

**METAL AND ASSIMILABLE ORGANIC CARBON REMOVAL IN  
DRINKING WATER WITH REVERSE OSMOSIS AND ACTIVATED  
CARBON POINT-OF-USE SYSTEMS**

by

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**A Thesis**

*Submitted to the Faculty of Purdue University*

*In Partial Fulfillment of the Requirements for the degree of*

**Master of Science in Engineering**



Division of Environmental and Ecological Engineering

West Lafayette, Indiana

May 2021

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## **ACKNOWLEDGMENTS**

I am grateful to my advisor and committee members for their help and patience in my training as a researcher: My advisor Dr. Zhi Zhou for patience, support, and help, Dr. Nadya Zyaykina for her help and training in chemical analysis theory and instrument operation, and Dr. Amisha Shah and Dr. Loring F. Nies for the contribution of their knowledge. I would like to thank Mr. Yifei Bi for his help in ICP-OES analysis and Ms. Arlene P. Rothwell for her help in ICP-MS analysis. For AOC analysis, I would like to thank my undergraduate team of Sungmin Hong, Benjamin Jolkovsky, Hannah Congleton, and Sophie Mendelson. Many thanks to Mr. Lingjun Xu for helping me with my TDS analysis. I would also like to thank Ms. Shreya Gupta, my fellow lab mate, for her support and the time we worked together. Without the help and support from all of them, this thesis would not have been possible.

*To my dearest parents and sister for their love and support through my graduate study*

## TABLE OF CONTENTS

LIST OF TABLES .....	7
LIST OF FIGURES .....	8
ABSTRACT .....	10
1. INTRODUCTION .....	11
1.1 Background .....	11
1.2 Research objectives .....	11
2. Literature review .....	12
2.1 POU water treatment systems .....	12
2.2 Emerging contaminants .....	12
2.3 Treatment technologies to remove emerging contaminants .....	13
2.4 Manganese .....	13
2.5 Uranium .....	14
2.6 Assimilable organic carbon .....	14
3. MATERIALS AND METHODS .....	16
3.1 Chemicals and bacteria .....	16
3.2 Analysis of heavy metals .....	17
3.3 Analysis of AOC .....	19
3.4 Analysis of TDS .....	22
3.5 Configuration of POU systems .....	23
4. RESULTS AND DISCUSSION .....	28
4.1 Removal of low concentration metals in AC systems .....	28
4.2 Removal of low concentration metals in RO systems .....	36
4.3 Removal of low concentration AOC in AC and RO systems .....	43
4.4 Removal of medium concentration metals in AC systems .....	46
4.5 Removal of medium concentration metals in RO systems .....	53
4.6 Removal of medium concentration AOC in AC and RO systems .....	60
4.7 Total dissolved solid and pH .....	62
5. CONCLUSIONS AND FUTURE WORK .....	64
5.1 Conclusion .....	64

5.2 Future work .....	65
REFERENCES .....	66

## LIST OF TABLES

<b>Table 1.</b> Concentrations of Mn, U, and AOC spiked in influent.....	16
<b>Table 2.</b> Method parameters for metals determination with ICP-OES .....	17
<b>Table 3.</b> Instrument (ICP-OES) operating parameters .....	18
<b>Table 4.</b> Metal concentrations in tap water, groundwater, and ultrapure water. ....	18
<b>Table 5.</b> Water quality of tap water and groundwater.....	23
<b>Table 6.</b> Specifications of AC systems tested in this study.....	23
<b>Table 7.</b> Specifications of RO systems tested in this study.....	25
<b>Table 8.</b> pH analysis for medium concentration AC samples .....	62
<b>Table 9.</b> pH analysis for medium concentration RO samples .....	62
<b>Table 10.</b> TDS analysis for medium concentration AC samples .....	63
<b>Table 11.</b> TDS analysis for medium concentration RO samples .....	63

## LIST OF FIGURES

<b>Figure 1.</b> Bacterial colonies of <i>Pseudomonas fluorescense</i> , P17 and <i>Spirillum</i> sp. Stain NOX .	20
<b>Figure 2.</b> Agar plates used in AOC testing. ....	20
<b>Figure 3.</b> Correlation between AOC and CFU.....	21
<b>Figure 4.</b> Configuration of AC systems. ....	24
<b>Figure 5.</b> Variation of flow rates of AC systems under 50 PSI. ....	25
<b>Figure 6.</b> Configuration of RO systems. ....	26
<b>Figure 7.</b> Removal of low concentration Ca in AC systems.....	30
<b>Figure 8.</b> Removal of low concentration Mg in AC systems.....	31
<b>Figure 9.</b> Removal of low concentration Fe in AC systems. ....	32
<b>Figure 10.</b> Removal of low concentration Cu in AC systems.....	33
<b>Figure 11.</b> Removal of low concentration Mn in AC systems.....	34
<b>Figure 12.</b> Removal of low concentration U in AC systems. ....	35
<b>Figure 13.</b> Removal of low concentration Ca in RO systems.....	37
<b>Figure 14.</b> Removal of low concentration Mg in RO systems.....	38
<b>Figure 15.</b> Removal of low concentration Fe in RO systems. ....	39
<b>Figure 16.</b> Removal of low concentration Cu in RO systems.....	40
<b>Figure 17.</b> Removal of low concentration Mn in RO systems.....	41
<b>Figure 18.</b> Removal of low concentration U in RO systems. ....	42
<b>Figure 19.</b> Removal of low concentration AOC in AC and RO systems.....	45
<b>Figure 20.</b> Removal of medium concentration Ca in AC systems.....	47
<b>Figure 21.</b> Removal of medium concentration Mg in AC systems.....	48
<b>Figure 22.</b> Removal of medium concentration Fe in AC systems. ....	49
<b>Figure 23.</b> Removal of medium concentration Cu in AC systems.....	50
<b>Figure 24.</b> Removal of medium concentration Mn in AC systems.....	51
<b>Figure 25.</b> Removal of medium concentration U in AC systems. ....	52
<b>Figure 26.</b> Removal of medium concentration Ca in RO systems.....	54
<b>Figure 27.</b> Removal of medium concentration Mg in RO systems.....	55



<b>Figure 28.</b> Removal of medium concentration Fe in RO systems. ....	56
<b>Figure 29.</b> Removal of medium concentration Cu in RO systems.....	57
<b>Figure 30.</b> Removal of medium concentration Mn in RO systems.....	58
<b>Figure 31.</b> Removal of medium concentration U in RO systems. ....	59
<b>Figure 32.</b> Removal of medium concentration AOC in AC and RO systems. ....	61

## ABSTRACT

Activated carbon (AC) systems and reverse osmosis (RO) systems are commonly used point-of-use (POU) water filtration systems as the last barrier to remove trace-level contaminants in tap water to protect human health. Limited studies have been done to evaluate trace-level manganese and uranium removal in tap water. Additionally, undesired microbial growth in POU systems may reduce treatment efficiencies of POU systems and limited studies have been done to evaluate microbial growth potential in POU systems. The overall research objective of this study was to systematically evaluate the removal of metals and assimilable organic carbon in POU systems. AC systems were operated to 200% of their designed treatment capacities and RO systems were operated for three weeks. The results indicated that AC systems were generally ineffective to remove metals in drinking water, while metals were effectively removed in RO systems. The results showed that calcium and magnesium were not effectively removed in AC systems with removal efficiencies of less than 1%. Various fractions of iron were removed with its removal efficiencies in AC systems ranged between 61% and 84%. Copper was effectively removed in AC systems with removal efficiencies greater than 95%, which was possibly related to its low influent concentration in drinking water ( $<30 \mu\text{g/L}$ ). Both manganese and uranium were ineffectively removed from AC systems. Different from AC systems, RO systems were consistently effective to remove all metals. Calcium, magnesium, iron, and copper were all removed with removal efficiencies greater than 98%, while removal efficiencies of manganese and uranium in RO systems were above 95%. Assimilable organic carbon was effectively removed from all AC and RO systems and high variability of AOC removal efficiencies were observed, which may be attributed to the heterogeneity of biofilm and microbial growth in POU systems. The new knowledge generated from this study can help improve our understanding of emerging contaminant removal in POU systems and develop better strategies for the design and operation of POU systems to remove emerging contaminants in drinking water and mitigate their health risks to humans.

# 1. INTRODUCTION

## 1.1 Background

Point-of-use (POU) water treatment systems in consumers' houses or buildings provide benefits of removing emerging contaminants that remain in treated municipal water, such as manganese (Mn) and uranium (U). Currently, the United States Environmental Protection Agency (EPA) standard for Mn in drinking water is 50 µg/L and the minimum reporting level of Mn in the Unregulated Contaminant Monitoring Rule 4 (UCMR 4) is 0.4 µg/L. The maximum contaminant level of U allowed by EPA is 30 µg/L in drinking water. However, limited studies have been done on Mn and U removal in POU systems. Reverse osmosis (RO) filtration systems and activated carbon (AC) filtration systems are two highly efficient POU systems to treat remaining chemicals in tap water, but their potential to remove U and Mn has not been fully explored. Additionally, undesired microbial growth on membrane surface significantly reduces membrane flux and microbial growth on AC may also reduce active adsorption sites for chemical removal. These issues may significantly reduce treatment efficiencies of POU systems and increase maintenance costs, but limited studies have been done to evaluate the efficiencies and mechanisms of metal removal and microbial growth in POU systems.

## 1.2 Research objectives

The *overall research objective* of this study was to systematically evaluate metal removal and AOC removal in both RO and AC POU systems. The *specific research objectives* were to:

- 1) evaluate the removal efficiencies and resulting effluent concentrations of representative metals in POU systems,
- 2) evaluate the removal efficiencies and resulting effluent concentrations of AOC in POU systems.

The *research hypotheses* are:

- 1) RO systems are more efficient to remove metals and AOC than AC systems,
- 2) metal concentrations in RO effluent are lower than existing drinking water standards,
- 3) AOC removal may be correlated with POU system performance.

## **2. LITERATURE REVIEW**

### **2.1 POU water treatment systems**

POU systems provide additional treatment capabilities in consumers' houses or buildings to remove contaminants in drinking water. The purpose of a POU water treatment system is to treat water right before consumption and is different from point-of-entry (POE) water treatment system, which is a water treatment system before the distribution of water for the whole house. POU water treatment systems can help achieve compliance with some of the maximum contaminant levels (MCLs) established in the National Primary Drinking Water Regulations (NPDWRs) of the Safe Drinking Water Act (SDWA) (Environmental Protection Agency, 2006). POU systems treat water close to consumers on an as-needed basis and provide additional protection for consumers through the removal of emerging contaminants that may be not regulated.

### **2.2 Emerging contaminants**

Emerging contaminants including “endocrine-disrupting chemicals (EDCs)”, “pharmaceuticals and personal care products (PPCPs)”, and “trace organic compounds (TOrcs)”, are defined as “natural and manmade substances, including elements and inorganic and organic chemicals, detected within the water and the environment” by the Water Environment Federation (WEF) (Salveson, Brown, & Zhou, 2010). Hundreds of chemicals have been implicated as potential emerging contaminants based on a variety of criteria (IEH, 2005). Emerging contaminants even at sub- $\mu\text{g/L}$  levels can pose various health risks to aquatic organisms, such as the feminization of male fish or masculinization of female fish. Although evidence is sparse on the adverse effects of most emerging contaminants on human health, their potential health risks to the human endocrine system cannot be neglected. The EPA established the Endocrine Disruptor Screening Program (EDSP) to develop standardized toxicity tests to determine whether a particular chemical is an EDC based on a chemical's interference with estrogen, androgen, or thyroid action (EPA-EDSP, 2008). So far, emerging contaminants are not regulated concerning their potential to cause endocrine disruption (U.S. EPA, 2005). Except for perchlorate, which was regulated in drinking water in Massachusetts based on its potential to interfere with thyroid function (Massachusetts DEP, 2006), no other state in the U.S. has regulated emerging contaminants in

drinking water. Because of the potential health concerns of emerging contaminants, there is an urgent need to remove emerging contaminants from drinking water.

### **2.3 Treatment technologies to remove emerging contaminants**

Various water treatment technologies have been developed to remove emerging contaminants. Most conventional water treatment methods were inefficient for emerging contaminant removal, except for AC adsorption, membrane filtration, and advanced oxidation processes (AOPs). Although emerging contaminants can be effectively removed by a few treatment technologies, high treatment costs prevent their wide application. Activated carbon filters need regular regeneration and non-regenerated activated carbon filters displayed no removal of emerging contaminants (Snyder et al., 2007). The construction costs and operation and maintenance costs of membrane systems are much higher than conventional water treatment techniques. Storage of O<sub>3</sub> and H<sub>2</sub>O<sub>2</sub> increases the operation costs of AOPs, and sleeve fouling and poor transmission in water samples may limit the efficiency of UV-based AOP processes. The scavenging of strong oxidants by natural organic matter (NOM) and other natural water components may also reduce the efficiency of AOPs. Enzymatic treatment and ferrate (VI) oxidation were also costly. As these treatment technologies are often expensive, they are not frequently used in drinking water treatments for unregulated emerging contaminants. POU systems could provide an additional barrier for trace-level emerging contaminants that remain at drinking water before consumption and studies are needed to evaluate emerging contaminant removal efficiencies in POU systems.

### **2.4 Manganese**

As an inorganic emerging contaminant, Mn is often detected in tap water with public health concerns on neurological impairment (Park & Berg, 2018), idiopathic Parkinson's disease (Tarale et al., 2018), metabolic syndrome (Lo et al., 2021), kidney stones (Ferraro, Gambaro, Curhan, & Taylor, 2018), and possible association with sex-specific kidney function and glucose metabolism interruptions (Yang et al., 2020). Previous studies have shown neurobehavioral impairment risk from long-term exposure to Mn, especially in neuro-degenerative elder aging populations (Weiss, 2010) and young age groups undergoing neuro-development (Lucchini, Aschner, Landrigan, &

Cranmer, 2018; Mora et al., 2018). Several studies have been done on metal removal in POU systems. The mechanism for metal removal in POU systems includes adsorption (S. Chen, Hong, Yang, & Yang, 2013; Nejadshafiee & Islami, 2019; Yakout, Metwally, & El-Zakla, 2013), ion exchange (Dong et al., 2018; Z. Liu et al., 2020; Oloibiri, De Coninck, Chys, Demeestere, & Van Hulle, 2017), and membrane separation (Azamat, Khataee, & Joo, 2014; X. Chen et al., 2020; X. Y. Lou, Xu, Bai, Resina-Gallego, & Ji, 2020). There are limited studies on Mn removal with AC (Coleman, Coronado, Maxwell, & Reynolds, 2003) and further studies are needed to evaluate Mn removal in drinking water with POU systems.

## **2.5 Uranium**

As another emerging contaminant, U is a heavy metal that is less commonly found in drinking water but can be a health concern for regions close to phosphorous fertilizers, untreated mining activity, mining waste, and weapon manufacture and testing (Bigalke, Schwab, Rehmus, Tondo, & Flisch, 2018; Redvers, Chischilly, Warne, Pino, & Lyon-Colbert, 2021; Semenova et al., 2020; Surdyk et al., 2021). U in drinking water has been linked to leukemia (Winde, Erasmus, & Geipel, 2017), nephrotoxic effects (Kurtio et al., 2002), and tumors and liver diseases (Banning & Benfer, 2017). Chronic U exposure in drinking water may result in cancer, kidney disease, and immune dysfunction (Bjorklund et al., 2020; Bolt, Medina, Lauer, Liu, & Burchiel, 2019). U can be effectively removed by RO and its removal efficiency was affected by pH and concentration polarization (Schulte-Herbrüggen, Semião, Chaurand, & Graham, 2016). U can also be removed by AC and removal efficiency was affected by pH, contact time, initial concentration, and temperature (Katsoyiannis & Zouboulis, 2013). Additional studies on the removal of U at different concentrations in tap water with POU systems are needed to evaluate the effectiveness of U removal in different POU systems.

## **2.6 Assimilable organic carbon**

Depending on the quality and age of household plumbing systems, biofilm may develop in plumbing systems that can leech out metals and carbon contents from the pipes between water entry and tap (Proctor, Reimann, Vriens, & Hammes, 2018). Water quality can vary significantly

by household and region, and POU systems may help remove metals and microbes before consumption (Bosscher, Lytle, Schock, Porter, & Del Toral, 2019; Chaidez & Gerba, 2004). However, one study showed that tap water treated with POU filtration contained higher concentrations of opportunistic bacterial pathogens than those in sample water without POU treatment (Chaidez & Gerba, 2004), suggesting that POU systems may promote microbial growth, and further studies on microbial growth potential in POU systems are needed.

Assimilable organic carbon (AOC) is widely detected in drinking water distribution systems (W. Liu et al., 2002a) been often used as an indicator for microbial growth. AOC are carbon sources readily available to be incorporated into microorganisms and may also be a good indicator for microbial growth potential in POU systems. AOC is a fraction (0.03-27%) of dissolved organic carbon (DOC) (Dirk van der Kooij, Visser, & Hijnen, 1982) or a fraction (0.1-9%) of total organic carbon (TOC) (D. van der Kooij, 1990). AOC has been used as an important indicator for the regrowth potential of heterotrophic bacteria (Escobar, Randall, & Taylor, 2001; Ma, Ibekwe, Leddy, Yang, & Crowley, 2012). Additionally, AOC has been correlated with the growth potential of pathogens (Vital, Stucki, Egli, & Hammes, 2010) or fouling development (L. Weinrich, LeChevallier, & Haas, 2016). The structural and chemical properties of AOC are still not well known, but as an indicator of bacterial growth and regrowth potential (Chen, 2018, Vital, 2010, Weinrich, 2016), the biostability criteria is 50-100  $\mu\text{g/L}$  (Liu, 2002) and increased differential pressure was especially associated with 50  $\mu\text{g/L}$  AOC in RO systems (Weinrich, 2016). The acceptable concentration of AOC under the condition of minimized chlorine residual level of 0.05 mg  $\text{Cl}_2/\text{L}$  in drinking water is 10.9  $\mu\text{g/L}$  (Ohkouchi, 2013). Therefore, it is useful to know whether POU systems can effectively treat AOC to be in the concentration range of 10.9 to 50  $\mu\text{g/L}$ . AOC is associated with biofouling (L. A. Weinrich, 2015), bacterial growth on water filters, and drinking water quality declination (Li et al., 2018; W. Liu et al., 2002b; Pick et al., 2019). AOC can also be used to calculate microbial regrowth capacity to evaluate how fast microbes grow back after disinfection and as an indicator of water quality stability (J. C. Lou et al., 2012; Ohkouchi, Ly, Ishikawa, Kawano, & Itoh, 2013). Therefore, removal of AOC may be evaluated to investigate how microbial growth potential may be affected in POU systems.

### 3. MATERIALS AND METHODS

#### 3.1 Chemicals and bacteria

Manganese chloride was used to prepare Mn stock solution and uranyl nitrate was used to prepare U stock solution. The oxidation states for Mn and U are +2 and +6, respectively. Sodium acetate was used to represent AOC according to the standard methods (Eaton et al., 1998). The analyte concentrations were 100 µg/L (Mn), 50 µg/L (U), and 100 µg/L (AOC) for low concentration testing, and analyte concentrations were 1,000 µg/L(Mn), 100 µg/L (U), and 200 µg/L (AOC) for medium concentration testing.

ATCC 49642 (*Pseudomonas fluorescense* P17) and ATCC 49643 (*Spirillum* sp. NOX) were purchased from ATCC and grown in agar plates. Vials free of AOC were used to minimize potential contamination of trace organics in glassware.

The concentrations of Mn, U, and AOC spiked in influent water in the feed tank are listed in Table 1. Two levels (low and medium) of metals and AOC were tested to cover the typical concentrations of Mn (Barbeau, Carriere, & Bouchard, 2011; Roitz, Flegal, & Bruland, 2002; Sly, Hodgkinson, & Arunpairojana, 1990), U (Banning & Benfer, 2017; Berisha & Goessler, 2013; Bigalke et al., 2018; Bjorklund et al., 2020; Jakhu, Mehra, & Mittal, 2016), and AOC (W. Liu et al., 2002a; J. C. Lou et al., 2012; Ohkouchi et al., 2013; D. van der Kooij, 1990) in drinking water. Only Mn and U were spiked into tap water in this study. Other metals, including calcium (Ca), magnesium (Mg), iron (Fe), and copper (Cu), were also monitored due to their wide occurrence in the tap water.

**Table 1.** Concentrations of Mn, U, and AOC spiked in influent.

Contaminant	Low influent concentration (µg/L)	Medium influent concentration (µg/L)
Mn	10 <sup>2</sup>	10 <sup>3</sup>
U	0.5×10 <sup>2</sup>	10 <sup>2</sup>
AOC	10 <sup>2</sup>	2×10 <sup>2</sup>



### 3.2 Analysis of heavy metals

Concentrations of Ca, Mg, Fe, Cu, and Mn were quantified with inductively coupled plasma-optical emission spectrometry (ICP-OES). U was first quantified by ICP-OES, and some of the RO permeates with a concentration lower than the detection limit of ICE-OES were analyzed by inductively coupled plasma mass spectrometry (ICP-MS).

According to the Drinking Water Health Advisory for Manganese proposed by the United States Environmental Protection Agency (U.S. EPA), many analytical methods and instruments are available to analyze Mn (U.S. EPA, 2004). EPA Standard Method 200.7 with ICP-OES was used to determine Mn concentration (U.S. EPA, 1994). Briefly, water samples were filtered through 0.45  $\mu\text{m}$  pore diameter membrane filters as soon as possible. The first 50-100 mL filtered sample was used to wash the flask and discarded.

Before and during analysis, ICP-OES was calibrated for quality control. ICP-OES iCAP7400 Duo (Thermo Scientific, Germany) was used for metals quantification and analysis for this study and consists of five major components: plasma torch and sample introduction parts, radiofrequency power generator, echelle polychromator optical system, CID detector with thermoelectric cooling, and interlock. According to EPA Method 200.7, recommended wavelength for Mn is 257.610 nm. As mentioned above, U is not included in EPA method 200.7, so there is no recommended wavelength for U. U has several spectral lines between 200-450 nm (Santos et al., 2010), and a wavelength of 385.96 nm with high sensitivity and low interferences was selected for ICP-OEC analysis in our study. Argon (99.995%) and UHP liquid nitrogen gases were purchased from Indiana Oxygen, USA. All ICP-OES standard solutions have one single element. The Limits of Detection (LOD) and Limits of Quantitation (LOQ) were established at the beginning of the project and shown in Table 2.

**Table 2.** Method parameters for metals determination with ICP-OES

Element		Plasma view	Wavelength (nm)	Echelle Order	LOD ( $\mu\text{g/L}$ )	LOQ ( $\mu\text{g/L}$ )
Ca	Calcium	Radial	184.006	483	16.3	54.2
Ca	Calcium	Radial	393.366	86	1.7	5.6
Ca	Calcium	Radial	422.673	80	59	197
Cu	Copper	Axial	224.700	450	1.9	6.5
Fe	Iron	Axial	259.940	130	0.5	1.7
Mg	Magnesium	Radial	285.213	118	0.2	0.6
Mg	Magnesium	Radial	279.553	120	0.5	1.8
Mn	Manganese	Axial	257.610	131	0.3	1.0
U	Uranium	Axial	385.958	87	21.7	72.4

The instrument operating parameters were the same for the radial and axial plasma views (**Error! Not a valid bookmark self-reference.**). Samples were analyzed without dilution after acidification with concentrated HNO<sub>3</sub> (Trace Metal Grade, Fisher Scientific) to 2% nitric acid.

