.
Non Carcinogenics	CTUh	Substances are chemicals of concern that are toxic to humans but do not cause cancer.
Respiratory effects	kg $PM_{2.5}$ eq	Fine particulate matter and precursors to particulates ambient in the air can be inhaled by a human and cause respiratory illnesses or even death.
Eutrophication	kg N eq	Excess of nutrients in a body of water results in dense growth of plants and algae and a reduction of oxygen.
Acidification	kg SO_2 eq	A decrease in the pH of water because of the uptake of CO2 and SOx.
Smog	kg O_3 eq	Ground level ozone can cause respiratory illnesses and ecosystem damages. Ozone is created with the presence of nitrogen oxides (NOx) and volatile organic compounds (VOCs).
Fossil fuel depletion	MJ surplus	Non-site specific use of fossil fuels.
Global warming	kg CO_2 eq	The raising of the Earth's atmospheric temperature due to the increase in CO2 and other greenhouse gas emissions. Global warming has many additional adverse climate and health effects.
Ecotoxicity	CTUe	Substances are chemicals of concern that are toxic to the ecosystem.

B2. Uncertainty and Sensitivity Data

Table B6. Uncertainty parameter values and basis. Low (L) and high (H) values were used for the Monte Carlo analysis, assuming a uniform probability distribution.

#	Uncertainty Parameter	Low Value	High Value	Basis
1	Head loss in UV chamber (m)	0.15	1.52	[116]
2	UV Electronics Mass (kg)	4.1	15	L= sum (UV Sensor, Ballast, min UVT analyzer), H= sum (UV Sensor, Ballast, max UVT analyzer), (Section B4)
3	Electronics Life Expectancy (yrs)	3	5	[116]
4	Quartz Sleeve Life Expectancy (yrs)	8	10	[116]
5	Quartz Sleeve/Wiper Thickness (cm)	0.10	0.20	[116]
6	Wiper Life Expectancy	3	5	[116]
7	UV Vessel Mass (kg)	26.4	39.6	Existing system ± 20%
8	UV Lamp Power at 40 mJ/cm^2 (kWh/m^3)	0.0079	0.0249	Range of existing systems (Section B4)
9	UV Lamp Length (m)	1.2	1.5	Range of existing systems (Section B4)
10	UV Lamp Weight (kg)	0.5	1.2	Range of existing systems (Section B4)
11	UV Lamp Life Expectancy (hrs)	8000	15000	[116]
12	UV Quartz Sleeve Outer Diameter (cm)	2.5	5	[116]
13	Chlorine Storage Tank Life Expectancy (yrs)	30	35	[79]
14	Chlorine Delivery Rate (hauling trips/week)	0.5	2	L=Every other week, H=Twice per week
15	Chlorine Pump Head (m)	1.22	70.3	[149]
16	Concrete Life Expectancy (yr)	30	60	[79]
17	Concrete Wall Thickness (m)	0.23	0.46	[201]
18	Concrete Base Thickness (m)	0.30	0.61	[201]
19	Unbaffled Tank Baffling Factor	0.1	0.3	[97]
20	Two Baffle Tank Baffling Factor	0.3	0.5	[97]
21	Baffle Thickness (cm)	3.8	4.5	L=[202], H=Derived geometrically (from Figure 1 in [203])
22	Polyethylene Cylindrical Contactor Life Expectancy (yrs)	30	60	[79]
23	Stainless Steel Life Expectancy (yrs)	30	60	[79]
24	Cylindrical Basin Baffling Factor	0.2	0.3	[97]
25	Cylindrical Basin Thickness (cm)	1.3	5.1	[97]
26	PVC Life Expectancy (yrs)	35	40	[79]
27	PVC Thickness (cm)	0.60	1.7	[150]
28	PVC Piping Baffling Factor	0.6	1	[97]
29	Filter Replacement (times/yr)	0.5	4	L=replace at least every 2 years, H= replace at most four times a year

B3. Filtration for *Cryptosporidium* Removal

The chlorine system needed filtration processes to achieve 2-log *Cryptosporidium* removal in every scenario. The main unit processes included for filtration were: filtration pumping energy, filter housing material, and filter material. Eq. B15 was used to determine the filtration energy required to maintain flow through the filter. Head loss was assumed to be between 0.7 to 11 meters,[141–146] with a maximum allowable head loss of 21.1 meters.[137] The expected value was 3.5 meters.[147] Cartridge filtration housing units are typically composed of stainless steel, and the quantity was determined from an existing cartridge filtration system's specification sheet.[141] Likewise, the mass of filters was estimated using existing cartridge filters, and the dominant material was borosilicate microglass.[141–146] Filter replacement (uncertainty parameter) was assumed to be between every other year at a minimum, which represented negligible particulate fouling, and seasonally at a maximum.

B4. Chlorine Disinfection Design

The chlorine disinfection alternatives included: (i) a place of contact (contact zone) where the chlorine was added to the water and allowed time to interact with the pollutants, (ii) a chlorine storage tank, (iii) energy for pumping the chlorine into the contact zone from the storage tank, and (iv) energy for pumping the water through the contact zone. The amount of each contact zone material, chlorine, and energy was quantified using the following equations and information.

B4.1 Chlorine Dose and Contact & CT

EPA tables were used to estimate concentration times time (CT) values (Reference (148)) based on inactivation target, water quality, and residual. The contact time was then calculated using the CT value and chlorine residual Eq. B1. The chlorine dose must be large enough to meet chlorine demand, so that the chlorine does not decay below the desired residual (1 mg/L) in the contact zone. Chlorine dose was determined using the U.S. EPA's Water Treatment Plant Model v2.0,[92] which predicts free chlorine concentration in water as a function of CT and the influent's total organic carbon (TOC) and UV transmittance (UVT); Eq. B2 was used to compute the chlorine dose needed to compensate for chlorine decay and residual for a given water quality.[92] This indeterminate equation was solved iteratively until convergence. The dose was calculated to be 1.5 mg/L for the following source water: 2.8 mg/L TOC, 82% UVT, 7.5 pH, and 15°C (national average values[89]).

$$\boldsymbol{T_{contact} = \frac{CT}{C_{res}}} \qquad \textbf{Eq. B1}$$

where:

$T_{contact}$ = chlorine contact time (min)
CT = concentration times time (mg*min/L)
C_{res} = desired free chlorine residual exiting the plant (mg/L)

$$\boldsymbol{C_{res} = \{-0.8404(C_0)\} * ln\left(\frac{C_0}{C_{res}}\right) + \left\{-0.404\left(\frac{C_0}{UVA}\right)^{-0.9108}\right\} * TOC * T_{contact} + C_0} \qquad \textbf{Eq. B2}$$

where:

C_{res} = residual free chlorine concentration (mg Cl_2/L)
C_0 = initial chlorine concentration = dose (mg Cl_2/L)
UVA = ultraviolet absorbance of raw water (1/cm)
TOC = total organic carbon in raw water (mg/L)
$T_{contact}$ = contact time (min)

B4.2 Contact Zone Material

Three types of contact zones were designed: (1) a rectangular, open, concrete basin with and without baffles; (2) a closed, cylindrical tank; and (3) PVC piping in a serpentine layout. The chlorine contact time Eq. B1 was used to determine the contact tank volume Eq. B3. All contact time was assumed to happen in the contact zone (e.g., there was no credit awarded for time spent in a clearwell), except for the UV disinfection alternatives were chlorine contact time for virus inactivation was assumed to happen between the injection point and entrance to the distribution system. The baffling factor was an based on typical values[97] (uncertainty parameter, Table B6).

$$\boldsymbol{V_{zone} = \frac{T_{contact} * Q}{BF}} \qquad \textbf{Eq. B3}$$

where:

V_{zone} = contact zone volume (m^3)
$T_{contact}$ = chlorine contact time (hr)
Q = flow rate (m^3/hr)
BF = baffling factor

B4.2.1 Concrete Tank

The dimensions of the concrete tank were based on traditional design guidelines, which state that the length must be 10 to 100 times larger than the channel width and depth must be 1 to 3 times larger than the channel width.[98] The middle value for each range was used (50 and 2, respectively). For baffled tanks, common design requires that the length must be 5 to 15 times the channel width and that depth must be 1 to 3 times larger than the channel width;[98] middle values for both were used (10 and 2, respectively). Also, the length must be 2 to 5 times larger than the tank width,[98] and this requirement was met given the previously chosen ratio values.

To calculate tank dimensions for either the baffled or unbaffled tank, the channel width was first calculated Eq. B4, then the tank length Eq. B5, depth Eq. B6, and width Eq. B7 were calculated. The corresponding volume of concrete was calculated assuming a concrete thickness (uncertainty parameter, Table B6) and using Eq. B8. The amount of reinforcing steel needed for this concrete was calculated by using Eq. B10 while assuming rebar will be spaced every 0.305 meters over the concrete cross for the length of each concrete segment.[97] The mass of baffling material was calculated using Eq. B9 and the following baffle assumptions: 4.5 cm thick,[203] height was the same as tank depth, and length was the tank length minus the channel width.

$$\boldsymbol{W_c = \sqrt[3]{\frac{V_{zone}}{(N_{channels}) * \left(\frac{L}{W_c}\right) * \left(\frac{D}{W_c}\right)}}} \qquad \textbf{Eq. B4}$$

$$\boldsymbol{L = \left(\frac{L}{W_c}\right) * W_c} \qquad \textbf{Eq. B5}$$

$$\boldsymbol{D = \left(\frac{D}{W_c}\right) * W_c} \quad \textbf{Eq. B6}$$

$$\boldsymbol{W_t = W_c * N_{channels}} \quad \textbf{Eq. B7}$$

Where:

W_c = width of channel (m)
W_t = width of tank (m)
D = depth of tank (m)
L = length of tank (m)
$N_{channels}$ = number of channels (equal to number of baffles + 1)
V_{zone} = Contact zone volume (m^3)

$$\boldsymbol{V_{concrete} = (L * W_t * t_B) + \left(4t_w^2 + 2L * t_w + 2\,W * t_w\right) * (D + t_B)} \quad \textbf{Eq. B8}$$

Where:

$V_{concrete}$ = concrete volume (m^3)
t_B = thickness of concrete base (m)
t_W = concrete outer wall thickness (Table B6) (m)

$$\boldsymbol{M_{baffle} = (L - W_c) * D * t_{baffle} * N_{baffles} * \rho} \quad \textbf{Eq. B9}$$

Where:

V_{baffle} = baffle volume (m^3)
t_{baffle} = baffle thickness (Table B6) (m)
$N_{baffles}$ = number of baffles
ρ = density of steel (kg/m^3): 7500 kg/m^3 from ([150])

$$\boldsymbol{M_{rebar} = \left(\frac{W_t * \mathrm{L} + \mathrm{D} * \mathrm{L} + \mathrm{D} * W_t}{W_s}\right) * \mathrm{R}_{M/L}} \quad \textbf{Eq. B10}$$

Where:

M_{rebar} = weight of reinforcing rebar (kg) (value was rounded up to near integer)
W_s = width of rebar spacing (m): 0.305 m from ([127])
$R_{M/L}$ = mass to length ratio of rebar (kg/m): 1 kg/m from ([127])

B4.2.2 Cylindrical Tank

The amount of plastic or stainless steel needed for the cylindrical container was based on the tank dimensions Eq. B11, which assumed the depth was twice the diameter and a material thickness (uncertainty parameter, Table B6) Eq. B12. The material type depended on flow and typical small systems data; plastic was used for volumes less than 19 m^3 and stainless steel for larger volumes.[97]

$$\boldsymbol{d_{cyl}} = \sqrt[3]{\frac{\mathbf{2} * \boldsymbol{V_{zone}}}{\boldsymbol{\pi}}} \qquad \textbf{Eq. B11}$$

Where:

d_{cyl} = diameter of cylinder (Table B6) (m)
V_{zone} = Contact zone volume (m^3)

$$\boldsymbol{M_{cyl}} = \left(\frac{\boldsymbol{H\pi}}{\mathbf{4}}\left(\left(\boldsymbol{d_{cyl}} + \boldsymbol{t_{cyl}}\right)^{\mathbf{2}} - \left(\boldsymbol{d_{cyl}}\right)^{\mathbf{2}}\right) + \frac{\boldsymbol{\pi} * \boldsymbol{t_{cyl}} * \left(\boldsymbol{d_{cyl}} + \boldsymbol{t_{cyl}}\right)^{\mathbf{2}}}{\mathbf{2}}\right) * \boldsymbol{\rho} \qquad \textbf{Eq. B12}$$

Where:

M_{cyl} = mass of cylinder material (kg)
t_{cyl} = thickness of contact tank (Table B6) (m)
H = height of cylinder (m): 2 * d_{cyl}
ρ = density of cylinder (kg/m^3): 7500 (steel) or 950 (polypropylene)[150]

B4.2.3 Serpentine Plastic Pipe

The pipe diameter was chosen based on the flow rate needed to achieve a baffling factor of one,[97] which was calculated based on typical minimum flow requirements, pipe length, pipe length to diameter ratios (must be greater than or equal to 160),[97] and the CT value. Then, pipe length was determined using Eq. B13. PVC volume was calculated by multiplying the pipe's cylindrical surface area times the pipe thickness (Table B6) Eq. B14.

$$\boldsymbol{L = \frac{V_{zone}}{\pi\left(\frac{d_p}{2}\right)^2}} \qquad \textbf{Eq. B13}$$

Where:

L = length of pipe (m)
$V_{zone.}$ = contact zone volume (m^3)
d_p = diameter of pipe (from reference (97)) (m)

$$\boldsymbol{M_{PVC} = L * \pi\left(\left(\frac{d_p + t_{pipe}}{2}\right)^2 - \left(d_p\right)^2\right) * \rho} \qquad \textbf{Eq. B14}$$

Where:

M_{PVC} = mass of PVC piping (kg)
t_{pipe} = thickness of PVC piping (Table B6) (m)
L = length of pipe (m)
d_p = diameter of pipe (from reference (97)) (m)
ρ = density of PVC (kg/m^3): 1390 kg/m^3 from ([150])

B4.3 Chlorine Storage and Pumping Energy

The chlorine chemical storage tank dimensions were calculated using the same method as for the cylindrical contact tank, using Eq. B11 and Eq. B12. The volume of chlorine was based on a once a week (every 7 days) delivery rate with the required sodium hypochlorite (NaOCl) mass (or flow). The mass of sodium hypochlorite was based on the amount of free chlorine required for disinfection goals. Specifically, the mass of free chlorine as Cl_2 was converted to a mass of NaOCl (dry), which is the sole source of chlorine, using molecular weights and molar relationships. Then, this dry mass of NaOCl was converted to a mass of 15% NaOCl solution. The NaOCl unit process only includes the impacts associated with the NaOCl, so the amount of water needed for this 15% solution was also calculated. The solution's density was assumed to be 1,200 kg/m^3.[150] The chlorine feed pump power was calculated using Eq. B15, which was based the chlorine flow rate and typical pump efficiency (60%) and pump pressure (70.3 m).[149] The power was multiplied by the functional timeframe to get energy and normalized to 1 m^3 of water treated.

$$\boldsymbol{P = \frac{(Q * \rho * g * \ H)}{\left(1000\frac{W}{kW} * \ \eta\right)}} \qquad \textbf{Eq. B15}$$

Where:

P = power (kW)
Q = flow rate (m^3/s)
ρ = density of liquid solution (kg/m^3)
g = gravity (9.81 m/s^2)
H = metering pump pressure or head loss (m)
η = efficiency

B4.4 Contact Zone Pumping Energy

Pumping energy required to overcome major head loss (due to friction) and minor head loss (due to bends in flow) in each contact zone were determined using the following equations. For open concrete basins, the major head losses associated with friction were determined using the Mannings equation Eq. B16. The Hazen-Williams equation was used to determine the major frictional head losses in the cylindrical tank and plastic pipe Eq. B17. Minor head losses in the plastic pipe, in a serpentine layout, were calculated using Eq. B18; the number of bends were determined by first setting the ratio of the segment length to pipe diameter to 40 (Colorado regulatory recommendation[97]) and then dividing the total pipe length by segment length. Pumping requirements associated with each contact zones total head loss was calculated using Eq. B15.

$$\boldsymbol{H_f = \left(\frac{v * n}{R_h^{\frac{2}{3}}}\right)^2 * L * N_{Channels}}$$ **Eq. B16**

Where:

H_f = frictional head loss (m)
v = water velocity (m/s)
R_h = hydraulic radius (m) = (W_c*D) / (W_c + 2D)
n = Manning's roughness coefficient (0.015)[150]
L = total flow length (m)
$N_{Channels}$ = number of channels

$$\boldsymbol{H_f = \frac{(0.2083) * \left(\frac{100}{C}\right)^{1.852} * Q^{1.852}}{d_p{}^{4.8655} * 100m} * L}$$ **Eq. B17**

Where:

C = Hazen-Williams roughness coefficient = 150 (for PVC and steel) and 140 (for polyethylene plastic)[150]
d_p = pipe/cylinder diameter (from reference (97)) (m)
L = length of pipe/height of cylinder (m)

$$\boldsymbol{H_m = \frac{\varepsilon v^2}{2g} N_{bends}}$$ **Eq. B18**

Where:

H_m = minor head loss (m)
ε = minor loss coefficient = 1.5 (return bend, threaded 180°)[150]
v = water velocity (m/s)
g = gravity constant (9.81 m/s^2)
N_{bends} = number of 180° return bends

B5. UV Disinfection Design

The low pressure UV (LPUV) system design included the: (i) energy required to operate UV lamps; (ii) materials for the vessel, quartz sleeve, and lamp; (iii) cleaning system materials (rubber wipers and aluminum wiper motor); (iv) control electronics; (v) vessel pumping energy (due to head loss); and (vi) for chlorine residual included liquid sodium hypochlorite production, storage tank production, and injection pumping energy (Section B4). Existing LPUV systems were characterized to estimate energy use and materials. All of these systems were small, commercially available, and validated according to U.S. EPA approved protocols for drinking water. Data was collected from reports, specification sheets, and personal communications([79,84,209–211,116,152,153,155,205–208]) and generalized to protect proprietary data.

B5.1 UV Energy

Energy values for existing LPUV systems operating at a 40 mJ/cm^2 dose were normalized to the corresponding flow rate (between 30 to 83 m^3/hr); the energy range was 7.9 to 25 Wh/m^3, which was an uncertainty parameter to account for variability between existing systems and operation. To estimate energy requirements for other doses, energy and dose were assumed to be linearly proportional.[212] The LT2ESWTR required UV dose was defined using U.S. EPA guidance.[84,116] The dose corresponding to each scenario's UV disinfection goal was selected and then multiplied by a validation factor to account for uncertainties inherent in the validation process. Table B7 shows the validation factors used for 1-log *Giardia* and 3-log *Crypto*. The energy requirements for pumping water through the UV vessel were determined using Eq. B15 and an assumed head loss between 0.15 to 1.52 meters.[116]

Table B7. Validation factors reported from UV disinfection validation centers.[152,153]

UVT	1-Log *Giardia*		3-Log *Cryptosporidium*	
	Min	Max	Min	Max
70%	1.9	4.4	1.4	1.9
82%	1.8	4.1	1.3	1.8
99%	1.4	2.0	1.2	1.7

B5.2 UV Materials

The modeled UV unit was assumed to be a one lamp system (typical of existing small systems). Table B8 shows the material data for a one lamp (300 to 750 W) system. Many components were assumed flow independent for small systems since minimum requirements would need to be met regardless of how low the flow was (e.g., at least one UV sensor would be needed). Lamp mass was estimated from existing systems (Table B8). The quartz sleeve mass was calculated using Eq. B19), using typical sleeve thicknesses and diameters (Table B6) and typical lamp lengths (Table B8). The main materials associated with the quartz sleeve cleaning system were rubber wipers. The wipers were assumed to have a torus shape, so the mass was calculated using Eq. B20; wiper thickness was assumed to have the same radial thickness as the quartz sleeve (Table B6). Wiper mass was estimated from this volume by assuming a rubber density of 1500 kg/m^3.[150] The wiper motor mass was estimated from existing wiper motors.[154] Major electronic components for UV systems include UVT analyzers, ballasts, UV intensity sensors, temperature sensors, and lamp electrical connections;[116] the mass of major electronics were summed. This total mass was an uncertainty parameter due to the large range of possible masses (e.g., UVT analyzer mass had a large range) and the uncertainty about the exact electronics to be used by a small system (e.g., some small systems may not use a UVT analyzer[155]). The electronics LCI data accounted for a mix of metals, plastics, wires, and circuit boards.

