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以太陽光驅動過硫酸鹽

去除廢水中抗生素之效能探討

The Efficiency Evaluation of Solar-Driven Peroxydisulfate
Oxidation for Removing Antibiotics in Wastewater

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中文摘要



為減少新興污染物排放至環境水體，臺灣環境部近年加嚴醫院放流水管理規範，不僅新增氨氮與自由餘氯排放標準，亦將抗生素及其他新興污染物納入監測項目。由於傳統生物處理程序難以有效去除多數藥物，且為降低鹵化消毒副產物 (Disinfection Byproducts, DBPs) 生成，自由餘氯排放濃度限制為 2 mg/L，因此需尋找兼具高降解效率與低副產物生成之替代氧化方式。

本研究評估過二硫酸鹽 (Peroxydisulfate, PDS) 應用於醫院廢水中微量抗生素降解之可行性，比較紫外光(UV) 與太陽光 (Solar) 活化之 PDS 系統(UV/PDS 與 Solar/PDS)，並以 Solar/Chlorine 作為比較基準。

結果顯示，UV/PDS 與 Solar/PDS 在廢水基質中皆具有良好的降解性，於反應 10 分鐘或 60 分鐘後，對 Sulfamethoxazole (SMX)及 Levofloxacin (LVX) 之去除率均達約 50%，且與以超純水為基質之實驗結果趨勢一致。相較之下，Solar/Chlorine 僅對 SMX 具有顯著降解效果，推測因自由餘氯與溶解性有機物 (Dissolved Organic Matter, DOM) 發生反應而快速消耗。另一方面，大環內酯類抗生素 Clarithromycin (CLR) 及 Azithromycin (AZM) 於各系統中皆呈現高度難降解特性；而 Sulfamethoxazole 主要代謝物 N-acetyl-sulfamethoxazole (NASMX) 僅能透過 UV 活化系統有效降解，顯示 UV 對特定代謝物去除具有關鍵作用。此外，PDS 系統產生之鹵化消毒副產物濃度顯著低於氯系統，各處理程序亦皆未造成顯著急性細胞毒性，且基因毒性隨反應進行呈下降趨勢。

本研究顯示 PDS 為醫院廢水處理中具發展潛力之替代氧化劑。抗生素及其代謝物於自由基氧化過程中展現不同之反應特性，顯示應依目標污染物特性選擇適當之 PDS 活化策略，以提升污染物去除效率，同時降低消毒副產物生成及毒理風險，作為未來醫院廢水高級處理技術之參考。

關鍵字：抗生素、高級氧化、過二硫酸鹽、去除效率、太陽光驅動

ABSTRACT



To reduce the release of emerging contaminants into environmental waters, recent regulatory actions by Taiwan's Ministry of Environment have tightened hospital effluent standards by imposing limits on ammonia nitrogen and free chlorine, and have introduced new monitoring requirements for antibiotics and other emerging contaminants. Since many antibiotics are poorly removed by conventional biological treatment, and free chlorine is now capped at 2 mg/L to reduce the formation of halogenated disinfection byproducts, alternative oxidants with lower potential for byproduct formation are needed.

This study evaluates peroxydisulfate (PDS) as a candidate oxidant for degrading trace-level antibiotics in hospital wastewater. Two PDS activation strategies (UV/PDS and Solar/PDS) were examined, with Solar/Chlorine included as a benchmark.

Results demonstrated that both the UV/PDS and Solar/PDS systems exhibited effective degradation performance, achieving approximately 50% removal of sulfamethoxazole (SMX) and levofloxacin (LVX) after 10 min (UV/PDS) or 60 min (Solar/PDS) of treatment, consistent with the trends observed in Milli-Q water. In contrast, the Solar/Chlorine benchmark was effective only against SMX, likely due to the rapid consumption of free chlorine by reacting with dissolved organic matter (DOM). Notably, macrolide antibiotics, including clarithromycin (CLR) and azithromycin (AZM), remained highly recalcitrant across all tested systems. Regarding metabolite degradation, UV/PDS was the only process capable of consistently degrading N-acetyl-sulfamethoxazole (NASMX), suggesting that UV activation plays an important role in its degradation. Concurrently, DBP profiling revealed that PDS-based processes generated

significantly lower concentrations of halogenated DBPs compared to chlorine-based AOPs. Toxicological assessments further indicated that while none of the processes induced significant acute cytotoxicity, genotoxicity decreased over the course of treatment.

In summary, these findings demonstrate that PDS is a promising alternative oxidant for hospital wastewater treatment. The distinct reactivity of antibiotics and their metabolites under radical-based oxidation underscores the importance of selecting appropriate PDS activation strategies according to target contaminants, thereby maximizing degradation efficiency while minimizing DBP formation and toxicological risks in practical wastewater treatment applications.

Keywords: Antibiotics, Advanced Oxidation Processes, Peroxydisulfate, Removal Efficiency, Solar-driven

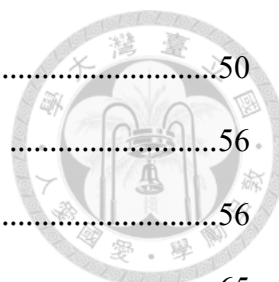
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Chapter 1 Introduction

1.1 Background

Antibiotics, among the Pharmaceutical and Personal Care Products (PPCPs), are used as medical treatments to prevent bacterial infections by killing or suppressing bacterial growth. However, many people overuse and misuse them worldwide, and have been infected with or died from antibiotic-resistant bacterial infections ^[1]. To address the issue, the Taiwan government also has its own monitoring system, known as the Taiwan Healthcare-associated Infection and Antimicrobial Resistance Surveillance System (THAS), which is based on the WHO guidelines.

Aside from human health, the environmental issue should also be considered: improper disposal and incomplete metabolism of the antibiotic in the body can lead to trace concentrations in hospital effluent and even in natural waters, increasing the risk of spreading antibiotic resistance genes and antibiotic-resistant bacteria ^[2-4]. However, traditional wastewater treatment processes, which include secondary treatment and post-chlorination, are still unable to remove or degrade these chemicals.

As a result, Advanced Oxidation Processes (AOPs) have been proposed to degrade these contaminants for decades. The AOPs generate radicals that degrade or mineralize recalcitrant micropollutants ^[5]. Still, the choice of oxidant and activation method remains a significant research focus, with application, energy consumption, and environmental sustainability as key considerations ^[6].

In addition to the emerging contaminant, recent regulatory actions by Taiwan's Ministry of Environment have tightened hospital wastewater effluent standards by imposing limits on ammonia nitrogen and free chlorine, capped at 2 mg/L to reduce the formation of halogenated disinfection byproducts. This further implies that alternative

oxidants with lower potential for byproduct formation are needed.

UV/Chlorine and UV/H₂O₂ are ultraviolet-based advanced oxidation technologies that have been widely studied. The former has the advantages of low cost and high radical-production efficiency, and it also meets the residual-chlorine standards for drinking water, making it widely favored and accepted. Although the combination of ultraviolet light and chlorine in advanced oxidation shows excellent performance in degrading PPCPs, halogenated disinfection by-products (DBPs) are a common issue in the UV/Chlorine process. In contrast, although the latter produces fewer DBPs, it is less selective and more likely to react with organic matter in the water ^[7,8].

The advantage of peroxydisulfate is that the sulfate radicals it generates have high oxidation potentials, similar to those of hydroxyl radicals, and possess longer half-lives ^[9]. In addition, in environments containing humic acid and in carbonate and phosphate buffer solutions, PDS exhibits higher oxidation capacity and better pharmaceutical degradation efficiency; therefore, advanced oxidation systems based on PDS have also attracted considerable attention ^[10].

Solar light is characterized as an environmentally friendly, non-polluting, and continuously available energy source. Therefore, using solar light to replace ultraviolet light as an activation way for oxidant generation in water has become a research focus in recent years ^[11]. The combination of solar light with peroxydisulfate may open a new pathway for wastewater treatment technologies with the ability of degrading trace-level contaminants and reduce the energy inputs.

1.2 Study aims

Despite extensive research into peroxydisulfate (PDS) oxidation mechanisms, existing literature predominantly relies on simplified water matrices (e.g., Milli-Q or RO water), leaving its behavior in complex wastewater and the resulting DBP formation largely unexplored. This study intends to fill that gap by evaluating Solar/PDS alongside a suite of comparative treatments: UV, Solar, UV/PDS, and Solar/Chlorine. By investigating DBP formation within real-world wastewater contexts, this study underscores the potential of solar-driven AOPs. Ultimately, it positions PDS as a sustainable oxidant that meets stringent discharge regulations while safeguarding human health and the environment.

Chapter 2 Literature Review

2.1 Medicine Uses from Taiwan to the World

Since the National Health Insurance system was launched in 1995, healthcare quality in Taiwan has improved, making healthcare more accessible and equitable for those with urgent medical needs ^[12]. According to the National Health Insurance System Statistics, drug expenditures reached 254 billion NTD in 2024, accounting for nearly 30% of the total NHI budget. When it comes to per-person consumption, the NHI covered 10,584 NTDs for medication. Analgesic-antipyretic is the largest proportion, especially Acetaminophen, which remained the most widely used drug by patient count, increasing by 200,000 from the previous year to 9.19 million. Besides, due to an aging society, consumption of the three-hypers series of medicines is gradually increasing. Among these drugs (anti-hypertensives, lipid-lowering agents, diabetes drugs), the diabetes medication Metformin ranked first in total dosage, with over 650 million pills consumed ^[13].

Aside from the analgesic-antipyretic and three-hypers series of medicines, antibiotics are the category being considered, even though consumption is lower than that of the former. Between 2016 and 2023, global consumption of human antibiotics increased from 29.5 billion to 34.3 billion Defined Daily Doses (DDDs), a 16.3% increase, with the growth primarily driven by low- and middle-income countries. Although it declined slightly during the COVID-19 period, it rebounded thereafter. This continuous increase is of significant global concern, as there is a direct link between higher antibiotic consumption and the rising prevalence of multidrug-resistant bacteria ^[14,15].



2.2 Environmental Occurrence and Distribution of Antibiotics

2.2.1 Antibiotic Concentrations in the Aquatic Environment

Antibiotics are now widely recognized as ubiquitous contaminants in aquatic systems, occurring in wastewater, rivers, lakes, groundwater, and even drinking water. Concentrations span many orders of magnitude, from ng/L to $\mu\text{g/L}$, and frequently reach levels relevant to ecological effects and resistance selection ^[3,4,16]. A recent One-Health-oriented review compiled 6305 concentration data for 142 antibiotics in surface and groundwater across 76 countries, reporting levels typically ranging from <1 ng/L to ~ 782 ng/L, with surface waters 7 times more frequently contaminated than groundwater ^[17]. A review of water, sediment, and biota found antibiotics in aquatic media ranging from <1 ng/L to $100 \mu\text{g/L}$, with sulfonamides (30%) and fluoroquinolones (34%) among the most abundant and hazardous classes ^[18]. Focusing on rivers and streams, another synthesis reported mean antibiotic concentrations of 0.4 and $0.9 \mu\text{g/L}$ in Europe and the Americas, respectively, but much higher means of $11.3 \mu\text{g/L}$ in the Asia-Pacific and $17.7 \mu\text{g/L}$ in Africa, reflecting weaker sanitation and treatment infrastructure in many low-income regions ^[19]. Regionally, the Western Pacific and South-East Asia show particularly widespread contamination: 92 antibiotics in the Western Pacific and 45 in South-East Asia were detected in wastewater, WWTP effluents, and natural waters, with many measurements exceeding predicted no-effect concentrations (PNECs) for resistance ^[20].

2.2.2 Sources and Pathways into the Aquatic Environment

Based on estimates for 40 commonly used antibiotics, approximately 31% (about 9500 tons) of the 30300 tons consumed annually enter river systems, with 11% ultimately flowing into oceans or inland catchment areas ^[15]. The pathways of antibiotics can vary; as Figure 2.1 ^[16] shows, the sources can be human or veterinary uses. Hospital effluents



contribute significantly to antibiotic levels in wastewater due to patient excretion and discarded pharmaceuticals ^[21]. Urban households, despite showing lower concentrations, mirror this issue through improper disposal and incomplete metabolism. Consequently, these parent compounds remain intact as they enter wastewater treatment plants (WWTPs) ^[2,22,23].

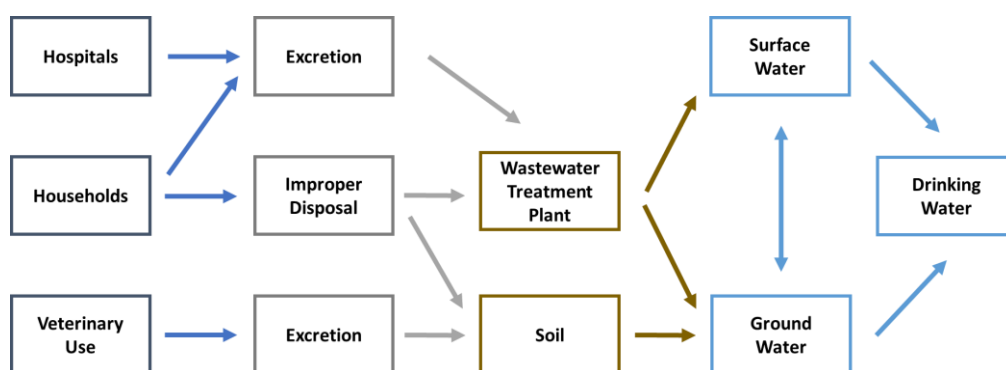


Figure 2.1 Pathways of antibiotics

The traditional wastewater treatment comprises three steps (Figure 2.2^[24]): primary treatment, secondary treatment, and disinfection. In the primary treatment, raw sewage undergoes screening and grit removal to eliminate large debris and inorganic solids, and heavier organic matter settles as sludge in the primary sedimentation tank. The process then transitions to the secondary treatment, where aerobic microorganisms in the aeration tank metabolize dissolved organic pollutants. Following the secondary sedimentation, the clarified water undergoes chlorination to eliminate pathogens. This comprehensive treatment ensures that the final effluent meets the environmental standards before discharge, while the separated sludge is diverted for specialized stabilization and processing ^[22,25].

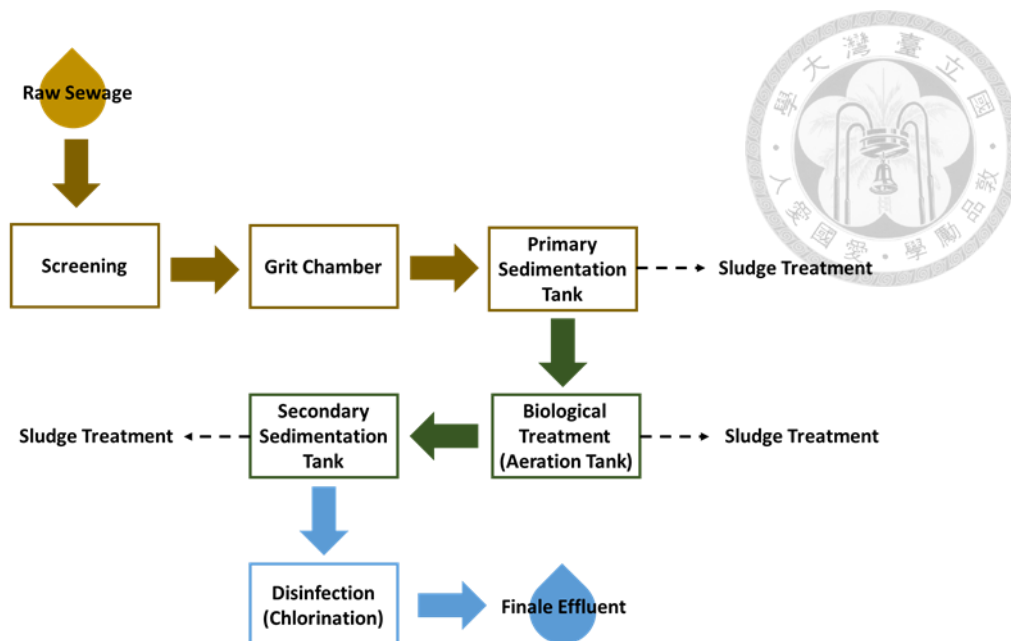
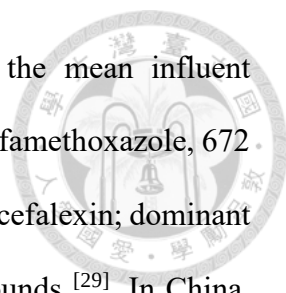


Figure 2.2 Traditional wastewater treatment

However, the traditional WWTP process ^[24] cannot completely remove these compounds. The biological system is designed for removing organic matter and nutrients in domestic wastewater. Chlorination, which serves as a disinfection process, is intended to inactivate microorganisms in the effluent, not to treat these trace contaminants. As a result, many pharmaceutical compounds can be detected in the surface water, groundwater, and even in the drinking water at trace levels ^[4,26], leading to potential human exposure as a consequence ^[24], raising the risk of spreading antimicrobial-resistant genes (ARGs) and antimicrobial-resistant bacteria (ARBs) to the ecosystem ^[27].

Besides, the removal efficiency is highly compound-dependent. Certain compounds exhibit higher removal in the traditional wastewater treatment processes. Research on four secondary WWTPs in Taiwan detected 57 pharmaceuticals in the influent and effluent. While most analgesics, caffeine, and certain estrogens achieved high removal rates of 72–100%, several antibiotics were barely removed ^[28], making WWTP effluents



a primary source of pollutants to receiving waters. In Kuwait, the mean influent concentrations at the Umm Al Hayman WWTP were 852 ng/L for sulfamethoxazole, 672 ng/L for ciprofloxacin, 592 ng/L for clarithromycin, and 491 ng/L for cefalexin; dominant effluent concentrations remained at 93–338 ng/L for several compounds ^[29]. In China, total antibiotic removal ranged from 54.7% to 75.7%, with efficiency positively related to influent concentrations ^[22,30]. In Finnish WWTPs, the treatment process enables efficient removal of sulfonamides, whereas macrolides show negative removal, indicating transformation or deconjugation during treatment ^[31]. In Qatar, penicillin and metronidazole showed nearly 99% removal, while clavulanic acid and ciprofloxacin persisted even after ultrafiltration ^[32].

All the findings indicate that these persistent antibiotics have become a source of contamination in downstream water bodies, highlighting the limitations of traditional secondary treatment in eliminating specific medicinal residues.

2.2.3 Antibiotic Selections for This Study

Hospital wastewater is a major source of pharmaceutical pollution, with antibiotic levels in particular being particularly alarming ^[21]. According to the Taiwan Ministry of Environment (MOENV), levels of critical antibiotics frequently exceed regulatory limits, posing a significant challenge to environmental safety. In 2024, the MOENV finalized a localized priority list for managing the Pharmaceuticals and Personal Care Products (PPCPs) in wastewater. This selection was the culmination of a multi-year program that initially screened 90 potential substances. After excluding 24 items with low domestic usage, 66 substances were rigorously evaluated based on their consumption volume, toxicological data, Risk Quotient (RQ), and international regulatory trends ^[33].

From the assessment, nine key substances were prioritized for localized management

due to their high scores across two consecutive years. This localized list allows Taiwan to focus its regulatory efforts on the substances posing the highest risk to its specific aquatic ecosystems and public health ^[34].

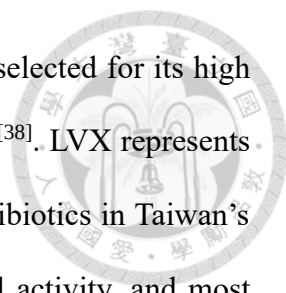


Table 2.1 The new regulated pharmaceuticals in the hospital effluents in Taiwan

Category	Compound Name	Regulatory Limited (mg/L)
Antibiotics	Ofloxacin	0.002
	Ciprofloxacin	0.0017
	Sulfamethoxazole	0.0031
	Erythromycin	0.009
	Clarithromycin	0.00095
	Cefazolin	0.0022
Analgesics	Acetaminophen	4.2
Diabetes Medicine	Metformin	7.8
Hormones	17 beta-Estradiol	0.0035

Aside from regulatory constraints, antibiotic consumption and degradation are the main considerations for representing the real scenario in water. Therefore, in this study, the antibiotics and metabolites chosen are Sulfamethoxazole (SMX) and its metabolite N-acetyl-sulfamethoxazole (NASMX), Trimethoprim (TMP), Levofloxacin (LVX), Clarithromycin (CLR), and Azithromycin (AZM).

SMX and TMP are frequently prescribed as a synergistic combination (co-trimoxazole) for urinary tract and respiratory infections. SMX is known for its high mobility in aquatic environments ^[35]. However, only about 14% of SMX is excreted unchanged, while nearly 50% is metabolized in humans to N-acetyl-sulfamethoxazole (NASMX) ^[36]. Therefore, NASMX was included in this study to evaluate the total environmental burden. Interestingly, a study reported the potential to revert to SMX ^[37], further emphasizing the importance of evaluating both simultaneously. TMP is a



dihydrofolate reductase inhibitor, usually used with SMX, and was selected for its high excretion rate: 80% of the dose is excreted as the active metabolite.^[38] LVX represents the fluoroquinolone class, which is among the most widely used antibiotics in Taiwan's healthcare system, characterized by its broad-spectrum antibacterial activity, and most levofloxacin is excreted unchanged in urine^[39]. CLR and AZM were selected as representatives of the macrolide class. CLR is a key target compound in emerging environmental surveillance and is increasingly regulated under new water quality policies due to its persistence and toxicity to aquatic organisms^[40]. AZM is included not only for its common use in treating respiratory tract infections but also for its significant surge in use during the COVID-19 pandemic, making it a critical marker for assessing the impact of domestic wastewater in the post-pandemic era^[40,41].

2.3 General Review for the AOP Treatments

The issue of PPCPs, of which antibiotics are one of the categories, has been discussed for decades, pointing out that the traditional wastewater treatment process cannot remove antibiotics completely, further indicating the requirement for advanced treatments, such as membrane filtration, granular activated carbon, or advanced oxidation processes, is necessary in the process^[23].