**Table 3.** Instrument (ICP-OES) operating parameters

Parameter	Settings
RF Power, Watt	1150
Cooling Gas Flow Rate, L/min	12
Auxiliary Gas Flow Rate, L/min	0.5
Nebulizer Gas Flow Rate, L/min	0.5
Exposure Time	15 s (UV), 5 s (visible)
Spray Chamber	Cyclonic, Quartz
Nebulizer	Concentric

Since the LOD/LOQ for uranium was unacceptably high with ICP-OES, a more sensitive analytical method was needed to detect this element in the samples. The instrument ELEMENT2 High Resolution Inductively Coupled Plasma Mass Spectrometer (Thermo Fisher Scientific, Bremen, Germany) was used in Purdue University iLab. An Aridus II Desolvating Sample Introduction system with a 100 uL/min PFA low flow concentric nebulizer (Teledyne Cetac Technologies, 14306 Industrial Rd, Omaha, NE 68144) was used to introduce the samples into the ICP-MS. The type of autosampler used for this purpose is the Teledyne Cetac Autosampler ASX-112FR. A 1 µg/L Thermo Fisher Tune-Up solution was used for tuning first before the samples were analyzed for m/z of U238 using Medium Resolution mode since this will separate the U238 from interference peaks. The LOD and LOQ of this method are below 1 ppt (< 1 pg/mL).

The results of typical metal concentrations in tap water used in this experiment are shown in Table 4 and metal concentrations in groundwater and ultra-pure water are also listed for comparison. When a metal concentration was below the detection limit, half of the detection limit was used.

**Table 4.** Metal concentrations in tap water, groundwater, and ultrapure water.

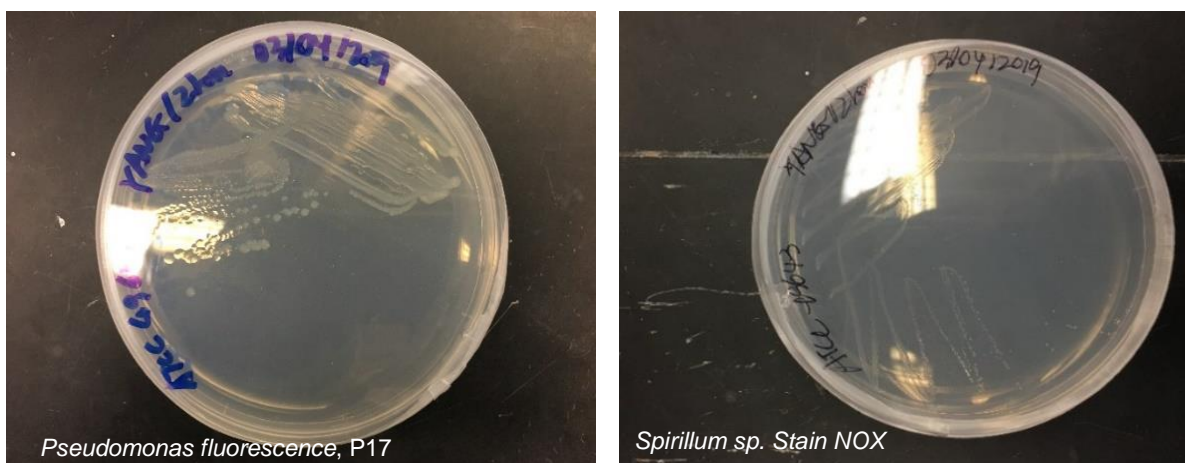
Metal	Tap water (µg/L)	Groundwater (µg/L)	Ultra-pure water (µg/L)	Detection method
Ca	94,220.6	64,774.9	<DL	ICP-OES
Cu	52.2	<DL	<DL	ICP-OES
Mg	34,419.8	32,963.3	2.3	ICP-OES
Mn	818.7	246.0	0.4	ICP-OES
U*	0.94	1.27	0.0002	ICP-MS

\*: Uranium analyzed with ICP-MS has a lower detection limit than those measured with ICP-OES  
DL: Detection limit. Cu DL: 0.32 µg/L, Ca DL: 11.21 µg/L

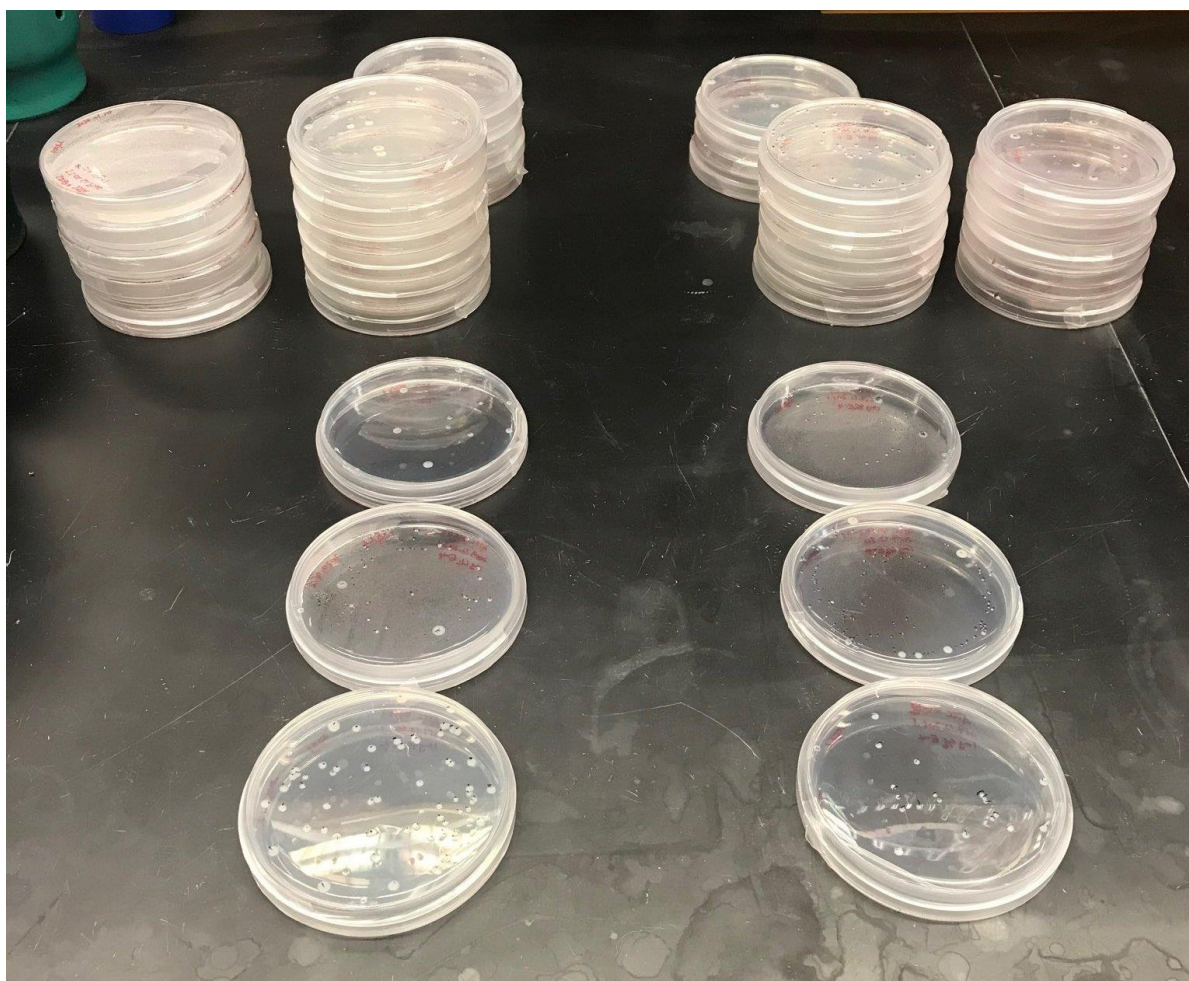
### 3.3 Analysis of AOC

The method chosen to follow for AOC analysis was from Standard Methods for the Examination of Water and Wastewater edited (Baird, Bridgewater, American Public Health Association, American Water Works Association, & Water Environment Federation, 2017). We use the two carbons in sodium acetate to represent AOC for the experiment. This conventional AOC measurement is conducted with pure bacterial cultures (*Pseudomonas fluorescense* P17 and *Spirillum* sp. NOX) (Clesceri, Greenberg, & Eaton, 1998; D. van der Kooij, Visser, & Oranje, 1982). The detection limit for this method is 5 to 10 µg AOC/L as stated in the method due to possible organic carbon contamination, though in theory any concentration below 1 µg C/L can be detected. The conventional AOC measurement method includes the following steps.

All glassware must be carbon-free. Briefly, containers and caps were washed with tap water and rinsed three times with ultrapure water. Then glassware was soaked in 10% HNO<sub>3</sub> for 4 hours. After that, glassware was rinsed with ultrapure water three times and heated at 550°C for 6 hours to oxidize any carbon residuals. Bacteria were recovered from the frozen condition through streaking on an R2A agar plate and incubated at room temperature (20-25 °C) for 3-5 days. A single colony was separated from the plate and inoculated into autoclaved organic carbon limited water. Each sample was transferred into ten 50mL vials and had 100 µl mineral salt solution added into each vial before autoclaving. Then 10 µL of each P17 and NOX bacterial suspension was transferred to every 50 ml autoclaved sample. The mixture was incubated at room temperature until it reached a stationary phase for 7 days. Collected water samples were collected into different vials as soon as possible and pasteurized in flasks at 70 °C for 40 min within 6 hours after sampling to kill any vegetative cells. After the water was cooled down to room temperature, P-17 and NOX were mixed into the water sample and incubated at 15 °C in dark for a week. On the 7th, 8th, and 9th days of incubation, the sample was shaken for 1 min before testing and the 1 ml sample was transferred with a sterile pipet in a diluted solution. Three plates were used for enumeration. Plates were incubated for 3-5 days and the number of colony-forming units (CFUs) on each plate were counted. Sample vials on three separate days were checked to evaluate if maximum density was reached. A consistent 20% increase indicates that the culture was not reaching a stationary phase. To minimize inhibition of P17 and NOX from residual chlorine in the tap water, a carbon pre-filter was used before emerging contaminants were mixed in the feed tank. Some examples of the colonies and agar plates used in AOC testing are shown in Figure 1 and Figure 2.



**Figure 1.** Bacterial colonies of *Pseudomonas fluorescence*, P17 and *Spirillum* sp. Stain NOX

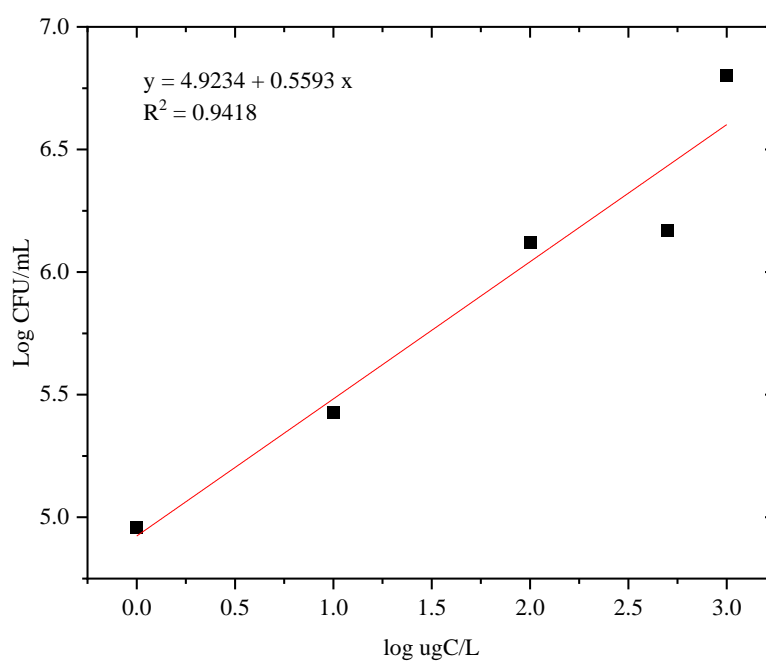


**Figure 2.** Agar plates used in AOC testing.

AOC concentration was calculated by using the empirical formula, as previously described (Baird et al., 2017).

$$\mu g \frac{AOC}{L} = \left[ \left( mean P_{17 \frac{CFU}{mL}} \right) * \frac{1 \mu g acetate_C}{4.1 * 10^6 CFU} + \left( mean NOX \frac{CFU}{mL} \right) \left( \frac{1 \mu g acetate_C}{1.2 * 10^7 CFU} \right) \right] * (1000 ml/L)$$

The correlation between acetate as the only AOC carbon source and microbial growth was tested. A positive correlation was observed (Figure 3) and AOC concentration as low as 1  $\mu g$  C/L was detected.



**Figure 3.** Correlation between AOC and CFU.

### 3.4 Analysis of TDS

Total dissolved solid (TDS) was tested in medium concentration samples to see the overall performance of contaminant removal under high influent concentrations of metals and AOC. The method followed the Standard Methods for the Examination of Water and Wastewater (Baird et al., 2017). Dry beakers were pre-weighed ( $W_1$ ) using an analytical balance with an accuracy of 0.1 mg (Mettler Toledo) before adding in 50 mL ( $V$ ) of well-mixed samples filtered through glass-fiber filter disks with 0.45 mm pore size with filtration apparatus. The volume of influent filtered should have about 500 mg of TDS. The beaker is then kept in the oven to dry under 110 °C (5 hours-overnight). The beakers are then put into a desiccator to cool for 20 minutes before weighing the final mass with the dry mass inside ( $W_2$ ).

The TDS (mg/L) was calculated as:

$$TDS\left(\frac{mg}{L}\right) = \frac{(W_1 (mg) - W_2 (mg)) * 1000}{V (mL)}$$

### 3.5 Configuration of POU systems

Tap water was used for emerging contaminant removal in this project. The actual water quality of tap water and groundwater is shown in Table 5.

**Table 5.** Water quality of tap water and groundwater.

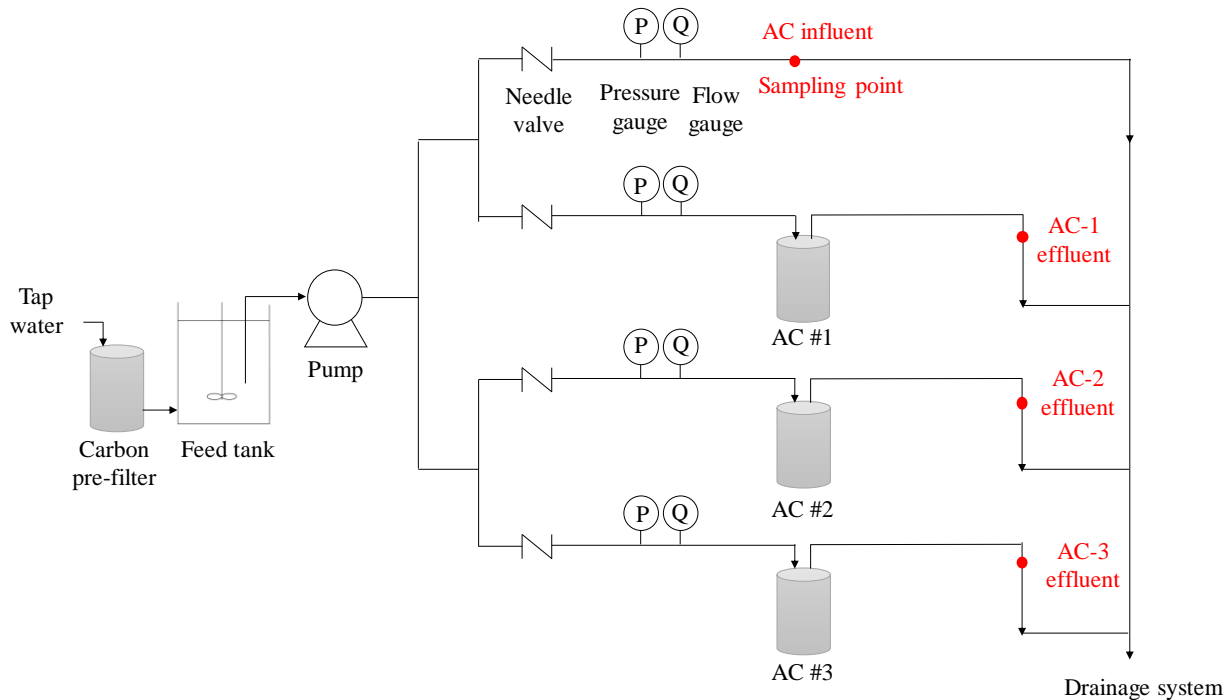
Parameters	Concentration in tap water	Concentration in groundwater
Conductivity	756.67 $\mu\text{s}/\text{cm}$	826.33 $\mu\text{s}/\text{cm}$
Dissolved oxygen (DO)	2.13 mg/L	5.90 mg/L
pH (7.5+/- 0.5)	7.56	8.39
temperature	22.37 $^{\circ}\text{C}$	21.80 $^{\circ}\text{C}$
Total dissolved solid (TDS)	491.83 mg/L	537.12 mg/L
Total organic carbon	0.83 mg/L	1.17 mg/L
Free available chlorine	0.11 mg/L	0.01 mg/L

The specifications of the three AC systems tested in this study are listed in Table 6. Since filtration capacities of AC systems are recommended by their manufacturers and are all different, comparison experiments were conducted with relatively capacity of their designed treatment capacity.

**Table 6.** Specifications of AC systems tested in this study.

System	Flow rate (gpm)	100% capacity (gallons)	Micron rating	Pressure (psi)	Certification	Filtration method
AC-1	0.5	500	0.5	30-100	NSF 42, 53, P473	Carbon block
AC-2	0.5	300	0.5	10-125	NSF 42, 53	Granular activated carbon with copper-zinc granules
AC-3	0.5	200	NA	NA	NSF 42, 53, 401, P473	Carbon block

These AC filtration systems were operated in the dead-end mode. The configuration of AC systems is shown in Figure 4.



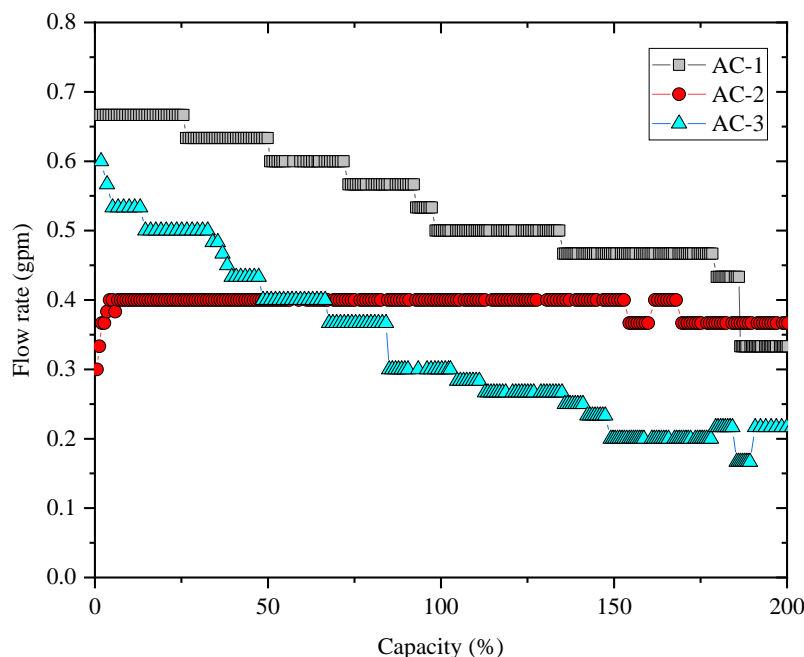
**Figure 4.** Configuration of AC systems.

Sample collection and the testing protocol was based on the industry-standard ANSI/NSF standard 53 “Drinking water treatment unit- Health effects”.

The systems were operated for 16 hours a day in a 6 min on / 54 min off cycle at the manufacture-rated maximum flow rate of the device. Samples were collected at the middle of the 6-minute on cycle. Samples were collected at 0%, 50%, 100%, 150%, 180%, and 200% capacity of systems. No performance indicator devices were included in any of the tested AC.

During one of the contaminant removal tests, the flow rate of AC-1 was observed to be 0.67 GPM and then slowly decreased to 0.33 GPM after it reached its 200% capacity. The flow rate of AC-2 started and maintained at 0.4 GPM during the majority of the time and decreased and kept at 0.37 GPM after it exceeded its 180% capacity. The flow rate of AC-3 decreased steadily from 0.6 GPM to 0.22 GPM. The reduction of flow rates of both AC-1 and AC-3 could be attributed to slow clogging and reduced porosity of carbon block AC filters. On the other hand, the relatively uniform flow rates in AC-2 may be related to its special configuration of granular activated carbon and copper-zinc granules that allows for tap water to flow evenly in the AC system. The flow rates of the AC systems are shown in Figure 5.





**Figure 5.** Variation of flow rates of AC systems under 50 PSI.

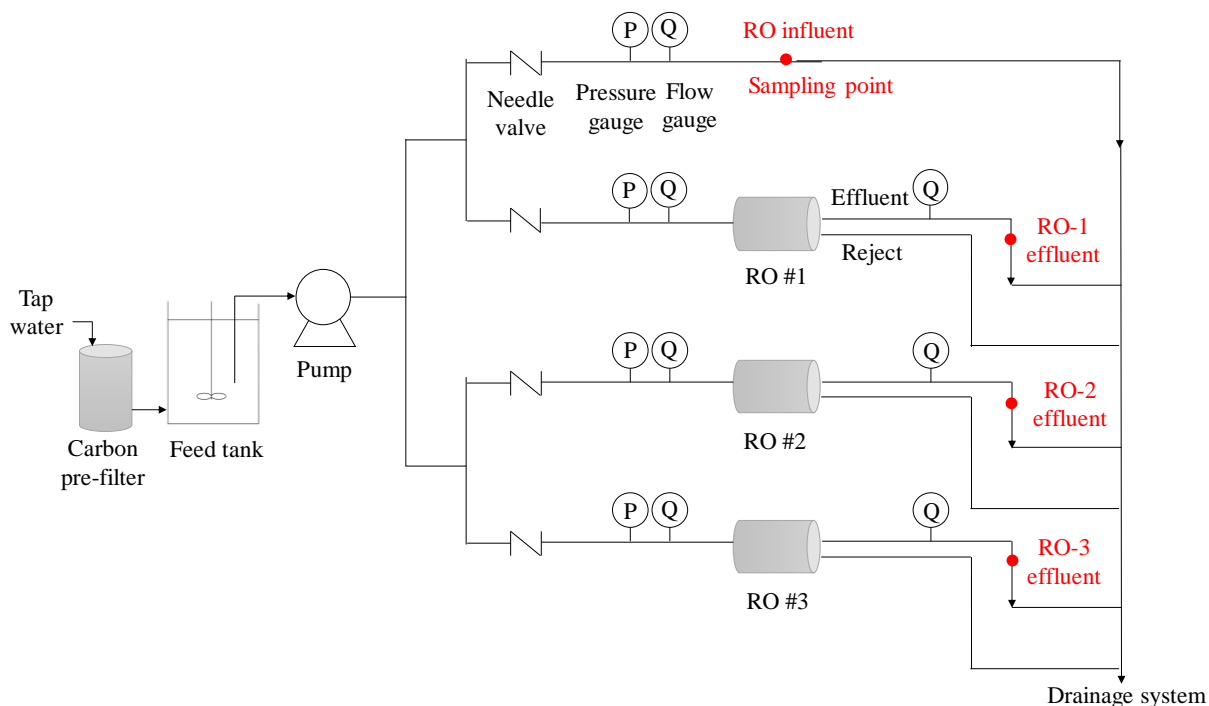
Typically, the RO filter system includes four stages: sediment prefilter, carbon prefilter, RO member, and carbon post filter, and only the RO membrane was tested in this study. RO products with the same flow rate (50 GPD) were selected to reduce the potential impact of variable flow rate on product performance. No performance indication device was included in any of the products that were tested.