Table B8. UV parameters used to quantify materials associated with UV.

UV System Component	Unit	Value(s)	Reference	Material Lifetime
Lamp mass	kg	0.5 – 1.2	[155,210,211]	8000 – 15000 hrs[116]
Lamp length	m	1.2 – 1.5	[206,207]	8000 – 15000 hrs[116]
Quartz sleeve mass	kg	0.41 – 2.05	Eq. B19, (Table B6)	8 – 10 yrs[116]
Wiper mass	g	0.96 – 7.66	Eq. B20	3 – 5 yrs[116]
Vessel mass	kg	33 (± 20%)	[206]	30 – 60 yrs[79]
UVT analyzer mass	kg	0.73 – 10	[213–215]	3 – 5 yrs[116]
UV sensor mass	kg	0.08 – 1.33	[155,210,211]	3 – 5 yrs[116]
Ballast mass	kg	2.00 – 3.19	[155,210,211]	3 – 5 yrs[116]

$$M_{Quartz} = \left(\frac{\pi}{4}\left({d_q}^2\right) * L_{lamp}\right) - \left(\frac{\pi}{4}\left({d_q}^2 - {t_q}^2\right) * L_{lamp}\right) * \rho \qquad \textbf{Eq. B19}$$

Where:

M_{Quartz} = mass of quartz sleeve (kg)

d_q = outer quartz diameter (Table B6) (m)

t_q = quartz thickness (Table B6) (m)

L_{lamp} = lamp length of existing systems (Table B6) (m)

ρ = density of quartz (kg/m^3): 2000 kg/m^3 from ([150])

$$M_{wiper} = \left(2 * \pi * r_q + 0.5 * t_q\right) * \pi * {t_q}^2 * \rho \qquad \textbf{Eq. B20}$$

Where:

M_{wiper} = mass of wiper material (kg)

r_q = outer quartz radius (Table B6) (m)

t_q = quartz sleeve and wiper thickness (Table B6) (m)

ρ = density of rubber (kg/m^3): 1522 kg/m^3 from ([150])

B6. Results

B6.1 Process Contributions

B6.1.1 Typical Source Water Scenario

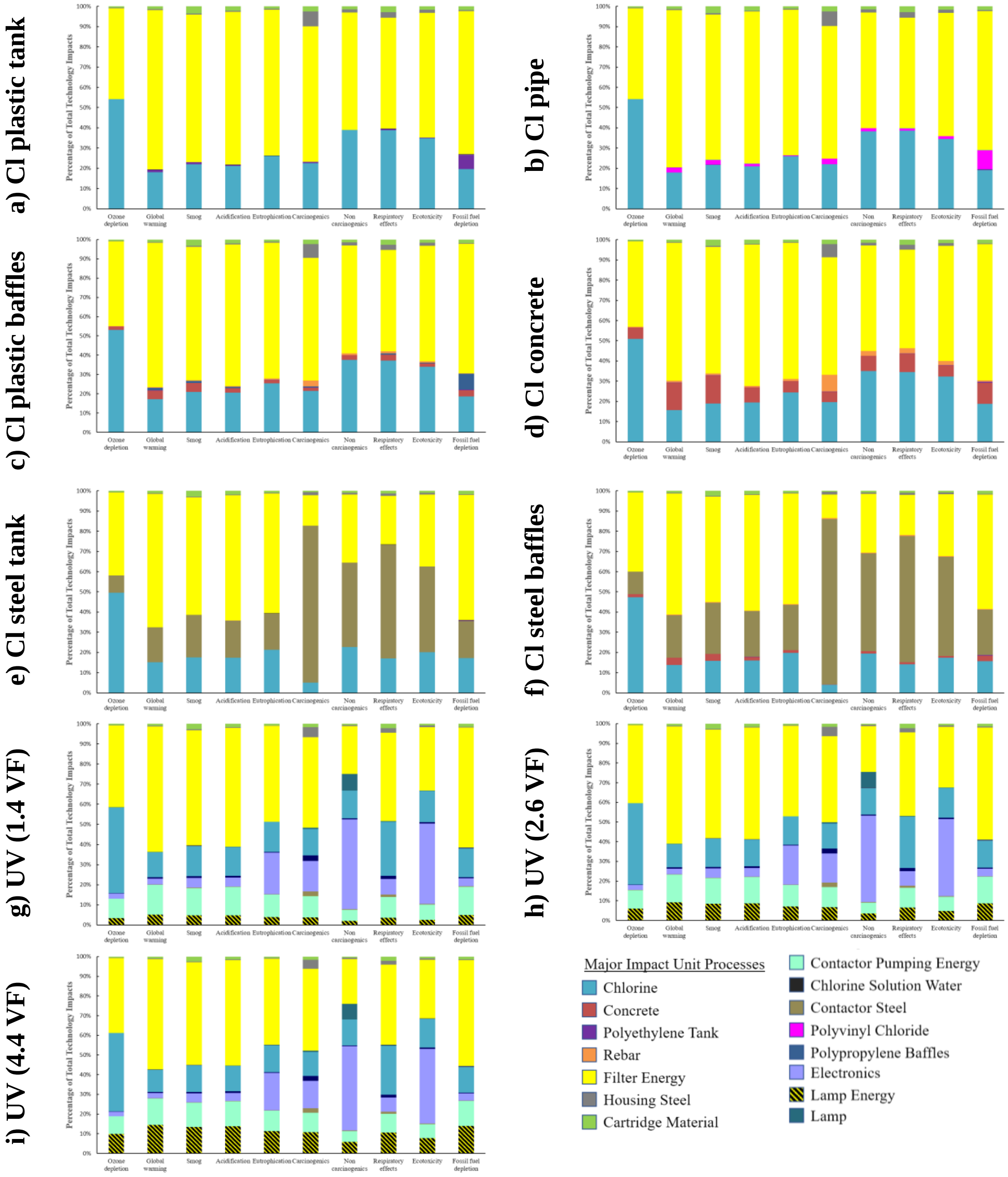


Figure B1. Process contribution graphs for each typical source water scenario treatment alternative: a) Cl plastic tank, b) Cl pipe, c) Cl concrete with plastic baffles, d) Cl concrete, e) Cl steel tank, f) Cl concrete with steel baffles, g) UV (1.4 VF), h) UV (2.6 VF), i) UV (4.4 VF). Note, VF is validation factor and Cl is chlorine.

B6.1.2 Filtration Exemption Scenario

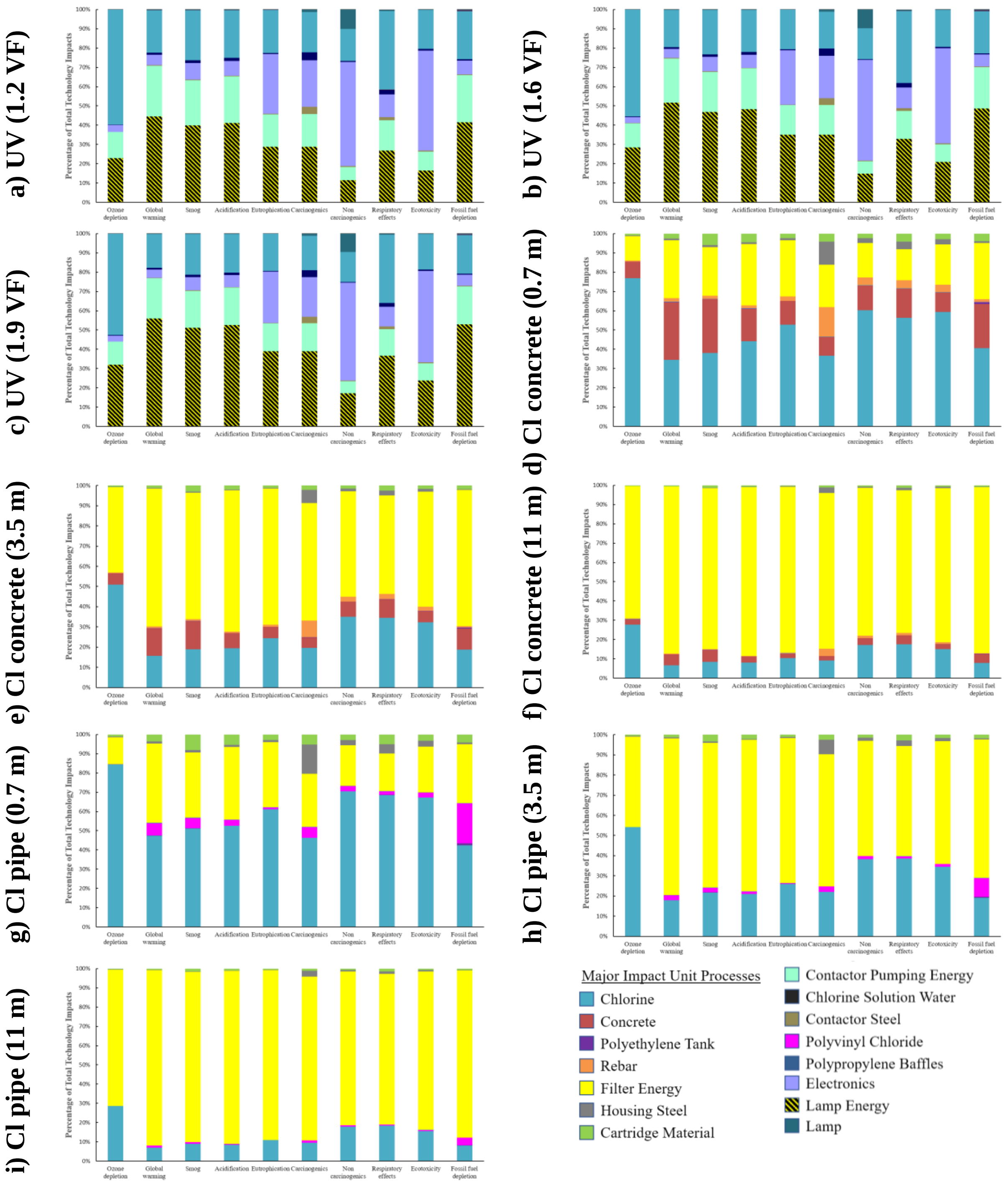


Figure B2. Process contribution graphs for each filtration exemption scenario treatment alternative: a) UV (0.2 VF), b) UV (1.6 VF), c) UV (1.9 VF), d) Cl Concrete (0.7 m), e) Cl Concrete (3.5 m), f) Cl Concrete (11 m), g) Cl Pipe (0.7 m), h) Cl Pipe (4 m), i) Cl Pipe (11 m). Note, VF is validation factor and Cl is chlorine.

B6.2 Uncertainty and Sensitivity

B6.2.1 Typical Source Water Scenario

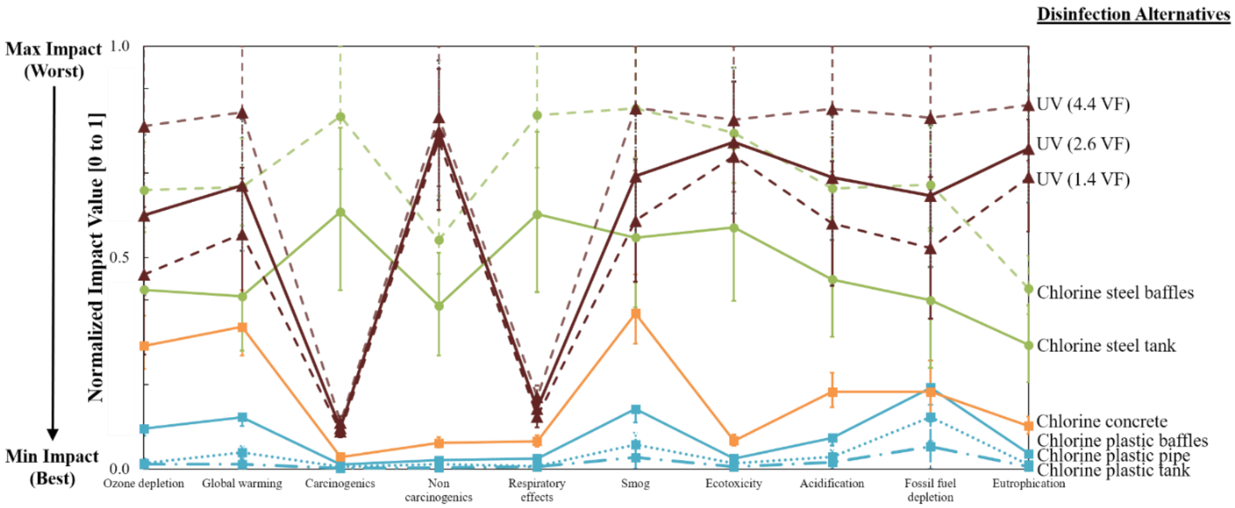


Figure B3. Parallel coordinate plot showing the comparative assessment with uncertainty ranges (error bars representing 25th to 75th percentiles) for 6 chlorine contactor alternatives and 3 UV validation factor alternatives (VF is validation factor).

Table B9. Typical source water scenario results that were sensitive to uncertainty parameters (|ρ| > 0.8).

Uncertainty Parameter	Alternative	Impact Categories
Filter Replacement Rate	Chlorine Plastic Pipe, Chlorine Plastic Tank	Acidification, Ecotoxicity, Eutrophication, Ozone Depletion, Respiratory Effects, Smog, Non Carcinogenics (for plastic tank only)
PVC Thickness	Chlorine Plastic Pipe	Fossil Fuel Depletion
Plastic Thickness for Cylindrical Contact Basin	Chlorine Plastic Tank	Fossil Fuel Depletion
Steel Thickness for Cylindrical Contact Basin	Cl Steel Tank	All Categories
Head Loss in UV Chamber	All UV Alternatives	Acidification, Fossil Fuel Depletion, Global Warming, Ozone Depletion, Smog, Respiratory Effects (Except for 4.4 VF)
UV Electronics Mass	All UV Alternatives	Ecotoxicity, Non Carcinogenics

B6.2.2 Chlorine Residual Exemption Scenario

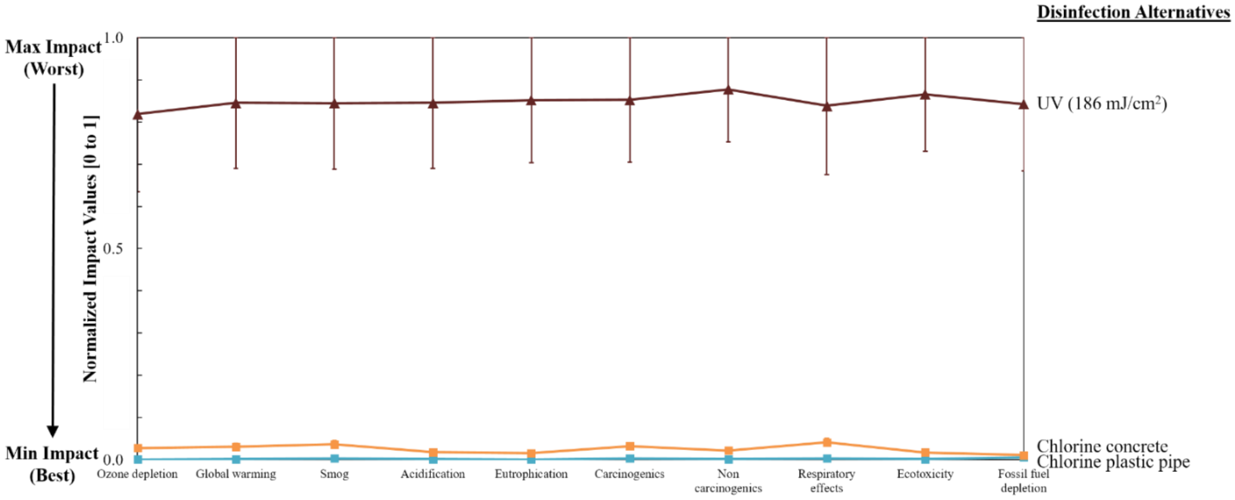


Figure B4. Parallel coordinate plot showing the comparative assessment with uncertainty ranges (error bars representing 25th – 75th percentile) for 2 chlorine contactor alternatives and 1 UV alternative (using virus inactivation dose and no chlorine addition). Note, error bars for the chlorine alternatives are smaller than the markers.

Table B10. Chlorine residual exemption scenario results that were sensitive to uncertainty parameters ($|\rho| > 0.8$).

Uncertainty Parameter	Alternative	Impact Categories
Filter Replacement Rate	Chlorine Plastic Pipe	Acidification, Ecotoxicity, Eutrophication, Ozone Depletion, Respiratory Effects, Smog
PVC Thickness	Chlorine Plastic Pipe	Fossil Fuel Depletion
UV Lamp Power	All UV Alternatives	All Categories

B6.2.3 Filtration Exemption Scenario

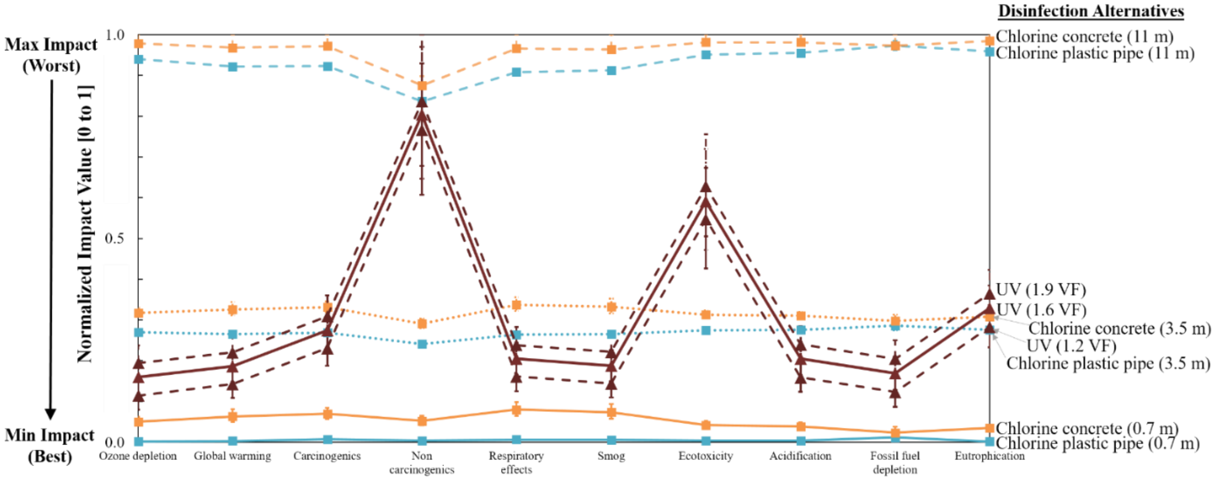


Figure B5. Parallel coordinate plot showing the comparative assessment with uncertainty ranges (error bars representing 25th – 75th percentile) for 2 chlorine contactor alternatives (concrete basin and PVC serpentine pipe) at 3 different filter pressures (0.7 m, 3.5 m, 11 m) and 3 UV validation factor alternatives (VF is validation factor). Note, most error bars for the chlorine alternatives are smaller than the markers.

Table B11. Filtration Exemption scenario results that were sensitive to uncertainty parameters (|ρ| > 0.8).

