

FINAL REPORT

Evaluation and Life Cycle Comparison of Ex Situ Treatment Technologies for PFAS in Groundwater

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May 2023

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REPORT DOCUMENTATION PAGE					Form Approved OMB No. 0704-0188	
The public reporting burden for this collection of information is estimated to average 1 hour per response, including the time for reviewing instructions, searching existing data sources, gathering and maintaining the data needed, and completing and reviewing the collection of information. Send comments regarding this burden estimate or any other aspect of this collection of information, including suggestions for reducing the burden, to Department of Defense, Washington Headquarters Services, Directorate for Information Operations and Reports (0704-0188), 1215 Jefferson Davis Highway, Suite 1204, Arlington, VA 22202-4302. Respondents should be aware that notwithstanding any other provision of law, no person shall be subject to any penalty for failing to comply with a collection of information if it does not display a currently valid OMB control number. PLEASE DO NOT RETURN YOUR FORM TO THE ABOVE ADDRESS.						
1. REPORT DATE (DD-MM-YYYY) 30/5/2023		2. REPORT TYPE ESTCP Final Report		3. DATES COVERED (From - To) 9/27/2018 - 9/25/2023		
4. TITLE AND SUBTITLE Evaluation and Life Cycle Comparison of Ex Situ Treatment Technologies for PFAS in Groundwater				5a. CONTRACT NUMBER 18-C-0048		
				5b. GRANT NUMBER		
				5c. PROGRAM ELEMENT NUMBER		
6. AUTHOR(S) Kenan Ozekin and Alice Fulmer: Water Research Foundation Chris Bellona and Chris Higgins: Colorado School of Mines Charles Schaefer: CDM Smith Detlef Knappe: North Carolina State University Sherri Cook: University of Colorado, Boulder				5d. PROJECT NUMBER ER18-5053		
				5e. TASK NUMBER		
				5f. WORK UNIT NUMBER		
7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES) Water Research Foundation 6666 W. Quincy Ave. Denver, CO 80235				8. PERFORMING ORGANIZATION REPORT NUMBER ER18-5053		
9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES) Office of the Deputy Assistant Secretary of Defense (Energy Resilience & Optimization) 3500 Defense Pentagon, RM 5C646 Washington, DC 20301-3500				10. SPONSOR/MONITOR'S ACRONYM(S) ESTCP		
				11. SPONSOR/MONITOR'S REPORT NUMBER(S) ER18-5053		
12. DISTRIBUTION/AVAILABILITY STATEMENT DISTRIBUTION STATEMENT A. Approved for public release: distribution unlimited.						
13. SUPPLEMENTARY NOTES						
14. ABSTRACT The principal objective of this research was to assess the effectiveness of GAC adsorption, IX, and membrane processes for the removal of PFAS from different source waters and quantify the benefits and dis-benefits (treatment capacity, life cycle costs, etc.) for each technology as either stand-alone processes or integrated treatment trains. Specific objectives were to: • Develop a comprehensive assessment framework for ex-situ PFAS treatment technologies, • Generate a PFAS treatment efficiency database to support development of a decision support tool. • Develop a treatment technology decision support tool. These objectives were achieved through systematic investigation of PFAS removal under various treatment scenarios and development of a life cycle impact and costing model to quantify and compare the technologies for different treatment scenarios.						
15. SUBJECT TERMS Life Cycle Comparison, Ex Situ Treatment Technologies, PFAS, Groundwater						
16. SECURITY CLASSIFICATION OF:			17. LIMITATION OF ABSTRACT	18. NUMBER OF PAGES	19a. NAME OF RESPONSIBLE PERSON	
a. REPORT	b. ABSTRACT	c. THIS PAGE			Kenan Ozekin	
UNCLASS	UNCLASS	UNCLASS	UNCLASS	162	19b. TELEPHONE NUMBER (Include area code) 303-734-3464	

FINAL REPORT

Project: ER18-5053

TABLE OF CONTENTS

	Page
ABSTRACT.....	vii
EXECUTIVE SUMMARY	ES-1
1.0 INTRODUCTION AND OBJECTIVES	1
1.1 INTRODUCTION	1
1.2 OBJECTIVES	1
1.3 REGULATORY DRIVERS	2
2.0 PERFORMANCE OBJECTIVES	4
3.0 RESULTS AND DISCUSSION	6
3.1 Granular Activated Carbon Adsorption and Ion Exchange	6
3.1.1 Background	6
3.1.2 Materials and Methods	7
3.2 PFAS removal by GAC	11
3.2.1 Effect of initial PFAS concentration and competition among co-occurring PFAS.....	11
3.2.2 Effect of empty bed contact time (EBCT) on scaled-up PFAS removal.....	11
3.2.3 Effect of GAC type on scaled-up PFAS removal.....	13
3.2.4 Effects of total organic carbon (TOC) on the scaled-up removal of PFAS.....	14
3.2.5 Effects of inorganic anions on the scaled-up removal of PFAS.....	16
3.2.6 Effect of aqueous film-forming foam (AFFF) on scaled-up PFAS removal.....	17
3.2.7 Comparison of treatment criteria: Impact on GAC use rates	18
3.3 PFAS removal by IX.....	20
3.3.1 Effects of initial PFAS concentration and competition between co-existing PFAS	20
3.3.2 Effect of empty bed contact time (EBCT) on scaled-up PFAS removal by IX.....	21
3.3.3 Effect of IX type on scaled-up PFAS removal.....	23
3.3.4 Effect of total organic carbon (TOC) on scaled-up PFAS removal	23
3.3.5 Effect of inorganic anions on scaled-up PFAS removal	25
3.3.6 Comparison of treatment criteria: Impact on IX use rates	26
3.4 High-Pressure Membranes.....	29
3.4.1 Background	29
3.4.2 Materials and Methods	30
3.5 PFAS Removal by High-Pressure Membranes.....	35
3.5.1 Comparison of Membrane Performance	35
3.5.2 Membrane System Permeate Flux and Recovery.....	41
3.5.3 Ionic Strength.....	43
3.5.4 PFAS Concentration and Adsorption.....	46
3.5.5 Comparison of Membrane Types Evaluated.....	47
3.5.6 Advantages and Disadvantages of High-Pressure Membranes	50
4.0 PFAS TREATMENT TECHNOLOGY DECISION SUPPORT TOOL.....	51
4.1 Stakeholder Weighting Inputs.....	52

TABLE OF CONTENTS (Continued)

	Page
4.2 Treatment Technology Options and Design Inputs	53
4.2.1 GAC Gravity Basin	53
4.2.2 GAC Pressure Filter	54
4.2.3 IX Filter with Cartridge Prefilter	54
4.2.4 IX Filter with Backwashing	55
4.2.5 Membranes (RO or NF)	55
4.3 Treatment Performance Parameters	56
4.3.1 Media Disposal and Hauling	58
4.4 Life Cycle Inventory Data Calculations	62
4.4.1 Energy	62
4.4.2 Materials	64
4.5 Evaluation Methods	69
4.5.1 Affordability	69
4.5.2 Facility Operation	71
4.5.3 Environmental and Human Health Impacts	72
4.6 Overall Score and Recommendation	72
4.7 RESULTS	74
4.7.1 Navigating Treatment Performance Uncertainty using Scenario Analysis	74
4.7.2 Advantages and Disadvantages of Treatment Processes	89
4.7.3 Case Study Analysis	89
4.8 Evaluation of Treatability Data with Decision Support Tool	93
4.8.1 Impact of PFAS Regulations and TOC on GAC Costs for the Treatment of PFAS-Impacted Groundwater	94
4.8.2 Evaluation of GAC Data from Pilot-scale Experiments and RSSCTs	98
4.8.3 Impact of PFAS Regulations and TOC on IX Costs for the Treatment of PFAS-Impacted Groundwater	99
4.8.4 Cost-comparison between Different Treatment Options (IX, GAC, and RO)	101
5.0 CONCLUSIONS, IMPLEMENTATION ISSUES AND DATA GAPS	103
6.0 REFERENCES	107
Appendix A BREAKTHROUGH CURVES FOR PFAS	A-1
Appendix B DECISION SUPPORT TOOL USER INTERFACE	B-1
Appendix C COST NUMBERS	C-1
Appendix D EXPERT PANEL WORKSHOPS	D-1

LIST OF FIGURES

	Page
Figure 1.1. Schematic of PFAS Treatment Technologies Evaluated.	2
Figure 3.1. Raw and Scaled-up PFBS, PFHxS, and PFOS Breakthrough Data from Single-solute, Bi-solute, and Tri-solute CD-RSSCTs.	12
Figure 3.2. Effect of Empty Bed Contact Time (EBCT) on Scaled-up Bed Volume to 10% and 50% Breakthrough.	13
Figure 3.3. Effect of GAC Selection on Scaled-up Bed Volume to 10% and 50% Breakthrough.	14
Figure 3.4. Effect of TOC on Scaled-up Bed Volume to 10% and 50% Breakthrough.....	15
Figure 3.5. Effect of TOC on Scaled-up Bed Volume to 10% and 50% Breakthrough (partial plots).....	15
Figure 3.6. Effect of Inorganic Anions on Scaled-up Bed Volume to 10% and 50% Breakthrough.....	16
Figure 3.7. Effect of 3 meq/L Inorganic Anions on Normalized Bed Volume to 10% Breakthrough (BV10/BV10 Control) After Scale-up.	17
Figure 3.8. Effects of AFFF Matrices (AFFF and Aged, Weathered AFFF) on Scaled-up Bed Volume to 10% and 50% Breakthrough for PFHxS and PFOS.	18
Figure 3.9. Effect of Different Treatment Goals on GAC Use Rates.	19
Figure 3.10. PFHxA and PFOA Breakthrough Data from Single-solute (Spiked with PFBS Only) and Tri-solute (Spiked with PFBS, PFHxA and PFOA) CD-RSSCTs.....	21
Figure 3.11. PFBS Breakthrough Data from Pseudo-single-solute (Dosed with 0.2 nM PFBS Only) and Tri-solute (Dosed with PFBS, PFHxA, and PFOA) CD-RSSCTs.	22
Figure 3.12. Effect of Empty Bed Contact Time (EBCT) on Scaled-up Bed Volume to 10% Breakthrough. IX.	22
Figure 3.13. Effect of GAC Selection on Scaled-up Bed Volume to 10% and 50% Breakthrough.	23
Figure 3.14. Effect of TOC on Scaled-up Bed Volume to 10% and 50% Breakthrough.....	24
Figure 3.15. Effect of TOC on KAIX.	24
Figure 3.16. Effect of Inorganic Anions on Scaled-up Bed Volume to 10% and 50% Breakthrough.....	25
Figure 3.17. Effect of Coexisting Inorganic Anions on KAIX.	26
Figure 3.18. Effect of Different Treatment Goals on IX Use Rates.....	28
Figure 3.19. Membrane Pilot-scale System Photographs.....	31
Figure 3.20. Schematic of 2540 Pilot-scale System.....	32
Figure 3.21. Illustration of a CCD-NF/RO System.....	33
Figure 3.22. Rejection of PFAAs by the NF270 Membrane (Left) and the ESPA2 Membrane (Right) During PFAA Amended DI Feed Water Experiments.....	37

LIST OF FIGURES

	Page
Figure 3.23. Rejection of PFAAs by the NF270 Membrane (Left) and the ESPA2 Membrane (Right) During AFFF Contaminated Groundwater Experiments.	37
Figure 3.24. Rejection of PFAAs by the NF270 Membrane (Top Left), NF90 membrane (Top Right), CR100 Membrane (Bottom Left), and SW30 (Bottom Right) Measured during CCD Experiments.....	39
Figure 3.25. Overall Rejection during CCD Experiments Up to 97% Recovery.....	40
Figure 3.26. Rejection of Selected PFAAs (PFPrS, PFOS) by the ESPA2 (Left) and NF270 (Right) in PFAA Amended DI Water as a Function of Recovery and Permeate Flux.....	41
Figure 3.27. Rejection of selected PFAAs by the NF270 (left) and ESPA2 (right) during High Recovery Experiments Treating AFFF GW.....	42
Figure 3.28. Rejection of Selected PFCAs (Left) and PFSAs (Right) by the NF270 as a Function of Recovery during CCD Experiments.....	43
Figure 3.29. Rejection of selected PFCAs (left) and PFSAs (right) by the CR100 as a function of recovery during CCD experiments.	43
Figure 3.30. Rejection of Selected PFAAs by the NF270 for No-ionic Strength Addition Experiments (Baseline), Addition of Na ₂ SO ₄ (SS) Experiments, and Addition of CaCl ₂ (CC) experiments.	45
Figure 3.31. Observed Rejection (%) and Feed-retentate Concentration (ng/L) of PFOS and PFBS as a Function of Recovery.....	47
Figure 3.32. Loss of PFAS Mass (Percent Adsorbed) for (a) PFCAs and (b) PFSAs in AFFF Amended Deionized Water Measured in Batch Suspensions of Different Membrane Spiral Wound Element Components.	48
Figure 4.1. Framework for the PFAS Treatment Technology Decision-Support Tool.....	52
Figure 4.2. Preset weighting schemes.	52
Figure 4.3. System Boundary for GAC Gravity Basin Option.	53
Figure 4.4. System Boundary for GAC Pressure Filter Option.....	54
Figure 4.5. System Boundary for IX Filter with Cartridge Prefilter Option.	54
Figure 4.6. System Boundary for IX Filter with Backwashing Option.....	55
Figure 4.7. System Boundary for RO or NF System Options.....	55
Figure 4.8. U.S. Energy Sectors	63
Figure 4.9. Example of Total Capital and O&M for an IX Filter Over Its Lifetime.....	71
Figure 4.10. Tool's OUTPUT Tab Showing Analysis Results in Decision Support Tool.....	72
Figure 4.11. Comparison of Global Warming Impacts	75
Figure 4.12. Comparison of Lifetime Cost.....	76
Figure 4.13. Comparison of Global Warming Impacts	77
Figure 4.14. Comparison of Lifetime Cost.....	78
Figure 4.15. Comparison of Global Warming Impacts	79
Figure 4.16. Comparison of Lifetime Cost for GAC Pressure Filter versus IX Filter with Prefilter	80

LIST OF FIGURES

	Page
Figure 4.17. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)	81
Figure 4.18. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal).....	82
Figure 4.19. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)	83
Figure 4.20. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal).....	84
Figure 4.21. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal).....	85
Figure 4.22. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal).....	86
Figure 4.23. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)	87
Figure 4.24. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal).....	88
Figure 4.25. Breakthrough Curves (Modeled Using the PSDM in AdDesignS) of PFOA, PFHxS, and PFOS	91
Figure 4.26. Comparison of Global Warming Impacts for GAC Pressure Filter and IX Filter with Prefilters at 0.31 mgd for Filter Media Disposal by Hazardous Waste Landfill.....	92
Figure 4.27. Comparison of Global Warming Impacts (Top) and Lifetime Cost (Bottom) for GAC Pressure Filter Versus IX Filter	93
Figure 4.28. O&M- and Lifetime-costs Versus TOC for GAC.....	97
Figure 4.29. O&M- and Lifetime-costs Versus TOC for IX.....	101

LIST OF TABLES

	Page
Table 1.1. Proposed PFAS Regulations.	3
Table 2.1. Summary of Performance Objectives, Data Requirements, and Success Criteria.	4
Table 3.1. Water Characteristics of the Three Groundwater (NC/CO/FL) Used in RSSCTs.	8
Table 3.2. Properties of the Five GACs Selected for RSSCTs.	9
Table 3.3. Properties of the Four Single-use IX Resins Selected for RSSCTs.	9
Table 3.4. Design and Operating Parameters for Constant Diffusivity (CD)-RSSCTs and Corresponding Full-scale GAC Condition.	9
Table 3.5. Design and Operating Parameters for CD-RSSCTs at NCSU and CSM and Corresponding Full-scale IX Condition.	10
Table 3.6. GAC Use Rates by Following Different Treatment Goals.	20
Table 3.7. IX Use Rates by Following Different Treatment Goals.	28
Table 3.8. Set Operating Conditions for CCD Membrane Experiments.	32
Table 3.9. Set Operating Conditions for CCD Membrane Experiments.	34
Table 3.10. Properties of Membranes Employed in the Study.	34
Table 3.11. Dissolved Constituents in DI and Contaminated Groundwater Used in 2540 Pilot-scale Membrane Experiments.	36
Table 3.12. Measured PFAS Concentrations in DI and Contaminated Groundwater Used in 2540 Pilot-scale Membrane Experiments.	36
Table 3.13. Average Feed Water Quality for CCD Experiments.	38
Table 3.14. Measured PFAA Feed Water Concentrations and Limits of Quantification (LOQ) for Sample Types Collected during CCD Experiments.	38
Table 3.15. Feed and Combined Permeate PFAA Concentrations and Calculated Rejection for CCD Experiments Up to 97% Recovery.	41
Table 3.16. Summarized Rejection Values for the Membrane Types Evaluated.	49
Table 4.1. Decision Criteria with Scoring Method and Units.	51
Table 4.2. Percent Rejection by PFAS Compound for RO and NF Membranes.	56
Table 4.3. State Regulations for PFAS in Drinking Water (MCLs) that Are Available as Pre-set Selections in the Tool.	57
Table 4.4. State Notification, Guidance, or Health Advisory Limits for PFAS in Drinking Water that Are Available as Pre-set Selections in the Tool.	57
Table 4.5. State Notification, Guidance, or Health Advisory Limits for PFAS in Groundwater that Are Available as Pre-set Selections in the Tool.	58
Table 4.6. Chosen Coordinates of PFAS Contamination Sites (Hu et al., 2016).	60
Table 4.7. Locations of PFAS Disposal Sites.	61

LIST OF TABLES

	Page
Table 4.8. Midpoint Hauling Distances (in Miles) for Fresh Media, Virgin Membrane, and Disposal	62
Table 4.9. Standard Sizing by Flow Rate for GAC Gravity Basin, GAC Pressure Filter, and IX Filter	66
Table 4.10. Standard Design Assumptions and RO and NF Systems	67
Table 4.11. PVC Piping Weight in kg/ft According to Pipe Diameter	67
Table 4.12. PVC Valve Weights by Pipe Diameter	68
Table 4.13. Instrumentation Included in the Analysis of Each Treatment Option.....	68
Table 4.14. Instrument Weights Used to Calculate LCA Inventory Data.....	69
Table 4.15. Material Densities Used in LCA Calculations	69
Table 4.16 Economic Parameters Used in Life Cycle Costing.....	71
Table 4.17. Life cycle Inventory Unit Processes.....	73
Table 4.18. Summary of the Advantages and Disadvantages of the Removal of PFAS.....	89
Table 4.19. Influent PFAS Concentrations and TOC in the Four Waters Used for Case Studies	90
Table 4.20. Calculated Media Usage Rate (lb/1000 gal) Based on Regulation/Treatment Objective Using Modeled Breakthrough Curves	92
Table 4.21. Investigated GW with Different TOC Levels	94
Table 4.22. Investigated Drinking Water Regulation/Health Advisory Level/ Notification Levels.....	94
Table 4.23. Assigned Weighting Schemes for Cost Comparison of GAC and IX.....	95
Table 4.24. Media Use Rates for GAC (F400) Determined through RSSCTs.....	95
Table 4.25. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GWs: 2016 USEPA HAL.	96
Table 4.26. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: New Jersey MCL.	96
Table 4.27. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: Michigan State MCL.	96
Table 4.28. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: Vermont MCL.	96
Table 4.29. Bed Lifetime (Months), Capital- O&M- and lifetime Costs of GAC for the Treatment of Three GW Sources: California Notification Level.	97
Table 4.30. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: EPA (2023) MCL.....	97
Table 4.31. Average PFAS Concentrations in Fountain GW Used for Pilot Test.	98
Table 4.32. GAC Use Rates of Pilot Versus RSSCT with CO-GW as Feed Water.....	98
Table 4.33. Evaluated GAC Product Use Rates When Treating CO-GW	99

LIST OF TABLES

	Page
Table 4.34. Evaluated GAC Product Costs When Treating CO-GW.....	99
Table 4.35. IX-use Rates to Meet Different Regulations During Treatment of CO and FL-GW	100
Table 4.36. IX Costs for Treatment of CO-GW	100
Table 4.37. IX Costs for Treatment of FL-GW	101
Table 4.38. Cost-comparison of IX (AER), GAC, and RO Membrane for the Treatment of CO-GW without Thermal Reactivation of GAC	102
Table 4.39. Cost-comparison of IX (AER), GAC, and RO Membrane for the Treatment of CO-GW with Thermal Reactivation of GAC	102
Table 4.40. Comparison of the Costs for an RO Membrane System with Two Different Concentrate Disposal Option.....	102
Table 5.1. Advantages and Disadvantages of PFAS Treatment Technologies Evaluated in this Project Based on Findings, Life Cycle Impact, and Cost Modeling	105

ACRONYMS AND ABBREVIATIONS

AER	Anion Exchange Resin
AFFF	Aqueous Film-Forming Foam
AOP	Advanced Oxidation Process
BET	Brunauer–Emmett–Teller
BV	Bed Volume
BV10	Bed Volume To 10% Breakthrough
BV50	Bed Volume To 50% Breakthrough
CCD	Closed-Circuit Desalination
CD	Constant Diffusivity
CMC	Critical Micelle Concentration
DI	Deionized
DOC	Dissolved Organic Carbon
EBCT	Empty Bed Contact Time
EPA	U.S. Environmental Protection Agency
GAC	Granular Activated Carbon
gpm	Gallons Per Minute
gfd	Gallons Per Square Foot Per Day
GW	Groundwater
HAL	Health Advisory Level
HLR	Hydraulic Loading Rate
IC	Ion Chromatography
ISO	International Organization for Standardization
IX	Ion-Exchange
LCA	Life Cycle Assessment
LCC	Life Cycle Cost
LCI	Life Cycle Inventory
LMH	Liters Per Square Meter Per Hour
LOQ	Limit Of Quantitation
LPM	Liters Per Minute
MCDA	Multi-Criteria Decision Analysis
MCL	Maximum Contaminant Level
MCLG	Maximum Contaminant Level Goal
MGD	Million Gallons Per Day
MWCO	Molecular Weight Cut-Off

NF	Nanofiltration
NTIF	Normalized Total Intraparticle Flux
O&M	Operation And Maintenance
PFAA	Perfluoroalkyl Acid
PFAS	Per- And Polyfluoroalkyl Substances
PFBA	Perfluorobutanoic Acid
PFBS	Perfluorobutane Sulfonic Acid
PFCA	Perfluoroalkyl Carboxylic Acid
PFDA	Perfluorodecanoic Acid
PFNA	Perfluorononanoic Acid
PFOA	Perfluorooctanoic Acid
PFOS	Perfluorooctane Sulfonate
PFSA	Perfluoroalkyl Sulfonic Acid
POTW	Publicly Owned Treatment Works
PSDM	Pore And Surface Diffusion Model
QSPR	Quantitative Structure-Property Relationship
RCRA	Resource Conservation And Recovery Act
RO	Reverse Osmosis
RSSCT	Rapid Small-Scale Column Test
SDI	Silt Density Index
SPDFR	Surface-To-Pore Diffusion Flux Ratio
TDS	Total Dissolved Solids
TOC	Total Organic Carbon

ACKNOWLEDGEMENTS

This project was successfully completed as a direct result of the critical contributions of many people from multiple organizations. Specific acknowledgment is extended to the following:

- Colorado School of Mines: Christopher Higgins, Conner Murray, Andrew Safulko, and Bahareh Tajdini
- North Carolina State University: Ping Ping Meng and Lan Cheng
- University of Colorado: Cameron Oden
- CDM Smith: Charles Schaefer
- Utility and industry participants who attended the stakeholder workshops and provided input to the decision support tool
- WRF's Project Advisory Committee – Robert Raczko (retired), Thomas Speth (USEPA), and Chris Tadanier (Black and Veatch)

Thank you all for your efforts and support.

ABSTRACT

INTRODUCTION AND OBJECTIVES

Per- and polyfluoroalkyl substances (PFAS) are a group of man-made chemicals that have been widely used since the 1950s in a variety of industrial and consumer applications due to their unique chemical properties. PFAS are surfactants used as active ingredients in stain repellents (e.g., Scotchgard) and firefighting foams. PFAS are of concern because of their toxicity and persistence in the environment and the human body. Perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) have been detected globally in humans and wildlife. Conventional water treatment processes such as flocculation, sedimentation, granular media filtration, and chlorine disinfection are ineffective for removing PFAS. Past research has also demonstrated that common advanced oxidation processes, such as ultraviolet light or ozone with peroxide, may not be as effective for perfluoroalkyl acid (PFAA) degradation and may transform polyfluorinated precursors to PFAAs. As a result, researchers and practitioners have focused on the use of adsorbents such as granular activated carbon (GAC) and ion exchange (IX) and, to a lesser extent, nanofiltration (NF) or reverse osmosis (RO) for the treatment of PFAS-contaminated water.

The principal objective of this research was to assess the effectiveness of GAC adsorption, IX, and membrane processes for the removal of PFAS from different source waters and quantify the benefits and dis-benefits (treatment capacity, life cycle costs, etc.) for each technology as either stand-alone processes or integrated treatment trains. Specific objectives were to:

- Develop a comprehensive assessment framework for ex-situ PFAS treatment technologies,
- Generate a PFAS treatment efficiency database to support development of a decision support tool.
- Develop a treatment technology decision support tool.

These objectives were achieved through systematic investigation of PFAS removal under various treatment scenarios and development of a life cycle impact and costing model to quantify and compare the technologies for different treatment scenarios.

RESULTS

All three technologies (GAC, IX, and high-pressure membranes) studied in this research showed great ability to remove PFAS from different source waters. However, the removal efficiency was impacted by various factors.

Overall, total organic carbon (TOC) concentration had the most dramatic effect on GAC use rate. In groundwater with a TOC concentration of 1.3 mg/L, the GAC use rate was 10 times that in groundwater with a TOC concentration of <0.3 mg/L. Empty bed contact time (EBCT) and GAC type were also important parameters that affected GAC use rates. Doubling EBCT from 10 minutes to 20 minutes decreased GAC use rates by a factor of about two for the tested GAC and water matrix combination. Out of four major inorganic anions investigated (chloride, bicarbonate, sulfate, and nitrate), nitrate had the most pronounced impact, negatively impacting the adsorption of short-chain PFAS. Fresh and weathered aqueous film-forming foam (AFFF) constituents had a negligible impact on the removal of targeted PFAS. GAC use rates were also strongly impacted by different regulatory scenarios; both lower concentration targets and inclusion of shorter-chain PFAS into treatment criteria led to substantial increases in GAC use rates.

Initial PFAS concentration and competition among co-occurring PFAS had a negligible effect on PFAS breakthrough curves for the selected IX resin. Doubling EBCT from 1.5 minutes to 3 minutes had a negligible impact on PFAS removal. TOC concentration negatively impacted IX performance, but not as strongly as GAC performance. Long-chain PFAS were more impacted by TOC than short-chain PFAS. Out of four major ions investigated (chloride, bicarbonate, sulfate, and nitrate), nitrate had the most pronounced impact, negatively impacting PFAS removal by IX. Short-chain PFAS were more strongly affected by nitrate than long-chain PFAS. Similar to GAC, IX use rates were strongly impacted by different regulatory scenarios; both lower concentration targets and inclusion of shorter-chain PFAS into treatment criteria led to substantial increases in IX use rates.

For the PFAAs investigated and over the range of operating conditions evaluated, tight NF and RO membranes provided higher separation or rejection than the loose NF membrane. For a DI water matrix, RO membrane provided greater than 99% rejection for all PFAAs, with permeate concentrations being near or below the limit of quantification. For waters with higher ionic strength, loose NF and to a lesser extent tight-NF and RO exhibited lower PFAA separation. In particular, PFAA separation by loose NF was observed to be significantly and detrimentally impacted by the presence of salts and subsequently high product water recoveries. The addition of sodium sulfate negatively impacted the rejection of perfluorobutane sulfonic acid (PFBS), PFOA, and PFOS by loose NF. Compared to the addition of sodium sulfate, calcium chloride had a minimal impact on the rejection of PFAAs with separation being marginally (5-10%) lower than baseline conditions across the recovery setpoints evaluated.

As part of the scope, an Excel[®] based decision support tool was developed. The tool provides a pathway to comprehensively evaluate PFAS treatment technology options consisting of GAC, IX, and high-pressure membrane systems for groundwater remediation. Treatability data generated during this project was used to evaluate the impact of various factors on costs associated with PFAS treatment using GAC, IX or high-pressure membranes. For GAC and IX treatment, the highest treatment costs were associated with treating high TOC groundwater to the proposed USEPA maximum contaminant level (MCL) for PFOA. High-pressure membrane treatment was found to be significantly more expensive than GAC and IX due to higher capital and operation and maintenance (O&M) costs, with concentrate management costs being substantial depending on the disposal option.

PUBLICATIONS

Murray, C.C., R. E. Marshall, C.J. Liu, H. Vatankhah and C. Bellona, PFAS treatment with granular activated carbon and ion exchange resin: Comparing chain length, empty bed contact time, and cost, *Journal of Water Process Engineering* 44 (2021).

PRESENTATIONS

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EXECUTIVE SUMMARY

Per- and polyfluoroalkyl substances (PFAS) are a group of man-made chemicals that have been widely used since the 1950s in a variety of industrial and consumer applications due to their unique chemical properties. PFAS are surfactants used as active ingredients in stain repellents (e.g., Scotchgard) and firefighting foams. They are also used in the manufacture of such materials as non-stick coatings, water-repellent fabrics, and microwave popcorn bags. PFAS are released into the environment directly by fluorochemical production facilities and indirectly through consumer use of products. PFAS are of concern because of their toxicity and persistence in the environment and the human body. Perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) have been detected globally in humans and wildlife.

Conventional water treatment processes such as coagulation, flocculation, sedimentation, granular media filtration, and chlorine disinfection are ineffective for removing PFAS. Past research has also demonstrated that common advanced oxidation processes, such as ultraviolet light or ozone with peroxide, may not be as effective for perfluoroalkyl acid (PFAA) degradation and may transform polyfluorinated precursors to PFAAs. As a result, researchers and practitioners have focused on the use of adsorbents such as granular activated carbon (GAC) and ion exchange (IX) and, to a lesser extent, nanofiltration (NF) or reverse osmosis (RO) for the treatment of PFAS-contaminated water. However, various other promising technologies for treatment of PFAS have been developed and evaluated, including novel adsorbents (e.g., biochar, metal organic frameworks, surface modified clays, zeolite), hybrid adsorption-filtration systems (e.g., functionalized electrodialysis, sub-micron powdered activated carbon with microfiltration), and destructive technologies (e.g., electrochemical oxidation, hydrothermal alkaline treatment, ultraviolet light based reductive processes, supercritical water oxidation).

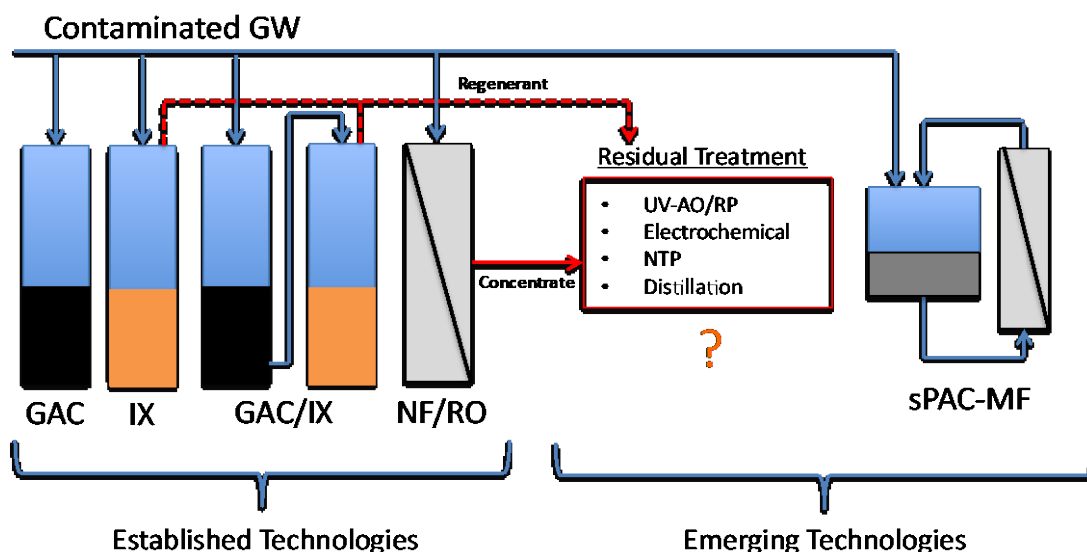
The concept of this project was conceived circa 2016 when PFAS contamination was beginning to gain attention in the scientific literature and the news, but there was limited understanding of effective treatment options. During this time, there were a significant number of publications and press on a variety of potential treatment approaches with little guidance on the implementability, maturity, effectiveness, and costs associated with many proposed PFAS treatment technologies. A primary goal of this project was to evaluate the most promising of these approaches through bench- and pilot-scale testing with the goal of developing a decision support tool to compare treatment approaches based on lifecycle costs and environmental impacts. While a significant number of treatment technologies have been proposed as effective PFAS solutions since the inception of this project, at the time of writing this report, many have proven to be ineffective for treatment and/or are exceedingly costly. Additionally, a number of promising destructive technologies are still in the development stage rendering development of costs and lifecycle impacts difficult.

OBJECTIVES

The principal objective of this research was to assess the effectiveness of GAC adsorption, IX, and membrane processes for the removal of PFAS from different source waters and quantify the benefits and dis-benefits (treatment capacity, life cycle costs, etc.) for each technology as either stand-alone processes or integrated treatment trains. Individual PFAS treatment processes we evaluated are summarized in Figure ES 1. Specific objectives were to:

- Develop a comprehensive assessment framework for ex-situ PFAS treatment technologies,
- Generate a PFAS treatment efficiency database to support development of a decision support tool.
- Develop a treatment technology decision support tool.

These objectives were achieved through systematic investigation of PFAS removal under various treatment scenarios and development of a life cycle impact and costing model to quantify and compare the technologies for different treatment scenarios.



* NTP is non-thermal plasma, UV-AO/RP is ultraviolet light advanced oxidation/reduction process

Figure ES-1. Schematic of PFAS Treatment Technologies Evaluated.

RESULTS

Granular Activated Carbon

Factors determining GAC use rates were assessed with rapid small-scale column tests (RSSCTs). A scale-up approach developed in a parallel research effort was used to predict GAC use rates for full-scale treatment systems from RSSCT data. The scale-up approach was based on the finding that intraparticle diffusivity is proportional to $d_p^{0.25}$, where d_p is the GAC particle diameter. The following factors were investigated to examine the PFAS removal by GAC:

- Effect of initial PFAS concentration and competition among co-occurring PFAS
- Effect of empty bed contact time (EBCT)
- Effect of GAC type
- Effects of total organic carbon (TOC)
- Effects of inorganic anions
- Effect of fresh and weathered aqueous film-forming foam (AFFF)
- Regulatory scenarios

Overall, total organic carbon (TOC) concentration had the most dramatic effect on GAC use rate. In groundwater with a TOC concentration of 1.3 mg/L, the GAC use rate was 10 times that in groundwater with a TOC concentration of <0.3 mg/L. Compared to groundwater with a TOC concentration of 1.3 mg/L, GAC use rates in groundwater with a TOC concentration of 4.3 mg/L increased another 10-fold. EBCT and GAC type were also important parameters that affected GAC use rates. Doubling EBCT from 10 minutes to 20 minutes decreased GAC use rates by a factor of about two for the tested GAC and water matrix combination. Out of four major inorganic anions investigated (chloride, bicarbonate, sulfate, and nitrate), nitrate had the most pronounced impact, negatively impacting the adsorption of short-chain PFAS. Fresh and weathered AFFF constituents had a negligible impact on the removal of targeted PFAS. GAC use rates were also strongly impacted by different regulatory scenarios; both lower concentration targets and inclusion of shorter-chain PFAS into treatment criteria led to substantial increases in GAC use rates.

Ion Exchange

Factors determining IX use rates were assessed with RSSCTs that were designed based on the assumption that intraparticle diffusivity is independent of IX particle diameter. A scale-up approach developed in a parallel research effort was used to predict IX use rates for full-scale treatment systems from RSSCT data. The scale-up approach was based on the finding that external film diffusion affected PFAS uptake capacities. To scale up RSSCT data, bed volumes treated in the RSSCT were multiplied by $(Sh_{pilot}/Sh_{RSSCT})^{0.5}$, where Sh is the Sherwood number. The following factors were investigated to examine the PFAS removal by IX:

- Effects of initial PFAS concentration and competition between co-existing PFAS
- Effect of EBCT
- Effect of IX type
- Effect of TOC
- Effect of inorganic anions

Results suggest that initial PFAS concentration and competition among co-occurring PFAS had a negligible effect on PFAS breakthrough curves for the selected IX resin. Doubling EBCT from 1.5 minutes to 3 minutes had a negligible impact on PFAS removal. TOC concentration negatively impacted IX performance, but not as strongly as GAC performance. Long-chain PFAS were more impacted by TOC than short-chain PFAS. Out of four major ions investigated (chloride, bicarbonate, sulfate, and nitrate), nitrate had the most pronounced impact, negatively impacting PFAS removal by IX. Short-chain PFAS were more strongly affected by nitrate than long-chain PFAS. Similar to GAC, IX use rates were strongly impacted by different regulatory scenarios; both lower concentration targets and inclusion of shorter-chain PFAS into treatment criteria led to substantial increases in IX use rates.

High-Pressure Membranes

Five different membranes were evaluated for the removal of PFAS from several source waters of differing quality. These membranes represent a spectrum of membrane classifications spanning loose NF, tight NF, low-pressure RO, and seawater RO. In addition to membrane classification, the impact of source water quality and operating conditions including permeate flux and product water recovery on the rejection of PFAAs was evaluated. The following sections detail the major findings related to the impact of membrane type and membrane system operating conditions.

Low-pressure NF ('loose NF') represents an attractive option for PFAA separation compared to RO given its low energy requirements and partial separation of salts. However, for the PFAAs investigated and over the range of operating conditions evaluated, tight NF and RO membranes provided higher separation or rejection than the loose NF membrane. For a DI water matrix, RO membrane provided greater than 99% rejection for all PFAAs, with permeate concentrations being near or below the limit of quantification. For waters with higher ionic strength, loose NF and to a lesser extent tight-NF and RO exhibited lower PFAA separation. In particular, PFAA separation by loose NF was observed to be significantly and detrimentally impacted by the presence of salts and subsequently high product water recoveries.

The impact of ionic strength of PFAA separation for NF membranes was evaluated by addition of CaCl_2 and Na_2SO_4 to dechlorinated tap water amended with AFFF. The addition of ionic strength is expected to increase PFAA flux by reducing the time require to sufficiently suppress the repulsive electrostatic forces between the negatively charged membrane surface and the PFAA anions. Results indicate that the addition of sodium sulfate negatively impacted the rejection of PFBS, PFOA, and PFOS by loose NF. Compared to the addition of sodium sulfate, calcium chloride had a minimal impact on the rejection of PFAAs, with separation being marginally (5-10%) lower than baseline conditions across the recovery setpoints evaluated.

A novel membrane system allowing high product water recoveries termed closed-circuit desalination was evaluated during this project. This system allowed for evaluation of the impact of system operating conditions, PFAA concentration, and ionic strength on PFAA separation. Evaluation of four membrane products spanning loose NF, tight NF, low-pressure RO, and seawater RO demonstrated that 2-log and greater PFAA separation is achievable over system conditions up to 97% recovery for tight NF and RO membranes evaluated.

Decision Support Tool

As part of the scope, an Excel[®] based decision support tool was developed. The tool provides a pathway to comprehensively evaluate PFAS treatment technology options consisting of GAC, IX, and high-pressure membrane systems for groundwater remediation. The tool quantifies each option's relative environmental impacts, costs, and facility requirements for a given scenario based on the assigned scoring. Each scenario is defined by a specific water quality (PFAS concentrations, TOC, pH) and quantity (0.03 to 75 million gallons per day); the functional unit, following ISO 14040 life cycle assessment (LCA) methodology to ensure comprehensive and accurate comparison, is the treatment of that water over a 40-year lifetime (Figure ES2).

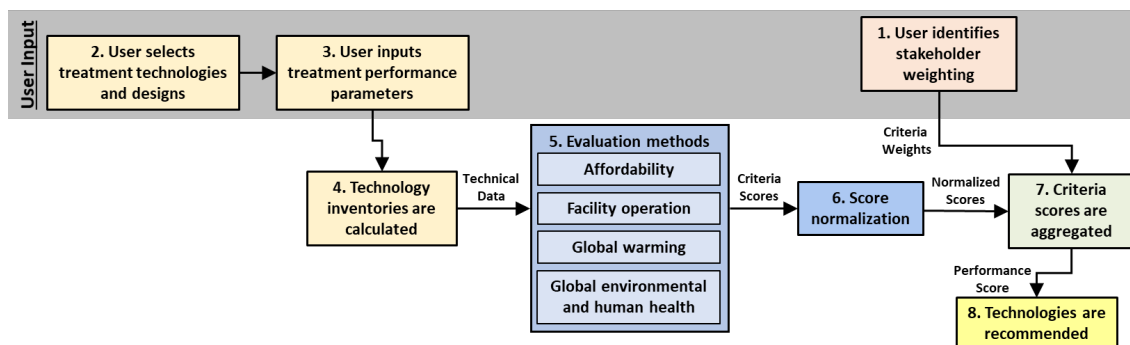


Figure ES2. Framework for the PFAS Treatment Technology Decision-Support Tool.

The tool's functionality was demonstrated using scenario analysis, case studies, and treatability data generated during the project. Through hypothetical scenario analyses comparing GAC, IX, and high-pressure membrane processes was determined that:

- Media usage rates of GAC and IX strongly impact which technology is more cost-effective and environmentally sustainable (e.g., global warming impacts);
- Lower media usage rates associated with IX compared to GAC make IX a more environmentally sustainable PFAS treatment technology based on global warming impacts;
- GAC typically has lower project lifecycle costs than IX when media usage rates are similar between the two processes;
- RO and NF have significantly higher project lifecycle costs and environmental impacts compared to GAC and IX;
- GAC and IX media disposal options evaluated do not significantly favor one process over another; and
- Thermal reactivation of GAC results in GAC becoming more favorable compared to IX for certain scenarios.

Treatability data generated during this project was used to evaluate the impact of various factors on costs associated with PFAS treatment using GAC, IX, or high-pressure membranes. For GAC and IX, treatment goals based on federal and state regulations were found to significantly impact project lifecycle costs, with a critical component being meeting PFOA regulations. Due to the negative impact of TOC on GAC and IX media usage rates, treatment of regulated PFAAs in source waters containing elevated TOC is challenging from a cost perspective. For GAC and IX treatment, the highest treatment costs were associated with treating high TOC groundwater to the proposed USEPA MCL for PFOA. High-pressure membrane treatment was found to be significantly more expensive than GAC and IX due to higher capital and O&M costs, with concentrate management costs being substantial depending on the disposal option.

1.0 INTRODUCTION AND OBJECTIVES

1.1 INTRODUCTION

Per- and polyfluoroalkyl substances (PFAS) are a group of man-made chemicals that have been widely used since the 1950s in a variety of industrial and consumer applications due to their unique chemical properties. PFAS are surfactants used as active ingredients in stain repellents (e.g., Scotchgard) and firefighting foams. They are also used in the manufacture of such materials as non-stick coatings (e.g., Teflon), water-repellent fabrics (e.g., GoreTex), and microwave popcorn bags. PFAS are released into the environment directly by fluorochemical production facilities and indirectly through consumer use of products. PFAS are of concern because of their toxicity and persistence in the environment and the human body. Perfluorooctanoic acid (PFOA) and perfluorooctane sulfonate (PFOS) have been detected globally in humans and wildlife. In addition, PFAS have been measured in human blood serum collected from the general population. Once present in human whole blood or serum, PFOA and PFOS have half-lives of 3.8 and 5.4 years, respectively (Betts 2007).

Conventional water treatment processes such as flocculation, sedimentation, granular media filtration, and chlorine disinfection are ineffective for removing PFAS (Quiñones and Snyder 2009; Xiao et al. 2013; Appleman et al. 2014). Past research has also demonstrated that common advanced oxidation processes, such as ultraviolet light or ozone with peroxide, may not be as effective for perfluoroalkyl acid (PFAA) degradation and may transform polyfluorinated precursors to PFAAs (Houtz et al. 2013; Appleman et al. 2014; Vecitis et al. 2009). While some success has been reported for PFAA degradation using alternative processes (such as sonolysis (or ultrasound), activated persulfate, and electrolysis), there are questions regarding the chemical and/or energy additions required for decomposition reactions to proceed, the implementability of these technologies, generation of secondary products, and their applicability to degrade a variety of PFAS (Vecitis et al. 2009; Liu et al. 2012; Mitchell et al. 2014; Schaefer et al. 2015). As a result, researchers and practitioners have focused on the use of adsorbents such as granular activated carbon (GAC) and ion exchange (IX) and, to a lesser extent, nanofiltration (NF) or reverse osmosis (RO) for the treatment of PFAS-contaminated water (Appleman et al. 2014; Yan et al. 2015; Dudley et al. 2015, Sun et al. 2016). However, relatively short breakthrough times have been reported for activated carbon for shorter-chain PFAAs, and both processes produce a residual requiring disposal or further treatment (Appleman et al. 2014).

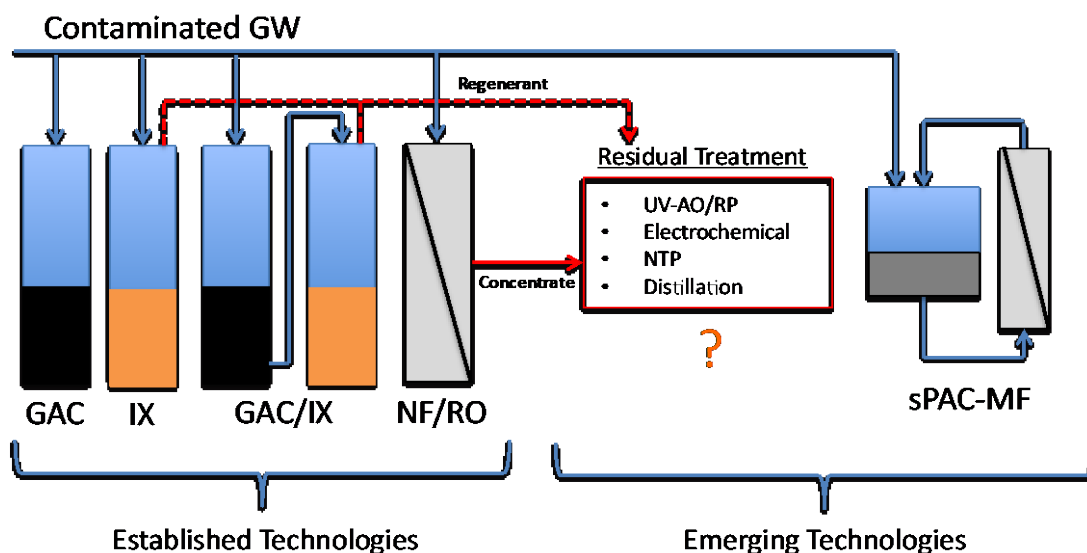
Since there is no ‘magic bullet’ treatment approach, there is a significant need to quantify the benefits and liabilities of the possible treatment approaches capable of protecting human health by limiting PFAS exposure through drinking water.

1.2 OBJECTIVES

The principal objective of this research was to assess the effectiveness of GAC adsorption, IX, and membrane processes for the removal of PFAS from different source waters and quantify the benefits and dis-benefits (treatment capacity, life cycle costs, etc.) for each technology as either stand-alone processes or integrated treatment trains. Individual PFAS treatment processes we evaluated are summarized in Figure 1.1. Specific objectives were to:

- Develop a comprehensive assessment framework for ex-situ PFAS treatment technologies,
- Generate a PFAS treatment efficiency database to support development of a decision support tool.
- Develop a treatment technology decision support tool.

These objectives were achieved through systematic investigation of PFAS removal under various treatment scenarios and development of a life cycle impact and costing model to quantify and compare the technologies for different treatment scenarios.



* NTP is non-thermal plasma, UV-AO/RP is ultraviolet light advanced oxidation/reduction process

Figure 1.1. Schematic of PFAS Treatment Technologies Evaluated.

1.3 REGULATORY DRIVERS

Due to the concerns about the potential health and environmental risks associated with PFAS, there has been increasing attention to regulating and managing these substances. In 2019, the U.S. Environmental Protection Agency (EPA) announced a comprehensive plan to address PFAS, including the development of enforceable drinking water standards for PFAS and listing certain PFAS as hazardous substances under the Superfund law, and the promotion of research into alternatives to PFAS.

In March 2023, the U.S. Environmental Protection Agency (EPA) announced its new proposed National Primary Drinking Water Regulations for six PFAS. The proposed rule would establish both enforceable Maximum Contaminant Levels (MCLs) and non-enforceable Maximum Contaminant Level Goals (MCLGs) for the six types of PFAS, including PFOA, PFOS, perfluorononanoic acid (PFNA), hexafluoropropylene oxide dimer acid (HFPO-DA, more commonly known as GenX), perfluorohexane sulfonic acid (PFHxS) and perfluorobutane sulfonic acid (PFBS). The proposed rule would set MCLs and MCLGs at the levels shown in Table 1.1.

Table 1.1. Proposed PFAS Regulations.

Type of PFAS	Proposed MCLG	Proposed MCL (enforceable levels)
PFOA	Zero	4 parts per trillion (ppt)
PFOS	Zero	4 ppt
PFNA	1.0 (unitless) Hazard Index	1.0 (unitless) Hazard Index
PFHxS		
PFBS		
HFPO-DA		

While the proposed rule sets 4 ppt MCLs for PFOA and PFOS, the mixture of four PFAS will be assessed using a tool known as a Hazard Index, which evaluates health risks of simultaneous exposure to mixtures of related chemicals. Water systems would monitor and compare the amount of each PFAS in drinking water to its associated Health-Based Water Concentration, which is the level at which no health effects are expected for that PFAS. The proposed rule would require public water systems to:

- Monitor for these PFAS
- Notify the public of the levels of these PFAS
- Reduce the levels of these PFAS in drinking water if they exceed the proposed standards.

In the proposed rule document, EPA identified GAC, IX, and high-pressure membranes (RO and NF) as the best available technologies for PFAS removal. These are the same technologies studied in this research project. The information generated by this research can help to understand the advantages and disadvantages of these technologies to remove PFAS.

2.0 PERFORMANCE OBJECTIVES

To address the three objectives established in Section 1.2, the project team collected data from previously conducted studies and performed laboratory-based experimentation to fill in data gaps related to PFAS removal by a variety of competing technologies including granular activated carbon (GAC), ion exchange (IX), nanofiltration (NF) and reverse osmosis (RO). Along with technology information, this data was used to develop the life cycle assessment (LCA) and life cycle cost (LCC) and, ultimately, a decision support tool for the DOD to aid in treatment technology selection. Quantitative and qualitative performance objectives are summarized in Table 2.1.

Table 2.1. Summary of Performance Objectives, Data Requirements, and Success Criteria.

Performance Objective	Data Requirements	Success Criteria	Success Criteria Achieved
Quantitative Performance Objectives			
Collect representative data (all technologies) from previous studies and literature	• Analysis QA/QC methodologies • Pre- and post-treatment PFAS concentrations • Background water matrix • Operating conditions • Technology specifications	• Compile minimum data to aid in LCA/LCC and framework development • Data gaps are determined	• Significant treatability data generated/compiled and used for LCA/LCC development and evaluation (Chapter 4) • Identified data gaps are presented in Chapter 5
Develop assessment framework	• Experimental operating conditions • Pre- and post-treatment PFAS concentrations in contaminated groundwater and surface water • Final list of PFAS to include	• Guidance document on PFAS data collection and system operation during testing • Successful application of assessment framework to lab-scale experiments	• Executed experimental protocol resulted in necessary data for comprehensive LCA/LCC investigation of investigated PFAS treatment technologies (Chapter 4) • Data generated during the project was input to the decision support tool to investigate costs associated with different treatment scenarios, treatment technologies, and treatment goals. Chapter 4.7 shows successful application to pilot-scale data and Chapter 4.8 shows successful application to treatability (lab-scale) experiments
Assess technology effectiveness at lab-scale (adsorbents)	• Pre- and post-treatment PFAS concentrations in contaminated groundwater and surface water • Adsorbent properties	• Reach breakthrough (>10%) for all relevant PFAS to provide sufficient data for LCA/LCC	• 10% breakthrough was achieved for all PFAS and adsorbent media evaluated and data were sufficient for performance metric calculation including media usage rates and specific costs for meeting relevant treatment goals (e.g., PFAS health advisory levels, maximum contaminant limits)
Assess NF and RO effectiveness at lab-scale	• PFAS rejection over range of operating conditions • Operating data (pressure, flux, recovery)	• >95% rejection of PFAAs and ionic PFAS • >90% of neutral PFAS • Minimal membrane fouling	• Achieved for anionic PFAS (Chapter 4) • Neutral PFAS not identified in source waters • Membrane fouling not observed through performance monitoring

**Table 2.1. Summary of Performance Objectives, Data Requirements, and Success Criteria.
(Continued)**

Performance Objective	Data Requirements	Success Criteria	Success Criteria Achieved
Qualitative Performance Objectives			
User friendly decision support tool	Feedback from stakeholders through workshops and meetings	- Stakeholder test runs successfully demonstrate ease of use during workshops - Project team members successfully demonstrate ease of use during toolbox development and testing	The decision support tool was presented to water professionals during two 2-hour virtual workshops in February 2022. During this workshop, the tool methodology was described, attendees were walked through a visual tutorial, and a demonstration of the tool was performed live for a selected scenario. Also, stakeholders provided feedback on the tool throughout the workshop, especially at the end during break-out (small group) discussions. Many found the tool helpful and usable and provided helpful comments for potential improvements. Feedback from these workshops was used to update the tool in several ways. For example, cartridge pre-filters were added as an option for fouling prevention in IX filters. The ability for the user to select if they want to use cartridge filters or backwashing during the operation of the IX filter system was added. The user input to calculate bed lifetime was changed to media usage rate in lb/1000 gal. Thermal reactivation was added as a disposal option for GAC processes

3.0 RESULTS AND DISCUSSION

3.1 GRANULAR ACTIVATED CARBON ADSORPTION AND ION EXCHANGE

3.1.1 Background

Granular activated carbon (GAC) adsorption and anion exchange (IX) are two promising treatment options for removing PFAS from water. In remediation applications, contaminated groundwater is pumped through pressure vessels containing either GAC or IX resin in a packed bed configuration. PFAS uptake kinetics determine the length of the mass transfer zone in the adsorbent bed. If the adsorbent bed is longer than the mass transfer zone of a given contaminant, the contaminant is essentially completely removed. As time progresses and the uptake capacity of the sorbent at the top of the bed is exhausted, the mass transfer zone moves downward through the bed. At some point in time, the mass transfer zone is no longer fully contained in the bed, and the contaminant begins to break through, meaning detectable contaminant levels appear in the effluent of the packed bed and increase over time. Once the effluent from the packed bed reaches the treatment criterion, the GAC or IX resin in the vessel needs to be replaced. Based on the total volume of water treated between sorbent changeouts and the mass of sorbent contained in the pressure vessel, a GAC or IX resin use rate can be calculated (e.g., pounds of sorbent per 1000 gallons of water treated), which is a key determinant of operation and maintenance (O&M) costs (i.e., higher sorbent use rates translate into higher O&M costs). Sorbent use rates depend on the characteristics of the sorbent, PFAS properties, and background water matrix constituents. In addition, O&M costs depend on the unit cost of the sorbent. On a per-pound basis, IX resins are substantially more costly than GAC, meaning that IX resin use rates have to be lower by an equivalent factor before they become cost competitive. Apart from costs associated with the removal of anionic perfluoroalkyl acids (PFAAs), other factors may influence the choice between GAC and IX processes. For example, GAC may be selected over IX if contaminants other than anionic PFAS (e.g., neutral/cationic PFAS, fuel hydrocarbons) need to be removed.

- **Granular Activated Carbon.** GAC contactors are widely used for the removal of PFAS from contaminated groundwater. While effective for the removal of long-chain PFAS, relatively early breakthrough has been observed for short-chain PFAAs (Liu et al. 2019, Rodowa et al. 2021, Kempisty et al. 2022). Apart from the length of the perfluorinated alkyl chain, the head group and branching affect PFAS adsorbability. Perfluoroalkyl sulfonic acids (PFSAAs) with a given number of fluorinated carbon atoms are more effectively removed than the corresponding perfluoroalkyl carboxylic acids (PFCAs); i.e., perfluorooctane sulfonate (PFOS) is more effectively removed than perfluorononanoate (PFNA) even though both compounds contain 8 fully fluorinated carbon atoms in their alkyl chain (e.g., Kempisty et al. 2022). Furthermore, branched isomers of compounds such as PFOS are less adsorbable than their linear analogs (e.g., Rodowa et al. 2021, Chow et al. 2022). GAC selection is also of critical importance. Generally, GACs with a larger volume of transport pores (Liu et al. 2019) and positive pore surface charge (Park et al. 2020, Cantoni et al. 2021) appear to be more effective for PFAS removal, with greater benefits observed for longer-chain PFAS. GACs with a large micropore surface area show advantages for the removal of short-chain PFAS (Park et al. 2020). From a design perspective, the empty bed contact time (EBCT) of an adsorber is important, but the effects of EBCT on GAC use rates for PFAS removal are poorly understood. Decreasing GAC use rates with increasing EBCT have been observed in some pilot studies that evaluated PFAS removal (e.g., CFPUA, 2018, Chow et al., 2022).

However, in other GAC data sets, the effect of increasing EBCT from ~10 to ~20 min was negligible (e.g., Gonzalez et al. 2021), inconclusive (e.g., Kempisty et al., 2022), or negative (Summers et al., 2022). In rapid small-scale column tests (RSSCTs), Park et al. (2020) observed that steeper breakthrough curves were obtained at longer EBCTs while adsorption capacity was similar, suggesting that a longer EBCT is beneficial if the treatment criterion is reached near the onset of breakthrough (e.g., 10% breakthrough) while EBCT may not impact GAC use rates if the treatment criterion allows operation of the GAC adsorber to a higher level of breakthrough (e.g., 50% breakthrough). Several other important factors controlling GAC use rates are also poorly understood. These include the effects of background water matrix constituents (e.g., dissolved organic matter, inorganic anions), initial PFAS concentration and co-occurring PFAS, and the effects of treatment criteria.