Specifications of the three RO units tested in this study are shown in Table 7. Several National Sanitation Foundation (NSF) / American National Standards Institute (ANSI) standards have been developed to certify POU systems for contaminant removal (International, 2021). NSF/ANSI 53 is used to certify POU or POE systems to reduce a contaminant with a health effect. Recovery is the percentage of RO effluent (permeate) water produced from RO influent water when RO effluent water is not flowing into the storage tank.

**Table 7.** Specifications of RO systems tested in this study.

System	Material	Flow rate (GPD)	The daily production rate in NSF 58 certificates (GPD)	Certification	Recovery
RO-1	Thin layer composite	50	15	NSF 58	15-20%
RO-2	Polyamide thin film composite	50	12	NSF 58	16%
RO-3	Thin film composite	50	17	NSF 58	28%

The configuration of RO systems is shown in Figure 6. The RO membrane modules were all housed in separate membrane enclosures and can be separated from complete RO systems. An external 1/4" quick connect flow restrictor (550 mL/min) was used in the concentrate line of each RO system. The performance of three RO systems was compared as they were operated under the same operation condition with the same permeate/reject ratio.



**Figure 6.** Configuration of RO systems.

Sample collection will be based on the industry-standard ANSI/NSF standard 58 “Reverse osmosis drinking water treatment system”. The systems were installed and operated based on NSF testing protocols, flushed and conditioned with tap water. A pre-filter was used to remove residual chlorine to protect the RO membrane. The test was conducted for three weeks.

The systems were tested with the following operating conditions.

- Day 1: The system is operated for 16 hours and the sample collected after 16 hours is analyzed.
- Day 2-4: Sample is collected every 6 hours and analyzed.
- Day 5,6: RO system is shut down.

- Day 7: The system is operated for 4 hours, and a sample is collected at the start point (144 hours) and another sample is collected after 4 hours of operation. Both samples are analyzed.

All removal efficiencies in this paper are calculated by the equation:

$$\text{Removal efficiency} = \left(1 - \frac{\text{effluent concentration}}{\text{influent concentration}}\right) \times 100\%$$

whereas effluent concentration is the concentration of the analyte analyzed from filtered sample water, the influent concentration is the concentration of the analyte from unfiltered influent water collected from the influent water sampling point (Figure 4 and Figure 6) prepared in the tank.

## 4. RESULTS AND DISCUSSION

### 4.1 Removal of low concentration metals in AC systems

The results of the AC systems operated up to their 200% designed treatment capacities (AC-1: 1,000 gallons, AC-2: 600 gallons, AC-3: 400 gallons) showed that Ca (Figure 7) was not effectively removed and its average ( $\pm$  standard error) removal efficiencies were  $0.06\% \pm 0.62\%$  (AC-1),  $0.13\% \pm 0.88\%$  (AC-2), and  $0.45\% \pm 0.27\%$  (AC-3). Mg was not effectively removed either (Figure 8) and its removal efficiencies were average ( $\pm$  standard error) removal efficiencies were  $-0.01\% \pm 0.59\%$  (AC-1),  $-0.64\% \pm 0.94\%$  (AC-2), and  $0.51\% \pm 0.23\%$  (AC-3). The low removal efficiencies of Ca and Mg, which are the main hardness ions in tap water, in AC systems in this study are consistent with the common knowledge that hardness is not effectively removed by AC systems. A water softener is often needed to remove Ca and Mg from tap water.

Significant fractions of Fe were removed in the AC systems (Figure 9) with average removal efficiencies of  $63.91\% \pm 5.92\%$  (AC-1),  $77.08\% \pm 2.41\%$  (AC-2), and  $50.57\% \pm 5.00\%$  (AC-3). Fe is a common element detected in groundwater and tap water and the results showed that a major fraction of Fe remained in tap water after filtration. A water softener may be also used to effectively remove Fe in tap water.

Cu was effectively removed in the AC systems (Figure 10) with average removal efficiency of  $98.70\% \pm 0.09\%$  (AC-1),  $98.83\% \pm 0.05\%$  (AC-2), and  $96.71\% \pm 2.08\%$  (AC-3). Cu is often detected in drinking water as copper pipes are commonly used as plumbing material and may leach into tap water. The relatively high removal efficiency of Cu may be related to its low influent concentration ( $12.47 - 13.69 \mu\text{g/L}$ ), which was much lower than the influent concentrations of Ca ( $8.95 \times 10^4 - 9.05 \times 10^4 \mu\text{g/L}$ ), Mg ( $3.05 \times 10^4 - 3.09 \times 10^4 \mu\text{g/L}$ ), and Fe ( $89.3 - 362.9 \mu\text{g/L}$ ) were much higher, and any adsorption of trace amount metal may be reflected as high removal percentage.

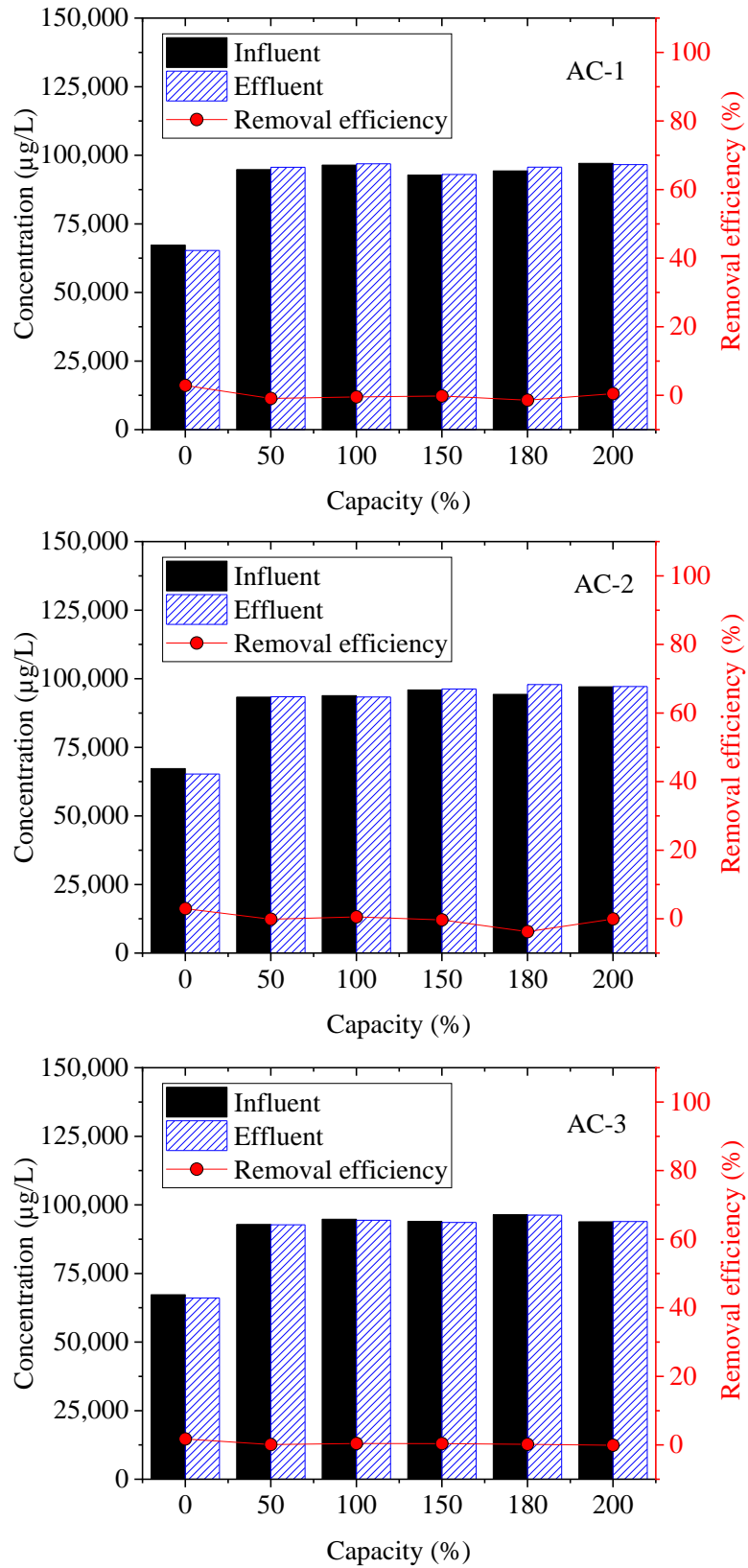
Mn was not effectively removed in all the AC systems (Figure 11) and its removal efficiencies were all below 30.54% on the first day of sample collection and then rapidly dropped with increased filtration time. The reduction of Mn removal efficiencies in AC systems was reported in a previous study in the literature. AC filtering performance on Mn was previously studied on GAC pour-through filters and solid block activated carbon and the results showed that

initial removal efficiencies of 60-100% decreased to 48-64% in the pour-through filters and less than 50% for solid block AC when the filters reached their 200% filtration capacities (Carrière, 2011). The background pH was 8.4 and influent Mn concentrations ranged between 81 µg/L and 916 µg/L (Carrière, 2011). In our study, the background pH was 8.8 and influent Mn concentrations ranged between 146.3 µg/L and 213.0 µg/L. In both cases, AC POU systems did not effectively remove Mn.

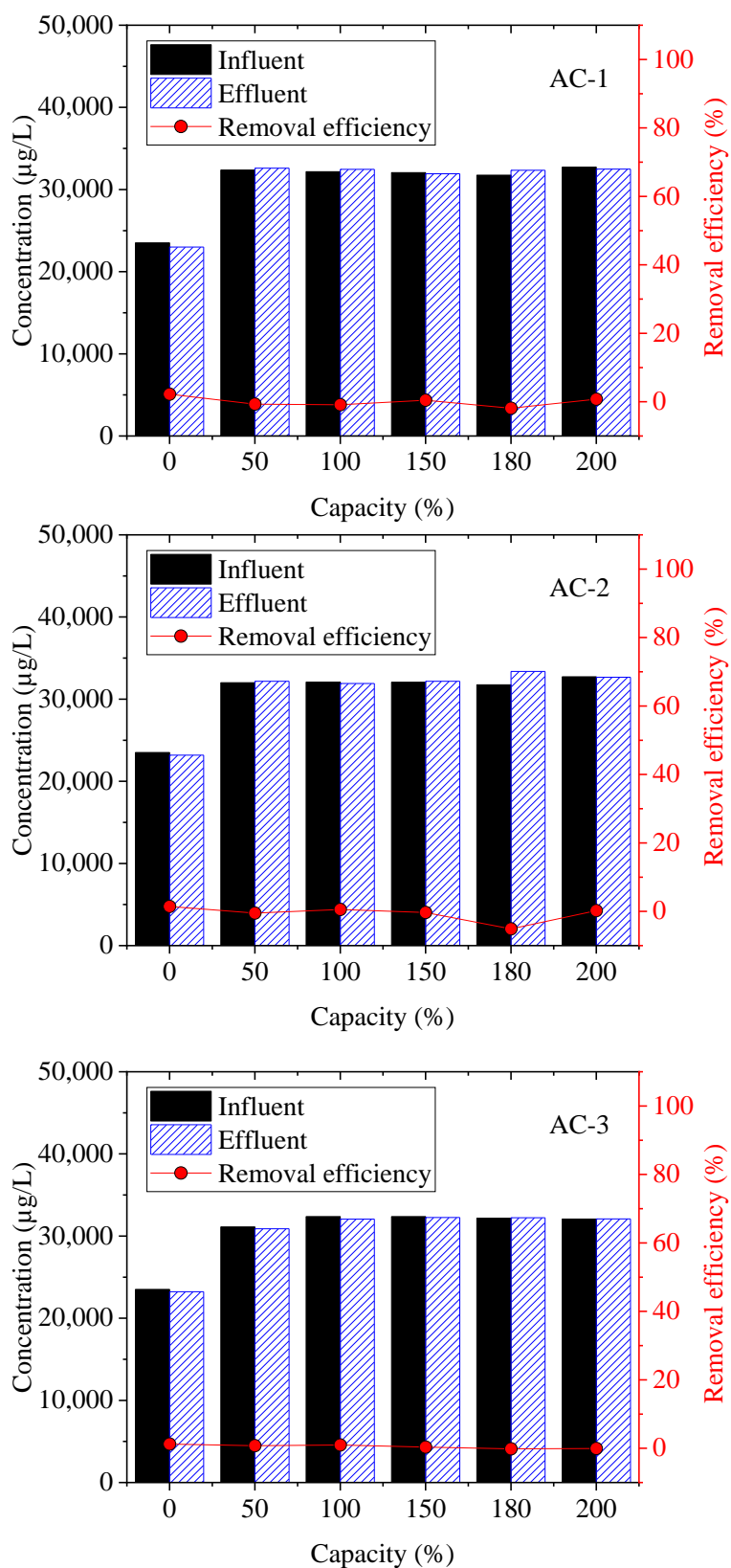
The observed negative removal efficiencies of Mn were unexpected, but increased Mn levels in drinking water systems has been previously reported. Accumulated scale, sediment, or biofilm in drinking water distribution systems may release Mn back to drinking water under different hydraulic and pressure conditions (Sly et al., 1990). In another study, a peak in total Mn and Fe was observed during tap flushing (Barbeau et al., 2011). In our study, AC systems were operated for 6 min after 54 min of stagnation and the transient pressure and hydraulic may have contributed to the observed high levels in AC effluents.

U was not effectively removed in AC systems (Figure 12) and its removal efficiencies were all below 22.97 (AC-1), 63.3 % (AC-2), and 0% on the first day and readily dropped with increased filtration time. Negative removal efficiencies and high removal efficiency variability were also observed, which may be also attributed to the release of temporarily adsorbed U during short filtration time (6 min) after 54 min of stagnation in AC systems.

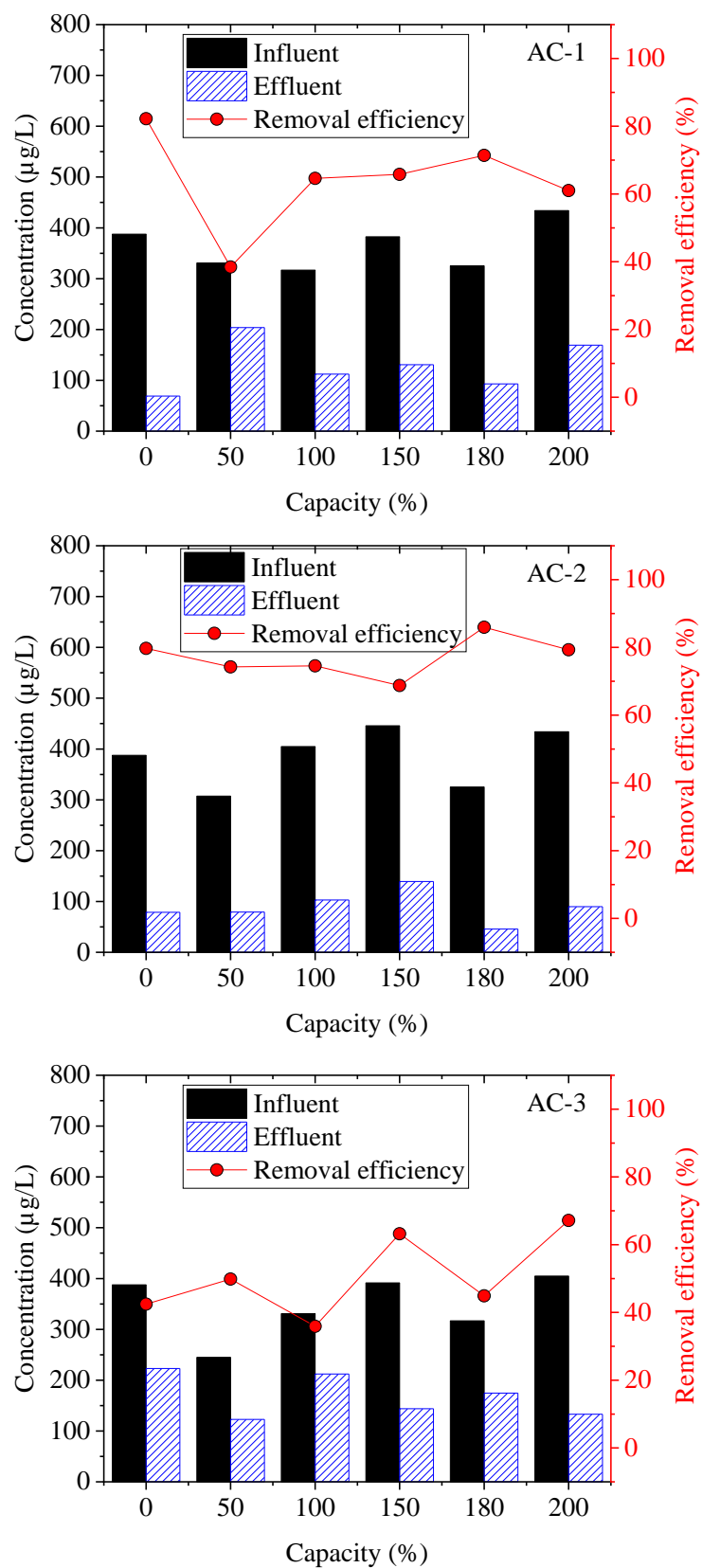
Overall, these results indicated that AC systems were generally ineffective to remove metals from drinking water.