Uncertainty Parameter	Alternative	Impact Categories
Filter Replacement Rate	Chlorine Plastic Pipe (all filter head losses)	Acidification, Ecotoxicity, Eutrophication, Ozone Depletion, Respiratory Effects, Smog
PVC Thickness	Chlorine Plastic Pipe (all filter head losses)	Fossil Fuel Depletion
UV Electronics Mass	All UV Alternatives	Ecotoxicity, Non Carcinogenics

B6.3 Electricity Source Impacts

B6.3.1 Typical Source Water Scenario

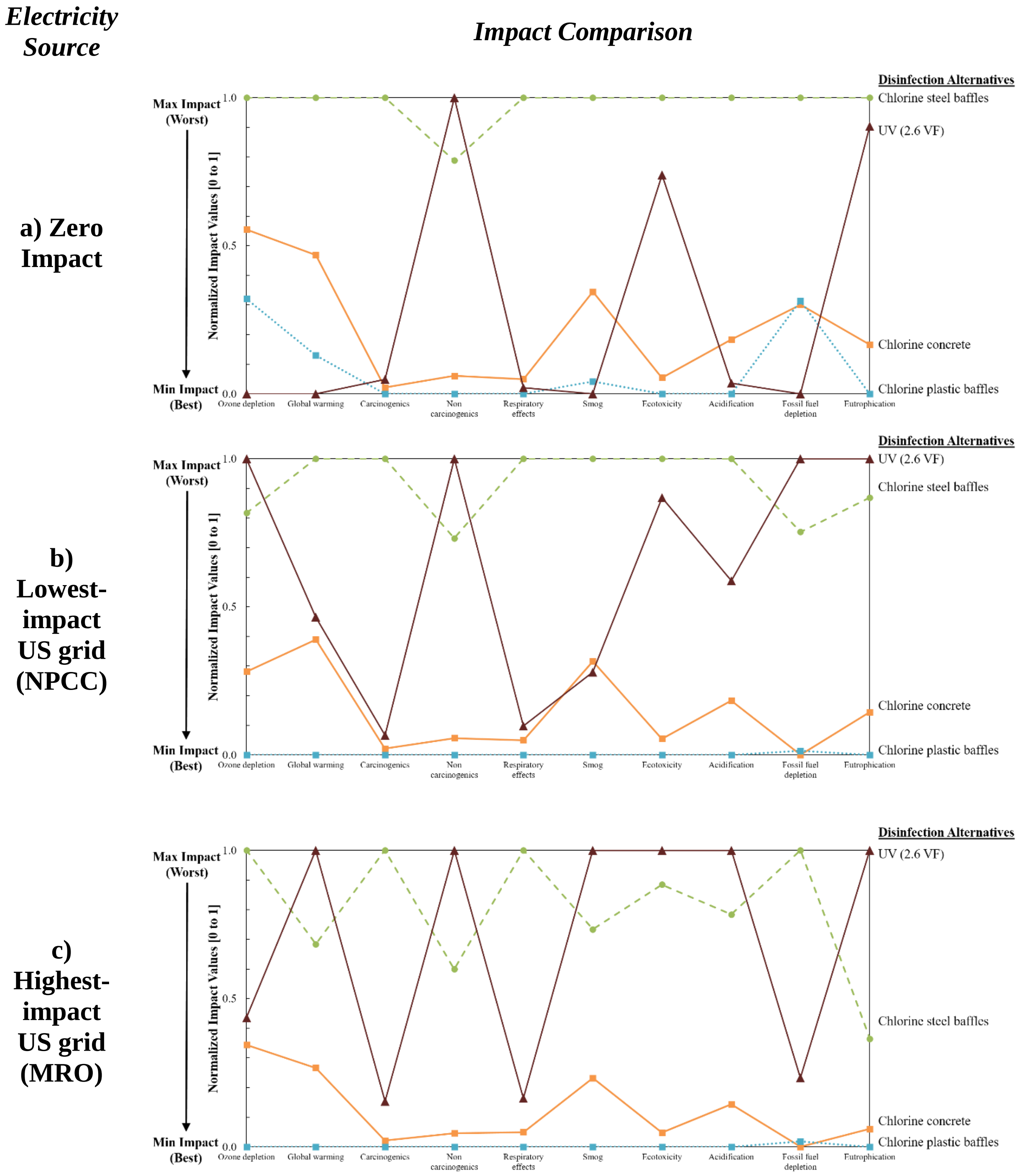
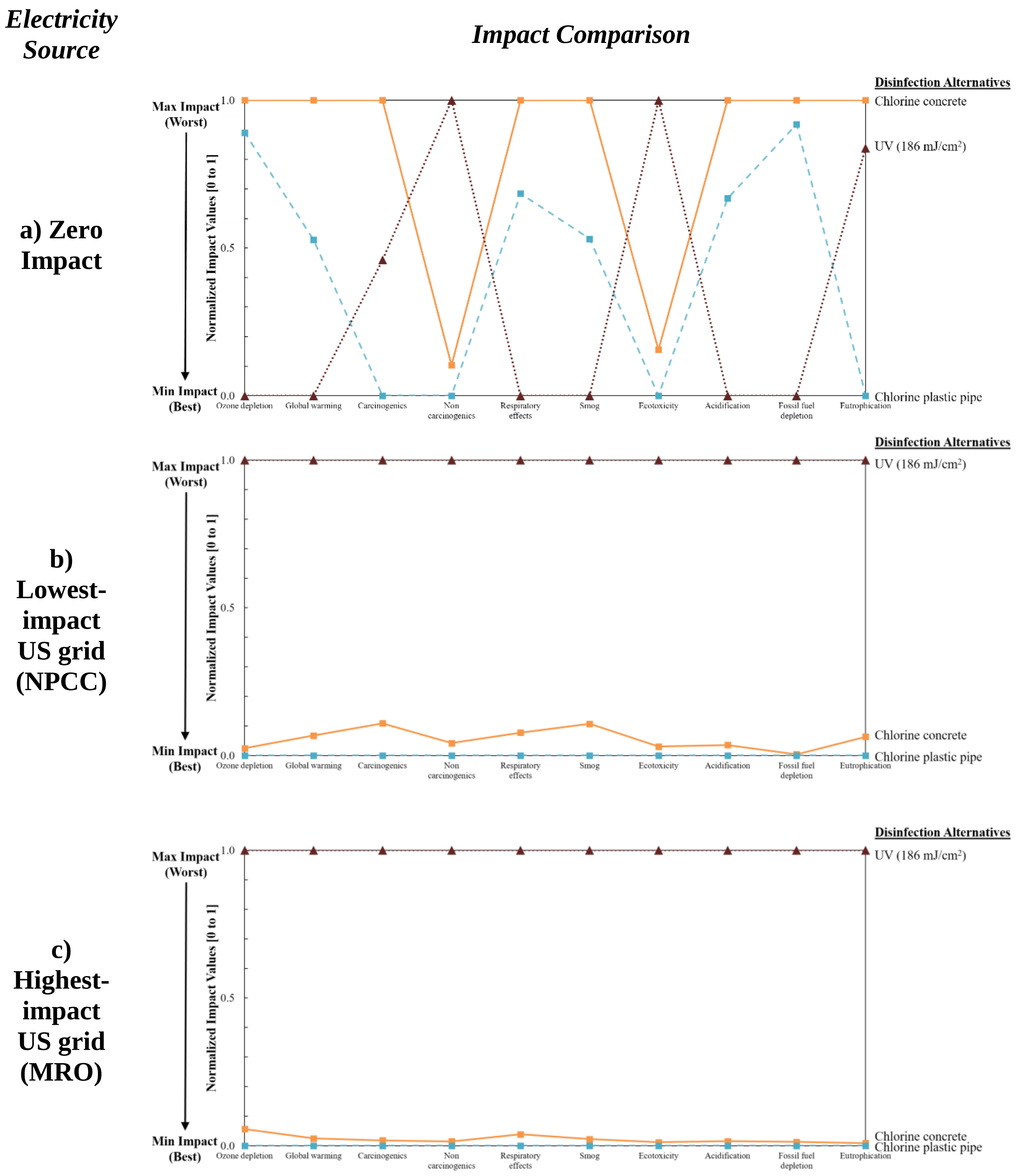


Figure B6. Typical source water scenario comparison of different electricity sources, (a) zero impact, (b) NPCC, and (c) MRO impacts on 3 chlorine alternatives with concrete basins (unbaffled, plastic baffles, and steel baffles) and for the UV average validation factor alternative. Data represents median uncertainty values. (VF is validation factor)

B6.3.2 Chlorine Residual Exemption Scenario

The chlorine residual exemption scenario had the same energy trends as scenario 1 (Figure S7). Except, the UV average validation factor alternative was better than the chlorine concrete alternative in 8 categories when negligible electricity emissions were assumed, instead of worse in all categories when assuming average electricity emissions.


Electricity Source

Impact Comparison

a) Zero Impact

b) Lowest-impact US grid (NPCC)

c) Highest-impact US grid (MRO)

Figure B7. Chlorine residual exemption scenario comparison of different electricity sources, (a) zero impact, (b) NPCC, and (c) MRO impacts on 2 chlorine alternatives (with concrete basin and plastic piping) and for 1 UV alternative (using virus inactivation dose and no chlorine addition). Data represents median uncertainty values.

B6.3.3 Filtration Exemption Scenario

For the third filtration exemption scenario, the energy source trends were opposite those in the other two scenarios (Figure S8). Assuming average U.S. electricity emissions, an average validation factor, a concrete contactor, and a 3.5 mWc pump pressure, UV disinfection was better than chlorine disinfection in 8 out of 10 TRACI categories. However, if negligible electricity-related emissions were assumed, then chlorine disinfection had lower impacts than UV disinfection in 6 out of 10 categories. This occurred because the chlorine alternative's impacts were very sensitive to electricity-related emissions. The chlorine alternative used significantly more energy (16 Wh/m^3 for filtration) than the UV alternative (5 Wh/m^3 for UV lamp and vessel pumping energy). Given this, chlorine disinfection had increasingly worse environmental performance as electricity-related impacts increased.

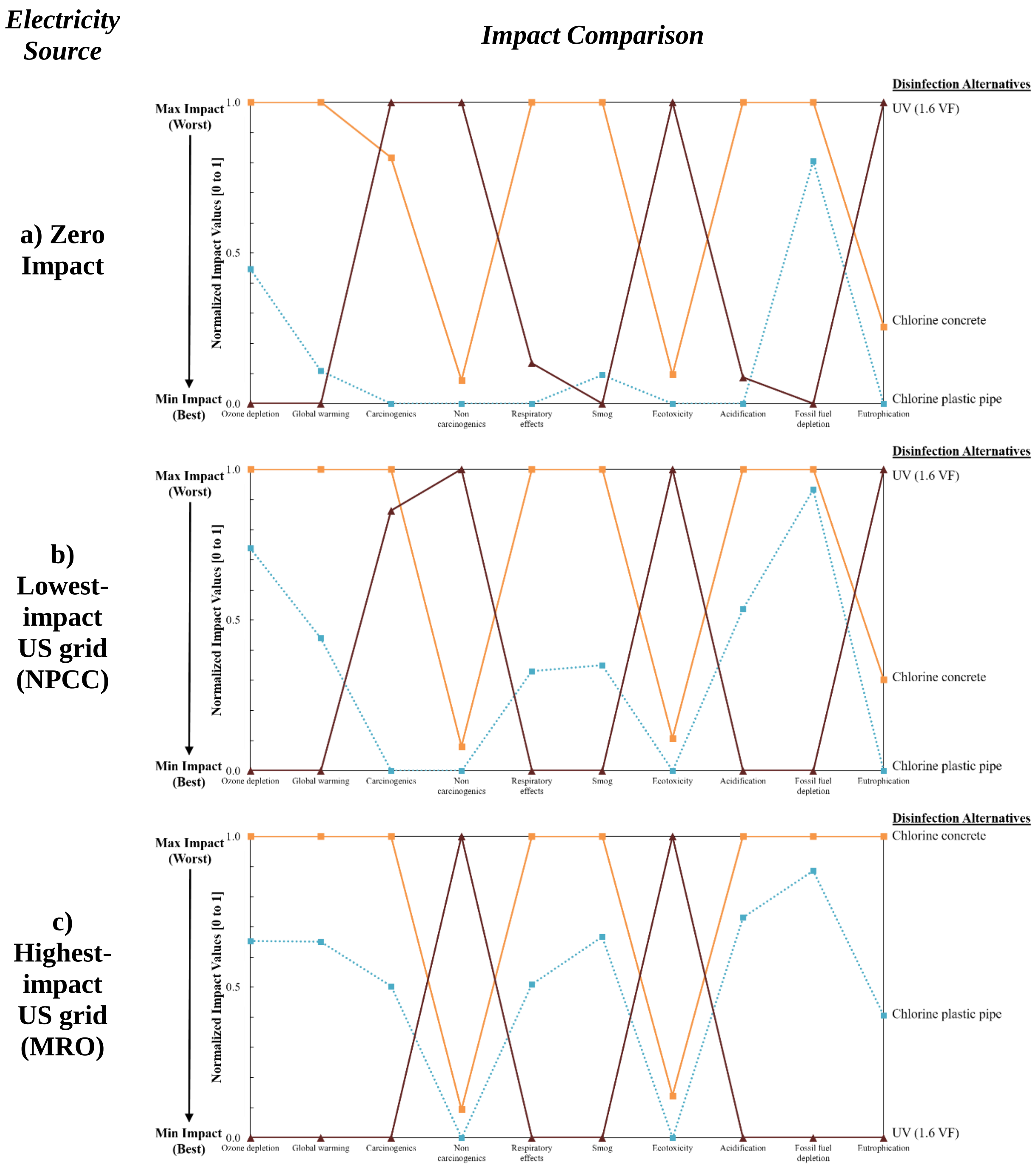


Figure B8. Filtration exemption scenario of different electricity sources, (a) zero impact, (b) NPCC, and (c) MRO impacts on 2 chlorine alternatives (chlorine concrete basin and chlorine plastic pipe) at the expected filter pressure (3.5 m) and on the UV lowest validation factor alternative. Data represents median uncertainty values. (VF is validation factor)

B6.4 Overdosing Scenario Impacts

B6.4.1 Typical Source Water Scenario

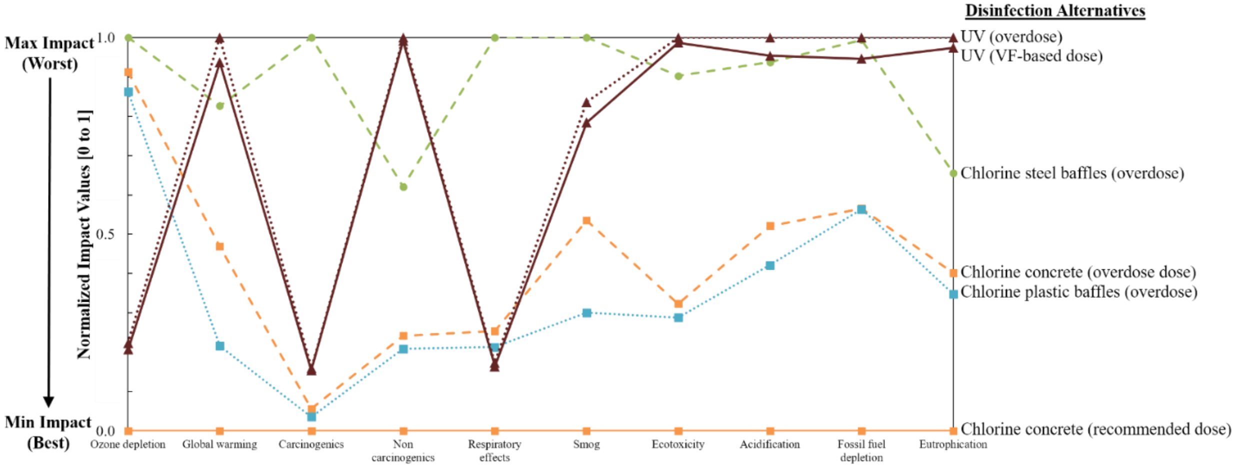


Figure B9. Overdosing of primary disinfectant for the chlorine in chlorine alternatives (double the chlorine dose from 1.5 mg/L to 3.0 mg/L) and for UV in the UV alternatives (minimum of 10 mJ/cm^2 [153]). Chlorine overdose alternatives have concrete basins (unbaffled, with plastic baffles, and with steel baffles). The graph includes the recommended doses for chlorine (1.5 mg/L, based on chlorine residual, decay, and demand calculations) and for UV (9.24 mJ/cm^2 times the highest validation factor of 4.4). (VF is validation factor)

B7. Filter Head Loss Alternatives

B7.1 Typical Source Water Scenario

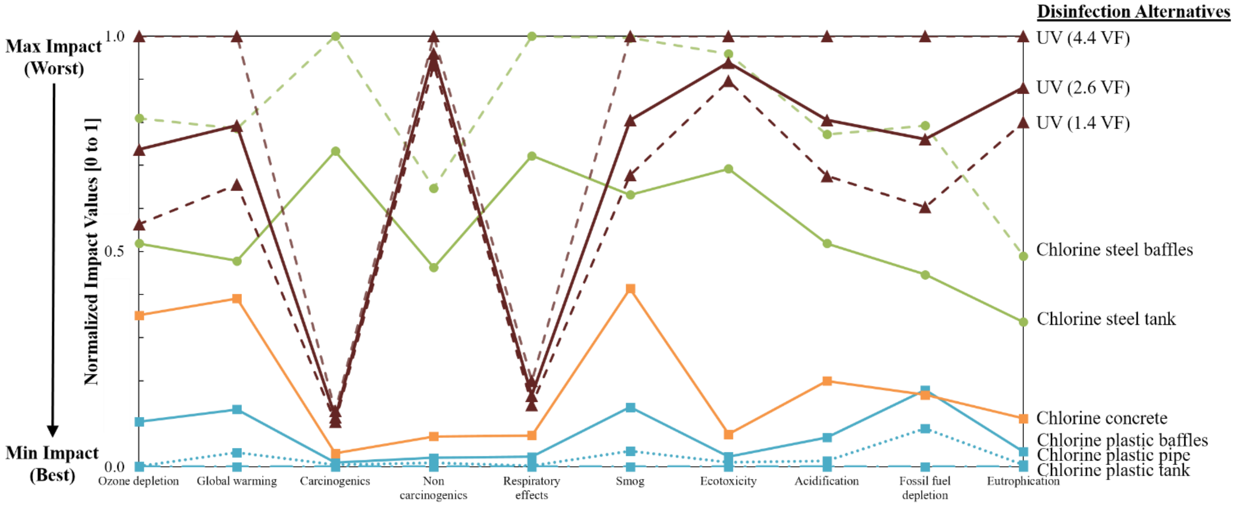


Figure B10. Same as Figure 3.3 in main text except the operational filter head is assumed to be 0.7 m (versus 3.5 m). Filter impacts alter each disinfection alternative equivalently, so there is no change in trends between this and Figure 3.3.

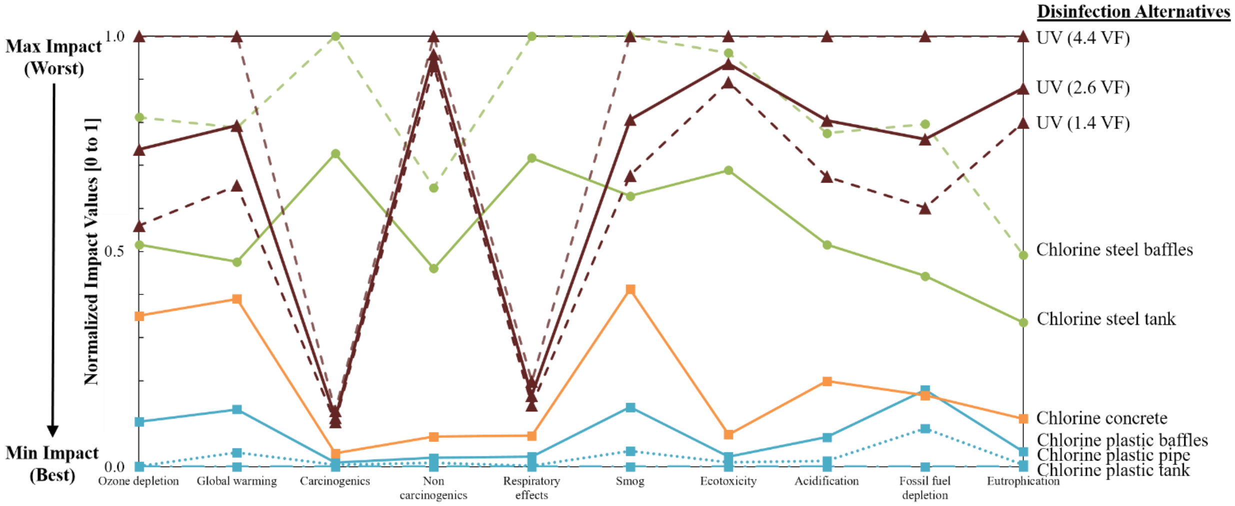


Figure B11. Same as Figure 3.3 in main text except the operational filter head is assumed to be 11 m (versus 3.5 m). Filter impacts alter each disinfection alternative equivalently, so there is no change in trends between this and Figure 3.3.

Appendix C: Supporting Information for: “A New Framework for Small Drinking Water Plant Sustainability Support and Decision-making”

The following manuscript and appendix are in preparation for submission for publication: Jones, C. H.; Meyer, J.; Seidel, C. J.; Cornejo, P. K.; Hogrewe, W.; Cook, S. M. A New Framework for Small Drinking Water Plant Sustainability Support and Decision-making.

C1. Decision-making Tool Inputs

Table C1. Source water quality parameters and valid model ranges included in SIPS.