- **Ion Exchange.** Although used less extensively than GAC, IX can be an effective alternative to GAC for PFAS remediation. When remediating relatively dilute PFAS plumes, single-use IX resins are often considered. Once the treatment criterion is reached, the spent single-use resin is replaced with fresh resin, and the spent resin is either incinerated or landfilled. In some situations, regenerable IX resins are preferred. In this case, exhausted IX resin can be regenerated onsite with brine/methanol mixtures; however, additional treatment of regenerant is required (e.g., distillation to recover methanol followed by electrochemical treatment of still bottoms, Fang et al. 2023) in the absence of viable disposal options. Past research has indicated that performance of GAC and IX was similar for removal of short and long-chain PFAAs (Appleman et al. 2014). However, later studies with different IX resins demonstrated that IX may be cost-competitive with GAC for PFAA removal, both for shorter-chain PFAAs (Zaggia et al. 2016) and longer-chain PFAAs, such as PFOA and PFOS (Woodard et al. 2017). From a resin use rate perspective, PFAS characteristics, background water matrix constituents, IX resin properties, and IX resin bed design and operating conditions are all expected to be important. With respect to PFAS properties, PFAS chain length, head group, and branching all affect IX use rates (e.g., Liu et al. 2022, Ellis et al. 2022, Chow et al. 2022). Similar to GAC, the uptake capacity of IX increases as the length of the perfluoroalkyl chain increases within a PFAS subclass. Also, IX resins have a higher uptake capacity for PFSA compared to PFCA of a given chain length, and the difference between PFSA and PFCA uptake capacities is much more pronounced for IX resins than for GAC. For GAC, PFSA behave like PFCA with 1-2 more $-CF_2-$ units, while for IX, PFSA behave like PFCA with 4-5 more $-CF_2-$ units (Chow et al. 2022). Branching also adversely affects PFAS uptake by IX, but it is less pronounced compared to GAC (Chow et al. 2022). With respect to treatment system design, increasing EBCT up to 3 min is beneficial, especially for short-chain PFAS removal (Murray et al. 2021, Chow et al. 2022). The effects of dissolved organic matter, inorganic anions, initial PFAS concentration, co-occurring PFAS, and treatment criteria on IX resin use rates are still poorly understood and were a focus of the research conducted in this study.

3.1.2 Materials and Methods

Water. Rapid small-scale column tests (RSSCTs) at North Carolina State University (NCSU) were completed with groundwater (GW) collected from three locations: Wake County, NC (not PFAS-impacted), Fountain, CO (PFAS-impacted), and Land O Lakes, FL (not PFAS-impacted).

Background water quality parameters for the three groundwaters were determined and are summarized in Table 3.1. **Error! Reference source not found..** The selected groundwaters differ in their total organic carbon (TOC) concentration (<0.3 mg C/L for NC GW, 1.4 mg C/L for CO GW, 4.6 mg C/L for FL GW) and in their total dissolved solids concentrations as indicated by the conductivity values (165 uS/cm for NC GW, 918 uS/cm for CO GW, 241 uS/cm for FL GW).

Table 3.1. Water Characteristics of the Three Groundwater (NC/CO/FL) Used in RSSCTs.

Parameters	Abbr.	Unit	NC GW	CO GW	FL GW	Method	MRL
Total organic carbon	TOC	mg C/L	< 0.30	1.35	4.60	Shimadzu NPOC	0.3
Dissolved organic carbon	DOC	mg C/L	< 0.30	1.34	4.51	Shimadzu NPOC	0.3
UV absorbance	UV ₂₅₄	abs/cm	0.003	0.026	0.082	Hach Spectrophotometer	-
Specific UV absorbance	SUVA	L/mg-m	-	1.9	1.8	Calculated	-
Conductivity	-	uS/cm	165	918	241	Hach Conductivity meter	-
pH	-	-	6.7	7.25	7.4	Hach pH meter	-
Calcium	Ca	mg/L	12.65	83.51	41.09	ICP-MS	0.005
Magnesium	Mg	mg/L	8.28	24.04	3.77	ICP-MS	0.005
Sodium	Na	mg/L	5.92	84.96	2.55	ICP-MS	0.005
Potassium	K	mg/L	1.32	6.01	0.32	ICP-MS	0.005
Iron	Fe	mg/L	< 0.001	< 0.001	< 0.001	ICP-MS	0.001
Manganese	Mn	mg/L	< 0.005	< 0.005	< 0.005	ICP-MS	0.001
Aluminum	Al	mg/L	< 0.005	< 0.005	< 0.005	ICP-MS	0.005
Lithium	Li	mg/L	< 0.005	0.04	< 0.005	ICP-MS	0.001
Copper	Cu	mg/L	0.018	< 0.005	< 0.005	ICP-MS	0.001
Zinc	Zn	mg/L	0.021	< 0.005	< 0.005	ICP-MS	0.001
Strontium	Sr	mg/L	0.067	0.56	0.14	ICP-MS	0.001
Barium	Ba	mg/L	0.028	0.05	0.007	ICP-MS	0.001
Silicon	Si	mg/L	16.65	8.18	3.11	ICP-MS	0.010
Phosphorus	P	mg/L	0.056	0.03	0.016	ICP-MS	0.005
Sulfur (as SO ₄)	S	mg/L	0.5	187.10	7.5	ICP-MS	5.000
Chlorine	Cl	mg/L	4.9	82.60	1.2	ICP-MS	5.000
Sulfate	SO ₄ ²⁻	mg/L	<0.5	190.0	7.5	IC	0.5
Chloride	Cl ⁻	mg/L	28.35	89.9	4.9	IC	0.5
Fluoride	F ⁻	mg/L	<0.5	1.65	<0.5	IC	0.5
Nitrate	NO ₃ ⁻	mg/L	10.7	17.0	<0.5	IC	0.5

PFAS. Prior to conducting RSSCTs, groundwaters were spiked with PFAS. Depending on the goal for each set of RSSCTs, the identities and concentrations of spiked PFAS varied, as described in Section 3.2. PFAS spiking involved (1) pipetting a desired volume of methanolic PFAS stock solution (prepared from PFAS salts purchased from Synquest, Alachua, FL) into a 15-mL Falcon tube, (2) evaporating the methanol, (3) dissolving the residue in the water matrix of interest, and (4) adding the dissolved residue to the feed tank. Steps 3 and 4 were repeated three times to assure essentially complete PFAS transfer from the Falcon tube to the 20-L Nalgene feed tank.

Adsorbents. Five GACs and four IX resins were evaluated. Properties of the GACs and the IX resins are summarized in Table 3.2 and Table 3.3, respectively. The selected GACs differed in their base material, surface area, and pore size distribution. For IX experiments, four single-use resins were evaluated. The selected IX resins differed in their structure and ion exchange capacity.

RSSCTs. Design and operating parameters for the RSSCTs simulating full-scale columns are listed in Table 3.4 and Table 3.5 for GAC and IX, respectively. For GAC, RSSCTs simulating a full-scale empty bed contact time (EBCT) of 10 minutes were conducted using groundwaters with different TOC levels, and an RSSCT simulating an EBCT of 20 minutes was conducted in the Fountain, CO groundwater using a GAC bed depth that was twice that listed in Table 3.4.

Table 3.2. Properties of the Five GACs Selected for RSSCTs.

	GAC				
	GAC1	GAC2	GAC3	GAC4	GAC5
	F400	F600	GCN1240	GAC400	AC1230
Material	Coal	Coal	Coconut	Coal	Coconut
BET surface area (m ² /g)	785	614	1107	963	1450
Total pore vol. (cm ³ /g)	0.27	0.16	0.21	0.30	0.57
Iodine number (mg/g)	1000	850	1000	1000	1100
Apparent density (g/mL)	0.54	0.62	0.51	0.49	0.49

Table 3.3. Properties of the Four Single-use IX Resins Selected for RSSCTs.

	IX			
	Resin 1	Resin 2	Resin 3	Resin 4
	CalRes 2301	CalRes 2304	PSR2+	PFA694E
Structure	Macroporous	Gel-type	Gel-type	Gel-type
Anion exchange capacity (eq/L)	≥ 0.51	≥ 0.65	≥ 0.7	0.81
Bulk density, wet (g/mL)	0.69	0.66-0.72	0.69	0.65

Table 3.4. Design and Operating Parameters for Constant Diffusivity (CD)-RSSCTs and Corresponding Full-scale GAC Condition.

Parameters	CD-RSSCT	Full-scale
Particle size (mm)	0.069	0.92
Scaling factor (SF)	13.3	-
X	0	-
EBCT (min)	0.056	10
Bed depth (cm)	0.99	-
Flow rate (mL/min)	1.4	-
Hydraulic loading rate (m/h)	10.6	-
Reynolds #	0.61	-
Minimum Reynolds # (for PFBA)	0.40	-
Time to reach 100k BV (day)	4	-
Water required to reach 100k BV (gal)	2	-

For IX, RSSCTs simulating a full-scale EBCT of 1.5 minutes were conducted using groundwaters with different TOC levels, and an RSSCT simulating an EBCT of 3 minutes was conducted in the Fountain, CO groundwater using an IX bed depth that was twice that listed in Table 3.5. The constant diffusivity (CD) design ($X = 0$) was used to conduct both GAC and IX RSSCTs. As shown in Table 3.4 and Table 3.5, the selected hydraulic loading rates (HLR) resulted in Reynolds numbers that were above the minimum Reynolds numbers. The minimum Reynolds numbers assured that dispersion was not overemphasized in RSSCTs operated at reduced HLRs. Minimum Reynolds numbers were derived from the product of the Reynolds and Schmidt numbers, which was set to a value of 500 for PFBA, the smallest evaluated PFAS. Dispersion can be ignored when the product of the Reynolds and Schmidt numbers ranges from 200 to 200,000 (Fried, 1975). RSSCTs conducted at NCSU and Colorado School of Mines (CSM) used different IX bed diameters and flow rate, but shared similar particle sizes, hydraulic loading rates, and the same simulated full-scale EBCT of 1.5 minutes. Comparison RSSCTs were performed at NCSU and CSM using the Fountain groundwater and CalRes 2301. Results generated by NCSU and CSM agreed reasonably well (within 30% difference) regarding the bed volume to 10% breakthrough (BV10) for most PFAS except for PFHxS, as shown in Figure A1 (see Appendix A).

Table 3.5. Design and Operating Parameters for CD-RSSCTs at NCSU and CSM and Corresponding Full-scale IX Condition.

Parameters	CD-RSSCT		Full-scale
	NCSU	CSM	
Particle diameter, d (mm)	0.070	0.063	0.698
Scaling factor (SF)	10	11	-
X	0	0	-
EBCT (min)	0.0151	0.012	1.5
Bed depth (cm)	0.425	0.4	-
Flow rate (mL/min)	2.23	4.3	-
Hydraulic loading rate (m/h)	16.9	17.9	-
Reynolds #	0.85	0.81	-
Minimum Reynolds # (for PFBA)	0.36	0.34	-
Time to reach 250k BV (day)	2.6	2.3	-
Water required to reach 250k BV (gal)	2.2	3.5	-

Scale-up of RSSCT data. To scale up RSSCT data, approaches developed in a parallel research project were used. The first step in the scale-up approach was to describe the RSSCT data with the pore-surface diffusion model. For GAC, we adjusted the input parameters of the pore and surface diffusion model (PSDM) [Freundlich capacity parameter K, the tortuosity (τ), and the surface-to-pore diffusion flux ratio (SPDFR)] to obtain the best fit of the CD-RSSCT data. We then computed the normalized total intraparticle flux (NTIF) and scaled it to vary with the scaling factor raised to the power 0.25, i.e.,

$$\text{NTIF}_{\text{full-scale}} = \text{SF}^{0.25} \text{NTIF}_{\text{CD-RSSCT}} \quad \text{where } \text{SF} = d_{\text{full-scale}}/d_{\text{CD-RSSCT}}$$

We then computed τ and SPDFR from the predicted full-scale NTIF and used the PSDM with the K value obtained from the CD-RSSCT and the computed τ and SPDFR for the full-scale condition to predict full-scale PFAS breakthrough curves.

For IX, CD-RSSCT data underpredicted pilot-scale performance by a factor of two. To predict full-scale performance, we used the PSDM and adjusted K , τ , and SPDFR to obtain the best fit of the CD-RSSCT data. We then multiplied the bed volume values associated with the CD-RSSCT by two while keeping the C/C_0 values for PFAS breakthrough constant to obtain the PSDM prediction for the full-scale column.

3.2 PFAS REMOVAL BY GAC

3.2.1 Effect of initial PFAS concentration and competition among co-occurring PFAS

To investigate the influence of initial PFAS concentration and co-existing PFAS, RSSCTs were conducted with GAC1 in a low-TOC, low-TDS groundwater (Wake County, NC). Raw and scaled-up breakthrough curves of PFBS, PFHxS, and PFOS in single-, bi-, and tri-solute systems containing PFAS spiked at either 0.2 or 10 nM are shown in Figure 3.1. Both the raw data as well as the scaled-up results suggest adsorption competition among PFAS in bi-solute and tri-solute systems had little effect on normalized PFAS breakthrough curves. Breakthrough of PFBS, PFHxS, and PFOS may occur slightly sooner at higher influent concentrations (10 nM) than at lower influent concentrations (0.2 nM) in this low-TOC, low-TDS groundwater (Figure 3.1). Nonetheless, all the results fell within 20% of the average result obtained for a targeted compound, suggesting that neither initial PFAS concentration nor co-occurring PFAS had a substantial effect on normalized breakthrough curves. It is expected that the small differences observed in the low-TOC, low-TDS groundwater become smaller in water with higher TOC and/or TDS concentrations.

3.2.2 Effect of empty bed contact time (EBCT) on scaled-up PFAS removal

To investigate the effects of EBCT, RSSCTs were conducted with GAC1 in the Fountain, CO groundwater, simulating field-scale EBCTs of 10 and 20 minutes. RSSCTs were conducted in groundwater with a moderate TOC level (1.3 mg/L, Fountain, CO) and low levels (10-40 ng/L) of native PFAS. This groundwater was amended with additional PFAS at a level of 0.2 nM each (C4-C10 PFCAs, PFBS, PFHxS, PFOS, 6:2 FTS, GenX). Parity plots showing the effect of EBCT on bed volumes of water that can be treated to reach 10% (BV10, Figure 3.5a) and 50% (BV50, Figure 3.5b) breakthrough are shown in Figure 3.2 for scaled-up RSSCT results.

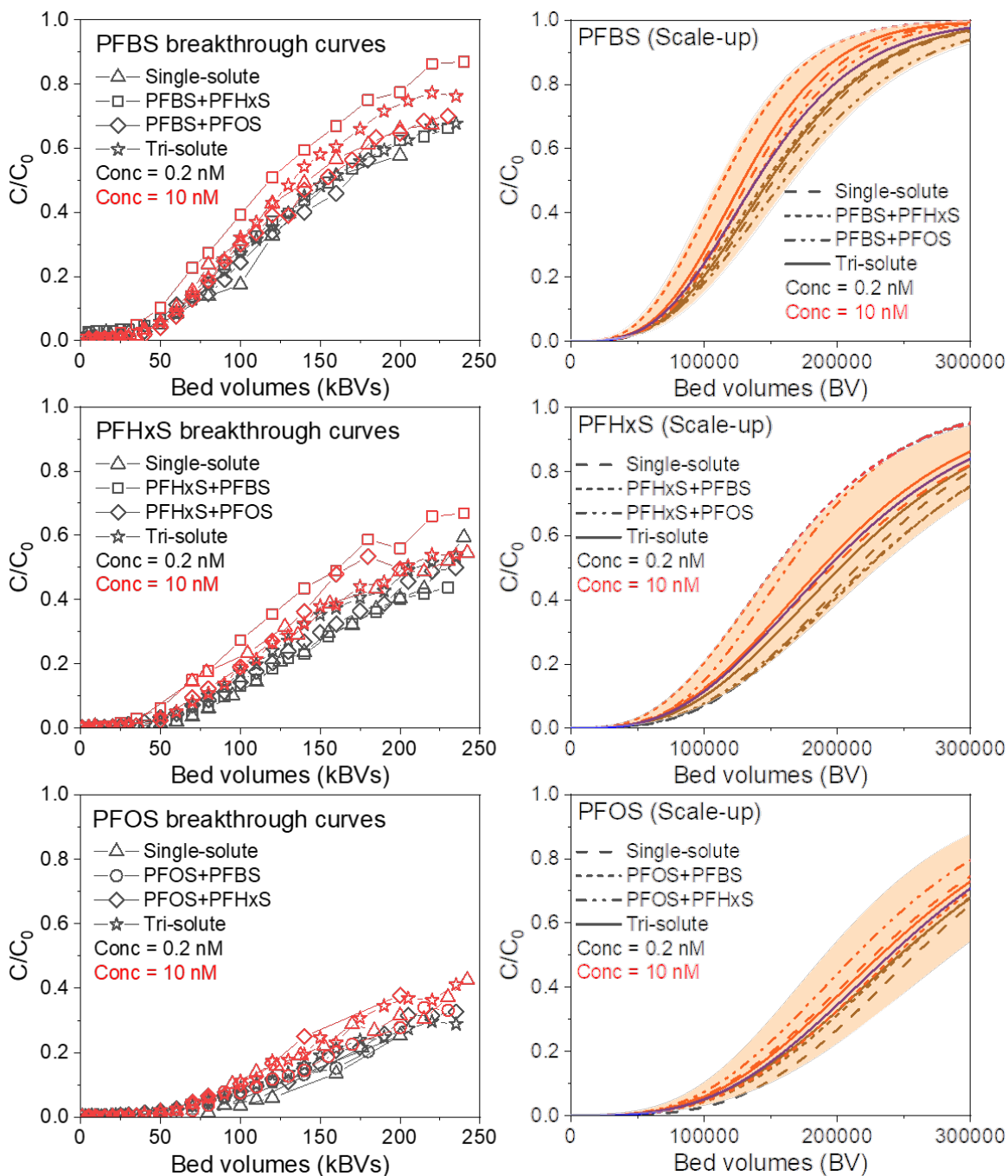


Figure 3.1. Raw and Scaled-up PFBS, PFHxS, and PFOS Breakthrough Data from Single-solute, Bi-solute, and Tri-solute CD-RSSCTs.

GAC: GAC1. Water: Wake County, NC, groundwater (TOC <0.3 mg C/L, specific conductivity = 165 mS/cm). Shaded envelopes show 20% variance around the average best fit obtained for all breakthrough curves obtained for a targeted compound.

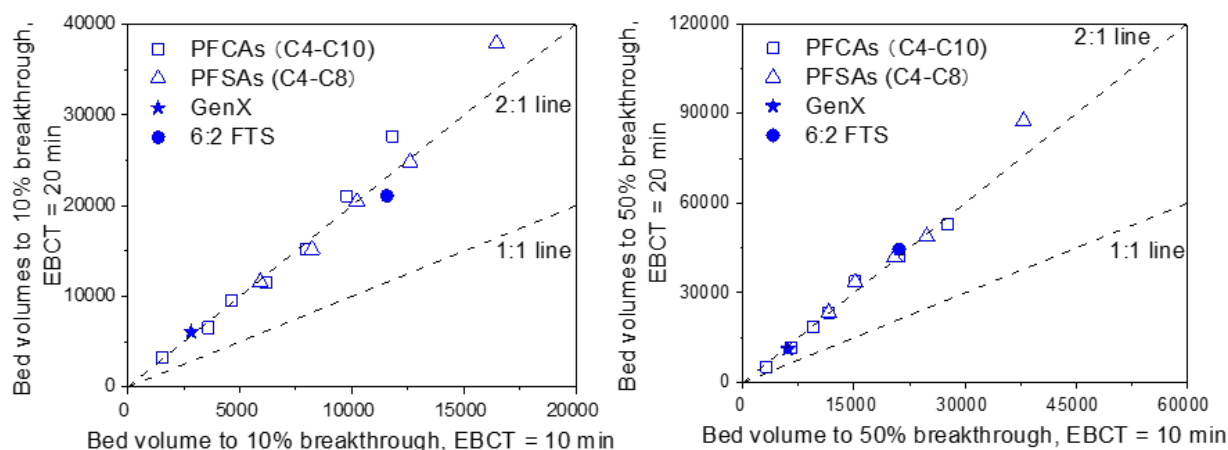


Figure 3.2. Effect of Empty Bed Contact Time (EBCT) on Scaled-up Bed Volume to 10% and 50% Breakthrough.

GAC: GAC1. Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm).

Figure 3.2 suggests that doubling EBCT from 10 minutes (x-axis) to 20 minutes (y-axis) approximately doubled BV10 and BV50, which is equivalent to decreasing GAC use rates by 50%. Additional data are needed to verify that the observed improvement in GAC performance with a doubling in EBCT can be reproduced at the full scale, in different water matrices, and/or with different GACs.

3.2.3 Effect of GAC type on scaled-up PFAS removal

To investigate the influence of GAC type, RSSCTs were conducted with five GACs (GAC1, GAC2, GAC4, GAC3, and GAC5) in Fountain, CO groundwater amended with PFAS at a level of 0.2 nM each (C4-C10 PFCA, PFBS, PFHxS, PFOS, 6:2 FTS). Figure 3.3 shows the effect of GAC type on BV10 and BV50 after scaling up the RSSCT data. BV10 and BV50 values obtained for GACs 2-5 were normalized by those obtained for GAC1. Figure 3.3 suggests that removal efficiency of PFAS by different GACs is PFAS-specific. For a short-chain PFAS, such as PFBA, the removal efficiency followed the order GAC4 > GAC1 ~ GAC2 > GAC5 > GAC3. For long-chain PFAS, such as PFOA and PFOS, the removal efficiency followed the order GAC4 > GAC5 > GAC1 ~ GAC3 > GAC2. Additional fundamental research is needed to develop a clearer understanding of the effects of GAC pore structure and surface chemistry on PFAS removal effectiveness from water with a wide range of background matrix characteristics.

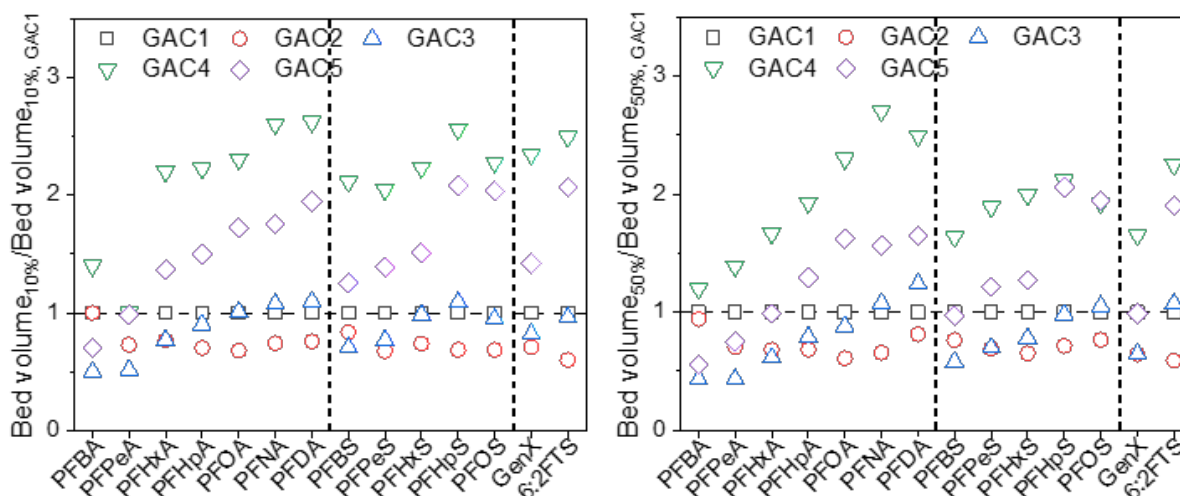


Figure 3.3. Effect of GAC Selection on Scaled-up Bed Volume to 10% and 50% Breakthrough.

Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm).

3.2.4 Effects of total organic carbon (TOC) on the scaled-up removal of PFAS

To investigate the effect of TOC on GAC use rates, RSSCTs were completed with three groundwaters containing TOC levels of <0.25, 1.3, 1.4 (diluted from high-TOC water), and 4.6 mg/L. Figure 3.4 shows the effect of TOC on BV10 and BV50 after scale-up of RSSCT data; panels in the left column show results for all four groundwaters, and panels in the right column exclude data for the low-TOC groundwater (TOC < 0.25 mg/L, Wake County, NC) to more clearly highlight differences among the remaining waters. Parity plots were also constructed to highlight the TOC effect, and the water with a TOC of 1.3 mg/L (Fountain, CO) served as the baseline for comparisons (x-axis in Figure 3.5). As shown in Figure 3.5a, the bed volumes of water that could be treated to 10% breakthrough increased by at least a factor of 10 when groundwater TOC decreased from 1.3 mg/L to <0.25 mg/L. On the other hand, the bed volumes of water that could be treated to 10% breakthrough decreased by as much as a factor of 10 when groundwater TOC increased from 1.3 mg/L to 4.6 mg/L (Figure 3.5b). Overall, results shown in Figure 3.4 and Figure 3.5 illustrate that GAC use rates increase dramatically with increasing TOC concentration.

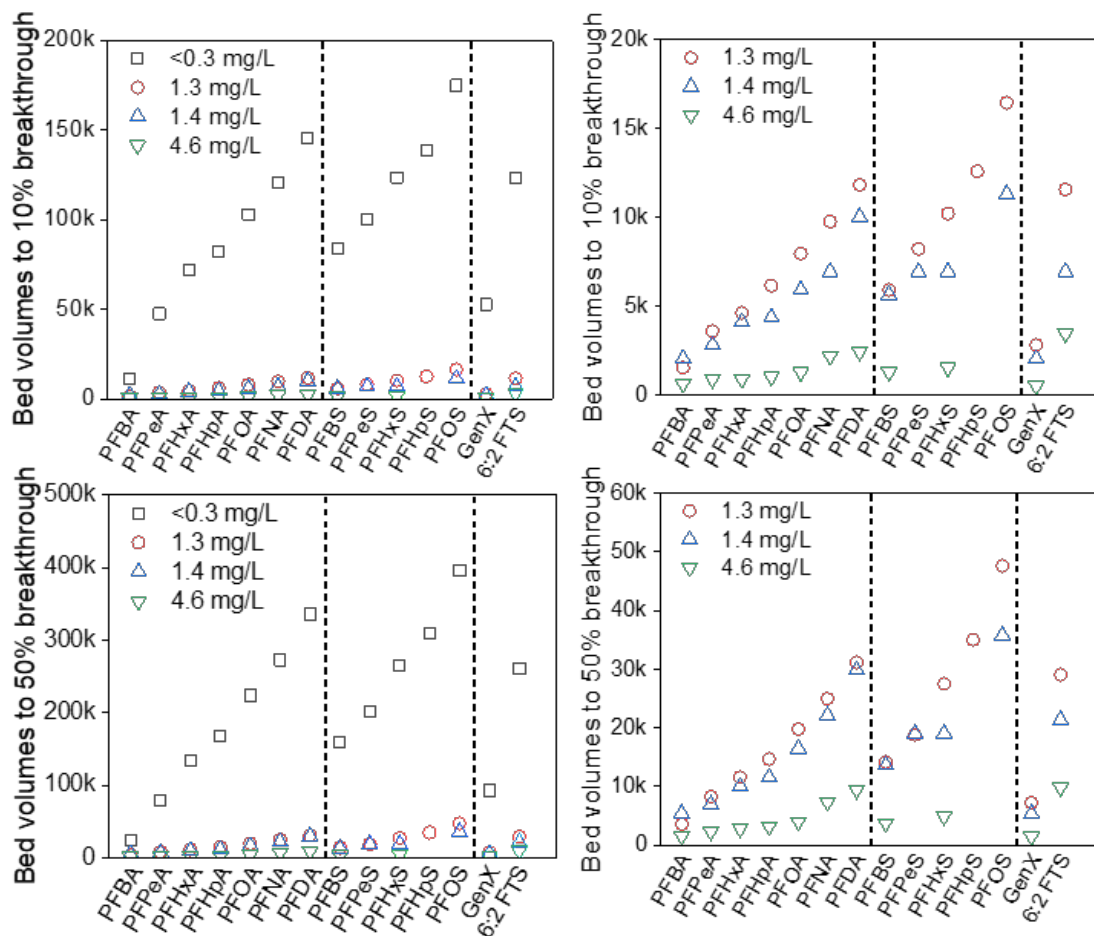


Figure 3.4. Effect of TOC on Scaled-up Bed Volume to 10% and 50% Breakthrough.

GAC: GACI.

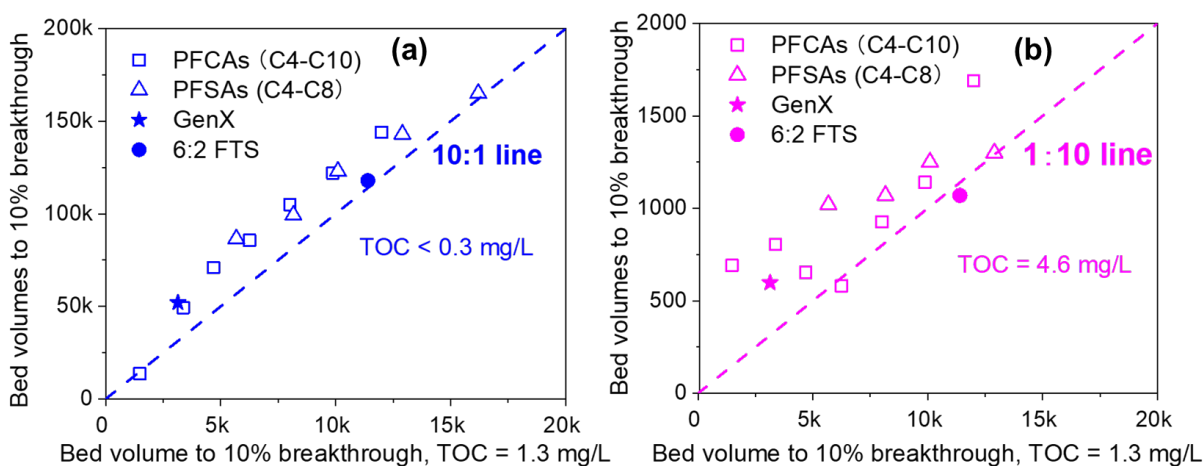


Figure 3.5. Effect of TOC on Scaled-up Bed Volume to 10% and 50% Breakthrough (partial plots).

GAC: GACI.

3.2.5 Effects of inorganic anions on the scaled-up removal of PFAS

To investigate the influence of major anions on PFAS breakthrough, baseline RSSCT data were obtained with GAC1 in a low-TOC, low-TDS groundwater (Wake County, NC) spiked with PFAS at an influent concentration of 0.2 nM. Baseline results were compared to RSSCT data obtained with the same groundwater spiked with a mixture of sodium salts to add 3 meq/L each of chloride (Cl^-), bicarbonate (HCO_3^-), sulfate (SO_4^{2-}), and nitrate (NO_3^-). Figure 3.6 shows the effect of added salts on BV10 and BV50 for each PFAS by normalizing BV10 and BV50 values obtained with the salt-amended water to those obtained in baseline water. Results highlight that the effect of the added salts depended were PFAS-specific. For PFBA and PFPeA, the salt mixture resulted in an earlier breakthrough. For PFHxA and GenX, the effect of the salt mixture was negligible. For the other 11 PFAS, adding the salts resulted in a later breakthrough.

To obtain additional insight into how major anions impact GAC performance for different PFAS, four RSSCTs were conducted with GAC1 in Wake County, NC groundwater amended individually with 3 meq/L Cl^- , HCO_3^- , SO_4^{2-} , and NO_3^- . Figure 3.7 suggests that nitrate (NO_3^-) negatively affected the adsorption of short-chain PFAS, while the other three anions benefitted PFAS removal. In the mixture of four anions, the nitrate effect dominated the adsorption of short-chain PFAS on GAC1 (Figure 3.7). Overall, the effects of major anions were more pronounced for BV10 than for BV50, suggesting that adsorption kinetics are impacted by the presence of major anions in addition to adsorption capacity, the latter being reflected by the BV50 results.

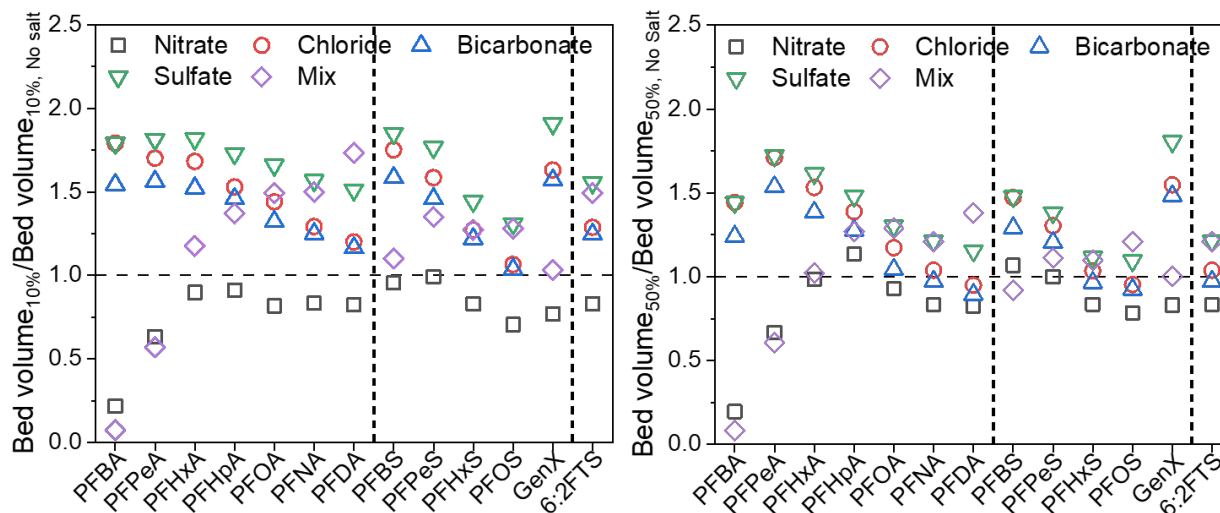


Figure 3.6. Effect of Inorganic Anions on Scaled-up Bed Volume to 10% and 50% Breakthrough.

GAC: GAC1. Water: Wake County, NC, groundwater (TOC <0.3 mg C/L, specific conductivity = 165 $\mu\text{S}/\text{cm}$).

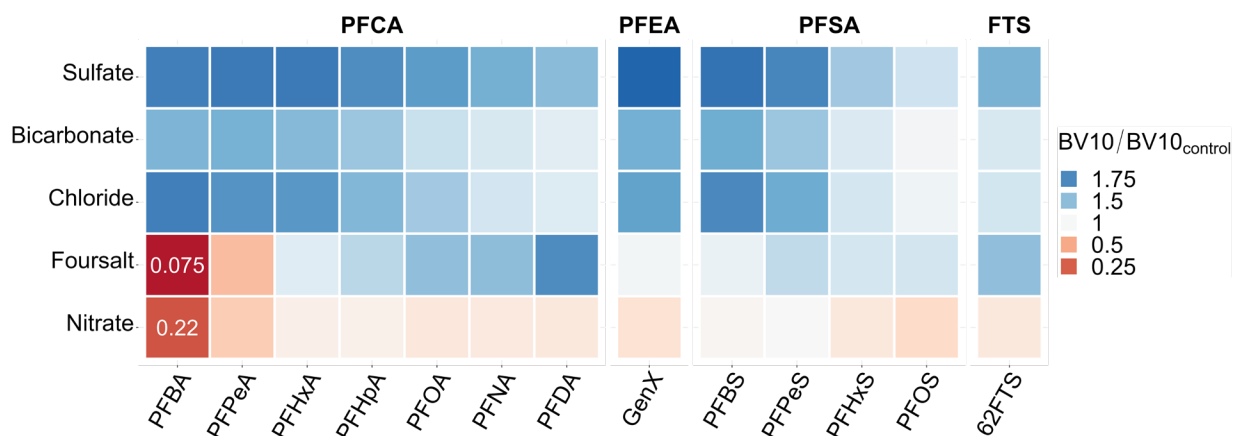


Figure 3.7. Effect of 3 meq/L Inorganic Anions on Normalized Bed Volume to 10% Breakthrough (BV10/BV10 Control) After Scale-up.

GAC: GAC1. Water: Wake County, NC, groundwater (TOC <0.3 mg C/L, specific conductivity = 165 μ S/cm).

3.2.6 Effect of aqueous film-forming foam (AFFF) on scaled-up PFAS removal

To investigate the influence of non-fluorinated constituents in fresh and aged AFFF, RSSCTs were conducted with GAC1 in two groundwaters containing TOC levels of < 0.3 and 4.6 mg/L. The fresh AFFF-impacted water amended with PFHxS and PFOS concentrations of 10 nM each was prepared by spiking around 9 μ L of AFFF concentrate (PFOS concentration: 10 g/L, PFHxS concentration: 1.2 g/L) into 5 gallons of groundwater and fortifying with PFHxS (from a neat stock solution). The aged AFFF-impacted water amended with PFHxS and PFOS concentrations of 0.2 nM was prepared by spiking around 190 mL of AFFF-impacted groundwater (PFOS concentration: 10 μ g/L, PFHxS concentration: 4 μ g/L) into 5 gallons of groundwater and fortifying with PFHxS. The non-AFFF water was prepared by spiking pure PFHxS and PFOS analytes. The TOC levels of the groundwater remained constant after spiking the AFFF concentrate but increased by around 0.1 mg/L after spiking the AFFF-impacted groundwater.

Figure 3.8 shows the effect of non-fluorinated constituents of fresh- and aged-AFFF on BV10 and BV50 after scaling up the RSSCT data. Fresh-AFFF constituents had a negligible effect on GAC performance in both groundwaters, while aged-AFFF constituents showed a negative impact on PFAS removal, which likely resulted from the slight elevation in TOC levels (0.1 mg/L) after the spiking the AFFF-impacted groundwater (190 mL into 5 gallons), while the TOC level remained the same after spiking the AFFF concentrate (9 μ L into 5 gallons).

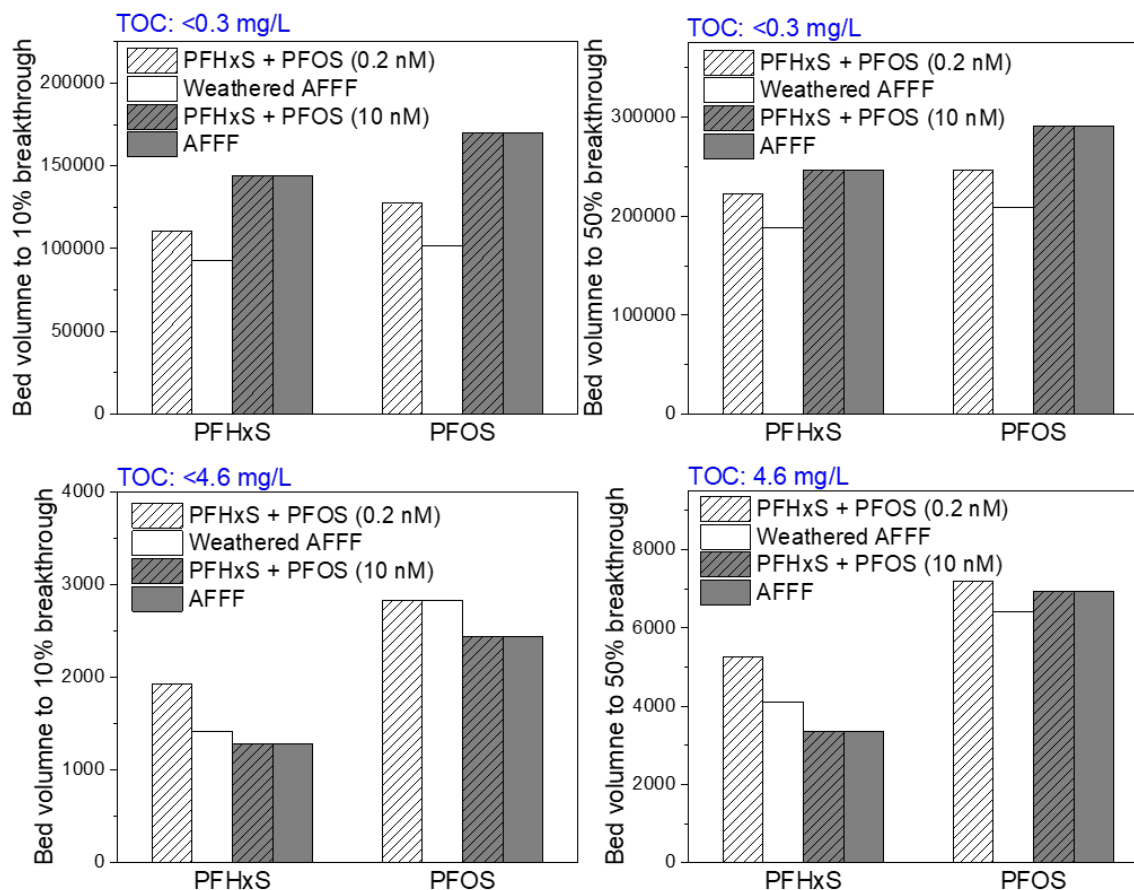


Figure 3.8. Effects of AFFF Matrices (AFFF and Aged, Weathered AFFF) on Scaled-up Bed Volume to 10% and 50% Breakthrough for PFHxS and PFOS.

GAC: GAC1. Water: Wake County, NC, groundwater (TOC <0.3 mg/L, specific conductivity = 185 μ S/cm); Land O Lakes, FL, groundwater (TOC = 4.6 mg/L).

3.2.7 Comparison of treatment criteria: Impact on GAC use rates

To investigate the effect of different treatment goals on GAC use rates, we assumed that the GAC influent contained 0.2 nM each of PFHpA (73 ng/L), PFOA (83 ng/L), PFNA (93 ng/L), PFHxS (80 ng/L) and PFOS (100 ng/L). Using scaled-up RSSCT results obtained with the PSDM, we applied different treatment goals to the resulting PFAS breakthrough curves and obtained GAC use rates (Figure 3.9). The evaluated treatment goals included (1) the 2016 US EPA lifetime health advisory level (<70 ng/L for the sum of PFOA and PFOS), (2) New Jersey MCLs (PFOA <14 ng/L, PFOS < 13 ng/L, PFNA < 13 ng/L), (3) Michigan MCLs (PFOA <8 ng/L, PFOS < 16 ng/L, PFNA < 6 ng/L), (4) Vermont MCL (<20 ng/L for the sum of PFHpA, PFOA, PFNA, PFHxS and PFOS), (5) California Notification Level (PFOA < 5 ng/L) and (6) the MCLs EPA proposed in 2023 for PFOS and PFOA (4 ng/L each). Scenarios were evaluated for 3 groundwaters (low TOC: < 0.25 mg/L, moderate TOC: 1.3 mg/L, and high TOC: 4.6 mg/L), and resulting GAC use rates are summarized in Table 3.6.

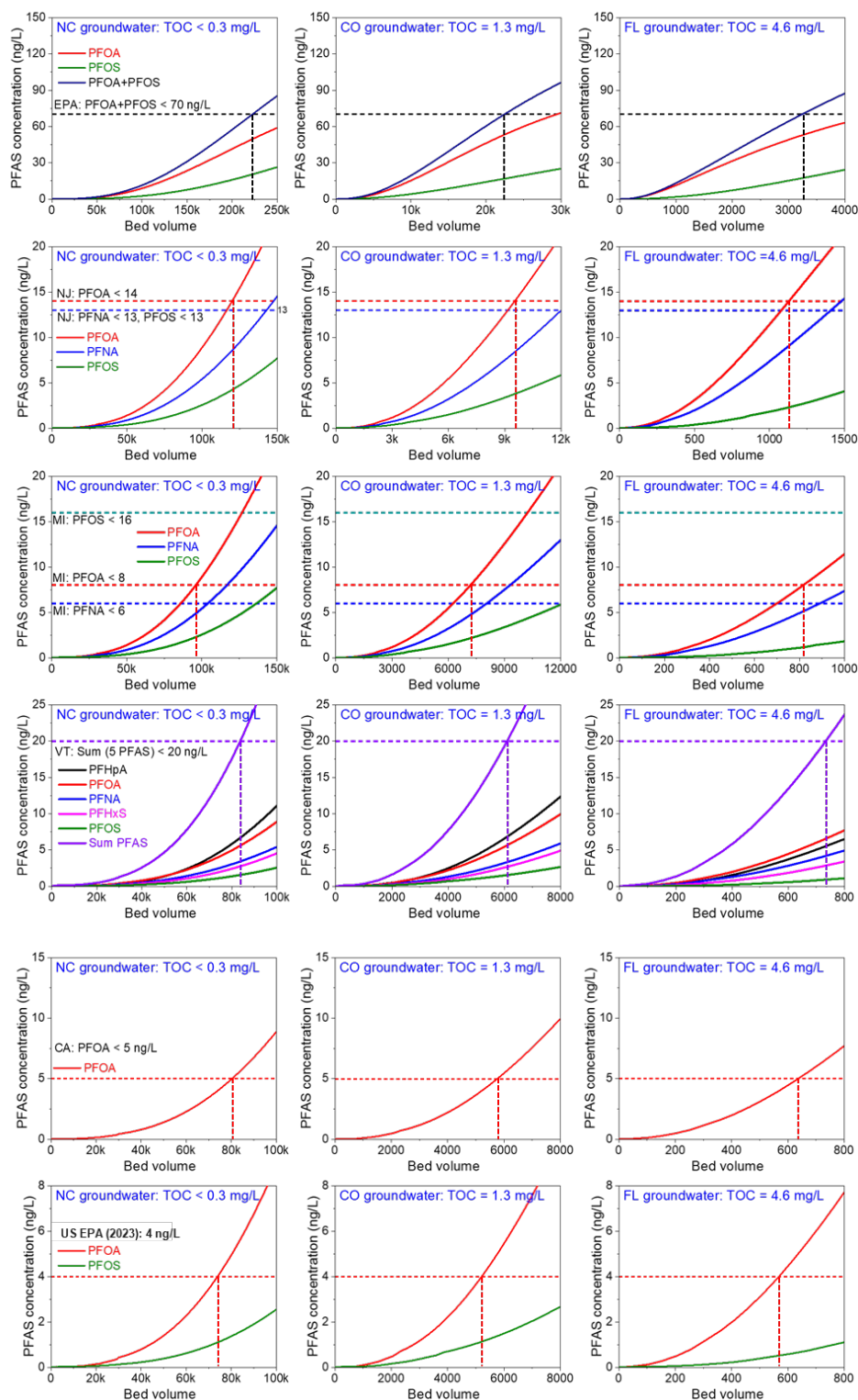


Figure 3.9. Effect of Different Treatment Goals on GAC Use Rates.

Table 3.6. GAC Use Rates by Following Different Treatment Goals.

Standard/ Guideline	Treatment criteria (ng/L)	GAC use rate (lb/1000 gal)		
		NC GW	CO GW	FL GW
US EPA (2016)	PFOA+PFOS<70	0.020	0.19	1.35
New Jersey	PFOA<14, PFOS<13, PFNA<13	0.036	0.48	3.99
Michigan	PFOA<8, PFOS<16, PFNA<6	0.044	0.60	5.26
Vermont	PFHpA+PFOA+PFNA+PFHxA+PFOS<20	0.050	0.71	5.75
California	PFOA<5	0.052	0.72	6.35
US EPA (2023)	PFOA<4, PFOS<4	0.057	0.79	7.45

The results highlight that GAC use rates are expected to increase dramatically with increasing TOC levels. For NC groundwater (TOC <0.25 mg/L), a GAC use rate of 0.057 lb GAC/1000 gal is predicted for the proposed EPA MCL; GAC use rates are predicted to increase to 0.79 lb GAC/1000 gal for CO groundwater (TOC = 1.3 mg/L) and to 7.45 lb GAC/1000 gal for FL groundwater (TOC = 4.6 mg/L). The stringency of the treatment goal also affects GAC use rates, especially in water with higher TOC levels. For NC groundwater, GAC use rates increased from 0.020 lb GAC/1000 gal for the 2016 EPA health advisory levels to 0.057 lb GAC/1000 gal for the 2023 EPA proposed MCLs, an increase of a factor of about 3. For CO groundwater, GAC use rates increased from 0.19 to 0.79 lb GAC/1000 gal (factor of 4.2), and for FL groundwater, from 1.35 to 7.45 lb GAC/1000 gal (factor of 5.5) when comparing 2016 and 2023 EPA criteria.

3.3 PFAS REMOVAL BY IX

3.3.1 Effects of initial PFAS concentration and competition between co-existing PFAS

To investigate the influence of initial PFAS concentration and co-existing PFAS on IX use rates, RSSCTs were conducted with a macroporous IX resin (CalRes2301) in a low-TOC, low-TDS groundwater (Wake County, NC) and a moderate-TOC groundwater (Fountain, CO). Native PFAS concentrations in the CO groundwater ranged from 10 ng/L to 20 ng/L, and we amended the water with PFBS at 0.2 nM (pseudo-single-solute system) or a tri-solute mixture of PFBS, PFHxA, and PFOA (tri-solute system) at 10 nM each. For the CO groundwater, Figure 3.10 shows the PFHxA and PFOA breakthrough curves in the as-received water (PFHxA and PFOA concentrations 10 ng/L each) and the same water amended with the tri-solute mixture (spiked with 10 nM each of PFBS, PFHxA, and PFOA). The breakthrough curves for PFHxA and PFOA in the two systems were very similar, suggesting that initial PFAS concentration and competition among co-occurring PFAS have a negligible effect on PFAS breakthrough curves for the selected IX resin.

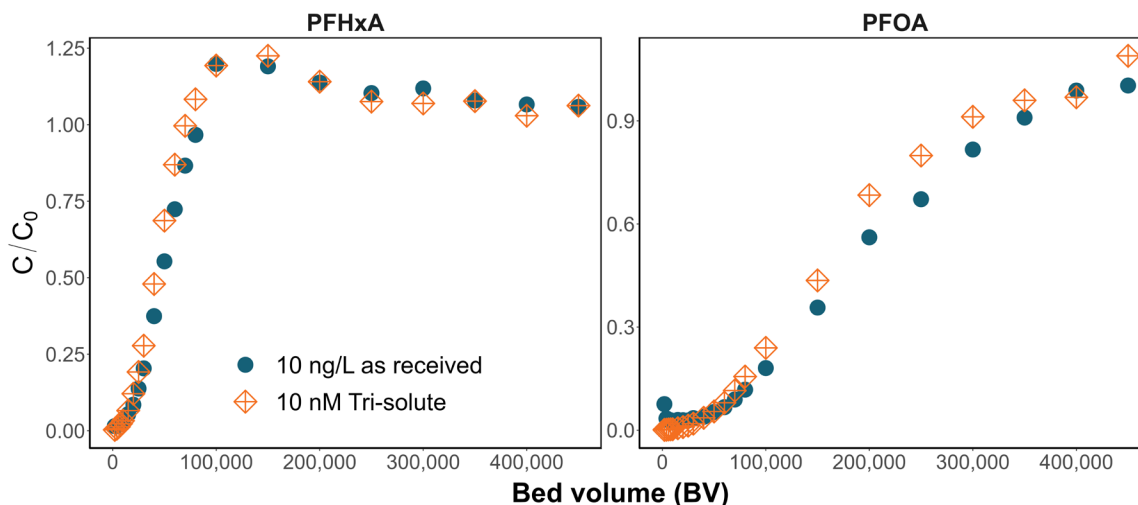


Figure 3.10. PFHxA and PFOA Breakthrough Data from Single-solute (Spiked with PFBS Only) and Tri-solute (Spiked with PFBS, PFHxA and PFOA) CD-RSSCTs.

IX: CalRes2301. Water: Fountain, CO, groundwater (TOC=1.3 mg/L).

Figure 3.11 shows the PFBS breakthrough curves in pseudo-single-solute and tri-solute systems containing PFAS spiked at either 0.2 or 10 nM into low-TOC or moderate-TOC groundwater. At both conditions, PFBS exhibited negligible breakthrough after treating 500,000 BVs of the low-TOC groundwater. In the moderate-TOC groundwater, the difference between two systems was small, with PFBS breakthrough occurring slightly sooner at the higher influent concentration (10 nM in a tri-solute system) than at the lower influent concentration (0.2 nM in a single-solute system). Compared to other factors we investigated, the effects of initial PFAS concentration and co-occurring PFAS on IX use rates were small or negligible. From a practical perspective, this means we can describe apparent adsorption equilibria with linear isotherms, which facilitates the prediction of PFAS breakthrough curves with the PSDM. In this case, the Freundlich capacity parameter K becomes a partition coefficient (K_{AIX} , L/g)

3.3.2 Effect of empty bed contact time (EBCT) on scaled-up PFAS removal by IX

To investigate the effects of EBCT, RSSCTs were conducted with one IX resin (CalRes2301) in the Fountain, CO groundwater, simulating field-scale EBCTs of 1.5 and 3 minutes. RSSCTs were conducted in groundwater with a moderate TOC level (1.3 mg/L, Fountain, CO) and low levels (10-40 ng/L) of native PFAS. This groundwater was amended with additional PFAS at a level of 0.2 nM each (C4-C8 PFCAs, PFBS, PFHxS, PFOS). Parity plots showing the effect of EBCT on bed volumes of water that can be treated to reach 10% breakthrough are shown in Figure 3.12 **Error! Reference source not found.** for scaled-up RSSCT results.

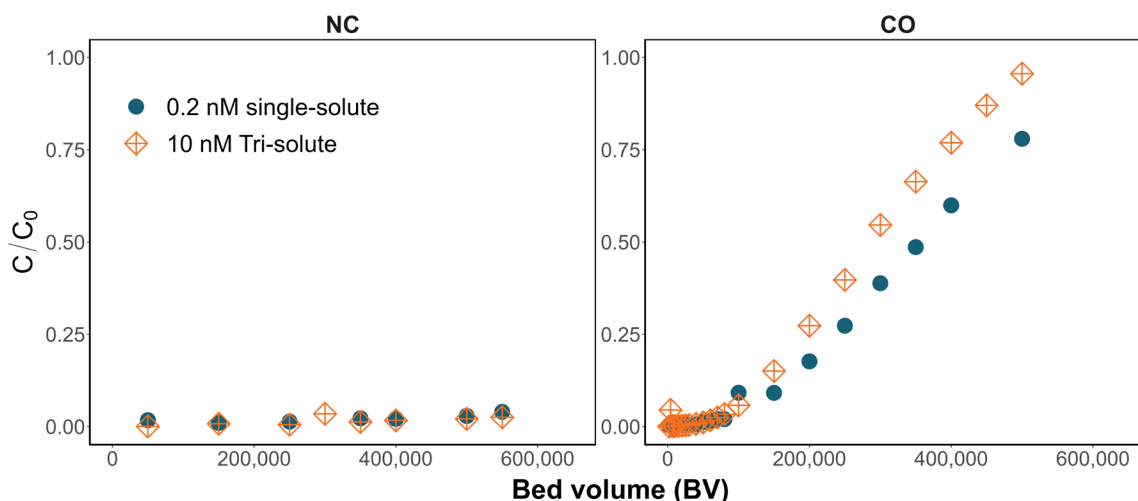


Figure 3.11. PFBS Breakthrough Data from Pseudo-single-solute (Dosed with 0.2 nM PFBS Only) and Tri-solute (Dosed with PFBS, PFHxA, and PFOA) CD-RSSCTs.

IX: CalRes2301. Water: Wake County, NC, groundwater (TOC <0.3 mg C/L) and Fountain, CO, groundwater (TOC=1.3 mg/L).

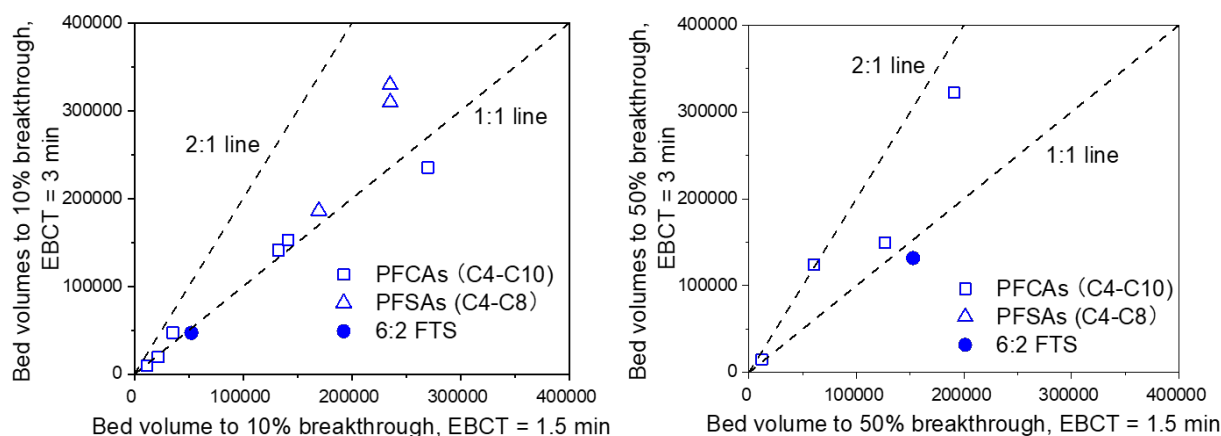


Figure 3.12. Effect of Empty Bed Contact Time (EBCT) on Scaled-up Bed Volume to 10% Breakthrough. IX.

Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm).

Figure 3.12 suggests that doubling EBCT from 1.5 minutes (x-axis) to 3 minutes (y-axis) had a negligible impact on PFCAs and 6:2 FTS breakthrough and resulted in slightly higher BV10 values for PFSA's. As the BV50 of most of the compounds during the treatment of CO groundwater was not reached, additional data are needed to verify that the apparent improvement in IX performance with a doubling in EBCT can be reproduced at the full scale, in different water matrices, and/or with different IXs.

3.3.3 Effect of IX type on scaled-up PFAS removal

To investigate the influence of IX type, RSSCTs were conducted with four IX (CalRes2301(1), CalRes2304 (2), PSR2+ (3), and Purolite 694E (4)) in Fountain, CO groundwater amended with PFAS at a level of 0.2 nM each (C4-C8 PFCA, PFBS, PFHxS, PFOS). Figure 3.13 shows the effect of IX type on BV10 and BV50 after scaling up the RSSCT data. BV10 and BV50 values obtained for IXs 2-4 were normalized by those obtained for IX 1. Figure 3.13 suggests that removal efficiency of PFAS by different IXs is PFAS-specific. For a carboxylate PFAS such as PFOA (C8), the removal efficiency followed the order PSR2+ (IX 3) > 694E (IX 4) > CalRes2304 (IX 2) > CalRes2301 (IX 1). For long-chain PFAS such as PFOS, the removal efficiency followed the order 694E (IX 4) > IX 1 ~ IX 2 ~ IX 3. Additional fundamental research is needed to develop a clearer understanding on the effects of IX properties on PFAS removal effectiveness from water with a wide range of background matrix characteristics.