**Figure 7.** Removal of low concentration Ca in AC systems.

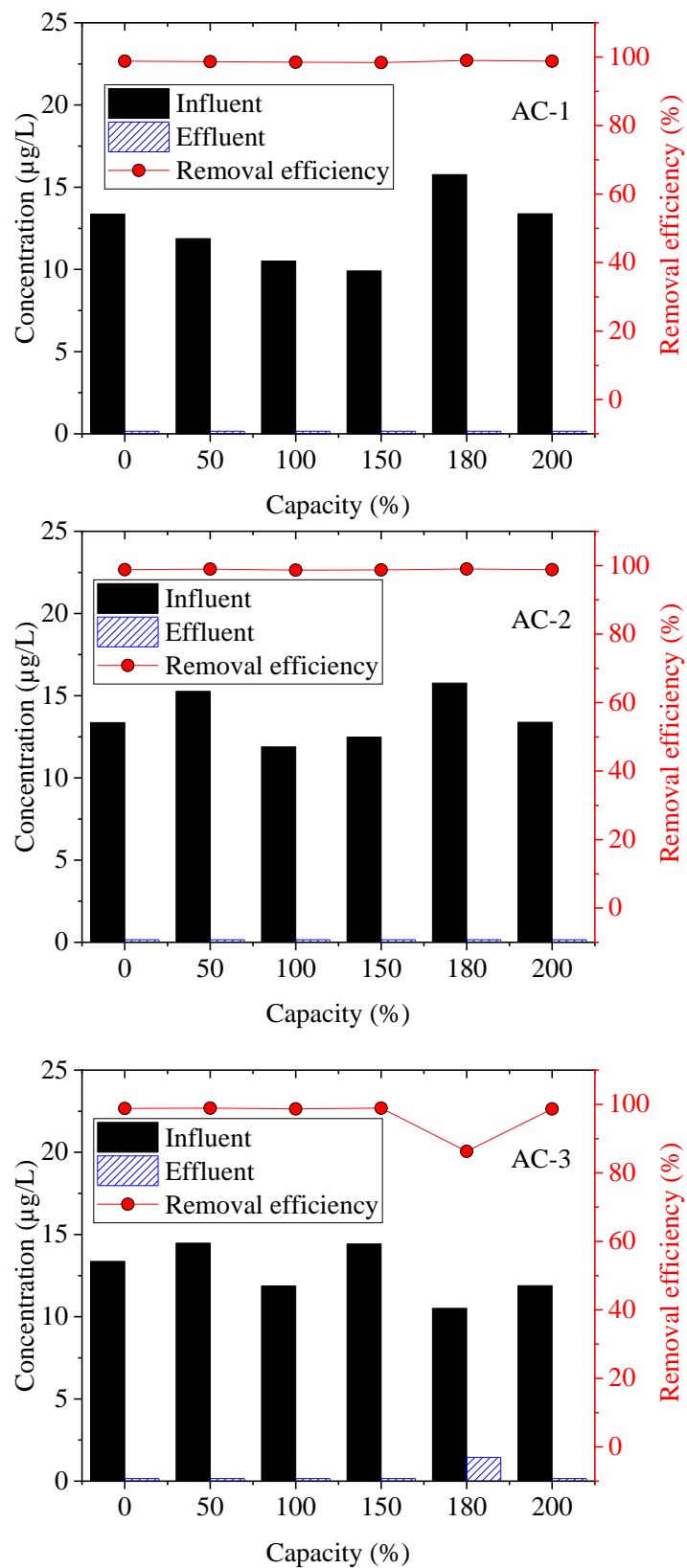


**Figure 8.** Removal of low concentration Mg in AC systems.

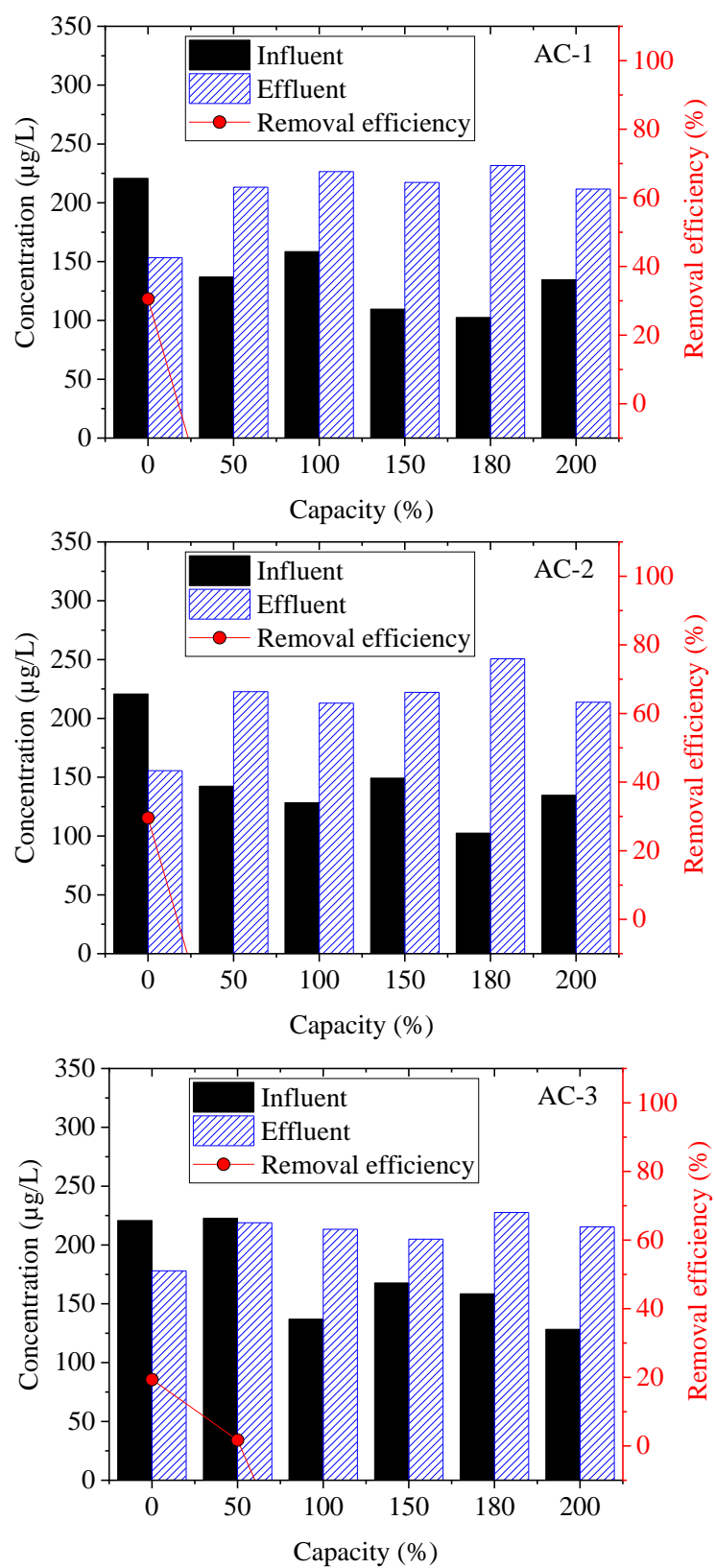


**Figure 9.** Removal of low concentration Fe in AC systems.

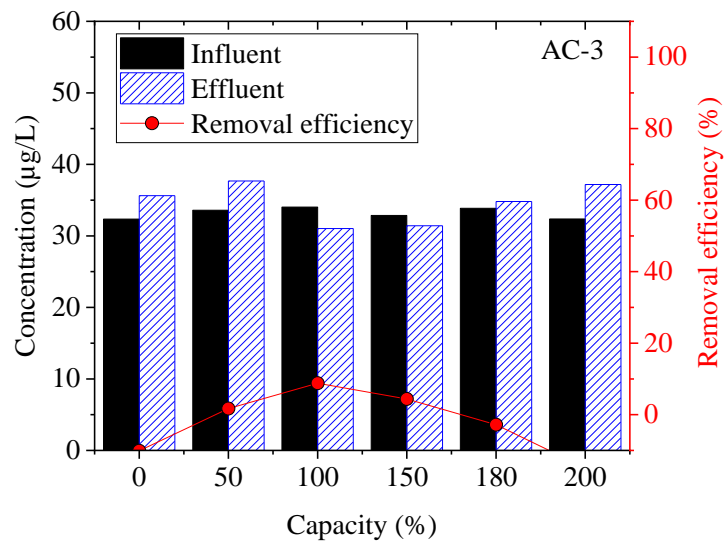
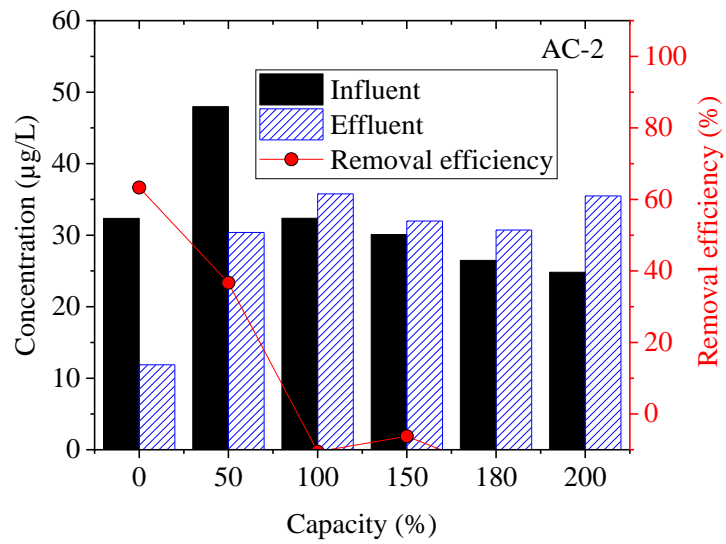
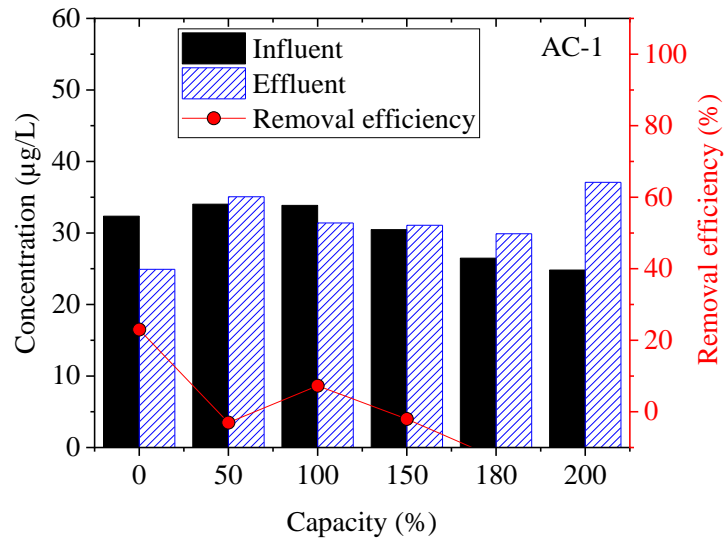




**Figure 10.** Removal of low concentration Cu in AC systems.



**Figure 11.** Removal of low concentration Mn in AC systems.



**Figure 12.** Removal of low concentration U in AC systems.

## 4.2 Removal of low concentration metals in RO systems

After three weeks of filtration, the average removal efficiencies of Ca (Figure 13. Removal of low concentration Ca in RO systems.), Mg (

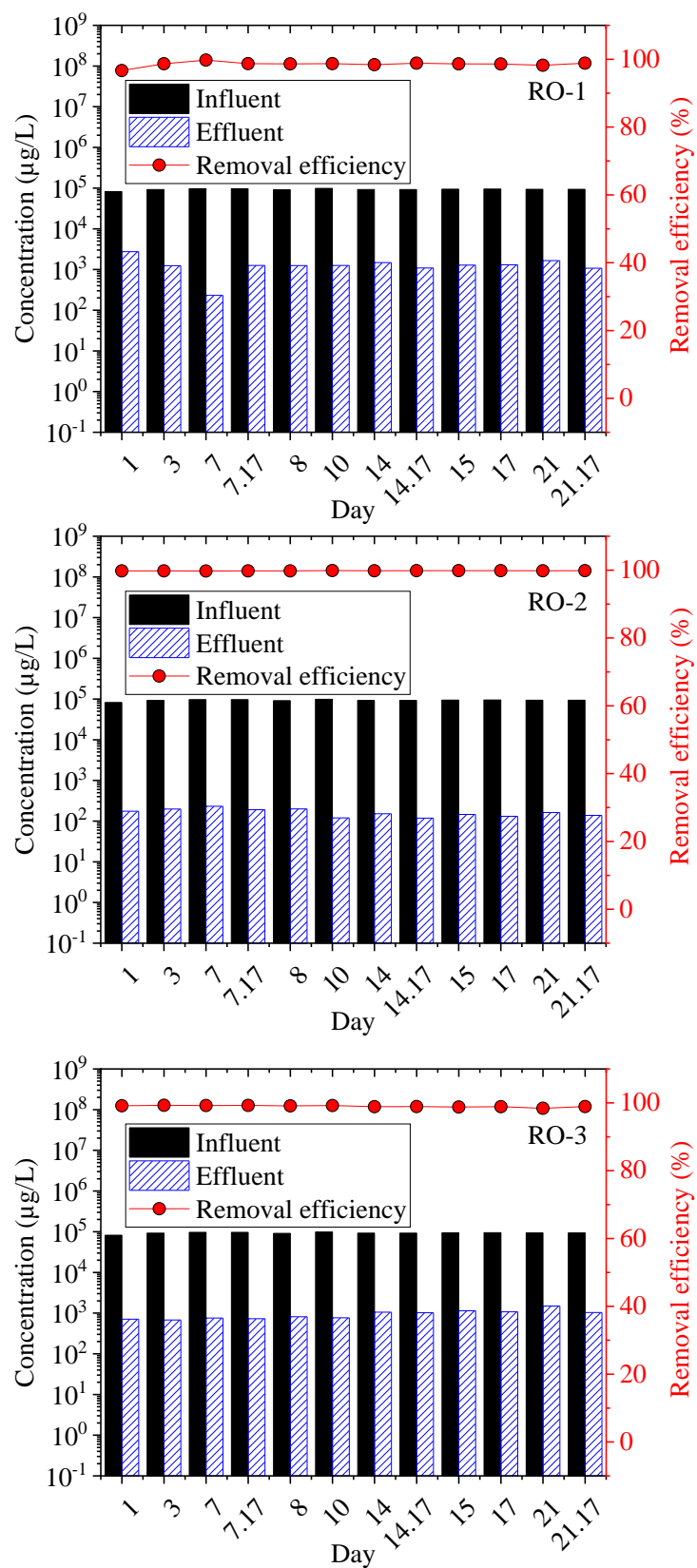
Figure 14. Removal of low concentration Mg in RO systems.), Fe (Figure 15), Cu (Figure 16), Mn (Figure 17), and U (Figure 18).

The removal efficiencies of Ca were  $98.56\% \pm 0.20\%$  (RO-1),  $99.82\% \pm 0.01\%$  (RO-2), and  $98.99\% \pm 0.07\%$  (RO-3). The removal efficiencies of Mg were  $98.65\% \pm 0.18\%$  (RO-1),  $99.83\% \pm 0.01\%$  (RO-2), and  $99.11\% \pm 0.05\%$  (RO-3). The removal efficiencies of Fe were  $99.47\% \pm 0.11\%$  (RO-1),  $99.70\% \pm 0.06\%$  (RO-2), and  $99.70\% \pm 0.05\%$  (RO-3). The removal efficiencies of Cu were  $99.19\% \pm 0.04\%$  (RO-1),  $99.19\% \pm 0.04\%$  (RO-2), and  $98.84\% \pm 0.32\%$  (RO-3).

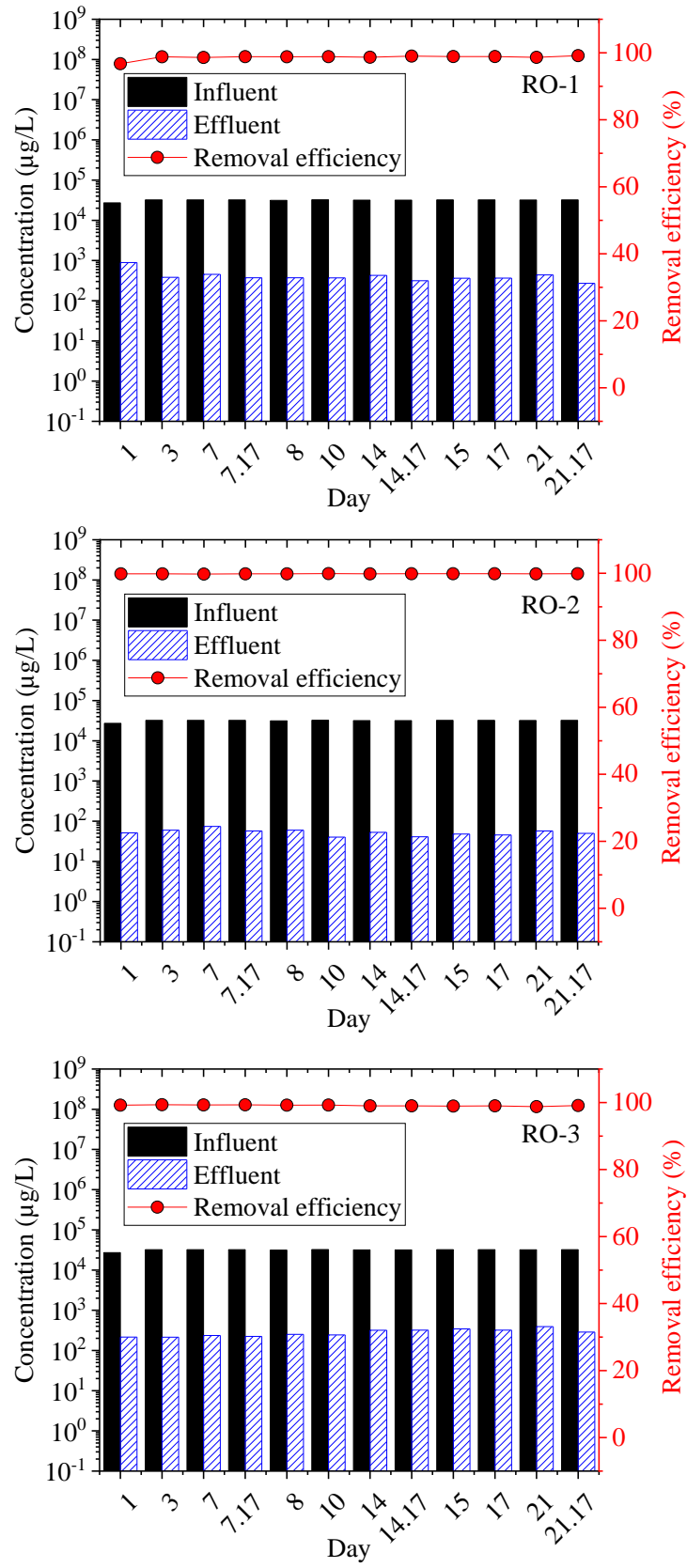
The removal efficiencies of Mn and U were relatively low compared with removal efficiencies of Ca, Mg, Fe, and Cu, but their typical removal efficiencies of Mn and U were still above 95% and 97%, respectively. The removal efficiencies of Mn were  $95.01\% \pm 1.45\%$  (RO-1),  $95.02\% \pm 1.44\%$  (RO-2), and  $95.09\% \pm 1.36\%$  (RO-3). The removal efficiencies of U were  $97.76\% \pm 0.43\%$  (RO-1),  $99.84\% \pm 0.03\%$  (RO-2), and  $99.21\% \pm 0.14\%$  (RO-3).

These results indicated that RO systems were generally effective to remove low concentration metals from tap water. The results were consistent with previous results. A previous study done on U removal with RO used removal of gross alpha and beta activity instead of concentration to evaluate membrane performance of over 90% (Figoli, 2016). Their results also showed that pH ranging from 3 to 11 can retain U and strong U adsorption was observed under pH 4 to 7. Our study showed a much lower U removal efficiency with AC systems, but consistent with high removal efficiency in RO systems.

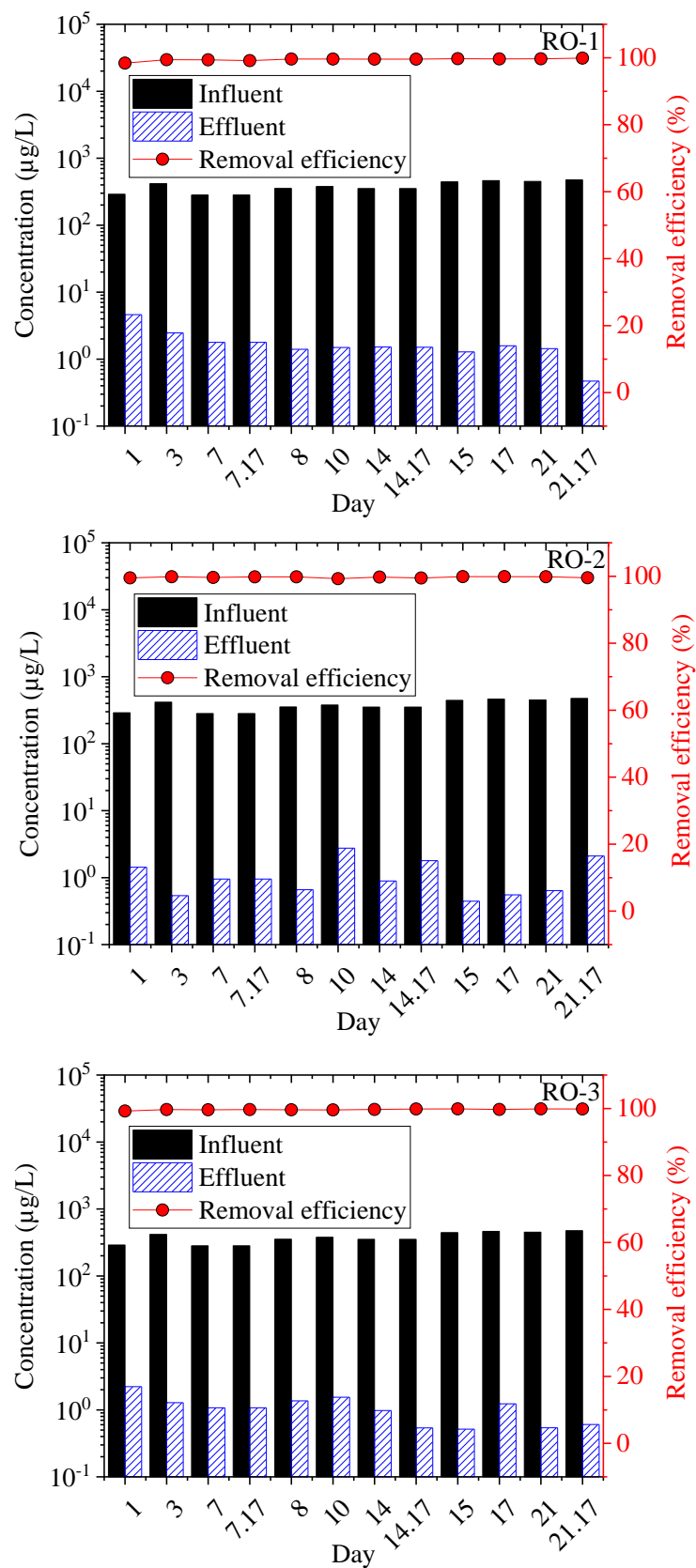
Metal removal efficiencies in RO systems were consistently higher than those in AC systems. As RO membrane can effectively reject monovalent ions, it is not surprising removal efficiencies of Ca, Mg, Fe, Cu, Mn, and U were effectively in RO systems in this study. On the other hand, dissolved metals are hydrophilic and are not effectively removed in hydrophobic carbon surfaces in AC systems.



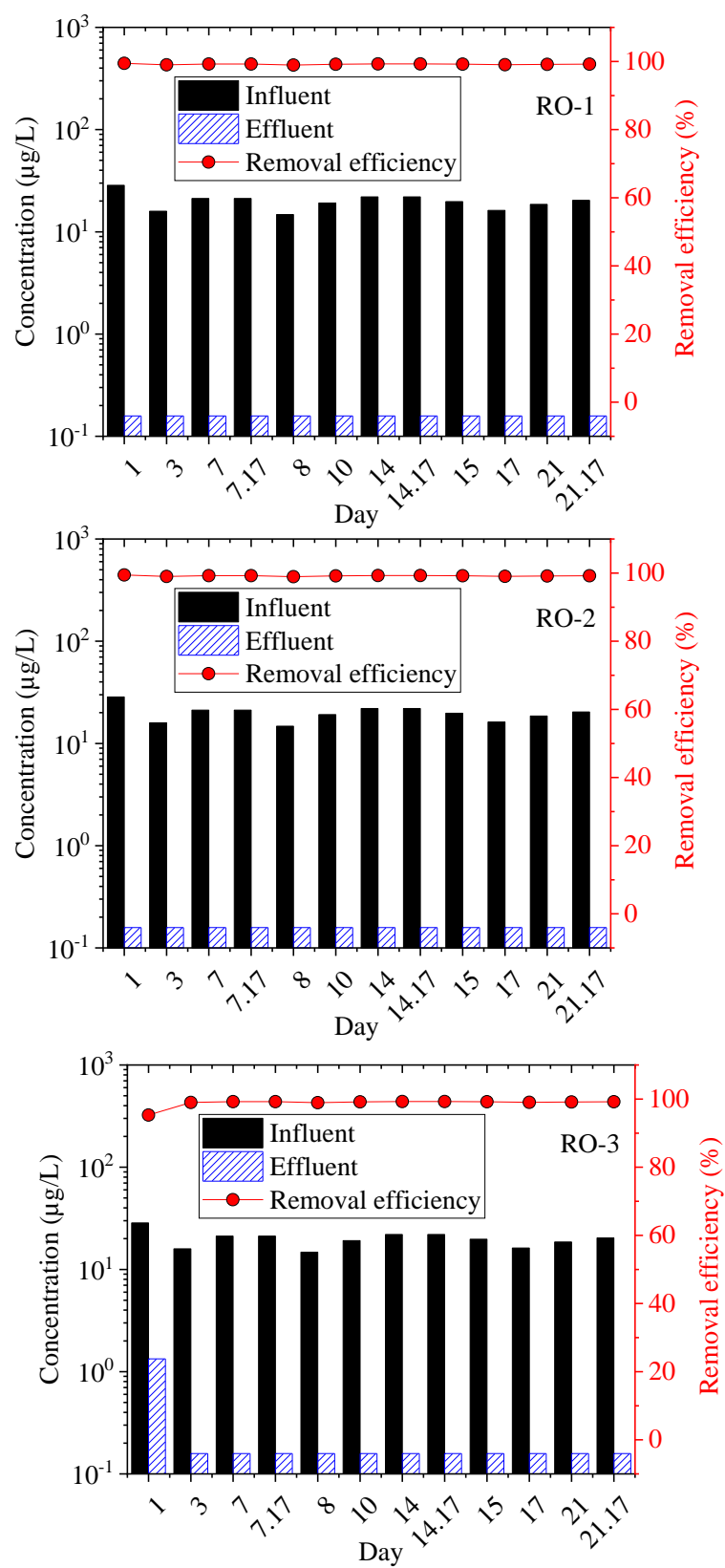
**Figure 13.** Removal of low concentration Ca in RO systems.