Source Water Input	**Model Range**	**Default Value**
Design Flow Rate	7,200 – 500,000 gal/day [3]	72,000 gal/day
UVT	67 – 99% [189]	82% [189]
pH	6 – 9 [148]	7.5 [189]
Temperature	0.5 – 25°C [148]	15°C [189]
TOC	0.4 – 5 mg TOC/L [189]	2.8 mg TOC/L [189]
Alkalinity	13 – 210 mg as $CaCO_3$/L [189]	93 mg as $CaCO_3$/L [189]
Average Turbidity	<1 or >1 NTU (indicative of pre-treatment needs such as coagulation) [131]	<1 NTU
Bromide	0.01 – 0.16 mg/L [189]	0.069 mg/L [189]
Electricity Sector	U.S. Energy Grids & Grid Mixes (Table S9)	Weighted Average of U.S. Energy Grids

Notes: Each range is based on existing drinking water process models' ranges[3,148]; if none, then typical range from over 7,500 values from U.S. drinking water systems stated.[189] Default values are typical, representative SDWS values, which were based on empirical data[189] when possible.

Table C2. All SDWS treatment process options with their inventory items and a reference to the method used to quantify each item over the life cycle functional unit. Maintenance labor, although not listed, was included in every treatment process (Section A4).

Treatment Processes	Inventory Item	Inventory Quantification Method
Coagulation Options		
Alum Coagulation	Aluminum sulfate (kg)	[167]
	Chemical & solids disposal hauling (tkm)	[167]
Ferric Coagulation	$FeCl_3$ (kg)	[167]
	Chemical & solids disposal hauling (tkm)	[167]
Filtration Options		
Bag Filtration	Housing steel (kg)	Section A7.2
	Operational energy (kWh)	Section A7.2
	Polypropylene bag (kg)	Section A7.2
Cartridge Filtration	Housing steel (kg)	[91]
	Operational energy (kWh)	[91]
	Borosilicate microglass cartridge (kg)	[91]
Conventional Filtration	Housing concrete (m^3)	[167]
	Housing rebar (kg)	[167]
	Operational energy (kWh)	[167]
	Sand & anthracite (kg)	[167]
	Static mixer energy (kWh)	Section A7.5
Membrane Filtration	Housing steel (kg)	Section A7.3
	Operational energy (kWh)	Section A7.3
	Polyethylidine chloride membrane* (kg)	Section A7.3
Nonozonated Biofiltration	Housing concrete (m^3)	[167]
	Housing rebar (kg)	[167]
	Operational energy (kWh)	[167]
	Sand & anthracite (kg)	[167]
	Static mixer energy (kWh)	Section 7.5
Ozonated Biofiltration	Ozone generator energy (kg)	[167]
	Ozone generator steel (kg)	[167]
	Housing concrete (m^3)	[167]
	Housing rebar (kg)	[167]
	Operational energy (kWh)	[167]
	Sand & anthracite (kg)	[167]
	Static mixer energy (kWh)	Section 7.5
Slow Sand Filtration	Housing concrete (m^3)	Section A7.4
	Housing rebar (kg)	Section A7.4
	Operational energy (kWh)	Section A7.4
	Sand & gravel (kg)	Section A7.4
	Housing rebar (kg)	Section A7.4
	Operational energy (kWh)	Section A7.4
	Sand & gravel (kg)	Section A7.4
Disinfection Options		
Chlorine Disinfection	Chlorine contact zone options: plastic pipe (kg), concrete (m^3) with rebar (kg), concrete (m^3) with rebar (kg) & steel baffles (kg), or concrete (m^3) with rebar (kg) & plastic baffles (kg)	[91]
	Sodium hypochlorite (kg)	[91]
	Polyethylene chlorine storage tank (kg)	[91]
	Contact zone head loss (kWh)	[91]
	Chemical hauling (tkm)	[167]
UV Disinfection	Lamp energy (kWh)	[91]
	Lamp (Lamp Equivalents)	[91]
	Vessel head loss (kWh)	[91]
	Control electronics (kg)	[91]
	Vessel steel (kg)	[91]
	Quartz sleeve (kg)	[91]
	Wiper rubber (kg)	[91]
	Motor aluminum (kg)	[91]
pH Adjustment Options		
Caustic	Sodium hydroxide (kg)	[167]
	Chemical hauling (tkm)	[167]
Lime	Calcium hydroxide (kg)	[167]
	Chemical hauling (tkm)	[167]

Note: * polyethylidine chloride was substituted for polyethylidine fluoride, which was not available in LCI databases.

Table C3. Operational preference parameters and their default values.

Parameter	Selection Options	Default Value
Chlorine Dose	Optimized value (calculated) or user selects: 1, 1.5, 2, 2.5, 3, 3.5, or 4 mg/L as free Cl2 from NaOCl	Calculated optimized value
Coagulant Dose	Optimized value (calculated) or user inputs dose	Calculated optimized value
Ozone Feed	Air or Oxygen	Air
Filter CT Credit	No or Yes	No
Effluent pH Target	User input (must be equal to or greater than source water pH)	8.2

Table C4. Technology design parameters with range of possible values and default value.

#	Design Parameter	Low Value	High Value	Default Value	Basis
1	Head loss in UV chamber (m)	0.15	1.52	0.83	L/H[191], D=median from Monte Carlo trials[91]
2	UV electronics mass (kg)	4.1	15	9.25	L=sum(sensor, ballast, min UVT analyzer), [91] H=sum(sensor, ballast, max UVT analyzer),[91] D=median from Monte Carlo trials[91]
3	UV electronics lifetime (yrs)	3	5	4	L/H[191], D=median from Monte Carlo trials[91]
4	UV quartz sleeve lifetime (yrs)	8	10	9	L/H[191], D=median from Monte Carlo trials[91]
5	UV quartz sleeve/wiper thickness (cm)	0.10	0.20	0.15	L/H[191], D=median from Monte Carlo trials[91]
6	UV wiper lifetime (yrs)	3	5	4	L/H[191], D=median from Monte Carlo trials[91]
7	UV vessel mass (kg)	26.4	39.6	33.0	Existing system ± 20%, D=median from Monte Carlo trials[91]
8	UV lamp power at 40 mJ/cm^2 (kWh/m^3)	0.0079	0.0249	0.01789	Range of existing systems[91], D=median from Monte Carlo trials[91]
9	UV lamp length (m)	1.2	1.5	1.4	Range of existing systems[91], D=median from Monte Carlo trials[91]
10	UV lamp weight (kg)	0.5	1.2	0.84	Range of existing systems[91], D=median from Monte Carlo trials[91]
11	UV lamp lifetime (hrs)	8000	15000	11497	L/H[191], D=median from Monte Carlo trials[91]
12	UV quartz sleeve outer diameter (cm)	2.5	5	3.74	L/H[191], D=median from Monte Carlo trials[91]
13	Chlorine storage tank lifetime (yrs)	30	35	33	L/H[79], D=median from Monte Carlo trials[91]
14	Chlorine delivery rate (hauling trips/week)	0.5	2	1.2	L=Every other week, H=Twice per week, D=median from Monte Carlo trials[91]
15	Chlorine pump head (m)	1.22	70.3	35.8	L/H[149], D=median from Monte Carlo trials[91]
16	Contact zone concrete lifetime (yr)	30	60	45	L/H[79], D=median from Monte Carlo trials[91]
17	Contact zone concrete wall thickness (m)	0.23	0.46	0.34	L/H[201], D=median from Monte Carlo trials[91]
18	Contact zone concrete base and filter base thickness (m)	0.30	0.61	0.46	L/H[201], D=median from Monte Carlo trials[91]
19	Contact zone unbaffled tank baffling factor	0.1	0.3	0.2	L/H[97], D=median from Monte Carlo trials[91]
20	Contact zone two baffle tank baffling factor	0.3	0.5	0.4	L/H[97], D=median from Monte Carlo trials[91]
21	Contact zone baffle thickness (cm)	3.8	4.5	4.2	L=[202], H=Derived geometrically (from Figure 1 in [203]), D=median from Monte Carlo trials[91]
22	Contact zone PVC lifetime(yrs)	35	40	38	L/H[79], D=median from Monte Carlo trials[91]
23	Contact zone PVC thickness (cm)	0.60	1.7	1.2	L/H[150], D=median from Monte Carlo trials[91]
24	Contact zone PVC piping baffling factor	0.6	1	0.8	L/H[97], D=median from Monte Carlo trials[91]
25*	Minimum allowable coagulant dose (mg alum/L)	6.0	17	10	L=25th percentile value[95], H=75th percentile value[95], D=median[95]
26*	Source water BDOC/TOC ratio	14%	27%	20%	L=min[85], H=max[85], D=median[85]
27*	Ozonated water BDOC/TOC ratio	20%	38%	30%	L=min[85], H=max[85], D=median[48]
28*	Biodegradable fraction of TOC removed by coagulation, when source water SUVA<3 L/mg/m	2.0%	5.0%	4.0%	L=25th percentile[52,72], H=75th percentile[52,72], D=average[52,72]
29*	Biodegradable fraction of TOC removed by coagulation, when source water SUVA>3 L/mg/m	7.5%	14%	9.0%	L=25th percentile[52,72], H=75th percentile[52,72], D=average[52,72]
30*	Nonozonated biofilter percent TOC removal (of the available biodegradable fraction of TOC) for 10°C to 20°C	5.0%	22%	10%	L=min[85], H=max[85], D=median[85]
31*	Ozonated biofilter TOC removal (of the available biodegradable fraction of TOC) for 10°C to 20°C	3.0%	47%	13%	L=min[85], H=max[85], D=median[85]

32*	Pre-ozonation dose (g O_3/g TOC)	0.25	1.6	0.50	L=min[85], H=max[85], D=median[85]
33	Air-fed ozone specific energy use (kWh/g O_3 generated)	0.018	0.022	0.020	L[101], H[102,103], D=average of L&H
34	Oxygen-fed ozone specific energy use (kWh/g O_3 generated)	0.008	0.013	0.010	L[106],H[102,216], D=average of L&H
35	RMF/BF backwash flowrate ($m^3/h/m^2$)	30	60	50	L/H[98], D=expert judgment
36	RMF/BF backwash pressure (m)	8	10	9	L/H[98], D=average
37	RMF/BF water height above media (m)	1.5	2.5	2	L/H[96], D=average
38	RMF/BF media lifetime (yr)	15	25	20	L/H/T[167]
39	RMF/BF hydraulic loading rate (m/h)	10	25	15	L/H[96], D=expert judgment
40	RMF/BF anthracite depth (m)	0.405	0.5	0.45	D= 0.45[96], L/H = ±10% of D
41	RMF/BF sand depth (m)	0.27	0.33	0.3	D= 0.3[96], L/H = ±10% of D
42	Landfill hauling distance (km)	20	100	20	L[204], H=expert judgement, D=most reliable estimate[204]
43	Chemical hauling distance (km)	20	100	20	L[204], H=expert judgement, D=most reliable estimate[204]
44	Ozone generator mass (kg/(g/hr))	0.8	18.7	9.8	L[105], H[104], D=average
45*	Cartridge filter average operational pressure head (m)	0.7	21.1	3.5	L/H[137], D[147]
46	Cartridge filter replacement (times/yr)	0.5	4	2.25	L=replace at least every 2 years, H=replace at most four times a year, D=median from Monte Carlo trials[91]
47	Slow sand filter HLR (m/h)	0.12	0.42	0.2	L/H[96], D=expert judgment
48	Slow sand filter sand depth (m)	1.0	1.4	1.3	L[217], H[218], D[217]
49	Slow Sand filter water height above media (m)	1.0	1.5	1.25	L/H[218], D=average
50	Freeboard (m)	0.2	0.3	0.25	L/H[218], D=average
51	Slow sand filter gravel support depth (m)	0.3	0.5	0.4	L/H[218], D=average
52	Slow sand filter percent TOC removal (of the available biodegradable fraction of TOC)	5.0%	25%	15%	L/H[175], D=average
53*	Bag filter average operational pressure head (m)	0.7	21.1	3.5	L/H[137], D[147]
54	Bag filter replacement (times/yr)	0.5	4	2.25	L=replace at least every 2 years, H= replace at most four times a year, D=median from Monte Carlo trials for cartridge filtration[91]
55	Bag filter housing mass (kg/housing)	31.8	45.4	38.6	L/H[219], D=Average
56*	Membrane filter average operational pressure head (m)	2.8	10.5	10.5	L/H[31,106,220,221], D=Most common from[31,106,220,221]
57	Membrane specific skid weight ($kg/(Mm^3/yr)$)	11000	42000	27000	L[222], H[223], D=Average
58	Membrane-filter replacement (times/yr)	0.5	4	2.25	L=replace at least every 2 years, H=replace at most four times a year, D=median from Monte Carlo trials for cartridge filtration[91]
59	Head loss across static mixer (m)	0.17	0.66	0.41	L/H[224], D=average

Note: * refers to parameters that used a triangular distribution instead of a uniform distribution during Monte Carlo simulations.

Table C5. All decision criteria initially considered, with justification if ultimately not included.

Potential Decision Criteria	Included	Notes if not Included
Capital Cost	X	
Global Acidification Impact		Midpoint LCA indicator that was accounted for in the ecosystem quality endpoint LCA indicator
Global Climate Change	X	
Global Ecosystem Destruction	X	
Global Ecotoxicity		Midpoint LCA indicator that was accounted for in the ecosystem quality endpoint LCA indicator
Global Eutrophication Impact		Midpoint LCA indicator that was accounted for in the ecosystem quality endpoint LCA indicator
Global Human Health	X	
Global Ozone Depletion		Midpoint LCA indicator that was accounted for in the human health and ecosystem quality endpoint LCA indicators
Global Resource Depletion	X	
Global Smog Formation		Midpoint LCA indicator that was accounted for in the human health and ecosystem quality endpoint LCA indicators
High Level of Automation		Accounted for in *maintenance requirements*
High Operational Flexibility	X	Expanded to be *resilience to regulation and source water*
High Water Conservation		Not included because considered low importance by SDWS stakeholders and unclear scoring (conservation could be considered negative or positive; e.g., the SDWS may require use for financial support or may be having difficulty securing a source water)
Independence from Outside Help	X	
Intrusiveness of Drinking Water System	X	
Local long-term Human health	X	
Low Complexity of Approval		Accounted for in *resilience to regulation and source water*, *operator training requirement*, and *maintenance requirements*
Low Frequency of Lab Testing		Expected to be the same across alternatives for finished water and added lab burdens due to treatment process complexity is accounted for in *maintenance requirements*
Low Waste Stream Handling		Accounted for in the affordability and global pollution decision criteria
Maintenance Requirements	X	
Operation and Maintenance Cost	X	
Operator Training Requirement	X	
Short Travel Distance for Spare Parts		Accounted for in *independence from outside help*
Small Land Footprint		Accounted for in *maintenance requirements*

Table C6. References that demonstrate each criterion's importance or applicability to SWDSs.

Decision Criteria	References
Capital Cost	[3,8,176,177,225]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Operation & Maintenance Cost	[3,8,176,177,225]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Global Climate Change	[17,18,34,91,167,178]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Global Ecosystem Destruction	[17,18,34,91,167,178]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Global Resources Depletion	[17,18,34,91,167,178]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Global Human Health	[17,18,34,91,167,178]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Local Long-term Health Risk	[168,179,226]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Resilience to Regulation and Source Water	[182,227,228]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Operator Training Requirement	[3,183,229]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Maintenance Requirements	[3,177,229]; RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
System Intrusiveness	RCAP technical providers; DERISK technical advisors; small system conference workshop attendees
Independence from Outside Help	RCAP technical providers; DERISK technical advisors; small system conference workshop attendees

Note: RCAP is the Rural Community Assistance Partnership; DERISK is Design of Risk-reducing, Innovative-implementable Small-system Knowledge.

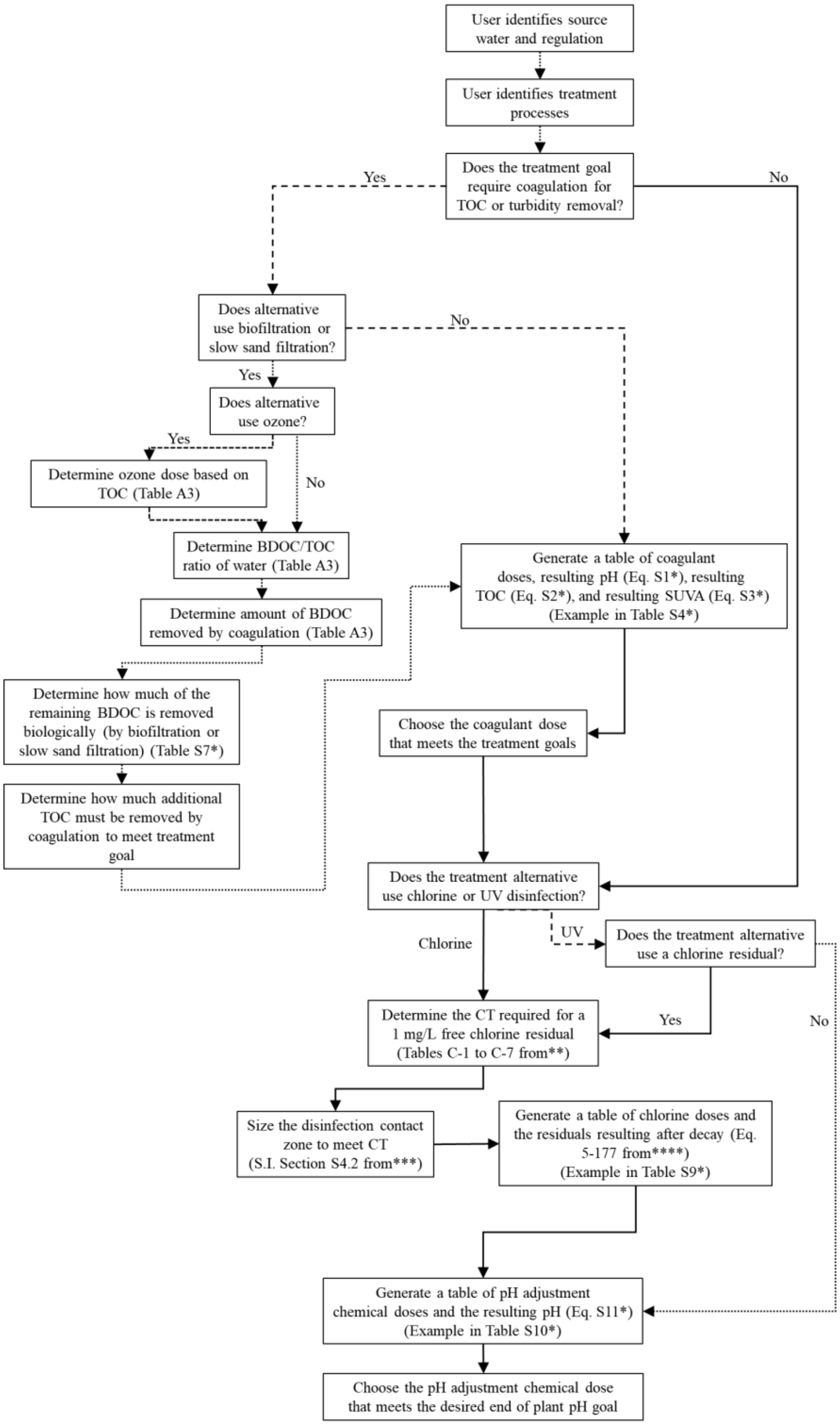


Figure C1. Summary of SIPS's water treatment process modeling calculation steps. The pH adjustment dose for UV followed by a chlorine residual uses the same dose required as chlorine-only disinfection because the pH after either disinfection is insignificantly different (i.e., the difference between chlorine doses is less than ~0.3 mg/L because hypochlorite is a weak acid).