AOPs are a tertiary treatment and have emerged as powerful technologies for removing recalcitrant organic pollutants and pathogens from water and wastewater. The process utilizes highly reactive species to mineralize contaminants, making it suitable for treating industrial effluents, PPCPs, and other emerging contaminants^[5,42,43]. However, large-scale applications of AOP remain relatively rare, and most experiments are conducted at the laboratory scale. To bridge the gap, the shift to engineering of in-situ radical production and the development of sustainable, cost-efficient reactor systems

became necessary ^[43,44]

2.3.1 Current AOP Applications in the Pilot Scale or the Real Field

Recent years have seen significant progress in scaling up AOPs from laboratory to pilot and real-world applications. Pilot-scale demonstrations of sulfate radical-based processes for simultaneous disinfection and removal of contaminants ^[45,46], UV/O₃ pressurization improved decolorization and COD removal in textile wastewater ^[47], UV/Chlorine AOPs have been used at pilot or full scale for water reuse in North America ^[48], which are more energy-efficient and less sensitive to the water matrix compared to UV/H₂O₂, with RCS contributing crucially to oxidation and microbial inactivation ^[49–51].

2.3.2 Comparison of Different AOPs

AOPs encompass a variety of chemical processes that generate reactive oxygen species (ROS), mainly hydroxyl radicals ($\bullet\text{OH}$), through methods such as ozonation (O₃), UV/H₂O₂ photolysis, Fenton reactions (Fe²⁺/H₂O₂), persulfate activation (SO₄^{•−}), photocatalysis (e.g., TiO₂/UV), ultrasound, electrochemical oxidation, and hybrid systems. Each method offers distinct advantages depending on the target contaminants and matrix complexity ^[6,52]. Table 2.3 also compares the removal efficiency of PPCP across various AOPs under different conditions.

Ozone-based AOPs are widely used for both disinfection and micropollutant removal in DWTPs; Fenton-based systems are common for industrial effluents; UV/H₂O₂ is prevalent for trace organic removal; UV/Chlorine AOPs have been used at pilot or full scale for water reuse in North America; sulfate radical-based processes are gaining attraction due to their high redox potential ^[48,51,53–55].

Table 2.2. Different Oxidation-Reduction Potential (ORP) of radicals ^[56]

Oxidant	Average Redox Potential (V)
Fluorine	3.0
Hydroxyl radical ($\bullet\text{OH}$)	2.8
Sulfate radical ($\text{SO}_4\bullet^-$)	2.5-3.1
Ozone (O_3)	2.1
PDS ($\text{S}_2\text{O}_8^{2-}$)	2.1
PMS (HSO_5^-)	1.82
Hydrogen peroxide (H_2O_2)	1.8
Chlorine (Cl_2)	1.4

To date, chlorine-based disinfectants are widely used worldwide. However, they pose a risk of generations of high concentrations of DBPs in the effluent, potentially threatening aquatic life and human health. To address the issue, MOENV in Taiwan also set a cap of 2 mg/L for free chlorine in hospital effluent. It is inevitable to use chlorine in the field because of its outstanding disinfection and sanitation capabilities ^[33]. Therefore, developing an alternative oxidant to address certain recalcitrant compounds seems an urgent need to reduce chlorine dosage.

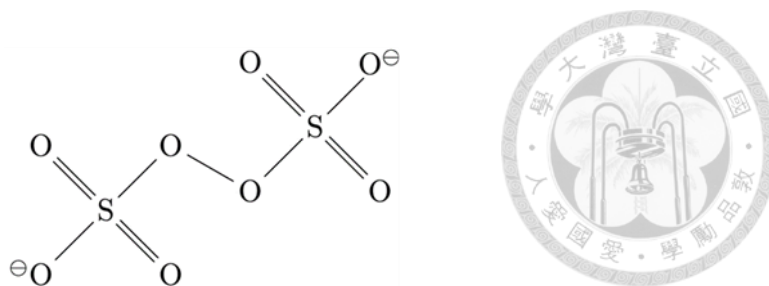


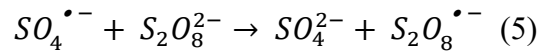
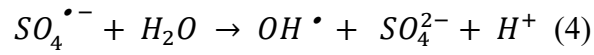
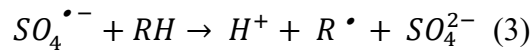
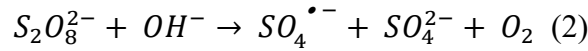
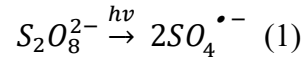
Figure 2.3. Structure of PDS

Peroxydisulfate (PDS) (Figure 2.3) is a strong oxidant ($E^\circ \sim 2.1$ V), highly soluble and stable as a sodium salt. Due to its strong oxidizing ability and long-distance transport capability, PDS has been widely investigated as an oxidant for in situ chemical oxidation (ISCO) [57,58]. It can generate sulfate radicals upon activation. Sulfate radicals have the advantages of their high redox potential ($E^\circ \sim 2.5\text{--}3.1$ V), greater selectivity, and a longer half-life ($3\sim 4 \times 10^{-5}$ s) than that of hydroxyl radicals (2×10^{-8} s). They are less reactive with natural organic matter (NOM). Furthermore, there are various activation methods, including radiation (UV, US) [9,59], heat, alkali, transition metals (homogeneous or heterogeneous), electricity, metal-free carbons, and other catalysts [60,61].

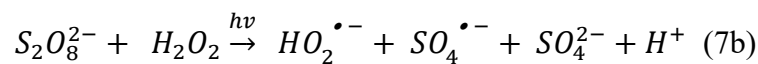
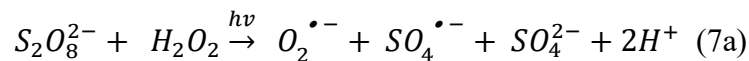
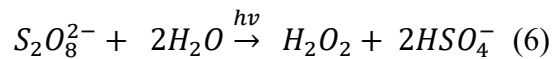
The sulfate radicals can react with contaminants via hydrogen abstraction, electron transfer, and addition to double bonds and aromatic rings, forming oxidized intermediates that eventually yield CO_2 and H_2O , thereby achieving mineralization [62,63]. Aside from the sulfate radical, hydroxyl, superoxide, carbonate radicals, and singlet oxygen can also form in specific situations, thereby modifying pathways and kinetics in water [10,64].

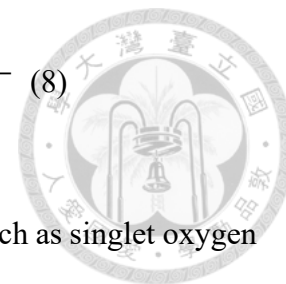
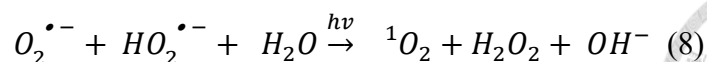
Moreover, studies in NOM solutions or synthetic wastewater at the same oxidant molar concentration showed that UV/PDS produced lower halogenated DBPs than UV/ H_2O_2 and UV/Chlorine, indicating the process might reduce the DBPFP (Disinfection by-product formation potential) of treated water [65,66].

On the other hand, sulfate radicals can also inactivate Gram-positive or Gram-negative bacteria; furthermore, they can reduce ARGs and ARBs by inflicting irreversible oxidative damage on cell envelopes, intracellular components, and nucleic acids [67–69].



All the mechanisms above are derived from Pattnaik et al. [70]. In the UV system, PDS is activated, generating two sulfate radicals (eq. 1). As expected, they will continuously react with pollutants (RH) in the water during the propagation process (eq. 3). These sulfate radicals can further react with water or other substrates to form hydroxyl radicals or other reactive oxygen species (ROS) (eq. 4). At an optimal concentration of PDS, the reaction will proceed via (eq. 2), but at higher concentrations, the scavenger effect will dominate, leading to less effective formation of sulfate radicals (eq. 5).





In some cases, other types of reactive oxygen species (ROS), such as singlet oxygen (1O_2) and superoxide radical ($O_2^{\bullet-}$) can be observed when applying with photocatalysts or under specific pH ^[70–72]. Superoxide itself is less dominant as an oxidant in the removal pathway, but may serve as an intermediate that leads to other ROS. Meanwhile, singlet oxygen formation is increasingly recognized as an important species. Unlike other radicals, it provides a non-radical pathway in AOPs, as shown in equations 6-8 ^[73,74].

Reported rate constants for sulfate radical and target compound reactions typically range from $\sim 1 \times 10^8$ to $> 1 \times 10^{10} \text{ M}^{-1} \text{ s}^{-1}$, depending on the substrate structure and system conditions. These values are still comparable to those for hydroxyl, but may exhibit greater selectivity ^[75,76].

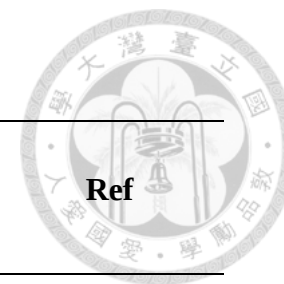
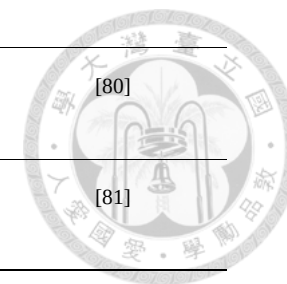
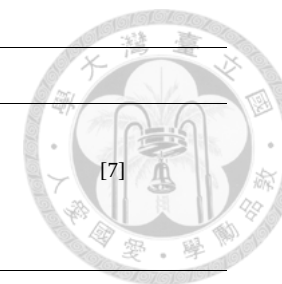


Table 2.3. Comparison of PPCP Removals with Different AOPs

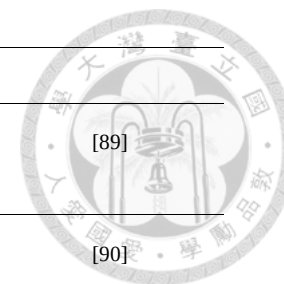
AOP	Target Compound	Initial con.	Time (min)	Dosage	Removal (%)	Ref
Ozonation	Trimethoprim	0.34±0.04 µg/L	18	5 mg/L	>85	[77]
	Sulfamethoxazole	0.62±0.05 µg/L			>92	
	Clarithromycin	0.21±0.02 µg/L			>76	
	Erythromycin	0.62±0.24 µg/L			>92	
	Azithromycin	0.58±0.06 µg/L	10.5	169 L/h 0.8 mg/mg TOC	~80	[59]
	Clarithromycin	0.38±0.06 µg/L				
	Ciprofloxacin	0.15±0.02 µg/L				
	Sulfamethoxazole	1.02±0.08 µg/L				
	Trimethoprim	0.27±0.03 µg/L	10	3.50 × 10 ⁻⁵ mol/L	100	[78]
	Ketoprofen	1.0 × 10 ⁻⁴ mol/L				
Fenton and Fenton-like	Amoxicillin	0.1 mM	1	Solar irradiation (UVA):20 W/m ² H ₂ O ₂ : 5.0 mM	84	[79]
	Trihydrate		5		98	
	Bezafibrate	0.055 mM	5		98	
	Paracetamol	0.1 mM	5		98	



	Ofloxacin	0.028 mM	30	Fe (II): 5 mg/L H ₂ O ₂ : 2.714 mM	100	[80]
		10 mg/L	15	Fe (II): 2 mg/L H ₂ O ₂ : 2.5 mg/L	78.1	[81]
UV/H ₂ O ₂	Sulfamethoxazole	187 ng/L			100	
	Clarithromycin	656 ng/L			<10	
	Levofloxacin	286 ng/L	5	65 W LPUV 7.8 mg/L	~25	[82]
	Trimethoprim	80 ng/L			~10	
	Azithromycin	116 ng/L			<10	
	Sulfamethazine	0.02 mM	45	15 W LPUV 0.2 mM	87.5	[83]
	Sulfamethoxazole	0.024 mM	120	2.8 kW MPUV 1 mM	96	[84]
	Trimethoprim				~60	
	Sulfamethoxazole	20 µM	30	40 W LPUV 20.4 mg/L	>95	[85]
	Ciprofloxacin				>95	
UV/Chlorine	Iopromide	10 µM	6	11 W LPUV 100 µM	78	[86]
	Clofibric acid	46.6 µM	120	40 W LPUV	85.8	[66]



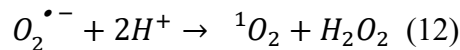
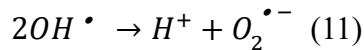
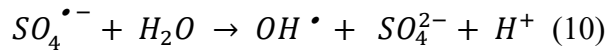
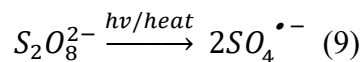
UV/O ₃	1.0 mM					[7]
	Naproxen	2500 ng/L	5	14 W LPUV 75 µM	100	
	Ibuprofen				75	
	DEET				56.5	
	Trimethoprim	20 µM	30	40 W LPUV 42.6 mg/L	~40	[85]
	Sulfamethoxazole				>95	
	Ciprofloxacin				>95	
	Norfloxacin	600 µg/L	10	15 W LPUV 4 mg/L with 1 L/min	100	[87]
	Ofloxacin					
	Roxithromycin					
	Azithromycin					
UV/PDS	Ketoprofen	1.0 × 10 ⁻⁴ mol/L	3	15 W LPUV 3.50 × 10 ⁻⁵ mol/L	100	[78]
	Tetracycline	5 mg/L	30	40 W LPUV 30 mg/L	95.73	[88]
	Sulfamethazine	0.02 mM	45	15 W LPUV 0.2 mM	96.5	[83]
	Sulfamethoxazole	0.024 mM	120	2.8 kW MPUV	97	[84]



			1 mM		
Penicillin G	0.02 mM	90	30 W UV 3 mM	94.28	[89]
Metronidazole	40 mg/L	35	8 W LPUV 2 g/L	99.5	[90]
Iopromide	10 μ M	6	11 W LPUV 100 μ M	31	[86]
Clofibric acid	46.6 μ M	120	40 W LPUV 1.0 mM	88.4	[66]
Ciprofloxacin	10 mg/L	30	8 W LPUV 8mM	95	[91]

2.3.3 Solar-Driven Peroxydisulfate Process

Since UV irradiation is relatively energy-intensive, using sunlight offers a promising advantage by reducing energy consumption, which is an environmentally friendly resource whose spectrum includes UVA, UVB, visible light, and IR [92]. Furthermore, it can inactivate or remove pathogens [93]. Compared with high-energy-demanding UV systems, Solar-driven systems offer an alternative pathway for activation. Solar-driven AOP is a sustainable technology that has attracted considerable interest in the field over the past decade, with a focus on optimizing catalyst design, activation mechanisms, and process integration to enhance pollutant removal efficiency under solar or visible-light irradiation [11,93,94]. A classical one is the heterogeneous photocatalysis of TiO₂; another type is the homogeneous Solar/Oxidant systems, including Solar/H₂O₂, Photo-Fenton, Solar/PDS, and Solar/Chlorine, which are often combined or hybridized to boost efficiency. To provide a performance baseline, Table 2.4 also includes the removal performance of chlorine-based AOPs, without further expanding on their underlying radical mechanisms.



In Solar-driven PDS systems, as in UV-driven systems, not only do they generate sulfate and hydroxyl radicals, but superoxide and singlet oxygen also play roles in the

degradation process ^[95], as shown in equations 9-12.

Although the main absorption wavelength of PDS is in the UV region ^[96], this results in an inefficient quantum yield under solar light. Several studies have focused on photocatalysts to enhance removal efficiency by creating active sites ^[97,98]. Nevertheless, beyond optimal loading, suspended particles increase wastewater turbidity and scatter or block light, so fewer photons reach the catalyst surface, thereby decreasing the reaction rates. Therefore, the process has an optimal catalyst concentration at which light penetration and active sites are balanced, a consideration in wastewater treatment ^[99,100].

The photothermal effect, in which solar light is converted into heat by engineered materials, thereby enhancing persulfate activation and accelerating pollutant degradation. The temperature required to activate PDS is approximately 40°C ^[101]. Under thermal and photo effects, the water temperature in a reactor can rise to 45°C during the summer, eliminating bacteria within 5 hours ^[102,103]. The combination of light absorption and heat generation not only boosts radical production but also broadens the operational spectrum of solar energy utilization, making these systems highly attractive for practical water treatment applications.

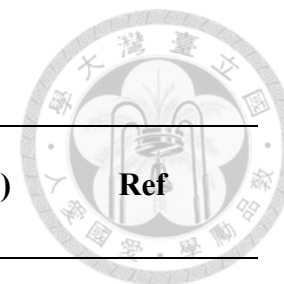
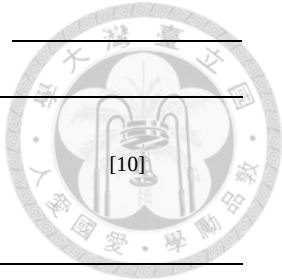


Table 2.4 Solar/Chlorine and Solar/PDS for PPCP removals

AOP	Target Compound	Initial con.	Time (min)	Condition	Removal (%)	Ref
Solar/ Chlorine	Azithromycin, Trimethoprim Sulfamethoxazole Acetaminophen Flumequine	2 µg/ L	20	2 mM phosphate buffer 300 W xenon lamp [Chlorine] ₀ : 1 mg/ L	100	[104]
	Flunixin Meglumine	10 µM	15	NOM solution	86.1	[105]
	Benzoic acid			300 W xenon lamp	69.2	
	Aspirin			[Chlorine] ₀ :	54.1	
	Nitrobenzene			200 µM	43.3	
	Carbamazepine	250 µg/L	10	500 W xenon lamp [Chlorine] ₀ : 3 mg/L pH: 7 *Chlorination for 1hr	51.3	[106]
	Sulfamethoxazole	200 µg/L	3	Natural Sunlight	90	[107]
	Diclofenac			Distilled Water		



Solar/PDS	Carbamazepine		6	[Chlorine] ₀ : 5 mg/L		
	Cephadrine		120	Fresh Urine	90	
	Caffeine	0.2 μM	240	1.5 kW xenon arc lamp	63	[10]
				[PDS] ₀ : 84 μM		
				Milli-Q Water		
	2-Nitrophenol	172.1 mg/L	45	Ag/S/TiO ₂	97.1	[108]
				Natural sunlight		
				[PDS] ₀ : 50 mg/L		
				Milli-Q Water		
	Ofloxacin	10 mg/L	10	300 W Xe lamp	100	[109]
				[PDS] ₀ : 0.1 mM		
	Flunixin Meglumine			NOM solution	97.5	
	Benzoic acid	10.0 μM	15	500 W stimulated	21.8	[110]
	Aspirin			sunlight lamp	72.6	
				[PDS] ₀ : 1.0 mM		
				Simulated Wastewater		
				1.8 kW Xe lamp		
	Levofloxacin	10 mg/L	10	400 W/m ²	93.1	[111]
				[PDS] ₀ :		
				200 μM		

				300W Xe lamp [PDS] ₀ : 1 mM *[CB] = 0.1 g/L pH 7.0 *Reaction with photocatalysts	100	 [112]
				Simulated Groundwater	91	
Sun/PDS	Acid Orange 7			System temperature: 42 ± 3 °C	89	
	Acetaminophen	2 mg/L	240	Ambient Temperature: 38 °C	62	
	Malachite Green			[PDS] ₀ : 600 μM		

2.3.4 Toxicity

Although advanced oxidation processes (AOPs) are highly efficient at degrading target organic contaminants, the complete removal of parent compounds does not guarantee reduced effluent toxicity ^[113]. Partial oxidation often yields transformation products (TPs) or intermediate oxidation products that can retain, alter, or even exacerbate biological toxicity relative to pristine contaminants ^[114,115]. Specific intermediates, such as nitro-compounds and halogenated disinfection by-products (DBPs), pose critical environmental hazards because they often exhibit higher toxicity than their parent compounds ^[116].

A study compared different UV-based photochemical processes under identical conditions, except for the oxidant ^[117]. Among the various systems, the UV/H₂O₂ process exhibited the lowest residual toxicity, with overall effluent toxicity increasing in the following order: UV/H₂O₂ < UV/PDS < UV < UV/PMS < UV/Chlorine. Although sulfate radicals (SO₄•-) and reactive chlorine species (RCS) often achieve superior target contaminant removals compared to hydroxyl radicals (•OH), their respective systems generated significantly higher secondary toxicity. This discrepancy suggests that, while sulfate and chlorine-derived radicals accelerate parent compound degradation, they concurrently promote the formation of more toxic TPs and halogenated DBPs. In contrast, the hydroxyl radical (•OH) mediated system yields intermediate compounds with negligible toxicity.

Nevertheless, different pharmaceutical compounds possess distinct chemical properties, leading to highly variable degradation pathways and toxicological outcomes. Regarding antibiotics specifically, the literature has demonstrated that while AOP treatments can eliminate antibacterial activity, they often fail to reduce the overall acute or chronic toxicity of the water matrix. Such studies frequently observe a dynamic profile

in which toxicity initially intensifies due to the accumulation of toxic TPs, followed by a decline as these intermediates undergo further oxidation. Consequently, optimizing the AOP reaction time is paramount for achieving true toxicity reduction rather than mere disappearance of the parent compound.