**Figure 14.** Removal of low concentration Mg in RO systems.

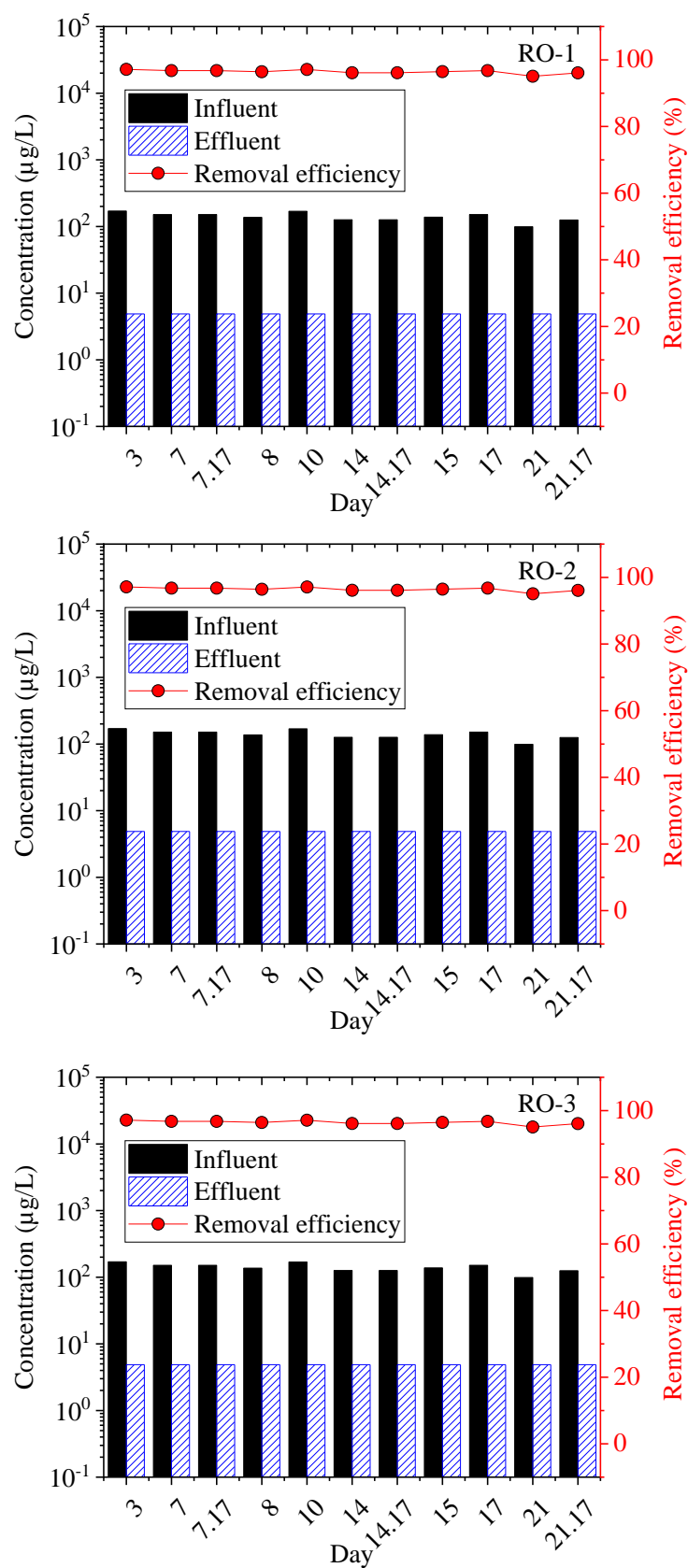


**Figure 15.** Removal of low concentration Fe in RO systems.

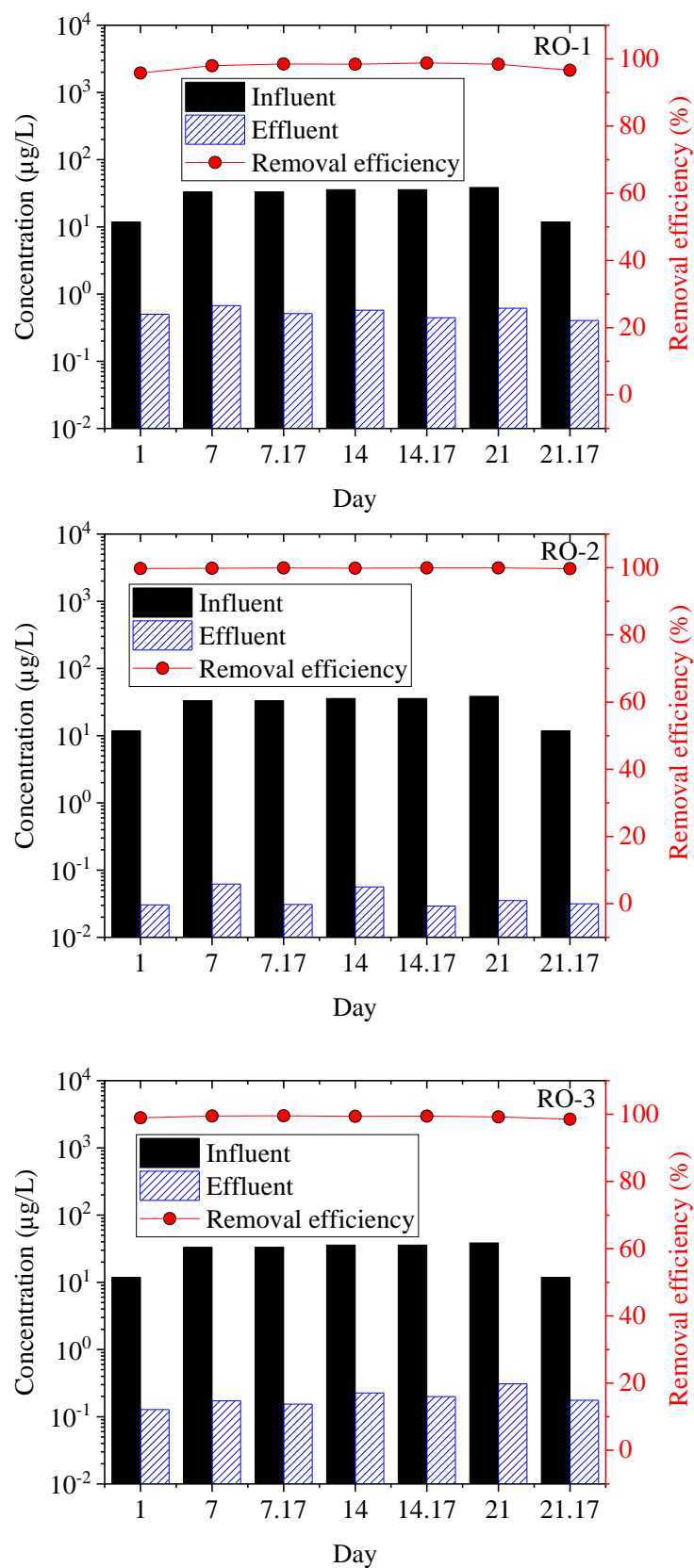


**Figure 16.** Removal of low concentration Cu in RO systems.





**Figure 17.** Removal of low concentration Mn in RO systems.



**Figure 18.** Removal of low concentration U in RO systems.

### 4.3 Removal of low concentration AOC in AC and RO systems

The results of low concentration AOC removal in AC and RO systems are shown in Figure 19.

Low concentration AOC was relatively effectively removed in all AC systems and the average effluent concentrations were 11.30  $\mu\text{g/L}$  (AC-1), 9.30  $\mu\text{g/L}$  (AC-2), and 18.89  $\mu\text{g/L}$  (AC-3), and the average influent concentration was 86.42  $\mu\text{g/L}$ . The average AOC removal efficiencies were  $85.60\% \pm 3.78\%$  (AC-1),  $81.40\% \pm 6.91\%$  (AC-2), and  $78.40\% \pm 6.19\%$  (AC-3), respectively. High variability of AOC was observed in all AC systems, which can be explained by the heterogeneity of biofilm in the AC systems. No clear trend of AOC removal was observed in the AC-1 system and AOC removal efficiencies were relatively high. AOC removal efficiencies in the AC-2 system gradually increased from 0% to 150% capacity and fluctuated. AOC removal efficiencies in the AC-3 system fluctuated initially, then gradually increased after 100% capacity. The observed increased AOC removal may be attributed to the biological removal of AOC in established biofilm in addition to chemical removal of AOC in POU systems.

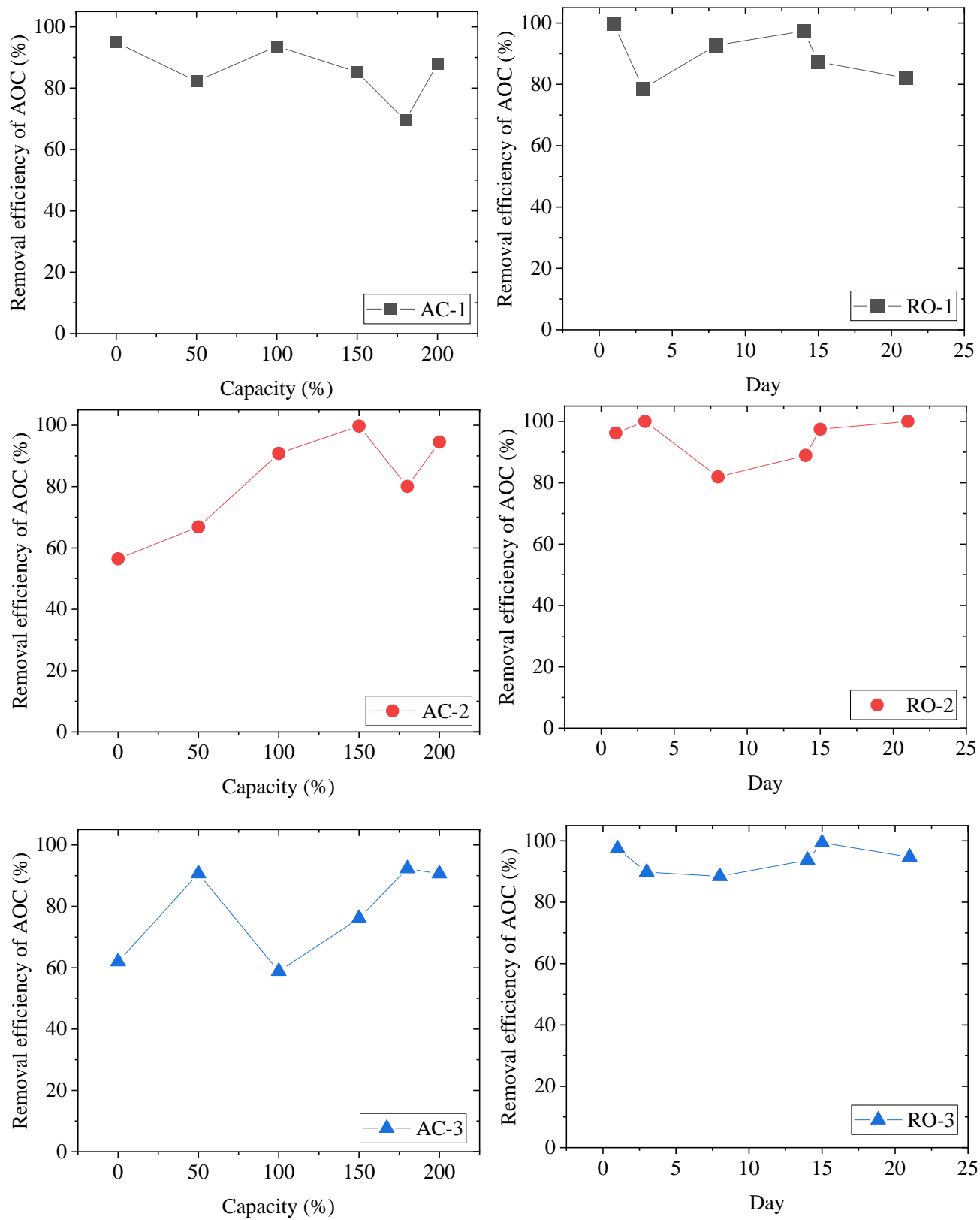
Low concentration AOC was relatively effectively removed in all RO systems and the average effluent concentrations were 11.16  $\mu\text{g/L}$  (RO-1), 6.52  $\mu\text{g/L}$  (RO-2), and 5.97  $\mu\text{g/L}$  (RO-3) with an average influent concentration of 96.76  $\mu\text{g/L}$ . The average AOC removal efficiencies were  $89.64\% \pm 3.44\%$  (RO-1),  $94.09\% \pm 2.94\%$  (RO-2), and  $93.93\% \pm 1.73\%$  (RO-3). Similar to AC systems, high variability of AOC removal was observed in all RO systems, which can also be explained by the heterogeneity of biofilm in the AC systems. No clear trend of AOC removal was observed in RO systems.

The AOC removal results were consistent with previous results in the literature. Previous studies have shown RO removal efficiencies of drinking water spiked with AOC level of 200  $\mu\text{g}$  acetate-C/L was more than 90% at a pH of 7.5 and more than 75% at a pH of 5.5 (Escobar, 2000). As RO membrane is negatively charged at high pH values, acetate can be more efficiently rejected in RO membrane at high pH values as acetate with a pKa of 4.36 is more ionized and negatively charged at high pH values.

Results in the literature also showed that AC was less effective to remove AOC. One study using GAC for AOC removal showed a wider removal efficiency range of 41-65% with an influent concentration of 91-226  $\mu\text{g}$  acetate-C/L (Huang, 2004). Another study showed that removal efficiencies of spiked AOC levels of 10-200  $\mu\text{g}$  acetate-C/L were about 40% to 67% (Lou, 2014).

In this study, the correlation between AOC removal efficiency and POU system performance was not observed, which is likely due to the complexity of POU systems that may be affected by many different factors. For example, commonly detected metals in drinking water, such as Ca and Mg, may contribute to water hardness in drinking water and Fe favors the growth of iron-oxidizing bacteria. A previous study has shown that AOC removal in RO was unlikely to be associated with size exclusion, but more likely to be charge repulsion from water hardness and ionic strength when tested by adding  $\text{CaCl}_2$  (Escobar, 2000). Fluctuation of background hardness may have introduced the variability observed AOC level observed in this study.

Further studies are needed to correlate metal concentration and AOC removal efficiency and the high variability of AOC in POU systems deserves further investigation.



**Figure 19.** Removal of low concentration AOC in AC and RO systems.

#### 4.4 Removal of medium concentration metals in AC systems

The overall trend of medium concentration metal removal is similar to that in low medium concentration metal removal in AC systems.

The results of the AC systems operated up to their 200% designed treatment capacities (AC-1: 1,000 gallons, AC-2: 600 gallons, AC-3: 400 gallons) showed that Ca (Figure 20) were not effectively removed and their average removal efficiencies were  $1.04\% \pm 0.20\%$  (AC-1),  $-0.12\% \pm 0.56\%$  (AC-2), and  $-0.11\% \pm 0.56\%$ . Mg was not effectively removed either (Figure 21) and their removal efficiencies were average removal efficiencies were  $1.13\% \pm 0.36\%$  (AC-1),  $-0.28\% \pm 0.59\%$  (AC-2), and  $-0.03\% \pm 0.58\%$  (AC-3).

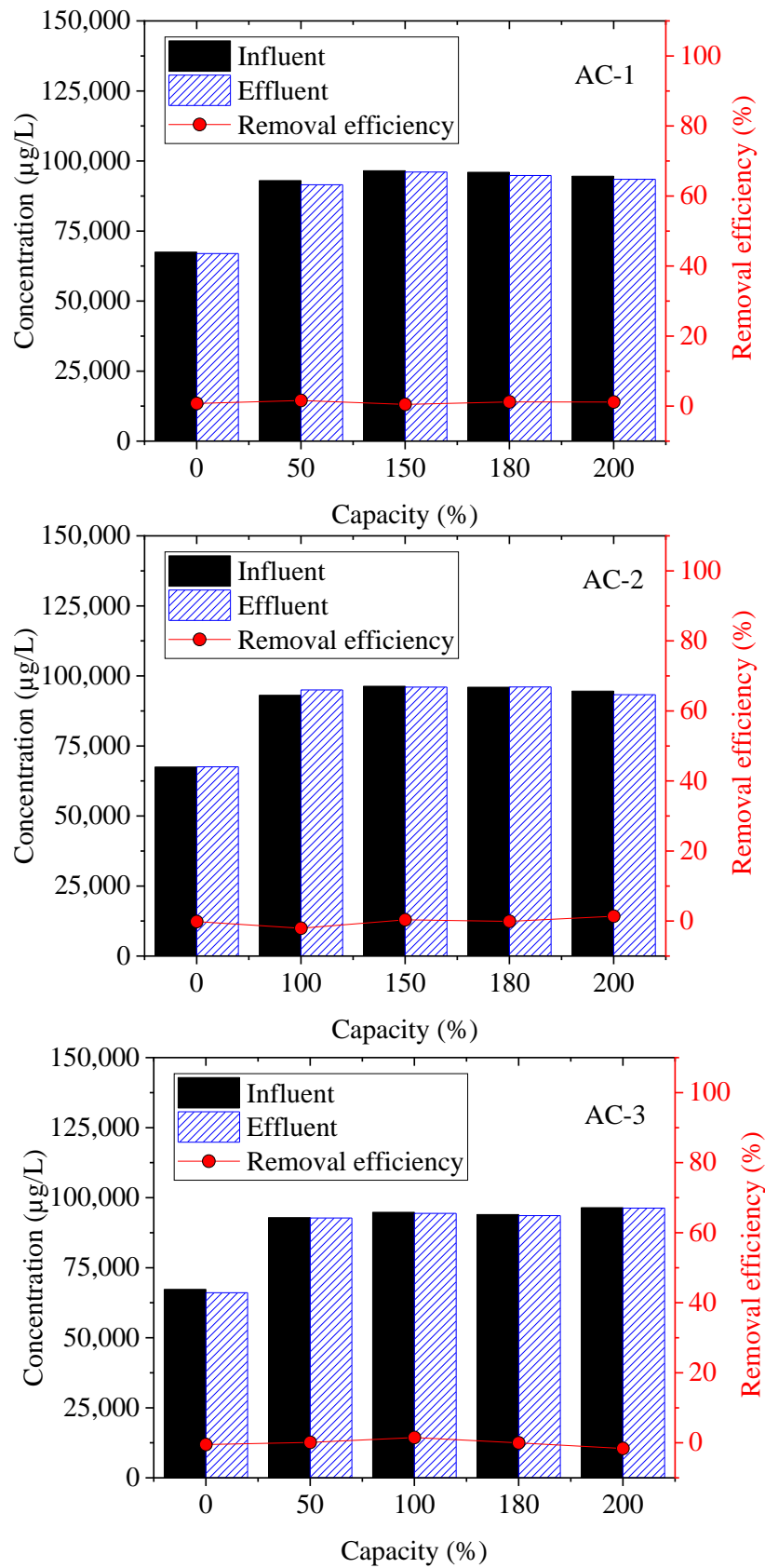
Similarly, a significant fraction of Fe was removed (Figure 22) in the AC-1 system with an average removal efficiency of  $61.35\% \pm 7.91\%$  (AC-1),  $84.14\% \pm 4.12\%$  (AC-2), and  $62.15\% \pm 7.37\%$  (AC-3).

Cu was also effectively removed (Figure 23) in the AC systems with average removal efficiencies of  $98.42\% \pm 0.23\%$  (AC-1),  $97.15\% \pm 1.09\%$  (AC-2), and  $95.25\% \pm 2.77\%$  (AC-3). The relatively high removal efficiency of Cu may be related to its low influent concentration ( $26.8 - 30.3 \mu\text{g/L}$ ).

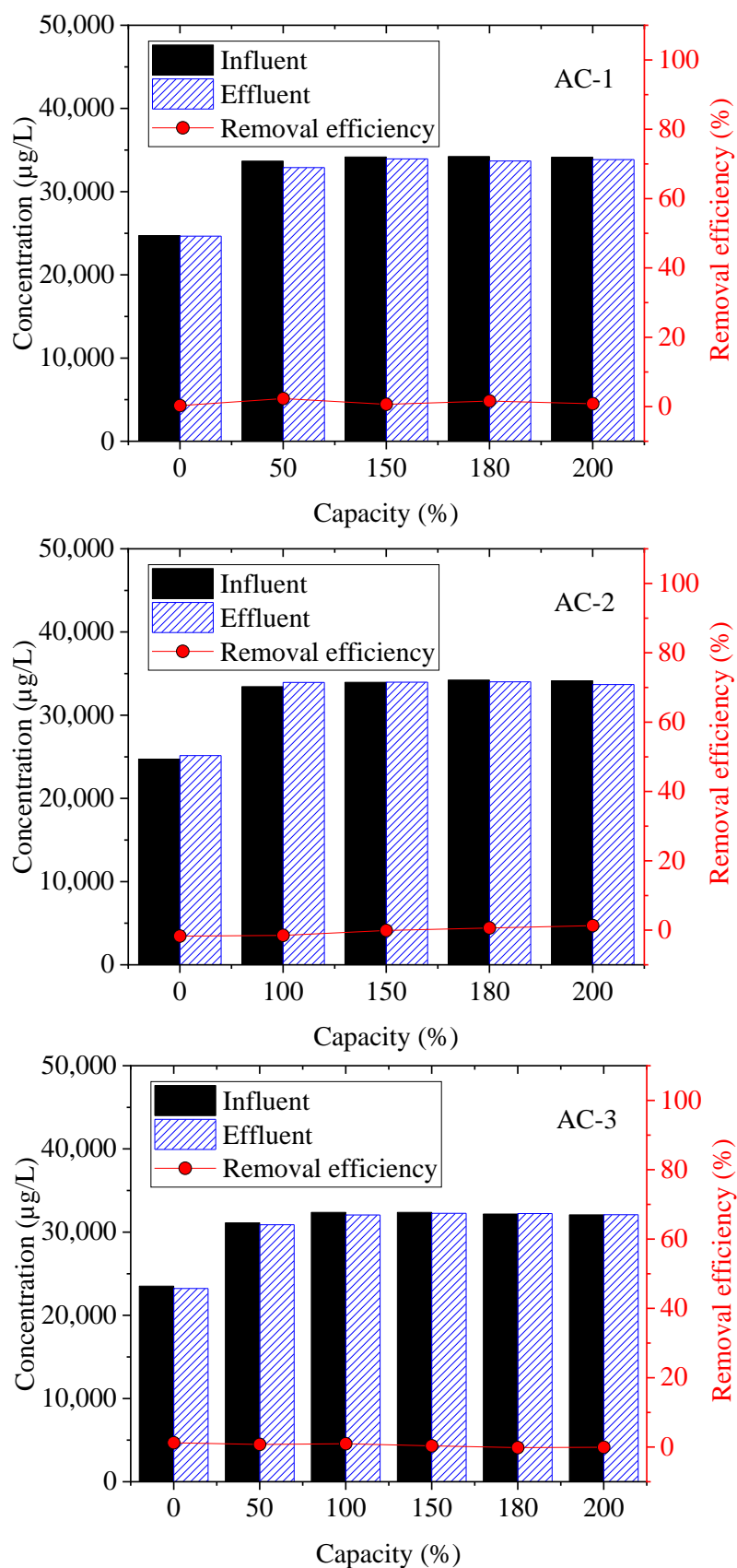
Mn was not effectively removed (Figure 24) by the AC systems and its removal efficiencies was  $0.87\% \pm 2.38\%$  (AC-1),  $2.48\% \pm 4.48\%$  (AC-2), and  $2.96\% \pm 0.68\%$  (AC-3). Compared with low concentration Mn removal testing, relatively low variability of Mn removal efficiencies was observed during medium concentration Mn removal testing.