Notes: * refers to a previous model of conventional filtration and biofiltration,[167] ** refers to U.S. EPA CT requirements,[90] *** refers to a previous disinfection LCA,[91] and **** refers to the water treatment plant model.[92]

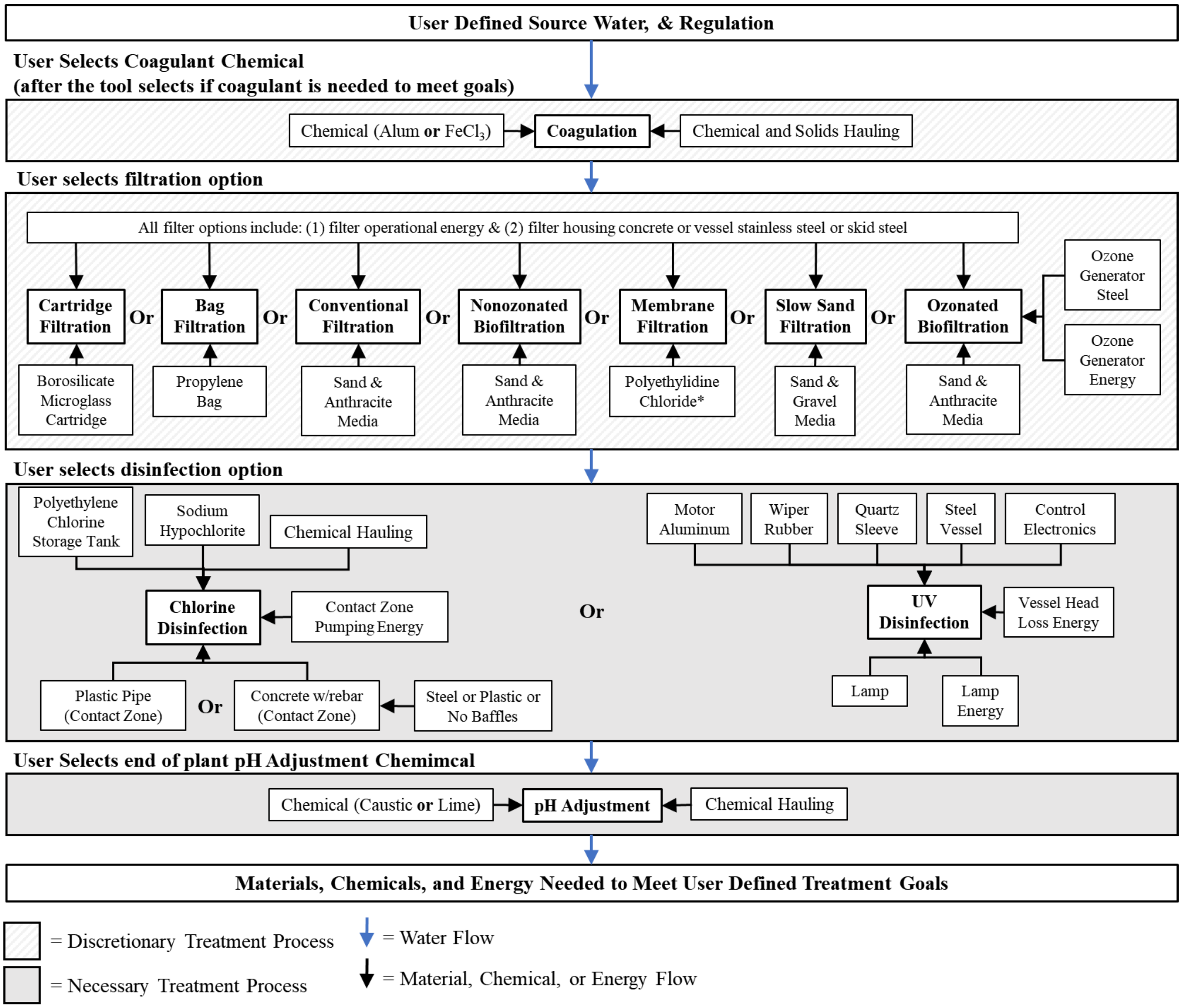


Figure C2. The LCA system boundary of all treatment process options.

Note: * polyethylidine chloride was substituted for polyethylidine fluoride, which was not available in LCI databases.

C2. MCDA Method Consideration

The overall SIPS framework was structured to reflect the four LCA phases (goal and scope definition, inventory analysis, impact assessment, and interpretation) and ensure comprehensive and accurate comparisons.[17,18,34,91,167] Decision criteria were developed collaboratively to effectively represent SDWS needs and values. Existing criteria scoring methods, such as the life cycle impact assessment method IMPACT 2002+[178] and the drinking water Relative Health Index (RHI) model,[168,179] were used to score (i.e., quantify) each SDWS decision criteria; new criteria evaluation methods were developed for novel SDWS criteria.

Since each criteria evaluation method had unique units and resulted in score values of varying magnitudes between criteria, commonly-used MCDA methods were leveraged to provide a comprehensive and transparent decision-making framework. ELECTRE[173] and PROMETHEE[230] MCDA methods use outranking and pair-wise comparison approaches, respectively, to recommend an alternative. ELECTRE is most effective when used to evaluate a large number of alternatives[186,231] while SDWSs have limited options. PROMETHEE requires complex data processing[232] that can conflict with the SDWS tool goals of simplicity and accessibility. To accomplish a similar rigorous approach that meets SDWS specific requirements, SIPS uses a normalization approach similar to the TOPSIS[233] method to synthesize criteria scores into a final recommendation (Eq. C1). Then, for simplicity and transparency, the weighted sum method,[184] the simplest data processing method with high clarity of results presentation,[234] was used to calculate a single-score for each alternative (Eq. C2). MCDA methods usually take stakeholder preferences into account for criteria weighting,[173,230,233] so SIPS was constructed to have the user follow the Analytical Hierarchy Process (AHP)[187] to develop a weighting scheme (i.e., quantify their priorities in a consistent manner). AHP is a common and effective method for incorporating stakeholder preferences in multiple-criteria decision problems[196,235,236] and has been used to make drinking water expert opinions and decision-making criteria scores more resilient to uncertainty.[174,237]

$$X_{i,j} = \frac{R_{i,j} - R_{best,j}}{R_{worst,j} - R_{best,j}} \qquad \textbf{Eq. C1}$$

Where: i = refers to the ith treatment train alternative, j = refers to the jth criterion, $X_{i,j}$ = normalized criterion score (between 0 and 1), $R_{i,j}$ = raw criterion score, $R_{best,j}$ = best raw criterion score, $R_{worst,j}$ = worst raw criterion score

$$\text{Performance Score}_i = \sum_{j=1}^{n} X_{i,j} * W_j \qquad \textbf{Eq. C2}$$

Where: i = refers to the ith treatment train alternative, j = refers to the jth criterion, n = number of decision criteria (12 for SIPS), $X_{i,j}$ = normalized criterion score, and W_j = criterion's weight

C3. Affordability Calculations

Capital costs were determined based on the initial construction of a treatment train (Eq. C3). The operation and maintenance costs were based on well-established cost equations[176] (Eq. C4). These equations were populated with material quantity (from the inventory of materials, chemicals, and energy) and cost information (Table C7). Discount rates were determined based on the literature[238] (Figure C3a), labor escalation rates were based on U.S. Bureau of Labor Statistics data for water operator wages[239] (Figure C3b), and projected average fuel price index was based on National Institute of Standards and Technology (NIST) recommendation[240] (Figure C3c). Escalation rates for materials, chemicals, and hauling were assumed to be zero over the 40 year study life.[176]

$$\mathbf{CC = Materials \times Cost\ Factor}$$ **Eq. C3**

Where:

CC = Capitol cost; one-time, initial costs
Materials = Quantity of materials per replacement life (from material, chemical, energy inventory)
Cost Factor = Unit cost of materials per unit quantity (Table C7)

$$\mathbf{O\&M = R + E + Chems + OM\&R - S}$$ **Eq. C4**

Where:

$$\mathbf{R = \frac{T_v\,(1+e)^t}{(1+d)^t}}$$ **Eq. C4a**

$$\mathbf{E = A \times \sum_0^n \left(\frac{I_{2018+y}}{(1+d_i)^y}\right)}$$ **Eq. C4b**

$$\mathbf{Chems = Chems_{t_1,d_1} + \left(Chems_{t_2,d_2} - Chems_{t_1,d_2}\right) + \left(Chems_{t_3,d_3} - Chems_{t_2,d_3}\right)}$$ **Eq. C4c**

$$\mathbf{Chems_{t_i,d_i} = \frac{A(1+d_i)^{t_i} - 1}{d(1+d_i)^{t_i}}}$$ **Eq. C4c-ii**

$$\mathbf{OM\&R = OM\&R_{t_1,d_1} + \left(OM\&R_{t_2,d_2} - OM\&R_{t_1,d_2}\right) + \left(OM\&R_{t_3,d_3} - OM\&R_{t_2,d_3}\right)}$$ **Eq. C4d**

$$\mathbf{OM\&R_{t_i,d_i} = A\frac{1+e}{d_i - e}\left[1 - \left(\frac{1+e}{1+d_i}\right)^{t_i}\right]}$$ **Eq. C4d-ii**

$$\mathbf{S = \frac{\left[T_v - \left(\frac{T_v}{t} \times n\right)\right](1+e)^n}{(1+d)^n}}$$ **Eq. C4e**

Where:

O&M = Operation and maintenance cost over the study time frame
R = Replacement costs
E = Energy costs
Chems = Chemical costs
OM&R = Operations, maintenance, and repair costs
S = Resale, scrap, salvage without disposal revenue
T_v = Value of a present non-fuel replacement cost in base year dollars; this is the product of material quantities and unit costs(Table C7)
d_i = Discount rate for the corresponding time interval (Figure C3a)
t_i = Defined time interval between discrete discount rate changes (Figure C3a)
t = Unit useful life (Table C4)

e = Escalation rate; zero for materials, chemicals, and hauling, and 2.41% for operator labor[239] (Figure C3b)
n = Study timeframe (40 years)
A = Constant yearly annuity value in base year dollars, which is the product of chemical, or energy inventory quantities and cost factors (Table C7)
I_{2018+t} = Projected average fuel price index[240] (Figure C3c)
y = time interval (1 year)
$Chems_{ti,di}$ = Cost of chemicals such as chlorine or alum for a time interval using a specified discount rate up to the end of study timeframe $OM\&R_{ti,di}$ = Cost of operations and maintenance for a time interval using a specified discount and escalation rate up to the end of study life

Table C7. Unit costs for materials, energy, chemicals, and labor.

Cost Parameter	Cost Factor (reference)
Materials	
Concrete	Average of range \$0.34/kg (41) - \$0.35/kg (241)
PVC Piping	Average of range \$12.66/kg (181) - \$15.28/kg (241)
304 Stainless Steel	Average of range \$2.56/kg (242) - \$4.08/kg (243)
Reinforcing Steel	Average of range \$0.64/kg (242) - \$0.80/kg (244)
Filter Anthracite Coal	Average of range \$2.64/kg (245) - \$3.70/kg (246)
Filter Sand	\$1.39/kg (246)
Filter Gravel	Average of range \$4.27/kg (246) - \$4.82/kg (245)
Polyethylene Baffle	Cost(\$/baffle) = 0.1478*(Contactor Design Volume (gal))+113.36 (Trend based on data from: 247)
LPUV Lamp	Average of range \$450/lamp (33) – 850/lamp (241)
Quartz Lamp Sleeve	Average of range \$70/sleeve (33) – 75/sleeve (241)
Wiper ring	\$18/ring (33)
UV UVT Monitor	\$7500/monitor (33)
UV Ballast	\$155/ballast (241)
UV Sensor	\$500/sensor (33)
UV Wiper Motor	\$149/motor kit (248)
Ozone Generator	\$274200 (41)
MF/UF Skid	Cost (\$/skid) = 0.9508*(flow (gpd))+51746 (Trend based on data from: 249,250)
MF/UF Membrane	\$7.3/ft^2 membrane area (241)
Cartridge Filter Housing	Cost (\$/housing) = 0.0023*(flow (gpd))+120.54 (Trend based on data from: 251)
Cartridge Filter	Cost (\$/filter) = 0.0005*(flow (gpd))+21.96 (Trend based on data from: 251)
Bag Filter Housing	\$342.17/housing (252)
Bag Filter	\$7.19/filter (252)
Non-Hazardous Waste Transportation	\$0.5/ton/mile (181)
Chemicals	
Alum ($AlSO_4$)	\$0.33/kg (49)
$FeCl_3$	\$0.44/kg (49)
Caustic	\$0.77/kg (49)
Quicklime	\$0.11/kg (49)
Chlorine from 12% NaOCl	\$10.1/kg (49)
Electricity by sector	
ASCC	\$0.189/kWh (253)
FRCC	\$0.137/kWh (253)
NPCC	\$0.159/kWh (253)
MRO	\$0.090/kWh (253)
RFC	\$0.124/kWh (253)
SERC	\$0.098/kWh (253)
SPP	\$0.080/kWh (253)
TRE	\$0.085/kWh (253)
WECC	\$0.106/kWh (253)
HICC	\$0.254/kWh (253)
Average	\$0.132/kWh (253)
Weighted Average	\$0.108/kWh (253)
Labor	
Operator Hourly Wage	\$22.19/hr (239)

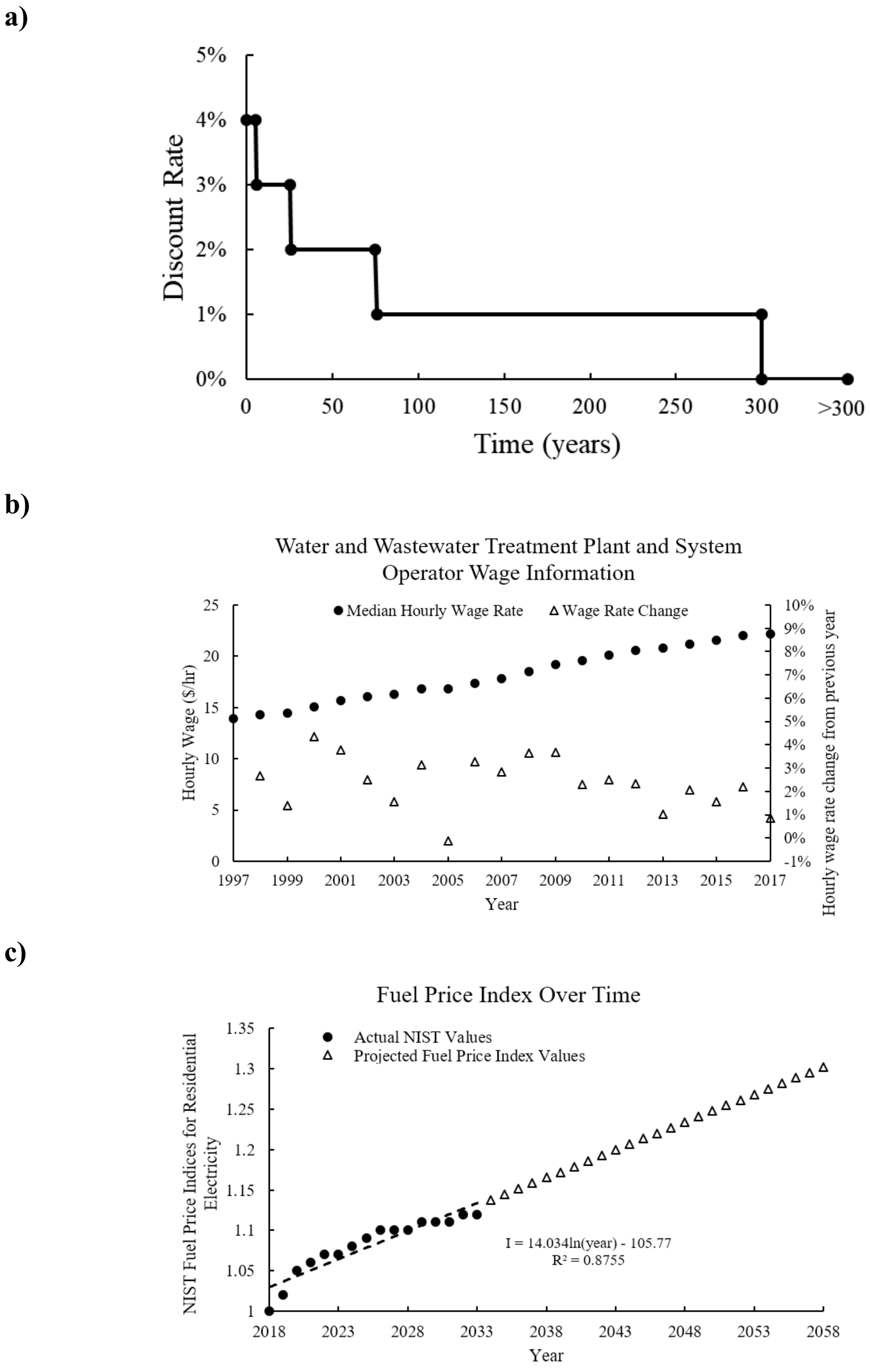


Figure C3. a) Discount rate based on time interval.[238] b) Change in water system operator wage over time; median yearly change was 2.41% (mean was 2.38% and a standard deviation was 1.12%).[239] c) Fuel price index values over 40 year study life using existing NIST values until 2033[240] and then projected values until 2058.

C4. Life Cycle Inventory Data

Table C8. LCI unit process names and descriptions. Unit processes were from the ecoinvent v3 database[81] except for anthracite coal, which was from the US-EI 2.2 database.[82] The percent contribution of US electricity produced by each grid is stated;[197] n/a is not available.

Description	LCI Unit Process Name	Application
Alum	Aluminium sulfate, powder {RoW}| production | Alloc Def, U	Coagulation
Aluminum	Aluminium, wrought alloy {GLO}| aluminium ingot, primary, to market	UV wiper motor
Anthracite	Anthracite coal, at mine NREL/RNA U	Filter media
Caustic Soda	Sodium hydroxide, without water, in 50% solution state {RoW}| chlor-alkali electrolysis, membrane cell | Alloc Def, U	pH adjustment
Chlorine	Sodium hypochlorite, without water, in 15% solution state {RoW}| sodium hypochlorite production, product in 15% solution state | Alloc Def, U	Disinfection (free chlorine from NaOCl and was adjusted as such)
Concrete	Concrete, 20MPa {RoW}| concrete production 20MPa, RNA only | Alloc Def, U	Chlorine contact basin
Electricity	Electricity, medium voltage {ASCC}| market for | Alloc Def, U	ASCC grid (n/a)
	Electricity, medium voltage {FRCC}| market for | Alloc Def, U	FRCC grid (6% US electricity)
	Electricity, medium voltage {NPCC, US only}| market for | Alloc Def, U	NPCC grid (7% US electricity)
	Electricity, medium voltage {MRO, US only}| market for | Alloc Def, U	MRO grid (17% US electricity)
	Electricity, medium voltage {RFC}| market for | Alloc Def, U	RFC grid (20% US electricity)
	Electricity, medium voltage {SERC}| market for | Alloc Def, U	SERC grid (17% US electricity)
	Electricity, medium voltage {SPP}| market for | Alloc Def, U	SPP grid (6% US electricity)
	Electricity, medium voltage {TRE}| market for | Alloc Def, U	TRE grid (9% US electricity)
	Electricity, medium voltage {WECC, US only}| market for | Alloc Def, U	WECC grid (18% US electricity)
	Electricity, medium voltage {HICC}| market for | Alloc Def, U	HICC grid (n/a)
Electronics	Electronics, for control units {RoW}| production	UV electronics (wires and housing)
Ferric Chloride	Iron (III) chloride, without water, in 40% solution state {RoW}| iron (III) chloride production, product in 40% solution state | Alloc Def, U	Coagulation
Glass	Glass tube, borosilicate {RoW}| production	UV quartz sleeve and cartridge filter (portion of total composition)
Hard Plastic	Polypropylene, granulate {RoW}| production	Plastic baffles (chlorine contact basin)
Hauling	Transport freight, lorry 3.5-7.5 metric ton, EURO6 {RoW}| transport, freight, lorry 3.5-7.5 metric ton, EURO6 | Alloc Def, U	Solids and chemical hauling
Lamp	Ultraviolet lamp {GLO}| ultraviolet lamp production, for water disinfection	UV lamp (Note: lamp equivalent units were converted to mass by summing the individual component masses in this unit process.)
PVC	Polyvinylchloride, suspension polymerised {RoW}| polyvinylchloride production, suspension polymerisation |	PVC pipe (chlorine contact basin)
Lime	Lime, hydrated, loose weight {RoW}| production | Alloc Def, U	pH adjustment
Reinforcing Steel	Reinforcing steel {RoW}| market for | Alloc Def, U	Chlorine contact basin
Rubber	Synthetic rubber {RoW}| production	UV sleeve wiper
Sand	Sand {GLO}| market for | Alloc Def, U	Sand (filter media)
Soft Plastic	Polyethylene, high density, granulate {RoW}| production | Alloc Def, U	Plastic cylindrical tank (contact basin)
Stainless Steel	Steel, chromium steel 18/8, hot rolled {RoW}| production | Alloc Def, U	Ozone generator, UV vessel, steel baffles (cl contactor), tank (cl contactor), filter housing
Tap Water	Tap water {RoW}| tap water production, conventional treatment | Alloc Def, U	Dilution water (chlorine solution)

C5. Maintenance Hours Calculations

Maintenance hour calculations were based on EPA work breakdown structure model relationships.[181] First, the number of instruments, number of vessels, plant area, and number of chemicals were determined. Second, the number of pumps and valves were calculated using Eq. C5, based on the number of vessels and chemicals, and Eq. C6 from the work breakdown structure model that has been released for GAC.[177] Next, time factors (Exhibit E-1[181]) were then used with the number and types of components associated with each treatment option (Table C9) to get the hourly requirement for each maintenance task per year (Eq. C7). Then hourly requirements per task were summed.

$$\mathbf{\#\ Pumps = \#Vessels + \#Chemicals} \qquad \textbf{Eq. C5}$$

$$\mathbf{\#\ Valves = 2 + (1 * \#Vessels) + (2 * \#Pumps)} \qquad \textbf{Eq. C6}$$

$$\mathbf{T_m = TF * C_\#} \qquad \textbf{Eq. C7}$$

Where:

T_m = Hours for a maintenance task (hours/year)
TF = Time factor year/(# or ft^2) (Exhibit E-1[181])
$C_\#$ = Number of components (Table C9)

Table C9. Summary of maintenance requirement inputs and maintenance hours output.[181] Input values (e.g.,, number of instruments) are adjustable in SIPS.