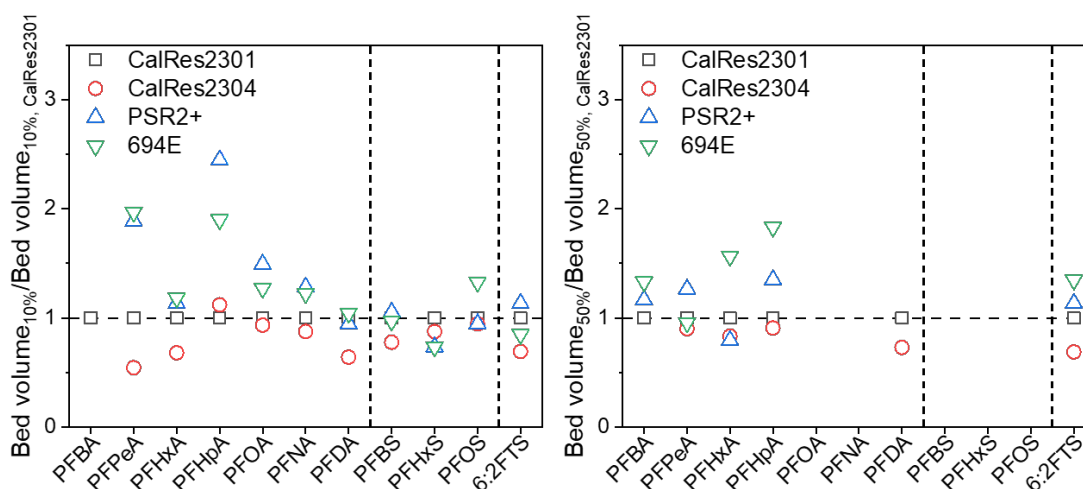


Figure 3.13. Effect of GAC Selection on Scaled-up Bed Volume to 10% and 50% Breakthrough.

Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm).

3.3.4 Effect of total organic carbon (TOC) on scaled-up PFAS removal

To investigate the effect of TOC on IX use rates, RSSCTs were completed with three groundwaters with TOC levels of <0.3, 1.3, and 4.6 mg/L. Figure 3.14 shows the effect of TOC on BV10 and BV50 after scale-up, suggesting that the bed volumes of water that could be treated to 10% or 50% breakthrough decreased substantially when groundwater TOC levels increased. For example, BV10 for PFHxA dropped from 226,000 BVs to 28,600 BVs and 16,400 BVs when groundwater TOC increased from <0.25 mg/L to 1.3 mg/L and 4.6 mg/L, respectively. In the low-TOC water, only PFBA, PFPeA, PFHxA, and GenX reached meaningful levels of breakthrough and, therefore, could be scaled up. The results in Figure 3.14 also suggest that long-chain PFAS are more impacted by TOC than short-chain PFAS. For example, BV10 for PFHxA and PFOA increased by a factor of 1.7 and 3.2, respectively, when groundwater TOC decreased from 4.6 mg/L to 1.3 mg/L.

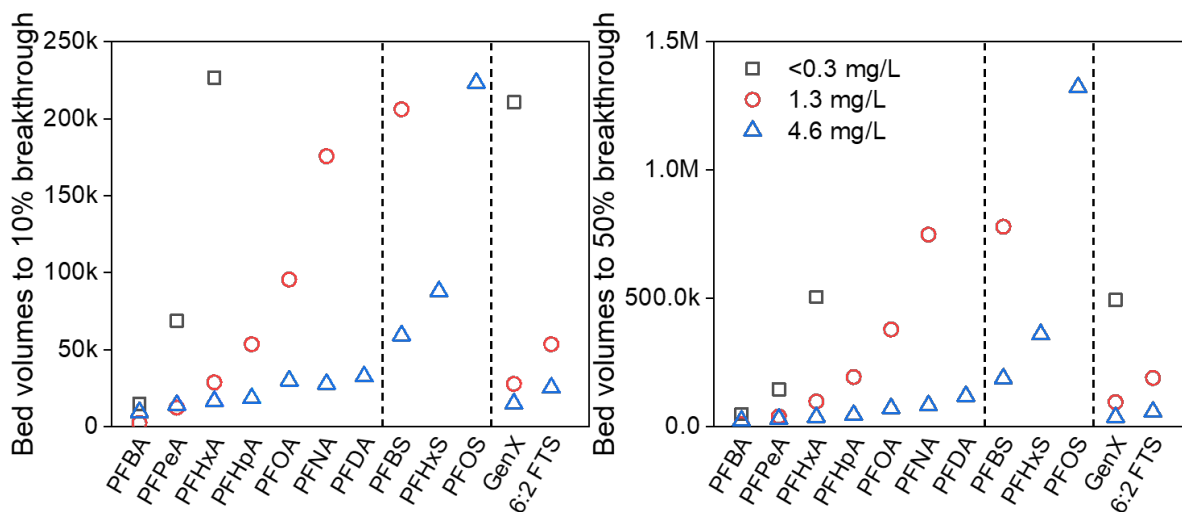


Figure 3.14. Effect of TOC on Scaled-up Bed Volume to 10% and 50% Breakthrough.

IX: CalRes2301.

The effect of TOC on K_{AIX} is shown in Figure 3.15. For PFAS that exhibited no measurable breakthrough at the completion of the RSSCT, K_{AIX} was predicted from a quantitative structure-property relationship (QSPR) model developed in a separate research project (lighter-shade symbols in Figure 3.15). Figure 3.15 further illustrates that the adverse effect of TOC on IX resin performance increases with increasing PFAS chain length in a given PFAS sub-class (PFCAs, PFSAAs).

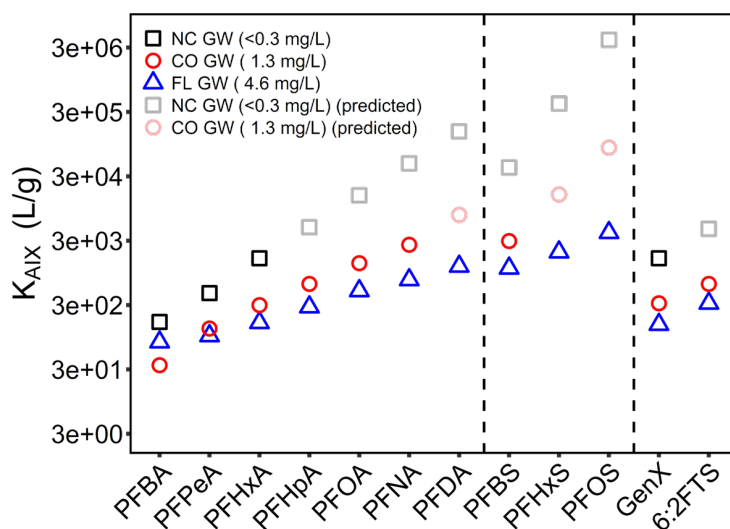


Figure 3.15. Effect of TOC on K_{AIX} .

IX: CalRes2301.

3.3.5 Effect of inorganic anions on scaled-up PFAS removal

To investigate the influence of inorganic anions on PFAS removal by IX, RSSCTs were conducted with CalRes2301 in a low-TOC, low-TDS groundwater (Wake County, NC) that was amended with a PFAS mixture such that the influent concentration of each PFAS was 0.2 nM. A baseline experiment was conducted using the PFAS-spiked groundwater without adding salts. Results from the baseline RSSCT were compared to those obtained with the same groundwater amended with 3 meq/L NO_3^- and an anion mixture containing 3 meq/L each of NO_3^- , Cl^- , HCO_3^- , and SO_4^{2-} . Figure 3.16 shows the effect of coexisting inorganic anions on BV10 and BV50 after scaling up the RSSCT data. Bed volumes of water that could be treated to 10% or 50% breakthrough decreased substantially in the presence of spiked nitrate and the anion mixture that included nitrate. In the control water, only PFBA, PFPeA, PFHxA, and GenX reached meaningful levels of breakthrough and could be scaled up. In the groundwater water dosed with nitrate or mixed anions, PFBA and PFPeA reached complete breakthrough before our first sampling point, while PFDA, PFHxS, and PFOS did not reach meaningful levels of breakthrough and, therefore, could not be scaled up. Based on the limited scaled-up data, we can conclude that nitrate negatively impacts PFAS removal by IX. The similar impacts of nitrate and the anion mixture containing nitrate, especially for shorter-chain PFAS, suggest that the impacts of bicarbonate, chloride, and sulfate were small compared to nitrate.

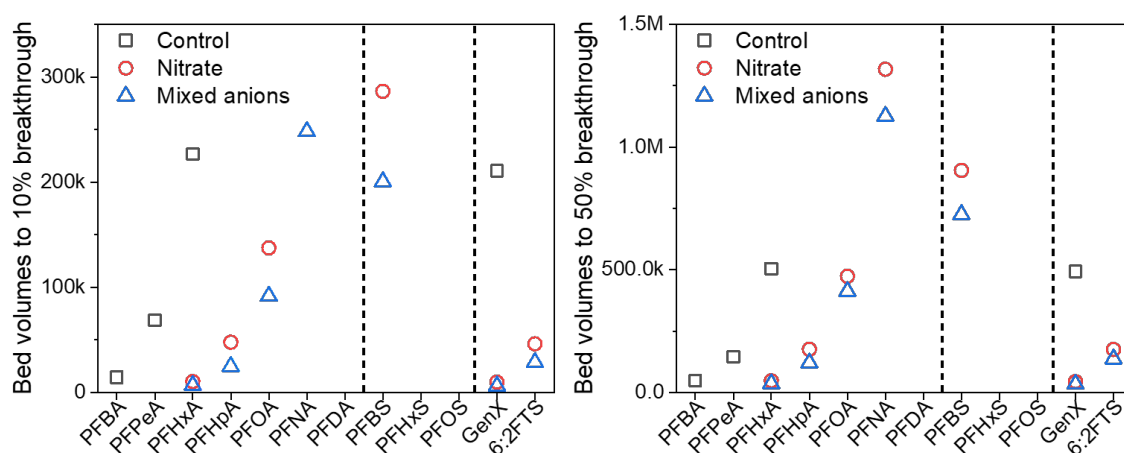


Figure 3.16. Effect of Inorganic Anions on Scaled-up Bed Volume to 10% and 50% Breakthrough.

IX: CalRes2301. Water: Wake County, NC, groundwater (TOC <0.3 mg C/L, specific conductivity = 165 $\mu\text{S}/\text{cm}$).

Figure 3.17 summarizes the effect of coexisting inorganic anions on K_{AIX} , suggesting the presence of nitrate alone or in a mixture adversely impacts the partition coefficient representing the PFAS uptake capacity of the IX resin. For PFAS that exhibited no measurable breakthrough at the completion of the RSSCT, K_{AIX} was predicted from a QSPR model developed in a separate research project (lighter-shade symbols in Figure 3.17). Figure 3.17 suggests the impact of nitrate alone on K_{AIX} is similar to that observed for nitrate in the presence of chloride, sulfate, and bicarbonate, suggesting that nitrate primarily impacted PFAS removal by CalRes2301 IX.

It is likely that IX resins designed for PFAS removal were originally developed for perchlorate removal. Perchlorate is a poorly hydrated ion, and resins for perchlorate removal are designed to be selective for poorly hydrated ions. It is expected, therefore, that there is competition for ion exchange sites among other poorly hydrated ions, such as nitrate and PFAS, while chloride, bicarbonate, and sulfate, which are better hydrated, have a lesser impact on the performance of IX resins that are selective for poorly hydrated ions.

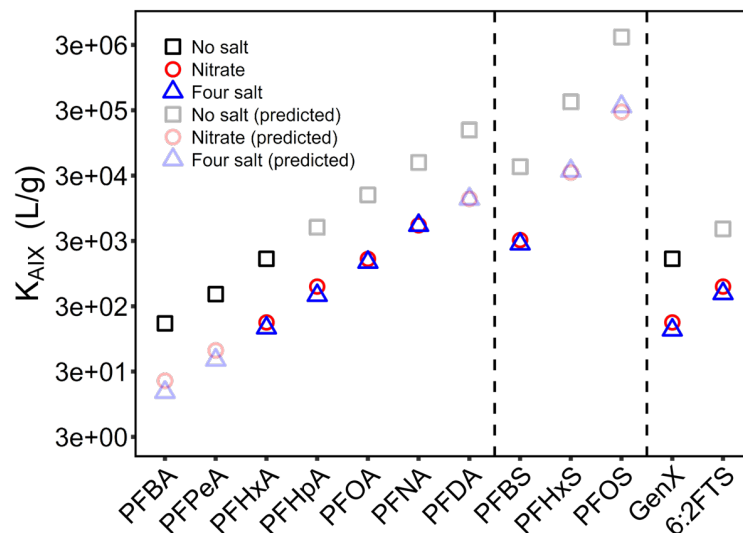
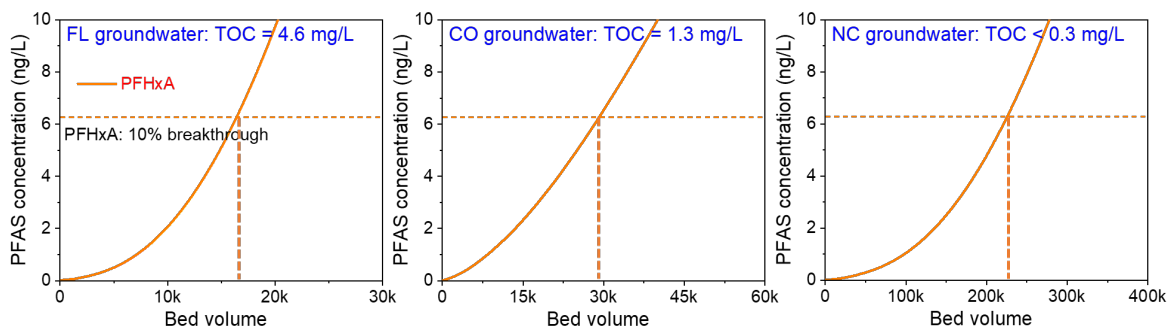


Figure 3.17. Effect of Coexisting Inorganic Anions on K_{AIX} .

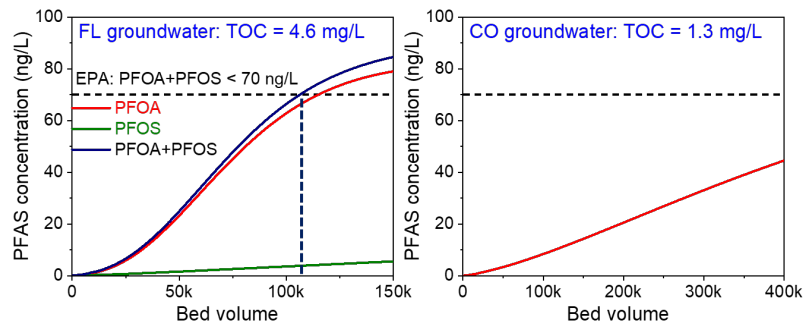
IX: CalRes2301.

3.3.6 Comparison of treatment criteria: Impact on IX use rates

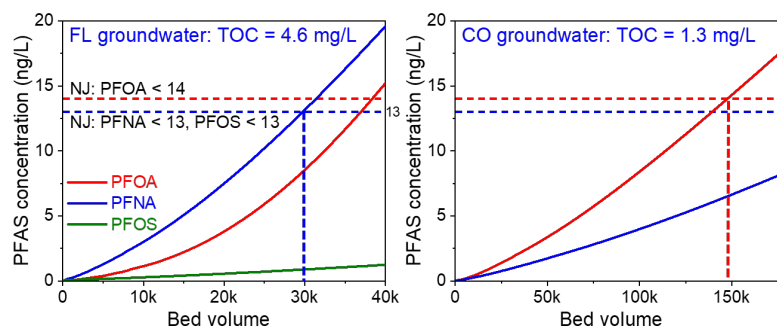
To investigate the effect of different treatment goals on IX use rates, we assumed that the IX influent contained 0.2 nM each of PFHpA (73 ng/L), PFOA (83 ng/L), PFNA (93 ng/L), PFHxS (80 ng/L) and PFOS (100 ng/L). Using scaled-up RSSCT results obtained with the PSDM, we applied different treatment goals to the resulting PFAS breakthrough curves and obtained IX use rates (Figure 3.18). The evaluated treatment goals included (1) the 2016 US EPA lifetime health advisory level (<70 ng/L for the sum of PFOA and PFOS), (2) New Jersey MCLs (PFOA < 14 ng/L, PFOS < 13 ng/L, PFNA < 13 ng/L), (3) Michigan MCLs (PFOA < 8 ng/L, PFOS < 16 ng/L, PFNA < 6 ng/L), (4) Vermont MCL (< 20 ng/L for the sum of PFHpA, PFOA, PFNA, PFHxS and PFOS), (5) California Notification Level (PFOA < 5 ng/L) and (6) the MCLs EPA proposed in 2023 for PFOS and PFOA (4 ng/L each). Scenarios were evaluated for 3 groundwaters (low TOC: < 0.25 mg/L, moderate TOC: 1.3 mg/L, and high TOC: 4.6 mg/L), and resulting IX use rates are summarized in Table 3.7. In the low-TOC NC groundwater, no meaningful breakthrough was observed for PFHpA, PFOA, PFNA, PFHxA, and PFOS, therefore, no exact IX use rates could be determined for most treatment scenarios. We therefore added an additional treatment scenario that corresponds to 10% breakthrough for PFHxA.



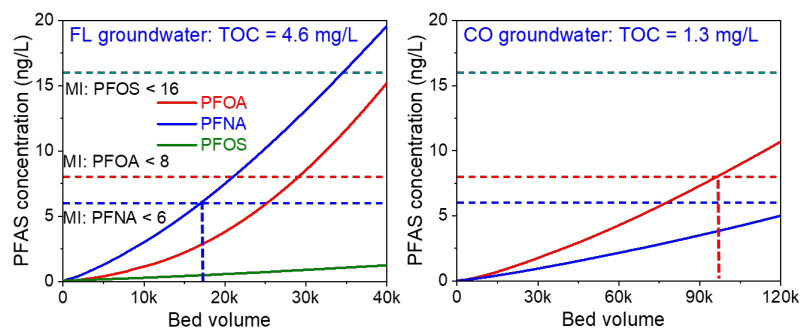
(a)



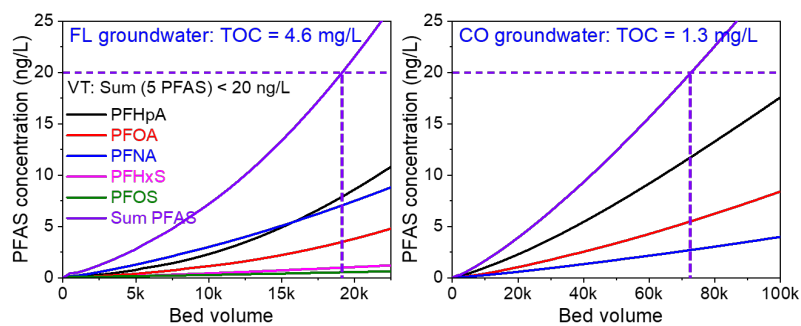
(b)



(c)



(d)



(e)

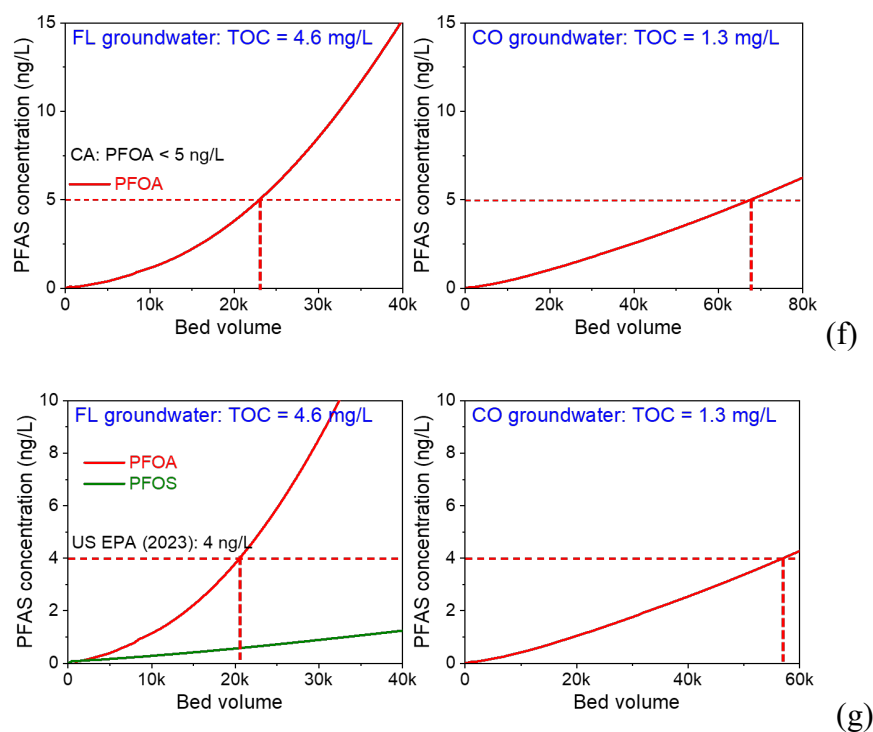


Figure 3.18. Effect of Different Treatment Goals on IX Use Rates

(a) 10% PFHxA breakthrough, (b) 2016 EPA HAL, (c) New Jersey MCL, (d) Michigan MCL, (e) Vermont MCL, (f) California notification level, (g) 2023 EPA proposed MCL

Table 3.7. IX Use Rates by Following Different Treatment Goals

Treatment goal	Threshold (ng/L)	IX use rate (lb/1000 gal)		
		NC GW	CO GW	FL GW
10% PFHxA breakthrough	PFHxA < 6.2	0.012	0.098	0.166
US EPA (2016)	PFOA+PFOS<70	<0.003 ¹	<0.006 ²	0.026
New Jersey	PFOA<14, PFOS<13, PFNA<13	<0.003 ¹	0.019	0.094
Michigan	PFOA<8, PFOS<16, PFNA<6	<0.003 ¹	0.029	0.161
Vermont	PFHpA+PFOA+PFNA+PFHxA+PFOS<20	<0.003 ¹	0.038	0.144
California	PFOA<5	<0.003 ¹	0.041	0.120
US EPA (2023)	PFOA<4, PFOS<4	<0.003 ¹	0.049	0.134

¹ Scaled-up based on the last sampling point (480,000 BV) for the low-TOC NC groundwater, which did not reach targeted breakthrough

² Scaled-up based on the last sampling point (232,000 BV) for the moderate-TOC CO groundwater, which did not reach targeted breakthrough.

The results highlight that IX use rates are expected to increase with increasing TOC levels. The TOC effect was more pronounced going from the low-TOC NC groundwater (TOC <0.25 mg/L) to the moderate-TOC CO groundwater (TOC = 1.3 mg/L) than going from the CO-groundwater to the high-TOC FL groundwater (TOC = 4.6 mg/L). For example, for NC groundwater, an IX use rate of <0.003 lb IX/1000 gal is predicted for the proposed EPA MCL; IX use rates are predicted to increase to 0.049 lb IX/1000 gal for CO groundwater (TOC = 1.3 mg/L) and to 0.13 lb IX/1000 gal for FL groundwater (TOC = 4.6 mg/L). The stringency of the treatment goal also affects IX use rates. For CO groundwater, IX use rates increased from <0.006 lb IX/1000 gal for the 2016 EPA health advisory levels to 0.049 lb GAC/1000 gal for the 2023 EPA proposed MCLs, an increase of a factor of >8. For FL groundwater, IX use rates increased from 0.026 to 0.134 lb IX/1000 gal (factor of 5.2) when comparing 2016 and 2023 EPA criteria.

3.4 HIGH-PRESSURE MEMBRANES

3.4.1 Background

Reverse osmosis (RO) and nanofiltration (NF) are reportedly viable PFAS treatment technologies due to their ability to separate dissolved ions and low molecular weight organic compounds (Bellona et al. 2008; Steinle-Darling et al. 2008; Liu et al. 2021a). Electrostatic and size exclusion of anionic perfluoroalkyl acids (PFAAs) from RO and NF membrane polymers is believed to lead to effective separation, and past studies have reported greater than 90% rejection of various PFAAs (Steinle-Darling et al. 2008; Liu et al. 2021a; Appleman et al. 2013). For example, Appleman et al. (2013) evaluated the separation performance of nine PFAAs by a loose NF (NF270) membrane across three synthetic water matrices and a range of flux conditions (17 to 75 LMH) and reported target PFAA rejection >95% except for PFBA (~93%). It is worth noting that most past studies were performed using small bench-scale membrane systems that may not represent conditions relevant to full-scale membrane systems. Additionally, reported high PFAA rejection values are often calculated using very low permeate concentrations or using detection limits due to non-detects in permeate samples (Steinle-Darling et al. 2008; Appleman et al. 2013).

System conditions such as permeate flux and recovery, membrane fouling, and background water matrix constituents can reportedly influence the rejection of PFAS by NF and RO membranes (Steinle-Darling et al. 2008; Liu et al. 2021a, Tang et al. 2006; Tang et al. 2007; Wang et al. 2018). Mechanisms influencing the separation of solutes by NF and RO membranes have been the focal point of numerous studies (Bellona et al. 2004; Agenson et al. 2003; Dang et al. 2014) and include solute and membrane properties, system operating conditions, and feed water quality. Solute properties believed to dictate separation efficiency include molecular size, charge, and adsorbability (e.g., hydrophobic interactions, hydrogen bonding) (Kiso et al. 2001; Bellona et al. 2005; Comerton et al. 2007; Verliefde et al. 2008, Ozaki and Li 2002; Kimura et al. 2003). PFAA's low pKa values render them anionic at near-neutral pH, and their molecular weights are generally near to or greater than molecular weight cut-offs (MWCO) of RO and most NF membranes. Given that separation or rejection of organic solutes by NF and RO is governed largely by size and electrostatic exclusion, PFAAs should be well removed by most NF and RO systems. Several recent studies, however, have reported that the unique properties of PFAAs, in particular their hydrophobicity and relatively low critical micelle concentration, may impact separation performance in certain scenarios (Wang et al. 2018; Mastropietro et al. 2021; Hang et al. 2015).

For example, Wang et al. (2018) reported increased PFOS and PFBS rejection with increasing feed water concentration, decreased rejection with increasing ionic strength (PFBS), and that adsorptive effects influenced separation (mainly PFOS). Liu et al. (2021a) reported significantly lower PFAA rejection by the NF270 membrane when performing experiments on contaminated groundwater compared to deionized water amended with PFAAs highlighting the potential influence of ionic strength on rejection.

One major drawback associated with high-pressure membranes is management of the retentate stream produced during treatment which has frequently been cited as the most divisive issue preventing the widespread adoption of NF and RO membrane processes (Greenlee et al. 2009; Shahmansouri et al. 2015). Within the last decade, closed-circuit desalination (CCD) has emerged as a promising high-recovery membrane process (Efraty et al. 2011; Efraty et al. 2012; Stover 2013) that uses recirculation of retentate to decouple flux, recovery, and crossflow velocity during operation. Compared to single-pass high-pressure membrane systems, a CCD system can achieve comparably high recovery in a significantly reduced footprint, with a lower specific energy requirement, and increased operating flexibility (Lin and Elimelech 2015; Lin and Elimelech 2017; Warsinger et al. 2016). A recent study by Warsinger et al. (2016) suggests that the operational transience that comprises the semi-batch CCD operation is a major factor in reducing the potential for inorganic fouling, even when foulants are concentrated beyond theoretical saturation limits. Considering the potential for significant volumetric recovery improvement, energy/cost savings, and increased operational flexibility, the CCD approach is a feasible alternative to conventional high-pressure membrane applications. In the context of PFAS treatment, high-recovery membrane applications may be beneficial; PFAS residuals could be concentrated, allowing advantages in terms of disposal, e.g., more effective application of PFAS destruction technologies, and several recent studies have evaluated a treatment train approach consisting of RO or NF with retentate based treatment (Liu et al. 2021b).

3.4.2 Materials and Methods

Two pilot-scale membrane systems were used for treatability study testing of RO and NF membranes for removal and concentration of PFAS. The first system, termed single-pass, utilizes three spiral wound membrane elements in series and is operated similarly to most conventional membrane systems. The second is a closed-circuit desalination system (CCD) with an internal recycle stream that allows for high product water recoveries. Both systems and experimental procedures are discussed in more detail below.

Single-Pass 2540 Pilot-scale System. A pilot-scale membrane system (Figures 3.19 and 3.20) containing three 2540 (2.5" diameter × 40" length, 61 mm diameter x 1016 mm length) spiral-wound membrane elements in series was used to evaluate the rejection of PFAS for this project. The system is mainly comprised of a feed pump capable of achieving pressure up to 250 psi, sensors for system monitoring (temperature, flow rate, pressure, conductivity), membrane pressure vessels and associated piping, fittings, and valves. The system was interfaced with a supervisory control and data acquisition system (SCADA) that controlled flow rates (feed and permeate) and monitored system conditions during experiments. Manufacturer specifications require between 1.5 and 4 gallons per minute (gpm; 5.7 – 15 liters per minute) of feed flow into the lead membrane element and no more than 15 percent product water recovery per element. The product water recovery is defined as Eqn (1):

$$\text{Product Water recovery (\%)} = \frac{Q_p}{Q_f} * 100 \quad (1)$$

Where Q_p is volumetric permeate flow rate, and Q_f is volumetric feed water flow rate. Given a maximum single element recovery of 15 percent, the maximum system recovery is 45 percent. To increase system recovery in a full-scale application, multiple membranes are operated in series and in staged arrays. The maximum recovery of a single-pass system is generally assumed to be 85 percent due to various limitations, including reduction of cross-flow velocity and permeate flux, and precipitation of sparingly soluble salts. This system was engineered to simulate the operation of higher recovery single-pass systems. Experiments were conducted on PFAS contaminated groundwater (CO groundwater), and feed water was prepared by spiking AFFF into dechlorinated tap water. The impact of permeate flux and system recovery on PFAS rejection was evaluated using this system. Two membranes were evaluated using this system including the ESPA2 membrane (Hydranautics, Oceanside, CA) and the NF270 membrane (Dupont/Filmtec). All PFAS rejection experiments were maintained at $20 \pm 2^\circ\text{C}$ and operated in batch recirculation mode where the concentrate and permeate were recycled back to the feed tank. Pertinent operating conditions used in experiments are presented in Table 3.8.

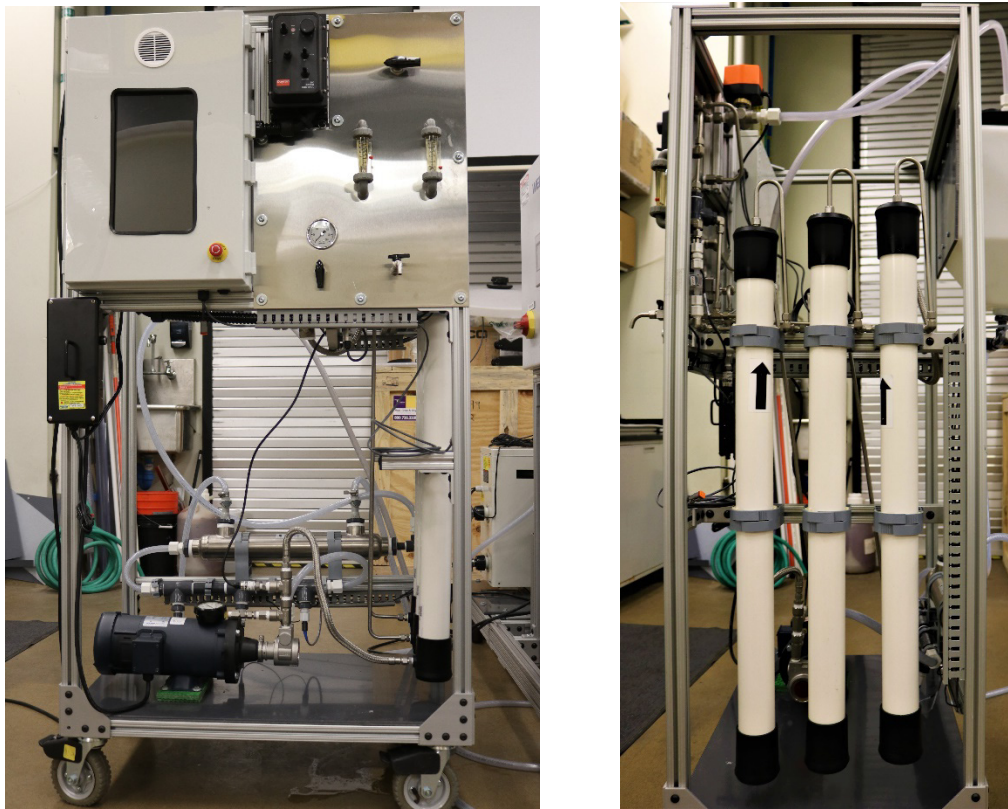


Figure 3.19. Membrane Pilot-scale System Photographs
(left-front side of membrane system; right-membrane elements in series)

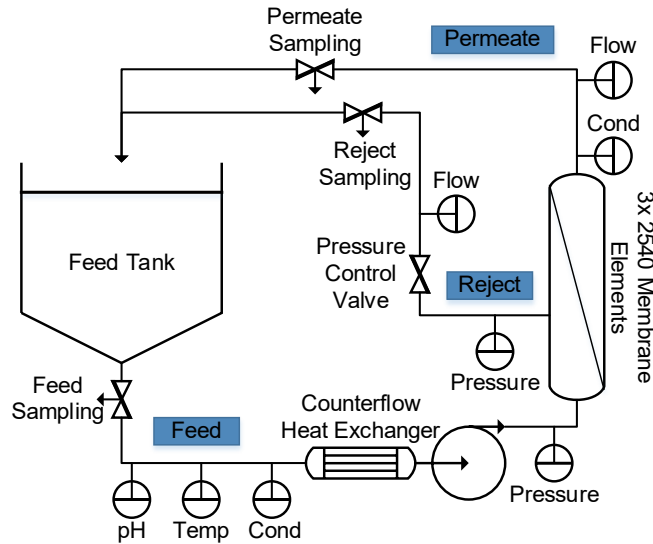


Figure 3.20. Schematic of 2540 Pilot-scale System

Table 3.8. Set Operating Conditions for CCD Membrane Experiments

Feed Flow, Q_f	1.5 – 3.0 gpm	5.7 – 11.2 LPM
Flux	4 - 30 gfd	7 - 50 LMH
Target Recovery	15 - 52% (single pass)	

Closed-Circuit Desalination System. Closed circuit desalination (CCD) is a relatively new development of membrane technology that allows for achievement of higher product recovery compared to standard single-pass membrane systems. Single-pass membrane systems (i.e., conventional membrane systems) utilize spiral membrane elements in series and arranged in arrays to maintain a relatively constant cross-flow velocity throughout the system while achieving relatively high product water recovery. A simplified schematic of a single-pass system is given in Figure 3.21. Due to restrictions on feed-concentrate flow rates in and out of spiral wound elements and single-element recoveries (~15% maximum), single-pass systems are restricted to a total product water recovery of approximately 85%. A membrane system operating at 85% recovery will generate 15% of the feed flow rate as waste requiring disposal. In addition, due to the configuration of single-pass membrane systems, filtration is performed under sub-optimal conditions, particularly in the second or third stage, where solute concentrations are high, and permeate flux is low. It is worth noting that during operation of a single-pass membrane system, permeate flux varies across the length of the membrane system and is relatively high at feed inlet of the system and relatively low at the tail end of the membrane system.

CCD was developed to overcome the recovery limitations of single-pass membrane systems while maintaining filtration conditions more conducive to achieving optimal separation performance. In a CCD system (Figure 3.21), concentrate from membrane system is recirculated internally while feed water is brought into the membrane system. The purpose of this internal recirculation is to maintain constant and high cross-flow velocity during operation and allows for high recovery.

During operation, the feed flow rate is set equal to the permeate flow rate so that, as water permeates the membrane, the internal recirculation loop becomes more and more concentrated. As such, CCD systems are operated in a batch mode where feed water is continuously brought to the membrane until a recovery setpoint is reached. After the recovery setpoint has been reached, the fluid inside the membrane elements on the feed-brine side and the reticulation loop are displaced through a plug-flow rinse and collected in a separate tank. As opposed to single-pass systems, the permeate flux can be more precisely controlled across the membrane system leading to better separation performance. The CCD design theoretically allows for much higher product water recoveries, potentially up to 98%. Product water recovery for CCD systems is calculated differently than single-pass systems and is given by Eqn. (2):

$$\text{Recovery (\%)} = 1 - \left(\frac{V_{\text{system}}}{V_{\text{permeate}} + V_{\text{system}}} \right) * 100 \quad (2)$$

Where, recovery is the percentage of volume recovered for the permeate (V_{permeate} = volume of permeate) produced during each batch or sequence (e.g., gallons) relative to volume of feed water used to flush this system which is related to the dead-volume (V_{system} = the dead-volume) of the membrane system on the feed-brine side of the membrane including the recirculation loop.

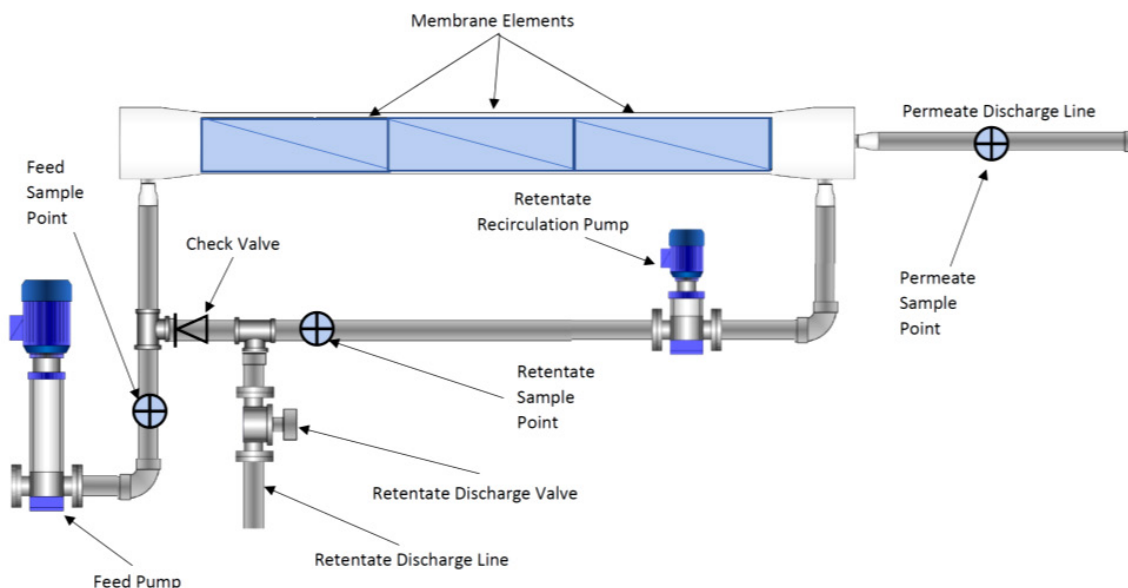


Figure 3.21. Illustration of a CCD-NF/RO System.

Prior to rejection experiments, membrane integrity was first evaluated by operating the system on dechlorinated tap water with normalized operating parameters compared to manufacturer specifications to verify integrity. A summary of operating conditions during experiments is listed in Table 3.9. Following validation of membrane break-in and integrity, high recovery PFAS rejection experiments were initiated. During all experiments feed flow rate was constant at 10 gpm (37.9 LPM) with feed pressure modulated to maintain a constant flux of 12 gfd (20 LMH). Generated permeate was continuously discharged to a 2,000-gallon tank. The retentate recirculation flow rate was set at 25 gpm during all experiments. Membrane experiments were generally conducted over two sequences where each sequence is defined by the period of CCD operation followed by the plug flow flush.

Table 3.9. Set Operating Conditions for CCD Membrane Experiments

Feed Flow, Q_f	10 gpm	37.9 LPM
Retentate Recirculation Flow, Q_r	25 gpm	94.6 LPM
Permeate Flow, Q_p	10 gpm	37.9 LPM
Flux	12 gfd	20 LMH
Target Recovery	97%	
Anti-Scalant Dose	1.6 mg/L	

Membranes Evaluated. Membranes used in this study spanned a range of characteristics from loose NF to tight RO. A summary of relevant properties of membranes evaluated is presented in Table 3.10. The NF270 (loose NF) was selected due to its low pressure and energy requirements and ability to separate organic solutes and multivalent ions from monovalent ions. Amongst the membranes evaluated, the NF90 (tight NF), ESPA2, and CR100 (low-pressure RO) shared similar characteristics, namely, high rejection of NaCl (>98%) and relatively low operating pressure and energy requirements. The SW30 is classified as a seawater RO membrane and can achieve greater than 99% NaCl rejection when treating seawater. As shown in Table 3.10, all membranes have a molecular weight cut-off (MWCO) of less than 180 g/mole, indicating a high degree of separation of organic compounds with molecular weights greater than the cut-off.

Table 3.10. Properties of Membranes Employed in the Study

Membrane:	NF270	ESPA2	NF90	CR100	SW30
Classification	Loose NF	RO	NF	RO	Seawater RO
Manufacturer	DuPont Filmtec	Hydranautics	DuPont Filmtec	DuPont Filmtec	DuPont Filmtec
Active Layer	Polypiperazine Thin Film Composite	Aromatic polyamide	Aromatic polyamide	Aromatic polyamide	Aromatic polyamide
Isoelectric Point	<3	< 4	<4	<3	<3
Membrane material	Polypiperazine Thin Film Composite	Polyamide Thin Film Composite	Polyamide Thin Film Composite	Polyamide Thin Film Composite	Polyamide Thin Film Composite
pH Range	2-11	2-10	2-11	2-11	2-11
MgSO₄ Rejection (%)	97	99%	-	-	-
NaCl Rejection (%)	40	99%	98.7	99.4	99.8
MWCO (g/mole)	180	100	120	-	<100
Membrane Active Area, ft²	400 for 8040 2.6 for 2540	400 for 8040 2.4 for 2540	400 for 8040	400 for 8040	400 for 8040

All information provided by manufacturer besides MWCO
MWCO provided by Drewes et al. 2013

Water Analysis. Samples were collected for analysis in 15 mL polypropylene centrifuge tubes (Falcon; Corning, NY). Samples were analyzed for targeted PFAS using liquid chromatography quadrupole time-of-flight mass spectrometry (LC-QToF-MS). PFAS were measured on a SCIEX X500R QTOF-MS system (Framingham, MA) using electrospray ionization in negative mode (ESI-) with SWATH® Data-Independent Acquisition for both TOFMS and MS/MS mode. Target analytes were identified based on retention time and mass compared to analytical standards and quantified using calibration standards ranging from 0.074 ng/L to 7.4 µg/L, with most analytes having a minimum limit of quantitation (LOQ) between 0.074 ng/L and 7.4 ng/L. The analytical column used was a Phenomenex (Torrance, CA) Gemini C18, 5 µm, 100 mm x 3 mm.

Targeted analysis of nine PFAS compounds was accomplished with authentic reference standards. Precursor ion data was collected for m/z 100-1200 for 1283 cycles with a total scan time of 842 ms and accumulation time of 20 ms, with ion spray voltage set at -4500 V and temperature set to 550 °C. The ion source, curtain, and collision (CAD) gas were set to 60 psi, 35 psi, and 10 psi, respectively. The collision energy was set to -5 V and the declustering potential to -20 V, both with no spread. Product ion (MS/MS) scanning was conducted for m/z 50-1200 Da. The accumulation time for each SWATH window was 50 ms and collision energy was -35 V with 30 V spread. The instrument was mass calibrated every 5 injections using SCIEX ESI Negative Calibration Solution.

Source water quality was characterized by total organic carbon (TOC), anions, cations, and metals. TOC was measured using a carbon analyzer (Shimadzu ion TOC-L, Columbia, MD). Anions were measured using ion chromatography (IC; ICS-900, Dionex, Sunnyvale, CA), while cations/metals were analyzed using inductively coupled plasma optical emission spectroscopy (ICPOES; optima 5300 DV, PerkinElmer, Fremont, CA).

3.5 PFAS REMOVAL BY HIGH-PRESSURE MEMBRANES

During this project, five different membranes were evaluated for the removal of PFAS from several source waters of differing quality. These membranes represent a spectrum of membrane classifications spanning loose NF, tight NF, low-pressure RO, and seawater RO. The following sections detail the major findings related to impact of membrane type and membrane system operating conditions.

3.5.1 Comparison of Membrane Performance

2540 Pilot-scale System. Several different source waters were used in membrane experiments. For experiments with the 2540 pilot-scale system, two feed waters were evaluated including: 1) deionized (DI) water spiked with PFAS and amended with NaHCO₃ and MgSO₄; and 2) PFAS impacted groundwater collected from Peterson AFB. A summary of relevant water quality parameters and PFAS concentrations for these waters are presented in Table 3.11 and Table 3.12, respectively. The ESPA2 (RO) and NF270 (loose NF) were operated in a series of experiments to characterize the rejection of perfluoroalkyl acids (PFAAs) for representative spiral wound elements for the decision support tool. During experiments, the feed and permeate flow rates were varied to achieve a range of operating conditions (Table 3.9) to generate an expected range of rejection values for each membrane and water type. Box and whisker plots (max, min, 25%, 50%, 75% values) summarizing rejection of quantified PFAAs in DI amended feed water by the NF270 and ESPA2 are presented in Figure 3.22. Box and whisker plots summarizing the rejection of quantified PFAAs in groundwater are presented in Figure 3.23.

For the PFAAs investigated and over the range of operating conditions evaluated, the RO membrane (ESPA2) provided higher separation or rejection than the NF membrane (NF270; Figure 3.22 and Figure 3.23). For a DI water matrix, the ESPA2 provided greater than 99% rejection for all PFAAs, with permeate concentrations being near or below the limit of quantification. While higher permeate PFAA concentrations were detected in NF270 permeate relative to the ESPA2, for the DI water matrix, the NF270 provided 98% or greater rejection of PFAAs evaluated. In general, an increase in rejection was observed for increasing molecular size within each sub-class of PFAAs (PFCAs and PFSA). Treatment of PFAS impacted groundwater resulted in a wider range of rejection values for both membranes evaluated (ESPA2, NF270; Figure 3.23) and particularly for the NF270, lower rejection. However, the rejection of investigated PFAAs by the ESPA2 membrane generally exceeded 98%, with the NF270 providing greater than 90% rejection. The impact of ionic strength on the rejection was further evaluated using the CCD system and is discussed below.

Table 3.11. Dissolved Constituents in DI and Contaminated Groundwater Used in 2540 Pilot-scale Membrane Experiments.

	AFFF Spiked DI (pilot-scale)	AFFF GW (pilot-scale system)
Analyte	mg/L	mg/L
Sodium (Na)	92 (amended)	56.2 ± 2.1
Potassium (K)	-	6.4 ± 0.1
Calcium (Ca)	-	250.4 ± 6.9
Magnesium (Mg)	20 (amended)	50.4 ± 1.3
Chloride (Cl)	-	154.0 ± 4.2
Sulfate (SO ₄)	80 (amended)	456.6 ± 4.3
Dissolved organic carbon	< 0.3	5.4 ± 0.3
pH (standard units)	7 ± 0.4	7 ± 0.4

Table 3.12. Measured PFAS Concentrations in DI and Contaminated Groundwater Used in 2540 Pilot-scale Membrane Experiments.

PFAA	Acronym	MW	AFFF Spiked DI	AFFF GW
		(g/mol)	(ng/L) n=6	(ng/L) n=6
Perfluoropentanoic acid	PFPeA	264	237±20	664±32
Perfluorohexanoic acid	PFHxA	314	894±82	1160±70
Perfluoroheptanoic acid	PFHpA	364	262±20	191±13
Perfluorooctanoic acid	PFOA	414	708±49	153±8
Perfluoropropane sulfonate	PFPrS	250	778±91	248±11
Perfluorobutane sulfonate	PFBS	300	1650±170	743±33
Perfluoropentane sulfonate	PFPeS	350	3480±560	754±59
Perfluorohexane sulfonate	PFHxS	400	12500±1100	2700±200
Perfluoroheptane sulfonate	PFHpS	450	1250±260	40±2
Perfluorooctane sulfonate	PFOS	500	58400±1700	139±10

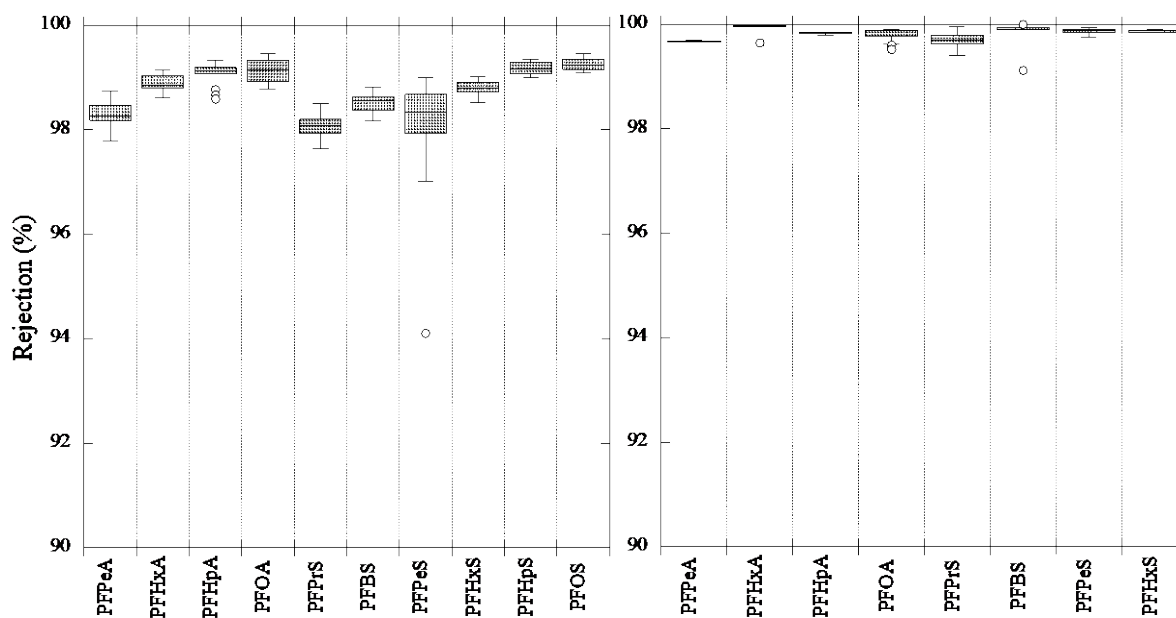


Figure 3.22. Rejection of PFAAs by the NF270 Membrane (Left) and the ESPA2 Membrane (Right) During PFAA Amended DI Feed Water Experiments.

(36 permeate samples were collected during triplicate experiments over a range of flux and recovery setpoints.)

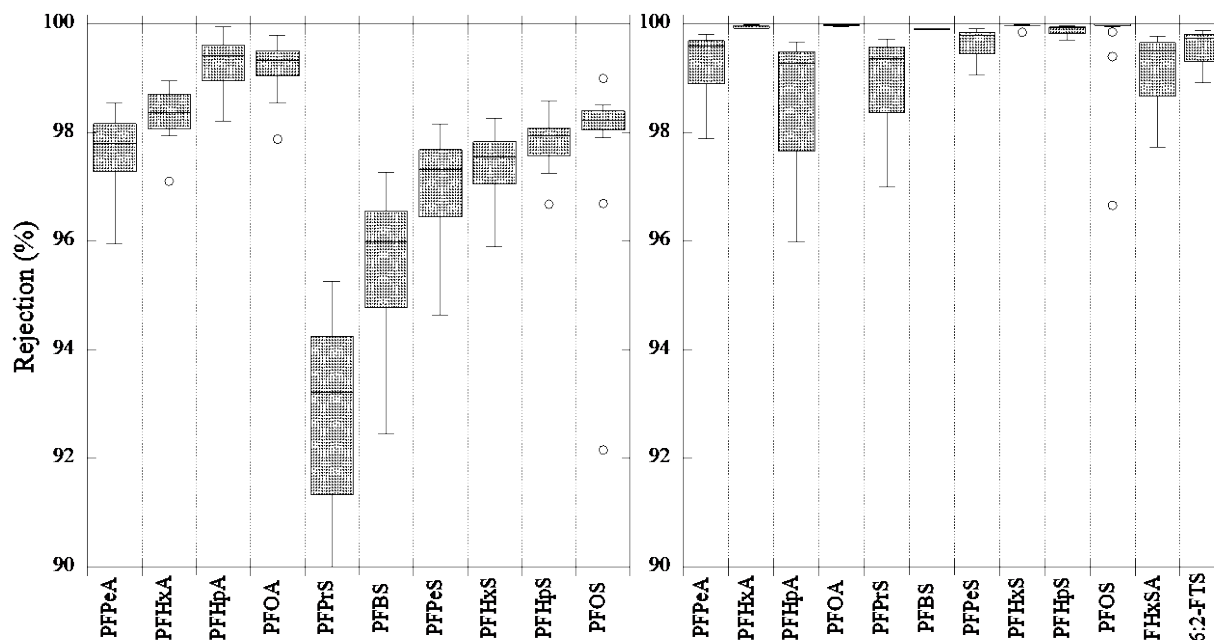


Figure 3.23. Rejection of PFAAs by the NF270 Membrane (Left) and the ESPA2 Membrane (Right) During AFFF Contaminated Groundwater Experiments.

(36 permeate samples were collected during triplicate experiments over a range of flux and recovery setpoints.)

CCD System. Four membranes (NF270, NF90, CR100, SW30) were evaluated using the CCD system using dechlorinated potable water (Denver Water, CO) spiked with a C8 based AFFF product. Relevant water quality of feed water and PFAA concentrations are presented in Table 3.13 **Error! Reference source not found.** and Table 3.14. Feed PFAA concentrations presented in Table 3.14 represent average values taken over four separate experiments with the different membranes.

Permeate samples collected over each CCD sequence as a function of recovery (50 to 97%) were analyzed and used with measured feed concentrations to calculate rejection values. Rejection values for each membrane were then developed into box and whisker plots presented in

to illustrate ranges of rejection for each PFAA. The NF270 operating under CCD mode was found to have highly variable rejection ranging from 20 to approximately 98% depending on the PFAA and the recovery at which permeate samples were collected. Because potable water was used as feed make-up water, it is hypothesized that the increase in ionic strength with increasing recovery had a negative impact on rejection as rejection values significantly decreased as a function of recovery (see sections 3.5.2 and 3.5.3 on impact of recovery and ionic strength on PFAA rejection). Compared to the NF270 membrane, the NF90 (tight NF), CR100 (low-pressure RO), and SW30 (seawater RO) provided significantly higher rejection, with all three membranes generally achieving 99% or greater rejection of all PFAAs regardless of recovery.

Table 3.13. Average Feed Water Quality for CCD Experiments

Parameter	Unit	Average (n=8)	Std Dev
pH	-	8.5	0.07
Alkalinity	mg/L as CaCO ₃	57	7
TOC	mg/L	2.96	0.74
Fluoride	mg/L	0.65	0.14
Chloride	mg/L	24.01	3.21
Sulfate	mg/L	88.23	13.72
Barium	mg/L	0.044	0.005
Calcium	mg/L	25.80	8.66
Potassium	mg/L	2.15	1.46
Magnesium	mg/L	5.64	1.24
Sodium	mg/L	20.40	2.64

Table 3.14. Measured PFAA Feed Water Concentrations and Limits of Quantification (LOQ) for Sample Types Collected during CCD Experiments.

	PFAS Analyte (n=36)									
	PFPeA	PFHxA	PFHpA	PFOA	PFPrS	PFBS	PFPeS	PFHxS	PFHpS	PFOS
Avg Feed (ng/L)	218	649	146	643	356	798	984	3824	318	20151
Std. Dev.	52	133	33	188	23	80	298	364	140	4675
Permeate LOQ ^a	2	4	2	2	1	1	1	1	1	4
Feed LOQ ^a	20	40	20	20	10	10	10	10	10	40
Retentate LOQ ^a	200	350	200	225	100	575	100	100	125	350

a. LOQ presented in ng/L units and vary due to sample dilutions.

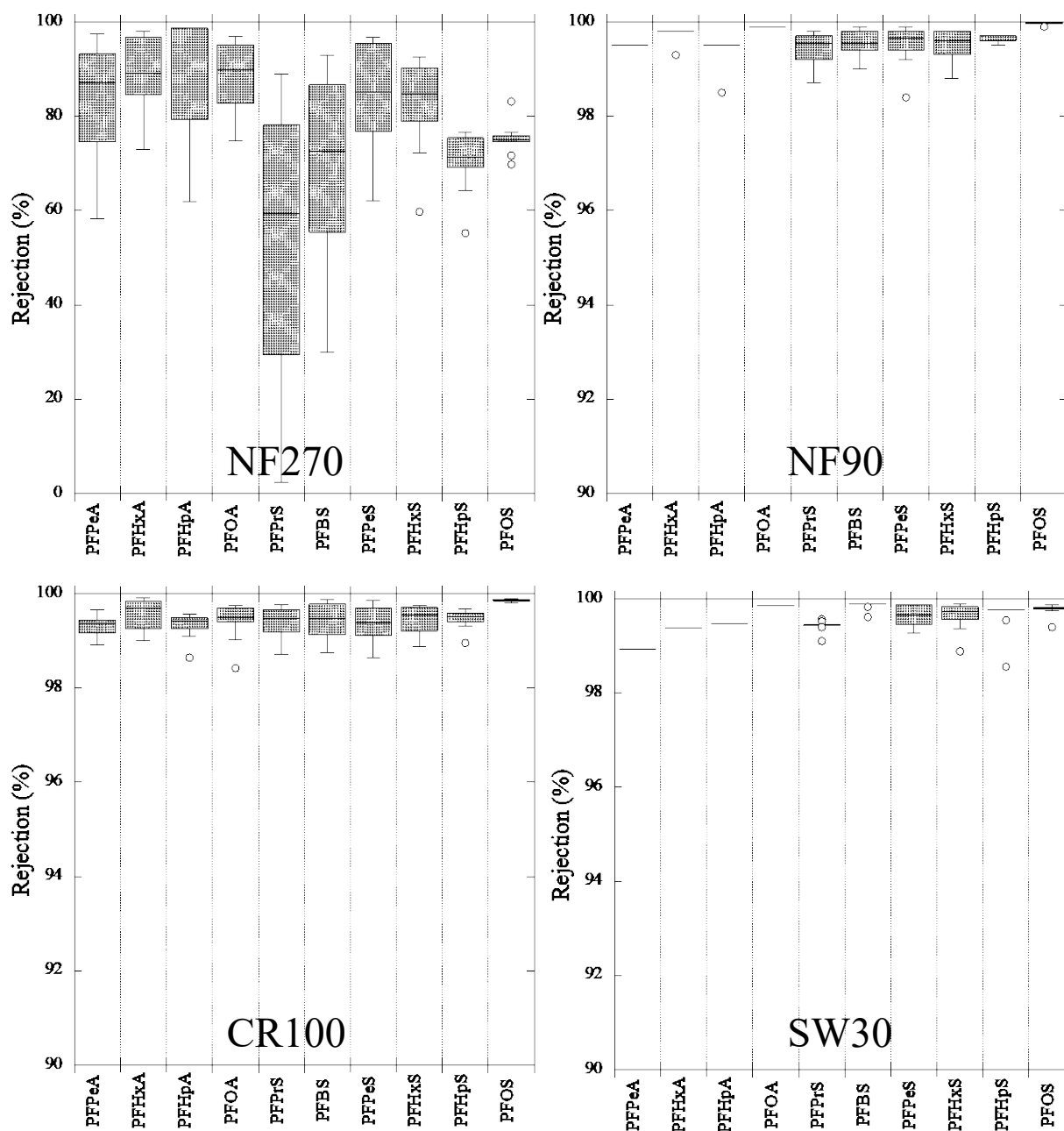


Figure 3.24. Rejection of PFAAs by the NF270 Membrane (Top Left), NF90 membrane (Top Right), CR100 Membrane (Bottom Left), and SW30 (Bottom Right) Measured during CCD Experiments

(10 permeate samples were collected for duplicate CCD sequences over a range of recovery setpoints (5 to 97% recovery). Boxes represented by lines had permeate concentrations below or at the LOQ for the given PFAA (for example NF90 and SW30 PFPeA data).