U was not effectively removed (Figure 25) by the AC systems and its average removal efficiencies was  $-1.95\% \pm 3.72\%$  (AC-1),  $0.58\% \pm 7.58\%$  (AC-2),  $-3.00\% \pm 1.92\%$  (AC-3).

Overall, these results indicated that AC systems were generally ineffective to remove medium concentration metals in drinking water.

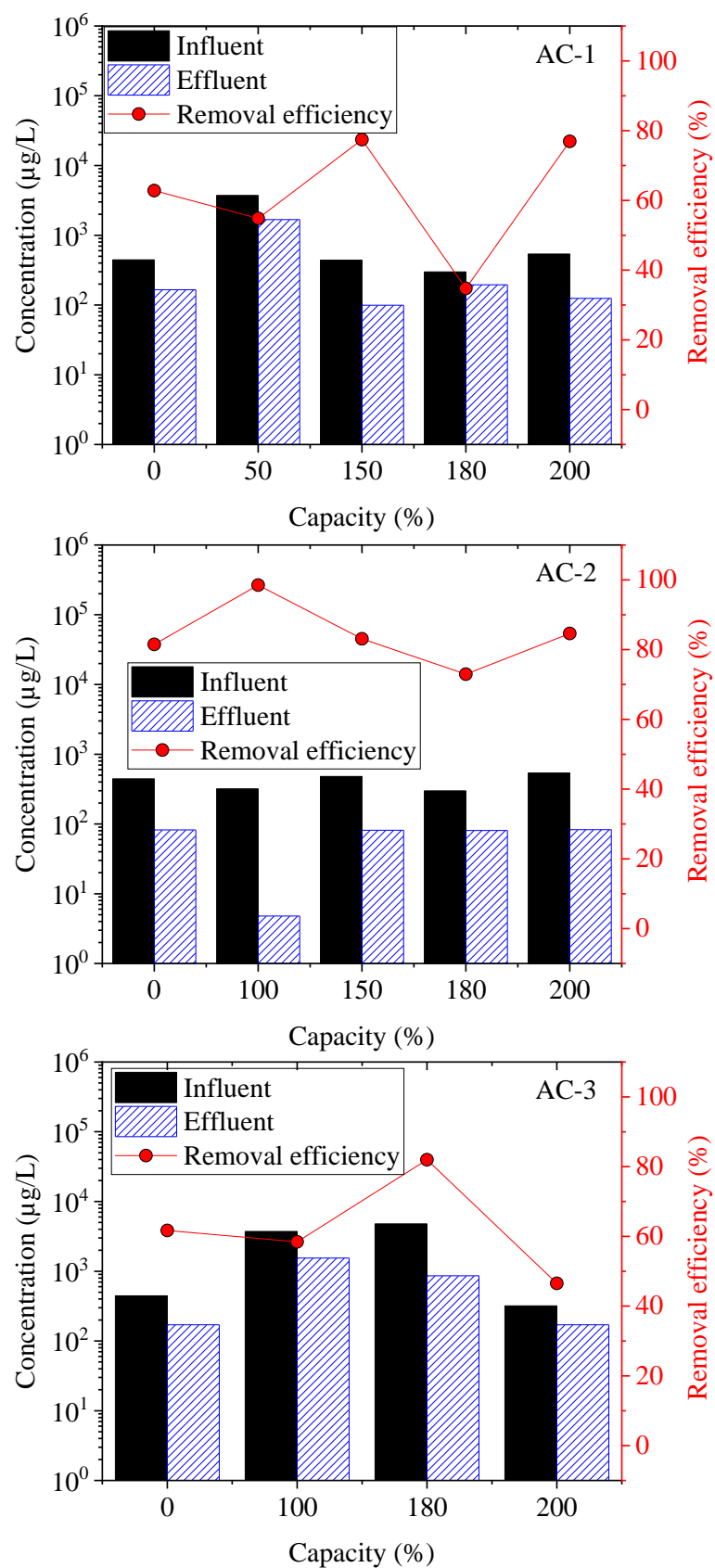


**Figure 20.** Removal of medium concentration Ca in AC systems.

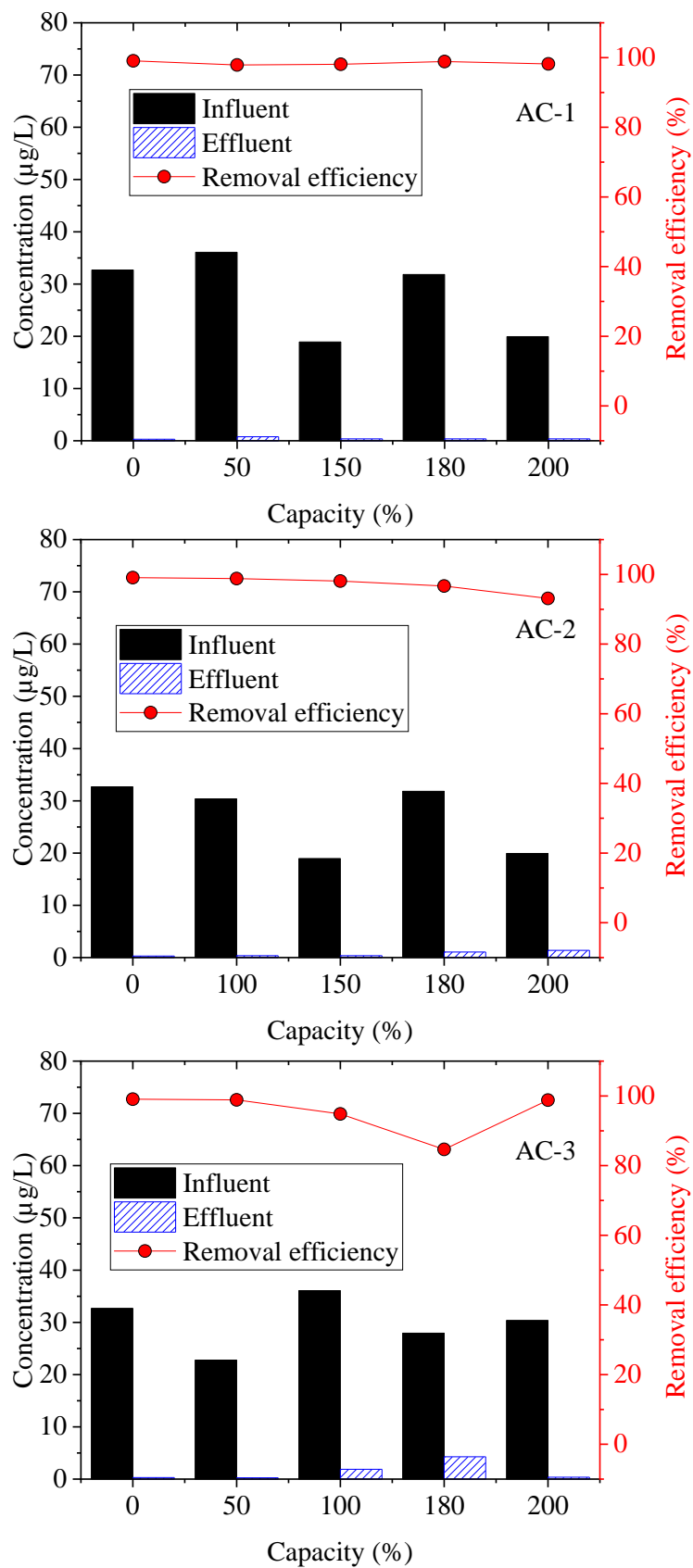


**Figure 21.** Removal of medium concentration Mg in AC systems.

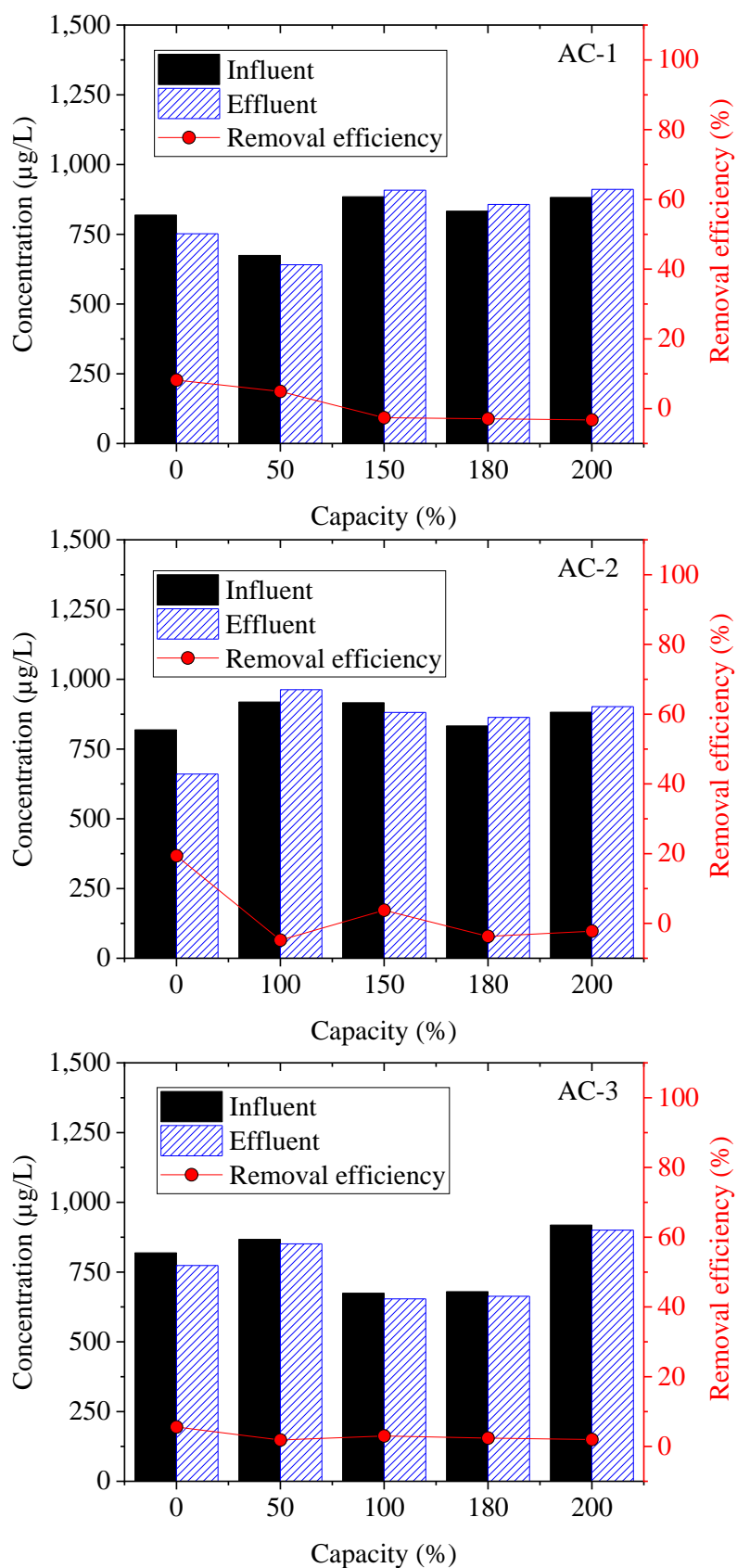




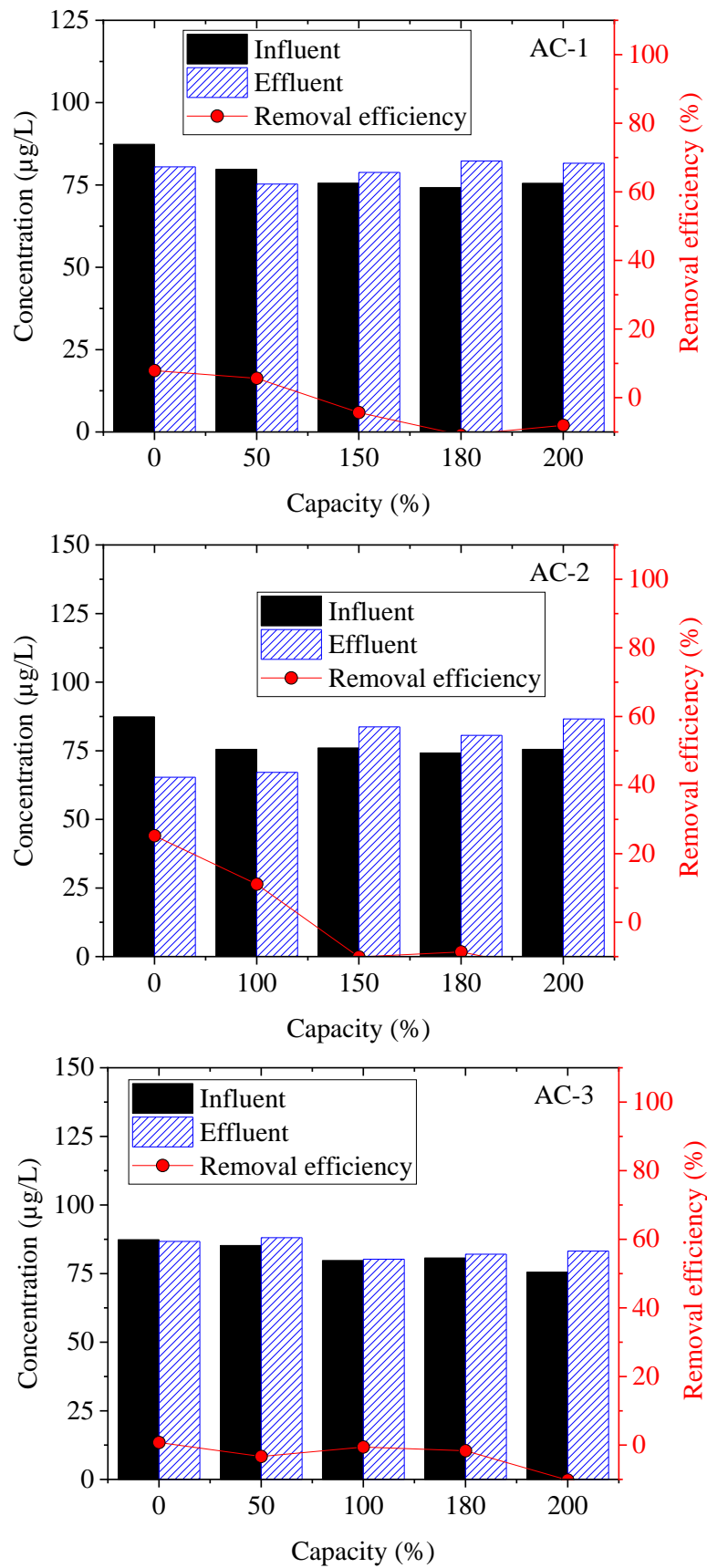
**Figure 22.** Removal of medium concentration Fe in AC systems.



**Figure 23.** Removal of medium concentration Cu in AC systems.



**Figure 24.** Removal of medium concentration Mn in AC systems.



**Figure 25.** Removal of medium concentration U in AC systems.

#### 4.5 Removal of medium concentration metals in RO systems

The overall trend of medium concentration metal removal is similar to that in low medium concentration metal removal in RO systems. Metals were effectively removed in RO systems and metal removal efficiencies were consistently higher than those in AC systems.

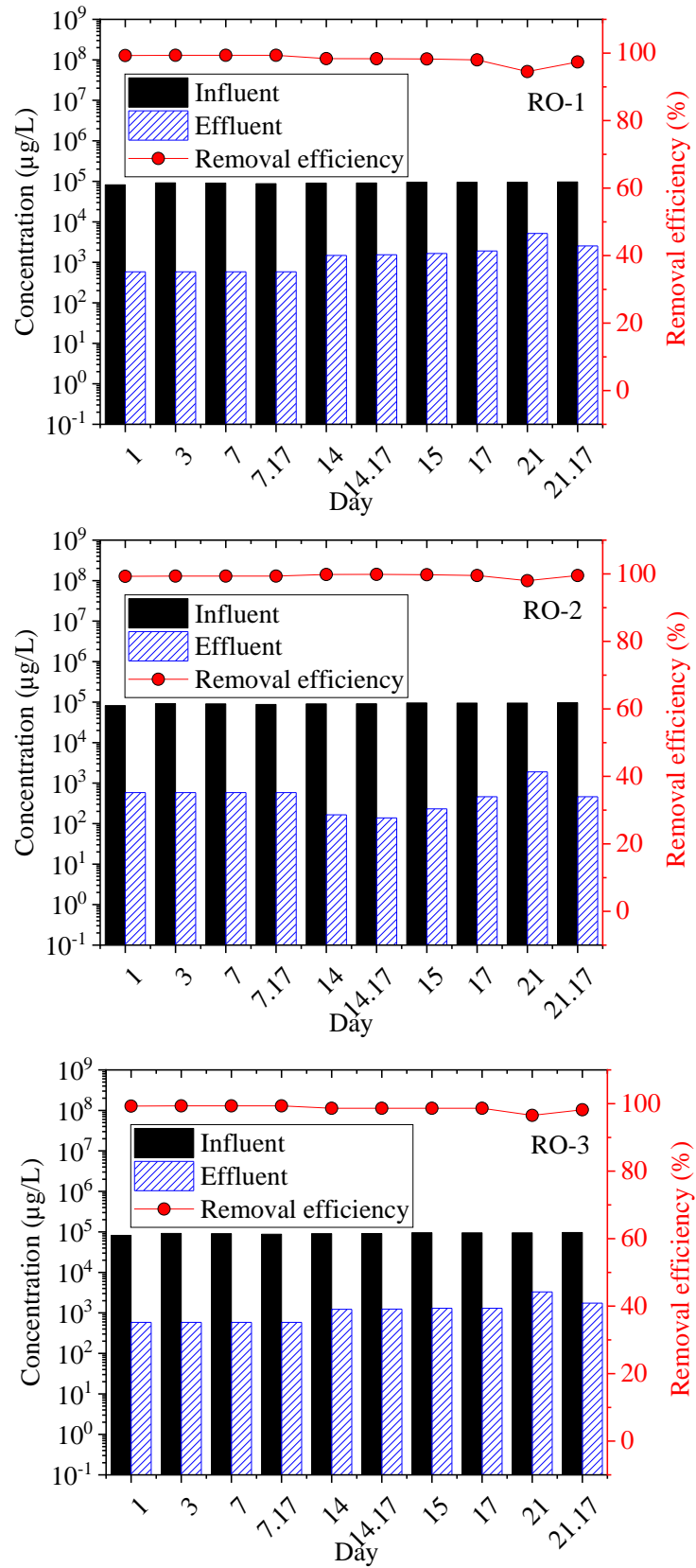
After three weeks of filtration, Ca (Figure 26), Mg (Figure 27), Fe (Figure 28), and Cu (Figure 29) were all effectively removed in RO systems.

The average removal efficiencies of Ca were  $98.22\% \pm 0.46\%$  (RO-1),  $99.38\% \pm 0.17\%$  (RO-2), and  $98.66\% \pm 0.27\%$  (RO-3). The average removal efficiencies of Mg were  $98.22\% \pm 0.46\%$  (RO-1),  $99.42\% \pm 0.15\%$  (RO-2), and  $98.94\% \pm 0.20\%$  (RO-3). The average removal efficiencies of Fe were  $98.18\% \pm 0.73\%$  (RO-1),  $97.75\% \pm 1.25\%$  (RO-2), and  $97.87\% \pm 1.08\%$  (RO-3). The average removal efficiencies of Cu were  $99.09\% \pm 0.27\%$  (RO-1),  $98.70\% \pm 0.46\%$  (RO-2), and  $97.53\% \pm 1.31\%$  (RO-3).

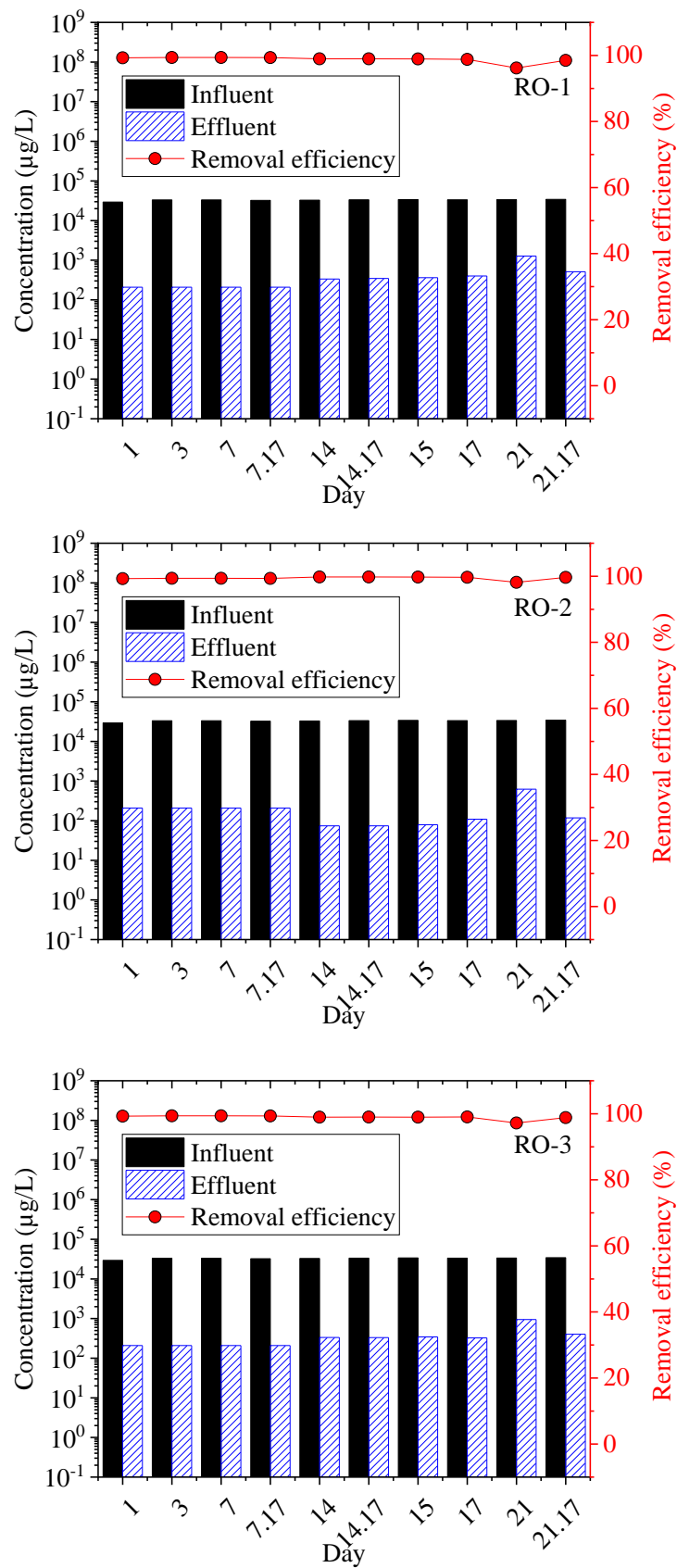
Mn (Figure 30) was effectively removed in the RO systems and its average removal efficiencies was  $97.09\% \pm 0.58\%$  (RO-1),  $97.81\% \pm 0.52\%$  (RO-2),  $97.49\% \pm 0.51\%$  (RO-3).