Treatment Process	**Instruments (#)**	**Vessels (#)**	**Area (ft^2)**	**Chemicals (#)**	**Maintenance Hours (hr/yr)**
Alum Coagulation (Includes pH adjustment)	2	2	100	2	309
Ferric Coagulation (Includes pH adjustment)	2	2	100	2	309
Bag Filtration	1	1	100	0	126
Cartridge Filtration	1	1	100	0	126
Conventional Filtration	2	3	100	2	357
Membrane Filtration	1	1	100	0	126
Nonozonated Biofiltration	2	3	100	2	357
Ozonated Biofiltration	3	4	100	3	489
Slow Sand Filtration	1	1	100	0	126
Chlorine Disinfection	1	1	100	1	177
UV Disinfection	2	1	100	1	209
Caustic	1	1	100	1	177
Lime	1	1	100	1	177

C6. Criterion Ranking Tables and Default Score Values

Table C10. Ranking table to define the *resilience to regulation and source water* criterion scores.

Score	Score Description
1	Treatment process can handle spikes in contaminants with a small change in operation (i.e., chemical dose change) and can remove contaminants of concern that are less common in source waters.
2	Treatment process can handle spikes in contaminants with a small change in operation (i.e., chemical dose change).
3	Treatment process cannot handle spikes in contaminants without adding a chemical process (e.g., coagulation for TOC removal) to meet regulation.
4	Treatment process cannot handle spikes in contaminants without adding a filtration process (e.g., roughing filter) to meet regulation.
5	Treatment process cannot handle spikes in contaminants without adding a disinfection process to meet regulation. (e.g., adding a disinfectant or increasing chlorine dose to meet the surface water treatment rule if E. coli is found in the source water).

Table C11. *Resilience to regulation and source water* criterion default scores (and basis).

Treatment Process	Score Default	Basis
Alum Coagulation	2	Not a standalone treatment process and can mitigate source water variability (i.e., coagulant chemical dose can be increased).
Ferric Coagulation	2	Not a standalone treatment process and can mitigate source water variability (i.e., coagulant chemical dose can be increased).
Bag Filtration	3	Not a standalone treatment process and does not have ability to mitigate source water variability (i.e., an additional treatment process must be added if source water degrades).
Cartridge Filtration	3	Not a standalone treatment process and does not have ability to mitigate source water variability (i.e., an additional treatment process must be added if source water degrades).
Conventional Filtration	2	Not a standalone treatment process and can mitigate source water variability (i.e., coagulant chemical dose can be increased).
Membrane Filtration	3	Not a standalone treatment process and does not have ability to mitigate source water variability (i.e., an additional treatment process must be added if source water degrades).
Nonozonated Biofiltration	2	Not a standalone treatment process and can mitigate source water variability (i.e., coagulant chemical dose can be increased).
Ozonated Biofiltration	1	Not a standalone treatment process and can mitigate source water variability (i.e., coagulant and ozone chemical doses can be increased).
Slow Sand Filtration	3	Not a standalone treatment process and does not have ability to mitigate source water variability (i.e., an additional treatment process must be added if source water degrades).
Chlorine Disinfection	4	Provides disinfection for pathogen reduction and can mitigate a limited amount of source water variability (i.e., chlorine chemical dose can be increased), but disinfection by-product formation limits how much the dose can be increased.
UV Disinfection	4	Provides disinfection for pathogen reduction and does not have ability to mitigate source water variability (i.e., an additional treatment process must be added if source water degrades). UV has potential benefits for removing contaminants of concern at high doses and when used with oxidants, but is ineffective with low UVT source water, so it was scored equivalently to chlorine in this model.
Caustic	5	Provides pH adjustment to prevent distribution system corrosion but does not provide contaminant removal.
Lime	5	Provides pH adjustment to prevent distribution system corrosion but does not provide contaminant removal.

Note: A standalone treatment process does not need any other treatment processes in the treatment train to meet regulation.

Table C12. Ranking table to define the *operator training requirement* criterion scores.

Score	Score Description
1	Treatment train does not require a certified operator.
2	Source water is from groundwater, or any of the following treatment processes are used: chlorine disinfection, UV disinfection, bag filtration, or cartridge filtration.
3	Any of the following treatment processes are used and followed by disinfection: membrane filtration or slow sand filtration.
4	Any of the following treatment processes are used and followed by disinfection: coagulation, conventional filtration, nonozonated biofiltration, ozonated biofiltration.
5	Drinking water system is large (serves more than 10,000 people).

Table C13. *Operator training requirement* criterion default scores, based on Colorado water treatment facility classification table.[254]

Treatment Process	Score Default
Alum Coagulation	4
Ferric Coagulation	4
Bag Filtration	2
Cartridge Filtration	2
Conventional Filtration	4
Membrane Filtration	3
Nonozonated Biofiltration	4
Ozonated Biofiltration	4
Slow Sand Filtration	3
Chlorine Disinfection	2
UV Disinfection	2
Caustic	2
Lime	2

Table C14. Ranking table to define the *system intrusiveness* criterion scores.

Score	Score Description*
1	No community nuisances are present.
2	One community nuisance is present.
3	Two community nuisances or one severe community nuisance is present.
4	Three community nuisances or two severe community nuisances are present.
5	Three severe community nuisances are present.

Note: Community nuisances include noticeability to the community and complaints about truck traffic as well as taste and odor.

Table C15. Ranking table to define the *independence from outside help* criterion scores.

Score	Score Description
1	No source water capacity received from other drinking water systems or parts or chemicals from third-party entities.
2	Up to 25% of either source water capacity received from other drinking water systems or parts or chemicals from third-party entities.
3	Up to 50% of either source water capacity received from other drinking water systems or parts or chemicals from third-party entities.
4	Up to 75% of either source water capacity received from other drinking water systems or parts or chemicals from third-party entities.
5	Up to 100% of either source water capacity received from other drinking water systems or parts or chemicals from third-party entities.

C7. Characterization of Additional Filtration Alternatives

The filtration alternatives not characterized in previously published models are discussed below.

C7.1 Operational Energy

Pumping power was determined using (Eq. A8). Pumping power was translated to operational energy by multiplying power by the number of hours in the functional unit timeframe (40 years).

$$\mathbf{P} = \frac{(\mathbf{Q} * \boldsymbol{\rho} * \mathbf{g} * \mathbf{H})}{\left(\mathbf{1000}\frac{\mathbf{W}}{\mathbf{kW}} * \boldsymbol{\eta}\right)} \qquad \textbf{Eq. C8}$$

Where:

P = Power (kW)
Q = Flow rate (m^3/s)
ρ = Density of liquid solution (kg/m^3): 1000 kg/m^3 for water
g = Gravity (9.81 m/s^2)
H = Head loss (m)
η = Efficiency (60%)

C7.2 Bag Filtration

The quantity of materials for the bag filters were determined based on accepted bag filtration technologies.[146,255] Filter mass was based on the total mass (i.e., total number of filters) needed for a given flow rate (i.e., based on allowable flow rates per filtration unit);[256] filter housing mass was determined the same way (based on existing filter housing masses and allowable flow rates).[219] The expected operational head loss of bag filters is 3.5 m;[147] the maximum allowable head loss is around 18 to 21 m.[137] A majority of bag filter configurations are filters in series or stand-alone filters;[146,255] the energy required to operate either was based on averaging the total power requirements of both configurations.

C7.3 Membrane Filtration

Membrane filtration refers to both microfiltration and ultrafiltration because those options are viewed similarly by regulations and have similar resource requirements.[137] Membrane module mass was based on a linear regression of dry module mass needed for a given flow rate (Figure C4), which were compiled from specification sheets of regulatorily accepted membranes.[257–259] These membranes are made of polyvinylidene fluoride; since data on this material is not available in LCI databases, LCI data for polyvinylidene chloride was used as a substitute; impacts of this substitution are expected to be minimal since impacts from the membrane material are expected to be negligible compared to operational energy use. Skid mass was determined by normalizing existing membrane skid weights to their flow rates[222,223,260] and then multiplying by the input flow rate. The expected operational head loss of membrane filtration is 10.5 m (with a range of 2.8 to 10.5 m);[31,106,220,221] the maximum allowable head loss can be up to 39 m.[137] This expected 10.5 m filter head loss results in 0.048 kWh/m^3 of operational energy, which is in agreement with literature expectations ranging from 0.029[106] to 0.046[31] kWh/m^3, given that the expected head loss is on the upper end of the expected range.

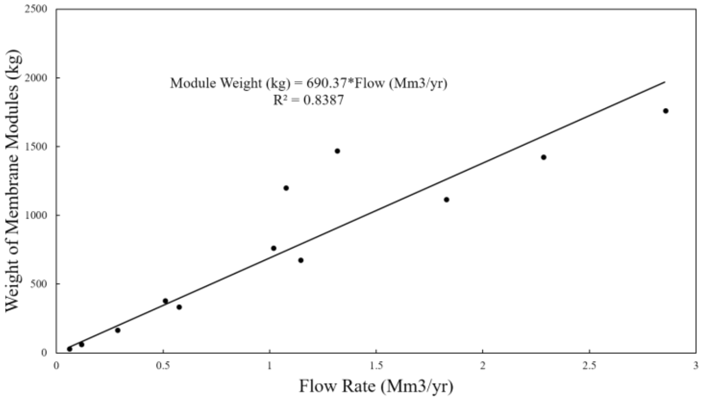


Figure C4. Membrane module weight versus flow for existing membrane modules.[257–259] Linear regression assumed a y-axis intercept of zero.

C7.4 Slow Sand Filtration

The slow sand filter area was calculated using Eq. A4, and the mass of media was calculated using Eq. A5. These equations used typical values for hydraulic loading rate and media depths of sand and gravel.[217,218] A freeboard of 0.3 m was also considered.[96] Then, the filter volume (based on filter area and depth), a typical concrete thickness,[201] and square cross-section were used to determine the mass of concrete (Eq. A7) and rebar (Eq. B10) needed for the filter housing. Rebar was separated by 0.305 meter spacing (along the width) for each concrete segment.[97] Filter operational energy calculations (Eq. A8) assumed the total filter head loss was the same as the total filter depth. The TOC removal due to the slow sand filter was based on previous literature.[175] As done in the published literature for rapid media biofiltration,[167] the slow sand filter TOC removal was assumed to be removed as biodegradable dissolved organic carbon (BDOC). Also, full biological degradation of BDOC in the filter may not be achieved if the BDOC entering the filter is below the expected filter removal; this can happen when the BDOC of the source water is low or when coagulation removes a significant amount of BDOC.[167] There is more uncertainty with the TOC removal estimates for slow sand filtration than for rapid media biofiltration due to limited experimental data.

$$\mathbf{A_T = \frac{Q}{HLR}} \qquad \textbf{Eq. C9}$$

Where:

A_T = Total filter area requirement (m^2)
Q = Plant capacity flow rate (m3/hr)
HLR = Filter design hydraulic loading rate (m/hr)

$$\mathbf{M_{media} = A_T * D_{media} * \rho_{media}} \qquad \textbf{Eq. C10}$$

Where:

M_{media} = Mass of filter media (kg)
D_{media} = Media Depth (m) (Table C4)
ρ_{media} = Media density (kg/m^3): (1,500 kg/m^3 sand,[51] and 1,362 kg/m^3 gravel[150])

$$\mathbf{V_{Filter}} = \mathbf{A_T} * (\mathbf{D_{media}} + \mathbf{H_{fb}} + \mathbf{H_L}) \qquad \textbf{Eq. C11}$$

Where:

V_{Filter} = Required filter volume (m^3)
H_{fb} = freeboard (m)
H_L = Operational above media filter head loss (m)
$D_{expansion}$ = Backwash filter expansion depth (m): Assumed 50% bed expansion[200]

$$V_{concrete} = \left(\left(\sqrt{\frac{\mathbf{V_{filter}}}{\mathbf{D_{total}}}}\right) * \mathbf{t_b}\right) + \left(\mathbf{4t_w}^2 + \mathbf{4}\left(\sqrt{\frac{\mathbf{V_{filter}}}{\mathbf{D_{total}}}}\right) * \mathbf{t_w}\right) * (\mathbf{D_{total}} + \mathbf{t_B}) \qquad \textbf{Eq. C12}$$

Where:

D_{total} = Sum of media depth, and filter head requirement (m)
t_b = Thickness of filter base (m)
t_w = Thickness of filter walls (m)

$$M_{rebar} = \left(\frac{W_t * \mathrm{L} + \mathrm{D} * \mathrm{L} + \mathrm{D} * W_t}{W_s}\right) * \mathrm{R}_{M/L} \qquad \textbf{Eq. C13}$$

Where:

M_{rebar} = weight of reinforcing rebar (kg) (value was rounded up to near integer)
W_s = width of rebar spacing (m): 0.305 m from ([127])
$R_{M/L}$ = mass to length ratio of rebar (kg/m): 1 kg/m from ([127])

C7.5 Static Mixer

Since conventional filtration, nonozonated biofiltration, and ozonated biofiltration must be coupled with coagulation, pumping energy for a static mixer (based on 0.41 m of head loss[224]) was also included in their inventories whenever a functional unit did not require any other filtration option to have coagulation (i.e., when treating low turbidity source water or when no TOC removal is required) to ensure similar treatment process system boundaries (i.e., an accurate life cycle comparison.).

C8. SIPS Preset Criteria Weighting Schemes

Table C16. Preset criteria weighting schemes in SIPS.

Decision Criteria	Equal Category Weights (%)	Equal Criteria Weights (%)	Health Focused Weights (%)	Pollution Focused Weights (%)	Cost Focused Weights (%)	Operation Focused Weights (%)
Affordability	**25**	**17**	**27**	**14**	**50**	**14**
Capital Cost	12.5	8	7	4	13	4
O&M Cost	12.5	8	20	10	37	10
Global Pollution	**25**	**25**	**14**	**50**	**14**	**14**
Global Climate Change	8.3	8	8	29	8	8
Global Ecosystem Quality	8.3	8	4	14	4	4
Global Resources	8.3	8	2	7	2	2
Human Health	**25**	**25**	**50**	**27**	**27**	**27**
Global Human Health	8.3	8	7	4	4	4
Local Long-term Health Risk	8.3	8	14	8	8	8
Resilience to Regulation and Source Water	8.3	8	29	15	15	15
Plant Operational Challenges	**25**	**33**	**9**	**9**	**9**	**45**
Operator Training Requirement	6.25	8	5	5	5	25
Maintenance Requirements	6.25	8	2	2	2	10
System Intrusiveness	6.25	8	1	1	1	5
Independence from Outside Help	6.25	8	1	1	1	5

C9. Criterion Scores for Hypothetical Scenarios

S*ystem intrusiveness* criterion scores: conventional, cartridge, and membrane filtration treatment trains were assigned a score of 2 because they were assumed to have one intrusiveness consideration in the form of taste and odor complaints; nonozonated biofiltration alternative was given a score of 1 because the biofilter was expected to remove taste and odor compounds.

Independence from outside help criterion scores: conventional and nonozonated biofiltration treatment trains were assigned a score of 2 because they use chemical coagulation, which may increase reliance on third-party entities; cartridge and membrane filtration trains were assigned a score of 1 because these had treatment processes that required no third-party aid for the purposes of this example. In Scenarios 2 and 3, chemical coagulation was added to both the cartridge and membrane filtration trains, so those alternatives were assigned a score of 2 (same as nonozonated biofiltration and conventional filtration).

Scenario 4’s *system intrusiveness* criterion scores: chlorine disinfection alternatives were given a score of 2 and UV a score of 1 because UV reduced chlorine use, which was expected to improve the taste and odor of the water. Scenario 4’s *independence from outside help* criterion scores: all disinfection alternatives were given a score of 1 because each was expected to have about the same reliance on third-party entities.

Appendix D: Supporting Information for: "Evaluating Small System Treatment Processes for Different Source Water Qualities, Regulations, and Stakeholder Preferences"

The following manuscript and appendix are in preparation for submission for publication: Jones, C. H.; Cook, S. M. Evaluating Small System Treatment Processes for Different Source Water Qualities, Regulations, and Stakeholder Preferences.

D1. Uncertainty Results

D1.1 Rank Range Plots

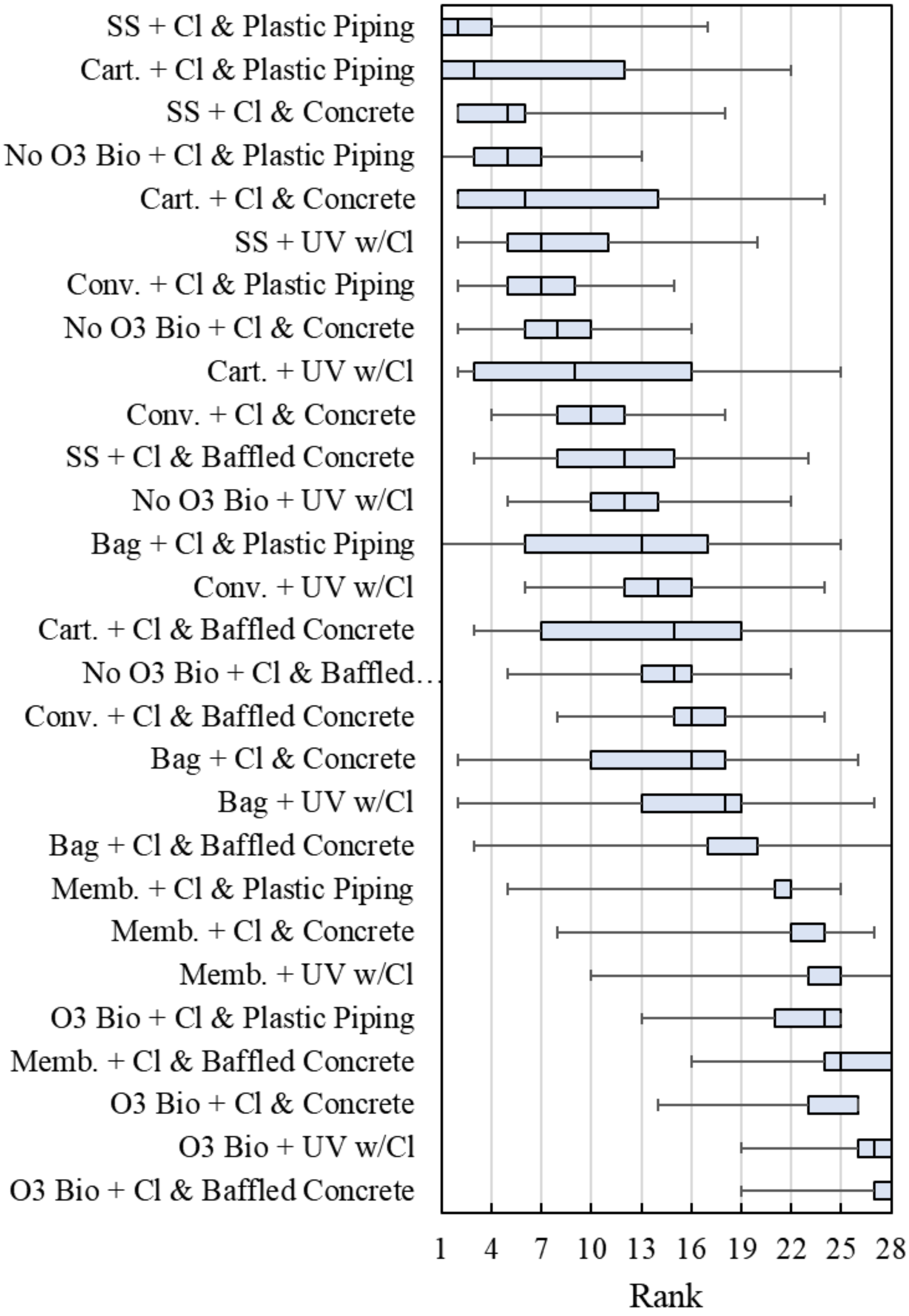


Figure D.1. Rank distributions by treatment alternative due to different design parameter sets for the low turbidity low TOC regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