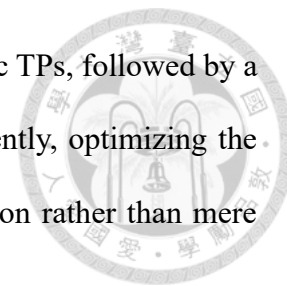
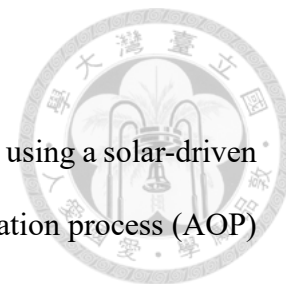


Table 2.5 Toxicity changes of different contaminants and reaction conditions

Contaminant	Reaction Condition	Bacteria	Toxicity Change	Ref
Sulfamethoxazole	[SMX] = 0.5 mM [PDS] = 10 mM 90 °C Time: 60 min	<i>Vibrio fischeri</i>	Increased first and then decreased	[118]
	[SMX] = 0.1 mM [PDS] = 2 mM UV240 W Time: 5 min	<i>Chlorella vulgaris</i>	Slight increase	[119]
	[SMX] = 0.1 mM [NaOCl] = 2 mM, UV 240 W Time: 5 min		Increase	
Metronidazole	[Metronidazole] = 10 mg/L UV 32W [PDS] = 100 mg/L [TiO ₂] = 2.5 g/L Time: 15 min	<i>Vibrio fischeri</i>	Increase	[120]
Caffeine	[CFD] = [CAF] = 0.2 mM 700 W/m ² 1.5 kW Xenon Lamp	<i>Vibrio fischeri</i>	No change	[10]
Cephadrine	[PDS] ₀ = 8.4 mM Time: 360 min			
Sulfamethoxazole	[SMX] = [TMP] = 100 µM, [PDS] = 3 mM Time: 6 min	<i>Vibrio qinghaiensis</i>	No antimicrobial property 20–40% higher	[121]
Trimethoprim				
Propranolol	[Propranolol] = 0.2 mM	<i>Vibrio</i>	Increase	[120]

	UV 4W [PDS] = 10 mM	<i>qinghaiensis</i>	first and then decrease	
Levofloxacin	[LVX] = 10 mg/L 400 W/m ² 1.5 kW Xenon Lamp [PDS] = 0.4 mM Time: 10 min	<i>E. coli</i>	Loss of antibacterial activity	[111]
Ribavirin	[RBV] = 50 mg/L UV500 W [H ₂ O ₂] = 1‰ [TiO ₂] = 50 mg/L Time: 120 min	Zebrafish	Increase first and then decrease	[122]
Aspirin	[ASA] = [FMME] = [NB] = [BA] = 10.0 μM 6.06 W/cm ² 500 W Xenon Lamp [PDS] = 1 mM Time: 15 min	Microalgae	Decrease	[110]
Flunixin				
Meglumine				
Nitrobenzene				
Benzoic Acid				
Carbamazepine	[CPZ] = 0.25 mM 500 W xenon lamp [Chlorine] = 3 mg/L Time: 10 min	<i>Aliivibrio fischeri</i>	Increase	[106]

Chapter 3 Materials and Methods



The aim of this study is to evaluate antibiotic removal efficiency using a solar-driven peroxydisulfate (PDS) oxidation process. Prior to the advanced oxidation process (AOP) experiments, method validation was conducted, including residual oxidant analysis, optimization of UPLC–MS/MS parameters, and recovery tests for the solid-phase extraction (SPE) procedure. After confirming the reliability of all analytical methods, the AOP experiments were performed using a bench-scale reactor. Figure 3.1 illustrates the overall research framework of this study.

In both Milli-Q water and post-secondary treated wastewater effluent, a series of AOPs were evaluated, including UV irradiation alone, simulated sunlight alone, and UV/PDS, Solar/PDS, and Solar/Chlorine systems operated at two oxidant concentrations (30 and 50 μM in Milli-Q water, and 30 and 60 μM in post-secondary treated wastewater). In both water matrices, antibiotic concentrations were determined using UPLC–MS/MS, while residual oxidant concentrations were monitored using a UV–Vis spectrophotometer.

The SOS ChromoTest was employed to evaluate cytotoxicity and genotoxicity. However, because the complex composition of post-secondary treated wastewater could interfere with the interpretation of toxicity changes associated with antibiotic degradation, toxicity assessments were initially conducted only in Milli-Q water. For the post-secondary treated wastewater experiments, additional water quality parameters were analyzed, including disinfection by-product (DBP) formation using GC–ECD and natural organic matter, expressed as non-purgeable dissolved organic carbon (NPDOC), using a TOC analyzer.

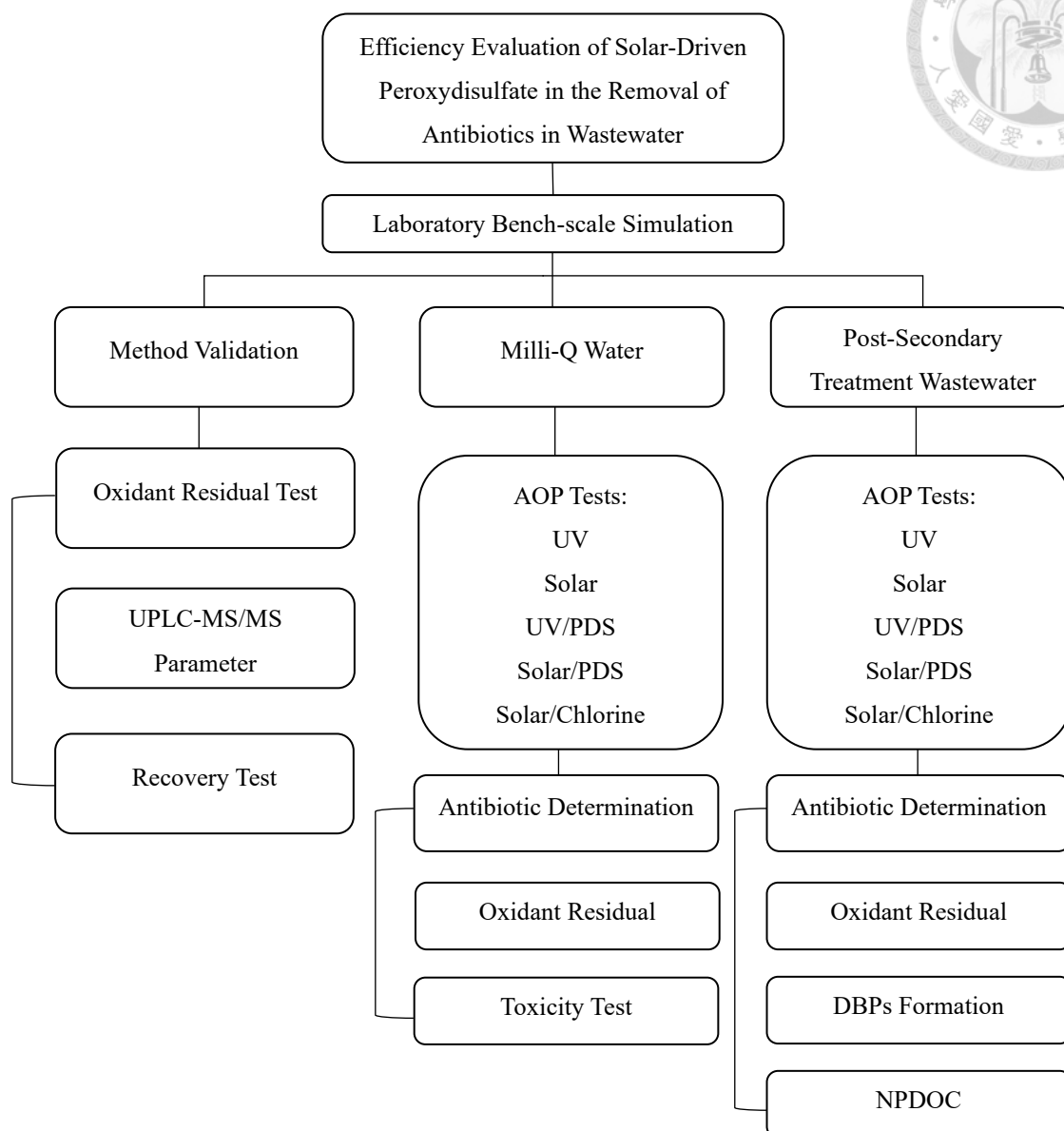
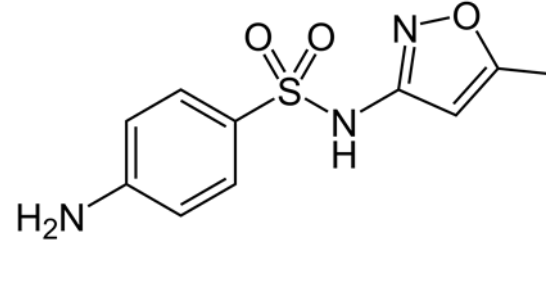
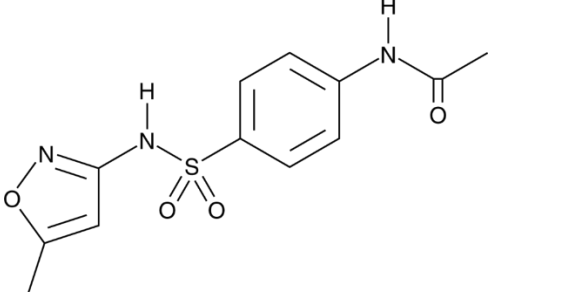
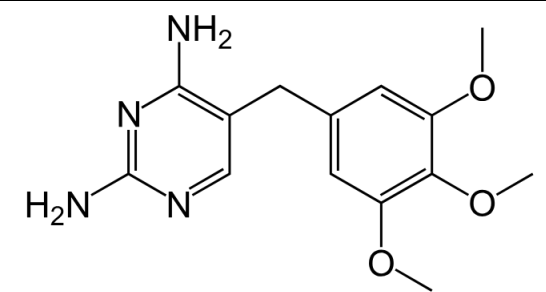
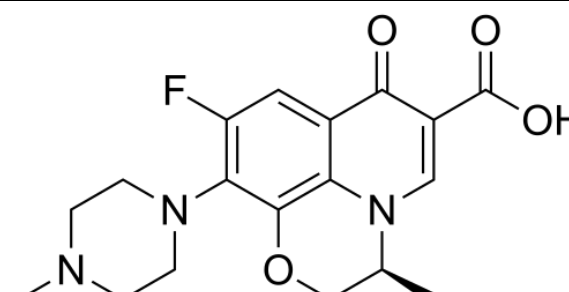
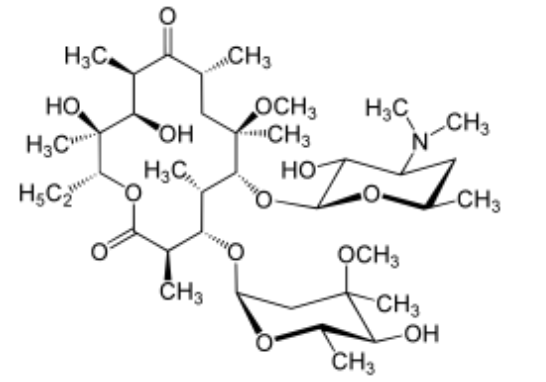
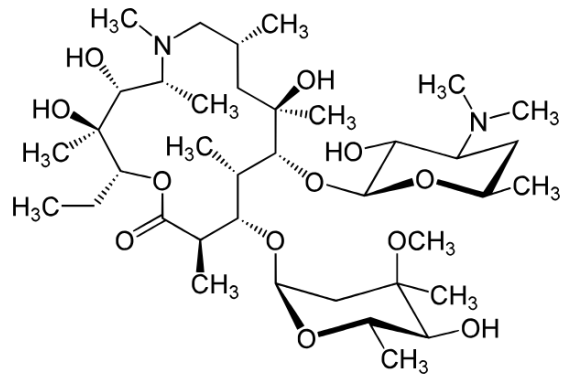


Figure 3.1 Research Framework of this Study

3.1 Chosen of the antibiotics

The study selected Sulfamethoxazole (SMX), its metabolite, N-acetyl Sulfamethoxazole (NASMX), Trimethoprim (TMP), Levofloxacin (LVX), Clarithromycin (CLR), and Azithromycin (AZM). All target compounds were of analytical grade, with comprehensive details of their manufacturers presented in the Appendices.

Table 3.1 Structural Formula of the Antibiotics Tested in this Study

	
Sulfamethoxazole	N-acetyl Sulfamethoxazole
	
Trimethoprim	Levofloxacin
	
Clarithromycin	Azithromycin

3.2 Sample Collection and Preparation

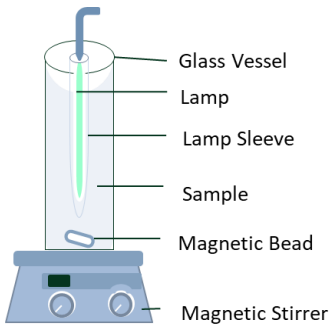
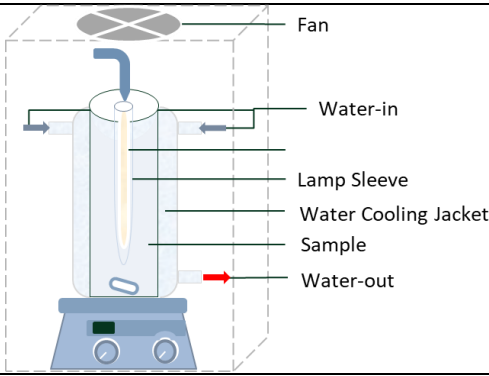
The wastewater matrix used in this study was the post-secondary-treatment effluent, unchlorinated, collected from the Neihu Wastewater Treatment Plant (WWTP) in Taipei. This plant is a second-largest WWTP in Taiwan, treating nearly 240,000 tons of domestic wastewater per day from areas including the Neihu and Dazhi districts. Sampling was conducted in January and March 2026. After collection, the wastewater samples were stored at 4°C in a refrigerator before AOP experiments and subsequent analyses. All the post-secondary treatment water was filtered by a 5 µm filter paper and a 1 µm glass microfiber filter, ensuring that large particles in the wastewater were removed.

3.3 Process of AOP experiments

3.3.1 AOP system set-up

The experimental setup consisted of a glass reactor, a lamp, a lamp sleeve, a magnetic bead, and a magnetic stirrer. Different light sources were paired with corresponding lamp sleeves. The UV-based experiments were conducted using a 14 W low-pressure mercury lamp emitting monochromatic radiation at 254 nm, enclosed in a quartz sleeve. In contrast, the sunlight-based experiments employed a long-arc xenon lamp to simulate natural sunlight, housed in a 7740 Pyrex glass sleeve. Additionally, the sunlight-based system was equipped with a fan and an air pump to supply cooling air to the lamp sleeve and to circulate water, thereby maintaining a stable system temperature. A water-cooling jacket was also incorporated to regulate temperature. The reactor was filled with 2.5 L of water spiked to 3 ng/mL, representing the upper range of pollutant levels observed under real-world conditions. Before a series of examinations, the irradiation or intensity of the UV and Solar systems would be measured first, and the UV lamp would be warmed up for 15 minutes to stabilize. Table 3.2 and Figure 3.2 show the characteristics of the light sources and the configuration of the AOP reactor.

Table 3.2 AOP set-up of different light sources

	UV-based	Solar light
Light Sources	14 W low-pressure lamp (254nm wavelength)	Long-arc Xenon lamp 500W
Lamp Sleeves	Quartz	7740 Pyrex glass
Experimental Set-ups		

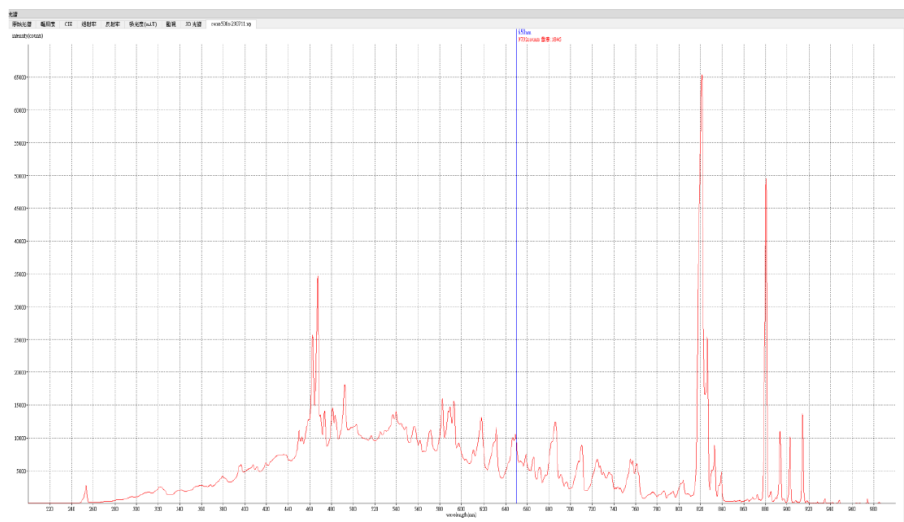


Figure 3.2 The spectrum of the solar simulator
(Prosper OptoElectronic Co., Ltd).



3.3.2 Reagents

1. Sodium peroxydisulfate ($\text{Na}_2\text{S}_2\text{O}_8$), ACS graded, $\geq 99.0\%$, Sigma-Aldrich, St. Louis, MO, USA
2. Sodium hypochlorite solution (NaOCl), 6-14% active chlorine basis, Honeywell, Charlotte, NC, USA
3. L (+)-Ascorbic acid, 100%, ACS grade, Honeywell, Charlotte, NC, USA
4. Sodium sulfite anhydrous, 98-100%, Honeywell, Charlotte, NC, USA

3.3.3 Solution preparations

1. 20 mM of $\text{Na}_2\text{S}_2\text{O}_8$ stock solution: prepared by dissolving 0.4762 g of the $\text{Na}_2\text{S}_2\text{O}_8$ salt in 100 mL of Milli-Q water. The solution was stored in an amber Duran bottle at 4°C .
2. 2000 mg/L of free chlorine stock solution: dilute the original solution and store in an amber Duran bottle at 4°C .

3.3.4 Procedure of the AOP experiment

1. Load a 2.5 L water sample (Milli-Q water or post-secondary treatment water) into the glass vessel.
2. Add the antibiotics standards in the water to form 3 ng/mL.
3. Wait for 10 minutes to ensure it is well-mixed.
4. Start the AOP experiments (For the AOP experiments with oxidants, add the oxidants first, then turn on the light sources.).
 - (1) UV
 - (2) Solar light
 - (3) UV/30 μM PDS
 - (4) UV/50 μM PDS in Milli-Q; UV/60 μM PDS in wastewater
 - (5) Solar/30 μM PDS

- (6) Solar/50 μM PDS in Milli-Q; Solar/60 μM PDS in wastewater
- (7) Solar/30 μM Chlorine
- (8) Solar/50 μM Chlorine in Milli-Q; Solar/60 μM Chlorine in wastewater



The total reaction time for the UV, solar light, Solar/PDS, and Solar/Chlorine groups was 60 min, with samples collected at 0, 10, 30, and 60 min. For the UV/PDS group, the total reaction time was 10 min, and samples were collected at 0, 3, 5, and 10 min. For the antibiotic analysis, the oxidants or radicals were quenched with ascorbic acid, whereas for the NPDOC and DBPs analyses, they were quenched with sodium sulfite.

3.4 Oxidant Residual Test

3.4.1 Residual of PDS

The residual concentration of PDS will be determined using modified iodometric spectrophotometry ^[123]. The concentration of the KI solution and the waiting time of the reaction are derived from the reaction kinetics of the study.

1. Reagents and Equipment

- (1) Potassium iodide (KI), $\geq 99.5\%$, UniRegion Bio-Tech, Taipei, Taiwan
- (2) Sodium bicarbonate (NaHCO_3), $\geq 99.7\%$, ACS grade, Sigma-Aldrich, St. Louis, MO, USA
- (3) UV–Vis spectrophotometer, UV-1800, Shimadzu, Japan
- (4) Two quartz cuvettes

2. Preparation of 2.4 KI_(aq) 100 mL

- (1) Weigh out 1.4 g of NaHCO_3 and 40 g of KI.
- (2) Put them into the brown serum bottle.
- (3) Add 100 mL Milli-Q to the bottle.
- (4) Tighten the lid of the bottle to reduce oxidation of the KI_(aq).

3. Procedure



- (1) Check the blank of KI (aq) before measuring the residual.
- (2) Take 100 μ L water sample, adding 2.9 mL of 2.4 M KI (aq) to the vial.
- (3) Wait for 45 seconds to reach equilibrium in the dark.
- (4) Transfer the solution to a cuvette, and measure it under a 352 nm wavelength.

3.4.2 Residual of free chlorine

1. Equipment

- (1) Pocket Colorimeter II, HACH, Loveland, Colorado, USA
- (2) DPD free chlorine reagent, Loveland, Colorado, USA
- (3) A vial with a 10mL mark

2. Procedure

- (1) Add the water sample to the vial, ensuring it reaches the 10 mL mark.
- (2) Autozero before adding DPD free chlorine reagent.
- (3) Add DPD free chlorine reagent to the sample, shake the vial until equilibrium.
- (4) Put the vial into the colorimeter, pressing the measure button to obtain the residual of free chlorine.

3.5 Antibiotic analysis

3.5.1 Equipment and reagents

1. UPLC I-Class/Xevo TQ-XS, Waters Acquity, Milford, MA, USA
2. SAVANT SPD1010 SpeedVac concentrator, Thermo Fisher Scientific, Waltham, MA, USA
3. Poroshell 120 EC-C18 column, 50 x 2.1 mm, 2.7 μ m, Agilent Technologies, Santa Clara, CA, USA
4. Methanol (MeOH), LC-MS grade, Duksan Pure Chemicals, Ansan, Gyeonggi-do, South Korea
5. Formic acid (FA), ACS grade, 88.0%, J.T. Baker, Phillipsburg, NJ, USA

6. Ammonium acetate (NH_4Ac), $\geq 99.0\%$, LC-MS grade, Sigma-Aldrich, St. Louis, MO, USA



3.5.2 Solid Phase Extraction (SPE)

The SPE procedure was adapted from the acid fraction of the US EPA Method 1694. For each 100 mL sample collected from the AOP experiments, internal standards were added, and the pH was adjusted to 2.7–3.0 using formic acid. The mixture was thoroughly homogenized before being split into two 50 mL subsamples. Table 3.3 shows the extraction processes, which were performed using Oasis HLB cartridges (6 cc, 200 mg) as follows:

1. Conditioning: The sorbent was conditioned with two 5 mL aliquots of methanol, followed by 5 mL of Milli-Q water and 5 mL of 0.1% formic acid in Milli-Q water.
2. Loading: Samples were loaded onto the cartridges at a controlled flow rate.
3. Washing: The sorbent was washed with 10 mL of Milli-Q water and then air-dried for 10 minutes.
4. Elution: Analytes were eluted with two 3 mL aliquots of 0.1% formic acid in methanol, followed by two 3 mL aliquots of pure methanol.
5. Concentration: The resulting 12 mL eluate was concentrated to a final volume of 0.5 mL using a SpeedVac concentrator at 50°C , so all the concentration in 0 min of analytes would be 300 ng/mL.