In addition to collecting permeate samples at defined recovery setpoints, samples were collected from the final permeate produced after two CCD sequences. Rejection values (Figure 3.25) and final permeate PFAA concentrations (Table 3.15) represent the effectiveness of the membrane as a barrier to PFAAs over two sequences up to 97% recovery. Overall, the tight NF membrane (NF90) provided greater than 2-log removal of all PFAAs evaluated, with the tight RO membrane (SW30) providing similar separation performance. The low-pressure RO membrane evaluated (CR100) provided a similar separation of PFAAs to the NF90 and SW30. Compared to other membranes evaluated, the NF270 was not an effective barrier for PFAAs evaluated in the CCD configuration.

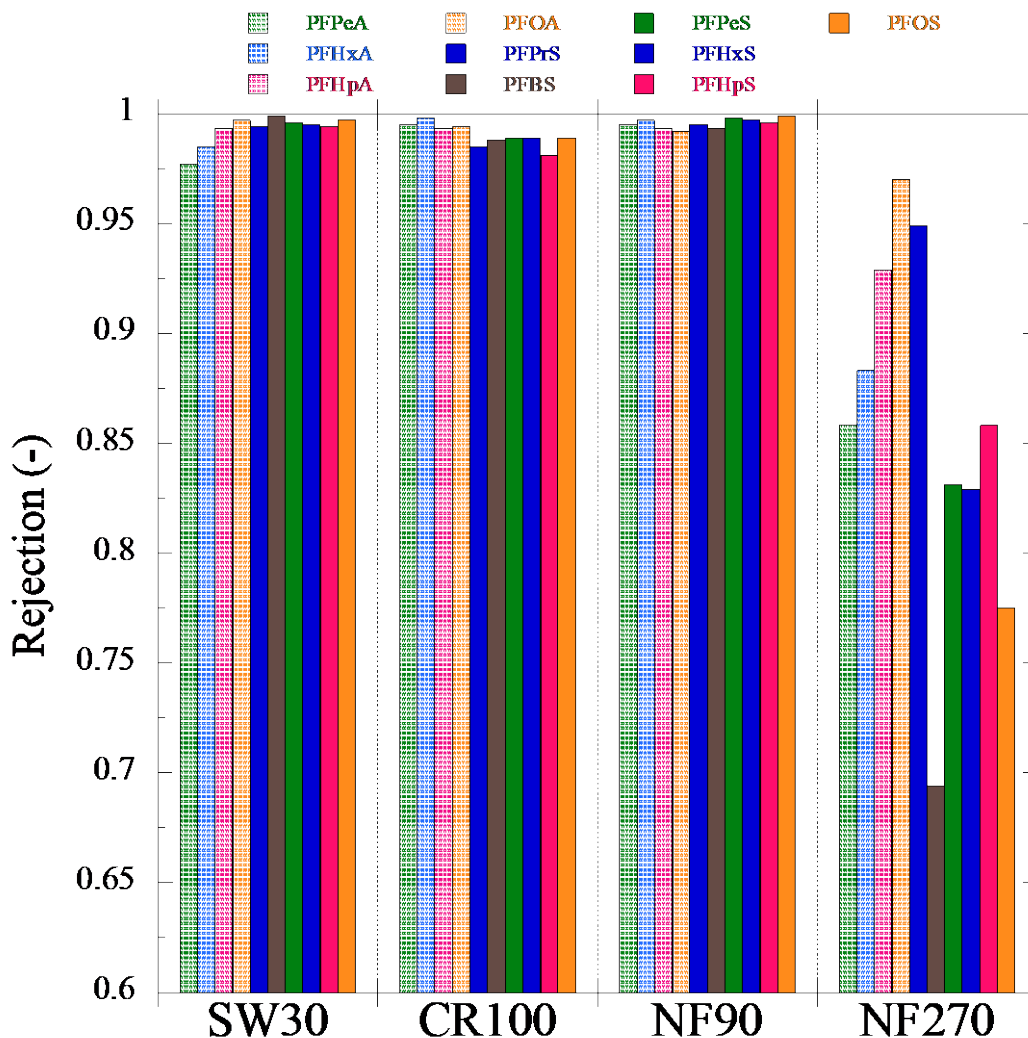


Figure 3.25. Overall Rejection during CCD Experiments Up to 97% Recovery.

Rejection was calculated using the samples collected from the combined permeate tank after two sequences had concluded.

Table 3.15. Feed and Combined Permeate PFAA Concentrations and Calculated Rejection for CCD Experiments Up to 97% Recovery

PFAS	PFPeA	PFHxA	PFHpA	PFOA	PFPrS	PFBS	PFPeS	PFHxS	PFHpS	PFOS
Average Feed Tank (ng/L)	218	649	146	643	356	798	984	3824	318	20151
Permeate SW30 (ng/L)	5	10	1	2	2	1	4	19	2	58
Overall SW30 Rejection (-)	0.977	0.985	0.993	0.997	0.994	0.999	0.996	0.995	0.994	0.997
Permeate CR100 (ng/L)	1	1	1	4	5	10	11	44	6	214
Overall CR100 Rejection (-)	0.995	0.998	0.993	0.994	0.985	0.988	0.989	0.989	0.981	0.989
Permeate NF90 (ng/L)	1	2	1	5	2	6	2	12	1	28
Overall NF90 Rejection (-)	0.995	0.997	0.993	0.992	0.995	0.993	0.998	0.997	0.996	0.999
Permeate NF270 (ng/L)	31	76	10	19	18	245	166	655	45	4531
Overall NF270 Rejection (-)	0.858	0.883	0.929	0.970	0.949	0.694	0.831	0.829	0.858	0.775

3.5.2 Membrane System Permeate Flux and Recovery

System Recovery. Experiments were conducted using the two membrane systems used during this study to evaluate the impact of system or product water recovery on PFAS rejection. For a conventional staged membrane system (i.e., the 2540 pilot-scale system), the impact of recovery on rejection is difficult to evaluate because, for a fixed feed flow rate, as recovery increases, permeate flux must also increase. These two system parameters (i.e., flux; recovery) tend to offset one another as is shown in Table 3.26. For example, for a system operating at a fixed recovery, increasing permeate flux (increasing permeate flow rate) results in dilution of solutes diffusing through the semi-permeable membrane and an increase in rejection. For a system operating at a fixed permeate flux, increasing recovery results in an increase in solute concentration and diffusion, which decreases rejection.

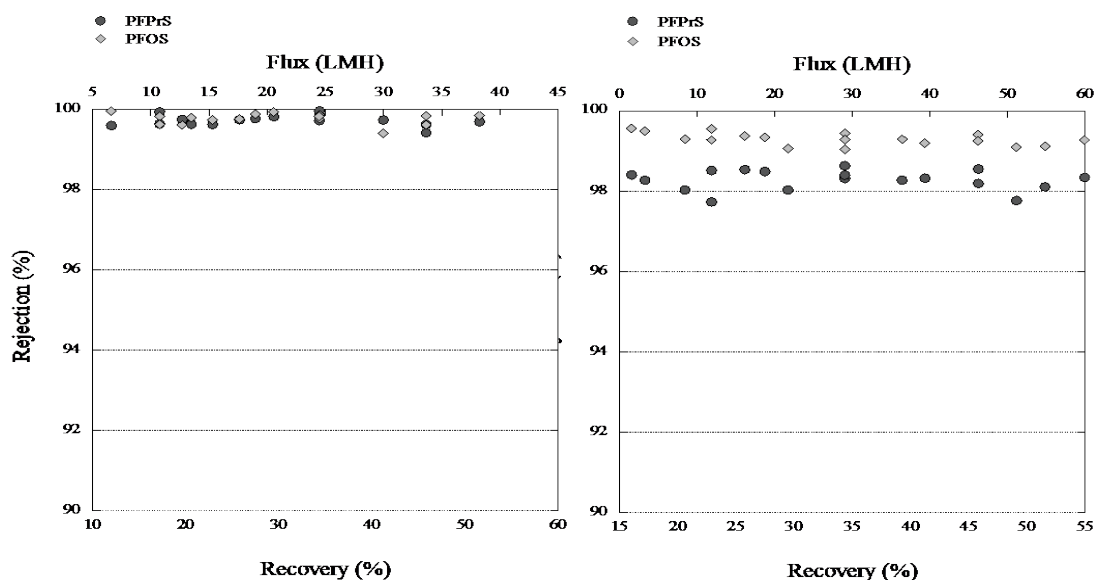


Figure 3.26. Rejection of Selected PFAAs (PFPrS, PFOS) by the ESPA2 (Left) and NF270 (Right) in PFAA Amended DI Water as a Function of Recovery and Permeate Flux.

2540 pilot-scale membrane experiments were subsequently conducted to evaluate the impact of recovery on PFAA rejection by concentrating feed water in the feed tank. This was accomplished by recirculating the concentrate stream back to the feed water tank while discharging the permeate. These experiments were performed with AFFG GW (Tables 3.11 and 3.12) with recovery evaluated up to 95-97%. During high recovery experiments with the NF270, rejection decreased significantly for several PFAAs above recovery values of 80% (Figure 3.27, left). Similar experiments with the ESPA2 showed less of an impact of recovery on rejection (Figure 3.27, right) with most PFAAs at concentrations below limits of quantification. Although a decrease in rejection was observed for AFFG GW in single-pass experiments (recovery between 5 and 50%) with the NF270 (Figure 3.23) compared to amended DI water (Figure 3.22), the much more significant drop in PFAA rejection was evaluated further using the CCD system.

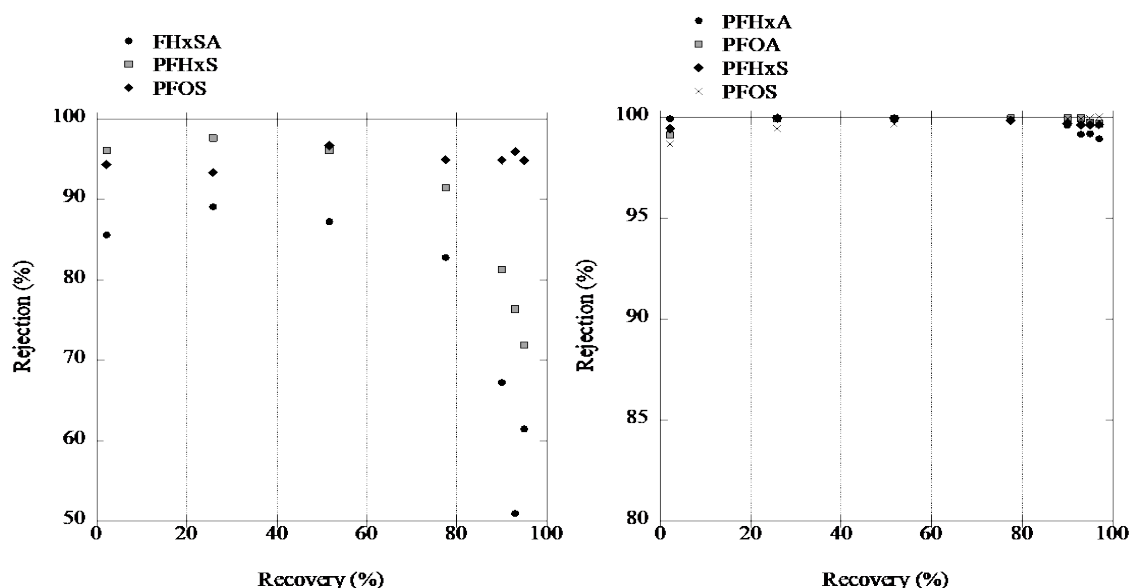


Figure 3.27. Rejection of selected PFAAs by the NF270 (left) and ESPA2 (right) during High Recovery Experiments Treating AFFG GW.

Note that y-axis scales are different between figures.

To evaluate the impact of recovery of the rejection of PFAS, samples were collected from the permeate and concentrate lines of the CCD system at discrete recoveries throughout a high-recovery (up to 97%) sequence. Similar to the results presented in Figure 3.27, high recovery experiments demonstrated a significant decline in PFAA separation by the loose NF membrane evaluated (NF270), specifically at recovery values greater than 90% (Figure 3.28). Although this phenomenon was observed for the other membranes employed during CCD experiments (NF90, CR100, SW30), the decrease in rejection was much less pronounced (Figure 3.29 provides a summary of selected CR100 PFAA rejection data). Given the significant detrimental impact of recovery on PFAA rejection by the NF270 and the observation that PFAA rejection was lower in a groundwater matrix compared to amended deionized water, experiments were conducted to evaluate the impact of ionic strength of PFAA rejection by the NF270.

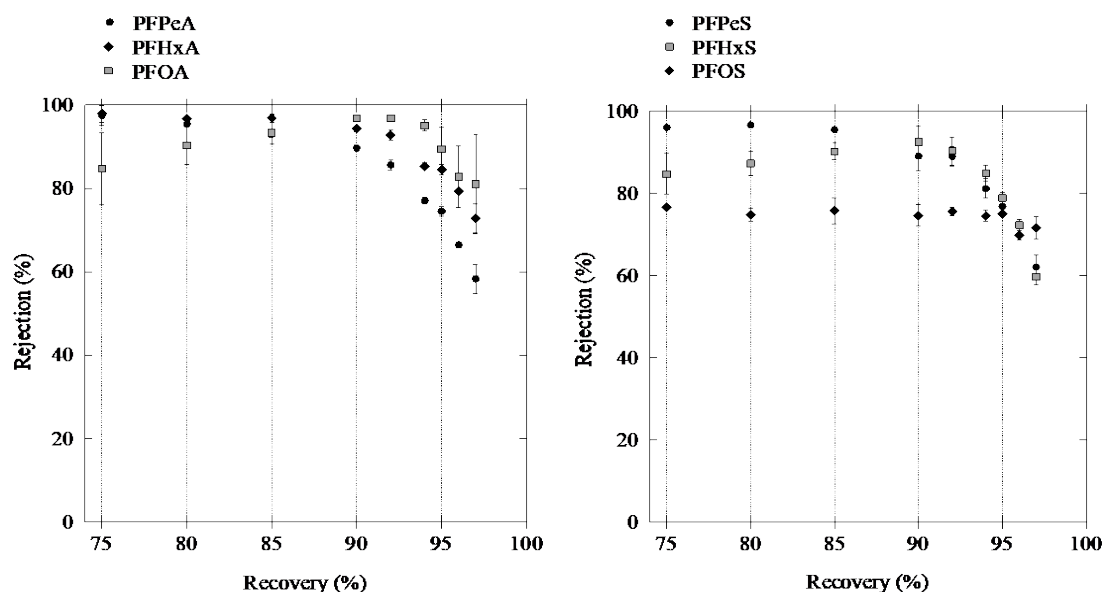


Figure 3.28. Rejection of Selected PFCAs (Left) and PFSA (Right) by the NF270 as a Function of Recovery during CCD Experiments

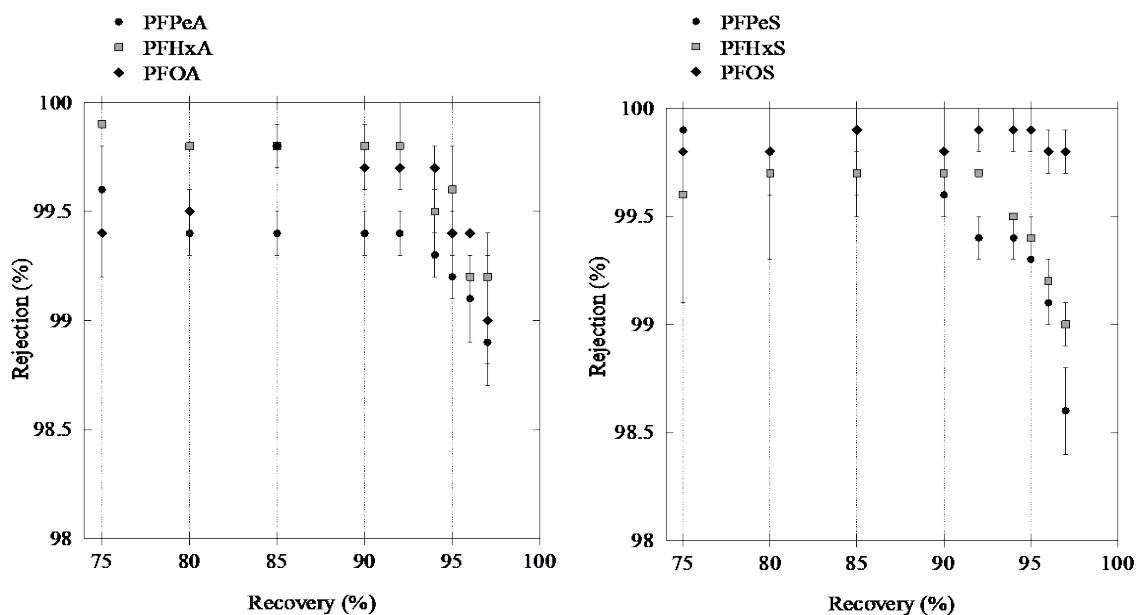


Figure 3.29. Rejection of selected PFCAs (left) and PFSA (right) by the CR100 as a function of recovery during CCD experiments.

3.5.3 Ionic Strength.

The impact of ionic strength on rejection of ionic solutes by high-pressure membranes has been evaluated in several past studies (Bellona and Drewes 2005; Verliefde et al. 2008; Ozaki and Li 2002). The impact of ionic strength of PFAA separation was indirectly evaluated during this study in high-recovery experiments as ions became more concentrated as recovery was increased.

Given the relatively large molecular weights of PFAAs however, past research suggests that ionic strength may only marginally impact their separation by the NF270 and to a much lesser extent, the NF90, CR100, and SW30. However, as was observed during high-recovery CCD experiments, PFAA rejection by the NF270 was significantly impacted at high recovery. Several additional experiments were conducted to further investigate the impact of ionic strength by the NF270, including 1) addition of 500 g CaCl_2 to 1000 gallons of dechlorinated tap water amended with AFFF; and 2) addition of 500 g Na_2SO_4 to 1000 gallons of dechlorinated tap water amended with AFFF.

The 500-gram supplement of sodium sulfate represented the addition of approximately 0.85 mM of sulfate ion concentration and approximately 1.7 mM of sulfate ionic strength. Sulfate salt was selected for addition to the feedwater because it is a large divalent co-ion to the PFAA anions. Sulfate is typically well rejected by the NF270 membrane and is expected to accumulate on the feed-retentate side of the system during CCD operation. The addition of both ionic strength and a PFAA co-ion is expected to have a two-fold effect. The addition of ionic strength is expected to increase PFAA flux by reducing the time required to sufficiently suppress the repulsive electrostatic forces between the negatively charged membrane surface and the PFAA anions. Second, the high retention of a large divalent PFAA co-ion on the feed-retentate side of the membrane might induce increased Donnan mediated migration of more mobile, shorter-chain PFAAs across the membrane. By comparing the rejection as a function of recovery between the baseline experiment and the sulfate-amended experiment, it is apparent that the addition of sodium sulfate negatively impacted the rejection of PFBS, PFOA, and PFOS by the NF270 (Figure 3.30). Compared to the baseline experiment, the average rejection significantly decreased by >30% for PFBS, ~20% for PFOA, and by ~40% for PFOS at 97% water recovery during the sodium sulfate amended experiment.

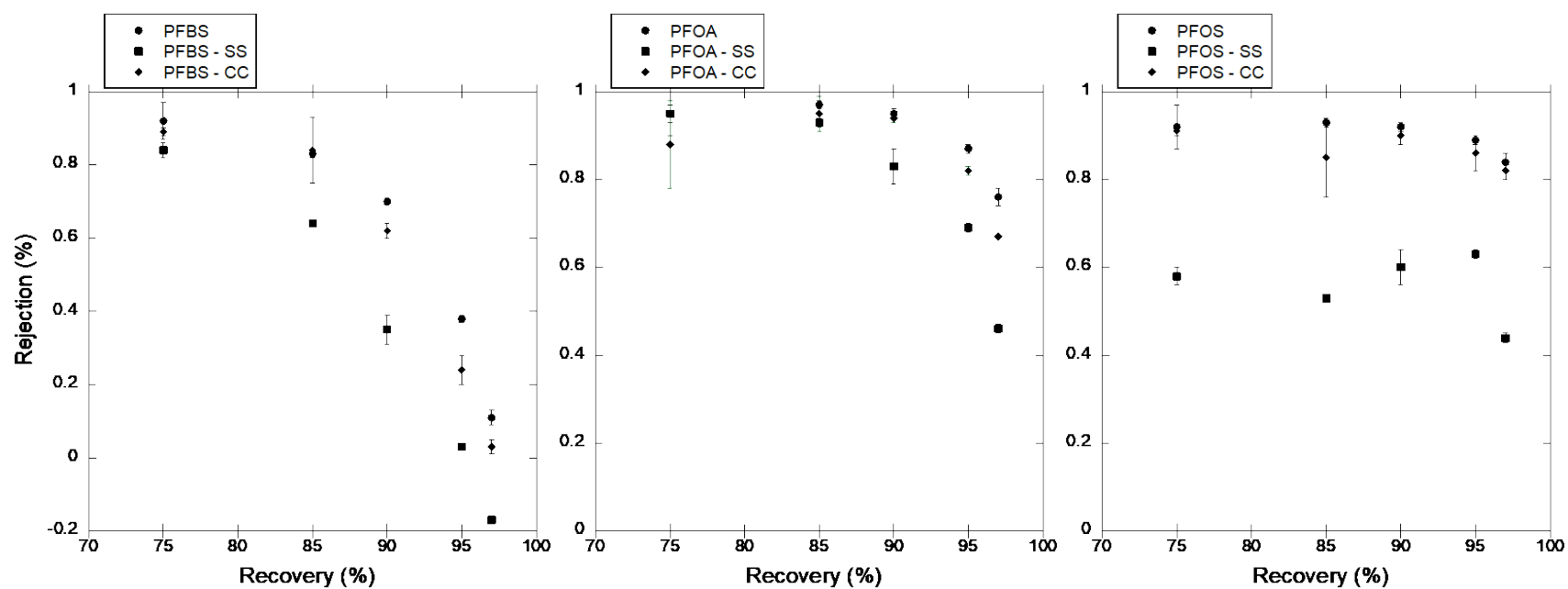


Figure 3.30. Rejection of Selected PFAAs by the NF270 for No-ionic Strength Addition Experiments (Baseline), Addition of Na₂SO₄ (SS) Experiments, and Addition of CaCl₂ (CC) experiments.

The 500-gram supplement of calcium chloride (CaCl_2) represented the addition of approximately 1.2 mM of calcium ion concentration and approximately 2.4 mM of calcium ionic strength. Calcium salt was selected because it is a large divalent counter-ion to the PFAA anions, and chloride is a mobile co-ion of the PFAAs. Calcium is relatively well rejected by the NF270 membrane and is expected to accumulate on the feed-retentate side of the system during CCD operation. The addition of both ionic strength and a PFAA counter-ion is hypothesized to impact the PFAA rejection in several ways. The addition of calcium is anticipated to shrink the electrical double layer, potentially allowing PFAAs to approach the membrane more readily, thereby increasing PFAA permeation. However, given that 2.4 mM of chloride was added, and chloride is more mobile than PFAAs (based on diffusion coefficients), chloride is expected to permeate the membrane more readily to preserve electroneutrality. The addition of a counter-ion to PFAAs (Ca^{2+}) may also promote PFAA complexation, causing an increase in molecular volume and a decrease in PFAA permeation. Zhao et al. also postulated that divalent calcium could serve as an “ion bridge” between the monovalent anionic membrane surface and PFAA functional group charges (Zhao et al. 2016). Compared to the addition of sodium sulfate, calcium chloride had a minimal impact on the rejection of PFAAs with separation being marginally (5-10%) lower than baseline conditions across the recovery setpoints evaluated (Figure 3.30).

3.5.4 PFAS Concentration and Adsorption

PFOS rejection was observed to be largely independent of permeate flux and recovery during experiments with different feed water and membranes (Figures 3.26, 3.27, 3.28, 3.29, and 3.30), which may be a result of concentration dependent behavior. Although the critical micelle concentration (CMC) of PFOS has been reported between 698 and 5,000 mg/L, it has been demonstrated that some degree of self-aggregation can occur at concentrations in the range of 0.001 and 0.01 of the CMC (Yu et al. 2009; Shih et al. 2013; Johnson et al. 2007; Costanza et al. 2019). At approximately 90% recovery, the average feed-retentate PFOS concentration was measured at approximately 0.132 mg/L, and at 97% recovery approximately 0.382 mg/L, significantly lower than the reported CMCs (Figure 3.31). However, the CMC of a given PFAA can be a function of several parameters, including solution pH, PFAA chain length, and PFAA counter ions. Additionally, concentration polarization at the membrane surface driven, in part, by the increased hydrophobicity and lower charge density of the PFOS molecule may have increased the PFOS concentration near the membrane surface, above the measured bulk solution concentrations. It is possible amphiphilic properties promoted the self-aggregation of PFOS, potentially resulting in increased supramolecular size (via aggregation) or the formation of a PFOS “gel” phase at the membrane surface.

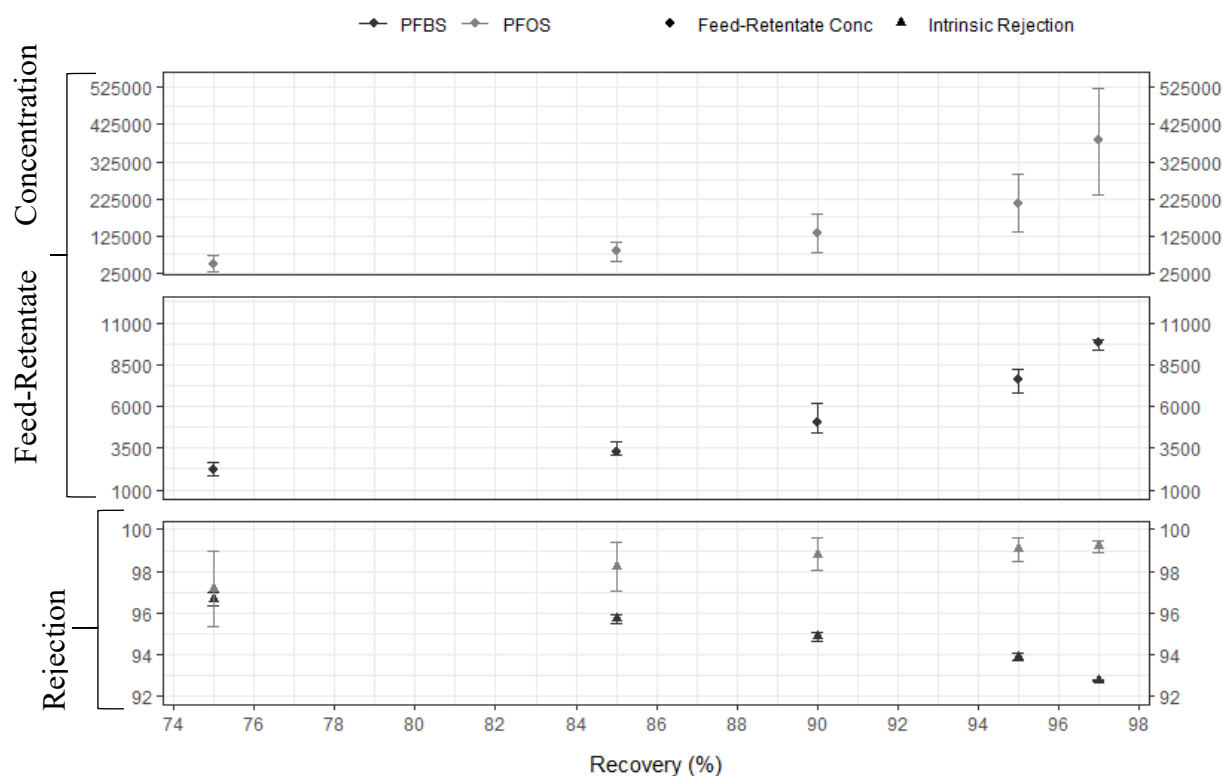


Figure 3.31. Observed Rejection (%) and Feed-retentate Concentration (ng/L) of PFOS and PFBS as a Function of Recovery.

*Data presented in **Error! Reference source not found.** incorporates a duplicate experiment, and error bars are deviations from average values calculated from duplicate experiments.*

Adsorption experiments were conducted by dismantling a spiral-wound NF-270 module and soaking three components in a solution of 1-L solution of AFFF Spiked DI water (Tables 3.11 and 3.12) and shaken for two-weeks: 1) membrane (0.56 cm²); 2) feed spacer (0.56 cm²); and 3) permeate spacer (0.56 cm²). Of the PFAAs evaluated, PFASs were observed to adsorb to components the most (Figure 3.32), with the permeate spacer adsorbing the most PFAS mass. While these experiments were performed in static equilibrium tests that do not mimic dynamic conditions of operating membrane systems, results indicate that PFOS could be accumulating at the membrane surface, resulting in better rejection during high-recovery experiments.

3.5.5 Comparison of Membrane Types Evaluated

Development of the decision support tool required inputting default rejection values for high-pressure membranes. PFAS rejection data generated during the project was summarized in Table 3.16 to provide conservative estimates of removal performance for categories of membrane products evaluated. For rejection values, the 25th and 75th percentile of rejection data generated during experiments described in previous sections were calculated and used for ranges of expected values across a wide range of operating conditions (e.g., permeate flux, recovery, feed water matrix).

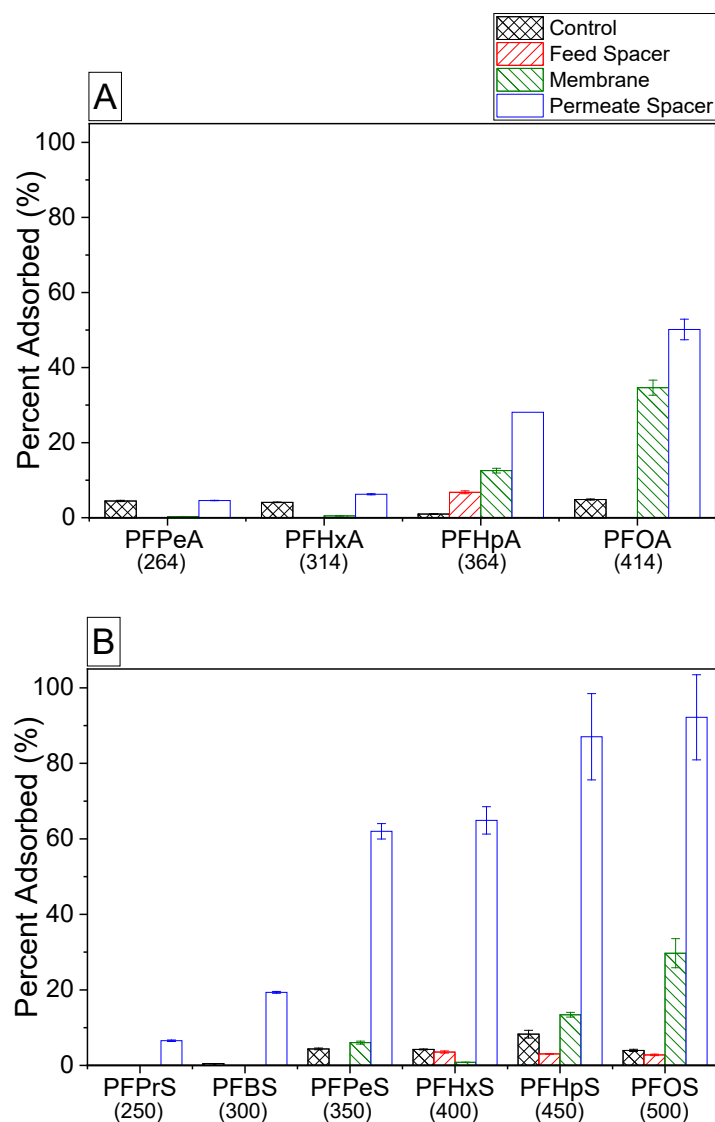


Figure 3.32. Loss of PFAS Mass (Percent Adsorbed) for (a) PFCAs and (b) PFSA in AFFF Amended Deionized Water Measured in Batch Suspensions of Different Membrane Spiral Wound Element Components.

Conditions for each experiment included: 0.56 m² of each membrane component was placed in 1L of solution over 13 days equilibration. Molecular weights are shown in parenthesis.

Table 3.16. Summarized Rejection Values for the Membrane Types Evaluated

PFAS	Abbreviation	Loose NF		Tight NF		Low-Pressure RO		Tight RO	
		25 th Percentile	75 th Percentile	25 th Percentile	75 th Percentile	25 th Percentile	75 th Percentile	25 th Percentile	75 th Percentile
Perfluorobutanoic acid	PFBA	41.7	77.9	N.A.	N.A.	N.A.	N.A.	97.8	99.9*
Perfluoropentanoic acid	PFPeA	76.5	94.4	99.5*	99.5*	99.2	99.7	98.9*	98.9*
Perfluorohexanoic acid	PFHxA	85	96.2	99.3	99.6*	99.4	99.9	99.4*	99.4*
Perfluoroheptanoic acid	PFHpA	82.8	98.7	99.5*	99.5*	94.8	99.9	99.5*	99.5*
Perfluorooctanoic acid	PFOA	84.2	96.9	99.2	99.3*	99.4	99.9	99.9*	99.9*
Perfluoropropane sulfonate	PFPrS	35.6	84.4	99.2	99.7	94.8	99.7	99.4	99.9
Perfluorobutane sulfonate	PFBS	67.3	90	99.3	99.8	99.3	99.8	99.9*	99.9*
Perfluoropentane sulfonate	PFPeS	72	92.6	99.5	99.8	98.4	99.8	99.5	99.9
Perfluorohexane sulfonate	PFHxS	75.5	88.2	99.5	99.8	99.3	99.7	99.6	99.8
Perfluoroheptane sulfonate	PFHpS	71.5	80.3	99.5	99.9	99.3	99.9	99.8*	99.8*
Perfluorooctane sulfonate	PFOS	72.7	75.1	99.96	99.99	99.4	99.9	99.9	99.9
Perfluorohexane sulfonamide	FHxSA	63.6	83.7	N.A.	N.A.	96.3	99.3	N.A.	N.A.
6:2-Fluorotelemer sulfonate	6:2-FTS	85	96.1	N.A.	N.A.	98.2	99.6	N.A.	N.A.

*Asterisk denotes values that used limits of quantification to calculate rejection values when permeate samples were less than LOQ

3.5.6 Advantages and Disadvantages of High-Pressure Membranes

High-pressure membrane systems are commonly used for various industrial and municipal drinking water applications, including separation of solutes, reclamation of impaired water resources (e.g., wastewater, seawater, brackish water), and as a barrier for inorganic and organic contaminants. Therefore, NF and RO products and systems are commercially available, have high technological maturity, and there is a significant amount of familiarity and knowledge regarding their implementation. There are currently a wide variety of membrane products spanning different MWCOs, permeabilities, and sizes, allowing for selection of membrane products for a given application. Advancements in membrane manufacturing over the past several decades have significantly reduced the energy requirements of NF and RO and led to the development of low-fouling membrane products and high-recovery membrane systems.

Compared to other PFAS removal processes, e.g., GAC and IX, high-pressure membrane systems are generally more complicated requiring effective pretreatment, requiring significantly more energy to operate, and are generally more capital expensive. However, compared to adsorbent systems, high-pressure membranes serve as a continuous barrier to PFAS (at pre-determined rejection or separation levels) without issues of breakthrough. Because high-pressure membranes are a separation process, PFAS and other constituents are concentrated into a retentate or concentrate stream requiring further treatment and/or disposal. Concentrate management is one of the most divisive issues impeding the widespread adoption of high-pressure membrane treatment systems in areas without concentrate disposal options (e.g., discharging to the ocean). There are currently significant research efforts towards the development of high-recovery membrane systems and/or concentrate minimization technologies. In the case of PFAS, concentration processes (e.g., high recovery membrane systems, foam fractionation) have merit as residual streams can be concentrated, allowing for more efficient operation of PFAS destruction technologies.

High-pressure membrane systems are also prone to operational upsets caused by fouling (adsorption of components to membrane polymer) and scaling (precipitation of minerals on the membrane surface). Effective pretreatment is needed for membrane systems generally consisting of pre-filtration (usually down to at least 1 μm), pH adjustment (between 6 and 6.5), and addition of antiscalants to reduce scaling. However, additional pretreatment may be necessary, including removal of iron, manganese, calcium, and magnesium as well as reduction of organic matter prior to filtration. Additionally, permeate produced by high-pressure membranes has low ion content and generally low pH, necessitating stabilization prior to introduction to a pipe network (e.g., drinking water distribution system) to prevent corrosion.

4.0 PFAS TREATMENT TECHNOLOGY DECISION SUPPORT TOOL

The PFAS treatment technology support tool provides a pathway to comprehensively evaluate PFAS treatment technology options. The tool quantifies each option's relative environmental impacts, costs, and facility requirements for a given scenario. Each scenario is defined by a specific water quality (PFAS concentrations, TOC, pH) and quantity (0.03 to 75 million gallons per day); the functional unit, following ISO 14040 life cycle assessment (LCA) methodology (International Standard ISO 14040) to ensure comprehensive and accurate comparison, is the treatment of that water over a 40-year lifetime.

Each treatment option's relative environmental impacts, costs, and facility requirements are quantified as 15 decision criteria. Each criterion has its own scoring (i.e., quantification) method (Table 4.1) and was selected based on stakeholder input, common issues, literature, and by considering all three pillars of sustainability. For a given scenario and treatment option, the 15 criteria scores are aggregated into one overall performance score using multi-criteria decision analysis (MCDA) methodologies. Since the final performance score for each option is calculated using weights provided by the tool user and therefore reflects their priorities and which decision criteria are important in their decision, the option with the lowest aggregated score is the option recommended for that scenario and user. This framework (Figure 4.1) was implemented in an Excel spreadsheet to provide a user-friendly tool.

Table 4.1. Decision Criteria with Scoring Method and Units.

<i>Category</i>	<i>Decision Criteria</i>	<i>Scoring Method</i>	<i>Units</i>
Affordability	Capital Cost	Total initial costs	\$
	O&M Cost	Total operation and maintenance costs	\$
Facility Operation	Footprint	Treatment process footprint	sq. ft.
	Maintenance	Total number of maintenance hours per year	hrs/yr
	Waste Generation	Volume of residual and liquid waste generated per year	gal/yr
Global Warming	Global warming	TRACI	kg CO2 eq
Global Environmental and Human Health	Ozone depletion	TRACI	kg CFC-11 eq
	Smog	TRACI	kg O3 eq
	Acidification	TRACI	kg SO2 eq
	Eutrophication	TRACI	kg N eq
	Ecotoxicity	TRACI	CTUe
	Fossil fuel depletion	TRACI	MJ surplus
	Carcinogenics	TRACI	CTUh
	Non carcinogenics	TRACI	CTUh
	Respiratory effects	TRACI	kg PM2.5 eq

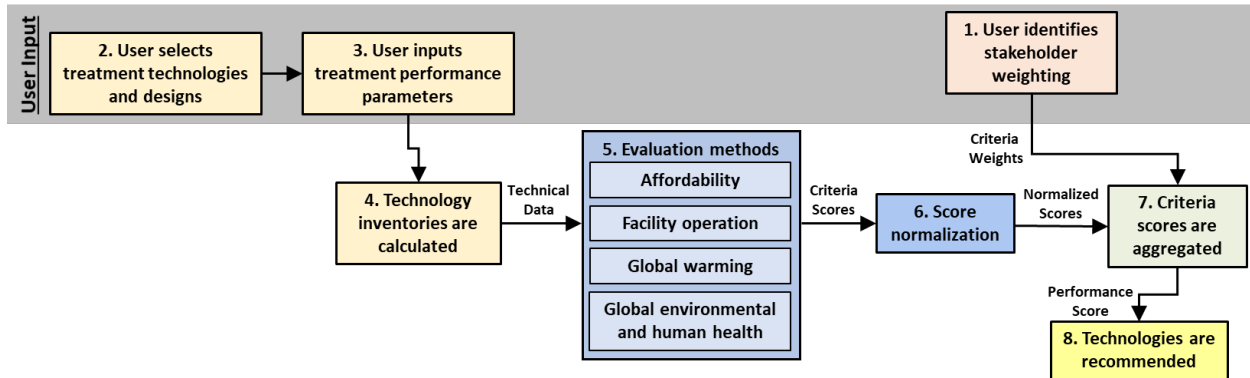


Figure 4.1. Framework for the PFAS Treatment Technology Decision-Support Tool.

4.1 STAKEHOLDER WEIGHTING INPUTS

The first tool input is the user’s criteria weighting scheme, which reflects the user’s priorities and which decision criteria are important in their decision. The user can input their own weighting scheme or select a preset weighting scheme (Figure 4.2). To input their own weighting scheme, the user will have to state the relative importance of each of the 15 decision criteria; the sum must be 100%, and users can input not at all important (i.e., do not include in the analysis) by inputting the value of 0%. The weighting scheme is the relative importance of each decision criteria, specific to that user, and is termed a “stakeholder weight” and used to weight the normalized criteria score in the final overall score calculations (see Section 4.6).

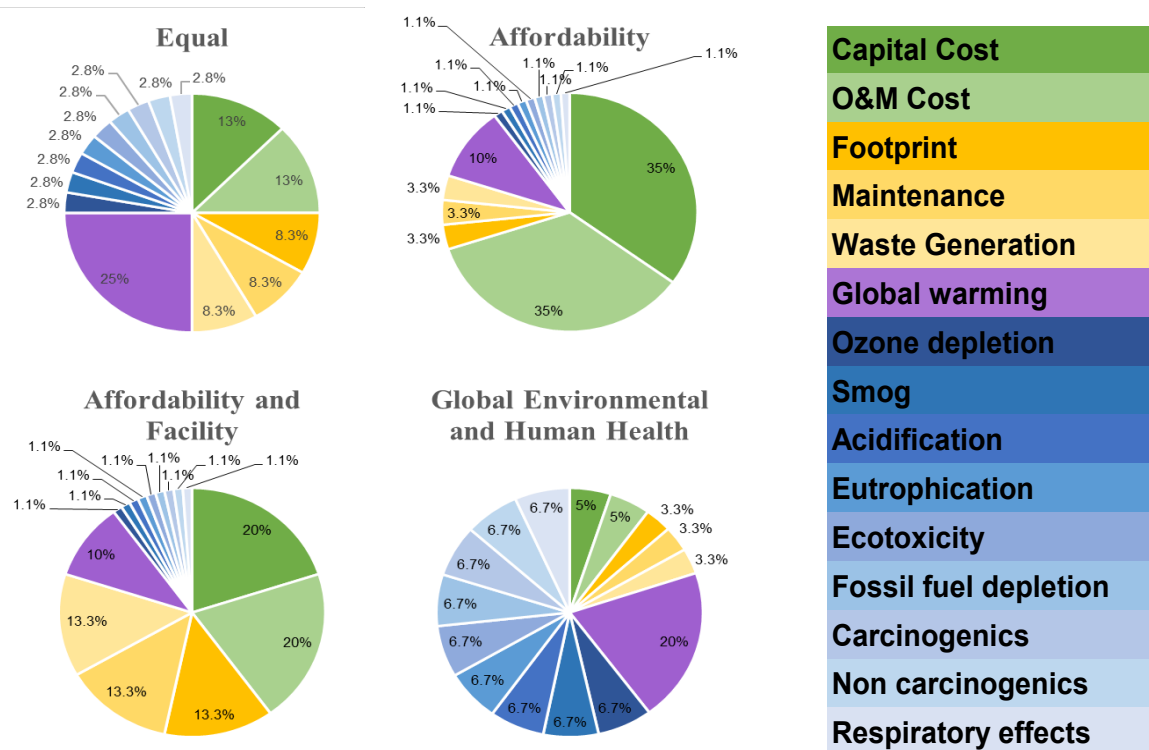


Figure 4.2. Preset weighting schemes.

4.2 TREATMENT TECHNOLOGY OPTIONS AND DESIGN INPUTS

Users can select from five treatment technology systems that use granular activated carbon (GAC), either in a gravity basin or pressure filter; anion exchange (IX) filter; reverse osmosis (RO) membranes; or nanofiltration (NF) membranes. Unfortunately, it is not possible to include combined treatment technology trains (i.e., GAC and IX) in the tool at this time due to a lack of standard design assumptions and values for estimating all the unit process sizing, energy use, chemical use, and infrastructure material requirements. However, the framework developed for this tool can be applied to a combined treatment train once design information and treatment data are available.

For all options, user input of flowrate is needed. If the user selects a preset flowrate option (0.03, 0.12, 0.31, 0.74, 2.2, 7.4, 23, or 75 mgd) then the tool uses default system design specifications based on that selection; if the user wants to use a customized flowrate or design specifications, then the user inputs the associated design specification values. The user also selects the energy sector (WECC, MRO, SPP, TRE, RFC, SERC, NPCC, or FRCC) relevant to their scenario (i.e., location), an average pump efficiency, and if a building structure is needed to house the treatment system.

4.2.1 GAC Gravity Basin

To include the GAC gravity basin in the analysis, the user inputs a media usage rate for each type of GAC to be compared (up to 3). The user also selects a disposal option for the GAC; disposal options include thermal reactivation, incineration, hazardous waste landfill, and PFAS-specific landfill. The user selects whether backwash residuals disposal (through publicly owned treatment works (POTW) discharge) is included in the cost analysis. The tool assumes basin sizings based on flowrate; the user can modify those values.

The cost, environmental impact, and facility requirements for this option are based on the main processes needed to fulfill the functional unit; in summary, the system boundary, which has the same scope and detail as the other option's system boundaries, includes GAC production and hauling (both to the treatment site and the disposal site); energy for groundwater pumping (only within the treatment system; all other pumping will be the same between the options); production of materials to construct and maintain the gravity basin and the holding tank for backwash residuals; production of materials to construct, and energy to operate, the building that houses the treatment system; and cost of backwash residuals disposal (Figure 4.3).

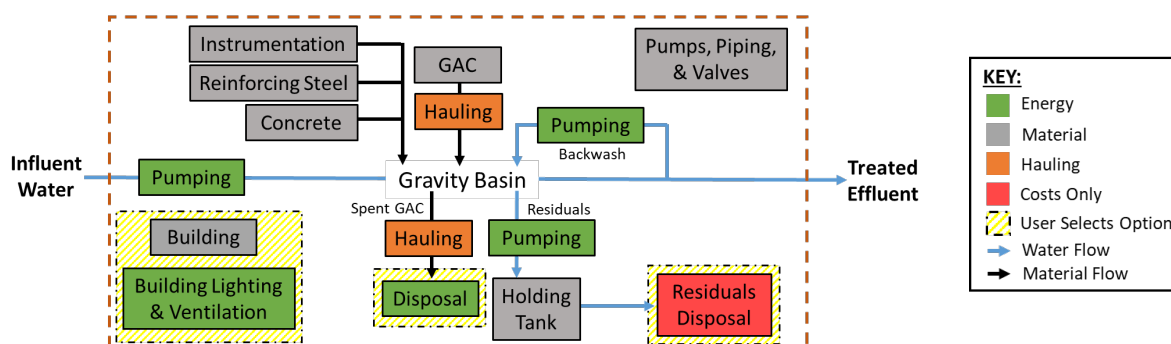


Figure 4.3. System Boundary for GAC Gravity Basin Option.

4.2.2 GAC Pressure Filter

To include the GAC pressure filter in the analysis, the user has the same inputs and selections as for the GAC gravity basin: media usage rate for each GAC (up to 3); GAC disposal option; and inclusion of backwash residuals disposal cost. The assumed pressure vessel sizings, based on flowrate, can be modified. The system boundary includes the same processes as the gravity basin option with three differences: different types of vessel materials, different instrumentation, and inclusion of PVC internals (i.e., influent and backwash distributors) (Figure 4.4).

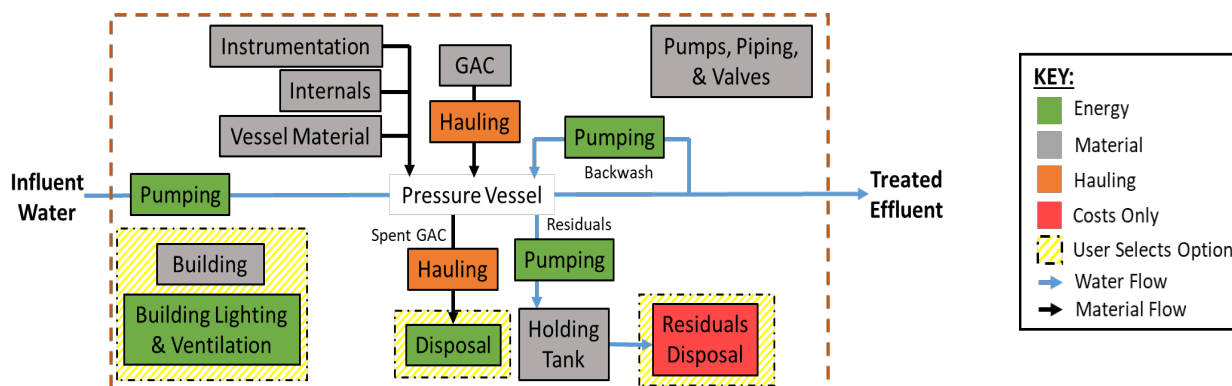


Figure 4.4. System Boundary for GAC Pressure Filter Option.

4.2.3 IX Filter with Cartridge Prefilter

To include IX with prefilter in the analysis, the user has similar inputs and selections as for the GAC pressure vessel: media usage rate for each IX resin (up to 3) and resin disposal option. The assumed pressure vessel sizings, based on flowrate, can be modified. The system boundary includes the same processes as the GAC pressure vessel option with two differences: production of IX resin instead of GAC; and production and disposal of cartridge filters instead of energy and residuals from backwashing (Figure 4.5).

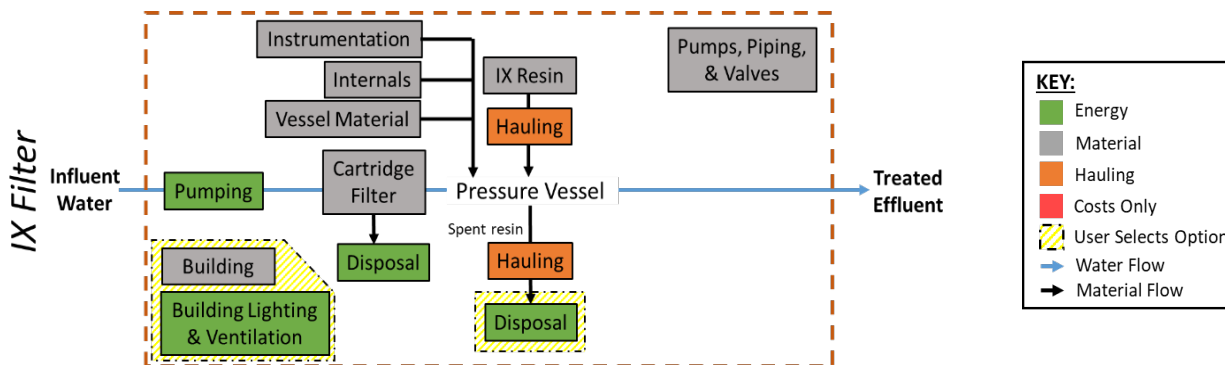


Figure 4.5. System Boundary for IX Filter with Cartridge Prefilter Option.

4.2.4 IX Filter with Backwashing

To include IX filter with backwashing in the analysis, the user has similar inputs and selections as for the GAC pressure vessel: media usage rate for each IX resin (up to 3); resin disposal option; and inclusion of backwash residuals disposal cost. The assumed pressure vessel sizings, based on flowrate, can be modified. The system boundary includes the same processes as the GAC pressure vessel option with one difference: production of IX resin instead of GAC (Figure 4.6).

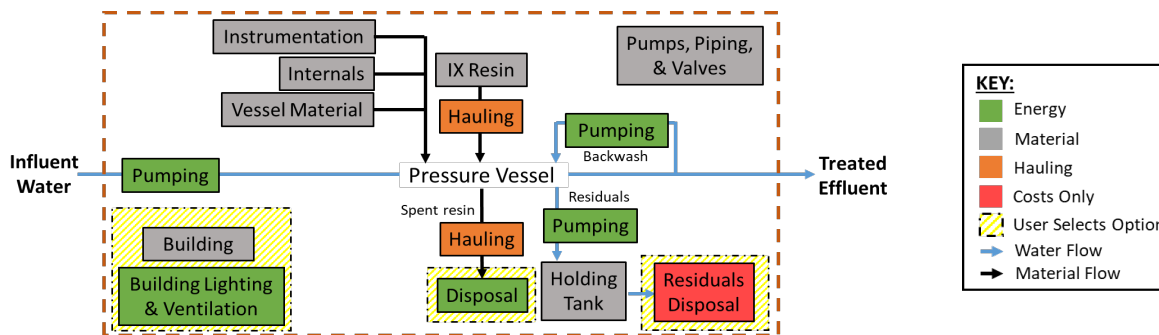


Figure 4.6. System Boundary for IX Filter with Backwashing Option.

4.2.5 Membranes (RO or NF)

To include the RO and/or NF options in the analysis, the user selects the source water quality (from a list of preset waters or user defined water quality); influent and effluent pH; water temperature; membrane disposal option (options include: hazardous waste landfill or PFAS-specific landfill); and if liquid disposal for concentrate and cleaning residuals is included, what disposal option (options include hazardous waste liquid disposal, deep well injection, or POTW discharge). For each RO and NF membrane type, the tool assumes membrane element sizes, recovery rates, and fluxes based on flow rate; the user can modify those values.

The system boundary for either RO and NF membranes is the same; it includes membrane production, disposal, and hauling (both to the treatment site and to the disposal site); energy for groundwater pumping (within the treatment system); production and disposal of cartridge filters (for membrane influent and cleaning); production of membrane cleaning chemicals; production of materials to construct and maintain the membrane trains and the holding tank for liquid waste; production of materials to construct, and energy to operate, the building that houses the treatment system; and cost of liquids disposal (Figure 4.7).

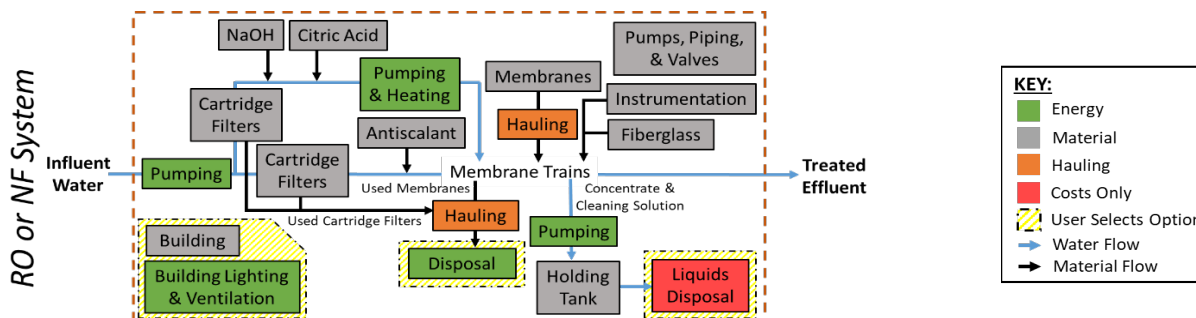


Figure 4.7. System Boundary for RO or NF System Options.

4.3 TREATMENT PERFORMANCE PARAMETERS

Due to the specificity of GAC and IX resin treatment performances, which depend on the exact media, water quality (influent PFAS concentrations, TOC, pH), and treatment objective (effluent PFAS concentrations), users must input media usage rates (in pounds per 1000 gallons) for each media being compared. Those usage rates determine the amount of media required over the functional unit lifetime. For an accurate comparison, the usage rates for all media need to be determined for the same water quality and treatment objective (i.e., scenario). See Section 4.7.1 *Navigating Treatment Performance Uncertainty using Scenario Analysis* if the user does not have media usage rate data.

For RO and NF, effluent PFAS concentrations are calculated by percent rejection [$C_{\text{eff}} = C_{\text{inf}} \cdot (1 - R)$] (Crone et al., 2019; Liu et al., 2021a) based on the user inputs of influent PFAS concentration and treatment objective. If the RO and/or NF option cannot meet that treatment objective, it is removed from the analysis. For both RO and NF, there are two sets of default values for percent rejection (Table 4.2); one for when recovery rate is greater than 75%, and one for when it is less than 75% (Liu et al., 2021a). Drinking water and groundwater regulations, notification levels, guidance levels, and health advisory levels (Table 4.3, Table 4.4, and Table 4.5) are included in the tool for selection by the user as the treatment objective, or the user can input their own. The treatment objective chosen for RO and/or NF needs to be the same as the treatment objective associated with any other technology options, specifically with the treatment objective used to determine the GAC and IX media usage rates.

Table 4.2. Percent Rejection by PFAS Compound for RO and NF Membranes.

Compound	Percent Rejection			
	RO		NF	
	Recovery		Recovery	
	<75%	>75%	<75%	>75%
PFPeA	98.5%	99.2%	98.9%	99.4%
PFHxA	99.9%	99.9%	98.7%	99.3%
PFHpA	96.9%	98.3%	98.0%	99.0%
PFOA	99.9%	99.9%	96.7%	98.4%
PFNA	99.9%	99.9%	96.7%	98.4%
PFPrS	98.0%	98.6%	84.1%	92.7%
PFBS	99.9%	99.9%	94.9%	94.6%
PFPeS	99.3%	99.5%	96.1%	97.6%
PFHxS	99.9%	99.9%	97.5%	97.7%
PFHpS	99.7%	99.9%	87.0%	93.9%
PFOS	99.4%	99.8%	97.4%	98.7%

Table 4.3. State Regulations for PFAS in Drinking Water (MCLs) that Are Available as Pre-set Selections in the Tool.

Compound	Effluent Concentration (ng/L)						
	CA	MA	MI	NH	NJ	NY	VT
PFBA							
PFPeA							
PFHxA			400,000				
PFHpA							
PFOA	10		8	12	14	10	
PFNA			6	11	13		
PFPrS							
PFBS	5000		420				
PFPeS							
PFHxS			51	18			
PFOS	40		16	15	13	10	
PFOA + PFOS							
PFHpA + PFOA + PFNA + PFDA + PFHxS + PFOS		20					20

Table 4.4. State Notification, Guidance, or Health Advisory Limits for PFAS in Drinking Water that Are Available as Pre-set Selections in the Tool.