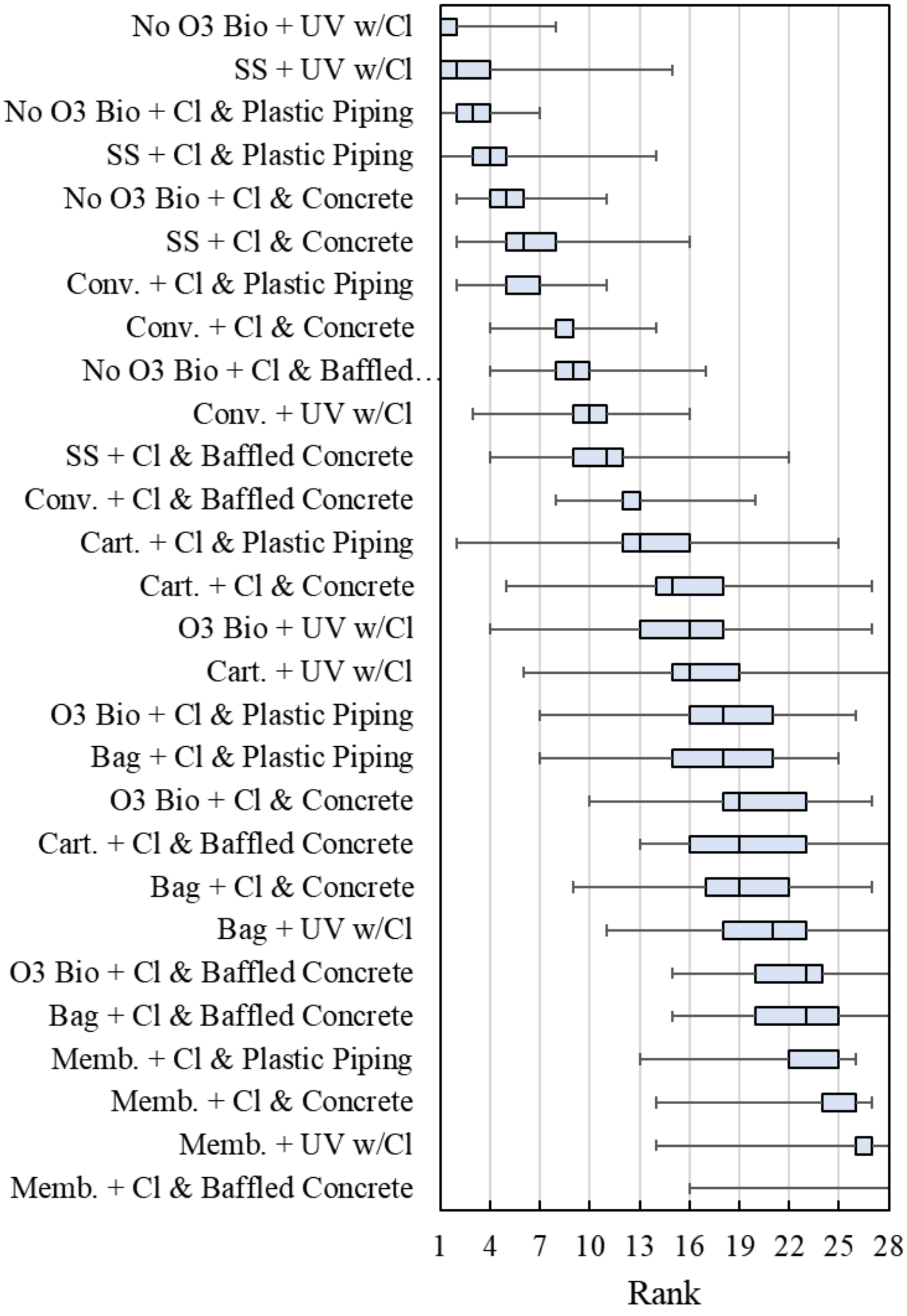


Figure D.2. Rank distributions by treatment alternative due to different design parameter sets for the high turbidity 30% TOC removal regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

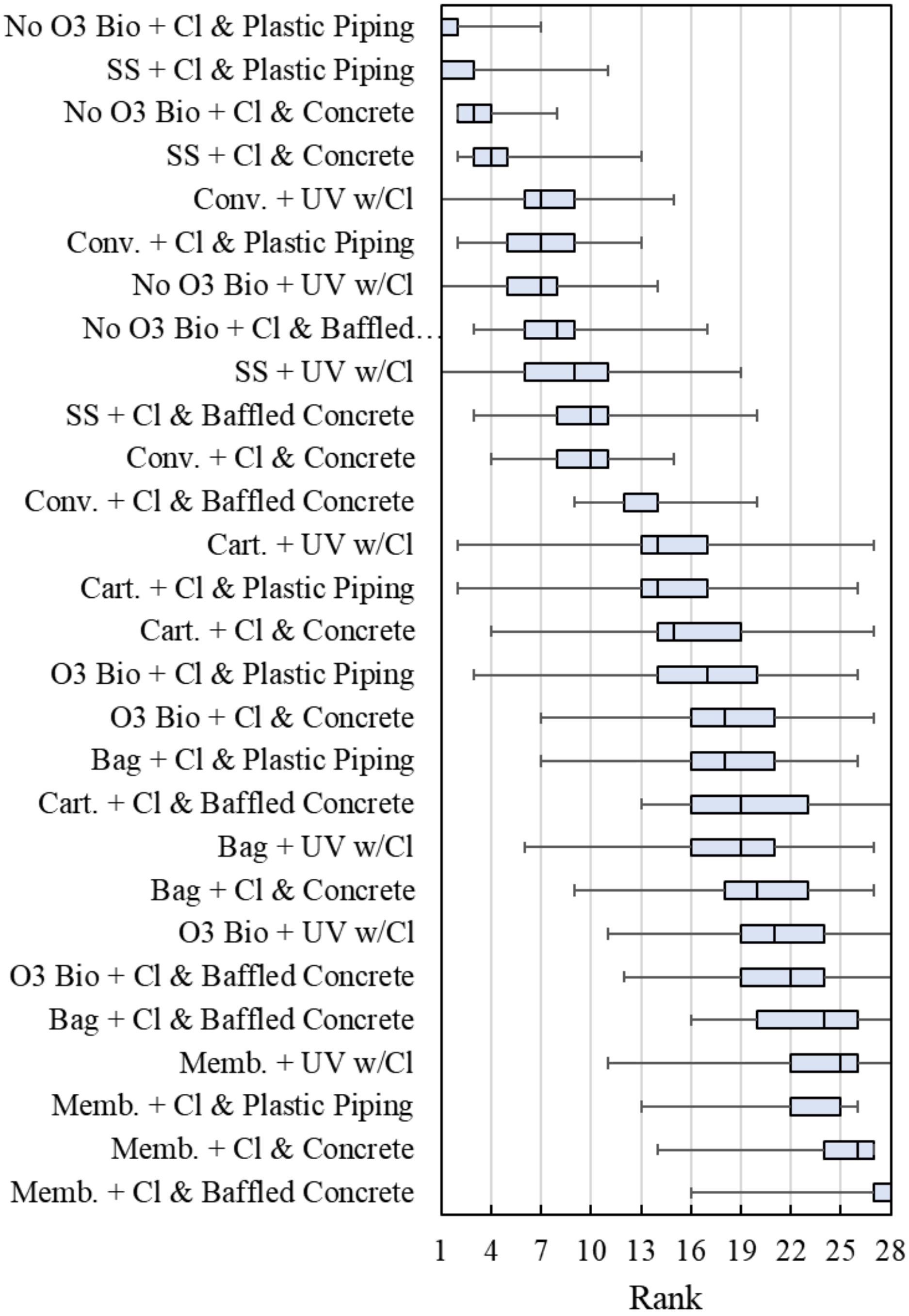


Figure D.3. Rank distributions by treatment alternative due to different design parameter sets for the high turbidity U.S. TOC removal regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

D1.2 Sensitivity Results

Table D.1. Summary of rank trends for the Regulatory Scenario#1 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	**Sensitive Parameter Spearman Correlation Coefficient |ρ|**
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	Bag Filter Head Loss Stage 1 (0.47), Bag Filter Head Loss Stage 2 (0.64)
Cartridge Filtration	Cartridge Filter Head Loss (0.90)
Conventional Filtration	Baseline Turbidity Coag. Dose (0.36), Cartridge Filter Head Loss (-0.51), Bag Filter Head Loss Stage 2 (-0.34)
Membrane Filtration	Pre-oxidation O3 Dose (-0.35), Membrane Head Loss (0.60)
Nonozonated Biofiltration	Baseline Turbidity Coag. Dose (0.40), Cartridge Filter Head Loss (-0.50)
Ozonated Biofiltration	Pre-oxidation O3 Dose (0.47), Membrane Head Loss (-0.52)
Slow Sand Filtration	Slow Sand Filter HLR (-0.47), Cartridge Filter Head Loss (-0.51)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	UV Chamber Head Loss (-0.30)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	N/A
UV W/Chlorine Residual	UV Chamber Head Loss (0.30)

Table D.2. Summary of rank trends for the Regulatory Scenario#2 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	**Sensitive Parameter Spearman Correlation Coefficient |ρ|**
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	Bag Filter Head Loss Stage 1 (0.38), Bag Filter Head Loss Stage 2 (0.58), Cartridge Filter Head Loss (-0.30), Pre-oxidation O3 Dose (-0.30)
Cartridge Filtration	Cartridge Filter Head Loss (0.77)
Conventional Filtration	Slow Sand Filter HLR (0.59)
Membrane Filtration	Membrane Head Loss (0.52)
Nonozonated Biofiltration	Slow Sand Filter HLR (0.72)
Ozonated Biofiltration	Pre-oxidation O3 Dose (0.64)
Slow Sand Filtration	Slow Sand Filter HLR (-0.80)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	UV Chamber Head Loss (-0.34)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	N/A
UV W/Chlorine Residual	UV Chamber Head Loss (0.35)

Table D.3. Summary of rank trends for the Regulatory Scenario#3 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	Sensitive Parameter Spearman Correlation Coefficient |ρ|
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	Bag Filter Head Loss Stage 1 (0.39), Bag Filter Head Loss Stage 2 (0.60)
Cartridge Filtration	Cartridge Filter Head Loss (0.78)
Conventional Filtration	N/A
Membrane Filtration	Membrane Head Loss (0.55)
Nonozonated Biofiltration	Slow Sand Filter HLR (0.65)
Ozonated Biofiltration	Bag Filter Head Loss Stage 2 (-0.31), Cartridge Filter Head Loss (-0.38), Pre-oxidation O3 Dose (0.62),
Slow Sand Filtration	Slow Sand Filter HLR (-0.69), Cartridge Filter Head Loss (-0.51)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	Baseline Turbidity Coag. Dose (-0.49)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	Baseline Turbidity Coag. Dose (-0.52)
UV W/Chlorine Residual	Baseline Turbidity Coag. Dose (0.57)

D2. Source Water Quality Results

D2.1 Rank Range Plots

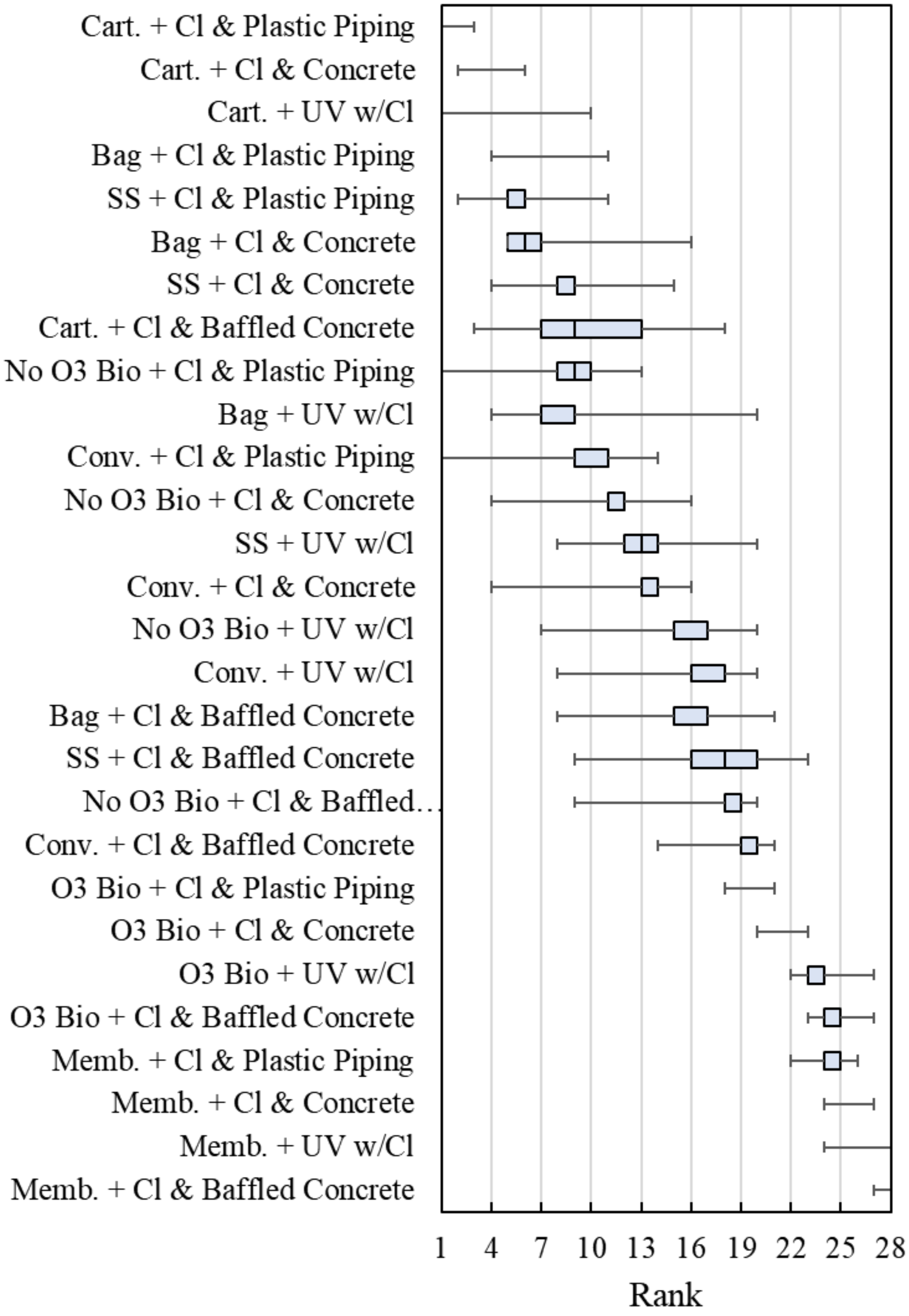


Figure D.4. Rank distributions by treatment alternative due to changing source water quality for the low turbidity low TOC regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

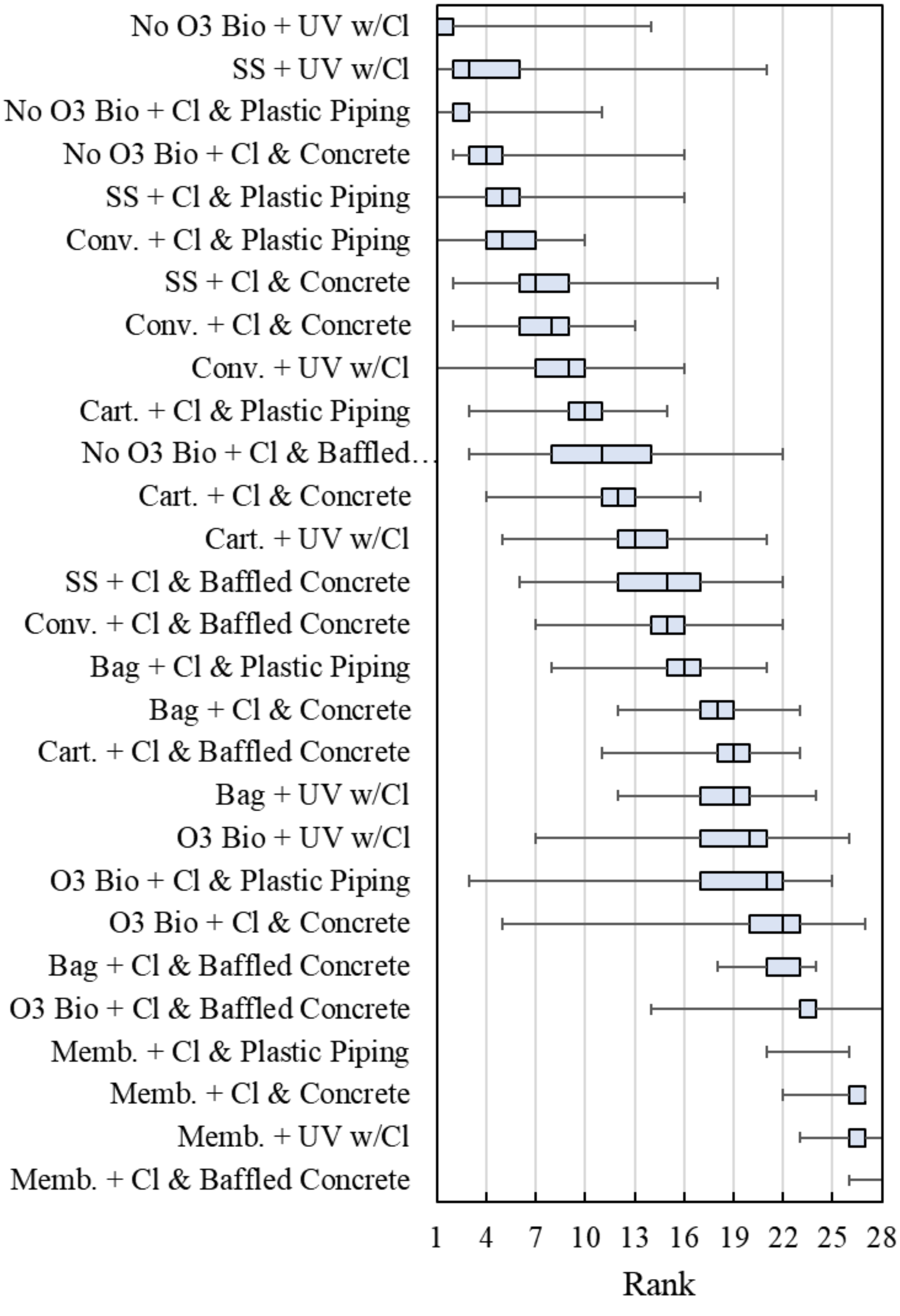


Figure D.5. Rank distributions by treatment alternative due to changing source water quality for the high turbidity 30% TOC removal regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