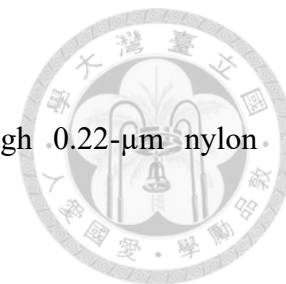
Table 3.3 SPE Processes in the Present Study



Oasis HLB 6cc 200mg	Soak (min: sec)	Air drying (min: sec)
Condition		
5 mL MeOH	03:00	00:00
5 mL MeOH	03:00	00:00
5 mL Milli-Q water	03:00	00:00
5 mL 0.1%FA/Milli-Q water	03:00	00:00
Loading samples	15:00	
Wash		
10 mL Milli-Q water	03:00	
Air drying		10:00
Elution		
3 mL 0.1%FA/MeOH	03:00	
3 mL 0.1%FA/MeOH	03:00	01:30
3 mL MeOH	03:00	
3 mL MeOH	03:00	02:00

3.5.3 UPLC–MS/MS Analysis of the Targeted Pharmaceuticals

Before UPLC–MS/MS analysis, samples were filtered through 0.22- μ m nylon membranes and transferred into 2 mL deactivated glass vials.



1. Preparation of the solutions

(1) Mobile phase solvent

- A. Milli-Q water
- B. 0.1% formic acid with 2 mM ammonium acetate in Milli-Q water as mobile phase A
- C. LC–MS grade methanol as mobile phase B

(2) Wash solvent

- A. 10% LC–MS grade methanol in Milli-Q water as a weak wash solution
- B. 100% LC–MS grade methanol as a strong wash solution

2. UPLC–MS/MS parameters

Table 3.4 shows the tuning parameters of MS. The mass spectrometer was operated in positive electrospray ionization (ESI+) mode. The source and desolvation temperatures were maintained at 150°C and 450°C, respectively. Nitrogen was used as the drying gas, with a cone gas flow of 150 L/hr and a desolvation gas flow of 800 L/hr. As for the LC part, Table 3.5 shows that the separation column is an EC-C18 column with dimensions of 50 $\times$ 2.1 mm and a particle size of 2.7 μ m. The column temperature was maintained at 35°C, the flow rate was set at 0.5 mL/min, and the sample injection volume was 2 μ L. Elution was performed with a linear gradient as follows: 0–0.5 min, 5% B; 0.5–4.0 min, 5% to 95% B; 4.0–4.5 min, 95% B; and finally returning to 5% B at 4.7 min to re-equilibrate the column until 6.0 min. Table 3.6 summarizes the optimized MS/MS parameters for the target analytes and their corresponding internal standards.

Table 3.4 Tuning Parameters of MS

	ESI+
Source temperature (°C)	150
Desolvation temperature (°C)	450
Cone gas flow (L/hr.)	150
Desolvation gas flow (L/hr.)	800



Table 3.5 Gradient of the LC

	ESI+	
Column	EC-C18 column (50 × 2.1 mm, 2.7 μm)	
Column temperature (°C)	35	
Flow rate (mL/min)	0.5	
Injection volume (μL)	2	
Mobile phase	A: 0.1% Formic Acid _(aq) with 2 mM Ammonium Acetate B: Methanol	
Gradient (min)	A (%)	B (%)
Initial	95	5
0.5	95	5
4	5	95
4.5	5	95
4.7	95	5
6	95	5

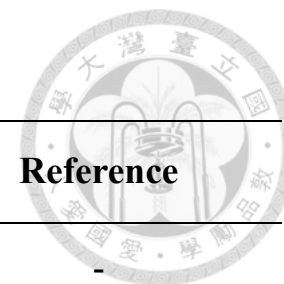


Table 3.6 Analyte Parameters in the MS

Compound	Cone Voltage (V)	Ion Transition (Collision Voltage, V)	Internal Standard	Reference
Sulfamethoxazole	20	(+)254 > 156 (10) (+)256 > 108 (20)	Sulfamethoxazole-D ₄	-
N-Acetyl- Sulfamethoxazole	30	(+)296.7 > 198 (15) (+)296.7 > 134 (25)		[27]
Sulfamethoxazole-D ₄	35	(+)258 > 160 (15) (+)258 > 112 (25)	-	-
Trimethoprim	35	(+)291 > 230 (25) (+)291 > 261 (25)	Trimethoprim-D ₉	-
Trimethoprim-D ₉	45	(+)300.1 > 234.04 (25) (+)300.1 > 264 (25)	-	-
Levofloxacin	35	(+)362.15 > 318.2 (20) (+)362.15 > 261.1 (25)	Levofloxacin-D ₈	-
Levofloxacin-D ₈	30	(+)370.2 > 265.1 (25)	-	-
Azithromycin	30	(+)749.5 > 591.2 (30) (+)749.5 > 158 (40)	Clarithromycin- ¹³ C-D ₃	[124]
Clarithromycin	45	(+)748.5 > 590 (20) (+)748.5 > 158 (30)		-
Clarithromycin- ¹³ C-D ₃	35	(+)752 > 593.4 (20)	-	-

3.6 Non-purgeable dissolved organic carbon (NPDOC) Analysis

In addition to antibiotic concentration analysis, NPDOC was also analyzed in the study to predict disinfection byproducts and precursors. All the samples were added sodium sulfite as a quencher and stored in a 40 mL amber vial at 4°C for further analysis.

3.6.1 Equipment and reagents

1. Wet oxidation total organic carbon (TOC) analyzer, Aurora Model 1030W, OI Analytical, USA
2. Potassium hydrogen phthalate (KHP), Merck, Darmstadt, Germany
3. Phosphoric acid (H_3PO_4), 85%, J.T. Baker, Phillipsburg, NJ, USA
4. Sodium peroxydisulfate ($\text{Na}_2\text{S}_2\text{O}_8$), ACS graded, $\geq 99.0\%$, Sigma-Aldrich, St Louis, MO, USA
5. MCE membrane, pore size $0.45\mu\text{m}$, Chrom Tech, Minnesota, USA

3.6.2 Preparation of the reagent

1. Calibration curve (0.625 mg/L - 10 mg/L): Dissolve 2.128 g of KHP in 1 L Milli-Q water to prepare the stock solution of 1000 mg C/L and dilute it to 10 mg C/L .
2. $5\% \text{ H}_3\text{PO}_4$ in Milli-Q water
3. $10\% \text{ Na}_2\text{S}_2\text{O}_8$ in Milli-Q water

3.6.3 Procedures

1. All the samples were filtered by a $0.45\mu\text{m}$ MCE membrane before further analysis.
2. The TOC analyzer analyzed all the samples within 7 days.

3.7 Disinfection By-Products (DBPs) Analysis

The measurement of disinfection byproducts is also one of the assessment elements in the study; the methods below are referred to as Method 551.1 for THMs and HANs, and Method 552.3 for HAAs by USEPA. For volatile DBPs, the sample should be headspace-free. After extracting, all the samples were analyzed within 7 days.

3.7.1 Equipment and reagents

1. Gas chromatography coupled with Micro-ECD (GC-ECD), Model 6890N, Agilent Technologies, CA, USA
2. GC-ECD column, DB1701, 30 m × 0.25 mm, 0.25 μm, Agilent Technologies, CA, USA
3. Sodium sulfate anhydrous (Na₂SO₄), ≥99.3%, granular, J.T. Baker, Phillipsburg, NJ, USA
4. Trihalomethanes calibration mixture, 2000 μg/L in methanol, SUPELCO, Bellefonte, PA, USA
5. Halogenated volatiles mixture, 2000 μg/L in acetone, SUPELCO, Bellefonte, PA, USA
6. Haloacetic acid mixed standard, 2000 μg/L in methanol, SUPELCO, Bellefonte, PA, USA
7. 1, 2, 3-trichloropropane, 1000 mg/L in MTBE, AccuStandard, New Haven, CT, USA
8. Methyl tert-butyl ether (MTBE), HPLC grade, ≥99.8%, Honeywell, Charlotte, NC, USA
9. Sodium bicarbonate (NaHCO₃), ACS grade, ≥99.7%, Sigma-Aldrich, St. Louis, MO, USA
10. Methanol (MeOH), HPLC grade, ≥99.9%, J.T. Baker, Phillipsburg, NJ, USA

11. Sulfuric acid (H_2SO_4), ACS grade, 95-98%, Sigma-Aldrich, St. Louis, MO, USA

3.7.2 Trihalomethanes (THMs) and Haloacetonitriles (HANs)

1. Solvent preparations

- (1) 1.00 $\mu\text{g/mL}$ of 1, 2, 3-trichloropropane in MTBE as internal standard

2. Procedures

- (1) Equilibrate samples at room temperature.
- (2) Take 30mL of each water sample to a 40 mL glass vial for extraction.
- (3) Add 3 mL of MTBE containing the internal standard to the sample
- (4) Add enough Na_2SO_4 inside, put the Teflon-coated cap back, and shake it vigorously immediately until most of the Na_2SO_4 dissolves to a saturated state.
- (5) Wait for the organic and aqueous layers to separate.
- (6) Transfer approximately 1–1.5 mL of the organic layer into a 1.8 mL amber vial using a glass dropper, ensuring no water is included.
- (7) Store at 4°C for further analysis.
- (8) Analyzed by GC–ECD within one week.

3.7.3 Haloacetic Acids (HAAs)

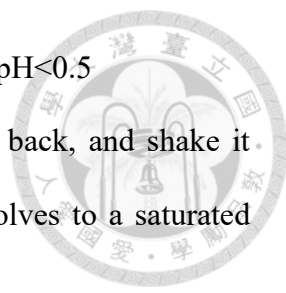
1. Solvent preparations

- (1) 1.00 $\mu\text{g/mL}$ of 1, 2, 3-trichloropropane in MTBE as internal standard
- (2) 150 g/L Na_2SO_4 solution
- (3) Saturated NaHCO_3 solution
- (4) 10% Sulfuric acid in methanol (10% $\text{H}_2\text{SO}_4/\text{MeOH}$)

2. Procedures

- (1) Equilibrate samples at room temperature.
- (2) Take 30 mL of each water sample to a 40 mL glass vial for extraction.



- 
- (3) Add 1.5 mL of sulfuric acid to the sample to adjust it to pH<0.5
 - (4) Add enough Na₂SO₄ inside, put the Teflon-coated cap back, and shake it vigorously immediately until most of the Na₂SO₄ dissolves to a saturated state.
 - (5) Add 3 mL of MTBE containing the internal standard to the sample, shaking it vigorously.
 - (6) Wait for the organic and aqueous layers to separate.
 - (7) Transfer 2 mL of the organic layer to a 15 mL glass centrifuge tube
 - (8) Add 2 mL of 10% H₂SO₄/MeOH to the glass centrifuge tube, put the Teflon cap on, and vortex.
 - (9) Water bath at 50 °C for 2 hours, wait for complete methylation.
 - (10) Cool down samples at the room temperature. Add 5 mL of Na₂SO₄ solution, vortex, and remove the aqueous layer until the remaining volume is less than 0.3 mL.
 - (11) Add 2 mL of the NaHCO₃ solution and vortex to neutralize the sample.
 - (12) Transfer approximately 1mL of the organic layer into a 1.8 mL amber vial using a glass dropper, ensuring no water is included.
 - (13) Store at 4 °C for further analysis.
 - (14) Analyzed by GC–ECD within one week.

3.8 Toxicity Assay

Given the potential genotoxicity of the selected antibiotic, the mechanisms of action of SMX/TMP and LVX involve DNA, particularly LVX, which interferes with bacterial DNA by inhibiting DNA gyrase and topoisomerase IV ^[125]. To evaluate cytotoxicity and genotoxicity, the SOS ChromoTest, a rapid diagnostic kit provided by EBPI, was employed. Besides, the advantage is that completion takes only one day. The sample pretreatment methodology is adapted from Zheng et al. ^[126].

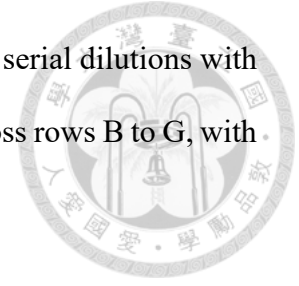
1. Equipment and Reagents

- (1) SOS ChromoTest Kit (containing lyophilized *Escherichia coli* PQ37), EBPI, Mississauga, ON, Canada
- (2) SpectraMax i3x multi-mode microplate reader, Molecular Devices, San Jose, CA, USA

2. Sample Pretreatment

- (1) Quench the water sample, ensuring there is no oxidant residual.
- (2) SPE from the HLB Cartridge (the steps of the SPE are the same as Table 3.3)
- (3) Dry the sample
- (4) Dissolve the sample with 30 μ L 10% Dimethyl sulfoxide (DMSO) in saline (equal to a concentration of 16.66 times, T_0 : 5 mg/L theoretically)
- (5) Prepare 10 μ g/mL 4-Nitroquinoline 1-oxide (4-NQO) in 10% DMSO in saline as a positive control, 10% DMSO in saline as a negative control
- (6) Column 2 serves as the positive control, column 3 as the negative control, columns 4 and 5 as T_0 , columns 6 and 7 as T_3 or T_{10} , columns 8 and 9 as T_5 or T_{30} , and columns 10 and 11 as T_{10} or T_{60} . Row H is designated as the machine blank.
- (7) Transfer 20 μ L of the sample, including the positive control, into row A of the

96-well plate (row A as 100%). Subsequently, perform six serial dilutions with DMSO (50%, 25%, 12.5%, 6.25%, 3.125%, 1.5625%) across rows B to G, with each well ultimately containing 10 μ L of the sample.



3. Incubation of Bacteria

- (1) Incubate the bacteria at 37 °C at 90 rpm for 15 hr
- (2) Measure the OD₆₀₀=0.15-0.2 (OD: Optical Density)
- (3) Dilute it with the medium to make the OD₆₀₀=0.05
- (4) Transfer 75 μ L of the bacteria to each well except the machine blank (row H)
- (5) Incubate the sample at 37 °C for 2.5 hr

4. Coloration

- (1) Prepare blue chromogen containing alkaline phosphatase, which is for testing β -galactosidase (β -gal) and alkaline phosphatase (AP)
- (2) After the incubation, transfer 75 μ L of the blue chromogen to each well
- (3) Incubate for 90 minutes at 37 °C

5. Plate Reading

- (1) Read absorbance at 420 nm to determine viability, and 600 nm for genotoxicity
- (2) The equations are below:

$$\text{Survival Rate \%} = \frac{\text{Sample OD}_{420\text{nm}} - \text{Blank OD}_{420\text{nm}}}{\text{Negative Control OD}_{420\text{nm}} - \text{Blank OD}_{420\text{nm}}} \times 100\%$$

$$\text{Growth Factor (G)} = \frac{\text{Sample OD}_{420\text{nm}} - \text{Blank OD}_{420\text{nm}}}{\text{Negative Control OD}_{420\text{nm}} - \text{Blank OD}_{420\text{nm}}}$$

$$\beta\text{-galactosidase } (\beta\text{-gal}) = \frac{\text{Sample OD}_{600\text{nm}} - \text{Blank OD}_{600\text{nm}}}{\text{Negative Control OD}_{600\text{nm}} - \text{Blank OD}_{600\text{nm}}}$$

$$\text{Induction Factor (IF)} = \frac{\beta\text{-gal}}{G}$$

- (3) Under the situation of survival rate over 80%. If IF>2, it means the positive result of genotoxicity; if IF<1.5, it means the negative result of genotoxicity.

Chapter 4 Results and Discussion

4.1 Water Parameter of Secondary Treatment Wastewater Effluents

The treated wastewater samples were obtained from the Neihu WWTP on two different dates in 2026, two major parameters were measured (Table 4.1): pH and NPDOC. The values are 7.38 and 4.34 mg/L on January 13th, and 8.33 and 4.65 mg/L on March 18th, for pH and NPDOC, respectively.

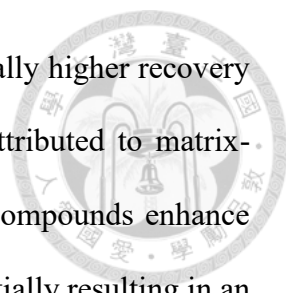
Table 4.1 Water Quality Parameters of Secondary Treated Wastewater Effluents

Secondary Treated Wastewater	JAN 13 th , 2026	MAR 18 th , 2026
pH@25°C	7.38	8.33
NPDOC (mg/L)	4.34	4.65

Recovery tests were conducted to evaluate the accuracy of the PPCP extraction method by determining the recovery of known spiked concentrations. In this study, recovery was calculated based on the internal standard method. A known concentration of 1 ng/mL was spiked into Milli-Q water and wastewater samples, which were subsequently processed through a series of SPE steps (as described in Table 3.3) to concentrate the PPCPs to 100 ng/mL.

As shown in Table 4.2, Milli-Q water was used as a baseline for comparison, while the results obtained from the January and March wastewater samples indicated the presence of matrix effects, suggesting that organic matter or other co-existing substances in the wastewater may have interfered with the analytical measurements.

For the Milli-Q water samples, all target compounds exhibited recovery rates within



the acceptable range of 90–120%. However, AZM showed substantially higher recovery rates in the wastewater samples. This elevated recovery may be attributed to matrix-induced ionization enhancement ^[127], in which co-eluting organic compounds enhance the ionization efficiency of AZM during LC–MS/MS analysis, potentially resulting in an overestimation of its actual concentration. Previous studies have reported AZM recovery rates as high as 190% in effluent samples ^[128], further supporting the possibility that matrix-induced signal enhancement contributed to the elevated recovery observed in this study.

In addition, Clarithromycin-N-methyl-¹³C, d₃ was selected as the internal standard for AZM based on a literature review. However, matrix effects may not have influenced AZM and the CLR internal standard to the same extent during MS/MS analysis ^[129]. The difference in their responses to matrix-induced ionization enhancement may therefore have contributed to the elevated AZM recovery observed in the wastewater samples.

Nevertheless, all relative percent difference (RPD) values for the target compounds were below 10%, indicating good analytical precision and demonstrating the stability and consistency of the overall experimental procedure.

In Table 4.3, the retention times of each PPCP range from 1.73 minutes for Trimethoprim to 3.25 minutes for CLR. The limits of detection (LOD) and quantification (LOQ) are calculated from the signal-to-noise ratio, with thresholds of 3 and 10, respectively. The linear range of the compounds is 1-500 ng/mL, with excellent linear performance (R^2), confirming that the instrument's response is theoretically stable and accurate for routine environmental monitoring.

Table 4.2 Recovery Rate of Each Compound in Different Water Matrices

Antibiotic Standard	Recoveries $\pm$ RPD% (n=2)		
	Milli-Q Water	JAN 13 th , 2026, post-secondary treated wastewater	MAR 18 th , 2026, post-secondary treated wastewater
Sulfamethoxazole	102 $\pm$ 4	73 $\pm$ 3	78 $\pm$ 3
N-acetyl Sulfamethoxazole	111 $\pm$ 6	127 $\pm$ 1	123 $\pm$ 2
Trimethoprim	97 $\pm$ 0	95 $\pm$ 6	93 $\pm$ 3
Levofloxacin	86 $\pm$ 7	88 $\pm$ 4	97 $\pm$ 3
Clarithromycin	108 $\pm$ 3	116 $\pm$ 3	136 $\pm$ 2
Azithromycin	96 $\pm$ 6	168 $\pm$ 6	161 $\pm$ 2

Table 4.3 Retention Time (RT), Limit of Detection (LOD), Limit of Quantification (LOQ), Calibration Curve Linear Range, and Coefficient of R-Square

Compound	Retention Time (RT) (min)	LOD (ng/L)	LOQ (ng/L)	Linear Range (ng/L)	R ²
Sulfamethoxazole	1.92	0.15	0.50	1-500	0.997
N-acetyl Sulfamethoxazole	2.25	0.13	0.48	1-500	0.997
Trimethoprim	1.73	0.02	0.06	1-500	0.997
Levofloxacin	1.84	0.04	0.12	1-500	0.997
Clarithromycin	3.24	0.02	0.06	1-500	0.992
Azithromycin	2.47	0.09	0.3	1-500	0.999

4.2 Oxidant Residuals

Aside from the removal efficiency, oxidant residuals are crucial for assessing treatment stability and safety in water treatment systems for informing operational decisions and ensuring compliance with safety standards. All systems containing oxidants are spiked with 30 μM and 50 μM oxidants in the Milli-Q water matrix and with 30 μM and 60 μM oxidants in the wastewater matrix to assess the matrix effect. The oxidant concentration was measured at designated reaction times (0, 3, 5, 10 min or 0, 10, 30, 60 min). The definition of 0 minutes (0 min) in the study is the sample collected after spiking with the oxidant and mixing for 1 minute.

In the Milli-Q water, as Figure 4.1 and Figure 4.2 showed, the PDS in all the UV/PDS or Solar/PDS systems consumed gradually, exceeded 50% in 10 minutes of UV-driven reaction, and around 50% in 60 minutes of solar-driven reaction, that is to say, as Figure 4.1 shows, there are $\sim 15 \mu\text{M}$ in the 30 μM group ($\sim 50\%$ PDS) and $\sim 21 \mu\text{M}$ in the 50 μM group (45% PDS), which have the potential to be activated by simulated sunlight and undergo the following removal reaction. For UV/PDS systems (Figure 4.2), $\sim 70\%$ of PDS (from 30 μM to 8 μM in the 30 μM system) and $\sim 60\%$ of PDS (from 50 μM to 20 μM in the 50 μM system) were consumed after UV/PDS processes. This is due to the absorption spectrum of PDS, which is mainly in the UV region^[96], meaning that relatively less PDS can be activated by the solar-driven system compared to the UV-driven ones over the same duration, suggesting that the activation time might be longer than the design assumed in the study.

On the other hand, when examining the Solar/Chlorine system (Figure 4.3), the concentration of chlorine declined sharply during the first 10 minutes of the reaction due to the overlapping absorption spectra of chlorine and simulated solar light^[130]. The quantum yield of chlorine is higher than that of PDS under the same reaction conditions.