Compound	Effluent Concentration (ng/L)												
	AK	CA	CT	CO	DE	MA	ME	MN	NM	NV	OH	OR	WA
PFBA								7000					
PFPeA													
PFHxA													
PFHpA													
PFOA		5.1						35		667			10
PFNA											21		14
PFPrS													
PFBS		500		400000				2000		667000	140,000		1300
PFPeS													
PFHxS				700		140		47			140		70
PFOS		6.5						15					
PFOA + PFOS	70				70				70		70		
PFOA + PFNA + PFOS				70									
PFOA + PFNA + PFHxS + PFOS												30	
PFHpA + PFOA + PFNA + PFHxS + PFOS			70										
PFHpA + PFOA + PFNA + PFDA + PFHxS + PFOS							20						

Table 4.5 State Notification, Guidance, or Health Advisory Limits for PFAS in Groundwater that Are Available as Pre-set Selections in the Tool

Compound	Effluent Concentration (ng/L)																
	AK	CO	DE	FL	IL	MA	ME	MI	MN	MT	NC	NH	NJ	PA	RI	VT	WI
PFBA																	
PFPeA																	
PFHxA					560000			400000									
PFHpA																	
PFOA	400	70			21		400	8	35		2000	12	10				
PFNA					21			6				11	13				
PFPrS																	
PFBS					140,000			420						690000			
PFPeS																	
PFHxS					140			51	47			18					
PFOS	400	70			14		400	16	15			15	10				
PFOA + PFOS			70	70	21					70				70	70		20
PFHpA + PFOA + PFNA + PFHxS + PFOS																20	
PFHpA + PFOA + PFNA + PFDA + PFHxS + PFOS						20											

4.3.1 Media Disposal and Hauling

4.3.1.1 Contaminated Media Disposal

PFAS waste disposal is currently unregulated. Therefore, for the disposal of PFAS-contaminated GAC and IX media, the three options supported by the Dec 2020 guidance document (USEPA 2020) are included: (1) hazardous waste landfills; (2) landfills specifically accepting PFAS waste (US Ecology, 2022); and (3) hazardous waste incinerators. Additionally, thermal reactivation was added as a GAC disposal option following the advice of water professionals from the February 2022 Virtual Stakeholder Workshops.

Landfill. Available LCA data on environmental impacts of hazardous waste (Resource Conservation and Recovery Act (RCRA) certified) landfilling is focused on electricity use, and no data is available to account for impacts from PFAS in landfill leachate. For hauling distance estimations, all RCRA-certified hazardous waste landfills were included, including landfills run by Chemical Waste Management; Reynolds Metals Company, LLC; Clean Harbours; Kettleman Hills Hazardous Waste Facility; Heritage Environmental Service; CWM Chemical Services; US Ecology; EnviroSAFE Services of Ohio; Waste Control Specialists, LLC; and Energy Solutions (USEPA 2019). U.S. Ecology operates two landfills (Grand View, ID and Beatty, NV) that have specifically stated are accepting PFAS waste (US Ecology, 2022). Environmental impacts and costs for the PFAS specific landfills are modeled the same as a hazardous waste landfill, but with hauling distances calculated to those two locations only.

Incineration. Incineration facilities that are approved for AFFF incineration according to the Defense Logistics Agency are included to determine hauling distances, including incinerators run by Clean Harbours, Ross Incinerator, Veolia, and Heritage Thermal Service (USEPA, 2019). Environmental impacts from incineration are calculated using the *hazardous waste, for incineration market for hazardous waste, for incineration APOS, U* unit process from Ecoinvent (Table 4.17).

GAC Thermal Reactivation. To calculate the associated costs and environmental impacts of GAC thermal reactivation, the amount of virgin GAC that could be offset by the reactivated GAC was calculated assuming 10% carbon loss and 25% sorption effectiveness loss due to the thermal reactivation process (Miguel et al., 2001). An associated income is calculated for reactivation, equal to the current market value of regenerated GAC. Cost of hauling to the reactivation facility is included with hauling distances calculated to thermal reactivation facilities operated by Calgon Carbon (Calgon, 2022) and Evoqua (Evoqua, 2022).

4.3.1.2 Residuals and Liquids Disposal

The cost of discharging the residuals generated from backwashing GAC and IX filters to a POTW is included in the tool. Three options are included for liquid (concentrate and cleaning) waste produced from RO and NF systems: discharge to POTW, hazardous waste disposal, and deep well injection. The cost of liquid waste disposal is included in the tool, as well as hauling for hazardous waste disposal and deep well injection. Costs for all are based on the EPA WBS unit cost estimates (See Table C10 in Appendix C). Environmental impacts associated with the disposal of liquid waste streams could not be included in the tool, so the amount of liquid waste generated (in gal/yr) is explicitly included as a decision criterion to help the user evaluate the practical considerations, beyond cost, of waste handling based on amount of waste.

4.3.1.3 Hauling Distances

Environmental impacts and costs are included for hauling of fresh media and virgin membranes from production facility to treatment system location, spent media and membranes from the treatment system location to disposal facility, and if selected, RO/NF liquid waste to hazardous waste treatment or deep well injection sites. Hauling of other materials and chemicals is included within those associated LCI unit processes (Table 4.17).

A treatment system's location is input by the user; the options are based on US energy grid (Figure 4.8). For each energy sector, default driving distances were the midpoint value expected between a treatment site and disposal facility and media production (Table 4.8). The distances were estimated using Google maps, common contamination site locations (Table 4.6), disposal facilities locations (Table 4.7), and manufacturing site locations. GAC manufacturing locations were Calgon Carbon Corporation plants in St. Louis, MS; Columbus, OH; Gila Bend, AZ, Pittsburgh, PA; and Cattlesburg, KY; and Cabot Norit plants in Marshall, TX and Pryor, OK. IX Resin manufacturing locations were the Purolite manufacturing plant in Philadelphia, PA; ResinTech in St. Camden, NJ; Dupont in Edna, MN; and Calgon Carbon in Pittsburgh, PA; Cattlesburg, KY; St. Louis, MS; Columbus, OH; and Gila Bend, AZ. Membrane manufacturing sites from the largest membrane producers were included for: DuPont (Edina, MN), Hydranautics (Oceanside, CA), and Koch Separations (Willmar, MN; Pickering, ON; and Wilmington, MA).

Table 4.6. Chosen Coordinates of PFAS Contamination Sites (Hu et al., 2016)

State	Energy Grid	Chosen Coordinates	State	Energy Grid	Chosen Coordinates
Washington	WECC	47.566916, -122.157965	Indiana	RFC	41.662777, -86.825732
California	WECC	39.948467, -121.961175	Michigan	RFC	42.886941, -85.636757
California	WECC	36.103579, -119.741087	Michigan	RFC	41.897027, -83.640962
California	WECC	33.062726, -116.477717	Kentucky	SERC	38.577340, -84.545158
Arizona	WECC	33.333094, -111.883998	Tennessee	SERC	35.805325, -86.689366
New Mexico	SPP	32.845664, -104.075386	Alabama	SERC	34.672164, -86.069393
Texas	TRE	31.686621, -99.050436	Florida	SERC	30.830265, -86.337942
Texas	TRE	29.069724, -95.903087	Georgia	SERC	33.319093, -83.128743
Colorado	WECC	39.577918, -105.296885	Florida	FRCC	28.492981, -82.474302
Kansas	SPP	38.306979, -98.299225	Florida	FRCC	26.671544, -80.630486
Oklahoma	SPP	36.042222, -97.263653	South Carolina	SERC	34.854112, -82.590875
South Dakota	MRO	44.989316, -100.770925	North Carolina	SERC	35.872912, -77.532788
Minnesota	MRO	47.174817, -94.516401	Virginia	RFC	36.800562, -81.645784
Minnesota	MRO	45.872548, -93.241056	New York	NPCC	44.011089, -75.763496
Wisconsin	MRO	44.977754, -89.110185	New Hampshire/ Massachusetts	NPCC	43.186579, -71.390806
Wisconsin	RFC	42.928333, -88.425297	New Jersey	RFC	40.151306, -74.596467
Illinois	SERC	40.926518, -90.670338	Pennsylvania/ Maryland	RFC	39.810972, -75.954717
Illinois	SERC	39.769176, -89.419543	Ohio	RFC	39.902478, -84.125559

Table 4.7. Locations of PFAS Disposal Sites

Facility Type	Facility Name	State
Incinerator	Clean Harbours - Arkansas	El Dorado, AR
	Veolia - Illinois	Sauget, IL
	Clean Harbours - Nebraska	Kimball, NE
	Ross incinerator	Grafton, OH
	Heritage Thermal Service	East Liverpool, OH
	Veolia - Texas	Beaumont, TX
	Clean Harbours - Texas	La Porte, TX
	Clean Harbours - Aragonite	Grantsville, UT
PFAS Landfill	US Ecology - Idaho	Grand View, ID
	US Ecology - Nevada	Beatty, NV
Haz Waste Landfill	CHEMICAL WASTE MANAGEMENT	Emelle, AL
	REYNOLDS METALS COMPANY LLC	Arkadelphia, AR
	CLEAN HARBORS BUTTONWILLOW	Buttonwill, CA
	Kettleman Hills Hazardous Waste Facility	Kettleman City, CA
	Clean Harbors Deer Trail	Deer Trail, CO
	HERITAGE ENVIRONMENTAL SERVICE	Roachdale, IN
	CWM Lake Charles	Sulphur, LA
	US Ecology Wayne Disposal Inc	Belleville, MI
	CWM CHEMICAL SERVICES	Youngstown, NY
	Envirosafe Services of Ohio, Inc.	Oregon, OH
	Clean Harbors Environmental Lone Mountain	Waynoka, OK
	Chemical Waste Management of the Northwest	Arlington, OR
	US Ecology Texas	Robstown, TX
	Waste Control Specialists LLC	Andrews, TX
	Energy Solutions	Grantsville, UT
	Clean Harbors Grassy Mountain	Wendover, UT
Thermal Reactivation	Calgon Carbon	Catlettsburg, KY
	Calgon Carbon	Pittsburgh, PA
	Evoqua	Darlington, PA
	Evoqua	Parker, AZ
	Evoqua	Red Bluff, CA
Deep well injection	US Ecology Texas	Winnie, TX
	Texas Molecular	Deer Park, TX
	Buckeye Brine	Coshocton, OH

Table 4.8. Midpoint Hauling Distances (in Miles) for Fresh Media, Virgin Membrane, and Disposal

	Fresh/Virgin			Disposal Options				
Energy Grid	GAC	IX	RO/NF	Incinerator	PFAS Landfill	Haz Waste Landfill	Thermal Reactivation	Deep Well Injection
FRCC	654	118	1382	939	2479	661	1016	952
MRO	911	489	194	579	1363	597	1040	946
NPCC	528	343	200	560	2619	358	630.5	623
RFC	248	285	581	283	2071	252	368	308
SERC	317	378	827	340	2030	284	569.5	452
SPP	766	712	771	548	1126	136	1086	608
TRE	568	457	1373	212	1397	196	1367	201
WECC	748	644	647	518	592	297	658	1694

4.4 LIFE CYCLE INVENTORY DATA CALCULATIONS

The types and amounts of energy, materials, and chemicals necessary for the operation and maintenance of each treatment system option over the functional unit lifetime are quantified by the tool using the following assumptions.

4.4.1 Energy

Costs and environmental impacts from energy generation are based on the energy sector (

Figure 4.8). Equations and values used for energy calculations are taken from EPA Work Breakdown Structures (USEPA, 2017) unless otherwise noted.



Figure 4.8. U.S. Energy Sectors

4.4.1.1 Pumping Energy

Pumping energy for GAC and IX influent pumping, backwash pumping, and residuals pumping:

$$P = \frac{\left(Q * \left(0.000063 \frac{m^3/h}{gpm} \right) * \rho * g * H * \left(0.305 \frac{m}{ft} \right) \right)}{\left(\left(10^6 \frac{W}{MW} \right) * \eta \right)} * \left(24 \frac{h}{d} \right) * \left(365 \frac{d}{yr} \right)$$

P = power (MWh/yr)

Q = flowrate (gpm)

ρ = density of water (kg/m³) = 1000 kg/m³

g = gravity = 9.81 m/s²

H = pumping head (ft) = 6.6 ft (GAC gravity basin) (Jones et al., 2019), 58 ft (GAC pressure and IX filter) (USEPA, 2017); 25 ft (residuals) (USEPA, 2017); 33 ft (backwash) (USEPA, 2017)

η = pump efficiency (%) = 75% (default)

Influent pumping energy for RO and NF membranes:

$$P_{MW} = \frac{Q * \left(\Pi + \frac{\Delta P}{2} + NDP \right) * \left(\frac{2.31 \text{ psi}}{\text{feet head}} \right) * \left(0.746 \frac{kW}{hp} \right)}{n * \eta * \left(\frac{3960 \text{ gal/min/ft}}{hp} \right) * \left(1000 \frac{kW}{MW} \right)} * \left(24 \frac{h}{d} \right) * \left(365 \frac{d}{yr} \right)$$

P_{MW} = power (MWh/yr)

Q = flowrate (gpm)

Π = osmotic pressure (psi)

ΔP = pressure drop in system (psi)

NDP = net driving pressure (psi)

n = number of pumps

η = pump efficiency (%) = 75% (default)

4.4.1.2 Building Lighting Energy

$$P = \left(\frac{t_{oper}}{\left(24 \frac{h}{d} \right) * \left(365 \frac{d}{yr} \right)} \right) * A * \left(\frac{P_{sf}}{\left(24 \frac{h}{d} \right) * \left(365 \frac{d}{yr} \right)} \right) / \left(10^6 \frac{W}{MW} \right)$$

P = power (MWh/yr)

t_{oper} = annual operator maintenance (h/yr)

A = building footprint (ft²)

P_{sf} = lighting power (W/ft²) = 1 W/ ft²

4.4.1.3 Building Ventilation Energy

$$P = \frac{(n_d) * \left(24 \frac{h}{d}\right) * \left(0.746 \frac{kW}{hp}\right) * ACPH * A * \Delta P * h}{33,000 * \left(10^3 \frac{kW}{MW}\right)}$$

P = power (MWh/yr)

N_d = number of mechanical ventilation days (days/yr) = 90 days/yr

ACPH = weighted average air changes per hour = 3 air changes/hr (filters), 20 air changes/hr (pumps)

A = building footprint (ft²)

ΔP = fan pressure drop (lbs/ft²) = 0.25 lbs/ft²

h = building height (ft) = 20 ft

4.4.1.4 Immersion Heater Energy (RO/NF cleaning solutions)

$$P = \frac{\Delta T * V}{\left(365 \frac{d}{yr}\right) * t * \left(1000 \frac{kW}{MW}\right)}$$

P = immersion heater energy (MWh/yr)

ΔT = temperature rise for cleaning (°F) = (95 °F - T_{influent})

V = volume of cleaning solution (gal) = volume of pressure vessels + volume of pipes

t = cleaning time (hr) = 5 hr

$$\text{Cleaning Frequency} = 12 - 4 * (SDI - 2)$$

SDI = silt density index - 15 min (included with water quality presets)

4.4.2 Materials

4.4.2.1 Gravity Basins and Pressure Vessels

The sizes of GAC and IX filter vessels are based on EPA WBS standard sizing (USEPA 2017) and are determined by flowrate (Table 4.9). Gravity basins are composed of 1 ft thick concrete walls and floor with reinforcing steel assumed at 0.0929 lbs/ft² (Jones et al., 2019). Over excavation for gravity basins is assumed to be 4 ft (See Table C2 in Appendix C for associated costs for over excavation and backfilling) (USEPA, 2017). Aluminum railings run the length of the outside

perimeter of concrete gravity basins and are assumed to weigh 2 lbs/ft (ULINE, 2022). Stairs for gravity basins have 15 risers per basin and are assumed to weigh 400 lbs/basin (Grainger, 2022).

Pressure vessels with volumes less than 900 gallons are made of fiberglass, and larger pressure vessels are made of carbon steel. Mass of material needed for pressure vessels is calculated by calculating the volume of material, assuming a cylindrical tank with hemispherical ends and a 0.5 in thickness (Jones et al., 2019), and multiplying by the associated density (Table 4.14). Pressure vessel internals are made of PVC pipes with an outer diameter of 1.3 in and a wall thickness 0.3 in. Internals are composed of 8 lengths of pipe equal to the diameter of the vessel and 1 length of pipe equal to the height of the vessel (SWT, 2022) Mass of material for internals is calculated by multiplying the volume of PVC by its density (Table 4.15).

4.4.2.2 Membrane Trains

Membrane element size, recovery rate, and design flux are based on membrane type (RO or NF), water quality (Table 4.10), and EPA WBS designs, which are based on flowrate. Membrane weights are 10, 40, or 250 lbs for 4 in, 8 in, and 16-18 in membrane elements, respectively. Membrane material percentage by weight is 30% polyamide and 70% polypropylene (Tarnacki et al., 2012) The mass of stainless steel, for the membrane housing, is calculated as 78% of the element weight (Tarnacki et al., 2012).

4.4.2.3 Pumps, Piping, and Valves

All pumps are made of aluminum and weigh 30 lbs/pump (Magnatex, 2022). All pipes and valves are PVC with diameters based on flow rate (USEPA, 2017). Weights for piping and valves are shown in Table 4.11 and Table 4.12.

4.4.2.4 Holding Tanks

Holding tanks are made of HDPE. Residuals holding tanks (GAC and IX) are sized to hold the volume of one backwashing event. Liquid waste holding tanks (RO and NF) are designed to hold 60 days of concentrate plus the volume of one cleaning event. Mass of HDPE needed for holding tanks is calculated by calculating the volume of material, assuming a cylindrical tank with hemispherical ends, with a height to diameter ratio of 2.5, thickness of 0.5 inches (Jones et al., 2019), and multiplying by HDPE density (Table 4.15).

4.4.2.5 Instrumentation

Included instrumentation for each treatment option are based on assumptions from the EPA WBS, as shown in Table 4.13. Weights for instrumentation were taken from product brochures, as shown in Table 4.14.

4.4.2.6 Concrete Pad

Volume of concrete for the concrete pad is calculated assuming the area is the same as the process footprint with a thickness of 6 inches. Mass of rebar in the concrete pad is 0.305 kg/m^2 (Jones et al., 2019). The footprint is calculated by rounding up the total area required to house the entire treatment train system (filters, membrane trains, pumps, etc.) to the nearest 10 square feet. The area required for GAC and IX options is calculated by the following equation:

$$\begin{aligned} \text{Process Area} = & (\text{Filter Footprint} + \text{Filter Access Space}) \\ & + (\text{Pump Footprint} + \text{Pump Access Space}) + (\text{Residuals Tank Footprint} \\ & + \text{Holding Tank Access Space}) \end{aligned}$$

Area required for RO and NF systems is calculated by the following equation:

$$\begin{aligned} \text{Process Area} = & \text{Pumping Footprint} + \text{Pretreatment Equipment Footprint} \\ & + \text{Cleaning System Footprint} + \text{Membrane Equipment Footprint} \\ & + \text{Concentrate Handling Footprint} \end{aligned}$$

4.4.2.7 Building

Buildings that are constructed to house the treatment system are assumed to be 20 ft tall and square (area equal to the process footprint) with steel columns and trusses, concrete brick walls, and gravel roofs (USEPA, 2017). Columns are spaced every 10 feet with dimensions of 6"x6"x20' and weigh 540 lbs/column. Concrete bricks for the walls weigh 58 lbs/ft² (Angelus, 2022). Roofs are composed of steel trusses with wood purlins covered in plywood, roofing felt, asphalt, and gravel (Malarkey, 2022). Truss spacing is 10 feet with purlin spacing at 4 feet (APB, 2022). Weights for roofing materials are as follows: 1.25 lbs/ft for wood purlins, 1.4 lbs/ft² for plywood, 0.21 lbs/ft² for roofing felt, 0.6 lbs/ft² for asphalt, and 4 lbs/ft² for gravel (Roof Online, 2022; The Engineering Toolbox, 2022a).

4.4.2.8 Chemicals (RO/NF only)

In RO and NF systems, antiscalant is dosed into the feedwater to control fouling, NaOH is used as a high pH cleaner, and citric acid is used as a low pH cleaner. Antiscalant dose is calculated by assuming a concentration of 15 mg/L in the concentrate stream (Dose = (15 mg/L)*(1 – Recovery)). Antiscalant is modeled as 19% polyacrylamide and 81% polycarboxylates by mass (Tarnacki et al., 2012). Cleaning frequency is calculated based on influent water silt density index (SDI), which is included with preset water quality selections. Minimum cleaning frequency is once every 18 months. Volume of cleaning solution is calculated as the total volume of membrane vessels and piping. High pH cleaning solution is 0.2% NaOH and low pH cleaning solution is 2.0% citric acid. NaOH storage tanks are included if the monthly storage requirement is greater than 5 gallons. Chemical storage tanks are sized to hold one month of NaOH and made of HDPE with a wall thickness of 0.5 inches (Jones et al., 2019).

Table 4.9. Standard Sizing by Flow Rate for GAC Gravity Basin, GAC Pressure Filter, and IX Filter

Flowrate (MGD)	0.03	0.12	0.31	0.74	2.15	7.37	22.6	75.1
GAC Gravity Basin								
Bed Depth (feet)	3	5	6	6	8	10	10	10
Basin Length (feet)	6	6	7.5	10	12	15	20	30
Basin Width (feet)	6	6	7.5	10	12	15	20	30
EBCT (min)	10	10	10	10	10	10	10	10
GAC Pressure Vessel								
Bed Depth (feet)	4.5	6.5	7	7	7	7	7	7
Vessel Height (feet)	6.5	10.5	11	11	11	11	14	14
Vessel Diameter (feet)	2.5	4	5	7	8.5	11	20	20
EBCT (min)	10	10	10	10	10	10	10	10

IX Filter								
Bed Depth (feet)	2.5	2.5	3	3	4	4	4.3	4.3
Vessel Height (feet)	4.5	4.5	5	5	7	7	7	10
Vessel Diameter (feet)	1.5	3.5	4.5	5.5	5.5	11	12.5	30
EBCT (min)	2	2	2	2	2	2	2	2

Table 4.10. Standard Design Assumptions and RO and NF Systems

(B – brackish RO membrane, LP – low pressure RO membrane).

Flowrate (MGD)	0.03	0.12	0.31	0.74	2.15	7.37	22.6	75.1
RO Membrane								
<i>Low Quality GW</i>								
Membrane	4 in B	4 in B	8 in B	8 in B	8 in B	8 in B	8 in B	8 in B
Recovery Rate	71%	65%	73%	75%	75%	75%	75%	75%
Design Flux	16	16	16	16	16	16	16	16
<i>Mid Quality GW</i>								
Membrane	4 in LP	4 in LP	8 in LP	8 in LP	8 in LP	8 in LP	8 in LP	8 in LP
Recovery Rate	72%	75%	74%	80%	80%	80%	80%	80%
Design Flux	17	17	17	17	17	17	17	17
<i>High Quality GW</i>								
Membrane	4 in LP	4 in LP	8 in LP	8 in LP	8 in LP	8 in LP	8 in LP	8 in LP
Recovery Rate	72%	75%	74%	80%	80%	80%	80%	80%
Design Flux	19	19	19	19	19	19	19	19
NF Membrane								
<i>Low Quality GW</i>								
Membrane	4 in	8 in	8 in	8 in	8 in	8 in	8 in	8 in
Recovery Rate	72%	75%	75%	75%	75%	75%	75%	75%
Design Flux	16	16	16	16	16	16	16	16
<i>Mid Quality GW</i>								
Membrane	4 in	4 in	8 in	8 in	8 in	8 in	8 in	8 in
Recovery Rate	72%	75%	75%	85%	85%	85%	85%	90%
Design Flux	17	17	17	17	17	17	17	17
<i>High Quality GW</i>								
Membrane	4 in	4 in	8 in	8 in	8 in	8 in	8 in	8 in
Recovery Rate	73%	75%	75%	85%	85%	85%	85%	90%
Design Flux	19	19	19	19	19	19	19	19

Table 4.11. PVC Piping Weight in kg/ft According to Pipe Diameter

(The Engineering Toolbox, 2022b).

Pipe Diameter (in)	Pipe Weight (kg/ft)
0.5	0.073
0.75	0.095
1	0.15
1.5	0.23
2	0.31
2.5	0.49
3	0.64
4	0.91
6	1.6
8	2.4

Table 4.12. PVC Valve Weights by Pipe Diameter

(Hayward, 2022; Plastomatic, 2022).

Pipe Diameter (in)	Valve Weight (kg)
0.5	0.29
0.75	0.43
1	0.66
1.5	1.28
2	1.88
3	7.12
4	11.80
6	13.7

Table 4.13. Instrumentation Included in the Analysis of Each Treatment Option

Instrument	GAC Gravity Basin	GAC Pressure Filter	IX Filter	NF System	RO System
Conductivity Meter	0	0	0	3 per operating rack + 2	3 per operating rack + 2
Flow Meter	2	2	2	3 per operating rack + 1 per cleaning chemical pump	3 per operating rack + 1 per cleaning chemical pump
Head Loss Sensor	0	1 per vessel	0	1 per operating rack	1 per operating rack
High/Low Alarm	0	0	0	1 per antiscalant tank	1 per antiscalant tank
Level Switch/ Alarm	1 per basin	1 per vessel	1 per vessel	1 per cleaning chemical	1 per cleaning chemical
pH Meter	0	0	0	1 per cleaning chemical	1 per cleaning chemical
Pressure Transducer	0	0	0		

Sampling Ports	1 per basin + 3	1 per basin + 3	1 per basin + 3	1 per operating rack + 1 per pressure vessel + 2	1 per operating rack + 1 per pressure vessel + 2
Temperature Meter	0	0	0	1 per cleaning chemical + 1	1 per cleaning chemical + 1
Turbidity Meter	1 per basin	0	0	0	0

Table 4.14. Instrument Weights Used to Calculate LCA Inventory Data

Instrument	Weight (kg)
Level switch	2
pH sensor	1.5
Temperature meter	1.5
Turbidimeter	1.5
Head loss sensor	1.5
Conductivity meter	1.5
Flow meter	12
Pressure sensor	5
Sampling Port	0.4

Table 4.15. Material Densities Used in LCA Calculations

(The Engineering Toolbox, 2022c).

Material	Density (kg/m ³)
Fiberglass	1500
HDPE	970
Carbon steel	7850
PVC	1380

4.5 EVALUATION METHODS

4.5.1 Affordability

The decision criteria in this category are capital costs and operation and maintenance (O&M) costs. Capital costs include initial capital costs (filter basins/vessels, membrane trains, building, concrete pad, tanks, pumps, piping, valves, and instrumentation) and indirect capital costs (engineering, surveying, building construction, and construction labor; estimated as 45% of initial capital costs). O&M costs include replacement costs for capital items whose useful life is less than 40 years, labor, energy, filter media, membranes, solids disposal, liquids disposal, and a miscellaneous allowance (10% of total O&M costs to account for unforeseen costs). Unit costs for each are based on the EPA's WBS unit costs (Table 4.17, and Tables C1 – C26 in Appendix C).

Life cycle costing methodology is used to calculate costs for the functional unit in present worth values using standard economic equivalence parameters (Table 4.16); an example is shown in Figure 10. If desired, the user is able to adjust the parameter values (discount rate and escalation rate). Equations used for life cycle costing are as follows:

$$LCC = CC + R + E + Chem + O\&M - S$$

$$CC = \text{Materials} \times \text{Cost Factor}$$

$$R = \frac{T_v(1 + e)^t}{(1 + d)^t}$$

$$E = A \times \sum_0^n \left(\frac{I_{2021+y}}{(1 + d_i)^y} \right)$$

$$Chem = Chem_{t_1, d_1} + (Chem_{t_2, d_2} - Chem_{t_1, d_2}) + (Chem_{t_3, d_3} - Chem_{t_2, d_3})$$

$$Chem_{t_i, d_i} = \frac{A(1 + d_i)^{t_i} + 1}{d(1 + d_i)^{t_i}}$$

$$O\&M = O\&M_{t_1, d_1} + (O\&M_{t_2, d_2} - O\&M_{t_1, d_2}) + (O\&M_{t_3, d_3} - O\&M_{t_2, d_3})$$

$$O\&M_{t_i, d_i} = A \frac{1 + e}{d_i - e} \left[1 - \left(\frac{1 + e}{1 + d_i} \right)^{t_i} \right]$$

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

LCC = life cycle costs

CC = capital cost; one-time, initial costs

R = replacement costs

E = energy costs

Chem = chemical costs

O&M = operation and maintenance costs

S = salvage value

Materials = quantity of materials per replacement life

Cost Factor = unit cost of materials per unit quantity

T_v = value in base year dollars

t = unit useful life

d = discount rate (Table 2.12)

e = escalation rate (zero for materials, chemicals, and hauling, and 2.41% for labor (Table 2.12))

A = constant yearly annuity value in base year dollars

t_i = defined time interval between discrete discount rate changes

y = time interval (1 year)

I_{2018+} = Projected average fuel price index

n = study timeframe (40 years)

Capital Costs	
Pressure Vessel	\$ 159,123
Plastic/HDPE Tanks	\$ 8,782
Process PVC Piping	\$ 2,476
Backwash PVC Piping	\$ 996
In Out PVC Piping	\$ 758
Residuals PVC Piping	\$ 694
Process PVC Manual Valve	\$ 34,781
Residuals PVC Manual Valve	\$ 703
Residuals PVC Check Valve	\$ 320
Residuals Pump	\$ 20,295
Level Switches/Alarms (for vessels)	\$ 8,822
SS Sampling Ports	\$ 1,313
Building	\$ 60,545
Concrete Pad	\$ 6,273
Indirect Capital Costs	\$ 97,658
Capital Total	\$ 403,540

Construction

O&M Costs	
Manager Labor	\$ 243,866
Clerical Labor	\$ 164,291
Operator Labor	\$ 1,720,383
Facility maintenance (materials and labor)	\$ 202,103
Process pumping energy	\$ 3,925,311
Energy for backwash pumps	\$ 20,114
Energy for residuals pumps	\$ 11,192
Energy for lighting	\$ 4,754
Energy for ventilation	\$ 5,771
Resin	\$ 3,921,342
Materials for residuals pumps	\$ 2,802
Spent media disposal	\$ 246,650
POTW discharge fee	\$ 69,497
Miscellaneous Allowance	\$ 1,046,858
O&M Total	\$ 11,584,932

Labor
Energy Consumption
Waste Disposal

Figure 4.9. Example of Total Capital and O&M for an IX Filter Over Its Lifetime

Table 4.16 Economic Parameters Used in Life Cycle Costing

(Weitzman, 2001, USBLS, 2018).

Economic Parameters		
Discount rate	0-5 years	0.04
	6-25 years	0.03
	> 25 years	0.02
Escalation rate		0.0241

4.5.2 Facility Operation

The facility operation category includes footprint (square ft), maintenance (hrs/yr), and waste generation (gal/yr). The footprint value used as a decision criterion value is the same as the building area required. The maintenance criterion includes the total hours of labor required (media changeouts, membrane filter changeouts, inspection, etc.). The waste generation criterion compares the total volume of backwash discharge generated from GAC and IX treatment train options, as well as concentrate and cleaning residuals from RO and NF systems.

4.5.3 Environmental and Human Health Impacts

Global warming and global environmental and human health decision criteria are calculated by translating an option's inventory data into environmental impacts using the life cycle impact assessment method TRACI with life cycle inventory (LCI) unit processes from EcoInvent and US LCI (Table 4.17).

4.6 OVERALL SCORE AND RECOMMENDATION

First, normalized criterion scores are calculated for each technology option, and each criterion (a normalized score of zero represents the lowest costs, lowest environmental impacts, or lowest facility requirements). Next, using the user identified weights, the overall score for each option is calculated and visually represented using stacked column graphs to show the contribution of each decision criteria category (Figure 4.10). The recommended technology has the lowest value and is highlighted in the table of outputs along with other key criteria (footprint, bed lifetime, capital cost, O&M cost, and lifetime cost) (Figure 4.10).

$$\text{Normalized Criterion Score} = \frac{\text{technology criterion score} - \text{best criterion score}}{\text{worst criterion score} - \text{best criterion score}}$$

$$\text{Overall Score} = \sum_{i=1}^{n \text{ criterion}} (\text{Normalized Criterion Score}_i * \text{Stakeholder Weight}_i)$$

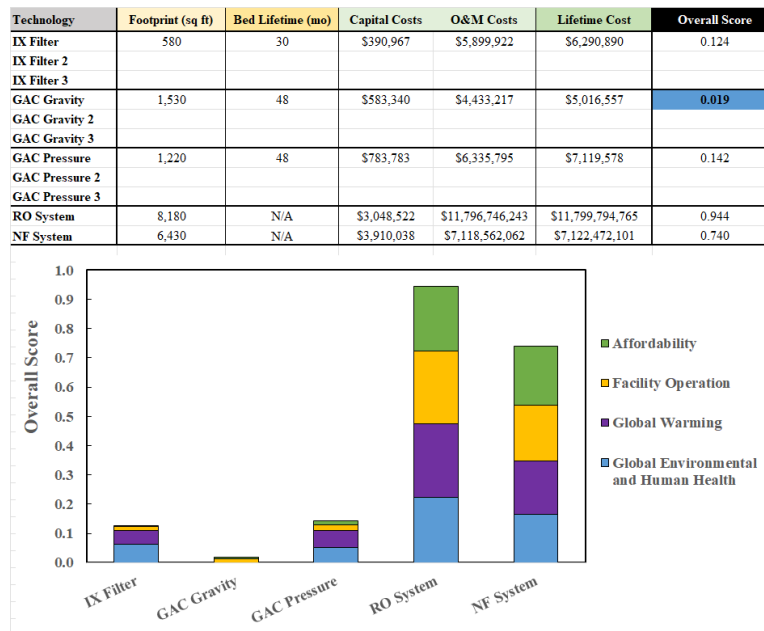


Figure 4.10. Tool's OUTPUT Tab Showing Analysis Results in Decision Support Tool

Table 4.17. Life cycle Inventory Unit Processes

	Unit Process		Units	Source	Use	Cost	Quantification
Filter Media							
1	Anion Exchange Resin	Anionic resin {RoW} market for anionic resin APOS, U	1 kg	Ecoinvent		\$300 /cf	Section 2.3.2
2	GAC	Activated carbon, granular {GLO} market for activated carbon, granular APOS, U	1 kg	Ecoinvent		\$1.90 /lb	
Materials							
3	Aluminum	Aluminum, cold rolling, at plant/kg/RNA	1 kg	Ecoinvent	Stairs, Gravity basin railing, pumps	Table C5	Section 2.3.3
4	Concrete for Basins	Concrete, 20MPa {GLO} market for APOS, U	1 m3	Ecoinvent	Building foundation, Gravity basin	\$601 /cy	Section 2.3.3
5	Electronics	Electronics, for control units {GLO} market for APOS, U	1 kg	Ecoinvent	Instrumentation	Table C1	Tables 2.8, 2.9
6	Fiberglass	Glass fibre reinforced plastic, polyamide, injection moulded {RoW} production APOS, U	1 kg	Ecoinvent	Pressure vessels, Roofing felt	Table C4	Section 2.3.4
7	HDPE	Polyethylene, high density, granulate, recycled {RoW} market for polyethylene, high density, granulate, recycled APOS, U	1 kg	Ecoinvent	Residuals storage, Chemical makeup	Table C12	Section 2.3.7
8	Polypropylene	Polypropylene, granulate {RoW} production APOS, U	1 kg	Ecoinvent	Cartridge filter, Membrane spacer	Table C17	Section 2.3.5
9	Polysulfone	Polysulfone {GLO} market for APOS, U	1 kg	Ecoinvent	Membranes	Table C16	
10	PVC	Polyvinylchloride, suspension polymerised {RoW} polyvinylchloride production, suspension polymerisation APOS, U	1 kg	Ecoinvent	Piping, Valves, Pressure vessel internals	Table C7, C8, C9	Tables 2.6, 2.7
11	Stainless Steel	Steel, chromium steel 18/8 {GLO} market for APOS, U	1 kg	Ecoinvent	Sampling ports, membrane housing	\$50 /unit	Table 2.6
12	Unalloyed Steel	Steel, unalloyed {RoW} steel production, converter, unalloyed APOS, U	1 kg	Ecoinvent	Carbon steel pressure vessels	Table C14, C15	Section 2.3.4
Building							
13	Bitumen (Asphalt)	Bitumen adhesive compound, hot {GLO} market for APOS, U	1 kg	US LCI	Roofing	Table C26	Section 2.5
14	Concrete Blocks	Concrete block {RoW} market for concrete block APOS, U	1 kg	Ecoinvent	Building walls		
15	Galvanized Steel	Galvanized steel sheet, at plant/RNA	1 kg	US LCI	Trusses		
16	Gravel	Gravel, crushed {RoW} market for gravel, crushed APOS, U	1 kg	Ecoinvent	Roofing		
17	Plywood	Plywood, for outdoor use {RoW} market for APOS, U	1 m3	Ecoinvent	Roofing		
18	Wood	Sawnwood, softwood, dried (u=10%), planed {RoW} market for APOS, U	1 m3	Ecoinvent	Purlins	\$648 /cy	Section 2.5.1.3
19	Concrete Pad	Concrete, 20MPa {GLO} market for APOS, U	1 m3	Ecoinvent	Building floor		
20	Reinforcing Steel	Reinforcing steel {GLO} market for APOS, U	1 kg	Ecoinvent	Concrete pad reinforcement	Included in 19	
Chemicals							
21	Citric Acid	Citric acid {GLO} market for APOS, U	1 kg	Ecoinvent	Membrane low pH cleaner	\$1.12 /lb	Section 2.6
22	NaOH	Sodium hydroxide, without water, in 50% solution state {GLO} market for APOS, U	1 kg	Ecoinvent	Membrane high pH cleaner	\$0.30 /lb	
23	Polyacrylamide	Polyacrylamide {GLO} market for APOS, U	1 kg	Ecoinvent	Anitscalant	\$1.84 /lb	
24	Polycarboxylates	Polycarboxylates, 40% active substance {RoW} market for polycarboxylates, 40% active substance APOS, U	1 kg	Ecoinvent	Antiscalant		
Transformation							
25	GAC Thermal Reactivation	Activated carbon, granular {RoW} treatment of spent activated carbon, granular from hard coal, reactivation APOS, U	1 kg	Ecoinvent	GAC thermal reactivation	\$269 /ton	Section 2.5.1
26	Haz Waste Incineration	Hazardous waste, for incineration {RoW} market for hazardous waste, for incineration APOS, U	1 kg	Ecoinvent	GAC and IX resin incineration		Same as 1, 2
27	Plastic Incineration	Waste plastic, mixture {RoW} treatment of waste plastic, mixture, municipal incineration APOS, U	1 kg	Ecoinvent	Membrane/cartridge filter incineration		Same as 8, 9
28	Landfilling	Municipal solid waste {RoW} treatment of, sanitary landfill APOS, U	1 kg	Ecoinvent	GAC, IX resin, and membrane landfill		Same as 1, 2
Hauling							
29	Hauling	Transport, freight, lorry, unspecified {RoW} market for transport, freight, lorry, unspecified APOS, U	1 tkm	Ecoinvent		Table C10	Section 2.7
Energy							
30	Electricity - ASCC	Electricity, medium voltage {ASCC} market for Alloc Def, U ASCC grid	1 kWh	Ecoinvent		\$0.189 /kWh	Section 2.4
31	Electricity - FRCC	Electricity, medium voltage {FRCC} market for Alloc Def, U FRCC grid	1 kWh	Ecoinvent		\$0.137 /kWh	
32	Electricity - NPCC	Electricity, medium voltage {NPCC, US only} market for Alloc Def, U NPCC grid	1 kWh	Ecoinvent		\$0.159 /kWh	
33	Electricity - MRO	Electricity, medium voltage {MRO, US only} market for Alloc Def, U MRO grid	1 kWh	Ecoinvent		\$0.090 /kWh	
34	Electricity - RFC	Electricity, medium voltage {RFC} market for Alloc Def, U RFC grid	1 kWh	Ecoinvent		\$0.124 /kWh	
35	Electricity - SERC	Electricity, medium voltage {SERC} market for Alloc Def, U SERC grid	1 kWh	Ecoinvent		\$0.098 /kWh	
36	Electricity - SPP	Electricity, medium voltage {SPP} market for Alloc Def, U SPP grid	1 kWh	Ecoinvent		\$0.080 /kWh	
37	Electricity - TRE	Electricity, medium voltage {TRE} market for Alloc Def, U TRE grid	1 kWh	Ecoinvent		\$0.085 /kWh	
38	Electricity - WECC	Electricity, medium voltage {WECC, US only} market for Alloc Def, U WECC grid	1 kWh	Ecoinvent		\$0.106 /kWh	
39	Electricity - HICC	Electricity, medium voltage {HICC} market for Alloc Def, U	1 kWh	Ecoinvent		\$0.254 /kWh	

4.7 RESULTS

4.7.1 Navigating Treatment Performance Uncertainty using Scenario Analysis

Since treatment performance data can be lacking or uncertain, we are conducting a scenario analysis that looks at a large range of technology treatment performances (i.e., media usage rates and membrane recovery percentages). This analysis will help evaluate the advantages and disadvantages of each technology option and can be used to determine which has lower environmental impacts or costs for a given scenario.

First, the analysis evaluates a large range of media usage rates (0.0125 to 0.50 lb/1000 gal) for GAC and IX. This range represents a large range of water qualities (e.g., differences in organic matter and/or influent PFAS concentrations), media (e.g., GAC type, such as F400, 1240CAT; IX resins, such as 2301, PFA694E), and treatment objectives. To represent a large range of possible performances for RO and NF systems, a range of multipliers (0.8 to 1.2) is applied to the standard membrane recovery rate (Table 4.10).

A series of scenarios have been evaluated using these media usage rates and recovery rate multipliers that investigate the comparison between treatment technologies as important design parameters are adjusted, including flow rate, media disposal option, pump efficiency, and treatment location. Figures 4.11, 4.13, and 4.15; and 4.12, 4.14, and 4.16 show the comparisons of global warming impacts and lifetime cost, respectively, for pairwise comparisons of IX filter with prefilter, GAC pressure filter, GAC gravity basin, NF system, RO system, and IX filter with backwash at flowrates of 0.03, 0.31, and 2.2 mgd.

These figures show several important trends when comparing treatment processes for the removal of PFAS: 1) media usage rate for IX resin and GAC can significantly impact which technology has lower environmental impacts or costs, Figure 4.11a, 4.13a, 4.15a; 2) IX filter with prefilter generally has lower global warming impacts than GAC pressure filter (GAC media usage rate would need to be significantly lower than IX media usage rate to have lower impacts), Figures 4.11a, 13a, 15a; 3) GAC pressure filter generally has lower lifetime cost than IX filter (due to the high cost of resin), Figures 4.12a, 14a, 16a; 4) change in recovery rate for RO and NF systems do not significantly impact comparisons of global warming impacts and lifetime cost, Figures 4.11d-i, 4.13d-i, 4.15d-i, 4.12d-i, 4.14d-i, 4.16d-i; 5) IX and GAC systems have lower environmental impacts than RO and NF at lower media usage rates, Figures 4.11d-i, 4.13d-i, 4.15d-I; 6) IX and GAC systems have lower lifetime costs than RO and NF systems, regardless of media usage rate, Figures 4.12d-i, 4.14d-i, 4.16d-I; 7) RO and NF have similar global warming impacts and lifetime costs at 0.03 and 0.31 mgd, but NF has lower impacts and costs at 2.2 mgd, Figures 4.11j, 4.13j, 4.15j, 4.12j, 4.14j, 4.16j; 8) using backwash instead of prefilters for IX filter does not significantly impact technology comparison with GAC pressure filter, Figures 4.11k-l, 4.13k-l, 4.15k-l, 4.12k-l, 4.14k-l, 4.16k-l.

Disposal of media by hazardous waste landfill, PFAS-specific landfill, or incineration does not significantly impact the comparison between GAC pressure filter or IX filter across flow rate (Figure 4.17 and Figure 4.18) and location (Figure 4.21 and Figure 4.22). Decreasing pump efficiency from 75% to 60% does not change the comparison; however, increasing pump efficiency to 90% shifts the comparison of global warming impacts to be even more favorable to IX filter with prefilter for all media disposal options (Figure 4.19). Disposing of GAC by thermal reactivation significantly decreases net environmental impacts and shifts the comparison of global warming impacts of GAC

pressure filter to be almost equally favorable to IX filter (Figure 4.17). Excluding filter redundancy and/or building for the treatment process does not significantly alter comparisons between GAC pressure filter and IX filter with prefilter, even across flow rates (Figure 4.23 and Figure 4.24).

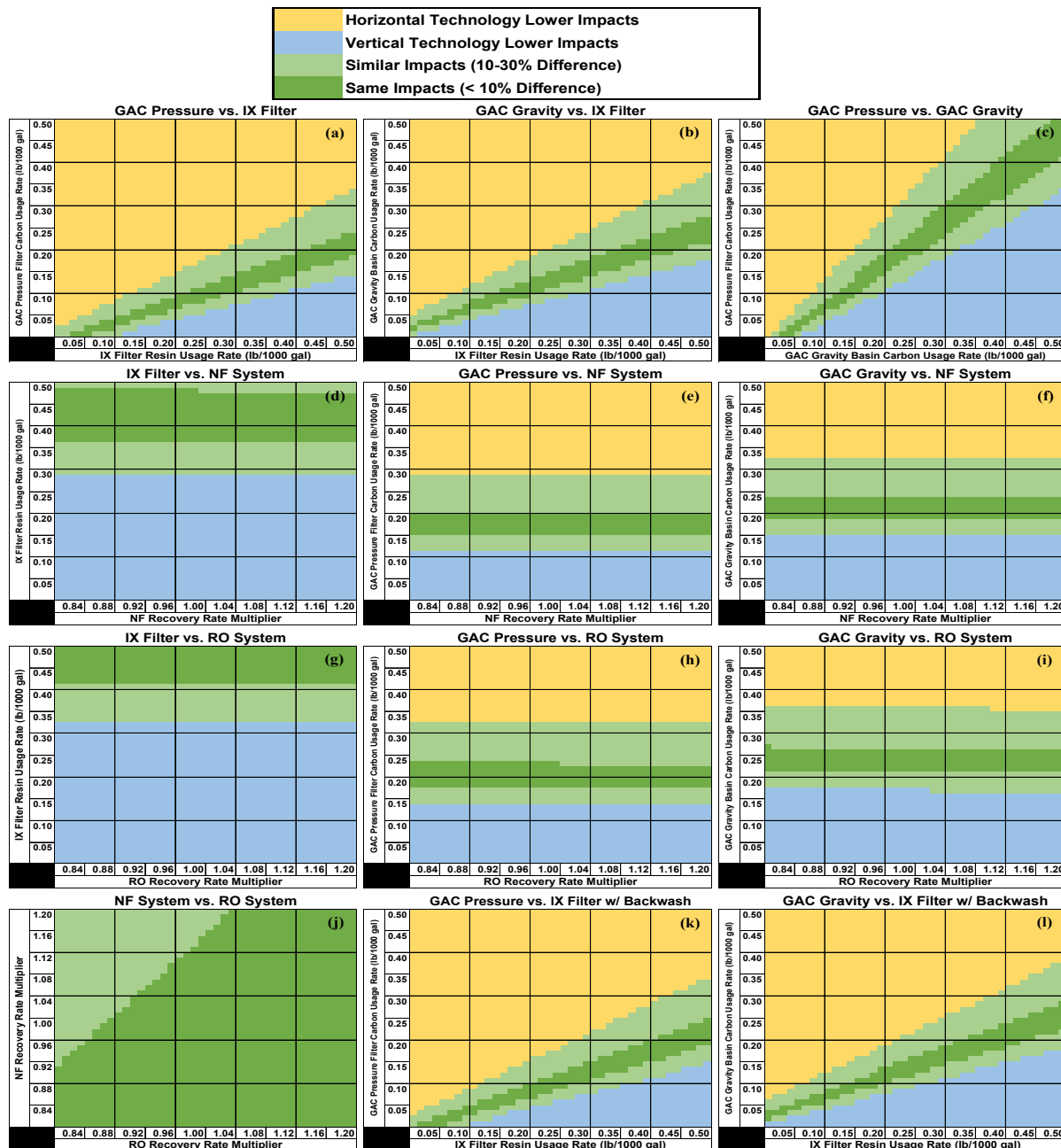


Figure 4.11. Comparison of Global Warming Impacts

a) GAC pressure filter versus IX filter with prefilter, b) GAC gravity basin versus IX filter with prefilter, c) GAC pressure filter versus GAC gravity basin, d) IX filter with prefilter versus NF, e) GAC pressure filter versus NF, f) GAC gravity basin versus NF, g) IX filter with prefilter versus RO, h) GAC pressure filter versus RO, i) GAC gravity basin versus RO, j) NF versus RO, k) GAC pressure filter versus IX filter with backwash, and l) GAC gravity basin versus IX filter with backwash. All comparisons are calculated at 0.03 mgd with media and membrane disposal in a hazardous waste landfill.

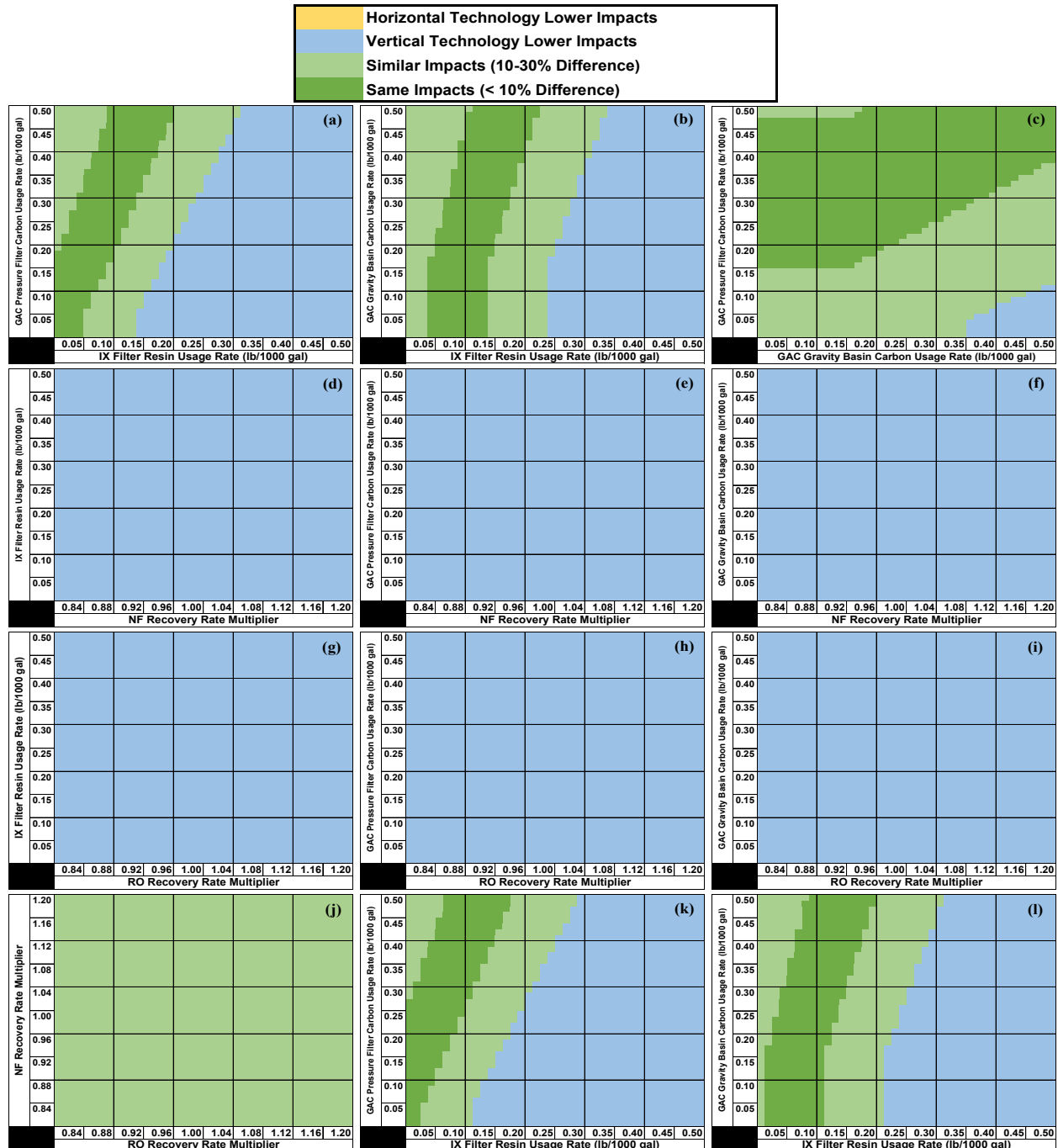


Figure 4.12. Comparison of Lifetime Cost

a) GAC pressure filter versus IX filter with prefilter, b) GAC gravity basin versus IX filter with prefilter, c) GAC pressure filter versus GAC gravity basin, d) IX filter with prefilter versus NF, e) GAC pressure filter versus NF, f) GAC gravity basin versus NF, g) IX filter with prefilter versus RO, h) GAC pressure filter versus RO, i) GAC gravity basin versus RO, j) NF versus RO, k) GAC pressure filter versus IX filter with backwash, and l) GAC gravity basin versus IX filter with backwash. All comparisons are calculated at 0.03 mgd with media and membrane disposal in a hazardous waste landfill.

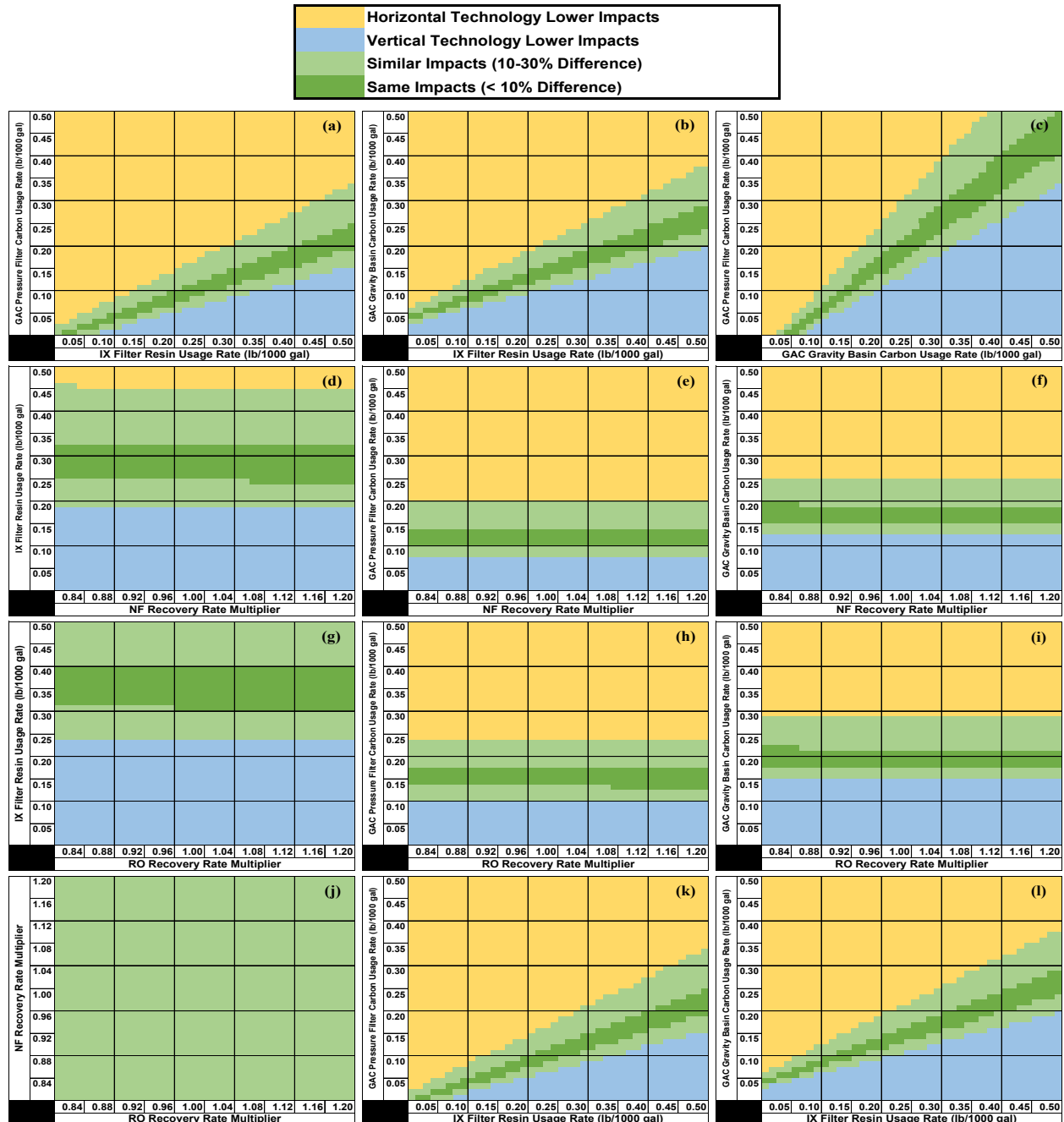


Figure 4.13. Comparison of Global Warming Impacts

a) GAC pressure filter versus IX filter with prefilter, b) GAC gravity basin versus IX filter with prefilter, c) GAC pressure filter versus GAC gravity basin, d) IX filter with prefilter versus NF, e) GAC pressure filter versus NF, f) GAC gravity basin versus NF, g) IX filter with prefilter versus RO, h) GAC pressure filter versus RO, i) GAC gravity basin versus RO, j) NF versus RO, k) GAC pressure filter versus IX filter with backwash, and l) GAC gravity basin versus IX filter with backwash. All comparisons are calculated at 0.31 mgd with media and membrane disposal in a hazardous waste landfill.