U (Figure 31) was also effectively removed in the RO-1 system and its average removal efficiencies was  $97.56\% \pm 1.53\%$  (RO-1),  $98.17\% \pm 1.58\%$  (RO-2),  $96.55\% \pm 2.06\%$  (RO-3).

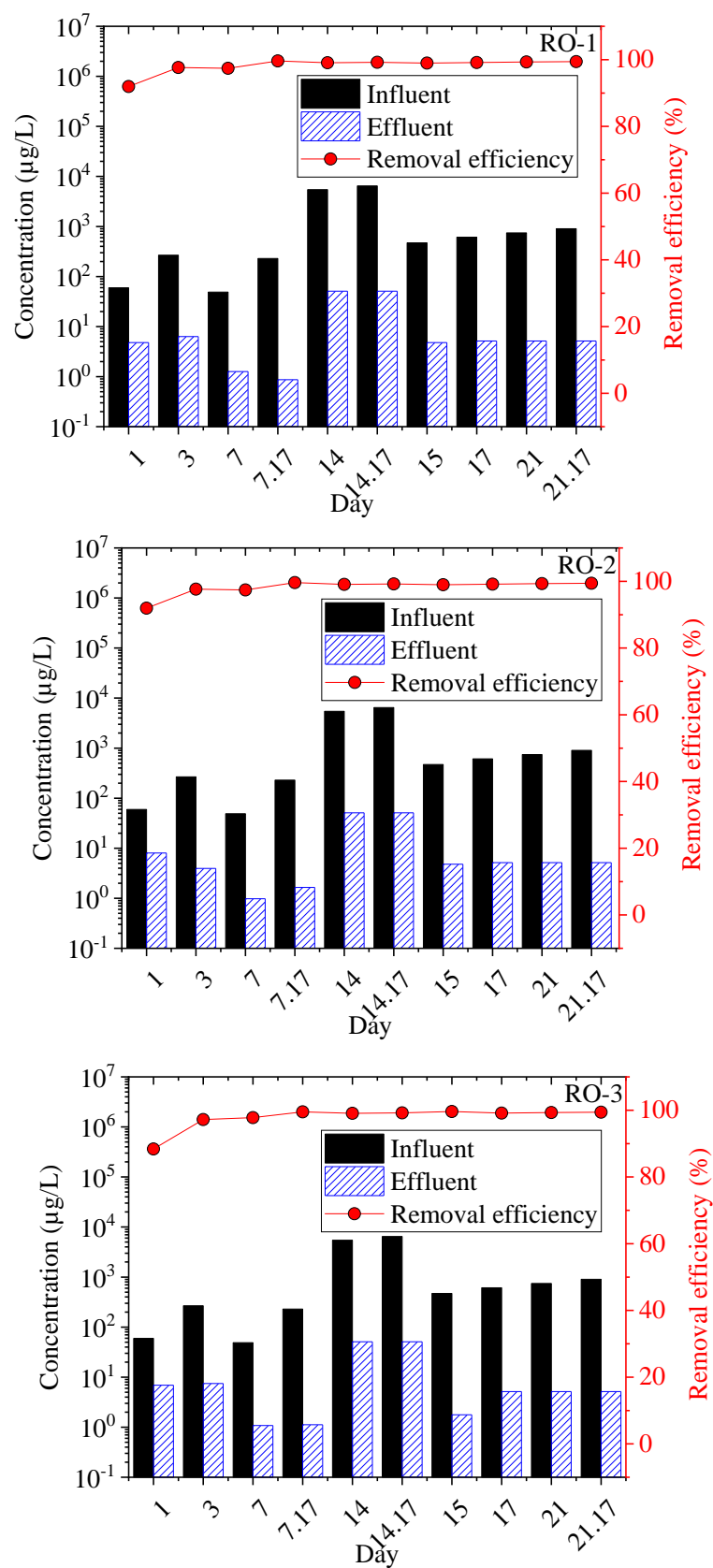
Overall, these results indicated that RO systems were effective to remove medium concentration metals in drinking water.



**Figure 26.** Removal of medium concentration Ca in RO systems.

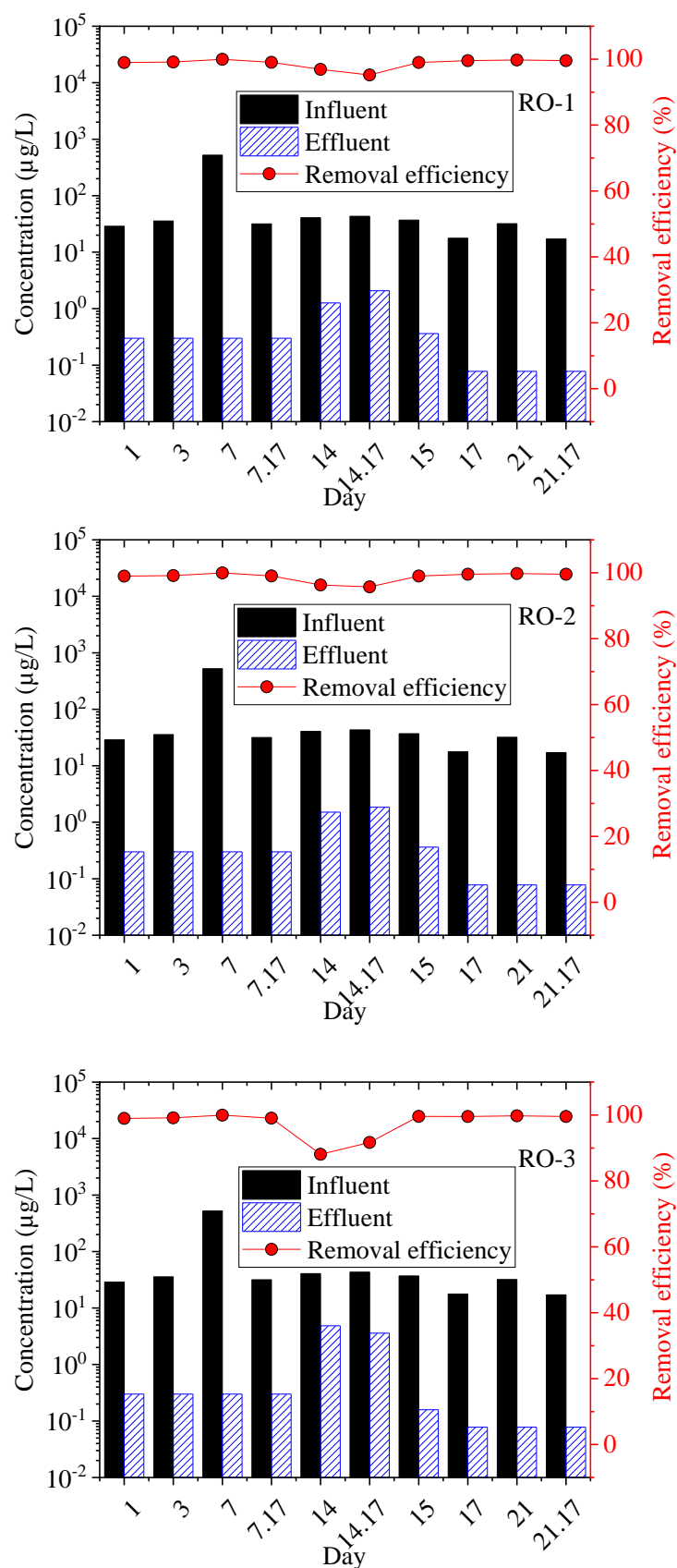


**Figure 27.** Removal of medium concentration Mg in RO systems.

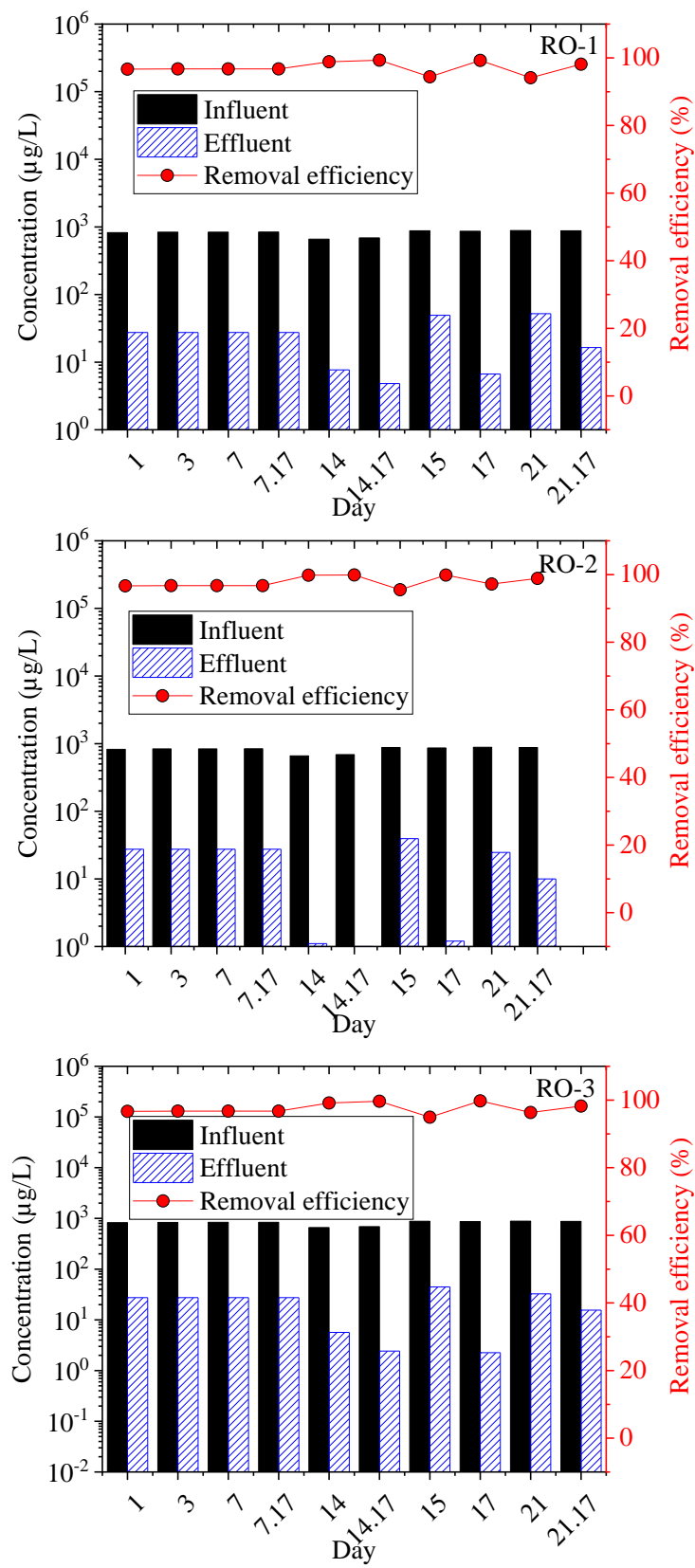


**Figure 28.** Removal of medium concentration Fe in RO systems.

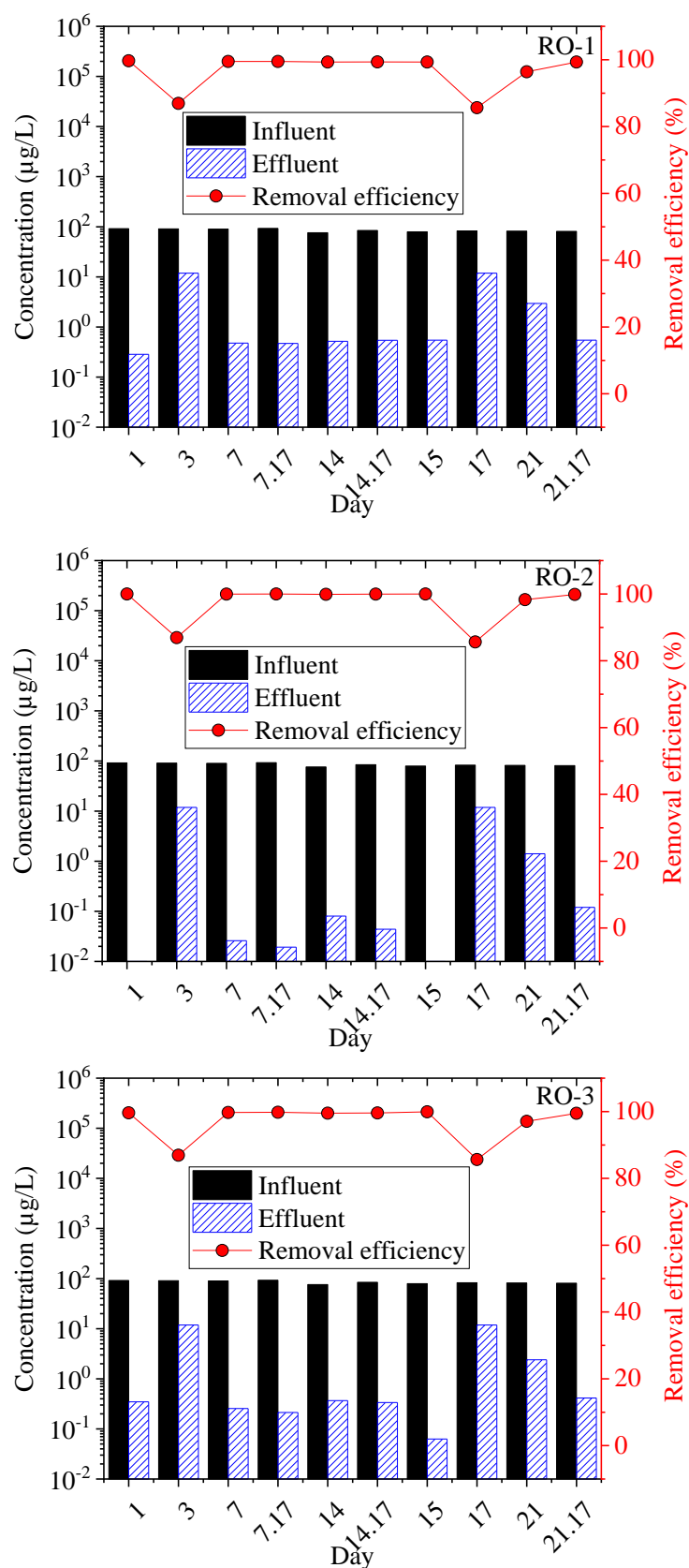




**Figure 29.** Removal of medium concentration Cu in RO systems.



**Figure 30.** Removal of medium concentration Mn in RO systems.



**Figure 31.** Removal of medium concentration U in RO systems.

#### 4.6 Removal of medium concentration AOC in AC and RO systems

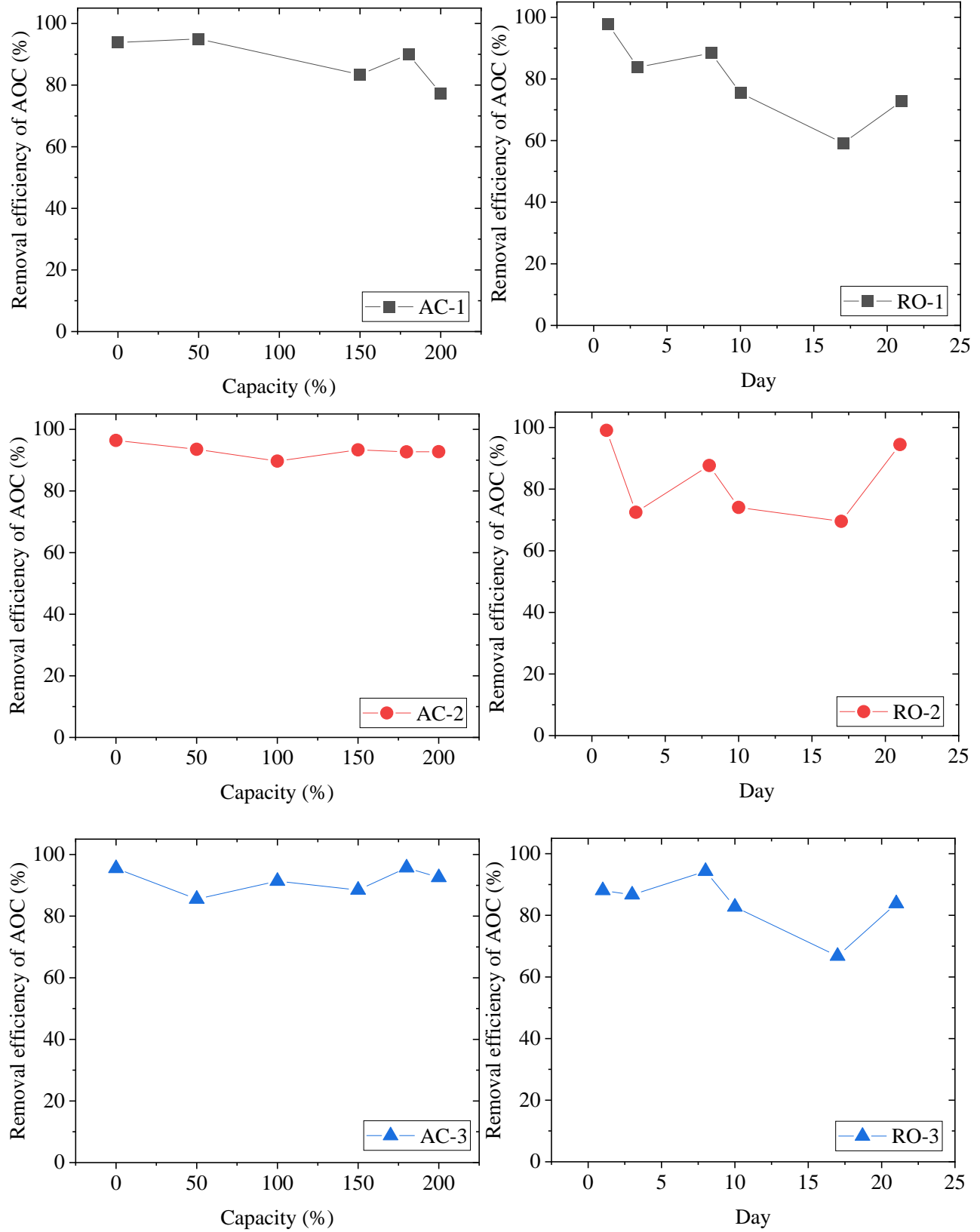
The results of medium concentration AOC removal in AC and RO systems are shown in Figure 32.

Medium concentration AOC was relatively effectively removed in all AC systems and the average AOC removal efficiencies were  $87.92\% \pm 3.32\%$  (AC-1),  $93.07\% \pm 0.96\%$  (AC-2), and  $91.52\% \pm 1.63\%$  (AC-3). There was a slight decrease in AOC removal efficiency in the AC-1 system, while AOC removal efficiencies were relatively stable in both the AC-2 system and AC-3 system.

The average AOC removal efficiencies were  $79.57\% \pm 5.49\%$  (RO-1),  $82.89\% \pm 5.11\%$  (RO-2), and  $83.73\% \pm 3.78\%$  (RO-3). High variability of AOC removal was observed in all RO systems, which can also be explained by the heterogeneity of biofilm in the RO systems. AOC removal efficiencies decreased in RO-1 and RO-3 systems, while no clear trend of AOC removal was observed in the RO-2 system.

There was no clear correlation between metal removal efficiencies and AOC removal efficiencies, suggesting the physical membrane separation process and microbial separation process are not directly linked, which was possibly attributed to microbial growth and microbial utilization of AOC during long-term filtration.

Variabilities of AOC removal efficiencies in RO systems were higher than those in AC systems, which may be related to microbial activities on membrane surfaces that may change membrane flux in RO systems. Additional experiments are needed to further examine the effects of microbial growth on AOC removal in POU systems.



**Figure 32.** Removal of medium concentration AOC in AC and RO systems.

#### 4.7 Total dissolved solid and pH

The results of medium concentration pH levels are shown in

Table 8 and Table 9. The pH values of AC influent (8.777) and AC effluent (AC-1: 8.756, AC-2: 8.759, AC-3: 8.756) were similar, suggesting that AC systems do not change pH significantly. Contrarily, pH value of RO influent was 8.643, which much lower pH values were observed in RO effluent (RO1: 7.636, RO-2: 7.729, RO-3: 7.594), suggesting that RO systems can effectively reduce to around neutral.

The typical values of pH vary between 6.5 to 9.5 in drinking water. Although pH may affect the water treatment and clarification performance during drinking water treatment, regular pH change in drinking water does not have direct adverse health effects on humans (WHO, 2007).

**Table 8.** pH analysis for medium concentration AC samples

Day	Influent pH	AC-1 effluent pH	AC-2 effluent pH	AC-3 effluent pH
1		8.681	8.520	8.429
2	8.706			8.656
4	8.816	8.690		8.633
5	8.620		8.754	
6	8.801			8.786
8	8.918			8.942
9	8.790		8.749	8.765
12	8.723	8.708		
13	8.806		8.758	
15	8.822	8.782	8.737	
16	8.772	8.845	8.797	
<b>Average</b>	8.777	8.756	8.759	8.756

**Table 9.** pH analysis for medium concentration RO samples

Day	Influent pH	RO-1 effluent pH	RO-2 effluent pH	RO-3 effluent pH
1	8.777	6.961	6.909	7.065
3	8.068	7.021	7.345	7.433
8	8.796	7.491	7.411	7.339
10	8.802	8.519	8.753	8.072
17	8.823	7.962	8.153	8.188
21	8.595	7.863	7.803	7.464
<b>Average</b>	8.643	7.636	7.729	7.594

The results of medium concentration total dissolved solids (TDS) removal in both AC and RO systems are shown in Table 10 and Table 11. The average TDS removal efficiencies were

88.75% (RO-1), 87.81% (RO-2), and 86.08% (RO-3). Similar to the variability of metal removal efficiency, high variability of TDS removal efficiencies was observed in AC systems.

TDS and pH are often used to describe drinking water quality. The World Health Organization (WHO) states that TDS can affect the taste of the water and different concentrations of TDS may be acceptable depending on the varying contents in the water (WHO, 2003). TDS criteria have been set based on their concentrations with less than 300 mg/L as excellent quality, 300 to 600 mg/L as good, 600 to 900 mg/L as fair, and 900 to 1200 mg/L as poor. Drinking water with TDS concentration of more than 1200 mg/L is deemed unacceptable.