The PPCP removal experiments (further discussed in Figures 4.13 and 4.14) provide evidence of chlorine consumption. Almost all antibiotics declined within the first 10 minutes of reaction, whereas the PDS group declined gradually as the oxidant consumption increased.

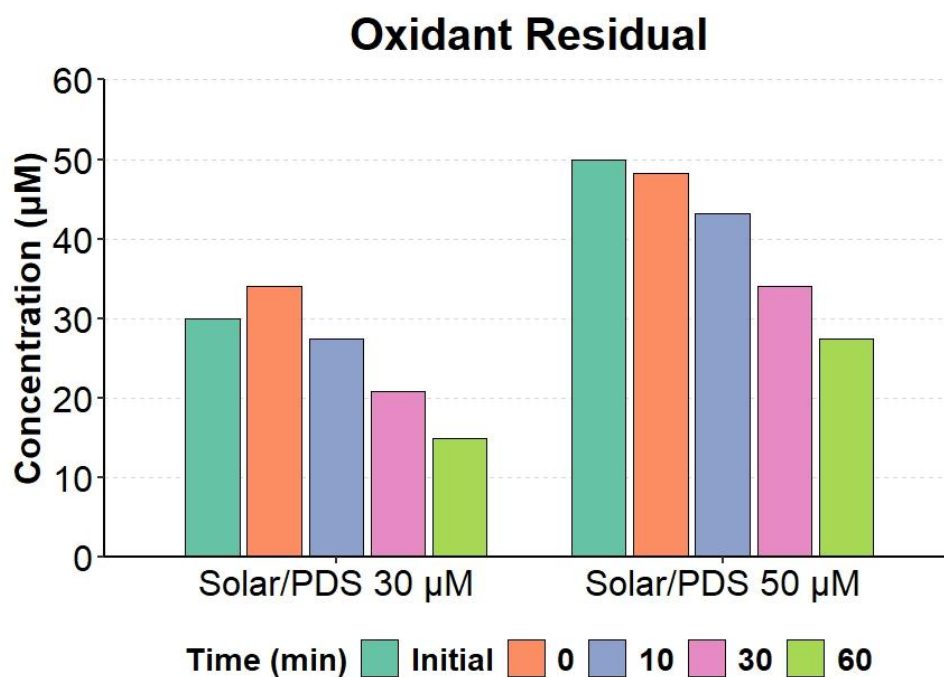


Figure 4.1 PDS residuals in the Solar/PDS systems in Milli-Q water
(Condition: Solar simulator 9 mW/cm², 500W)

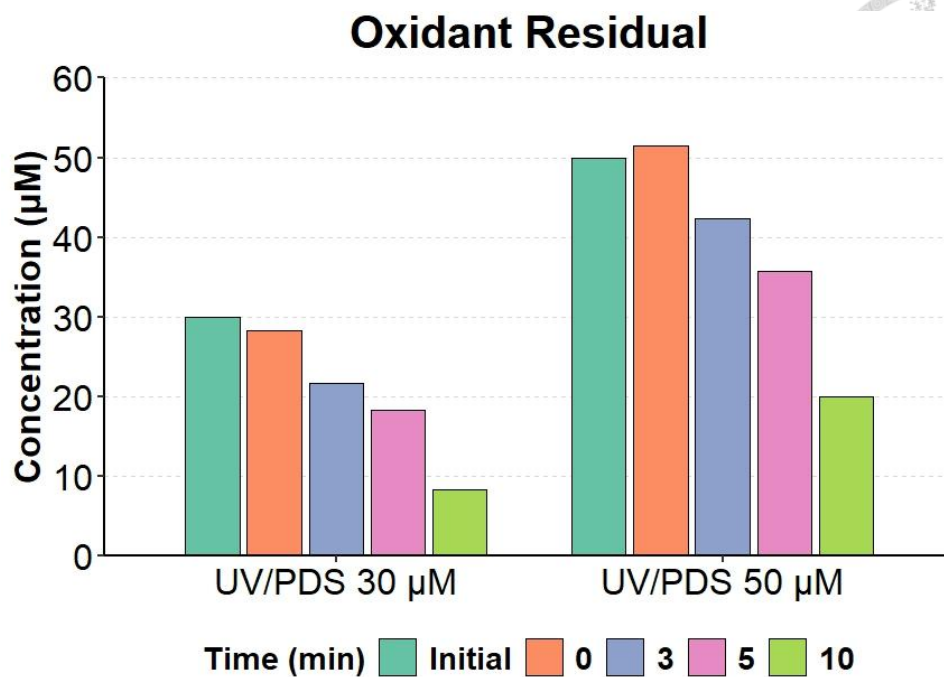


Figure 4.2 PDS residuals in the UV/PDS systems in Milli-Q water
(Condition: UV: 8 mW/cm², 14W, 254nm LPUV)

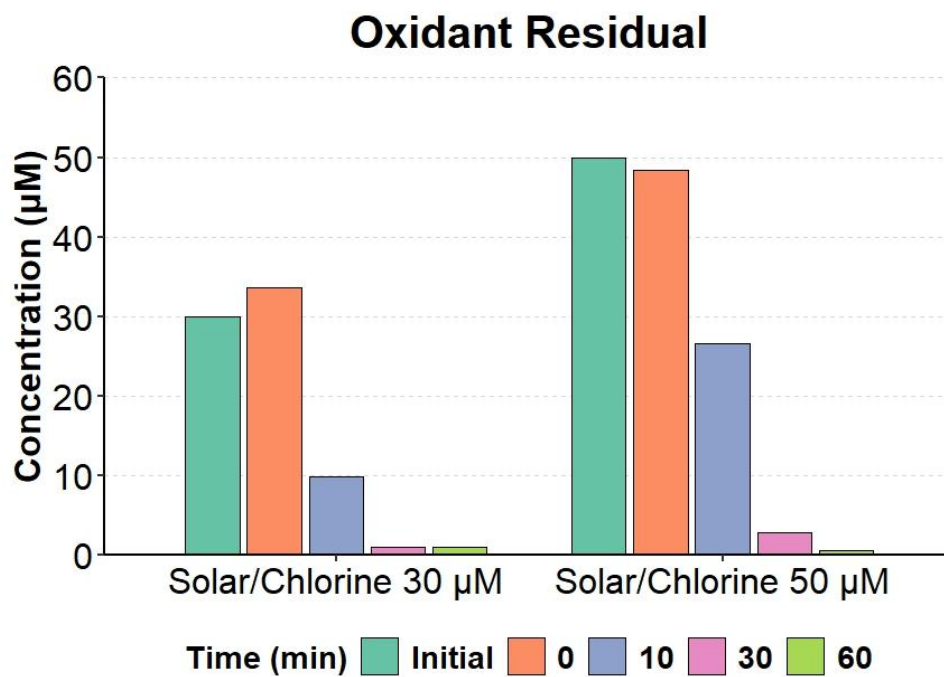
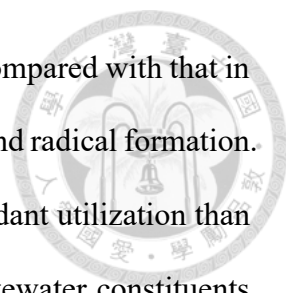


Figure 4.3 Chlorine residuals in the Solar/Chlorine systems in Milli-Q water
(Condition: Solar simulator 9 mW/cm², 500W)



In the wastewater matrix, PDS utilization generally decreased compared with that in Milli-Q water because the complex matrix reduced light penetration and radical formation. However, the Solar/PDS 30 μM system exhibited slightly higher oxidant utilization than the corresponding Milli-Q water group, suggesting that certain wastewater constituents may enhance PDS activation under this condition. As shown in Figure 4.4, approximately 18 μM of PDS was consumed in the Solar/PDS system with an initial concentration of 30 μM (~60% PDS), whereas approximately 20 μM was consumed in the 60 μM group (~33% PDS). In the UV/PDS system (Figure 4.5), approximately 20 μM of PDS was consumed in the 30 μM group (~66% PDS), while approximately 28 μM was consumed in the 60 μM group (~40% PDS)

To address this, future work should explore specific catalysts, such as g- C_3N_4 or carbonaceous materials, to enhance the catalytic AOP oxidation capacity. Some relevant studies [97,100,131–133] have shown that combining light-absorbing photocatalysts, photothermal heating, and co-oxidants improves radical generation and charge transfer. Carbonaceous or g- C_3N_4 -based catalysts might enhance oxidation efficiency by providing additional reaction surface area or active sites to stimulate PDS, leveraging the photothermal effect of catalysts that convert solar light to heat and undergo thermal activation. Nevertheless, the catalyst should be at an optimal dosage, as turbidity from suspended catalysts blocks and scatters light, reducing penetration and further lowering removal rates in the wastewater matrix [100].

In the Solar/Chlorine group (Figure 4.6), most of the chlorine spiked into the system almost disappeared at the beginning stage of the AOP experiment. After 10 minutes of reaction, chlorine concentration declined by about 50%, likely because the free chlorine reacted with the wastewater matrix before being activated by the simulated solar light. This rapid initial consumption highlights the challenge of maintaining stable residual

chlorine levels, which are critical for consistent disinfection and oxidation in wastewater treatment, underscoring the need for process control.

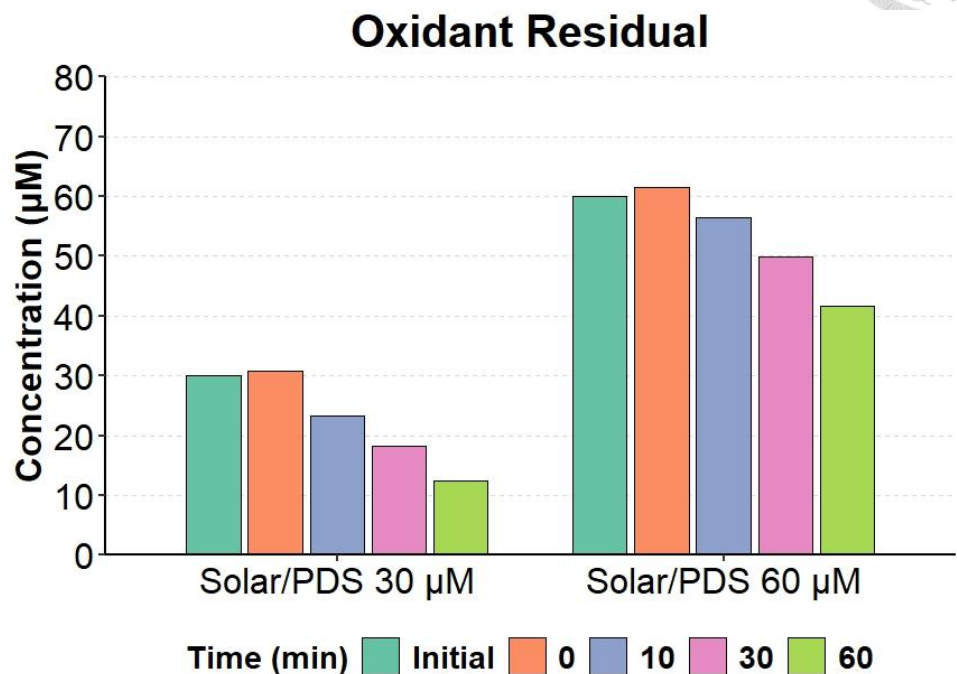


Figure 4.4 PDS residuals in the Solar/PDS systems in wastewater
(Condition: Solar simulator 9 mW/cm², 500W)

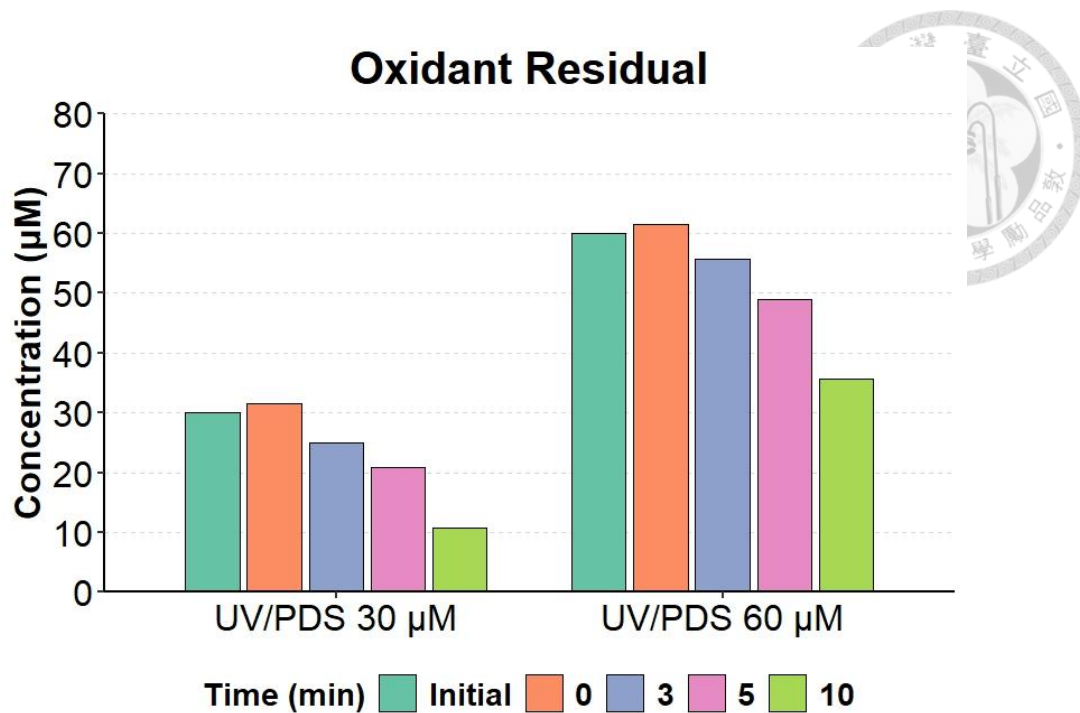


Figure 4.5 PDS residuals in the UV/PDS systems in wastewater
(Condition: UV: 8 mW/cm², 14W, 254nm LPUV)

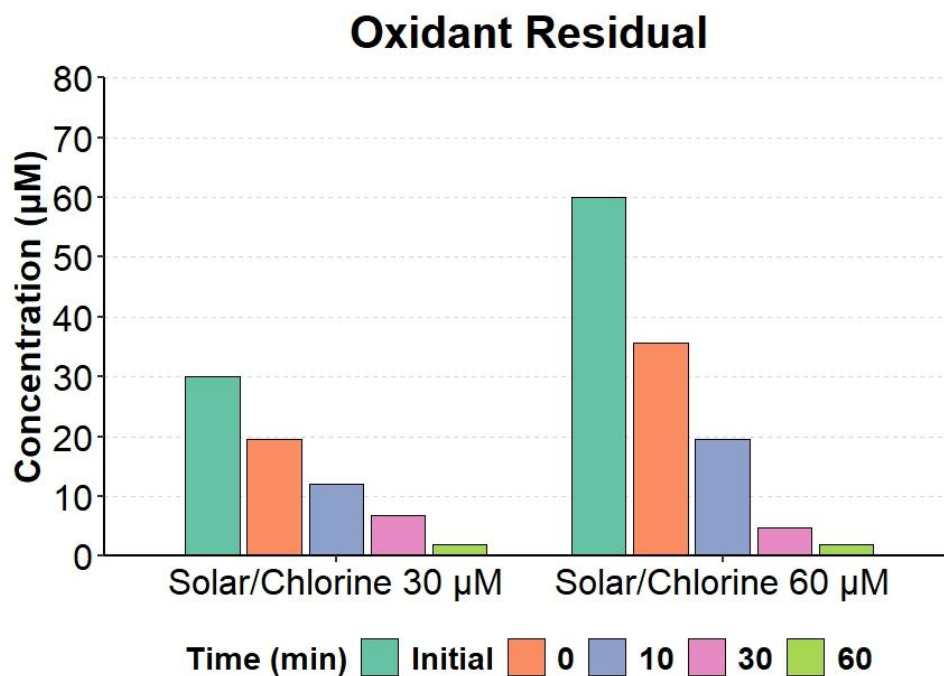


Figure 4.6 Chlorine residuals in the Solar/Chlorine systems in wastewater
(Condition: Solar simulator 9 mW/cm², 500W)

4.3 PPCP Removal Efficiencies in Different AOP Treatments

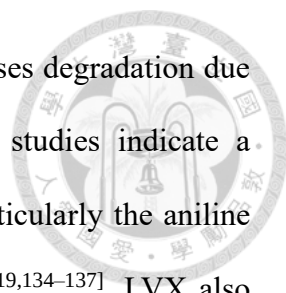
To evaluate treatment performance in a real wastewater matrix, preliminary experiments were first conducted with Milli-Q water. Following these experiments, the study proceeded to AOP experiments using secondary-treated wastewater as the reaction matrix. Each 2.5 L water sample was spiked with antibiotic standards to achieve a concentration of 3 ng/mL and stirred for 10 minutes to ensure complete mixing. For the combined oxidant group, 30 μ M or 50 μ M of chlorine or PDS was added in the Milli-Q water experiments. In contrast, in the wastewater experiment, the oxidant concentrations were 30 μ M or 60 μ M, accounting for the natural scavengers in the wastewater.

Subsequently, the initial sample (C_0) was collected from UV-only, solar-only, Solar/PDS, and Solar/Chlorine systems. Samples were then taken at 0, 10, 30, and 60 min of reaction. In the UV/PDS groups, samples were collected at 0, 3, 5, and 10 min of reaction. After concentration, all samples were concentrated 100-fold, and the C_0 sample was assumed to contain 300 ng/mL during LC–MS/MS analysis.

4.3.1 Experiments Conducted in Milli-Q water

1. UV Systems

In the UV-only system (Figure 4.7), SMX and NASMX achieved nearly complete removal within 10 min of reaction. This enhanced degradation can be attributed to their molecular structures, which exhibit high molar absorptivity in the 250–300 nm wavelength range ^[134]. Upon absorbing UV radiation, SMX molecules are promoted to an excited state and subsequently undergo direct photolytic degradation or participate in energy- and electron-transfer reactions ^[134]. Consequently, direct photolysis serves as an effective degradation pathway for both SMX and NASMX under UV irradiation. TMP, AZM, and CLR exhibited relatively low removal efficiencies in the UV-only system.



As for the UV/PDS system (Figure 4.8-4.9), the SMX also causes degradation due to attack by sulfate radicals. Quantum calculations and product studies indicate a preferred attack on the aromatic/heterocyclic moieties of SMX, particularly the aniline and isoxazole rings, and cleavage of the S–N/sulfonamide bond ^[119,134–137]. LVX also exhibited appreciable degradation in both the UV and UV/PDS systems. Approximately 50% removal was achieved within 60 min of reaction under UV irradiation alone, whereas complete removal was observed within 3 min in the UV/PDS system. In addition to the direct photolysis, radical-mediated oxidation likely contributed to the enhanced degradation. Previous studies have suggested that sulfate radicals can attack the piperazinyl moiety of LVX, leading to dealkylation and ring-opening reactions ^[138]. The addition of PDS significantly improved the TMP degradation in UV-based systems. Based on a review of the literature, the pyrimidine moiety is the most reactive site, and the methylene bridge is a major site of product formation, leading to α -hydroxytrimethoprim and α -ketotrimethoprim as the dominant intermediates^[121,139]. Interestingly, TMP removal was higher in the 30 μ M UV/PDS group than in the 50 μ M UV/PDS group, reaching approximately 90% and 50%, respectively. This phenomenon may be attributed to the radical-scavenging effect of excess PDS, which scavenges reactive radicals, thereby reducing the overall oxidation efficiency. For AZM and CLR, both of which belong to the macrolide antibiotic class, degradation efficiencies remained lower than those of the other target compounds, even in the presence of PDS. Under UV/PDS conditions, AZM and CLR achieved removal efficiencies of approximately 75% and 60%, respectively, with little difference observed between the two oxidant concentrations tested. Regarding the degradation pathway mechanism, most evidence supports hydroxyl radical-based AOPs, in which ring hydroxylation predominates, leading to further breakdown ^[140]. To date, no study directly maps $\text{SO}_4^{\bullet-}$ driven ring cleavage for CLR or AZM. However, a broad

review of persulfate notes that $\text{SO}_4^{\bullet-}$ and $\bullet\text{OH}$ readily attack C–H, S–N, C–N, C=C, and amino groups in antibiotics, thereby initiating structural breakdown ^[141].