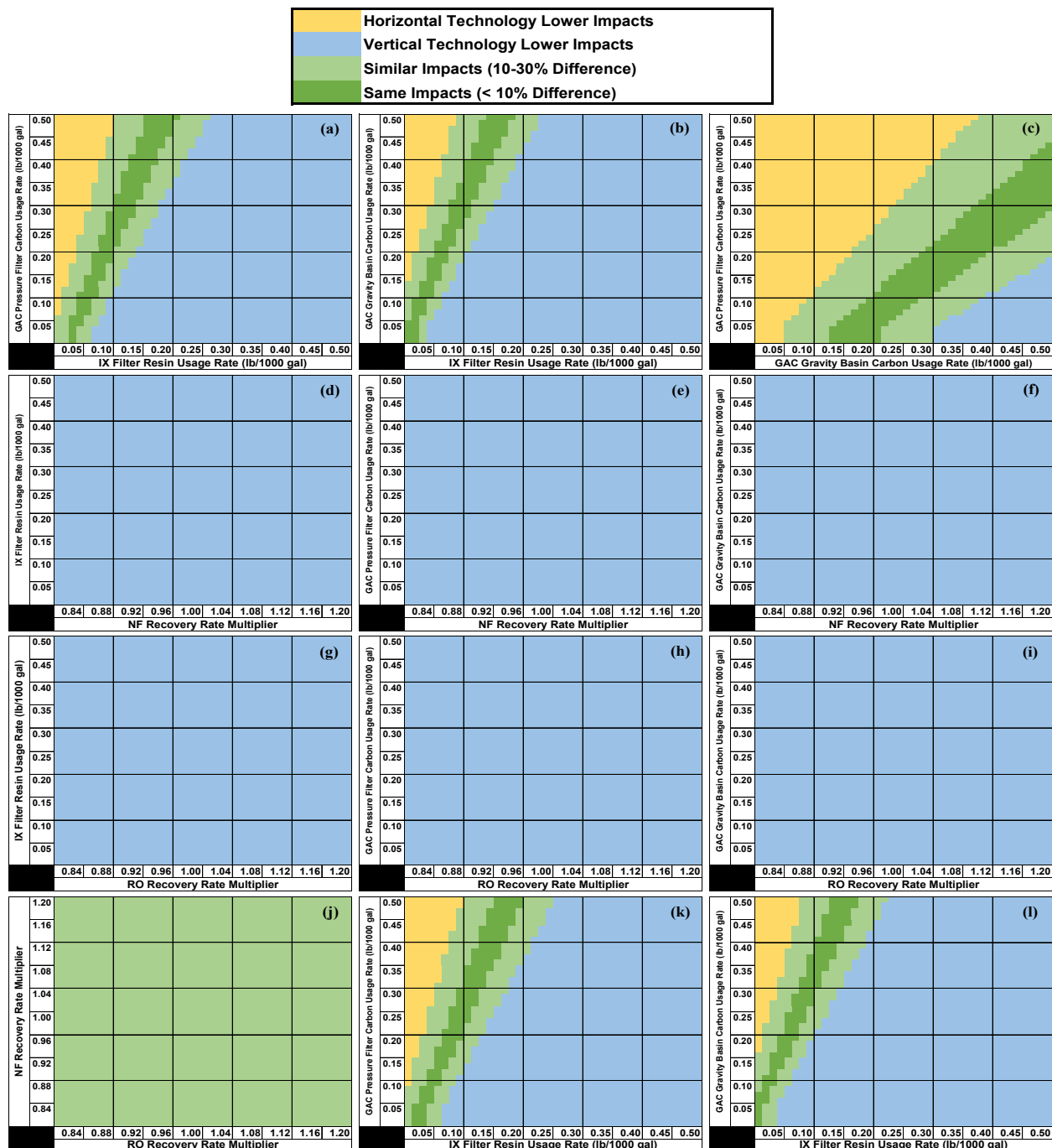


Figure 4.14. Comparison of Lifetime Cost

a) GAC pressure filter versus IX filter with prefilter, b) GAC gravity basin versus IX filter with prefilter, c) GAC pressure filter versus GAC gravity basin, d) IX filter with prefilter versus NF, e) GAC pressure filter versus NF, f) GAC gravity basin versus NF, g) IX filter with prefilter versus RO, h) GAC pressure filter versus RO, i) GAC gravity basin versus RO, j) NF versus RO, k) GAC pressure filter versus IX filter with backwash, and l) GAC gravity basin versus IX filter with backwash. All comparisons are calculated at 0.31 mgd with media and membrane disposal in a hazardous waste landfill.

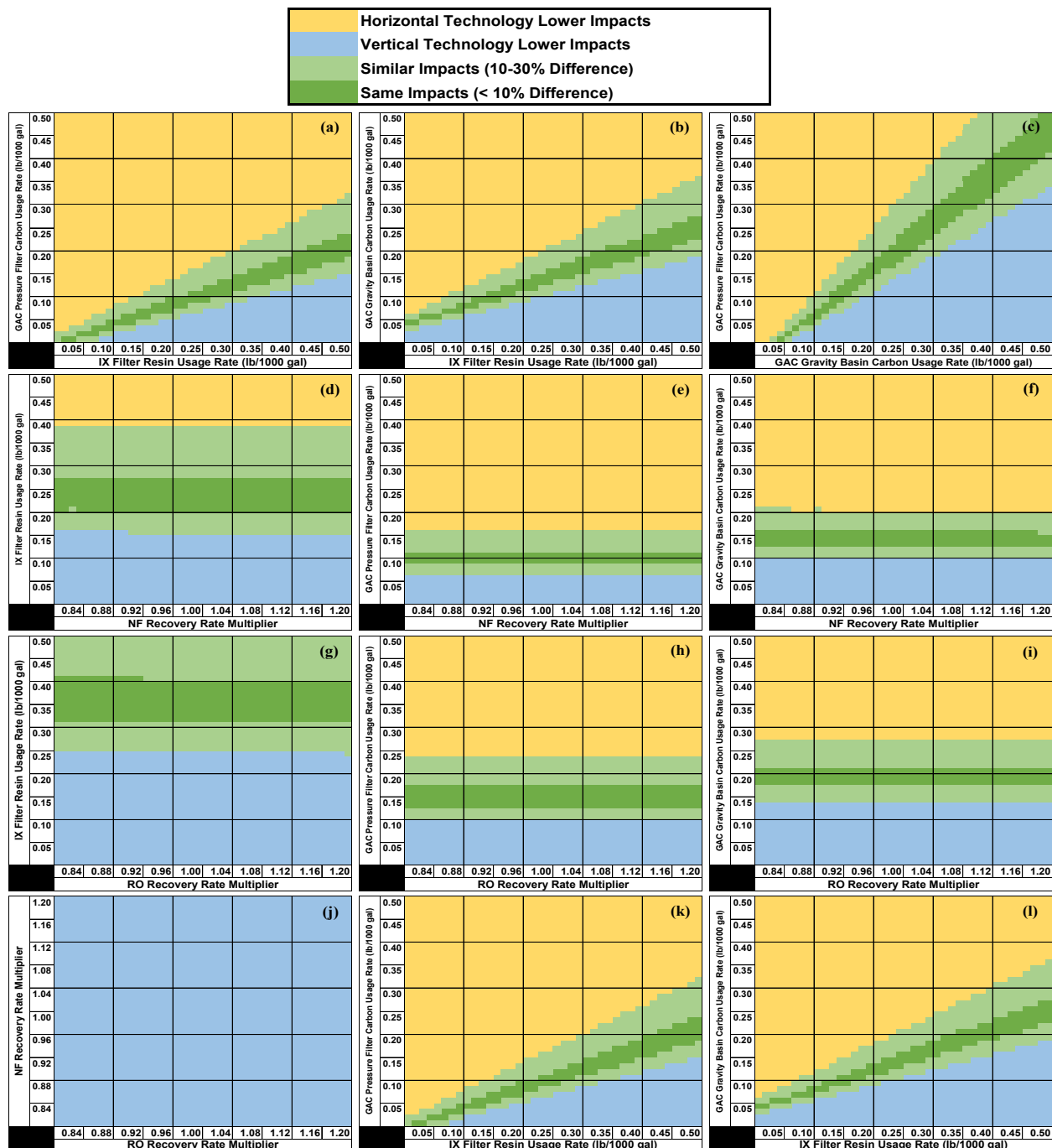


Figure 4.15. Comparison of Global Warming Impacts

a) GAC pressure filter versus IX filter with prefilter, b) GAC gravity basin versus IX filter with prefilter, c) GAC pressure filter versus GAC gravity basin, d) IX filter with prefilter versus NF, e) GAC pressure filter versus NF, f) GAC gravity basin versus NF, g) IX filter with prefilter versus RO, h) GAC pressure filter versus RO, i) GAC gravity basin versus RO, j) NF versus RO, k) GAC pressure filter versus IX filter with backwash, and l) GAC gravity basin versus IX filter with backwash. All comparisons are calculated at 2.2 mgd with media and membrane disposal in a hazardous waste landfill.

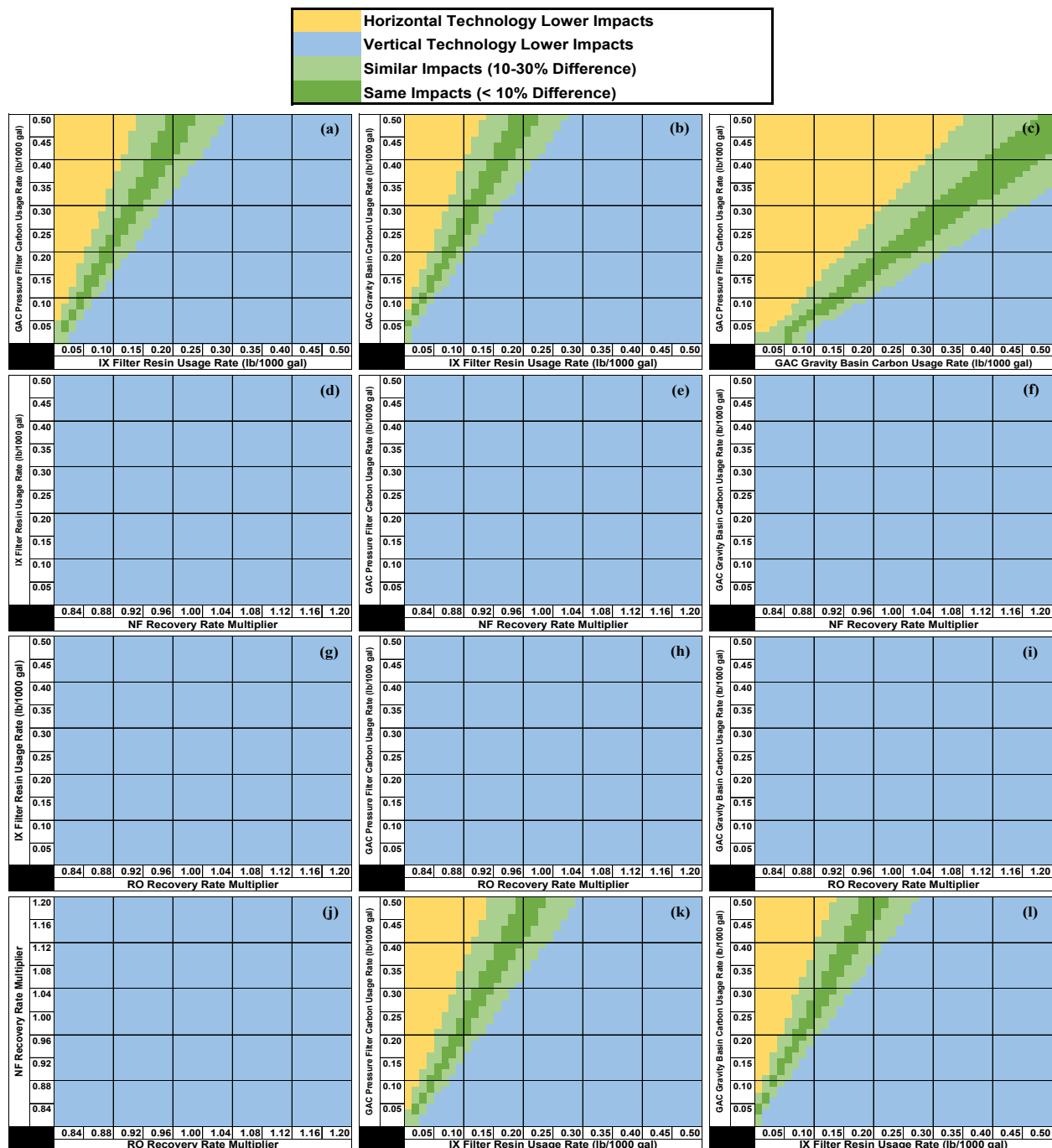


Figure 4.16. Comparison of Lifetime Cost for GAC Pressure Filter versus IX Filter with Prefilter

b) GAC gravity basin versus IX filter with prefilter, c) GAC pressure filter versus GAC gravity basin, d) IX filter with prefilter versus NF, e) GAC pressure filter versus NF, f) GAC gravity basin versus NF, g) IX filter with prefilter versus RO, h) GAC pressure filter versus RO, i) GAC gravity basin versus RO, j) NF versus RO, k) GAC pressure filter versus IX filter with backwash, and l) GAC gravity basin versus IX filter with backwash. All comparisons are calculated at 2.2 mgd with media and membrane disposal in a hazardous waste landfill.

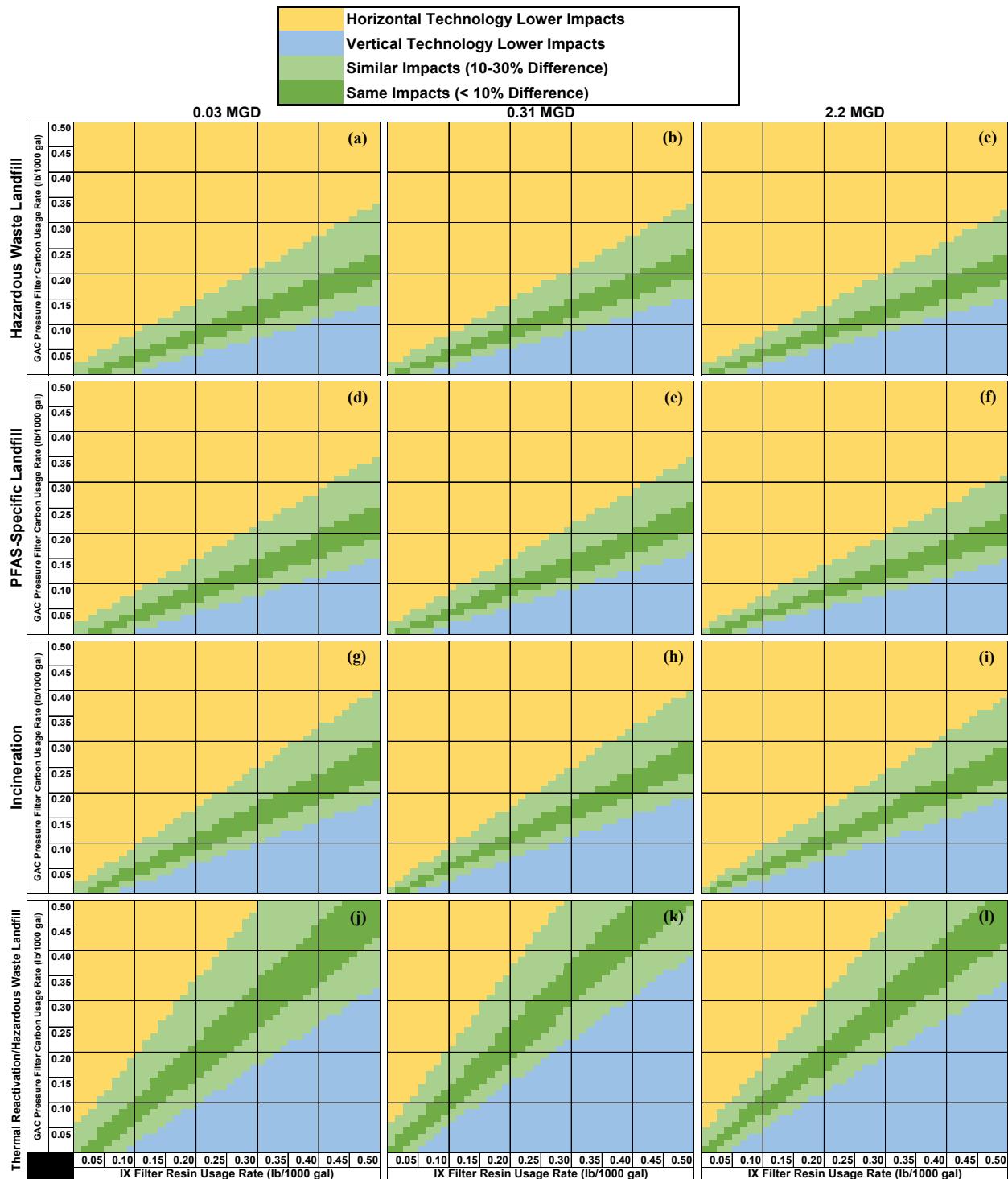


Figure 4.17. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

at 0.03 (left), 0.31 (center), and 2.2 (right) mgd with filter media disposed by hazardous waste landfill (1st row), PFAS specific landfill (2nd row), incineration (3rd row), and thermal reactivation (GAC) and hazardous waste landfill (IX) (4th row).

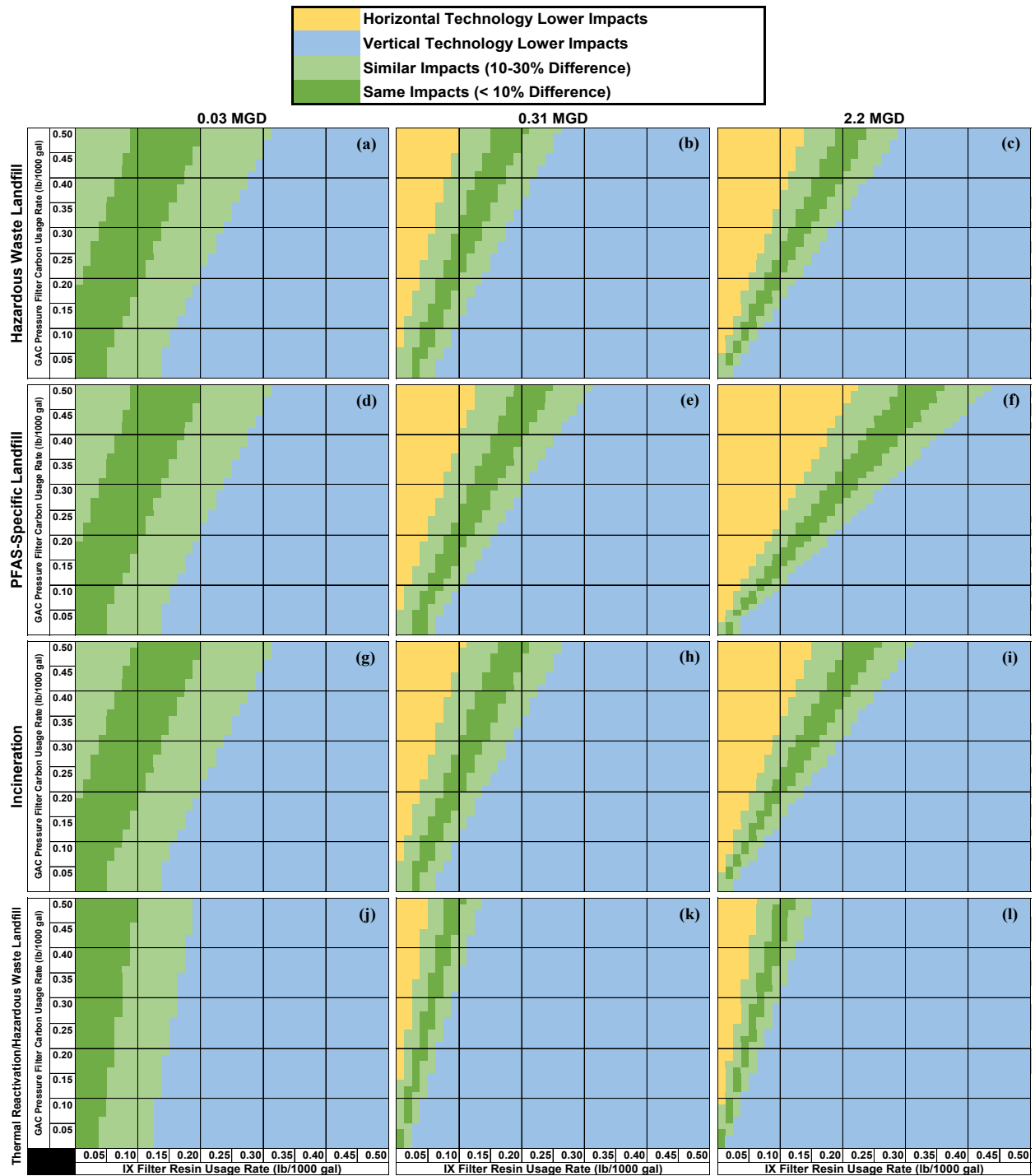


Figure 4.18. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

at 0.03 (left), 0.31 (center), and 2.2 (right) mgd with filter media disposed by hazardous waste landfill (1st row), PFAS specific landfill (2nd row), incineration (3rd row), and thermal reactivation (GAC) and hazardous waste landfill (IX) (4th row).

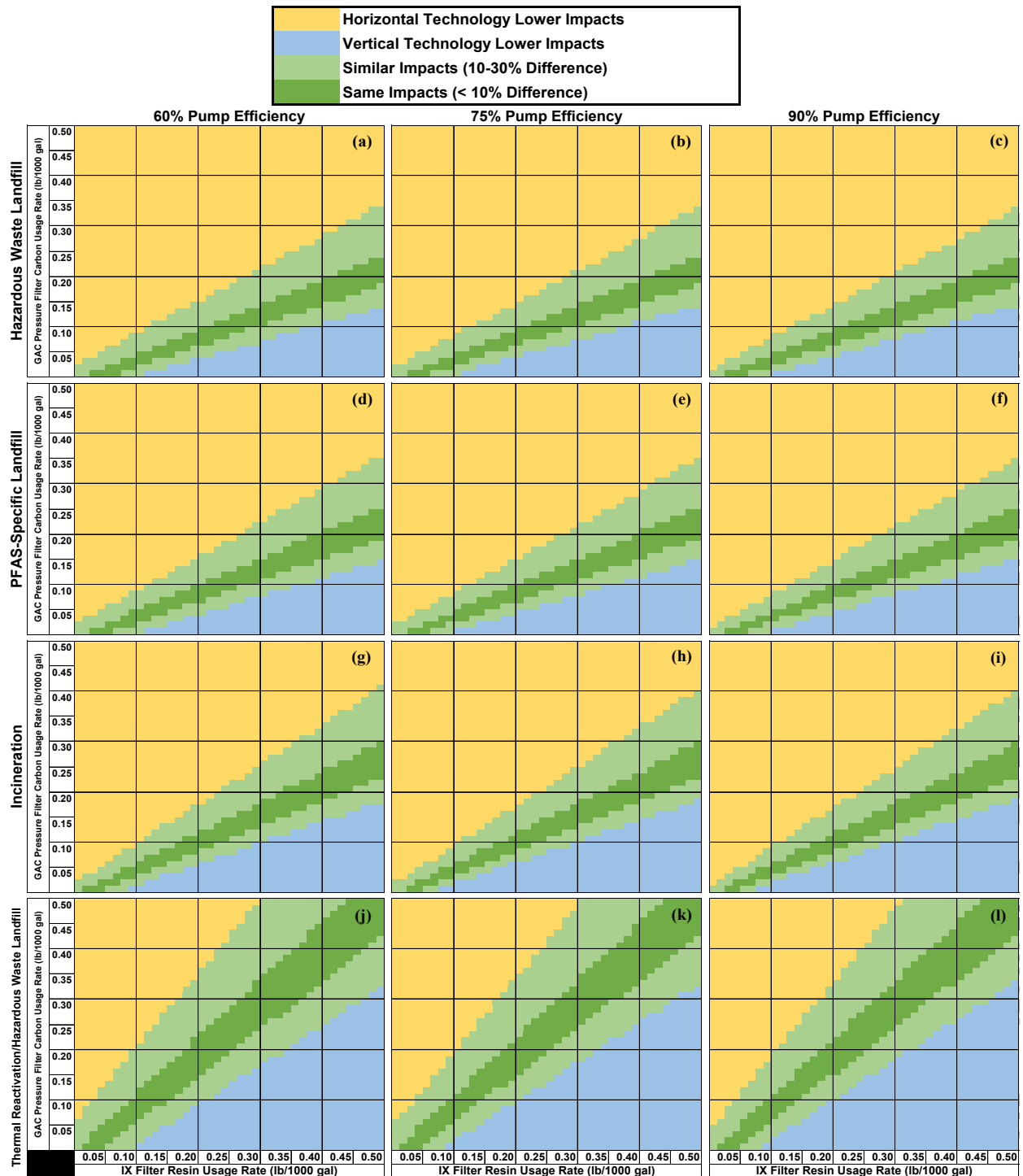


Figure 4.19. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

at 0.03 mgd with pump efficiency of 60% (left), 75% (center), and 90% (right) with filter media disposed by hazardous waste landfill (1st row), PFAS specific landfill (2nd row), incineration (3rd row), and thermal reactivation (GAC) and hazardous waste landfill (IX) (4th row).

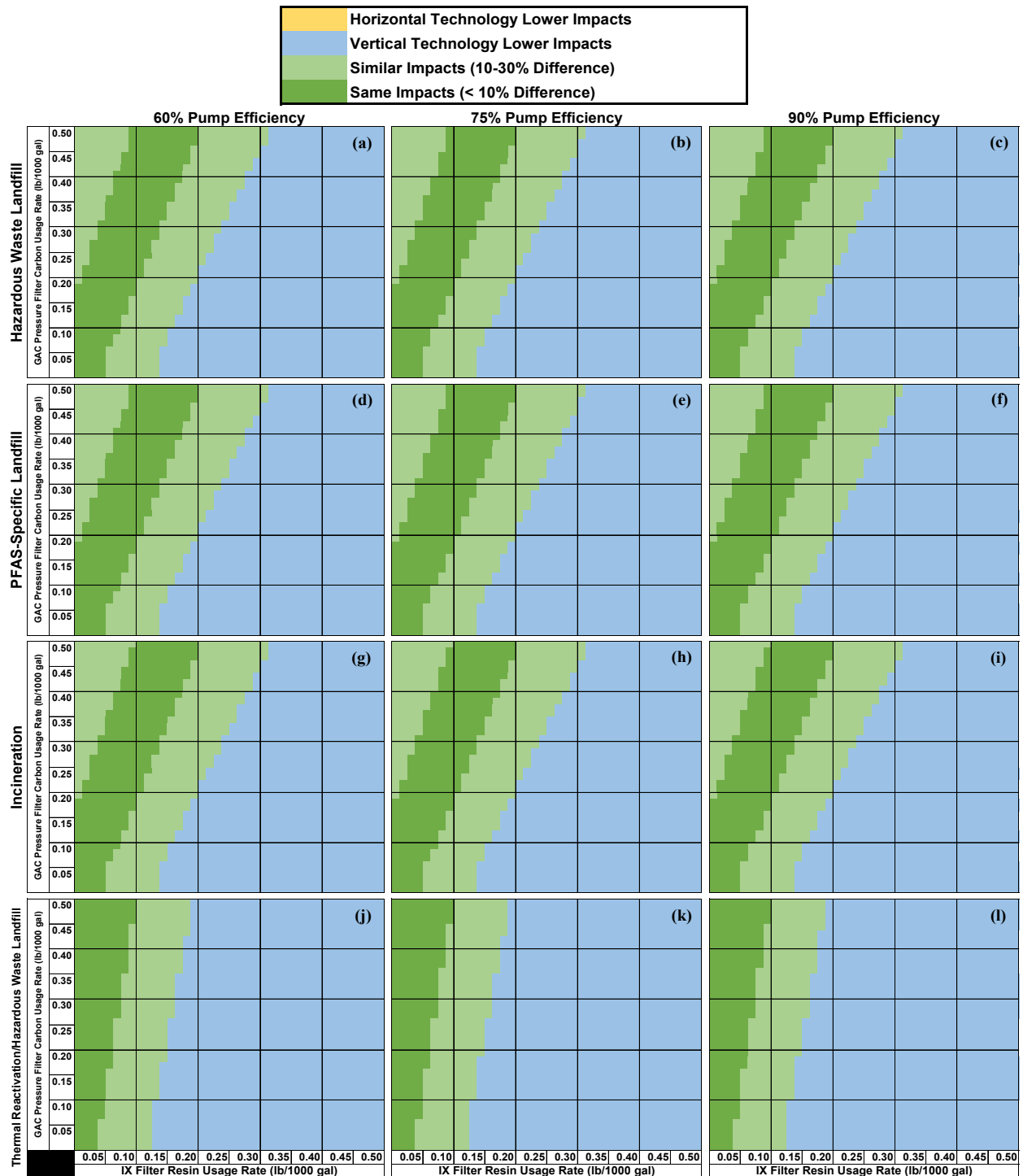


Figure 4.20. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

at 0.03 mgd with pump efficiency of 60% (left), 75% (center), and 90% (right) with filter media disposed by hazardous waste landfill (1st row), PFAS specific landfill (2nd row), incineration (3rd row), and thermal reactivation (GAC) and hazardous waste landfill (IX) (4th row).

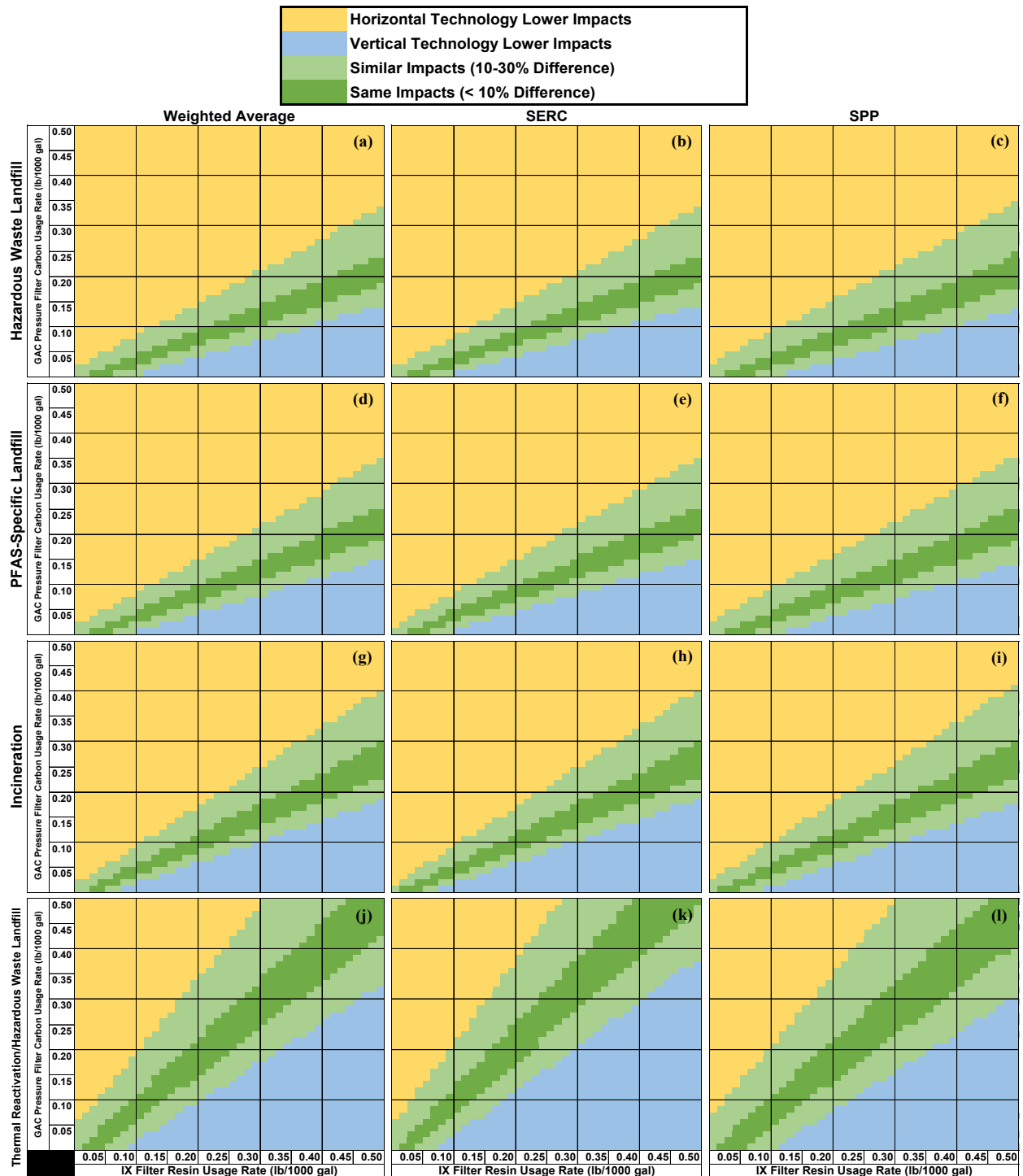
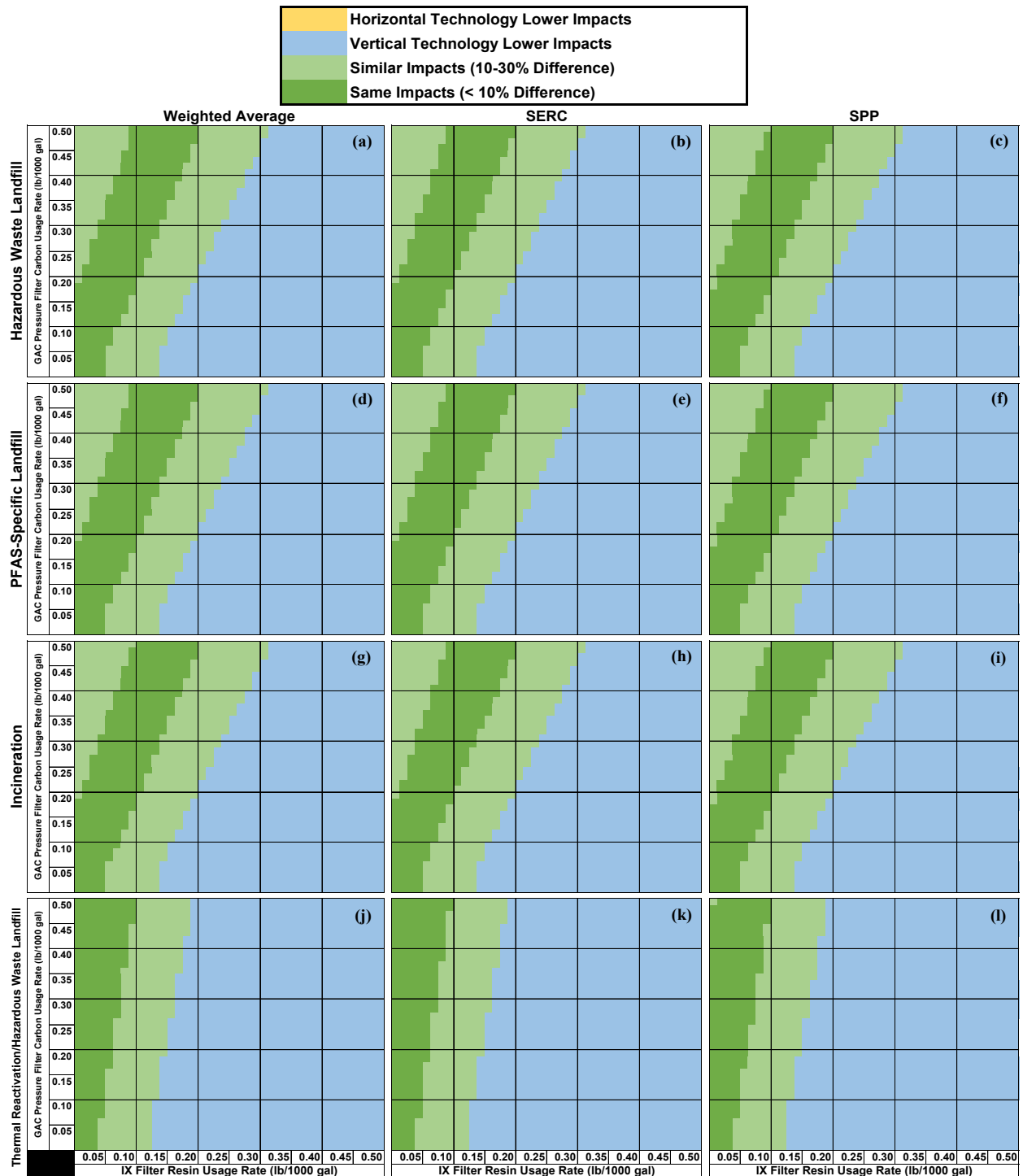


Figure 4.21 Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

at 0.03 mgd with location of weighted average (left), SERC (center), and SPP (right) with filter media disposed by hazardous waste landfill (1st row), PFAS specific landfill (2nd row), incineration (3rd row), and thermal reactivation (GAC) and hazardous waste landfill (IX) (4th row).



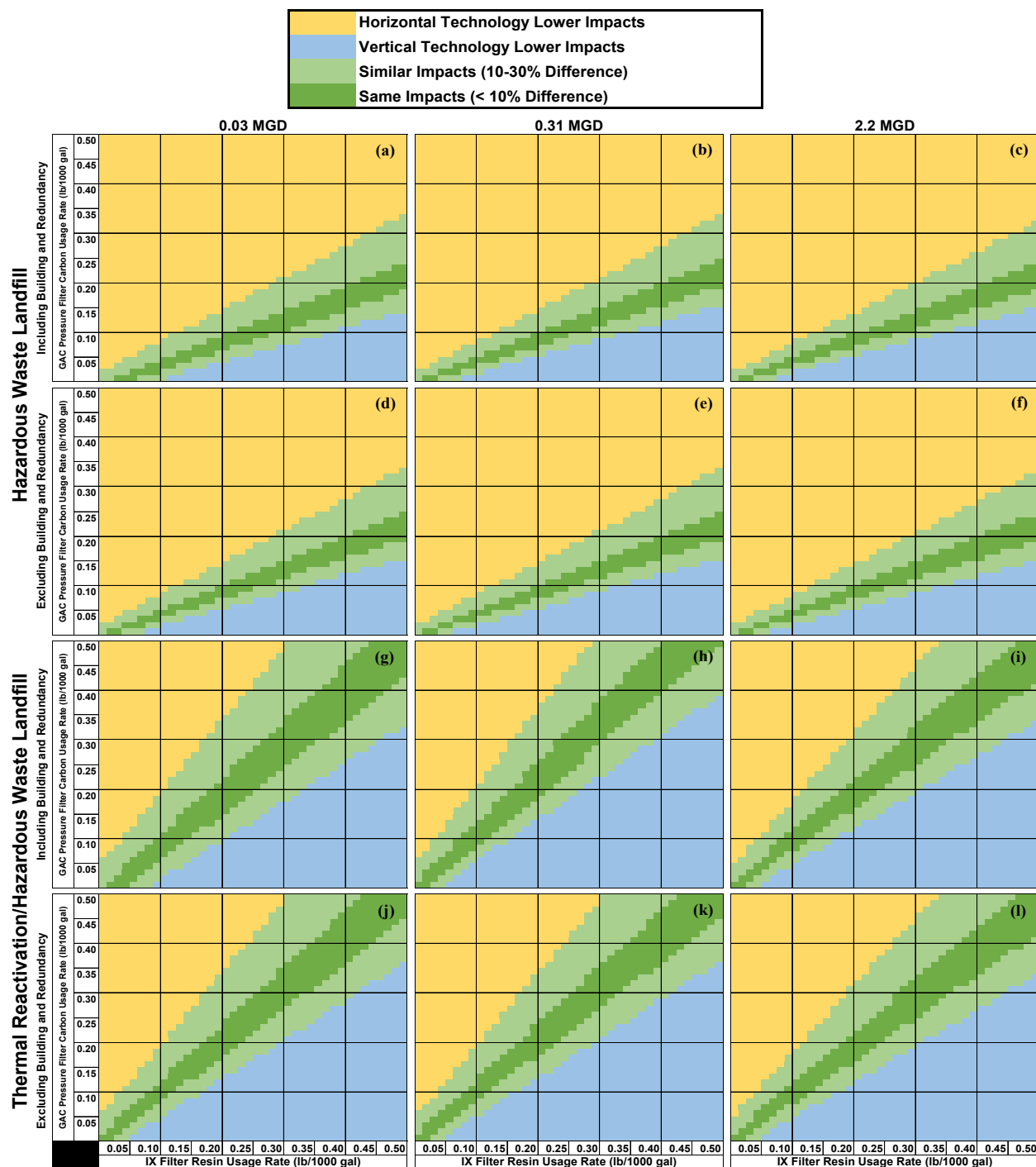


Figure 4.23. Comparison of Global Warming Impacts for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

with media disposed by hazardous waste landfill with a building (1st row) and without building or redundant filter (2nd row), and with media disposed by thermal reactivation (GAC) and hazardous waste landfill (IX) without a building (3rd row) and with a building or redundant filter (4th row); at flowrates of 0.03 MGD (left), 0.31 MGD (center), and 2.2 MGD (right).

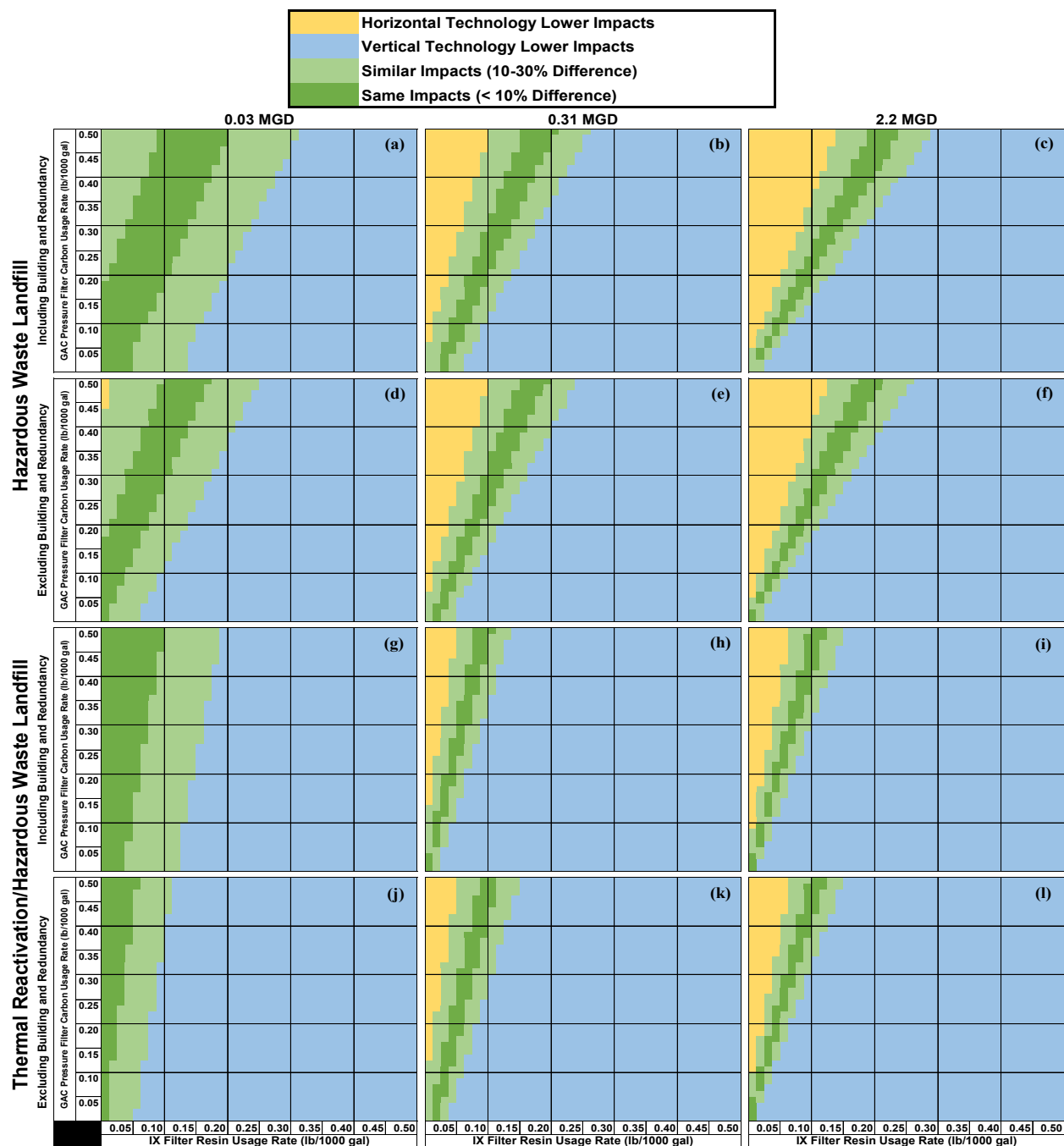


Figure 4.24. Comparison of Lifetime Cost for GAC Pressure Filter (Vertical) Versus IX Filter with Prefilter (Horizontal)

with media disposed by hazardous waste landfill with a building (1st row) and without building or redundant filter (2nd row), and with media disposed by thermal reactivation (GAC) and hazardous waste landfill (IX) without a building (3rd row) and with a building or redundant filter (4th row); at flowrates of 0.03 MGD (left), 0.31 MGD (center), and 2.2 MGD (right)

4.7.2 Advantages and Disadvantages of Treatment Processes

A table summarizing the advantages and disadvantages (along with improvement opportunities) for each treatment technology with respect to our specific analysis is shown below (Table 4.18).

Table 4.18. Summary of the Advantages and Disadvantages of the Removal of PFAS Using IX Filter, GAC Pressure Filter, and RO/NF Systems Identified through the Use of LCA.

Technologies	Advantages	Disadvantages
IX Contactor	• Low treatment footprint • Lower media usage rate than GAC, generally.	• High cost of resin – <i>Reduce cost/improve removal capacity</i> • Residuals discharge – <i>Decrease backwash volume</i> • Significantly impacted by water quality
GAC Pressure Contactor	• Lower treatment footprint than RO/NF • Thermal reactivation possible	• Higher media usage rate than IX, generally • Higher environmental impacts than IX filter (except ozone depletion) with media disposal by landfill or incineration – <i>Investigate use of coconut derived GAC</i>
RO/NF Systems	• Lower environmental impacts when GAC and IX media usage rate (changeout) is high • Water quality does not significantly impact performance (change in recovery rate does not impact comparison) • High removal of PFAS regardless of water quality	• Largest treatment footprint • Highest treatment cost – <i>Reduce pumping requirements and concentrate volume</i> • Concentrate stream – <i>Disposal/treatment of concentrate stream</i> – <i>Increase recovery rate to reduce concentrate</i>

4.7.3 Case Study Analysis

The decision support tool can also be used to look at specific cases when treatment performance data is available. To highlight this ability, four case studies (based on treatment data from three sources: Fountain, CO (Liu et al., 2019); Orange County Water District (Pannu, 2021); and Zeng et al., 2020) were analyzed. These four case studies represent a range of media, influent PFAS concentrations, and water quality (Table 4.19). First, the treatment data from GAC media and IX resin was fit using the pore surface diffusion model (PSDM) in AdDesignS. The fitted model parameters were then used to produce breakthrough curves assuming a 0.31 mgd flow rate (Figure 4.25). Second, multiple groundwater regulations (i.e., treatment objectives) were applied to the breakthrough curves to estimate the resulting range of media usage rates (Table 4.20).

Third, the environmental performance and costs associated with those usage rates was evaluated, with the total range of usage rates from each case study shown as a box; global warming impacts are shown here as an example (Figure 4.26).

For the Fountain, OCWD, and Zeng 1240CAT/PFA694E data, the IX generally had lower media usage rates across the regulations used. Because of this, IX had lower global warming impacts when looking at those three case studies. However, global warming impacts were similar for both technology options when using the Zeng F400/PFA694E data.

As previously discussed, however, the specific design scenario can significantly impact the comparison of technologies. Using thermal reactivation for the spent GAC shifts the comparison such that the GAC pressure filter has lower or similar impacts for the OCWD and Zeng data (Figure 4.27b). When looking at lifetime cost, the GAC pressure filter has similar or lower impacts for the Zeng (F400) data, the GAC pressure filter and IX filter have similar cost for the OCWD and Zeng (1240CAT) data, and the IX filter has similar or slightly lower impacts for the Fountain data when media is disposed of by hazardous waste landfill (Figure 4.27c). However, when GAC media is disposed of by thermal reactivation, the GAC pressure filter has a lower cost for the Zeng (F400 and 1240CAT) data and a similar cost for the OCWD and Fountain data (Figure 4.27d).

Table 4.19. Influent PFAS Concentrations and TOC in the Four Waters Used for Case Studies

Conc (ng/L)	Fountain, CO	OCWD	Zeng, 1240CAT	Zeng, GAC400
PFPeA	16			
PFHxA	18	2.8	21	490
PFHpA	6.6		4.1	210
PFOA	22	16	6.2	120
PFNA		2.2		
PFPrS	3.3			
PFBS	2.4	14	18	42
PFPeS	9.4			
PFHxS	42	11	75	380
PFHpS	1.2			
PFOS	51	23	32	9.7
Total PFAS:	172	69	156	1252
TOC (mg/L)	2.7	1.2	0.21	0.40

GAC Breakthrough Curves

IX Breakthrough Curves

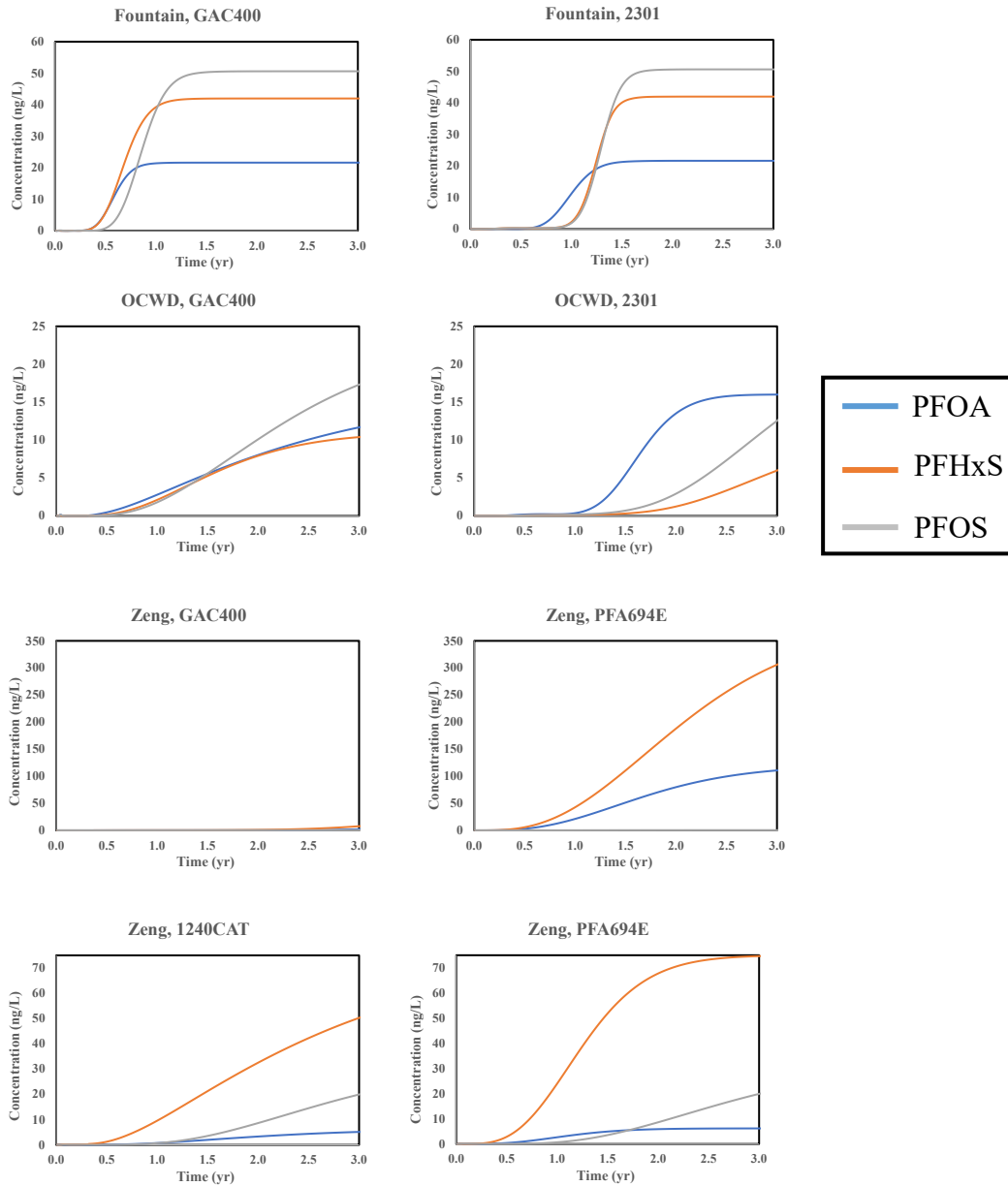


Figure 4.25. Breakthrough Curves (Modeled Using the PSDM in AdDesignS) of PFOA, PFHxS, and PFOS Modeled at 0.31 mgd Using PSDM for GAC (Left) and IX (Right) from Fountain, OCWD, and Zeng Breakthrough Data.

Table 4.20. Calculated Media Usage Rate (lb/1000 gal) Based on Regulation/Treatment Objective Using Modeled Breakthrough Curves

REG	Zeng 2020				Fountain		OCWD	
	GAC Pres GAC 400	IX PFA694E	GAC Pres 1240CAT	IX PFA694E	GAC Pres GAC400	IX 2301	GAC Pres GAC400	IX 2301
EPA HAL	0.030	0.025	N/A	N/A	0.11	0.026	N/A	N/A
IL GW	0.034	0.047	0.061	0.042	0.20	0.036	0.059	0.014
NH GW	0.045	0.069	0.12	0.050	0.24	0.042	0.056	0.023
WI GW	0.035	0.050	0.058	0.038	0.23	0.040	0.070	0.020
NJ GW	0.040	0.065	0.072	0.050	0.26	0.043	0.076	0.025
VT GW	0.055	0.11	0.13	0.062	0.27	0.044	0.091	0.021
Min	0.030	0.025	0.058	0.038	0.114	0.026	0.056	0.014
Max	0.055	0.112	0.126	0.062	0.272	0.044	0.091	0.025

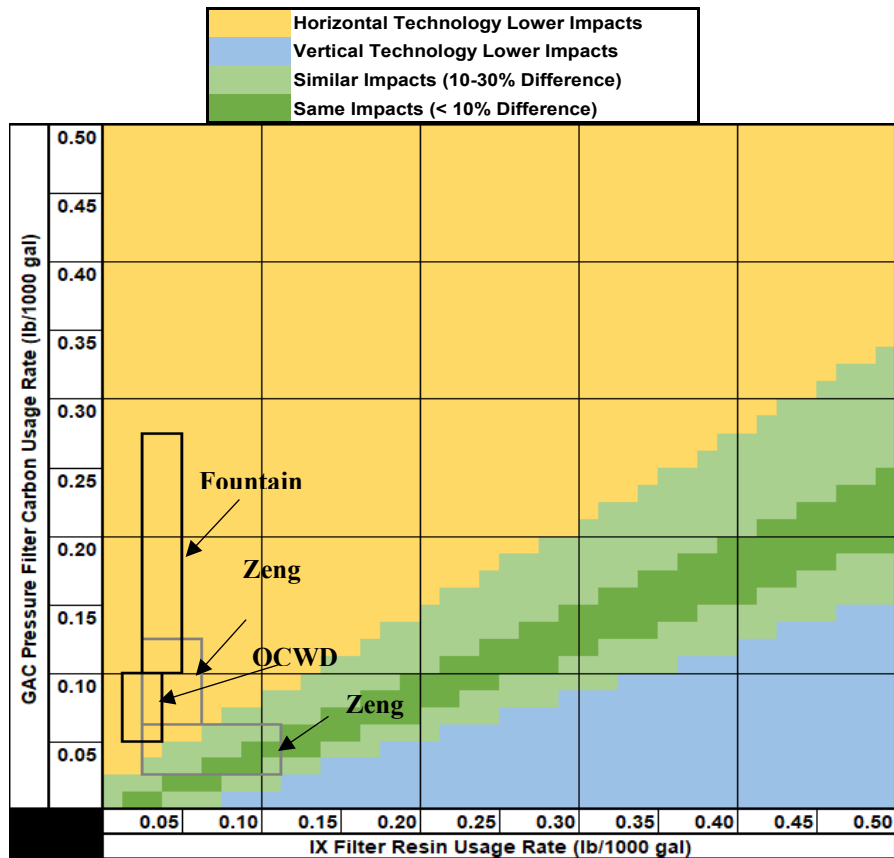


Figure 4.26. Comparison of Global Warming Impacts for GAC Pressure Filter and IX Filter with Prefilters at 0.31 mgd for Filter Media Disposal by Hazardous Waste Landfill.

Boxes show range of media usage rates, possible over a range of treatment objectives (Table 4.19), estimated using data from Fountain, CO, Orange County Water District, and Zeng et al. 2020.

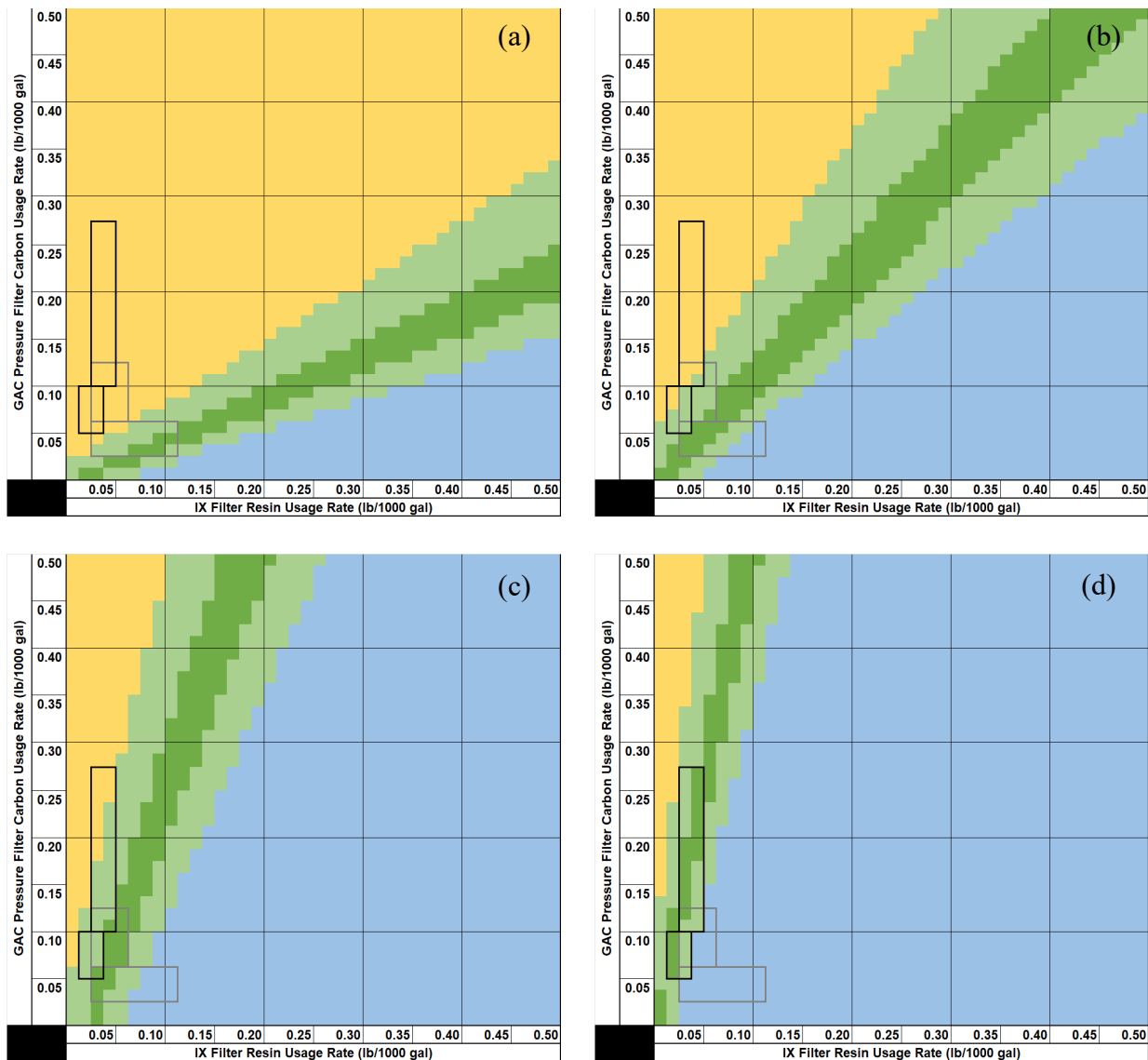


Figure 4.27. Comparison of Global Warming Impacts (Top) and Lifetime Cost (Bottom) for GAC Pressure Filter Versus IX Filter

with prefilter with media disposed by hazardous waste landfill (left) and thermal reactivation (GAC) and hazardous waste landfill (IX) (right) with range of calculated media usage rates from Fountain, CO; OCWD; and Zeng et al, 2020.