The average TDS concentrations were 466 mg/L for RO influent and 431 mg/L for AC influent in this study, which were in the range of 300 to 600 mg/L as good quality drinking water based on the criteria set by WHO. The results in this study showed that RO systems were more effective to remove TDS than AC systems, which was consistent with the result of metal removal in RO and AC systems.

**Table 10.** TDS analysis for medium concentration AC samples

Day	Influent		AC-1		AC-2		Ac-3	
	Average (mg/L)	Standard deviation	Average (mg/L)	Standard deviation	Average (mg/L)	Standard deviation	Average (mg/L)	Standard deviation
1			353	82	420	115	401	57
2	433	12					446	27
4	390	14	485	12			446	35
5	451	28			435	18		
6	440	37					326	143
8	377	31					439	26
9	442	42			460	10	429	40
12	446	27	443	48				
13	433	12			436	30		
15	353	82	438	16	421	30		
16	401	57	413	4	347	51		
<b>Average</b>	417	34	426	32	420	42	414	55

**Table 11.** TDS analysis for medium concentration RO samples

Day	Influent		RO-1		RO-2		RO-3	
	Average (mg/L)	Standard deviation	Average (mg/L)	Standard deviation	Average (mg/L)	Standard deviation	Average (mg/L)	Standard deviation
1	359	292	71	44	78	78	158	161
3	489	97	179	174	54	49	89	39
8	441	79	59	68	66	10	39	58
10	456	97	49	41	80	5	112	67
17	483	27	77	57	69	16	54	20
21	343	141	232	195	121	8	136	48
<b>Average</b>	428	122	111	97	78	28	98	65

## 5. CONCLUSIONS AND FUTURE WORK

### 5.1 Conclusion

AC systems and RO systems are commonly used point-of-use (POU) water filtration systems to remove trace-level emerging contaminants. In this study, metal removal and AOC removal in POU systems were evaluated. Mn and U were spiked into tap water and removal efficiencies of Ca, Mg, Fe, Cu, Mn, and U in three AC filters, and three RO filters were measured. AC systems were tested up to 200% of their designed treatment capacities, which RO systems were tested up to three weeks of filtration.

The results indicated that AC systems were generally ineffective to remove metals in drinking water, while metals were effectively removed in RO systems.

The results showed that Ca was not effectively removed in AC systems with removal efficiencies of less than 1%. Removal efficiencies of Mg in AC systems were also less than 1%. Various fractions of Fe were removed with its removal efficiencies in AC systems ranged between 61% and 84%. Cu was effectively removed in AC systems with removal efficiencies greater than 95%, which was possibly related to its low influent concentration in drinking water ( $<30 \mu\text{g/L}$ ). Mn was ineffectively removed from AC systems and removal efficiencies of Mn in AC systems were consistently low. The highest Mn removal efficiency of 31% was observed during the first day of AC filtration, then Mn was not effectively removed throughout the AC filtration experiment. U was not effectively removed either. The highest U removal efficiency of 63% was also observed during the first day of AC filtration, then U was not effectively removed throughout the AC filtration experiment.

Different from AC systems, RO systems were consistently effective to remove all metals. Ca, Mg, Fe, and Cu were all removed with removal efficiencies greater than 98%, while removal efficiencies of Mn and U in RO systems were above 95%.

AOC was effectively removed from all AC systems and all RO systems and high variability of AOC removal efficiencies were observed, which may be attributed to the heterogenicity of biofilm and microbial growth in POU systems. High variability of AOC removal efficiencies was observed in both AC and RO systems.



The new knowledge generated from this study can help improve our understanding of emerging contaminant removal in POU systems and develop better strategies for the design and operation of POU systems to remove emerging contaminants in drinking water and mitigate their health risks to humans.

## **5.2 Future work**

Several studies can be done to further understand emerging contaminant removal and microbial growth in POU systems.

In addition to the comparison of removal efficiencies based on relative treatment capacities of different AC systems, removal efficiencies can be plotted in their treated volumes and bed volumes for individual AC systems.

A comprehensive study can be done to evaluate the removal efficiencies of other emerging contaminants and microbial growth potentials of commonly detected bacteria in tap water.

The impact of different materials used in POU systems can be further evaluated.

Membrane fouling that may significantly affect RO removal efficiency can be examined for different types of organic contaminants that may promote microbial growth on the RO membrane surface.

The effects of water quality parameters, such as pH and temperature, can be evaluated and correlated with AOC removal in POU systems.

Mechanistic studies can be done to evaluate the effect of water quality on biofilm development in POU systems and maintenance requirements to minimize microbial growth in POU systems.

Although emerging contaminants can be removed in POU systems, additional microbial growth due to increased organic carbon concentration in POU systems may increase the possibilities of microbial growth and health risks associated with POU systems.

The results from these experiments can be used to design more efficient POU systems and also provide guidance on the operation and maintenance of POU systems to remove emerging contaminants while keeping microbial growth potential at a low level.

Last but not least, a cost analysis can be done to better understand the costs and benefits of using POU systems at residential homes as the last barrier for emerging contaminant removal.

## REFERENCES

- Azamat, J., Khataee, A., & Joo, S. W. (2014). Separation of a heavy metal from water through a membrane containing boron nitride nanotubes: molecular dynamics simulations. *Journal of Molecular Modeling*, 20(10), 2468. doi:10.1007/s00894-014-2468-1
- Baird, R., Bridgewater, L., American Public Health Association, American Water Works Association, & Water Environment Federation. (2017). *Standard methods for the examination of water and wastewater, 20rd edition*: APHA American Public Health Association.
- Banning, A., & Benfer, M. (2017). Drinking water uranium and potential health effects in the German Federal State of Bavaria. *International Journal of Environmental Health Research*, 14(8). doi:10.3390/ijerph14080927
- Barbeau, B., Carriere, A., & Bouchard, M. F. (2011). Spatial and temporal variations of manganese concentrations in drinking water. *Journal of Environmental Science and Health, Part A Toxic/Hazardous Substances and Environmental Engineering*, 46(6), 608-616. doi:10.1080/10934529.2011.562854
- Berisha, F., & Goessler, W. (2013). Uranium in Kosovo's drinking water. *Chemosphere*, 93(9), 2165-2170. doi:10.1016/j.chemosphere.2013.07.078
- Bigalke, M., Schwab, L., Rehmus, A., Tondo, P., & Flisch, M. (2018). Uranium in agricultural soils and drinking water wells on the Swiss Plateau. *Environmental Pollution*, 233, 943-951. doi:10.1016/j.envpol.2017.09.061
- Bjorklund, G., Semenova, Y., Pivina, L., Dadar, M., Rahman, M. M., Aaseth, J., & Chirumbolo, S. (2020). Uranium in drinking water: a public health threat. *Arch Toxicol*, 94(5), 1551-1560. doi:10.1007/s00204-020-02676-8
- Bolt, A. M., Medina, S., Lauer, F. T., Liu, K. J., & Burchiel, S. W. (2019). Minimal uranium immunotoxicity following a 60-day drinking water exposure to uranyl acetate in male and female C57BL/6J mice. *Toxicol Appl Pharmacol*, 372, 33-39. doi:10.1016/j.taap.2019.04.003
- Bosscher, V., Lytle, D. A., Schock, M. R., Porter, A., & Del Toral, M. (2019). POU water filters effectively reduce lead in drinking water: a demonstration field study in flint, Michigan. *J Environ Sci Health A Tox Hazard Subst Environ Eng*, 54(5), 484-493. doi:10.1080/10934529.2019.1611141
- Chaidez, C., & Gerba, C. (2004). Comparison of the microbiologic quality of point-of-use (POU)-treated water and tap water. *International Journal of Environmental Health Research*, 14(4), 253-260. doi:10.1080/09603120410001725595
- Chen, S., Hong, J., Yang, H., & Yang, J. (2013). Adsorption of uranium (VI) from aqueous solution using a novel graphene oxide-activated carbon felt composite. *Journal of Environmental Radioactivity*, 126, 253-258. doi:10.1016/j.jenvrad.2013.09.002
- Chen, X., Chen, D., Li, N., Xu, Q., Li, H., He, J., & Lu, J. (2020). Modified-MOF-808-loaded polyacrylonitrile membrane for highly efficient, simultaneous emulsion separation and heavy metal ion removal. *American Chemical Society Applied Materials & Interfaces*, 12(35), 39227-39235. doi:10.1021/acsami.0c10290
- Clesceri, L. S., Greenberg, A. E., & Eaton, A. D. (1998). *Standard Methods for the Examination of Water and Wastewater, 20th Edition*: APHA American Public Health Association.

- Coleman, S. J., Coronado, P. R., Maxwell, R. S., & Reynolds, J. G. (2003). Granulated activated carbon modified with hydrophobic silica aerogel-potential composite materials for the removal of uranium from aqueous solutions. *Environ Sci Technol*, 37(10), 2286-2290. doi:10.1021/es020929e
- Dong, L., Hou, L., Wang, Z., Gu, P., Chen, G., & Jiang, R. (2018). A new function of spent activated carbon in BAC process: Removing heavy metals by ion exchange mechanism. *J Hazard Mater*, 359, 76-84. doi:10.1016/j.jhazmat.2018.07.030
- Eaton, A. D., Clesceri, L. S., Greenberg, A. E., Franson, M. A. H., American Public Health Association., American Water Works Association., & Water Environment Federation. (1998). *Standard methods for the examination of water and wastewater* (20th ed.). Washington, DC: American Public Health Association.
- Environmental Protection Agency. (2006). *Point-of-use or point-of-entry treatment options for small drinking water systems*. (EPA 815-R-06-010).
- EPA-EDSP. (2008). *Endocrine disruptor screening program*. Retrieved from Washington, DC: <http://www.epa.gov/endo/>
- Escobar, I. C., Randall, A. A., & Taylor, J. S. (2001). Bacterial Growth in Distribution Systems: Effect of Assimilable Organic Carbon and Biodegradable Dissolved Organic Carbon. *Environmental Science & Technology*, 35(17), 3442-3447. doi:10.1021/es0106669
- Ferraro, P. M., Gambaro, G., Curhan, G. C., & Taylor, E. N. (2018). Intake of trace metals and the risk of incident kidney stones. *Journal of Urology*, 199(6), 1534-1539. doi:<https://doi.org/10.1016/j.juro.2018.01.077>
- IEH. (2005). *Chemicals purported to be endocrine disruptors: a compilation of published lists*. Retrieved from Leicester, United Kingdom: <http://www.silsoe.cranfield.ac.uk/ieh/pdf/w20.pdf>
- International, N. (2021). NSF standards for water treatment systems. Retrieved from <https://www.nsf.org/knowledge-library/standards-water-treatment-systems>
- Jakhu, R., Mehra, R., & Mittal, H. M. (2016). Exposure assessment of natural uranium from drinking water. *Environmental Science: Processes and Impacts*, 18(12), 1540-1549. doi:10.1039/c6em00514d
- Katsoyiannis, I. A., & Zouboulis, A. I. (2013). Removal of uranium from contaminated drinking water: a mini review of available treatment methods. *Desalination and Water Treatment*, 51(13-15), 2915-2925. doi:10.1080/19443994.2012.748300
- Kurtio, P., Auvinen, A., Salonen, L., Saha, H., Pekkanen, J., Mäkeläinen, I., . . . Komulainen, H. (2002). Renal effects of uranium in drinking water. *Environmental Health Perspectives*, 110(4), 337-342.
- Li, W., Zhang, J., Wang, F., Qian, L., Zhou, Y., Qi, W., & Chen, J. (2018). Effect of disinfectant residual on the interaction between bacterial growth and assimilable organic carbon in a drinking water distribution system. *Chemosphere*, 202, 586-597. doi:10.1016/j.chemosphere.2018.03.056
- Liu, W., Wu, H., Wang, Z., Ong, S. L., Hu, J. Y., & Ng, W. J. (2002a). Investigation of assimilable organic carbon (AOC) and bacterial regrowth in drinking water distribution system. *Water Research*, 36(4), 891-898. doi:[https://doi.org/10.1016/S0043-1354\(01\)00296-2](https://doi.org/10.1016/S0043-1354(01)00296-2)
- Liu, W., Wu, H., Wang, Z., Ong, S. L., Hu, J. Y., & Ng, W. J. (2002b). Investigation of assimilable organic carbon (AOC) and bacterial regrowth in drinking water distribution system. *Water Res*, 36(4), 891-898. doi:10.1016/s0043-1354(01)00296-2

- Liu, Z., Lompe, K. M., Mohseni, M., Berube, P. R., Sauve, S., & Barbeau, B. (2020). Biological ion exchange as an alternative to biological activated carbon for drinking water treatment. *Water Res*, 168, 115148. doi:10.1016/j.watres.2019.115148
- Lo, K., Yang, J. L., Chen, C. L., Liu, L., Huang, Y. Q., Feng, Y. Q., & Yang, A. M. (2021). Associations between blood and urinary manganese with metabolic syndrome and its components: Cross-sectional analysis of National Health and Nutrition Examination Survey 2011-2016. *Sci Total Environ*, 780, 146527. doi:10.1016/j.scitotenv.2021.146527
- Lou, J. C., Lin, C. Y., Han, J. Y., Tseng, W. B., Hsu, K. L., & Chang, T. W. (2012). Comparing removal of trace organic compounds and assimilable organic carbon (AOC) at advanced and traditional water treatment plants. *Environmental Monitoring and Assessment*, 184(6), 3491-3501. doi:10.1007/s10661-011-2203-7
- Lou, X. Y., Xu, Z., Bai, A. P., Resina-Gallego, M., & Ji, Z. G. (2020). Separation and recycling of concentrated heavy metal wastewater by tube membrane distillation integrated with crystallization. *Membranes*, 10(1). doi:10.3390/membranes10010019
- Lucchini, R. G., Aschner, M., Landrigan, P. J., & Cranmer, J. M. (2018). Neurotoxicity of manganese: Indications for future research and public health intervention from the Manganese 2016 conference. *Neurotoxicology*, 64, 1-4. doi:10.1016/j.neuro.2018.01.002
- Ma, J., Ibekwe, A. M., Leddy, M., Yang, C.-H., & Crowley, D. E. (2012). Assimilable Organic Carbon (AOC) in Soil Water Extracts Using *Vibrio harveyi* BB721 and Its Implication for Microbial Biomass. *Public Library of Science One*, 7(5), e28519. doi:10.1371/journal.pone.0028519
- Massachusetts DEP. (2006). *Perchlorate fact sheet*. Massachusetts Department of Environmental Protection Retrieved from <http://mass.gov/dep/toxics/pchlorqa.pdf>.
- Mora, A. M., Cordoba, L., Cano, J. C., Hernandez-Bonilla, D., Pardo, L., Schnaas, L., . . . van Wendel de Joode, B. (2018). Prenatal manganese exposure, excess manganese, and neurodevelopment at 1 year of age in the infants' environmental health (ISA) study. *Environmental Health Perspectives*, 126(5), 057007. doi:10.1289/EHP1955
- Nejadshafiee, V., & Islami, M. R. (2019). Adsorption capacity of heavy metal ions using sultone-modified magnetic activated carbon as a bio-adsorbent. *Materials Science and Engineering: C Materials for Biological Applications*, 101, 42-52. doi:10.1016/j.msec.2019.03.081
- Ohkouchi, Y., Ly, B. T., Ishikawa, S., Kawano, Y., & Itoh, S. (2013). Determination of an acceptable assimilable organic carbon (AOC) level for biological stability in water distribution systems with minimized chlorine residual. *Environmental Monitoring and Assessment*, 185(2), 1427-1436. doi:10.1007/s10661-012-2642-9
- Oloibiri, V., De Coninck, S., Chys, M., Demeestere, K., & Van Hulle, S. W. H. (2017). Characterisation of landfill leachate by EEM-PARAFAC-SOM during physical-chemical treatment by coagulation-flocculation, activated carbon adsorption and ion exchange. *Chemosphere*, 186, 873-883. doi:10.1016/j.chemosphere.2017.08.035
- Park, R. M., & Berg, S. L. (2018). Manganese and neurobehavioral impairment. A preliminary risk assessment. *Neurotoxicology*, 64, 159-165. doi:10.1016/j.neuro.2017.08.003
- Pick, F. C., Fish, K. E., Biggs, C. A., Moses, J. P., Moore, G., & Boxall, J. B. (2019). Application of enhanced assimilable organic carbon method across operational drinking water systems. *Public Library of Science One*, 14(12), e0225477. doi:10.1371/journal.pone.0225477

- Proctor, C. R., Reimann, M., Vriens, B., & Hammes, F. (2018). Biofilms in shower hoses. *Water Research*, 131, 274-286. doi:10.1016/j.watres.2017.12.027
- Redvers, N., Chischilly, A. M., Warne, D., Pino, M., & Lyon-Colbert, A. (2021). Uranium exposure in American Indian communities: Health, policy, and the way forward. *Environmental Health Perspectives*, 129(3), 35002. doi:10.1289/EHP7537
- Roitz, J. S., Flegal, A. R., & Bruland, K. W. (2002). The biogeochemical cycling of manganese in San Francisco Bay: Temporal and spatial variations in surface water concentrations. *Estuarine, Coastal and Shelf Science*, 54(2), 227-239. doi:10.1006/ecss.2000.0839
- Salveson, A., Brown, J., & Zhou, Z. (2010). *Monitoring microconstituents in an advanced wastewater treatment (AWT) facility and modeling discharge of reclaimed water to surface canals for indirect potable use*. Retrieved from Alexandria, VA, USA:
- Santos, J. S., Teixeira, L. S. G., dos Santos, W. N. L., Lemos, V. A., Godoy, J. M., & Ferreira, S. L. C. (2010). Uranium determination using atomic spectrometric techniques: An overview. *Analytica Chimica Acta*, 674(2), 143-156. doi:<https://doi.org/10.1016/j.aca.2010.06.010>
- Schulte-Herbrüggen, H. M. A., Semião, A. J. C., Chaurand, P., & Graham, M. C. (2016). Effect of pH and pressure on uranium removal from drinking water using NF/RO membranes. *Environmental Science & Technology*, 50(11), 5817-5824. doi:10.1021/acs.est.5b05930
- Semenova, Y., Pivina, L., Zhunussov, Y., Zhanaspayev, M., Chirumbolo, S., Muzdubayeva, Z., & Bjorklund, G. (2020). Radiation-related health hazards to uranium miners. *Environmental Science and Pollution Research*, 27(28), 34808-34822. doi:10.1007/s11356-020-09590-7
- Sly, L. I., Hodgkinson, M. C., & Arunpairojana, V. (1990). Deposition of manganese in a drinking water distribution system. *Applied and environmental microbiology*, 56(3), 628-639. doi:10.1128/AEM.56.3.628-639.1990
- Snyder, S. A., Adham, S., Redding, A. M., Cannon, F. S., DeCarolis, J., Oppenheimer, J., . . . Yoon, Y. (2007). Role of membranes and activated carbon in the removal of endocrine disruptors and pharmaceuticals. *Desalination*, 202(1-3), 156-181.
- Surdyk, S., Itani, M., Al-Lobaidy, M., Kahale, L. A., Farha, A., Dewachi, O., . . . Habib, R. R. (2021). Weaponised uranium and adverse health outcomes in Iraq: a systematic review. *British Medical Journal Global Health*, 6(2). doi:10.1136/bmjgh-2020-004166
- Tarale, P., Daiwile, A. P., Sivanesan, S., Stöger, R., Bafana, A., Naoghare, P. K., . . . Krishnamurthi, K. (2018). Manganese exposure: Linking down-regulation of miRNA-7 and miRNA-433 with  $\alpha$ -synuclein overexpression and risk of idiopathic Parkinson's disease. *Toxicology in Vitro*, 46, 94-101. doi:<https://doi.org/10.1016/j.tiv.2017.10.003>
- U.S. EPA. (1994). *Determined of metal and trace elements in water and wastes by inductively coupled plasma-atomic emission spectrometry* Retrieved from <https://www.epa.gov/sites/production/files/2015-06/documents/epa-200.7.pdf>.
- U.S. EPA. (2004). *Drinking water health advisory for manganese*. Retrieved from [https://www.epa.gov/sites/production/files/2014-09/documents/support\\_ccl\\_manganese\\_dwreport\\_0.pdf](https://www.epa.gov/sites/production/files/2014-09/documents/support_ccl_manganese_dwreport_0.pdf).
- U.S. EPA. (2005). *Ambient Water Quality Criteria - Nonylphenol. Final*. Retrieved from <http://www.epa.gov/waterscience/criteria/nonylphenol/final-doc.pdf>
- van der Kooij, D. (1990). Assimilable organic carbon (AOC) in drinking water. In G. A. McFeters (Ed.), *Drinking Water Microbiology: Progress and Recent Developments* (pp. 57-87). New York, NY: Springer New York.

- van der Kooij, D., Visser, A., & Hijnen, W. A. M. (1982). Determining the concentration of easily assimilable organic carbon in drinking water. *Journal (American Water Works Association)*, 74(10), 540-545.
- van der Kooij, D., Visser, A., & Oranje, J. P. (1982). Multiplication of fluorescent pseudomonads at low substrate concentrations in tap water. *Antonie van Leeuwenhoek*, 48(3), 229-243. doi:10.1007/bf00400383
- Vital, M., Stucki, D., Egli, T., & Hammes, F. (2010). Evaluating the growth potential of pathogenic bacteria in water. *Appl Environ Microbiol*, 76(19), 6477-6484. doi:10.1128/aem.00794-10
- Weinrich, L., LeChevallier, M., & Haas, C. N. (2016). Contribution of assimilable organic carbon to biological fouling in seawater reverse osmosis membrane treatment. *Water Res*, 101, 203-213. doi:10.1016/j.watres.2016.05.075
- Weinrich, L. A. (2015). The Impact of Assimilable Organic Carbon on Biological Fouling of Reverse Osmosis Membranes in Seawater Desalination. In: ProQuest Dissertations Publishing.
- Weiss, B. (2010). Lead, manganese, and methylmercury as risk factors for neurobehavioral impairment in advanced age. *International Journal of Alzheimer's Disease*, 2011, 607543. doi:10.4061/2011/607543
- Winde, F., Erasmus, E., & Geipel, G. (2017). Uranium contaminated drinking water linked to leukaemia—Revisiting a case study from South Africa taking alternative exposure pathways into account. *Science of The Total Environment*, 574, 400-421. doi:<https://doi.org/10.1016/j.scitotenv.2016.09.035>
- Yakout, S. M., Metwally, S. S., & El-Zakla, T. (2013). Uranium sorption onto activated carbon prepared from rice straw: Competition with humic acids. *Applied Surface Science*, 280, 745.
- Yang, J., Yang, A., Cheng, N., Huang, W., Huang, P., Liu, N., & Bai, Y. (2020). Sex-specific associations of blood and urinary manganese levels with glucose levels, insulin resistance and kidney function in US adults: National health and nutrition examination survey 2011-2016. *Chemosphere*, 258, 126940. doi:10.1016/j.chemosphere.2020.126940