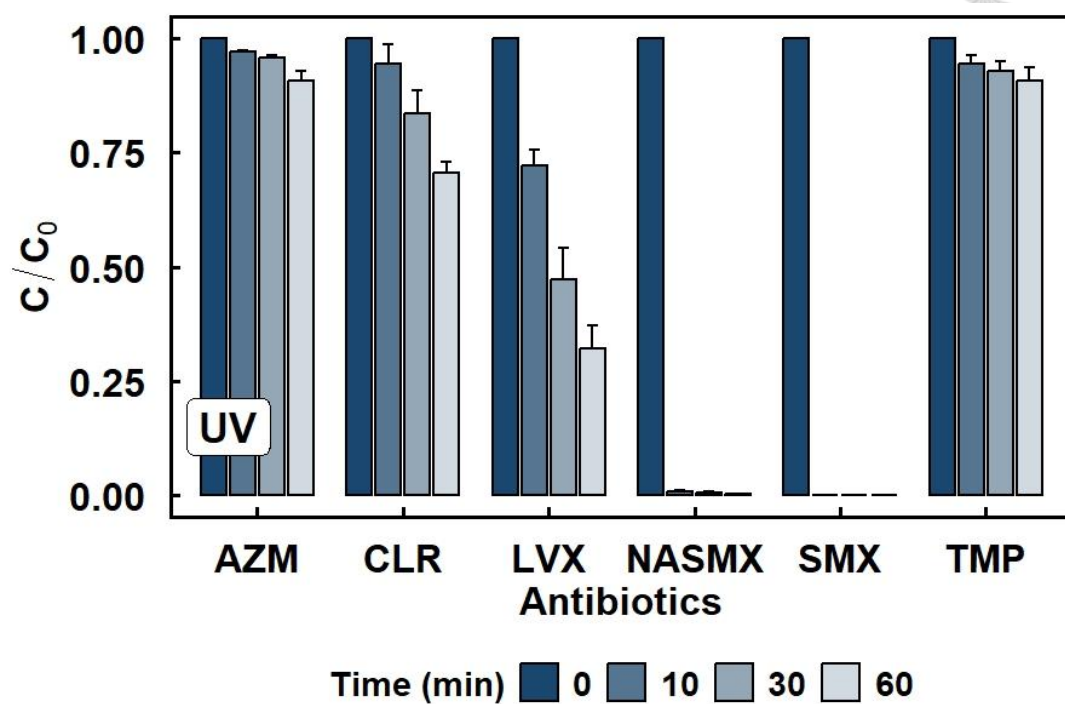


Figure 4.7 PPCP removal under the UV-alone system in Milli-Q water ([PPCP]₀=3 ng/mL, UV condition: 8 mW/cm², 14W, 254nm LPUV)

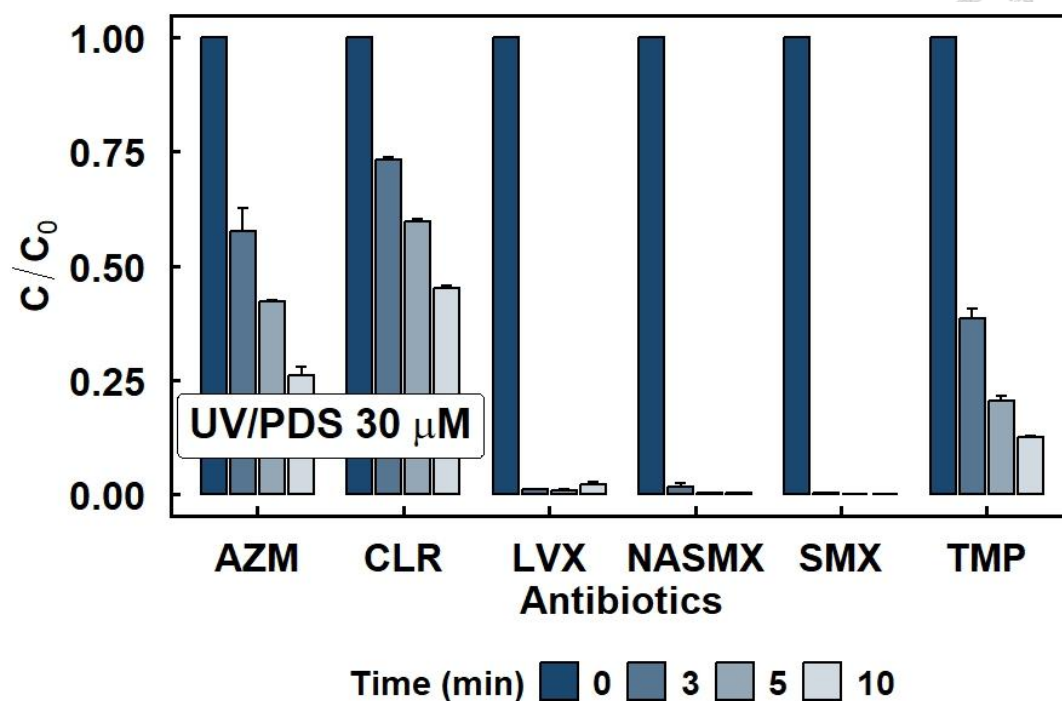


Figure 4.8 PPCP removal under the UV/PDS (30 μM) system in Milli-Q water
 ([PPCP]₀=3 ng/mL, UV condition: 8 mW/cm², 14W, 254nm LPUV)

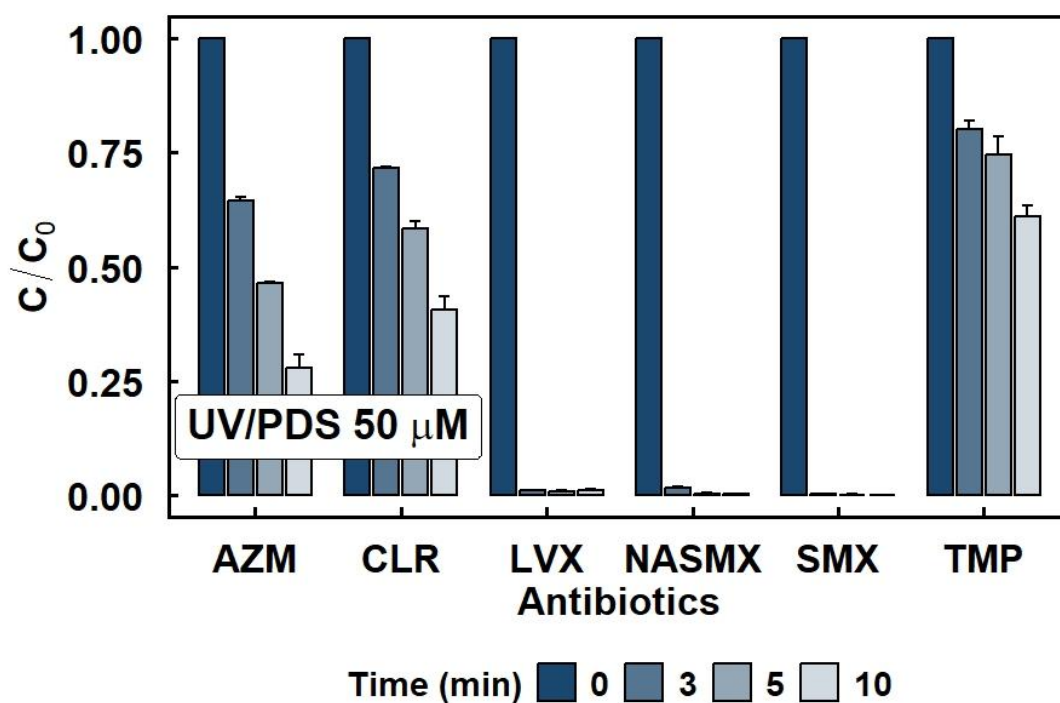


Figure 4.9 PPCP removal under the UV/PDS (50 μM) system in Milli-Q water
 ([PPCP]₀=3 ng/mL, UV condition: 8 mW/cm², 14W, 254nm LPUV)

2. Solar light Systems

Under solar irradiation alone (Figure 4.10), no significant removal of the target compounds was observed, indicating that although solar light can effectively inactivate microorganisms, its oxidation capacity is insufficient to degrade the selected antibiotics under the tested conditions.

In the Solar/PDS system (Figure 4.11-4.12), most target compounds exhibited moderate removal efficiencies of approximately 40–50%. The concentration of SMX gradually decreased over the reaction period in the Solar/PDS systems, whereas negligible degradation was observed in the solar-only system. This result suggests that SMX removal was not primarily driven by solar photolysis, but was enhanced by sulfate radical oxidation generated through PDS activation. In contrast, NASMX showed little to no change over the 60-min experiment, further suggesting that UV activation may be critical for the degradation of this compound. As for TMP, a trend similar to that observed in the UV/PDS system was found: a lower PDS dosage resulted in higher removal efficiency, likely due to self-quenching in the system.

Different from TMP, LVX showed a dependence on the PDS concentration in the Solar/PDS system; increasing the PDS concentration from 30 to 50 μM significantly enhanced its degradation, resulting in near-complete removal by the end of the 60-min reaction. For AZM and CLR, the degradation behavior was consistent with that observed in the UV-driven system (Figure 4.7-4.9). Although both compounds were removed to a limited extent, AZM consistently exhibited higher removal efficiency than CLR, suggesting that little structural difference would influence their reactivity toward reactive radical species.

Among all solar-driven processes, the Solar/Chlorine system (Figure 4.13-4.14) achieved the highest removal efficiencies, resulting in overall removal rates exceeding

50% except for NASMX. SMX, LVX, and TMP were rapidly degraded, with nearly complete removal occurring within the first 10 min of reaction. Previous studies have reported that TMP readily reacts with free chlorine and undergoes extensive transformation, producing numerous chlorinated and hydroxylated byproducts rather than complete mineralization ^[142].

AZM and CLR also showed improved degradation in the Solar/Chlorine system, reaching approximately 50% removal within 10 min. Intriguingly, after this rapid initial "burst degradation" in the first 10 min, the residual concentrations of both remained relatively stable up to 60 min. This plateau suggests the rapid depletion of available active chlorine species early in the process. Although no report exists on the effect of Solar/Chlorine on AZM or CLR degradation, chlorination mechanisms have been detailed for other macrolides. The primary reactive site is the tertiary amine in the desosamine sugar, which undergoes demethylation and chlorination, yielding multiple chlorinated transformation products ^[143].

In summary, in terms of removal efficiency in wastewater, in most of the systems assessed, AZM exhibited higher degradation efficiency than CLR. Notably, NASMX remained largely unaffected in Solar/Chlorine systems. The persistence of NASMX under solar-driven conditions suggests that UV activation may be necessary for its effective degradation.

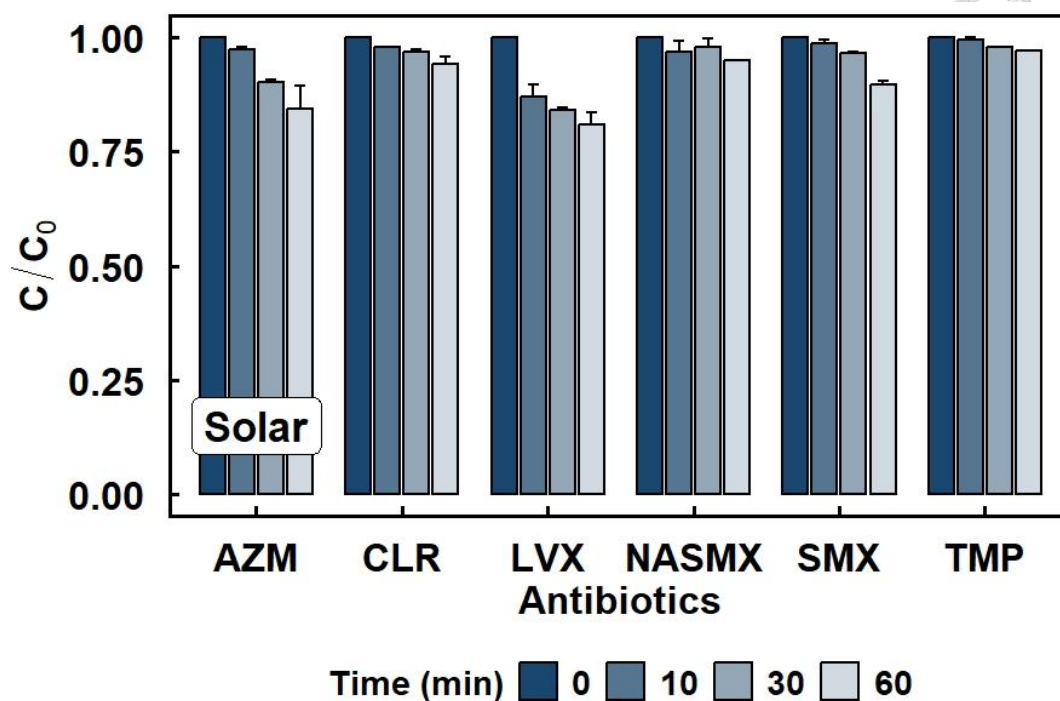


Figure 4.10 PPCP removal under the Solar light alone system in Milli-Q water
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

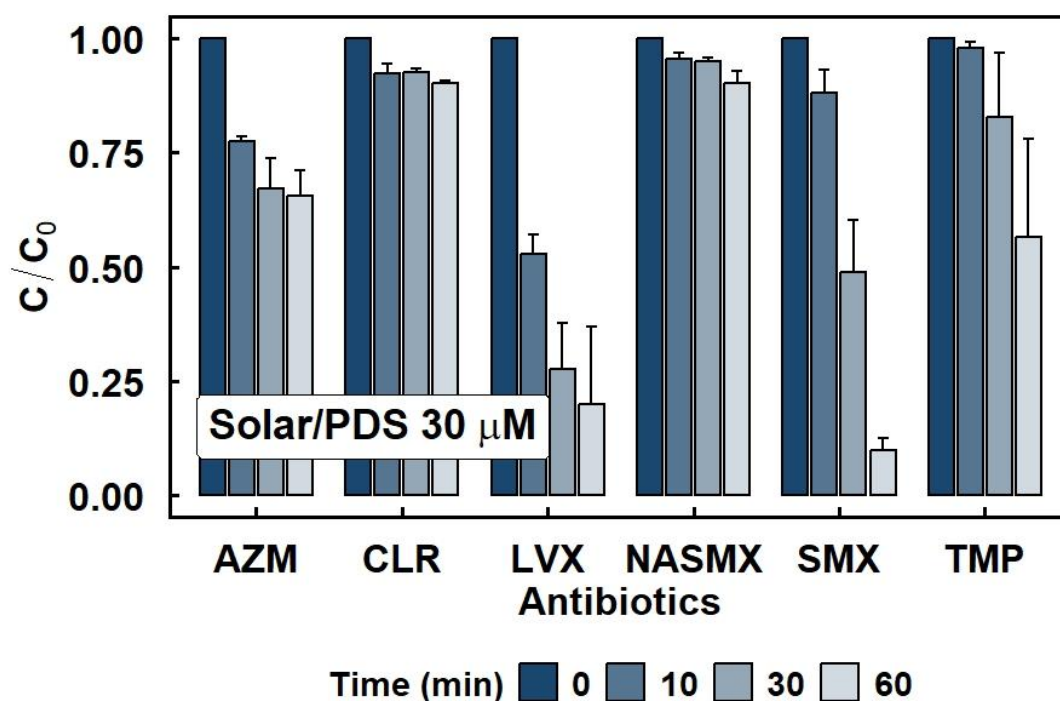


Figure 4.11 PPCP removal under the Solar/PDS (30 μM) system in Milli-Q water
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

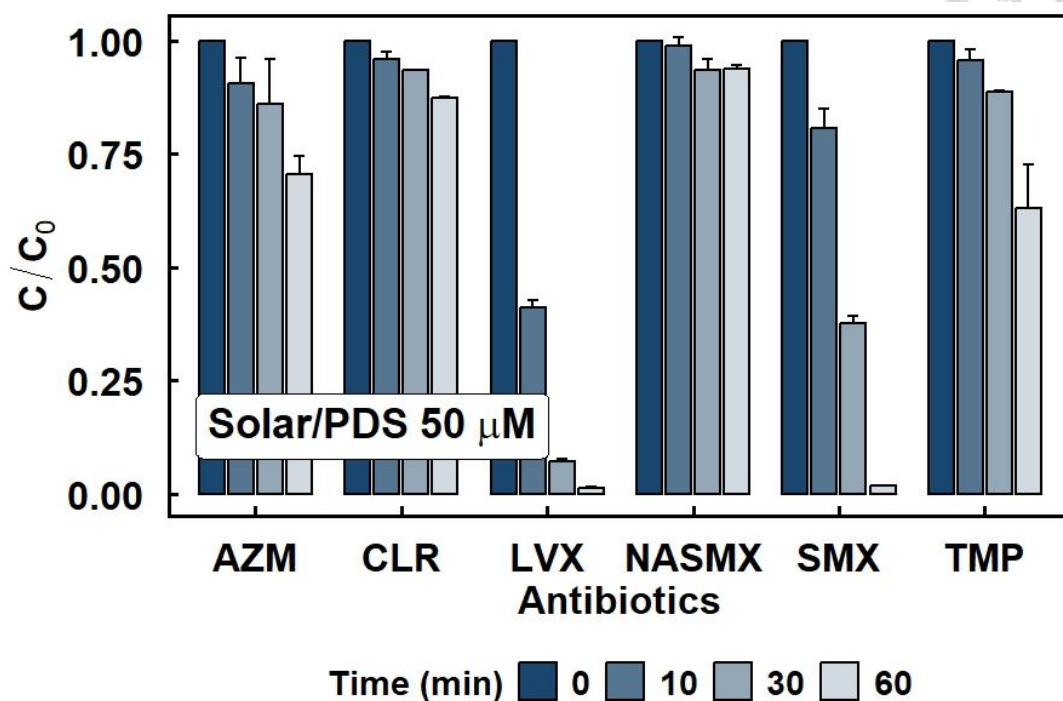


Figure 4.12 PPCP removal under the Solar/PDS (50 μM) system in Milli-Q water
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

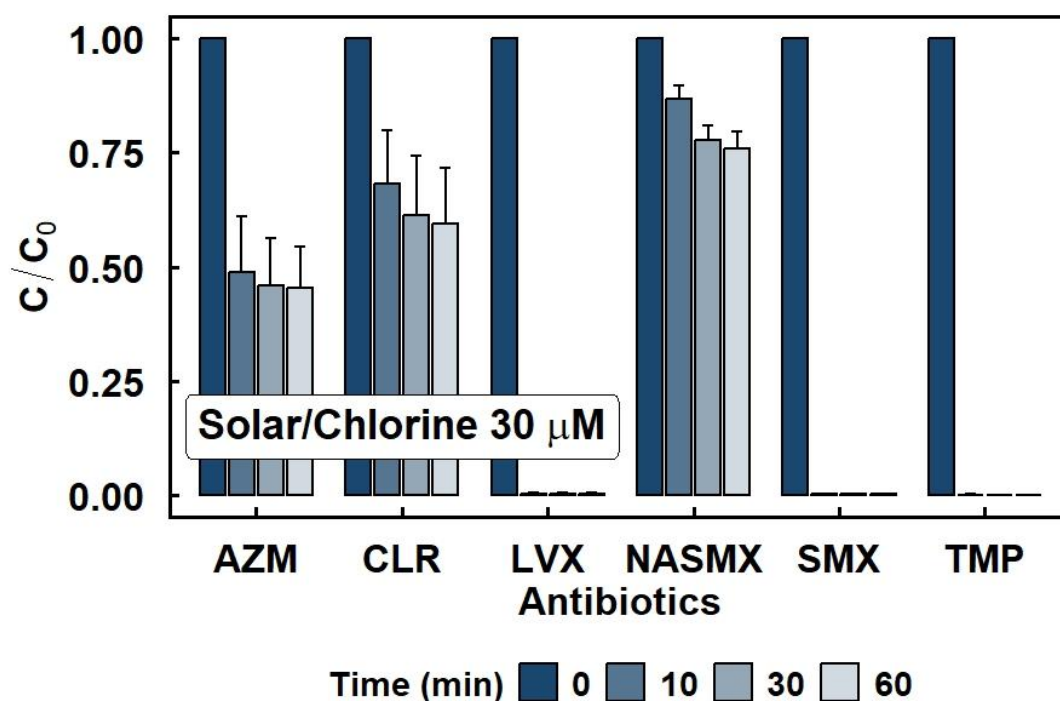


Figure 4.13 PPCP removal under the Solar/Chlorine (30 μM) system in Milli-Q water
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

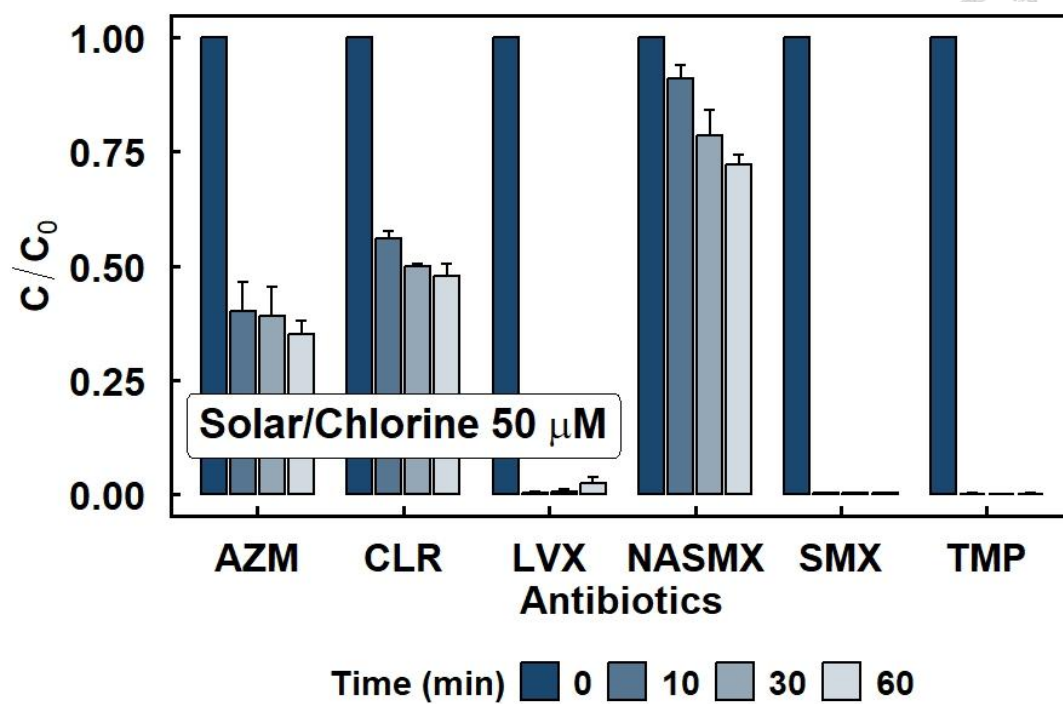
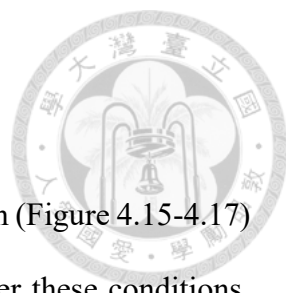


Figure 4.14 PPCP removal under the Solar/Chlorine (50 μM) system in Milli-Q water
 ($[\text{PPCP}]_0 = 3 \text{ ng/mL}$, Solar simulator condition: 9 mW/cm^2 , 500W)

4.3.2 Experiment in the wastewater matrix

1. UV Systems



In wastewater, the PPCP degradation efficiency of UV irradiation (Figure 4.15-4.17) typically decreases due to the complexity of the water matrix. Under these conditions, SMX, NASMX, and LVX still exhibited the highest removal efficiencies within the UV-only system (Figure 4.15), reaching approximately 100%, 100%, and 70% within 60 minutes of reaction, respectively. Conversely, AZM, CLR, and TMP remained recalcitrant, with removal efficiencies falling below 25%, a trend that mirrors the results obtained in Milli-Q water. In the UV/PDS systems (Figure 4.16-4.17), LVX, SMX, and NASMX again demonstrated the highest degradation performance, yielding removal efficiencies of 99%, 99%, and 65% with 30 μM of PDS, and 99%, 99%, and 70% with 60 μM of PDS, respectively.