4.8 EVALUATION OF TREATABILITY DATA WITH DECISION SUPPORT TOOL

Data generated during the project was input to the decision support tool to investigate costs associated with different treatment scenarios, treatment technologies, and treatment goals. The following sections detail the impact of regulatory levels (treatment goals), organic matter (TOC), and treatment technology (GAC, IX, RO) on operation and maintenance and lifecycle costs.

4.8.1 Impact of PFAS Regulations and TOC on GAC Costs for the Treatment of PFAS-Impacted Groundwater

To investigate the impact of total organic carbon (TOC) presence on the performance of GAC (F400) for the treatment of PFAS-impacted groundwater (GW), three sources of GW with different TOC levels were investigated during the treatability portion of the study. The GW sources and their corresponding TOC levels are summarized in Table 4.21.

Table 4.21. Investigated GW with Different TOC Levels

GW source	TOC level (mg/L)
North Carolina (NC)	<0.3
Fountain, Colorado (CO)	1.3
Florida (FL)	4.6

RSSCT experiments were conducted with each GW source (see Section 3.2) and the results were used to estimate the associated costs of implementing these adsorbents to comply with 6 different drinking water regulation/health advisory level/ notification levels (Table 4.22) with a design flow rate of 0.74 MGD. For this comparison, affordability was chosen as the weighting scheme to sort treatment technologies according to the highest score. The weights assigned to different decision criteria in this scenario are summarized in Table 4.23. For analysis of GAC systems, both pressure and gravity driven systems were evaluated. However, thermal reactivation as a disposal method was not included but is discussed in the technology comparison selection below.

To rule out the effect of initial concentration, all RSSCT influents for different GW sources were spiked with 0.2 nM of each investigated PFAS (PFHpA, PFOA, PFNA, PFHxS, and PFOS). The breakthrough data of GAC were fit using the pore surface diffusion model (PSDM) in AdDesignS. The fitted model parameters were then used to generate breakthrough curves for the estimation of the number of bed volumes (BVs) at which each regulation criteria were reached (GAC data presented in Table 4.24). These BVs were then used for the calculation of GAC use rates as input to the decision support tool (Table 4.24).

Table 4.22. Investigated Drinking Water Regulation/Health Advisory Level/ Notification Levels.

Federal/State	Year	Criteria (ng/L)
US EPA	2016	PFOA + PFOS = 70
New Jersey (NJ)	2020	PFOA = 14, PFOS = 13, PFNA = 13
Michigan	2020	PFOA = 8, PFOS = 16, PFNA = 6
Vermont	2020	PFHpA + PFOA + PFNA + PFHxS + PFOS = 20
California	2020	PFOA = 5
US EPA	2023	PFOA = 4, PFOS = 4

Table 4.23. Assigned Weighting Schemes for Cost Comparison of GAC and IX.

Weighting Scheme	Score
Global warming	0.1
Global Environmental and Human Health	0.1
Affordability	0.7
Facility operation	0.1

For GAC, the costs of gravity basin and pressure vessels were compared for each GW type and regulation scenario or treatment goal (Table 4.25 to 4.30). The lower overall score (marked with blue) represents the higher affordability of a specific treatment, and for this example, whether or not GAC gravity basins or pressure vessels are preferable based on scoring criteria (mostly weighted by cost). Except for the former EPA health advisory limit regulation, GAC gravity basins received a better score compared to GAC pressure vessels for low TOC GWs (TOC<0.3 mg/L) due to longevity of treatment. For CO-GW (mid-TOC levels), GAC pressure vessel had better scores than the GAC gravity basins. For FL-GW (with high-TOC levels) the overall score of GAC-pressure vessels could not be calculated with the tool as the bed changeout criteria was reached in less than a month, making the application of GAC treatment very inefficient.

Table 4.24. Media Use Rates for GAC (F400) Determined through RSSCTs.

	BVs to reach criterion	Use rate (lb/1000 gal)
EPA former health advisory limit		
NC	223,000	0.02
CO	22,600	0.19
FL	3,200	1.35
New Jersey State MCLs		
NC	120,000	0.036
CO	9,000	0.48
FL	1,080	3.99
Michigan State MCLs		
NC	96,500	0.044
CO	7,200	0.6
FL	820	5.26
Vermont State MCL		
NC	84,000	0.05
CO	6,100	0.71
FL	750	5.75
California Notification Level		
NC	80,200	0.05
CO	5,800	0.73
FL	650	6.53
EPA MCL (2023)		
NC	~77,000	0.057
CO	~5,800	0.79
FL	~580	7.45

Table 4.25. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GWs: 2016 USEPA HAL.

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
GAC gravity	NC	EPA	48.0	\$512,151	\$82,615	\$3,816,733	0.78
GAC pressure	NC	EPA	48.0	\$388,648	\$82,120	\$3,673,460	0.20
GAC gravity	CO	EPA	12.6	\$512,151	\$157,789	\$6,823,714	0.43
GAC pressure	CO	EPA	7.6	\$388,648	\$158,051	\$6,710,705	0.55
GAC gravity	FL	EPA	1.8	\$512,151	\$780,581	\$31,735,409	0.85
GAC pressure	FL	EPA	1.1	\$388,648	\$708,282	\$28,719,912	0.12

Table 4.26. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: New Jersey MCL.

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
GAC gravity	NC	NJ MCL	48	\$512,151	\$82,615	\$3,816,733	0.43
GAC pressure	NC	NJ MCL	39.9	\$388,648	\$85,004	\$3,788,793	0.55
GAC gravity	CO	NJ MCL	5.0	\$512,151	\$440,541	\$18,133,771	0.83
GAC pressure	CO	NJ MCL	3.0	\$388,648	\$359,136	\$14,754,073	0.14
GAC gravity	FL	NJ MCL	0.6	\$512,151	\$3,254,102	\$130,676,222	0.96
GAC pressure	FL	NJ MCL	0.4	\$388,648	\$2,488,595	\$99,932,447	N/A

Table 4.27. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: Michigan State MCL.

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
GAC gravity	NC	Michigan State MCL	48	\$512,151	\$95,844	\$4,345,926	0.78
GAC pressure	NC	Michigan State MCL	32.6	\$388,648	\$94,622	\$4,173,512	0.20
GAC gravity	CO	Michigan State MCL	4.0	\$512,151	\$536,731	\$21,981,376	0.85
GAC pressure	CO	Michigan State MCL	2.4	\$388,648	\$431,938	\$17,666,155	0.12
GAC gravity	FL	Michigan State MCL	0.5	\$512,151	\$4,272,114	\$171,396,710	0.98
GAC pressure	FL	Michigan State MCL	0.3	\$388,648	\$3,259,083	\$130,751,972	N/A

Table 4.28. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: Vermont MCL.

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
GAC gravity	NC	Vermont State MCL	47.80	\$512,151	\$95,859	\$4,346,519	0.43
GAC pressure	NC	Vermont State MCL	28.7	\$388,648	\$98,262	\$4,319,116	0.55
GAC gravity	CO	Vermont State MCL	3.4	\$512,151	\$624,905	\$25,508,348	0.86
GAC pressure	CO	Vermont State MCL	2.0	\$388,648	\$498,673	\$20,335,562	0.11
GAC gravity	FL	Vermont State MCL	0.4	\$512,151	\$4,664,890	\$187,107,764	0.97
GAC pressure	FL	Vermont State MCL	0.2	\$388,648	\$3,556,358	\$142,642,970	N/A

Table 4.29. Bed Lifetime (Months), Capital- O&M- and lifetime Costs of GAC for the Treatment of Three GW Sources: California Notification Level.

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
GAC gravity	NC	California Notification	48.0	\$512,151.23	\$95,859.21	\$4,346,519.47	0.43
GAC pressure	NC	California Notification	28.7	\$388,648.48	\$98,261.69	\$4,319,116.05	0.55
GAC gravity	CO	California Notification	3.3	\$512,151.23	\$640,936.60	\$26,149,615.34	0.96
GAC pressure	CO	California Notification	2.0	\$388,648.48	\$510,806.52	\$20,820,909.14	0.01
GAC gravity	FL	California Notification	0.4	\$512,151.23	\$5,290,126.16	\$212,117,197.82	0.98
GAC pressure	FL	California Notification	0.2	\$388,648.48	\$4,029,571.22	\$161,571,497.29	N/A

Table 4.30. Bed Lifetime (Months), Capital- O&M- and Lifetime Costs of GAC for the Treatment of Three GW Sources: EPA (2023) MCL.

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
GAC gravity	NC	EPA (2023)	42.1	\$512,151	\$101,470	\$4,570,963	0.43
GAC pressure	NC	EPA (2023)	25.2	\$388,648	\$102,508	\$4,488,987	0.55
GAC gravity	CO	EPA (2023)	3.0	\$512,151	\$689,032	\$28,073,418	0.96
GAC pressure	CO	EPA (2023)	1.8	\$388,648	\$547,208	\$22,276,950	0.01
GAC gravity	FL	EPA (2023)	0.3	\$512,151	\$6,027,584	\$241,615,504	0.10
GAC pressure	FL	EPA (2023)	0.2	\$388,648	\$4,587,720	\$183,897,453	N/A

Cost data generated for GAC under three TOC treatment scenarios (Table 4.21) and six regulatory limits (Table 4.22) were summarized to compare the impact of TOC levels and regulations on cost. Figure 4.28 summarizes the impact of TOC concentration on O&M and lifetime-costs of GAC for the treatment of PFAS-impacted GW. As can be inferred from Tables 4.25 to 4.30, increasing the TOC concentrations would substantially decrease the lifetime of GAC, leading to significantly higher costs for complying with various regulations.

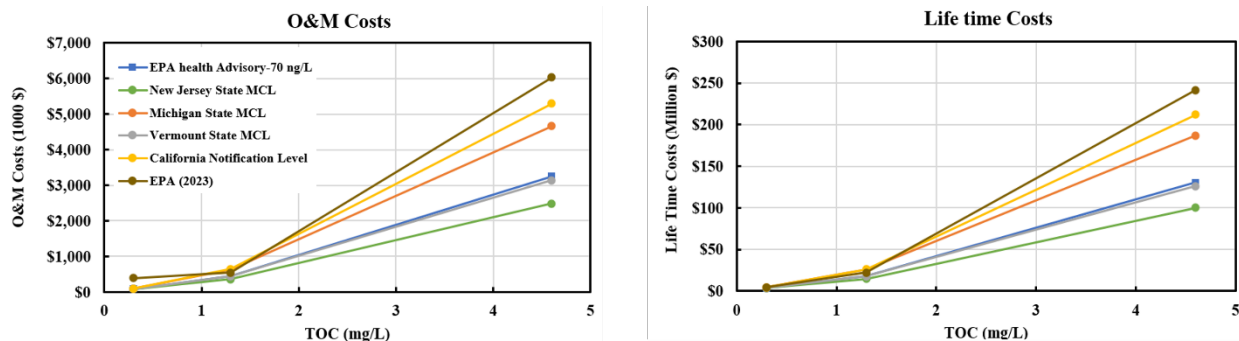


Figure 4.28. O&M- and Lifetime-costs Versus TOC for GAC

4.8.2 Evaluation of GAC Data from Pilot-scale Experiments and RSSCTs

The results from F400-RSSCT experiments during the treatment of CO-GW were also compared with the results from an F400-pilot study with GW from the same well (Liu et al., 2019), although sampled at different times. PFAS concentrations measured during the pilot-scale experiment in adsorption system influent are summarized in Table 4.31.

Table 4.31. Average PFAS Concentrations in Fountain GW Used for Pilot Test.

PFAS	Acronym	Average Concentration (ng/L)
Perfluoropentanoic acid	PFPeA	22.2 ± 3.3
Perfluorohexanoic acid	PFHxA	20.6 ± 1.9
Perfluoroheptanoic acid	PFHpA	7.2 ± 0.7
Perfluorooctanoic acid	PFOA	19.0 ± 3.2
Perfluoropropane sulfonate	PFPrS	6.1 ± 2.4
Perfluorobutane sulfonate	PFBS	25.8 ± 3.1
Perfluoropentane sulfonate	PFPeS	11.9 ± 2.8
Perfluorohexane sulfonate	PFHxS	42.8 ± 2.4
Perfluorooctane sulfonate	PFOS	31.5 ± 4.5

GAC usage rates for F400 determined through pilot-scale testing and RSSCTs are summarized in Table 4.32. GAC-RSSCTs have shown promising results with respect to scale-up of full sized media systems (pilot- and full-scale), and the difference between the F400 use rates of the RSSCT (Table 4.24) and pilot scale system (Table 4.31) are likely due to difference in initial concentrations between studies (pilot-scale influent concentrations listed in Table 4.31; RSSCT influent concentration was 0.2 nM for each PFAS) and the relatively low effluent concentrations required to meet the state regulations listed in Table 4.22. Because RSSCT influent concentrations were higher than non-amended CO-GW, usage rates are expected to be higher.

Table 4.32. GAC Use Rates of Pilot Versus RSSCT with CO-GW as Feed Water

(Liu et al. 2019)

	Colorado GW- TOC: 1.3 mg/L	
Regulation	Use rate-Pilot (lb/1000 gal)	Use rate-RSSCT (lb/1000 gal)
EPA	0.10	0.19
NJ	0.23	0.48
Michigan	0.32	0.6
Vermont	0.24	0.71
California	0.38	0.73

GAC usage rates for the other three GAC products used in the pilot-scale study (Liu et al, 2019) were calculated to compare costs associated with variations in removal performance between GAC products. Due to the low concentrations of PFAS in CO-GW feeding the pilot system, 10% breakthrough of PFOA (~2 ng/l) was selected for the determination of different commercially

available GAC product use rates (Table 4.33). GAC usage rates were input to the decision support tool to calculate costs for treatment of 0.74 MGD of CO-GW (Table 4.34). Over a 30-year project lifetime, use of F400 over a coconut-based carbon tested (GCN1240) would yield approximately \$3.5M savings. Both mesoporous bituminous coal-based GAC products (F400, N400) tested yielded similar lifetime costs, with the microporous bituminous coal-based GAC product (F600) being approximately \$1.7M more costly.

Table 4.33. Evaluated GAC Product Use Rates When Treating CO-GW

GAC-Product	BVs to reach 10% PFOA breakthrough	Use rate (lb/1000 gal)
F400	22,950	0.196
GAC400	17,568	0.241
F600	13,125	0.313
GCN1240	12,820	0.341

Table 4.34. Evaluated GAC Product Costs When Treating CO-GW

GAC Product	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)
F400	7.3	\$388,648	\$186,837	\$7,862,148
GAC400	6.0	\$388,648	\$214,138	\$8,954,179
F600	4.6	\$388,648	\$257,819	\$10,701,427
GCN1240	4.2	\$388,648	\$274,807	\$11,380,913

4.8.3 Impact of PFAS Regulations and TOC on IX Costs for the Treatment of PFAS-Impacted Groundwater

To investigate the impact of influent TOC concentration on the performance of IX (CalRes 2301) for the treatment of PFAS-impacted groundwater (GW), three sources of GW with different TOC levels were investigated during the treatability portion of the study employing RSSCTs. The GW sources and their corresponding TOC levels are summarized in Table 4.21. RSSCT experiments were conducted with each GW source (see Section 3.3), and the results were used to estimate the associated costs of implementing these adsorbents to comply with 6 different drinking water regulation/health advisory level/ notification levels (Table 4.22) with a design flow rate of 0.74 MGD. No significant PFAA (besides PFHxA) breakthrough was observed during the duration of IX RSSCTs during the treatment of NC-GW (TOC level of <0.3), and therefore, IX-use rates are only calculated for CO- and FL-GWs. Summarized IX usage rates for the two GWs evaluated and regulatory derived treatment goals are presented in Table 4.35. Significantly higher usage rates when treating FL GW is the result of quicker breakthrough of PFAAs associated with higher TOC GW. Tables 4.36 and 4.37 summarize calculated costs for IX (CalRes 2301) treatment of CO-GW and FL-GW, respectively. In general, costs increased with more stringent regulations and higher influent TOC (Figure 4.29), although impacts from TOC were not increased as significantly as was observed for GAC (Figure 4.28). For example, the calculated lifetime cost for meeting New Jersey MCLs increased by approximately 100% when treating FL-GW compared to CO-GW, demonstrating the significant impact of TOC on selective IX resin.

Table 4.35.IX-use Rates to Meet Different Regulations During Treatment of CO and FL-GW

	BVs to reach criterion	Use rate (lb/1000 gal)
EPA former health advisory limit		
CO	-	-
FL	106,695	0.026
New Jersey State MCLs		
CO	147,731	0.019
FL	29,775	0.094
Michigan State MCLs		
CO	96,197	0.029
FL	17,369	0.161
Vermont State MCL		
CO	73,293	0.038
FL	19,354	0.144
California Notification Level		
CO	67,567	0.041
FL	23,324	0.12
US EPA MCL (2023) for PFOS/PFOA		
CO	57,260	0.049
FL	20,843	0.134
10% PFHxA breakthrough		
NC	226,178	0.012
CO	28,630	0.098
FL	17	0.171
EPA MCL (2023)		
NC	-	-
CO	~58,000	0.049
FL	~20,500	0.134

Table 4.36. IX Costs for Treatment of CO-GW

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)
AER filter	CO	EPA	-	-	-	-
AER filter	CO	NJ	19.2	\$211,787	\$91,561	\$3,874,231
AER filter	CO	Michigan	12.6	\$211,787	\$103,970	\$4,370,602
AER filter	CO	Vermont	9.3	\$211,787	\$121,964	\$5,090,345
AER filter	CO	California	8.9	\$211,787	\$121,057	\$5,054,056
AER filter	CO	EPA MCL	7.4	\$211,787	\$132,448	\$5,509,692
AER filter	CO	10% PFHxA	3.7	\$211,787	\$202,217	\$8,300,462

Table 4.37. IX Costs for Treatment of FL-GW

	GW	Regulation	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)
AER filter	FL	EPA	13.5	\$211,787	\$103,722	\$4,360,676
AER filter	FL	NJ	3.9	\$211,787	\$196,521	\$8,072,644
AER filter	FL	Michigan	2.3	\$211,787	\$291,920	\$11,888,595
AER filter	FL	Vermont	2.6	\$211,787	\$277,019	\$11,292,529
AER filter	FL	California	3.0	\$211,787	\$245,096	\$10,015,609
AER filter	FL	EPA MCL	2.7	\$211,787	\$266,378	\$10,866,889
AER filter	FL	10% PFHxA	2.1	\$211,787	\$306,159	\$12,458,140

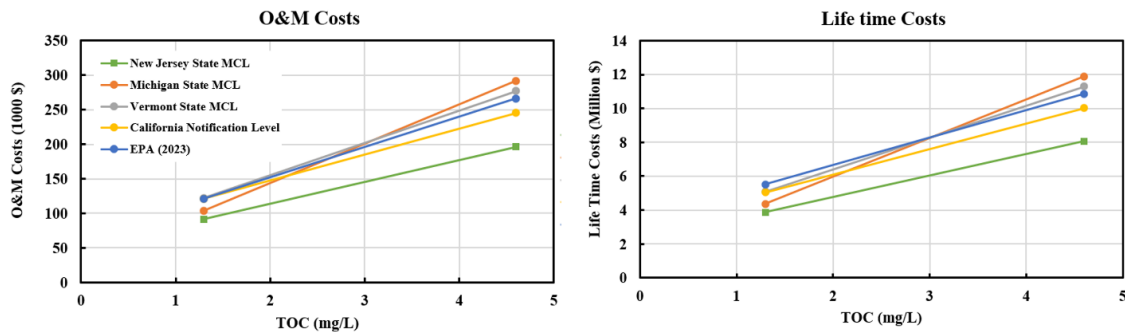


Figure 4.29. O&M- and Lifetime-costs Versus TOC for IX

4.8.4 Cost-comparison between Different Treatment Options (IX, GAC, and RO)

Costs of PFAA treatment between the technologies included in the decision support tool were evaluated for a similar treatment scenario as discussed above. Media usage rates for GAC and IX were obtained through RSSCT testing, and RO rejection values were obtained through membrane treatability testing. The cost comparison was conducted for the medium TOC groundwater (CO-GW, Table 4.21) and a flow rate of 0.74 MGD with capital, O&M, and lifetime (30-year) costs calculated for the different regulatory treatment goals summarized in Table 4.22 based on an influent PFAA concentration of 0.2 nM each. Summarized costs for various regulatory treatment goals are presented in Table 4.38. For RO data presented in Table 4.38, concentrate disposal was assumed to be to a publicly owned treatment plant (POTW; wastewater treatment plant). Overall scoring provided for each technology in Table 4.38 was based on the weights selected for comparison that are presented in Table 4.23 and is primarily based on costs of treatment. The GAC system being compared in Table 4.39 was selected as pressure driven without thermal regeneration as a disposal option (no discount given). Based on the comparison, for all regulatory driven treatment goals, IX would yield the lowest project cost followed by GAC and RO (Table 4.38).

For comparison purposes, the analysis was redone to include thermal reactivation for GAC disposal (Table 4.39). Although costs for GAC treatment are considerably lower than the previous analysis, IX would still be the favorable treatment process based on cost. Finally, the impact of RO concentrate disposal options on treatment costs was evaluated by comparing hazardous waste disposal to disposal through a POTW (Table 4.40). Results of the analysis demonstrate the significant challenge posed by concentrate management with the hazardous waste disposal option being economically infeasible.

Table 4.38. Cost-comparison of IX (AER), GAC, and RO Membrane for the Treatment of CO-GW without Thermal Reactivation of GAC

GW	Treatment	Regulation	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall score
CO	AER	NJ	\$211,787	\$91,561	\$3,874,231	0.012
CO	GAC	NJ	\$388,648	\$359,136	\$14,754,073	0.382
CO	RO	NJ	\$1,524,999	\$346,873	\$15,399,926	0.932
CO	AER	Michigan	\$211,787	\$103,970	\$4,370,602	0.012
CO	GAC	Michigan	\$388,648	\$431,938	\$17,666,155	0.460
CO	RO	Michigan	\$1,648,116	\$356,303	\$15,900,216	0.729
CO	AER	Vermont	\$211,787	\$121,964	\$5,090,345	0.011
CO	GAC	Vermont	\$388,648	\$498,673	\$20,335,562	0.585
CO	RO	Vermont	\$1,524,999	\$346,873	\$15,399,926	0.672
CO	AER	California	\$211,787	\$121,057	\$5,054,056	0.011
CO	GAC	California	\$388,648	\$510,807	\$20,820,909	0.493
CO	RO	California	\$1,524,999	\$346,873	\$15,399,926	0.609
CO	AER	EPA (2023)	\$211,787	\$132,448	\$5,509,692	0.011
CO	GAC	EPA (2023)	\$388,648	\$547,208	\$22,276,950	0.585
CO	RO	EPA (2023)	\$1,648,116	\$355,066	\$15,850,755	0.702

Table 4.39. Cost-comparison of IX (AER), GAC, and RO Membrane for the Treatment of CO-GW with Thermal Reactivation of GAC

GW	Treatment	Regulation	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall score
CO	AER	NJ	\$211,787	\$91,561	\$3,874,231	0.012
CO	GAC	NJ	\$388,648	\$199,778	\$8,379,779	0.382
CO	RO	NJ	\$1,524,999	\$346,873	\$15,399,926	0.932
CO	AER	Michigan	\$211,787	\$103,970	\$4,370,602	0.012
CO	GAC	Michigan	\$388,648	\$262,957	\$10,906,918	0.460
CO	RO	Michigan	\$1,648,116	\$356,303	\$15,900,216	0.903
CO	AER	Vermont	\$211,787	\$121,964	\$5,090,345	0.011
CO	GAC	Vermont	\$388,648	\$404,706	\$16,576,901	0.460
CO	RO	Vermont	\$1,524,999	\$346,873	\$15,399,926	0.879
CO	AER	California	\$211,787	\$121,057	\$5,054,056	0.011
CO	GAC	California	\$388,648	\$268,451	\$11,126,670	0.466
CO	RO	California	\$1,524,999	\$346,873	\$15,399,926	0.789
CO	AER	EPA (2023)	\$211,787	\$132,448	\$5,509,692	0.011
CO	GAC	EPA (2023)	\$388,648	\$284,932	\$11,785,923	0.475
CO	RO	EPA (2023)	\$1,648,116	\$355,066	\$15,850,755	0.702

Table 4.40. Comparison of the Costs for an RO Membrane System with Two Different Concentrate Disposal Option

(Hazardous waste disposal versus POTW disposal)

GW	Regulation	Waste disposal	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)
CO	EPA (2016)	POTW	\$1,524,999	\$346,873	\$15,399,926
CO	EPA (2016)	Hazardous waste disposal	\$1,524,999	\$101,530,802	\$4,062,757,094

5.0 CONCLUSIONS, IMPLEMENTATION ISSUES AND DATA GAPS

The principal objective of this project was to assess the benefits and disbenefits of the best available PFAS treatment approaches, including adsorption [i.e., granular activated carbon (GAC), ion-exchange] and high-pressure membranes [i.e., nanofiltration (NF), reverse osmosis (RO)]. This objective was achieved through systematic investigation of their efficacy for PFAS removal under various treatment scenarios and development of a life cycle impact and costing model to quantify and compare the technologies for different treatment scenarios. Factors impacting performance and life cycle impacts and costs were evaluated for each technology, including process variables, water quality, and regulatory limits. A key deliverable from this project was the development of a life cycle impact and costing tool allowing quantitative comparison of evaluated PFAS treatment technologies.

In general, and as one of the main drivers for the inception of this project, the investigated treatment technologies (i.e., adsorbents and membranes) can be implemented to provide effective separation of PFAS from impacted source waters. Therefore, a main component of this project was to generate treatability data, allowing for quantitative comparison of investigated technologies based on costing and life cycle impact metrics. To this end, an experimental matrix was developed and implemented to evaluate the factors impacting treatment performance. These factors included source water quality, process design and operating parameters, PFAS type and characteristics, and technology products (e.g., different IX resins, GAC products, and membrane types). Adsorbent treatability data were mainly generated through lab-scale experiments using rapid small-scale column tests (RSSCTs) with additional treatability data obtained from project team member pilot-scale studies and the literature. Membrane treatability data were generated through lab-scale experimentation using pilot-scale systems.

Scale-up approaches for GAC and IX RSSCTs developed in a parallel project were used to predict full-scale media use rates. PFAS type strongly affected both GAC and IX use rates, with short-chain perfluoroalkyl carboxylic acids (PFCAs) requiring the highest media use rates. For GAC, the total organic carbon (TOC) concentration had the most dramatic effect on media use rate. In groundwater with a TOC concentration of 1.3 mg/L, the GAC use rate was 10 times that in groundwater with a TOC concentration of <0.3 mg/L and 10 times less than for groundwater with a TOC concentration of 4.3 mg/L. TOC concentration was also observed to impact IX performance negatively, but not as strongly as GAC performance. Out of four major inorganic anions investigated (chloride, bicarbonate, sulfate, and nitrate), nitrate had the most pronounced impact on GAC and IX, negatively impacting the adsorption of short-chain PFAS by both sorbents, as well as the uptake of long-chain PFAS by IX. Doubling EBCT (from 10 minutes to 20 minutes) decreased GAC use rates by a factor of about two for the tested GAC and water matrix combination but had a negligible impact (from 1.5 minutes to 3 minutes) on IX use rates. Different regulatory scenarios also strongly impacted GAC and IX use rates; lower concentration targets and inclusion of shorter-chain PFAS into treatment criteria led to substantial increases in media use rates. GAC and IX types also significantly impacted media use rates, indicating that media selection is an important consideration for PFAS treatment projects.

Five membranes were evaluated for the removal of PFAS from several source waters of differing quality. They represented a spectrum of membrane classifications spanning loose NF, tight NF, low-pressure RO, and seawater RO. In addition to membrane classification, the impact of source water quality and operating conditions, including permeate flux and product water recovery, on the rejection of PFAAs was evaluated. PFAS type and characteristic had minimal impact on the separation performance of tight-NF and RO membranes evaluated, and these membranes can achieve >98% PFAS separation regardless of feed water quality and operating conditions. However, increased PFAS permeate concentrations were generally observed at high product water recovery for all membranes evaluated. Membrane type was found to impact treatment effectiveness with a low-pressure NF exhibiting significantly lower PFAS separation than expected. Follow-up experiments indicated that this membrane's PFAS treatment performance is negatively impacted by ionic composition (specifically increased multivalent anions, i.e., sulfate) and strength and system operating conditions (i.e., product water recovery).

An Excel® based decision support tool was developed as part of the project scope. The tool provides a pathway to comprehensively evaluate PFAS treatment technology options consisting of GAC, IX, and high-pressure membrane systems for groundwater remediation. The tool quantifies each option's relative environmental impacts, costs, and facility requirements for a given scenario based on the assigned scoring. The tool's functionality was demonstrated using scenario analysis, case studies, and treatability data generated during the project. Through scenario analyses comparing GAC, IX, and high-pressure membrane processes, it was determined that media use rates of GAC and IX strongly impact which technology is more cost-effective and environmentally sustainable (e.g., global warming impacts). Lower media use rates were often associated with IX compared to GAC, making IX a more environmentally sustainable PFAS treatment technology based on global warming impacts. However, GAC typically has lower project lifecycle costs than IX when media use rates are similar between the two processes. RO and NF have significantly higher project lifecycle costs and environmental impacts compared to GAC and IX; GAC and IX media disposal options evaluated do not significantly favor one process over another. Thermal reactivation of GAC results in GAC becoming more favorable compared to IX for certain scenarios. For GAC and IX, treatment goals based on federal and state regulations were found to significantly impact project lifecycle costs, with a critical component being meeting PFOA regulations. Due to the negative impact of TOC on GAC and IX media use rates, treatment of regulated PFAAs in source waters containing elevated TOC is challenging from a cost perspective. For GAC and IX treatment, the highest treatment costs were associated with treating high TOC groundwater to the proposed USEPA MCL for PFOA. High-pressure membrane treatment was found to be significantly more expensive than GAC and IX due to higher capital and O&M costs, with concentrate management costs being substantial depending on the disposal option. These findings have been summarized in Table 5.1.

Table 5.1. Advantages and Disadvantages of PFAS Treatment Technologies Evaluated in this Project Based on Findings, Life Cycle Impact, and Cost Modeling

Technology	Advantages	Disadvantages
IX Contactor	• Low treatment footprint • Lower media usage rate than GAC, generally.	• High cost of resin
GAC Pressure Contactor	• Lower treatment footprint than RO/NF • Thermal reactivation possible	• Residuals discharge • Significantly impacted by water quality • Higher media use rate than IX, generally • Higher environmental impacts than IX (except ozone depletion) with spent GAC disposal by landfill or incineration
RO/NF Systems	• Lower environmental impacts when GAC and IX media use rate is high • Water quality does not significantly impact performance • High PFAS removal regardless of water quality	• Largest areal treatment footprint • Generally highest treatment cost • Concentrate stream

Numerous data gaps and PFAS treatment technology needs were identified throughout this project. At the project's onset, the main objective was to quantify the benefits and disbenefits of numerous purportedly viable PFAS treatment processes (see Figure 1.1) to provide guidance on technology selection. However, low technology readiness levels coupled with unwillingness of intellectual property holders and vendors to provide inventories of technology components made this task insurmountable. Because several of these technologies, for example, PFAS destructive technologies such as plasma, electrochemical treatment, and super-critical water oxidation, have reached technology readiness levels enabling their application at larger scales, future studies should examine their PFAS treatment effectiveness and lifecycle impacts and costs to allow for technology comparison and selection based on quantitative data.

Rapid small-scale column tests (RSSCTs) were used in this project to quantify the performance of various adsorbents and evaluate parameters impacting PFAS treatment performance. RSSCTs are being applied more frequently across the PFAS research and development sectors to predict pilot- and full-scale adsorbent performance, down-select from a list of potentially viable adsorbent products, assess the impact of water quality on PFAS adsorption, and evaluate pretreatment options. While research conducted in this project and team member's parallel research indicates that RSSCTs are a useful tool for screening potential adsorbents, more research is needed to examine the use of RSSCTs to predict adsorbent performance at larger scales (up-scaling) and assess the performance of novel PFAS adsorbents. This research need requires generation of more pilot- or full-scale adsorbent breakthrough datasets coupled with RSSCT experiments performed on the same source water. Additionally, there is a need to perform RSSCTs on large-scale adsorbent feed water over longer time periods to assess the impact of water quality changes on RSSCT-derived scale-up predictions.

Findings from this project demonstrate that source water quality (e.g., TOC, nitrate) can have a detrimental impact on adsorbent performance and that the rapid breakthrough observed for certain PFAS (e.g., short-chain PFCAs) through adsorbent systems is a significant challenge. There is a

clear need to develop more selective adsorbents for PFAS with higher adsorptive capacity to extend the lifetime of adsorption-based processes. Additionally, the observed deleterious impact of dissolved organic matter on PFAS adsorption suggests that pretreatment processes, for example, coagulation/flocculation, oxidation, and biological processes, could improve PFAS removal by GAC and IX. Research is needed to examine the tradeoffs between potential PFAS removal capacity improvements resulting from pretreatment and costs of implementing pretreatment.

High-pressure membranes were found to be an effective approach for the removal of a wide variety of PFAS without exhibiting breakthrough behavior associated with adsorbents such as GAC and IX. However, lifecycle modeling indicates that RO and NF are generally more cost and energy-intensive than adsorbent-based processes, with concentrate management being a significant barrier to implementation. Emerging membrane technologies have been developed to minimize concentrate production [i.e., high-recovery systems, e.g., closed-circuit desalination (CCD)] and may provide the means to significantly concentrate aqueous PFAS residuals. Implementation of high-recovery NF or RO may be advantageous for concentrating PFAS-impacted source waters and residuals prior to PFAS destructive technologies. This approach requires additional evaluation in terms of costs and energy requirements in comparison to other concentration technologies (e.g., foam fractionation) prior to PFAS destruction.

Finally, this project mainly focused on the removal of PFAS with regulatory relevance, specifically PFAAs. Given the large number of PFAS, estimated in the thousands, there is a significant need to evaluate established treatment processes for additional PFAS of interest. Currently, there is a dearth of treatability data on other PFAS classes, including sulfonamides, zwitterionic and cationic PFAS, fluorotelomer alcohols, ultra-short-chain PFAS, and perfluoroalkyl ether acids among other emerging PFAS classes.

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APPENDIX A BREAKTHROUGH CURVES FOR PFAS

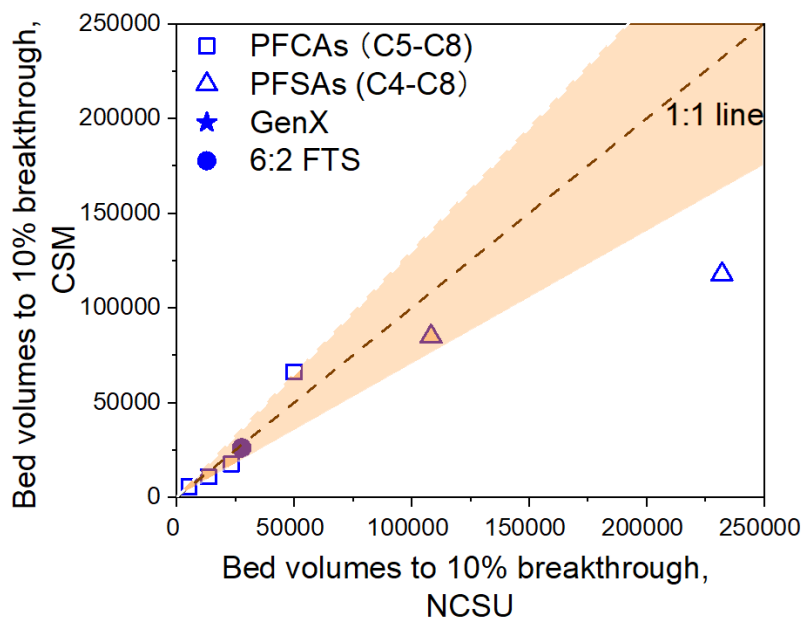


Figure A1. Comparison of RSSCT Data (bed volume to 10% Breakthrough) Generated at NCSU and CSM. IX: CalRes2301.

Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm). Shaded envelopes show 30% variance around the 1:1 line.

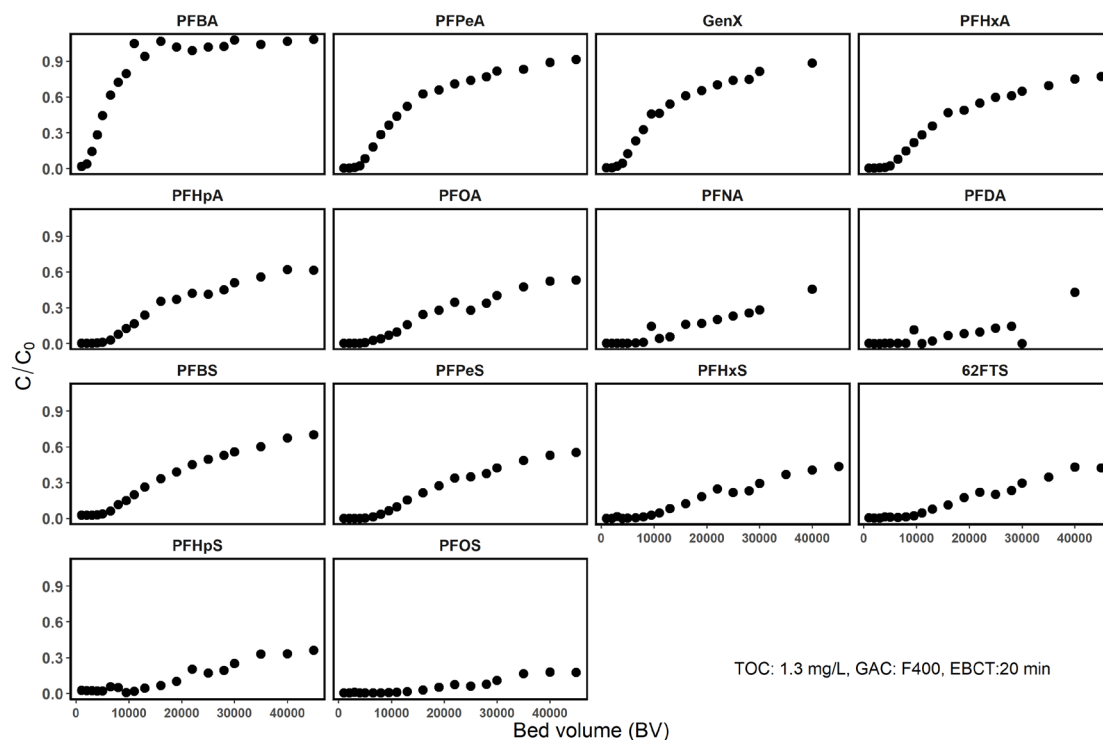
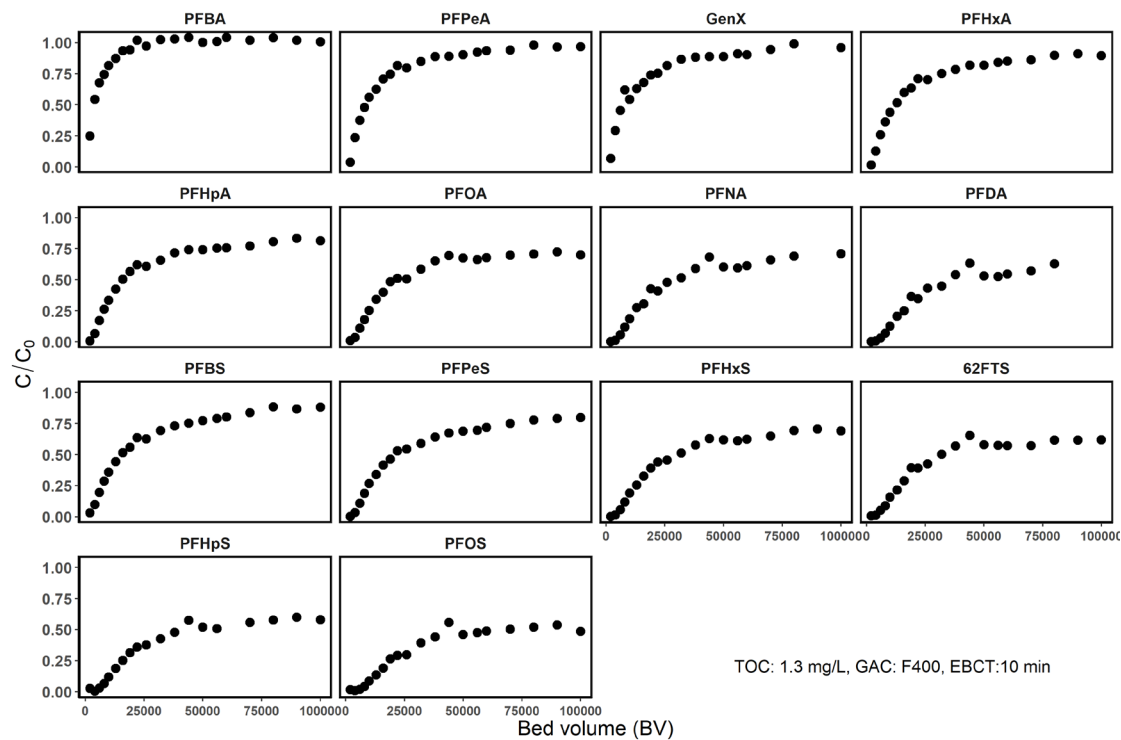


Figure A2. Effect of Empty Bed Contact Time (EBCT) on PFAS Removal.

GAC: GAC1. Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm).

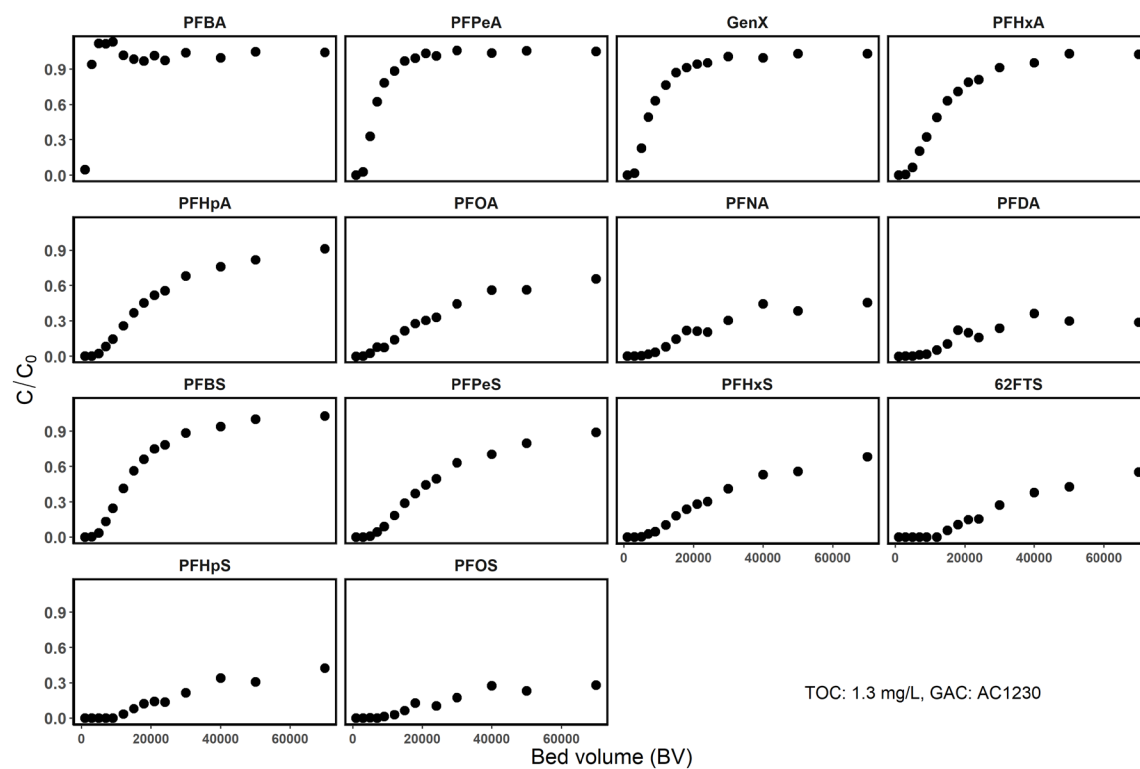
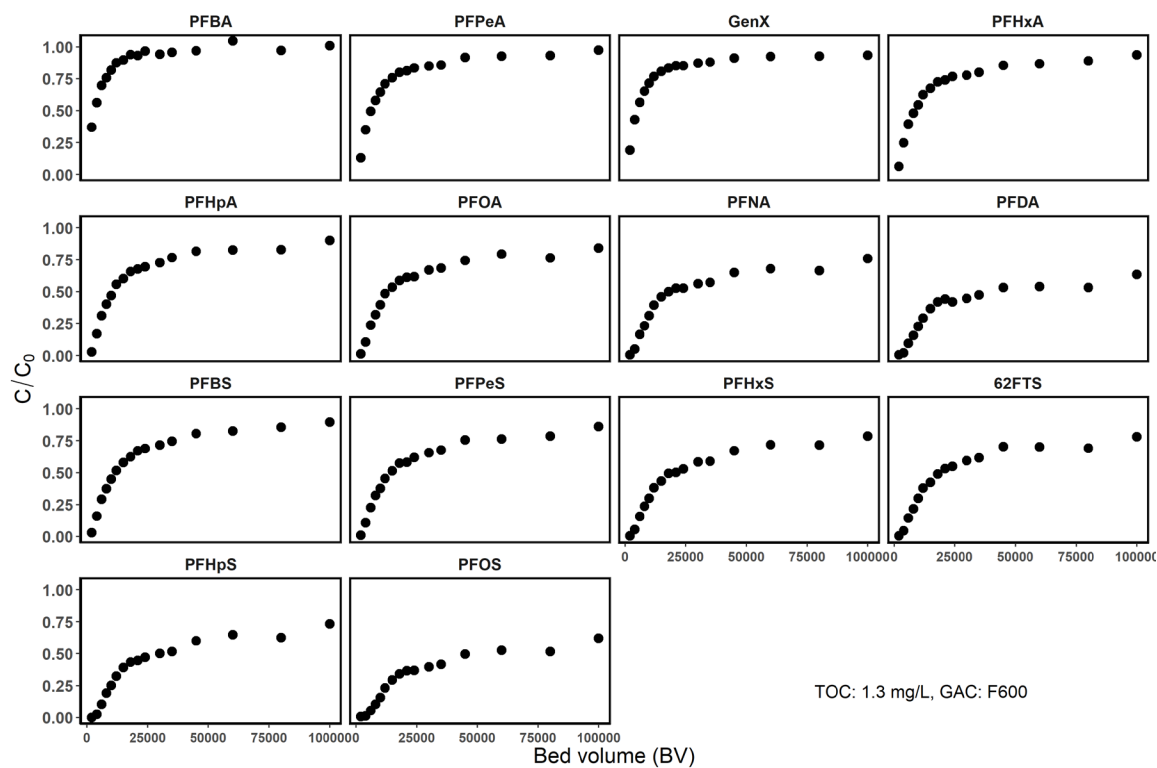
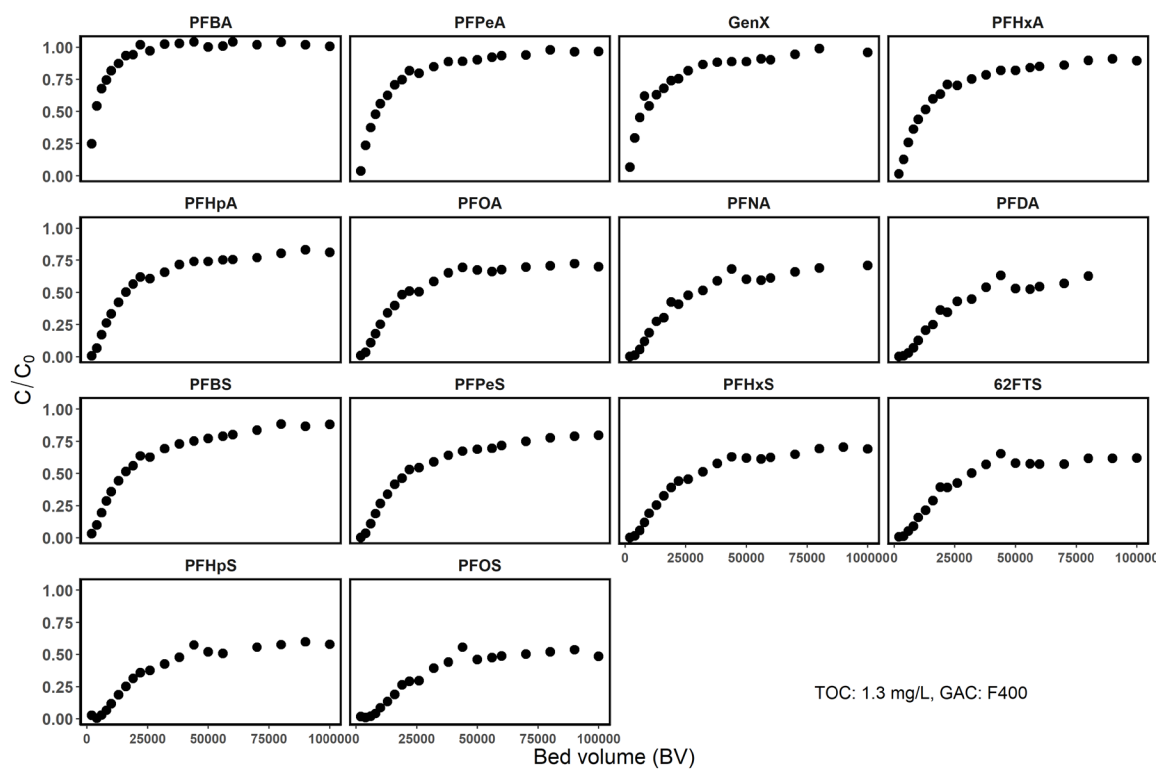


Figure A3. Effect of GAC Selection on PFAS Removal.

Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 $\mu\text{S}/\text{cm}$).



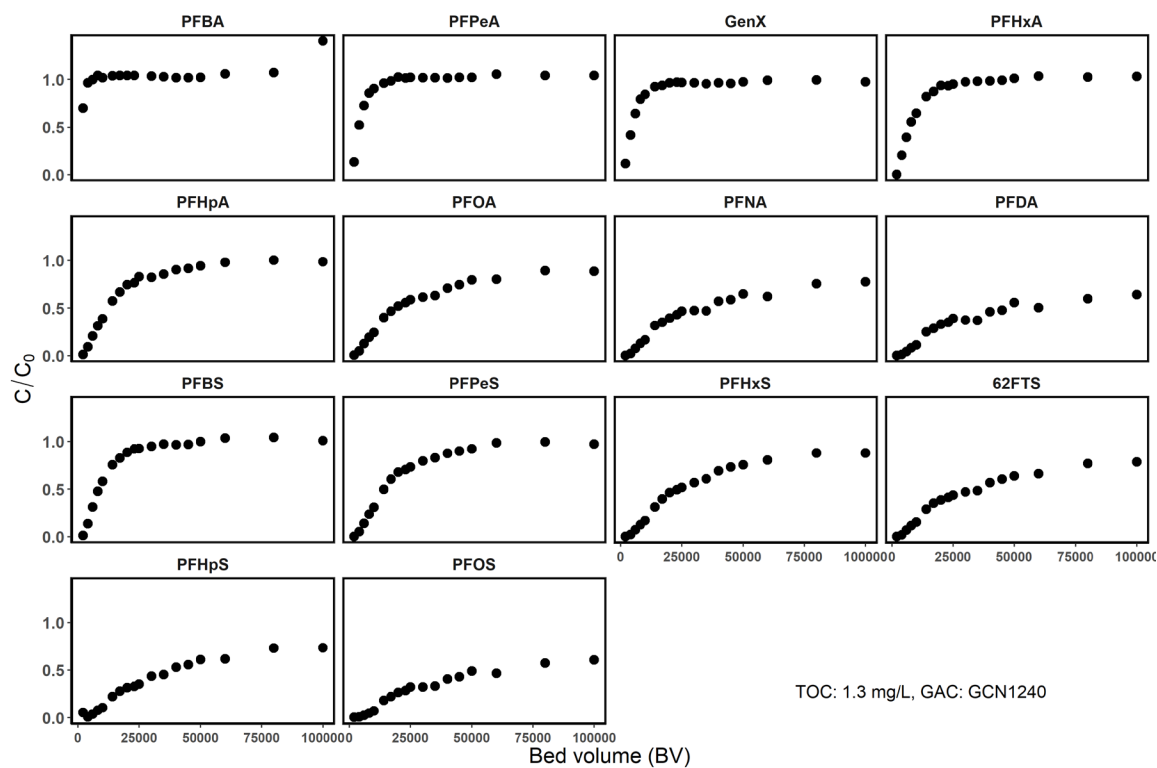
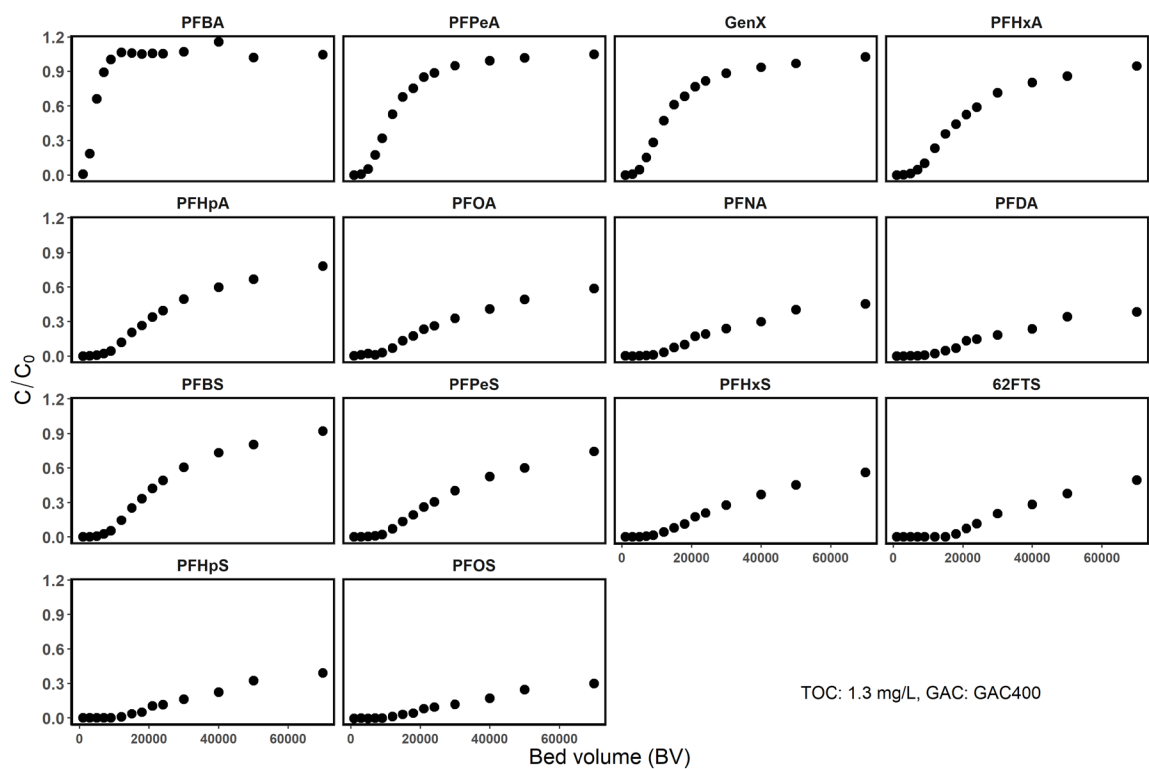
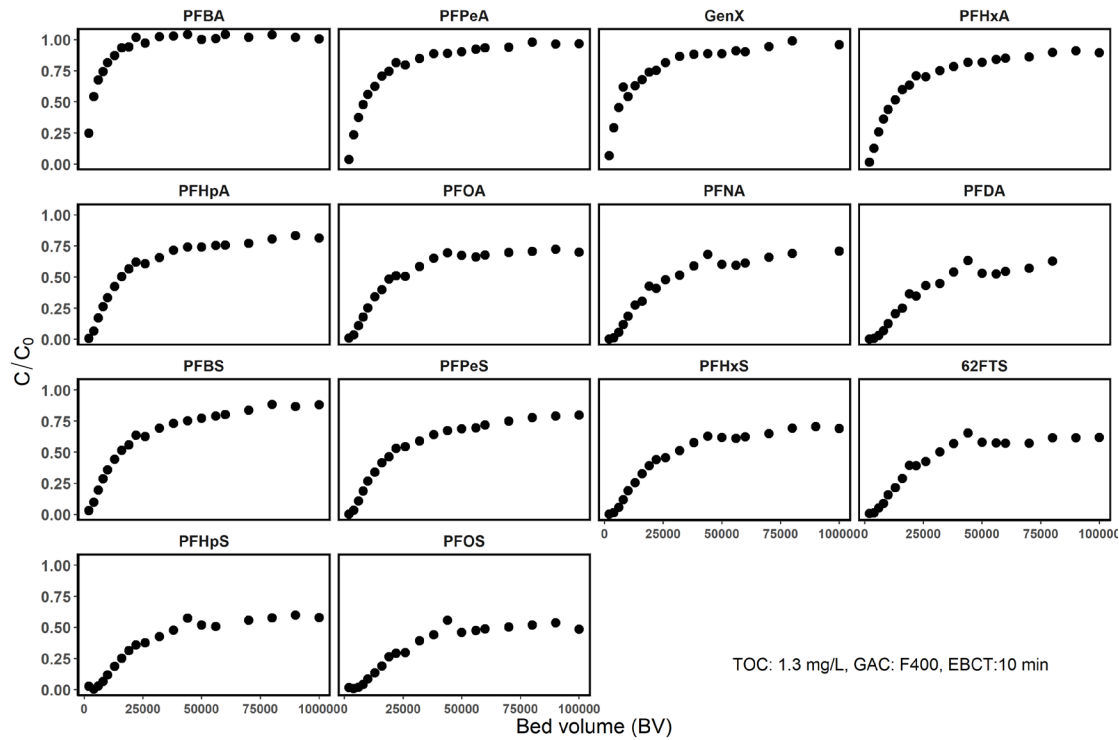
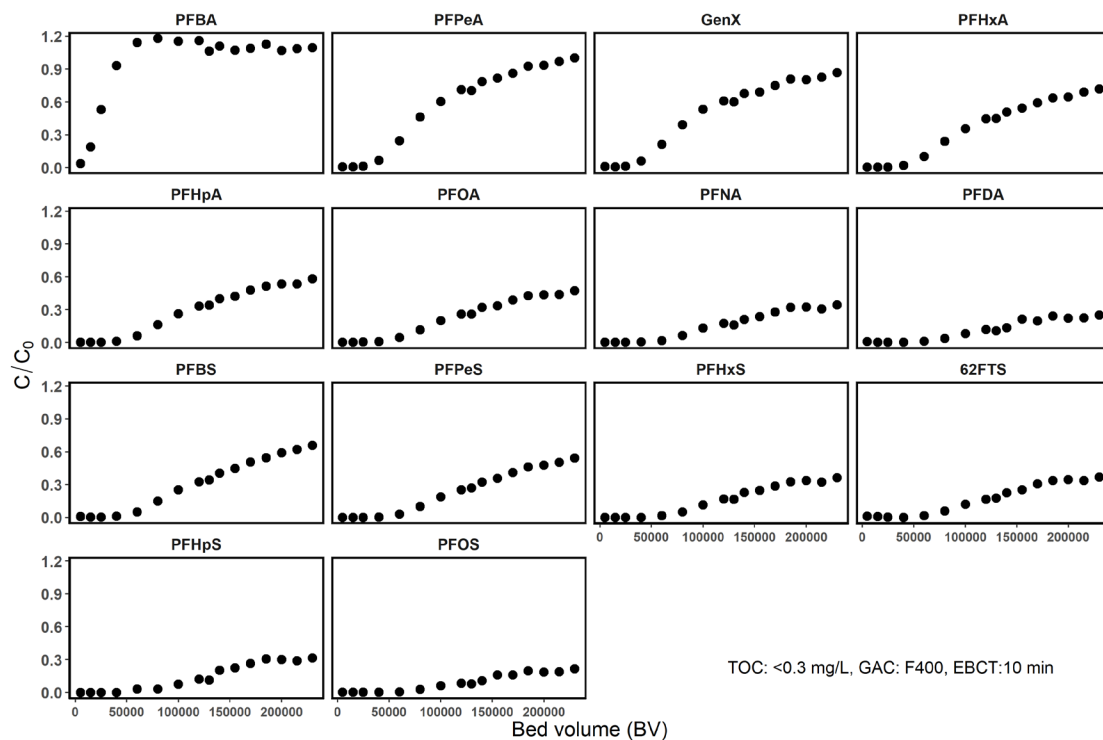


Figure A4. Effect of GAC Selection on PFAS Removal.