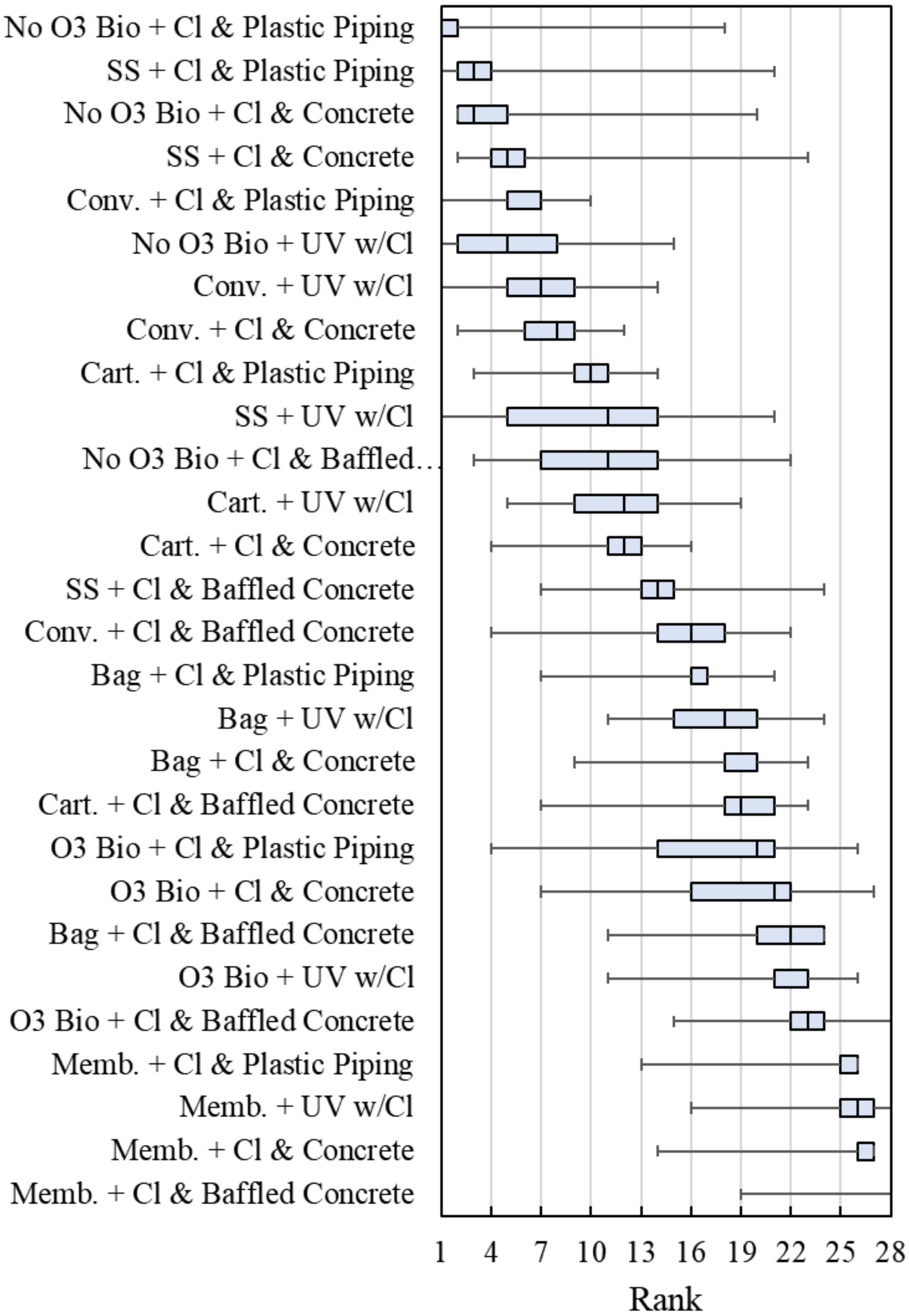


Figure D.6. Rank distributions by treatment alternative due to changing source water quality for the high turbidity U.S. TOC removal regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

D2.3 Sensitivity Results

Table D.4. Summary of rank trends for the Regulatory Scenario#1 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	Sensitive Parameter Spearman Correlation Coefficient |ρ|
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	pH (-0.42)
Cartridge Filtration	N/A
Conventional Filtration	N/A
Membrane Filtration	N/A
Nonozonated Biofiltration	N/A
Ozonated Biofiltration	N/A
Slow Sand Filtration	pH (0.51)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	pH (0.34), Temperature (-0.40)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	N/A
UV W/Chlorine Residual	pH (-0.37), Temperature (0.43)

Table D.5. Summary of rank trends for the Regulatory Scenario#2 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	**Sensitive Parameter Spearman Correlation Coefficient |ρ|**
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	TOC (-0.66)
Cartridge Filtration	TOC (-0.39)
Conventional Filtration	Alkalinity (0.31)
Membrane Filtration	N/A
Nonozonated Biofiltration	Temperature (-0.32)
Ozonated Biofiltration	TOC (0.68)
Slow Sand Filtration	Alkalinity (-0.30)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	Temperature (-0.44)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	N/A
UV W/Chlorine Residual	Temperature (0.43)

Table D.6. Summary of rank trends for the Regulatory Scenario#3 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	**Sensitive Parameter Spearman Correlation Coefficient |ρ|**
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	TOC (-0.82)
Cartridge Filtration	TOC (-0.51)
Conventional Filtration	N/A
Membrane Filtration	N/A
Nonozonated Biofiltration	Temperature (-0.42)
Ozonated Biofiltration	TOC (0.85)
Slow Sand Filtration	Temperature (0.43)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	Temperature (-0.54)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	Temperature (-0.40)
UV W/Chlorine Residual	Temperature (0.53)

D3. Decision-criteria Weights Results

D3.1 Rank Range Plots

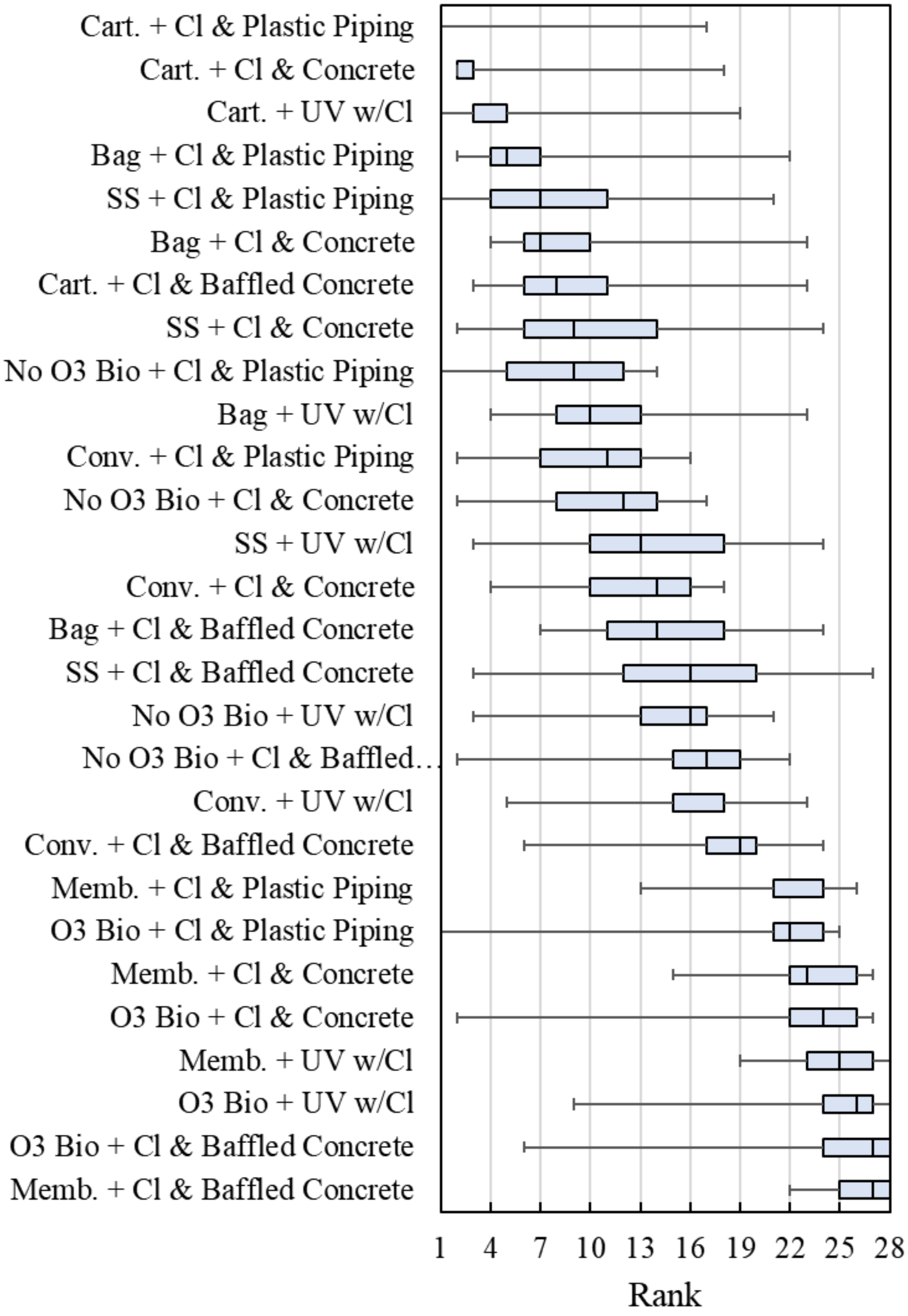


Figure D.7. Rank distributions by treatment alternative due to changing stakeholder preferences for the low turbidity low TOC regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value

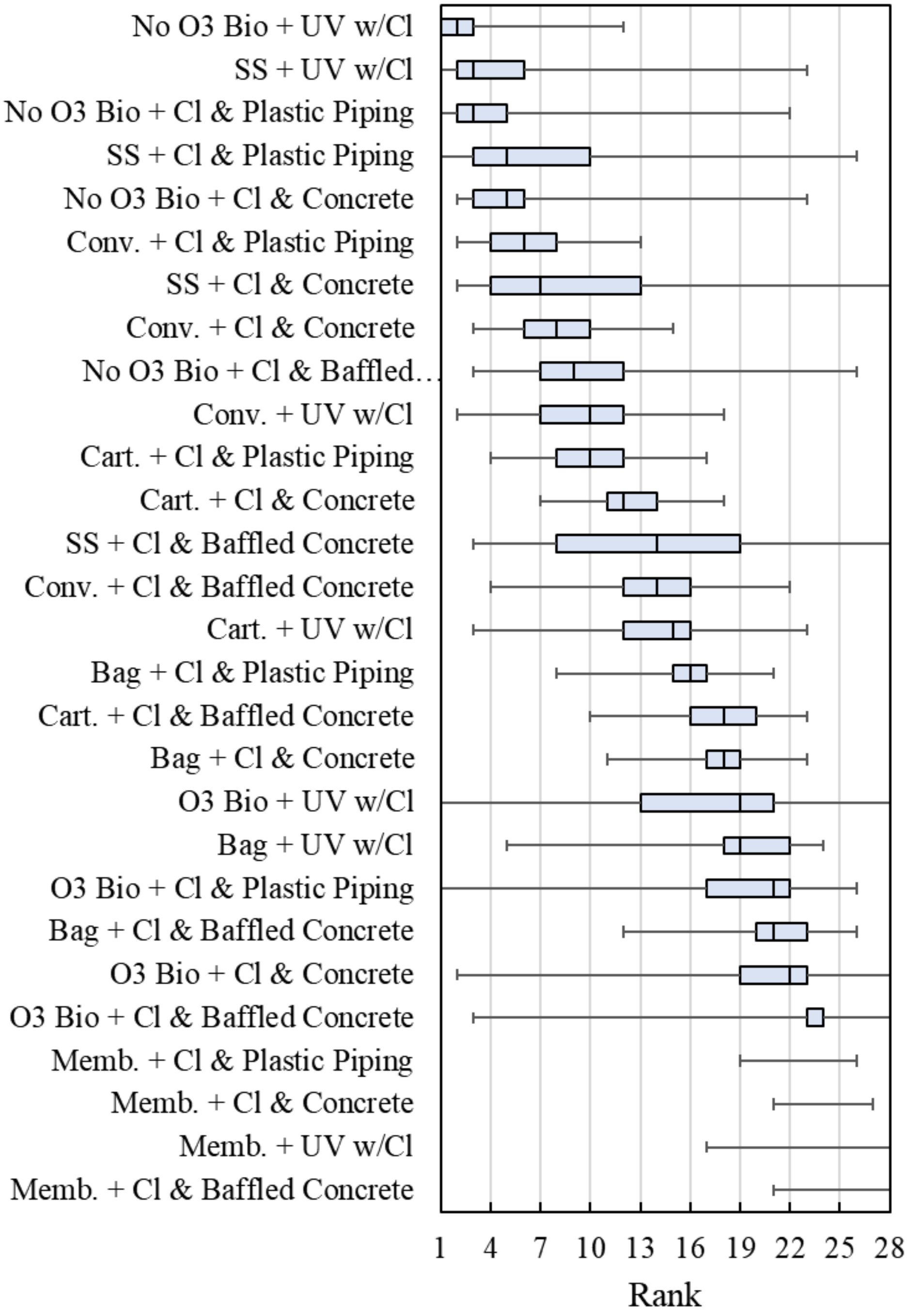


Figure D.8. Rank distributions by treatment alternative due to changing stakeholder preferences for the high turbidity 30% TOC removal regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

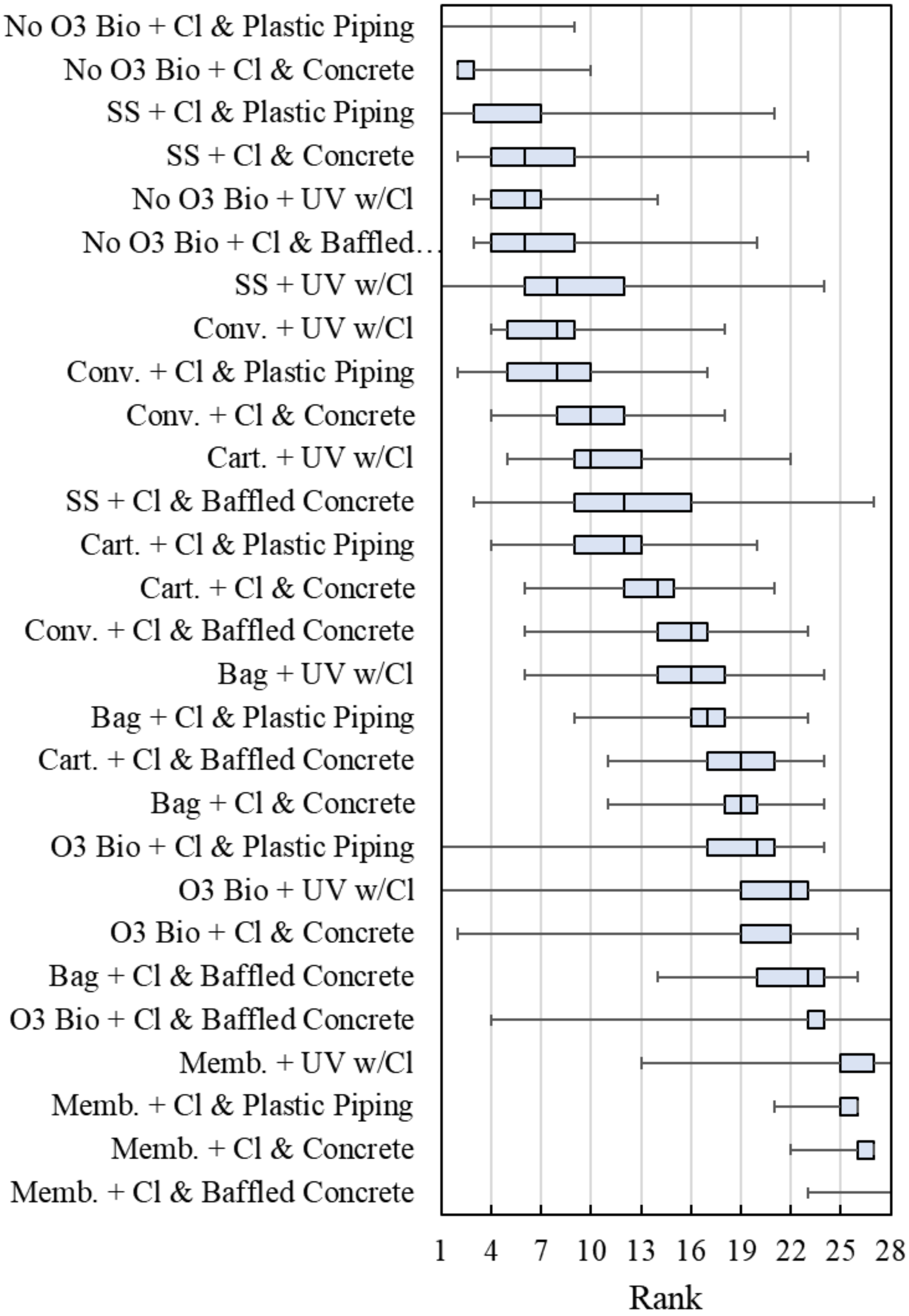


Figure D.9. Rank distributions by treatment alternative due to changing stakeholder preferences for the high turbidity U.S. TOC removal regulation standard. The box represents the 25, 50, and 75 percentiles while error bars represent minimum and maximum ranks. Note: when the min max or quartiles are not visible one or many of those are the same value.

D3.2 Sensitivity Results

Table D.7. Summary of rank trends for the Regulatory Scenario#1 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	Sensitive Parameter Spearman Correlation Coefficient |ρ|
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	Capital Cost (-0.34), Local Long-term Health Risk (0.40), Operator Training Requirement (-0.57)
Cartridge Filtration	Local Long-term Health Risk (0.30), Operator Training Requirement (-0.44)
Conventional Filtration	Local Long-term Health Risk (-0.45), Operator Training Requirement (0.43)
Membrane Filtration	Local Long-term Health Risk (0.49), Resilience to Regulation and Source Water (0.49)
Nonozonated Biofiltration	Local Long-term Health Risk (-0.51), Operator Training Requirement (0.43)
Ozonated Biofiltration	Local Long-term Health Risk (-0.49), Resilience to Regulation and Source Water (-0.50)
Slow Sand Filtration	Capital Cost (0.67), Maintenance Requirements (-0.38)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	N/A
Chlorine W/Concrete & Steel Baffles	Global Ecosystem Quality (0.38)
Chlorine W/Plastic Piping	N/A
UV W/Chlorine Residual	Local Long-term Health Risk (-0.35), Operation & Maintenance Cost (0.33)

Table D.8. Summary of rank trends for the Regulatory Scenario#2 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	**Sensitive Parameter Spearman Correlation Coefficient |ρ|**
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	Capital Cost (-0.45), Local Long-term Health Risk (-0.42), Resilience to Regulation and Source Water (0.34)
Cartridge Filtration	Capital Cost (-0.48), Local Long-term Health Risk (-0.44), Maintenance Value (0.30)
Conventional Filtration	Capital Cost (-0.43), Local Long-term Health Risk (-0.55)
Membrane Filtration	N/A
Nonozonated Biofiltration	Capital Cost (-0.50)
Ozonated Biofiltration	Capital Cost (0.32), Local Long-term Health Risk (0.35), Resilience to Regulation and Source Water (-0.53)
Slow Sand Filtration	Capital Cost (0.66), Local Long-term Health Risk (0.32), Maintenance Requirements (-0.47)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	Local Long-term Health Risk (0.47), Operation & Maintenance Cost (-0.33)
Chlorine W/Concrete & Steel Baffles	Local Long-term Health Risk (0.33), Global Ecosystem Quality (0.42)
Chlorine W/Plastic Piping	N/A
UV W/Chlorine Residual	Local Long-term Health Risk (-0.54), Operation & Maintenance Cost (0.38)

Table D.9. Summary of rank trends for the Regulatory Scenario#3 from 10,000 Monte Carlo trials. Filtration alternatives were all followed by chlorine w/plastic piping, and disinfection alternatives were all preceded by cartridge filtration. Note: N/A indicates that changing design parameters did not result in a treatment process model-based change in recommendation rank for an alternative.

Treatment Process	**Sensitive Parameter Spearman Correlation Coefficient |ρ|**
Filtration Rank Sensitivity (Chlorine W/Plastic Piping was the Disinfectant after each Filter)	
Bag Filtration	N/A
Cartridge Filtration	N/A
Conventional Filtration	Capital Cost (-0.32)
Membrane Filtration	N/A
Nonozonated Biofiltration	Capital Cost (-0.35)
Ozonated Biofiltration	N/A
Slow Sand Filtration	Capital Cost (0.42)
Disinfection Rank Sensitivity (Cartridge Filtration was the Filter used Before each Disinfectant)	
Chlorine W/Concrete	Local Long-term Health Risk (0.76)
Chlorine W/Concrete & Steel Baffles	N/A
Chlorine W/Plastic Piping	Local Long-term Health Risk (0.81)
UV W/Chlorine Residual	Local Long-term Health Risk (-0.84)

D3.2 Pilot Survey Data

Survey data based on three small drinking water system stakeholders (IRB# 17-0333) suggested that stakeholders value human health criteria the most followed by affordability (Figure 5.5). When paired with results from simulating stakeholder inputs, actual stakeholders seemed to value three out of the four most sensitive decision criteria (*local long-term health risk*, *resilience to regulation and source water*, and *capital cost*) highly (even though the sample size is three stakeholders). Treatment alternatives that used more complex processes had the potential to overtreat the water could be preferentially recommended due to their performance in *Resilience to regulation and source water* and *local long-term health risk*. Not all stakeholders will align with these criteria preferences, which is why stakeholder preferences should be determined on a case by case basis.