The removal of NASMX after 10 minutes of reaction was nearly identical in both the UV-only and UV/PDS systems, thereby reconfirming that direct UV photolysis is the primary driver of NASMX degradation. For the remaining compounds in the UV/PDS system, natural matrix scavengers, such as chloride and carbonate ions, removed accessible radicals in the wastewater^[144]. Furthermore, increased turbidity hindered PDS activation, thereby reducing the radical generation and resulting in lower removal efficiencies than those observed in the Milli-Q water groups.

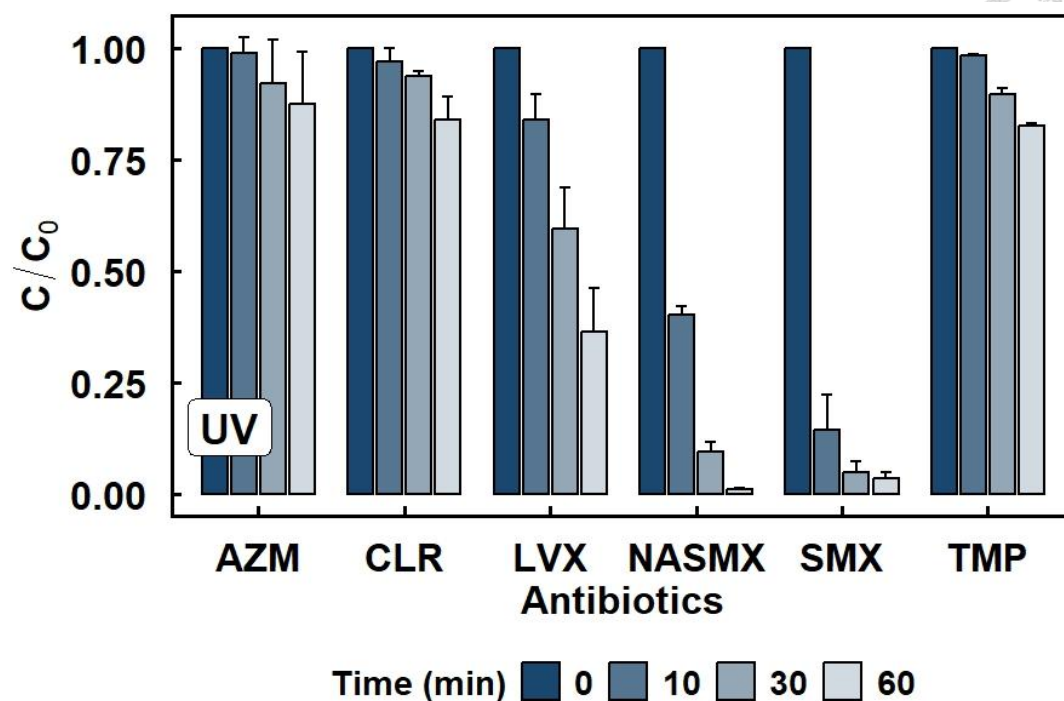


Figure 4.15 PPCP removal under the UV-alone system in wastewater ([PPCP]₀=3 ng/mL, UV condition: 8 mW/cm², 14W, 254nm LPUV)

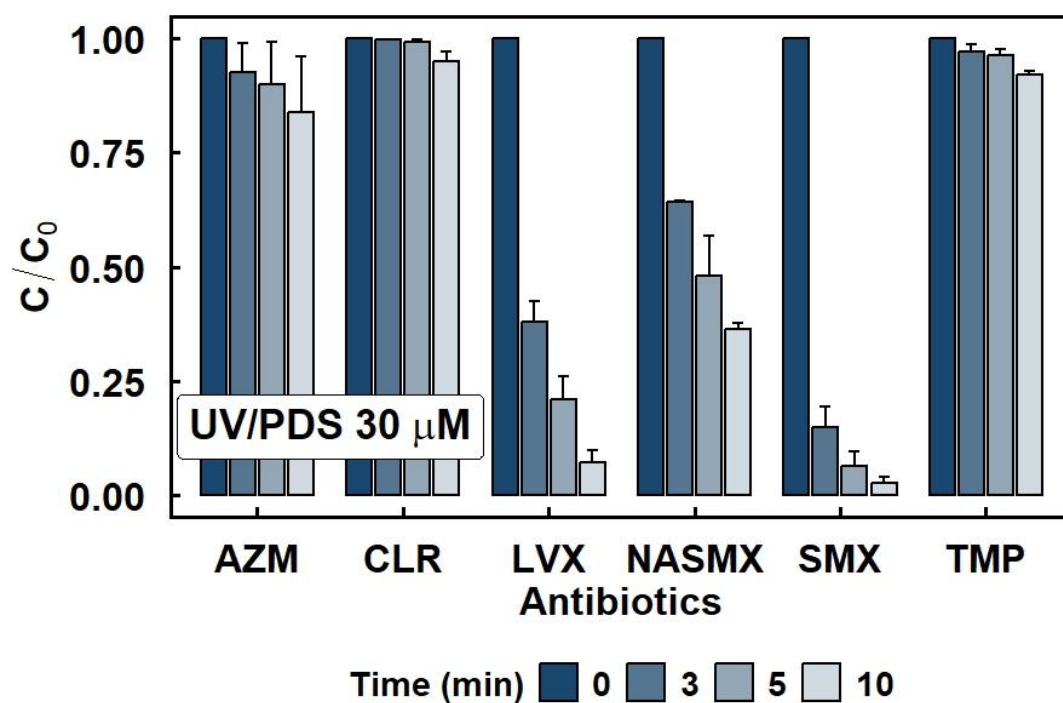


Figure 4.16 PPCP removal under the UV/PDS (30 μM) system in wastewater ([PPCP]₀=3 ng/mL, UV condition: 8 mW/cm², 14W, 254nm LPUV)

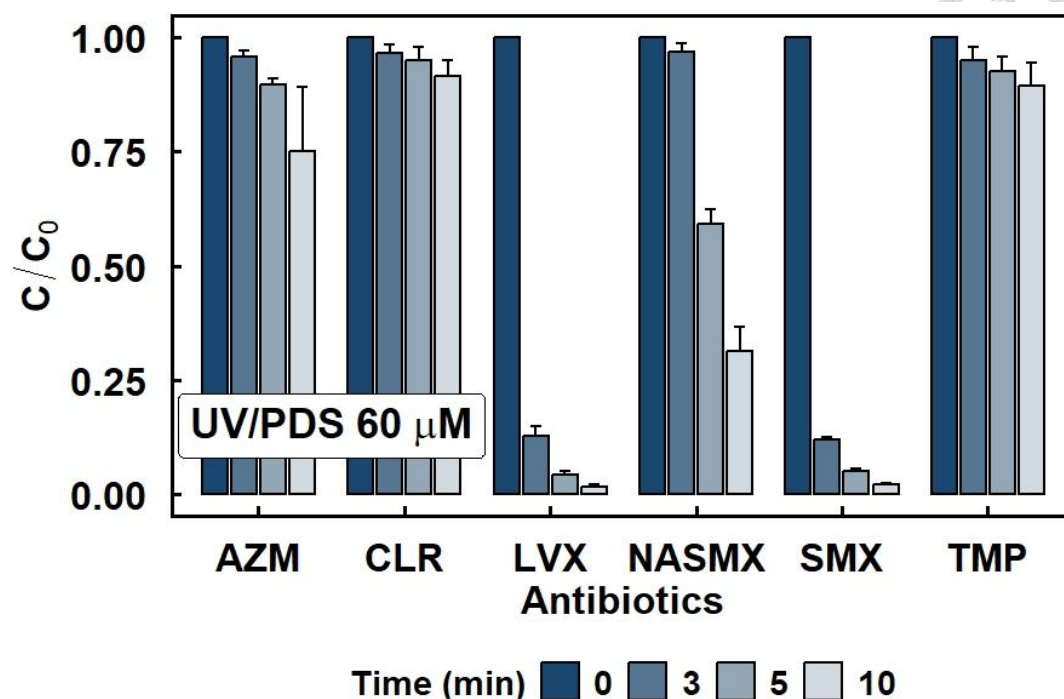


Figure 4.17 PPCP removal under the UV/PDS (60 μ M) system in wastewater ([PPCP]₀=3 ng/mL, UV condition: 8 mW/cm², 14W, 254nm LPUV)

2. Solar light Systems

Regarding the solar irradiation-only system (Figure 4.18), the negligible removal of all target compounds under turbid conditions highlights the severe limitations of direct solar photolysis in real wastewater matrices, primarily due to the light-shielding effect of suspended solids.

In the Solar/PDS groups, only LVX and SMX showed substantial removal, mirroring the trends observed in RO water. Specifically, in the Solar/PDS (30 μ M) system (Figure 4.19), the 60-minute removal efficiencies for both compounds reached approximately 50%. When the PDS dosage was increased to 60 μ M (Figure 4.20), a distinct dose-dependent behavior was observed for LVX, with its removal efficiency improving to 75%. Conversely, the final removal efficiency for SMX remained static at 50%. SMX degradation was markedly accelerated in wastewater, reaching a

maximum of 50% removal within the first 10 minutes of reaction, whereas it proceeded much more slowly in the Milli-Q water. This rapid initial transformation can be attributed to dissolved organic matter (DOM) in the matrix, which likely acts as a photosensitizer, generating reactive oxygen species (ROS) that facilitate initial degradation ^[145].

However, the subsequent plateauing of SMX removal from 10 to 60 minutes suggests a rapid depletion of PDS driven by competing wastewater constituents. In contrast to SMX, LVX exhibited a more progressive degradation profile over the 60 minutes of reaction. Nevertheless, its overall removal was lower than that in Milli-Q water, suggesting that the water matrix may concurrently exert a scavenging or suppressive effect on LVX degradation pathways. Meanwhile, its metabolite, NASMX, as well as AZM, CLR, and TMP, remained highly recalcitrant across both PDS dosages, showing no significant removal.

In the Solar/Chlorine systems (Figures 4.21 and 4.22), a similar target-specific degradation pattern occurred. Chlorine species appeared to react preferentially and rapidly with the wastewater matrix rather than with the target antibiotics, resulting in significantly lower treatment efficiencies than in Milli-Q water. SMX remained the only compound vulnerable to this process, achieving 30–40% removal within the first 10 minutes before reaching a complete plateau. Increasing the chlorine dosage from 30 to 60 μM yielded negligible improvements for the remaining compounds. This behavior supports the hypothesis of greater reactivity between reactive chlorine species (RCS) and the wastewater matrix (such as natural organic matter and ambient ions) ^[146], thereby rapidly exhausting the available oxidant and severely hindering the degradation of most target pharmaceuticals.

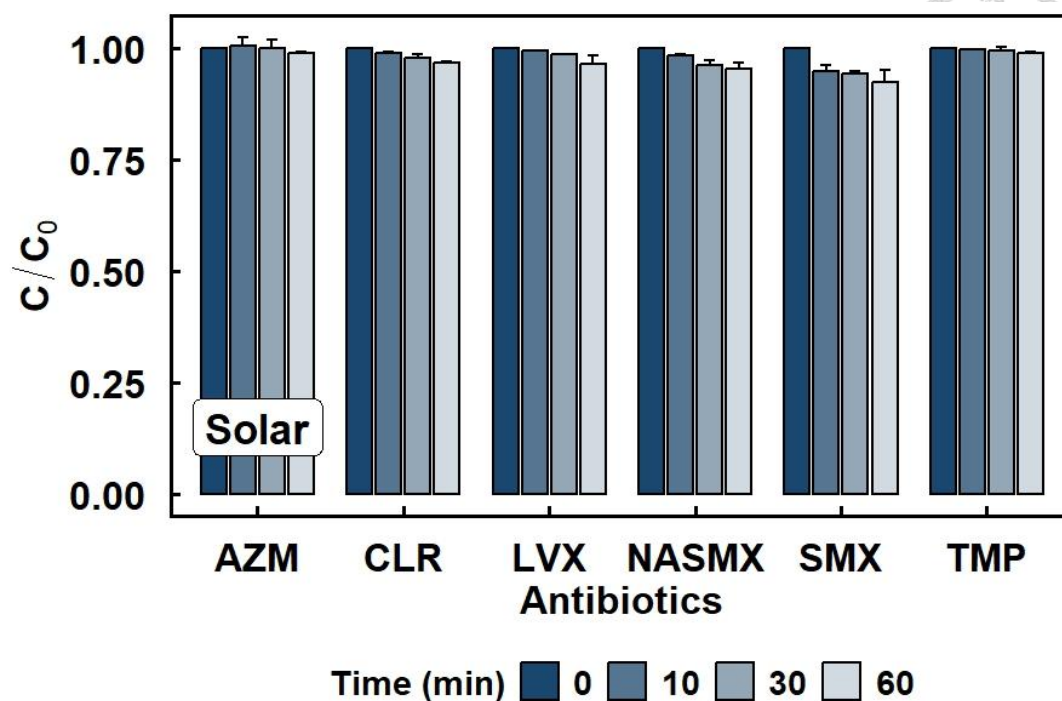


Figure 4.18 PPCP removal under the Solar light alone system in wastewater
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

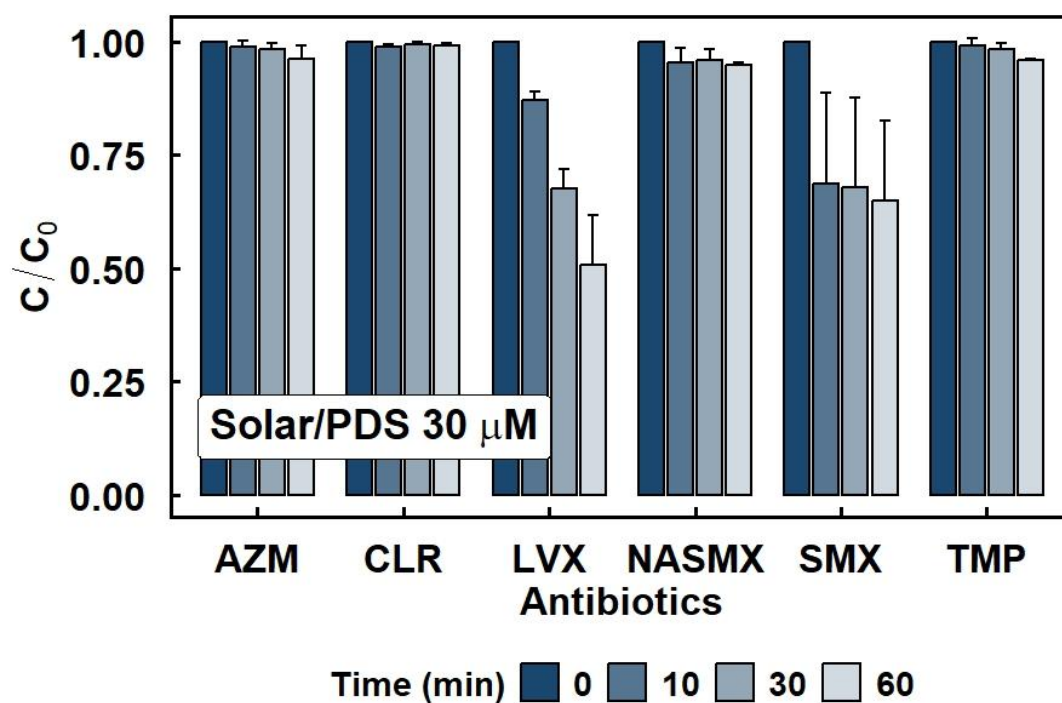


Figure 4.19 PPCP removal under the Solar/PDS (30 µM) system in wastewater
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

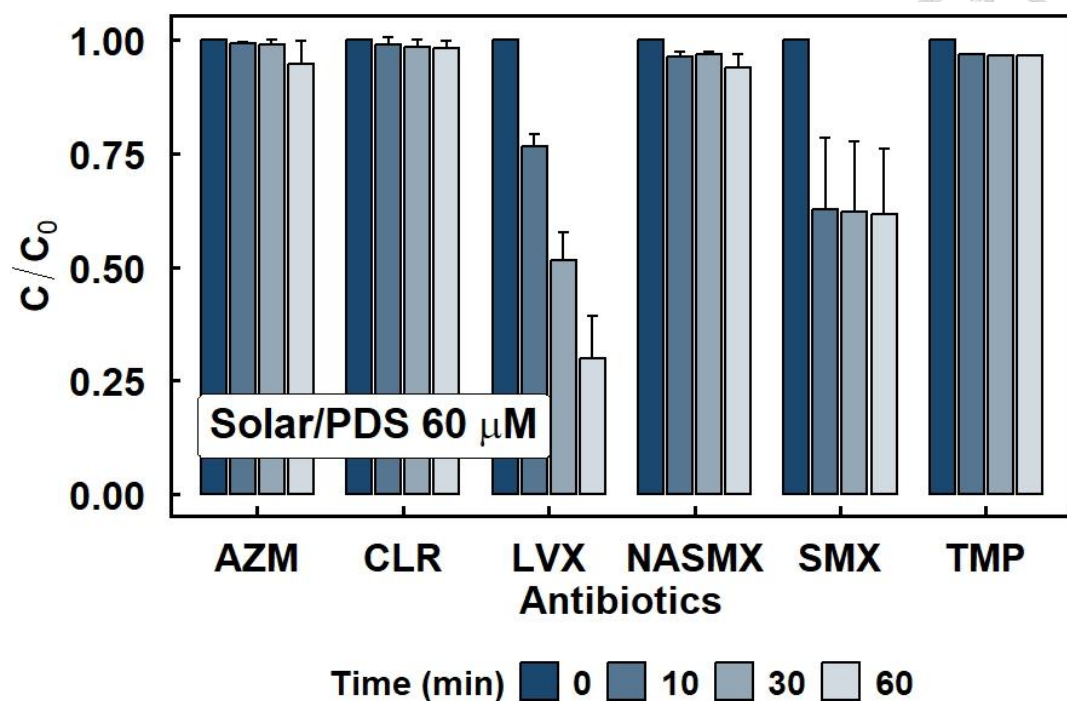


Figure 4.20 PPCP removal under the Solar/PDS (60 μM) system in wastewater
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

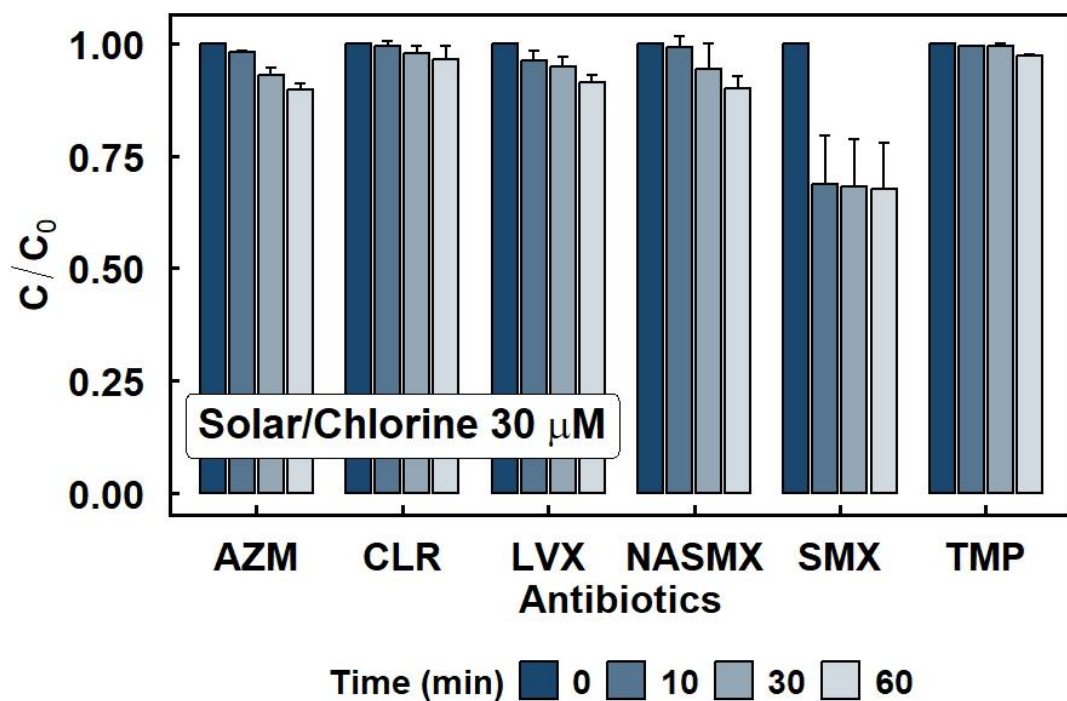


Figure 4.21 PPCP removal under the Solar/Chlorine (30 μM) system in wastewater
 ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

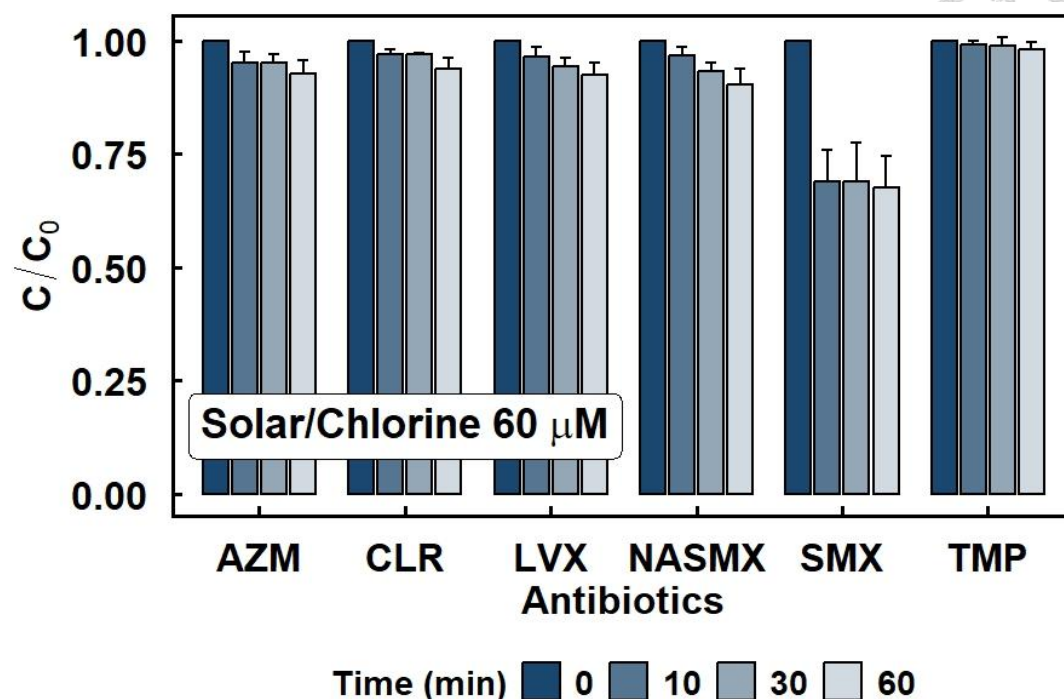


Figure 4.22 PPCP removal under the Solar/Chlorine (60 μM) system in wastewater ([PPCP]₀=3 ng/mL, Solar simulator condition: 9 mW/cm², 500W)

4.4 DBPs Formation in the Different AOPs

Figures 4.23 to 4.25 depict the formation profiles of trihalomethanes (THMs), haloacetic acids (HAAs), and haloacetonitriles (HANs) across the various treatment systems. As illustrated in Figure 4.23, THM formation was highly specific to chlorination-based processes^[147,148], with the Solar/Chlorine systems yielding the highest concentrations, predominantly trichloromethane (TCM). TCM concentrations in the Solar/Chlorine groups initially increased up to 30 minutes of reaction, then declined noticeably after 60 minutes of reaction. This downward trend at the later stage of the reaction suggests potential volatilization of TCM or its subsequent photolytic transformation under solar irradiation.