Water: Fountain, CO, groundwater (TOC=1.3 mg/L, specific conductivity = 918 μ S/cm).



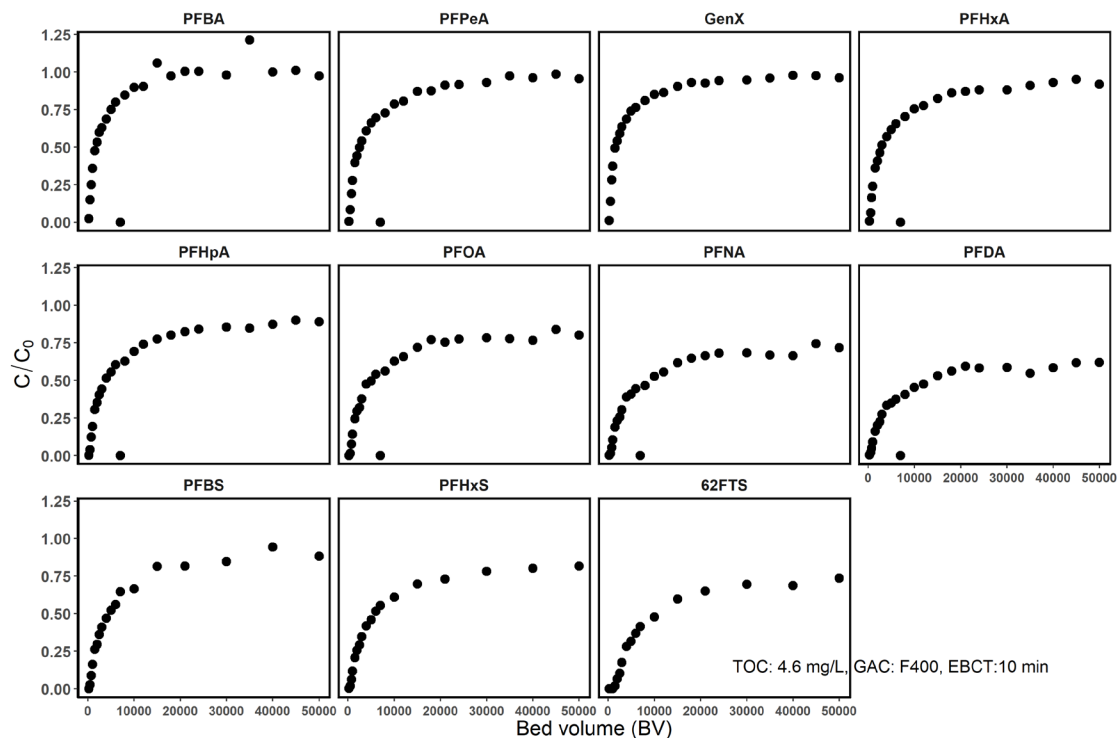
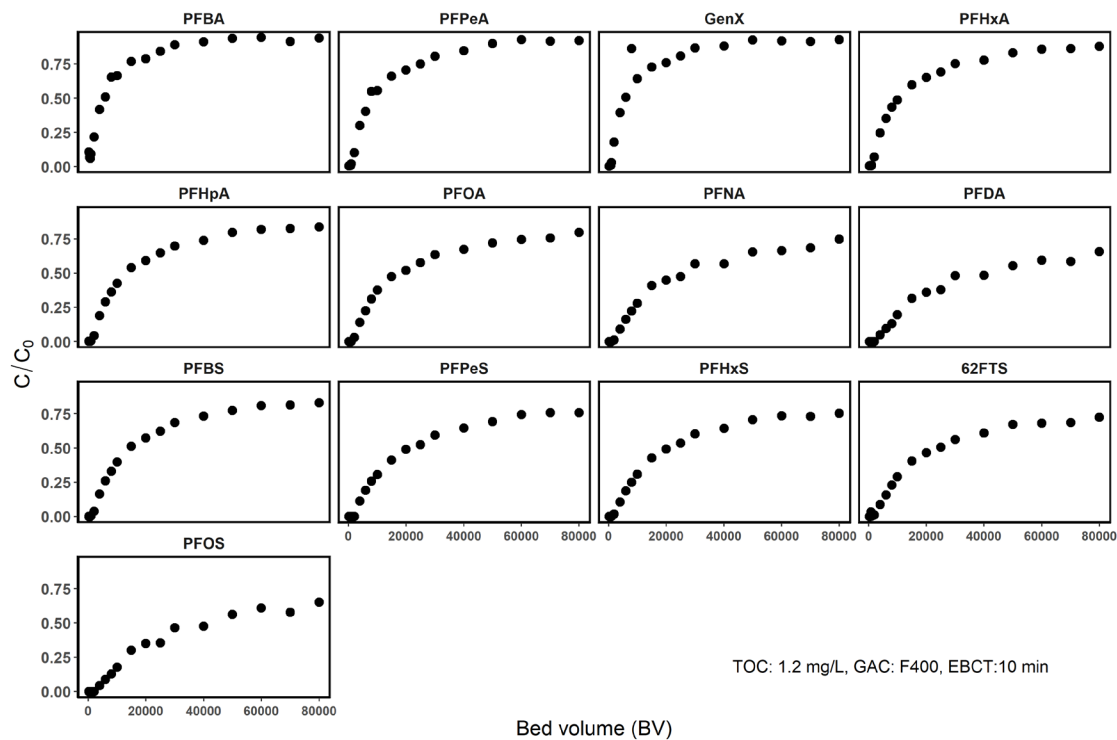
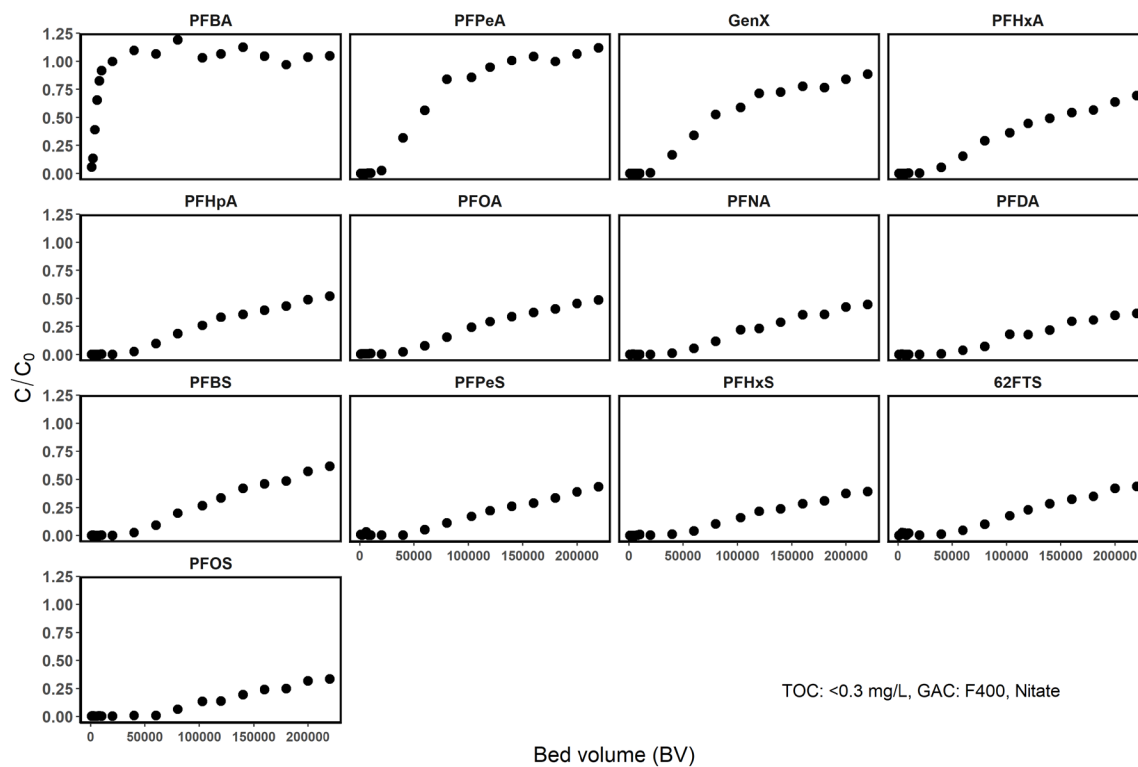
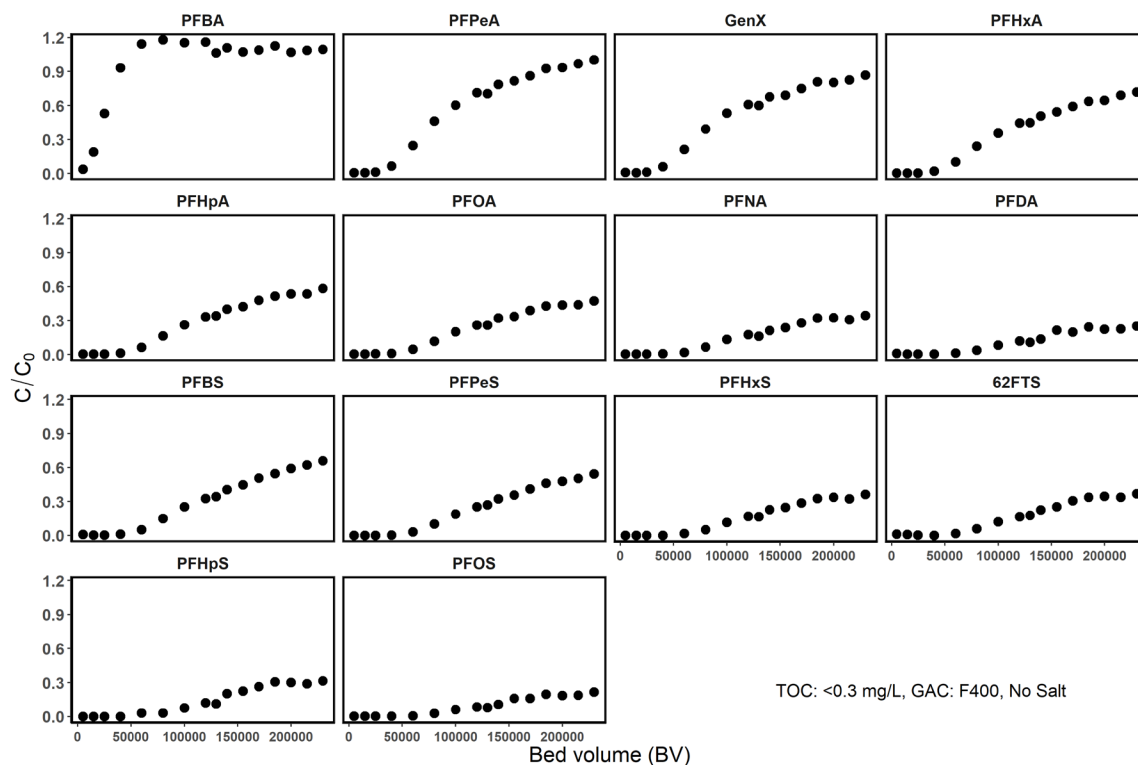
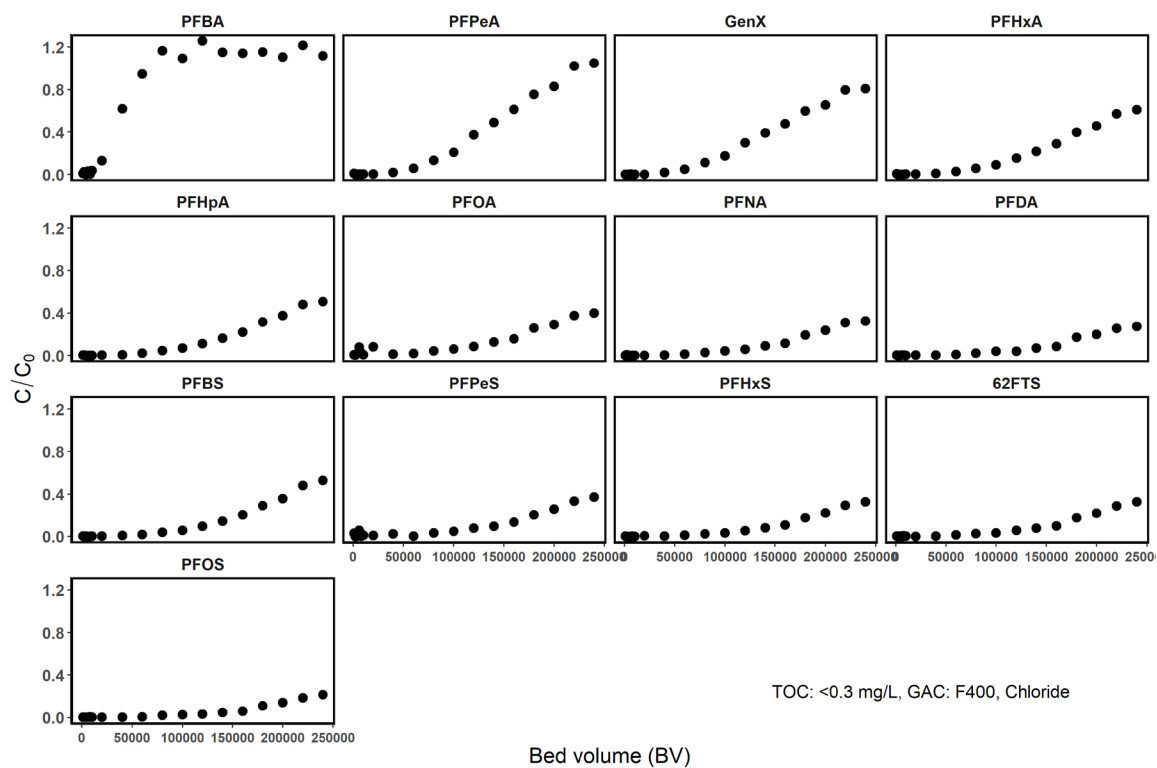
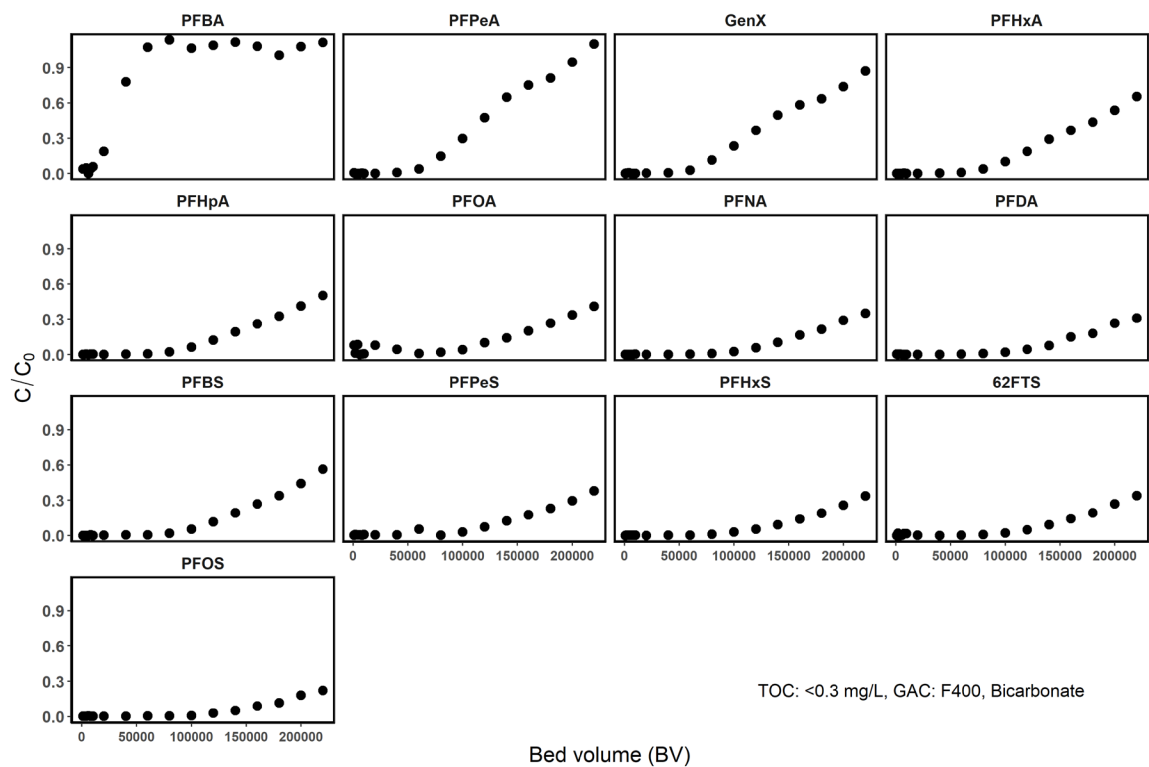


Figure A5. Effect of TOC on PFAS Removal.

GAC: GAC1.





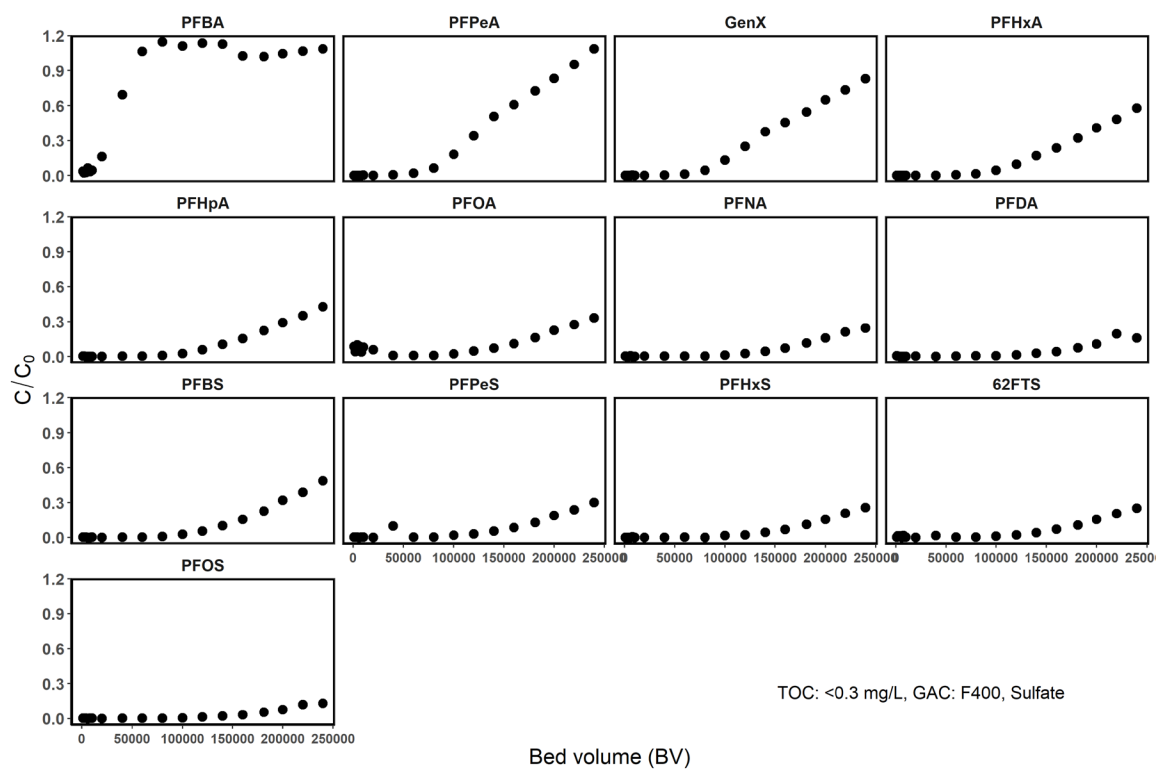


Figure A6. Effect of Inorganic Anions on PFAS Removal.

GAC: GAC1. Water: Wake County, NC, groundwater (TOC <0.3 mg C/L, specific conductivity = 165 $\mu\text{S}/\text{cm}$).

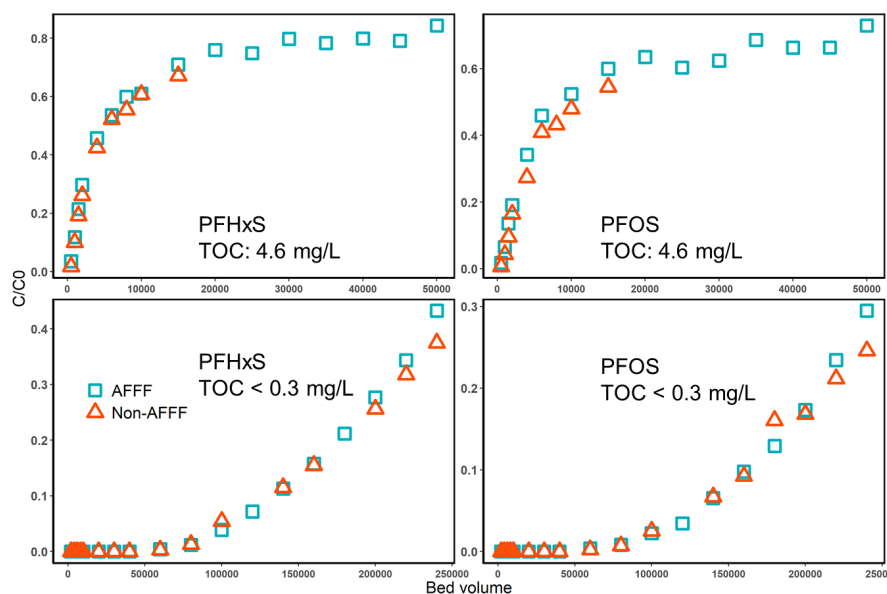


Figure A7. Effect of Aqueous Film-forming Foam (AFFF) on PFAS Removal.

GAC: GAC1. Water: Wake County, NC, groundwater (TOC < 0.3 mg/L, specific conductivity = 185 μ S/cm); Land O Lakes, FL, groundwater (TOC = 4.6 mg/L).

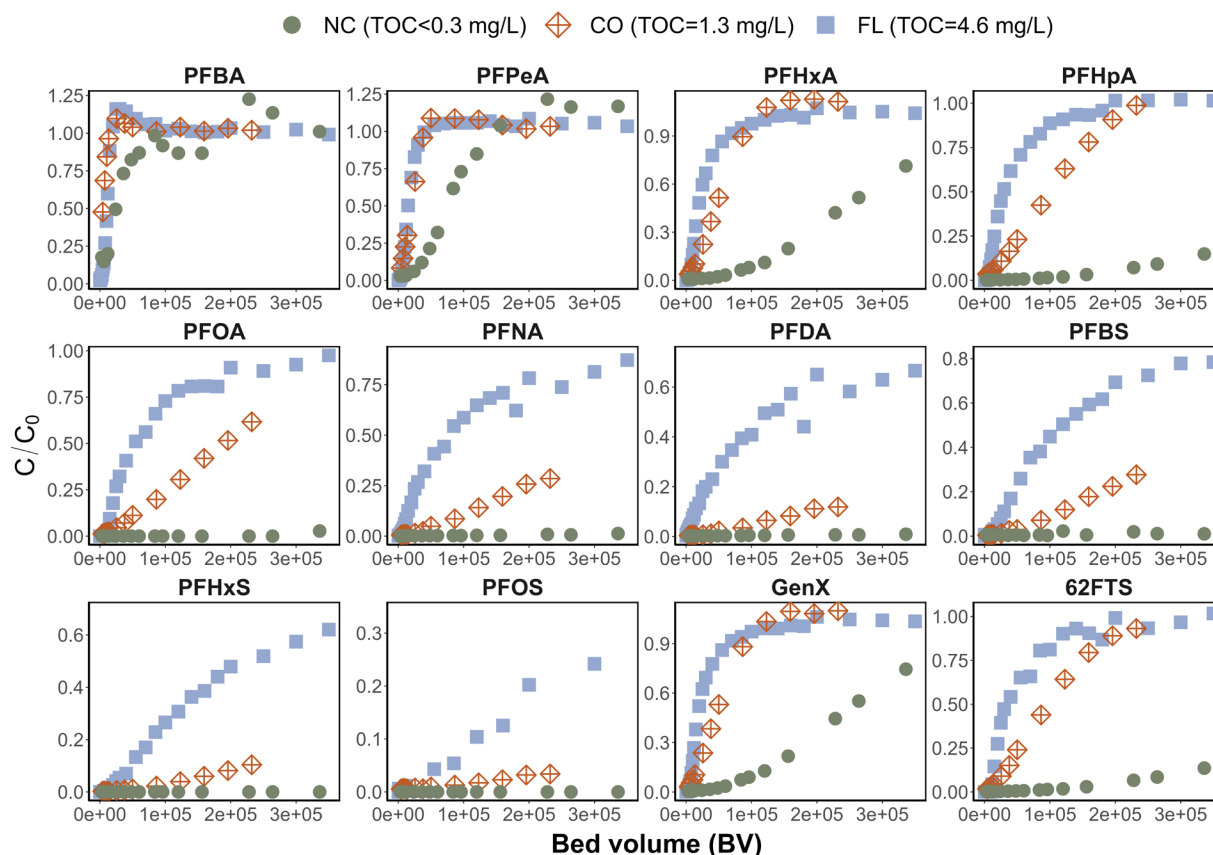


Figure A8. Effect of TOC on PFAS Breakthrough. IX: CalRes2301.

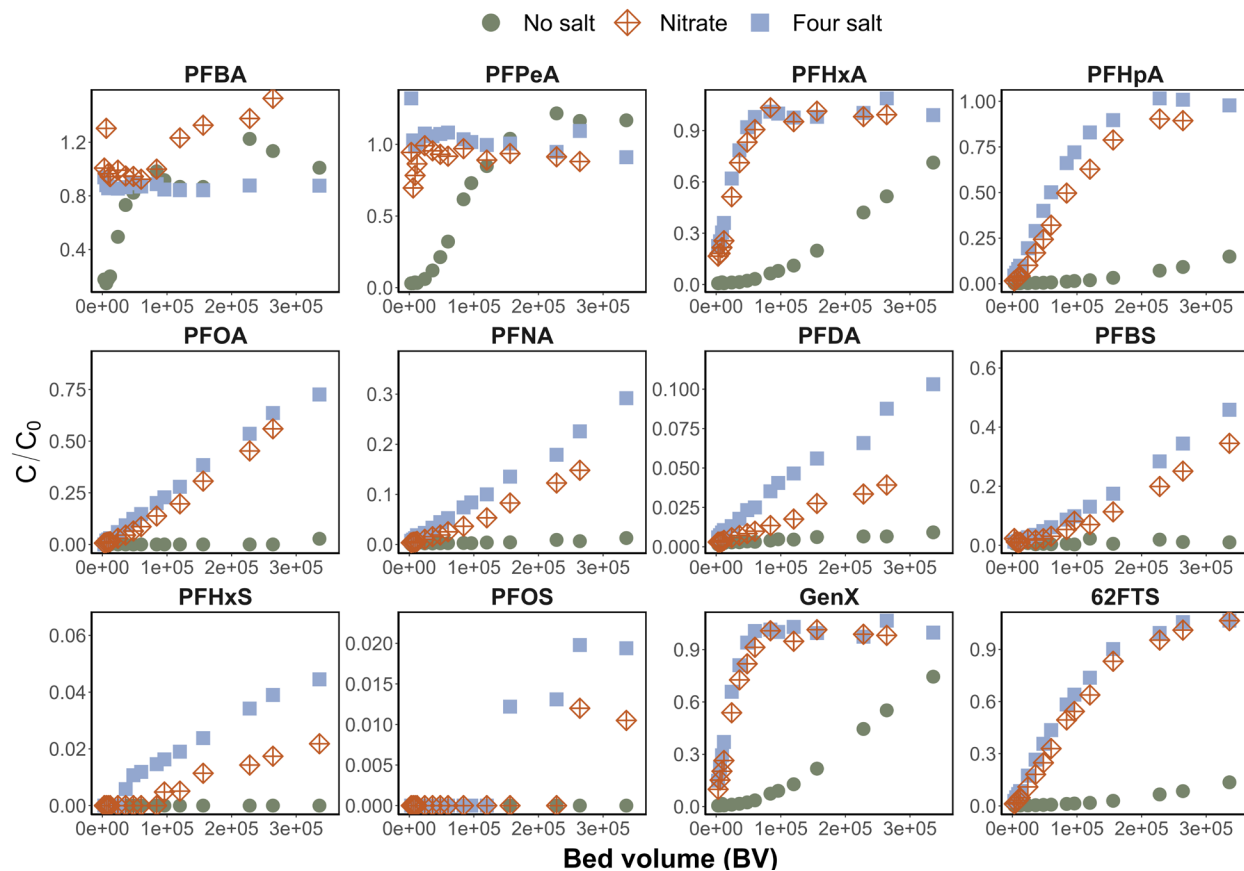


Figure A9. Effect of Nitrate and Coexisting Inorganic Anions on PFAS Breakthrough.

IX: CalRes2301.

APPENDIX B DECISION SUPPORT TOOL USER INTERFACE

The following images show the tool's user interface in Excel along with the steps for user inputs. The yellow highlighted cells indicate fields which require user inputs, and the green highlighted cells indicate option user inputs.

Step 1 Weighting Scheme		
Weighting Scheme	Equal	(Dropdown)

Selected Weighting Scheme:

Global Warming	0.25
Global Environmental and Human Health	0.25
Affordability	0.25
Facility Operation	0.25

Step 1

- Select weighting scheme

If "User Defined" selected

- Go to "Weighting Schemes" tab to input values

Step 2 General Design Info		
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Flowrate Presets	0.03 MGD	2.2 MGD
	0.12 MGD	7.4 MGD
	0.31 MGD	23 MGD
	0.74 MGD	75 MGD

Region/Electrical Grid	SERC	(Dropdown)
Include Building Structure?	No	(Dropdown)
Influent TSS	2	mg/L
Effluent TSS	0.5	mg/L
Pump Efficiency	0.75	

Step 2

- Select flowrate (8 options; see Step 5 if different value needed)
- Select Region/Electrical Grid

For the following, you can update default values

- Select if building structure is included
- Input influent and effluent TSS
- Input pump efficiency

Step 3 GAC and IX Info		
*Check to include		Media Usage Rate
GAC Gravity Basin	☒	GAC Type 1 0.056 (lb/1000 gal)
	☐	No input required
	☐	No input required
GAC Pressure Vessel	☒	GAC Type 1 0.056 (lb/1000 gal)
	☐	No input required
	☐	No input required
IX Filter	☒	Resin Type 1 0.023 (lb/1000 gal)
	☐	No input required
	☐	No input required

GAC Disposal	RCRA Landfill	(Dropdown)
IX Disposal	RCRA Landfill	(Dropdown)
IX Foulant Prevention	Cartridge Prefilter	(Dropdown)
Include GAC/IX Discharge Fees?	Yes	(Dropdown)

Step 3

- Select if GAC Gravity Basin, GAC Pressure Vessel, and/or IX Filter are included for comparison (up to 3 media each)
- Input media usage rate (lb/1000 gal) for each selected media; usage rate is based on media, water quality, and treatment objective
- Select disposal options for spent GAC and IX filter media
- Select fouling prevention for IX filter
- Select if GAC/IX discharge fees for filter backwash water are included

Step 4 RO and NF Info **Step 4**

*Check to include

RO System	☒
NF System	☒

Source Water	Low Quality Groundwater	(Dropdown)
Influent pH	7.0	
Effluent pH Goal	7.5	
Water Temperature	55	°F
RO/NF Membrane Disposal	Incineration	(Dropdown)
RO/NF Concentrate Disposal	Haz Waste Disposal	(Dropdown)
Include RO/NF Discharge Fees	Yes	(Dropdown)

- Select if RO and/or NF are included for comparison
- Select source water quality
- Input influent and effluent pH
- Input water temperature
- Select RO/NF used membrane disposal method
- Select RO/NF concentrate disposal method
- Select if RO/NF discharge fees for membrane cleaning and concentrate are included

Select Regulation	EPA Health Advisory (Dropdown)			
Treatment Objective Type	Sum of Concentrations	(Dropdown)		
	Treatment Objective	Influent Concentration (ng/L)	RO Percent Removal	NF Percent Removal
PFAS				
PFPeA	100000	0	98.5%	98.9%
PFHxA	100000	0	99.9%	98.7%
PFHpA	100000	0	96.9%	98.0%
PFOA	12	52	99.9%	96.7%
PFNA	11	24	99.9%	96.7%
PFPrS	100000	0	98.0%	84.1%
PFBS	100000	0	99.9%	94.9%
PFPeS	100000	0	99.3%	96.1%
PFHxS	18	47	99.9%	97.5%
PFHpS	100000	0	99.7%	87.0%
PFOS	15	88	99.4%	97.4%
Sum	70	211.0		

Step 4 (Continued)

- Select Treatment Objective/Regulation
- If “User Defined” is selected for treatment objective:
- Select Treatment Objective Type
 - If “Sum of Concentrations”, then Input the “Sum” for the treatment objective
 - If “Individual Concentrations”, then Input individual effluent concentrations for each relevant PFAS compound
 - Input relevant influent concentrations
 - Default percent removal values are included for RO and NF, user can adjust if desired

Step 5 Flowrate and Design Specifications **Step 5**

Design Flowrate	2.152	MGD
GAC Gravity Basin		
Bed Depth	8	ft
Basin Width	12	ft
EBCT	10	min
GAC Pressure Vessel		
Vessel Geometry	Upright	(Dropdown)
Bed Depth	7	ft
Vessel Height/Length	11	ft
Vessel Diameter	8.5	ft
EBCT	10	min
IX Filter		
Vessel Geometry	Upright	(Dropdown)
Bed Depth	4	ft
Vessel Height/Length	7	ft
Vessel Diameter	5.5	ft
EBCT	2	min
RO System		
Membrane Element	8 inch Low Pressure RO	(Dropdown)
Recovery Rate	80%	
Design Flux	17	gfd
NF System		
Membrane Element	8 inch NF	(Dropdown)
Recovery Rate	85%	
Design Flux	17	gfd

If Preset Flowrate not selected:

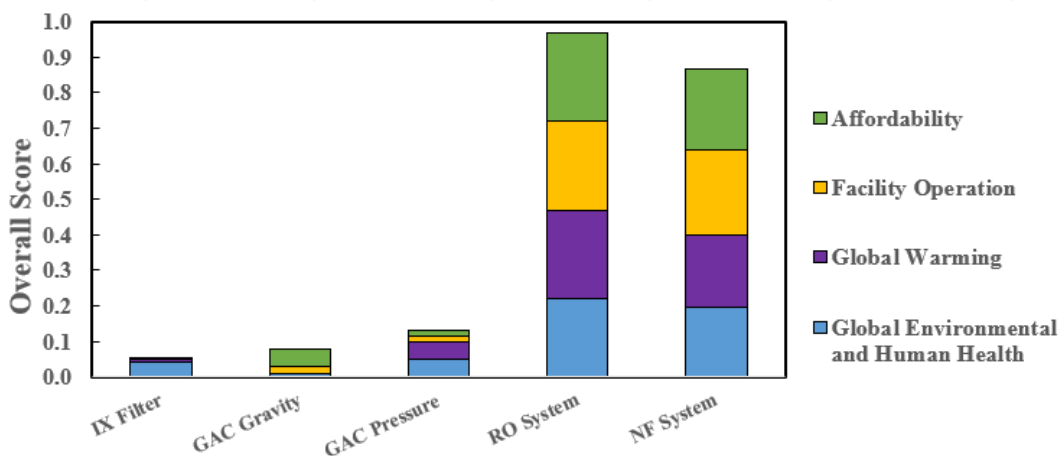
- Input Design Flowrate
- Input Bed Depth, Basin Width, and EBCT for GAC Gravity Basin
- Select Vessel Geometry, Input Bed Depth, Vessel Height/Length, Vessel Diameter and EBCT for GAC Pressure Vessel
- Select Vessel Geometry, Input Bed Depth, Vessel Height/Length, Vessel Diameter and EBCT for IX Filter
- Select Membrane Element, Input Recovery Rate and Design Flux for RO System
- Select Membrane Element, Input Recovery Rate and Design Flux for NF System

Output

Go to “OUTPUT” tab

- Only technologies that were selected for analysis are shown
 - If RO or NF system does not meet selected treatment objective, no output will be shown
- For each technology included in analysis, the tool displays:
 - Treatment system footprint in square feet
 - Bed lifetime (for GAC and IX processes) in months
 - Capital costs in present day value
 - Total operation and maintenance (O&M) costs in present day value
 - Lifetime costs (Capital + O&M costs) in present day value
 - Overall score (overall weighted score based on decision criteria)
- A bar graph of overall score, broken down by decision criteria category, is shown for each selected technology

Technology	Footprint (sq ft)	Bed Lifetime (mo)	Capital Costs (\$)	O&M Costs (\$/yr)	Lifetime Cost (\$)	Overall Score
IX Filter	280	11	\$ 106,349	\$ 46,322	\$ 1,959,216	0.051
IX Filter 2						
IX Filter 3						
GAC Gravity	650	48	\$ 310,204	\$ 41,662	\$ 1,976,690	0.077
GAC Gravity 2						
GAC Gravity 3						
GAC Pressure	520	39	\$ 188,174	\$ 45,061	\$ 1,990,607	0.132
GAC Pressure 2						
GAC Pressure 3						
RO System	2,010	N/A	\$ 634,770	\$ 26,525,406	\$ 1,061,651,023	0.966
NF System	1,990	N/A	\$ 646,443	\$ 21,575,617	\$ 863,671,129	0.867



APPENDIX C COST NUMBERS

Table C1 - Instrumentation	
Instrument	\$/unit
Alarms	593
Conductivity Meters	1616
Head Loss Sensor	2121
pH Meter	2632
Temperature Meter	581
Turbidimeter	5118

Table C3 - Backwash Distributors for GAC Gravity Basins	
Size Needed (cell length)	\$/unit
0	41,969
9	44,371
11	46,773
13	58,656
15	61,942
17	79,261
19	83,559
21	93,546
23	96,327
25	119,081
27	137,537
29	142,847
31	contact vendor

Table C6 - PVC Check Valves	
Size needed (in. diam)	\$/unit
0	80
0.6	87
0.76	104
1.1	159
1.51	208
2.1	848
2.6	513
3.1	709
4.1	1,462
6.1	2,163
8.1	3,223
10.1	4,519
12.1	contact vendor

Table C8 - PVC Piping			
Size needed (in. diam)	\$/lf	Size needed (in. diam)	\$/lf
0	3	10.1	31
0.6	4	12.1	23
0.76	6	14.1	29
1.1	9	16.1	36
1.6	2	18.1	44
2.1	3	20.1	60
2.6	5	24.1	104
3.1	8	30.1	147
4.1	11	36.1	197
6.1	16	42.1	247
8.1	24	48.1	contact vendor

Table C2 - Construction	
Process	\$/cy
Backfill and Compaction	14.41
Excavation	31.07

Table C4 - Fiberglass Pressure Vessels, Plastic Internals	
Size needed (gal)	\$/unit
0	4,262
188	5,458
265	7,392
346	9,164
464	17,070
901	contact vendor

Table C5 - Metalwork		
Aluminum Railing	36.00	\$/lf
Aluminum Stairs	541.43	\$/riser

Table C7 - Polypropylene/PVC Manual Valves	
Size needed (in. diam)	\$/unit
0	123
0.6	136
0.76	153
1.1	325
1.6	330
2.1	358
3.1	471
4.1	556
6.1	1,129
8.1	1,892
10.1	2,446
12.1	7,399
14.1	11,868
16.1	15,277
18.1	17,021
20.1	24,298
24.1	contact vendor

Table C9 - Labor Costs			
Design Flow (mgd)	Technical Rate (\$/hr)	Managerial Rate (\$/hr)	Clerical Rate (\$/hr)
0.03	31.12	44.12	29.72
0.12	31.12	44.12	29.72
0.31	31.12	44.12	29.72
0.74	31.12	44.12	29.72
2.2	33.21	50.46	29.72
7.4	35.05	56.21	38.38
23	36.59	65.59	38.38
75	42.76	70.07	38.38

Table C10 - Disposal and Hauling	
Non-Hazardous Landfill	\$ 53.72 /ton
Non-Hazardous Incineration	\$ 64.43 /ton
Non-Hazardous hauling	\$ 0.52 /ton-mile
Hazardous waste disposal	\$ 268.60 /ton
Hazardous waste minimum charge per shipment	\$ 3,132.43 /shipment
Hazardous waste hauling	\$ 1.04 /ton-mile
Liquid hazardous waste disposal	\$ 3.05 /gal
Liquid hazardous waste minimum charge per shipment	\$ 3,132.43 /shipment
Liquid hazardous waste hauling	\$ 0.00 /gal-mile
POTW discharge base rate (average)	\$ 15.66 /month
POTW discharge volume rate (average)	\$ 4.42 /1,000 gallons

Table C11 - Land Costs		
System Size greater than (MGD)	Assumed Population	\$/acre
0	25-100	22,071
0.03	101-500	22,047
0.12	501-1,000	16,524
0.31	1,001-3,300	19,718
0.74	3,301-10,000	36,189
2.2	10,001-50,000	46,532
7.4	50,001-100,000	52,022
23	100,001-1,000,000	111,428

Table C12 - HDPE Backwash Tanks	
Size Needed (gal)	Avg. Unit Cost
0	1,700
301	1,501
551	1,992
851	3,433
906	1,978
1001	3,894
1051	2,203
1201	2,418
1501	2,689
2001	3,807
2501	4,072
3001	5,652
4201	8,327
5201	9,370
6501	12,242
8001	7,764
10001	12,509
12001	17,317
15001	contact vendor

Table C13 - Centrifugal Pumps (Backwash)	
Size needed (gpm)	Avg. Unit Cost
0	5,569
11	5,796
31	6,608
51	7,768
56	8,244
76	8,244
101	10,543
121	8,886
126	10,543
141	9,861
201	10,404
251	10,951
301	14,818
501	15,245
751	17,479
1001	20,907
1501	31,038
2001	35,661
3001	40,615
3501	42,600
4001	88,269
6001	79,061
10001	contact vendor

Table C14 - Carbon Steel Pressure Vessels, Stainless Steel Internals	
Size needed (gal)	Avg. Unit Cost
0	14,195
181	17,221
301	20,641
651	22,793
751	25,297
881	27,376
951	37,238
1401	41,788
1701	40,440
2001	50,393
2901	54,965
4301	61,648
4701	62,881
6801	198,842
17001	260,763
27101	contact vendor

Table C15 - Carbon Steel Pressure Vessels, Plastic Internals	
Size needed (gal)	\$/unit
0	12,226
181	12,252
301	12,656
651	13,581
751	14,487
881	15,706
951	20,732
1401	22,181
1701	26,495
2001	27,643
2901	36,007
4301	36,623
4701	45,721
6801	103,845
17001	169,140
27101	contact vendor

Table C16 - Membrane Elements - NF/RO	
Element Selected	\$/sf
4 inch NF	3.58
8 inch NF	1.80
16-18 inch NF	1.17
4 inch Low Pressure RO	2.76
8 inch Low Pressure RO	1.47
16-18 inch Low Pressure RO	1.14
4 inch Brackish Water RO	2.64
8 inch Brackish Water RO	1.51
16-18 inch Brackish Water RO	1.13

Table C17 - Cartridge Filters			
Cartridge Filter Systems for Pretreatment		Replacement Cartridge Filters for RO	
Size needed (mgd)	\$/unit	Size needed (mgd)	\$/unit
0	5020	0	182
0.217	16918	0.051	182
0.649	31231	0.145	205
1.081	43264	0.201	285
1.729	43353	0.501	285
2.305	49371	1.001	285
2.881	contact vendor	2.001	285
		5.001	contact vendor

Table C18 - Chemicals	
Chemical	\$/lb
Antiscalant - Basic	1.84
Citric Acid (food grade)	1.12
Hydrochloric Acid	0.33
Sodium Hydroxide - Large Qty	0.13
Sodium Hydroxide - Mid Qty	0.21
Sodium Hydroxide - Small Qty	0.30
Sulfuric Acid - Large Qty	0.11
Sulfuric Acid - Small Qty	0.31

Table C19 - Chemical Pumps	
Size needed (gph)	\$/unit
0	1159
300	1186
600	1848
1020	1516
1200	1964
1500	6527
11400	6800
14400	21576
30000	contact vendor

Table C20 - Magnetic Flow Meters	
Size needed (in. diam)	\$/unit
0	5267
0.76	3737
1.1	3831
1.6	4209
2.1	4106
2.6	4481
3.1	4714
4.1	7747
6.1	9979
8.1	12077
10.1	14358
12.1	17614
14.1	21512
16.1	25120
18.1	28272
20.1	contact vendor

Table C22 - Teflon Immersion Heaters	
Size needed (kW)	\$/unit
0	633
0.5	760
1	1004
2	1303
3	1469
4	1672
5	1911
6	2212
8	2478
9	2499
10	2848
12	3444
15	3866
18	5319
24	contact vendor

Table C21 - Plastic (XLPE) Chemical Storage Tanks			
Size Needed (gal)	\$/unit	Size Needed (gal)	\$/unit
0	606	551	1918
61	486	601	1660
66	1589	751	1771
101	1465	826	2507
126	1385	851	2109
131	1550	1001	2075
151	1612	1026	3624
201	1943	1051	2996
206	1465	1201	2885
221	1538	1501	2652
231	1764	1601	3312
251	2053	2001	3595
301	1774	2401	4072
311	2003	2501	6544
326	1725	2551	4798
331	1679	3001	7675
451	2302	4201	10973
501	1943	5201	12611
526	2994	5301	14060
541	1825	6501	14183

Table C23 - Membrane Trains			
16-18 inch RO/NF Pressure Vessels - Base		16-18 inch RO/NF Pressure Vessels - Per Element	
Size needed (psi)	\$/unit	Size needed (psi)	\$/unit
0	contact vendor	0	contact vendor
150.1	7760	150.1	411
300.1	contact vendor	300.1	contact vendor
4 inch RO/NF Pressure Vessels - Base		4 inch RO/NF Pressure Vessels - Per Element	
Size needed (psi)	\$/unit	Size needed (psi)	\$/unit
0	318	0	78
300.1	382	300.1	70
600.1	contact vendor	600.1	contact vendor
8 inch RO/NF Pressure Vessels - Base		8 inch RO/NF Pressure Vessels - Per Element	
Size needed (psi)	\$/unit	Size needed (psi)	\$/unit
0	797	0	81
150.1	792	150.1	81
300.1	860	300.1	92
450.1	contact vendor	450.1	contact vendor
Steel Beams for Pressure Vessel Rack			
Size needed (in. diam)	\$/unit		
0	33		

Table C24 - Feed Water Pumps	
300 PSI Feed Water Pumps for Reverse Osmosis	
Size needed in (gpm)	\$/unit
0	4694
6	2994
8	5440
26	7902
40	9518
62	37584
80	23176
120	28835
300	contact vendor
350 PSI Feed Water Pumps for Reverse Osmosis	
Size needed in (gpm)	\$/unit
0	2994
4	5005
14	5440
21	25832
71	24152
160	34555
350	contact vendor
500 PSI Feed Water Pumps for Reverse Osmosis	
Size needed in (gpm)	\$/unit
0	5173
15.1	16287
33.1	17084
75.1	21718
115.1	30860
185.1	contact vendor

Table C25 - Stainless Steel Piping	
Size needed (in. diam)	\$/lf
0	20
0.6	25
0.76	28
1.1	33
1.26	42
1.6	60
2	79
3	96
3	115
4	208
6	302
8	423
10	540
12	168
16	1098
36	1676
48	contact vendor

Table C26 - Low Quality Building, No Heating or Cooling	
Size needed (sf)	\$/sf
0	90.90
500.1	80.18
1000.1	75.10
2000.1	72.77
3000.1	69.32
4000.1	66.19
5000.1	64.49
7500.1	64.08
8000.1	61.57
10000.1	62.12
12000.1	59.96
15000.1	58.92
18000.1	57.58
20000.1	58.62
24000.1	56.77
25000.1	56.34
30000.1	56.42
36000.1	54.92
42000.1	54.72
48000.1	54.27
50000.1	54.29
54000.1	54.45
60000.1	52.50
100000.1	51.01

APPENDIX D EXPERT PANEL WORKSHOPS

As part of the project scope, we held two expert panel workshops to present the findings from the treatability evaluation, solicit input on the draft tool, and inform potential users about the tool.

Workshop #1

The objective of the first workshop was to obtain industry buy-in to the life cycle assessment and costing framework prior to treatability studies. It was held on February 12, 2020 in Denver, Colorado. Twenty-eight people attended the meeting, including eighteen professionals with expertise in PFAS treatment and LCA/LCC, research team members, and students.

Workshop #1 Participants

Name	Organization
Andrew Henderson	Noblis
Chris Olsen	OCWD
Chris Tadanier	Black & Veatch
Dana Gonzales	HRSD
Elisabeth Hawley	Geosyntec
Eric Dickenson	SNWA
Jacqueline Rhoades	Hazen and Sawyer
Jeff McDonough	Arcadis
Jeremy Guest	University of Illinois
Kyle Thompson	SNWA
Robert Raczko	SUEZ
Scott Grieco	Jacobs
Timothy J. Strathmann	CSM
Tom Speth	USEPA
Trevor Boyer	ASU
Vincent Monaco	American Water
YuJung Chang	CDM Smith
Zaid Chowdhury	Garver
Research Team and Students	
Chris Bellona	CSM
Andrew Salfuko	CSM
Charles Schaefer	CDM Smith
Charlie Liu	CSM
Conner Murray	CSM
Detlef Knappe	NCSU
Kenan Ozekin	WRF
Pingping Meng	NCSU
Robert Eric Marshall	CSM
Sherri Cook	CU-Boulder

Prior to the workshop, the research team had developed a comprehensive agenda that included a number of presentations to set the stage and provide information on ongoing work being conducted by the research team members. After the presentations, the participants were divided into three different subgroups: Life Cycle Cost Assessment, Treatment, and Treatment Scenarios to discuss the available information and parameters that should be measured under each topic. Research team members facilitated the discussions. The outcome from each subgroup has been presented to the whole group for input. Below are the high-level summaries from each subgroup discussion. Input received from the workshop participants was used to finalize the treatability studies design for the project.

Treatment Breakout

GAC and IX

- Water matrix
 - TOC, anions (sulfate, nitrate), pretreatment (Fe/Mn/SO₄/NO₃), PFAS feed concentration, Co-Contaminants (TCE, PCE, 1-4 Dioxane, Hydrocarbons)
- Scalability
 - Breakthrough modeling, Batch/RSSCT/Pilot, Data Interpretation
- Representation of performance
 - Carbon usage rate, bed volumes vs. gallons treated
- Breakthrough classification
 - Standardize percent to breakthrough
- Regeneration
 - Byproducts, SOP, incineration of IX/GAC, standardized terminology (mineralization/ sequential defluorination)
- Treatment Goals
 - Suspect screening, polyfluorinated PFAS, branched PFAS
 - Potable or remediation

High-Pressure Membranes

- Energy cost / operating problems
 - Concentrate management, permeate flux loss
- Combined treatment train

Treatment Scenarios Breakout

- Source Characterization:
 - Groundwater
 - Surface Water
 - Landfill Leachate
 - Fire-Fighting Activities Collection Areas
 - What are the x number of parameters that matter...
 - What is the list for IX and GAC?
 - Getting a specific type of NOM characterization.... EEMS, UV₂₅₄, TSS.
 - Source of the TOC
 - Developing surrogates for PFAS.

- Ionic composition
- What is the level of Iron and Manganese that is impactful?
- Co-contaminants
- Ionic Strength
- Species: PFCAs, PFSAAs
 - Non-Target?
 - TOP-Assay?
 - What suite of analysis was performed? UCMR List, EPA 537
- Concentrations: Low vs High, what is the balance?
- Source of PFAS: foam, industrial etc.

Life Cycle Cost Assessment Breakout

- Previous Literature
- Scenario Analyses – designing analysis around key determinants (replacement rate, influent)
 - Where data availability is limited: media capacity, capital costs (emphasize O&M/recurring costs)
 - Variability among regions, regulations, and treatment goals as well as options for waste management
- System Boundary – scope of the LCA
 - Waste as a major source of impact – cannot be ignored – and transport for incineration
 - End-of-life must include waste management and how incineration is to be dealt with (water %, haz waste)
- Inventory analysis – which databases acceptable to DoD, how to choose between them
- Impact Assessment – all impacts results US-oriented and run through TRACI
 - Also give consideration to environmental liability & worker health concerns
- Uncertainty Assessment - Emphasis on ranking. Uncertainty in design rather than inventory
- Costing Data – avoiding vendor-specific data, potential for NDAs, updated EPA excel model
- Other topics – macro for Monte Carlo analysis, distribution options, prioritizing sensitivity
- Form revision:
 - Pilot and full scale data reporting
 - Point person...Kenan?
 - Email Follow-up
 - Compile from past presentations/reports/literature studies
 - Treatment Studies Experimental Design:
 - Sending out experimental matrix for feedback
 - Collaboration Potential:
 - RSSCT design sharing
 - Perhaps a PFAS RSSCT IX and GAC manual for standard comparison
 - Reaching out to Batelle for additional info
 - New adsorbents potentially competitive

- Destructive technologies
 - Developing baseline for comparison
 - Energy comparison

Workshop #2

Due to Covid restrictions, the second stakeholder workshop was held virtually on February 14 and 17, 2022. Twenty-five people representing water utilities, consultants, regulators, and the research team attended the meeting (see Table XX). The goal of the virtual workshop was to share high-level results from the project, demonstrate the tool, and seek participants' feedback. The workshop was opened with introductions, followed by a presentation from the research team sharing results from the lab-scale experiments. After briefly discussing the experiments, the research team provided an overview of the tool and sought feedback. Feedback from these workshops was used to update the tool in several ways, making the design assumptions more accurate and increasing usability.

The following changes were incorporated into the tool based on the input received from the workshop participants: (1) Adding cartridge pre-filters for the IX filter system so that the user can select if they want to use cartridge filters or backwashing to prevent fouling. (2) The user input of bed lifetime was changed to media usage rate in pounds per thousand gallons. Using media usage rate allows the tool to accurately calculate bed lifetime (how often media needs to be replaced) even if inputs such as flow rate, filter dimensions, etc., are changed. (3) Thermal reactivation was added as a disposal option for GAC media, assuming that virgin media is still used for treatment. (4) The tool user guide will be expanded to include explicit statements on the scope and purpose of the decision support tool, how life cycle inventory is calculated and design assumptions; how to check hydraulic loading rate values to ensure proper design; using the tool to model and evaluate lead-lag configurations; and how a user can conduct uncertainty and sensitivity analyses.

Workshop #2 Participants

Name	Organization
Ai Jia	MWD
Alex Gorzalski	Hazen and Sawyer
Balaji Seshasayee	Geosyntec
Carel Vandermeijden	Cape Fear Public Utility Authority
Chris Olsen	OCWD
Chris Tadanier	Black & Veatch
Dana Gonzales	HRSD
Jason Dadakis	OCWD
Jason Speicher	Navy
Jeff McDonough	Arcadis
Megan Plumlee	OCWD
Mike Pickel	Horsham Water and Sewer Authority
Robert Raczko	SUEZ – Retired
Scott Grieco	Jacobs
Stephanie Schlea	ASDWA
Tim Hagey	Warminster Municipal Authority
Tom Speth	EPA
Vincent Monaco	American Water
Zaid Chowdhury	Garver
Research Team	
Chris Bellona	CSM
Charles Schaefer	CDM Smith
Detlef Knappe	NCSU
Kenan Ozekin	WRF
Sherri Cook	CU-Boulder
Cameron Oden	CU-Boulder

Workshop Agenda

Evaluation & Life Cycle Comparison of Ex-Situ Treatment Technologies for PFAS in Groundwater

Expert Panel Workshop

February 11-12, 2020

February 11

Dinner at 7:00 pm – Vista Montagne - Gaylord Rockies Resort & Convention Center

February 12

Location – Gaylord Rockies Resort & Convention Center, Maple 1 & 2 Room

Agenda

Time	Topic	Presenter
7:30 – 8:30 am	Breakfast (Maple/Spruce Pre-function Area)	
8:30 – 9:00 am	Introductions, Background, Overview and Goals of Project	Kenan
9:00 – 9:45 am	Current Projects and Database Development	Charles/Chris/Detlef
9:45 – 10:00 am	LCA/LCC Modeling and Overall Developed Tool Concept	Sherri
10:00 – 10:30 am	Discussion of Technologies and Data Needs	Detlef/Chris/Sherri
10:30 – 10:45 am	Break	
10:45 am – 12:00 pm	Breakout Sessions (LCA, Treatment, Treatment Scenarios)	Facilitated Discussion – LCA (Sherri), Treatment (Detlef/Chris), Treatment Scenarios (Charles)
12:00 – 1:00 pm	Lunch	
1:00 – 2:00 pm	Report Out	Kenan
2:00 – 2:45 pm	Open Discussion on Knowledge Gaps	Kenan
2:45 – 3:15 pm	Development of List of Action Items and Collaborations	Detlef/Charles
3:15 pm	Adjourn	