Figure 4.24 presents the temporal formation of various HAA species. In parallel with the THM trends, the Solar/Chlorine processes generated the highest total HAA

concentrations, exceeding 3 µg/L at the 60 µM dosage. This substantial accumulation was primarily driven by a sharp, time-dependent rise in dichloroacetic acid (DCAA) levels, whereas the other treatment groups remained relatively stable and low. This phenomenon was also observed in a previous study, which indicates that TCM, DCAA, and Trichloroacetic acid (TCAA) are widely present in the drinking water distribution system [147].

Distinctive variations were observed regarding the nitrogenous DBPs (N-DBPs); N-DBPs yielded relatively lower concentrations compared to C-DBPs. Figure 4.25 illustrates the formation of HANs, where the UV-only group showed a progressive increase over time, dominated by trichloroacetonitrile (TCAN) and dibromoacetonitrile (DBAN). This significant upward trend indicates that UV irradiation can alter the structural properties of dissolved organic matter (DOM), thereby exposing nitrogenous precursors that subsequently react with background halide ions naturally present in the wastewater [149]. Conversely, under Solar/Chlorine conditions, dichloroacetonitrile (DCAN) emerged as the heavily predominant HAN species over time. This dominance aligns with the literature, which consistently reports DCAN as the primary N-DBP species generated during the chlorination of nitrogenous organic matter [150].

For the control and PDS groups where no external chlorine was introduced, minor background levels of DBPs were still detected (approximately 0.5 µg/L for THMs and 1.0 µg/L for HAAs). This baseline formation is attributable to the interaction between active radical species and background chloride or DBP precursor ions inherently present in the real wastewater matrix [151].

In general, chlorine-based systems yield the highest DBP levels. It reconfirms that free chlorine or reactive chlorine species (RCS) preferentially attack and are heavily consumed by the wastewater matrix to form these regulated DBPs, thereby leaving

insufficient oxidants to degrade the target recalcitrant antibiotics.

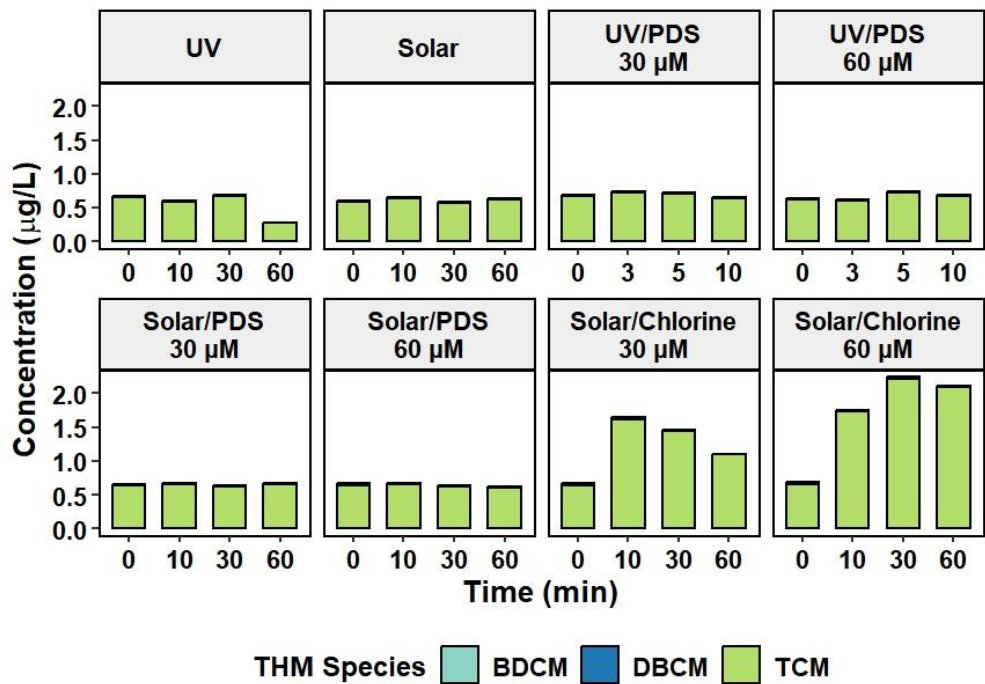


Figure 4.23 THMs formation in the PPCP treatment with different AOPs
(Condition: Solar simulator: 9 mW/cm², 500W; UV: 8 mW/cm², 14W, 254nm LPUV)

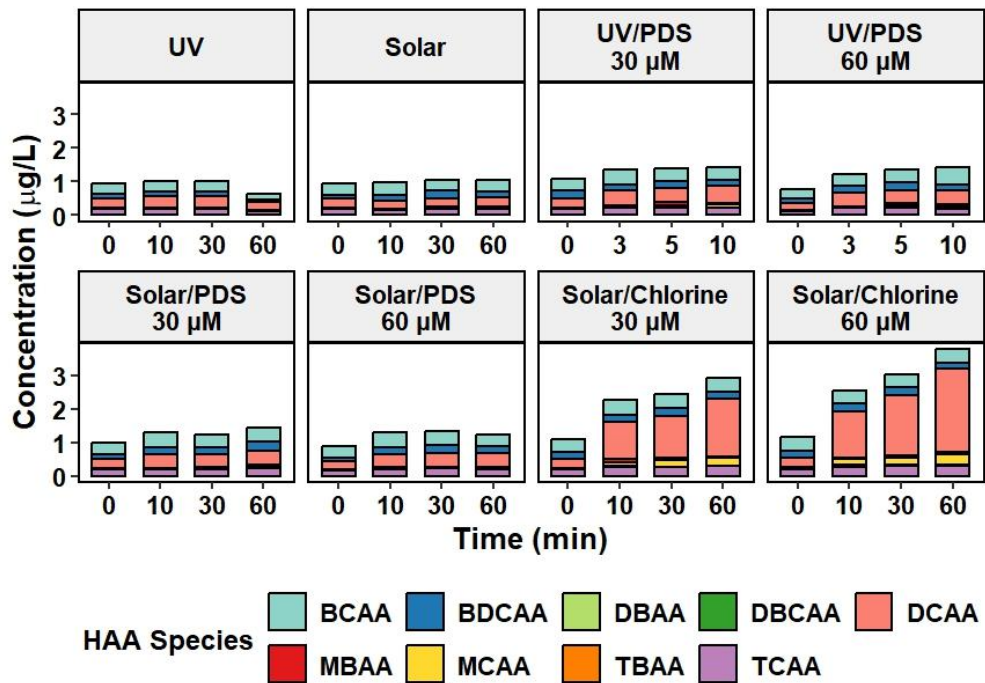


Figure 4.24 HAAs formation in the PPCP treatment with different AOPs
(Condition: Solar simulator: 9 mW/cm², 500W; UV: 8 mW/cm², 14W, 254nm LPUV)

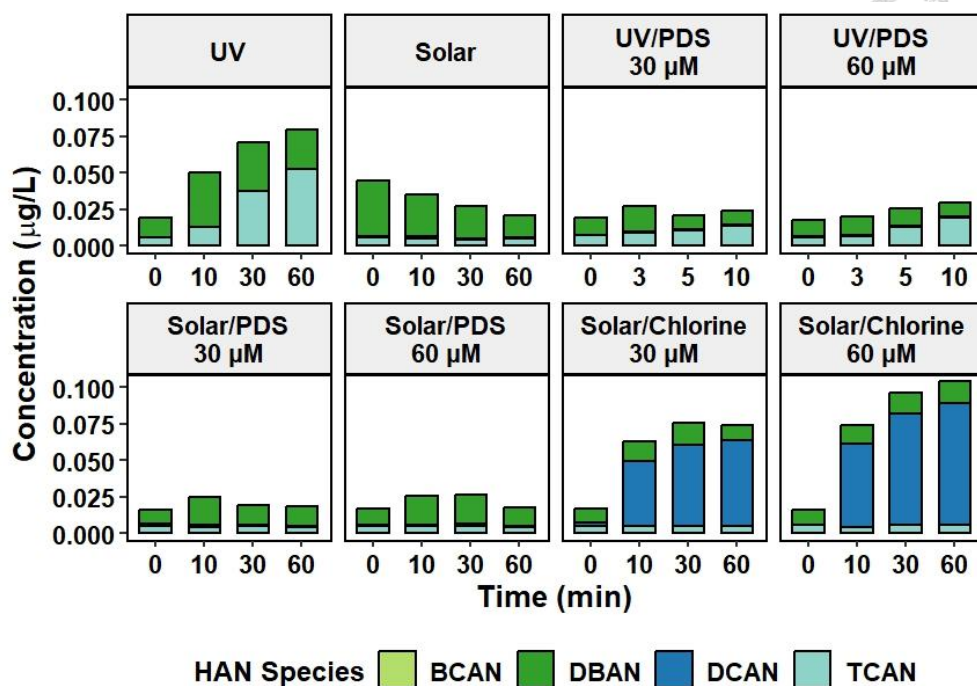
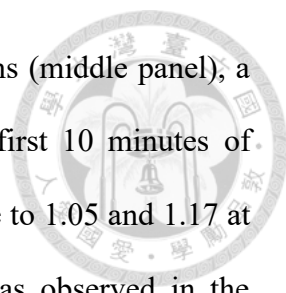


Figure 4.25 HANs formation in the PPCP treatment with different AOPs
(Condition: Solar simulator: 9 mW/cm², 500W; UV: 8 mW/cm², 14W, 254nm LPUV)

4.5 Changes of NPDOC

Figure 4.26 illustrates the changes in non-purgeable dissolved organic carbon (NPDOC) across the different treatment systems in wastewater. Overall, the extent of mineralization of organic matter was negligible, likely due to the relatively low oxidant doses applied.

In the UV/Solar only group (left panel), the solar-irradiation-only system maintained a highly stable NPDOC profile ($C/C_0 \sim 0.99$ - 1.02 throughout the 60-minute reaction time). Conversely, the UV-only group exhibited an initial increase to $C/C_0 = 1.09$ (+9%) after 10 minutes of reaction, followed by a slight decrease to 1.04 at 60 minutes. This distinct initial peak suggests that UV light alone can cleave the particulate organic matter (POM) or larger colloids into smaller dissolved organic molecules, thereby increasing the fraction of NPDOC that is detectable.



With the introduction of oxidants in the solar activation systems (middle panel), a rapid structural transformation of the matrix emerged within the first 10 minutes of reaction. For instance, in the Solar/PDS groups, the C/C_0 values rose to 1.05 and 1.17 at 30 μM and 60 μM PDS, respectively. A similar upward trend was observed in the Solar/Chlorine groups, where C/C_0 reached 1.12 at 30 μM of chlorine and 1.03 at 60 μM of chlorine. This phenomenon indicates that at lower treatment doses, the generated reactive species (such as sulfate radicals and reactive chlorine species) preferentially and aggressively attack complex dissolved organic matter (DOM) rather than the trace contaminants. The large, complex organic aggregates were partially oxidized and broken down into smaller dissolved intermediates, temporarily elevating the NPDOC concentration. Subsequently, from 10 to 60 minutes of reaction, these smaller intermediates underwent slow, progressive oxidation, resulting in a gradual decrease in C/C_0 (to 1.04 for Solar/Chlorine at 30 μM of chlorine and to 0.93 at 60 μM of chlorine), although the net mineralization remained minimal. In the Solar/PDS groups, the change in NPDOC concentration is merely small after a 60-minute reaction. The final C/C_0 are 1.06 and 1.16 because the oxidation ability of DOM in Solar/PDS is lower.

In the UV activation systems (right panel), this partial oxidation occurred at an accelerated rate due to the higher energy input. Under both UV/PDS dosages (30 and 60 μM), the NPDOC concentration peaked rapidly within 3 to 5 minutes of reaction, reaching C/C_0 values of 1.15 and 1.11, respectively, before beginning to decline toward 10 minutes of reaction ($C/C_0 \sim 1.09$). This rapid accumulation of partially oxidized DOM intermediates reinforces the previous finding that natural matrix organic scavengers dominate the consumption of available oxidants, thereby directly impairing the degradation efficiency of the target antibiotics.

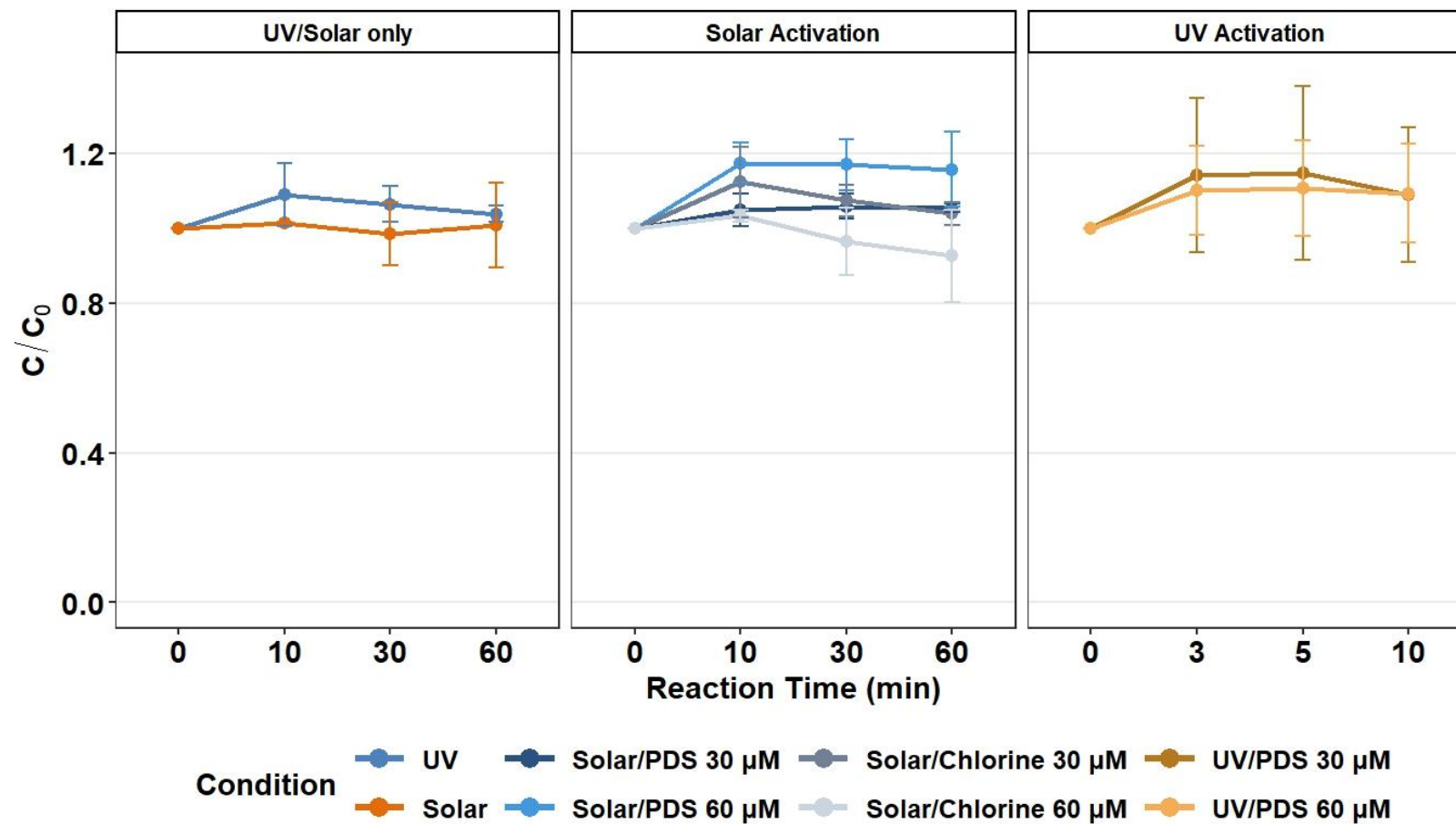


Figure 4.26 Changes of NPDOC concentrations during the different AOP treatment of PPCPs
(Condition: Solar simulator: 9 mW/cm², 500W; UV: 8 mW/cm², 14W, 254nm LPUV)

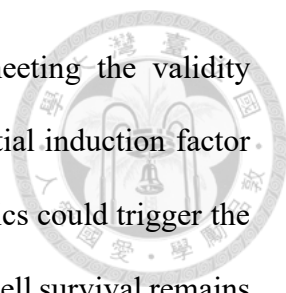
4.6 Results of Toxicity Assay for AOP treated waters

Although AOPs effectively degrade parent compounds, their degradation does not necessarily reduce the overall toxicity. The formation of DBPs or TPs may either increase or decrease toxicity depending on the reaction pathways. Therefore, both cytotoxicity and genotoxicity were evaluated to better understand the evolution of toxicity during antibiotic degradation.

To clarify antibiotic toxicity trends in the study, toxicity tests were first conducted in Milli-Q water groups. Since oxidant-containing systems are expected to generate more complex reaction products than light irradiation alone, only the three oxidation processes (Solar/Chlorine, Solar/PDS, and UV/PDS), each operated at an oxidant dosage of 50 μM , were further investigated. Although serial dilutions were prepared across each row of the 96-well plate, only four representative concentrations were selected for presentation of the results. Accordingly, the concentrations shown in Figure 4.27 represent the final well concentrations, where 100%, 25%, 6.25%, and 3.13% correspond to the initial sample concentrations (T_0) of 5, 0.125, 0.3125, and 0.156 ppm, respectively.

All measurements were performed in triplicate ($n = 3$). The cytotoxicity results, expressed as *E. coli* survival, showed no obvious toxic effects throughout the three AOP systems. Although relatively large error bars were observed, they were likely due to sample pretreatment and concentration losses. Instead of reaction time, cell survival was primarily governed by the dilution effect. Survival remained consistently low at the undiluted (100%) samples and gradually increased with dilution, reaching approximately 90% at a dilution of 3.13%.

Across all treatment times (0 - 60 min), only minor variations were observed, suggesting that none of the three AOPs substantially altered acute toxicity over the treatment period. Genotoxicity was subsequently evaluated using the 6.25% dilution, as



cell survival at this concentration consistently exceeded 80%, meeting the validity criterion for the SOS ChromoTest. As shown in Figure 4.28, the initial induction factor (IF) for all systems was close to 2, indicating that the parent antibiotics could trigger the *E. coli* SOS response before oxidation. When the IF exceeds 2 while cell survival remains above 80%, the sample is generally considered to exhibit potential genotoxicity. In the three AOPs, the Solar/Chlorine system exhibited the most rapid reduction in genotoxicity, with the IF decreasing markedly within the first 10 minutes of reaction and remaining relatively stable thereafter. This behavior is likely associated with the rapid degradation of the parent compounds, as shown in Figure 4.14.

The Solar/PDS system displayed an apparent lag period during the first 10 minutes. This temporary delay may indicate the formation of intermediate products that exert stronger DNA-damaging effects than the parent compounds, which are subsequently degraded after prolonged treatment. Although UV/PDS also reduced genotoxicity more rapidly than Solar/PDS, the current sampling intervals do not provide sufficient evidence to conclude that UV/PDS detoxifies more rapidly than Solar/Chlorine.

Although the main pattern of toxicity tests showed an increase first and then a decrease in the literature review, previous studies have reported similar patterns of toxicity evolution ^[118,120,122], in which genotoxicity initially decreases as toxic transformation products are degraded, but subsequently increases due to the formation of smaller, potentially more toxic intermediates. However, the toxicity of these intermediates gradually decreases after longer reaction time.

Furthermore, sulfate radical-based oxidation has also been reported to preferentially produce nitrated transformation products with relatively higher toxicity than hydroxyl radical systems, whereas chlorine-based AOPs tend to shift the toxicological risk toward halogenated DBPs ^[113,117,122,152].

No significant changes in cytotoxicity were observed during treatments, all three AOPs effectively reduced genotoxicity compared with the untreated samples; UV/PDS and Solar/Chlorine reduced genotoxicity significantly in the first 3 or 10 minutes; Solar/PDS also showed a descending trend after longer treatment time. Nevertheless, these findings should be considered preliminary, as the SOS ChromoTest provides only a screening-level assessment of genotoxicity, not a comprehensive evaluation of ecotoxicological risk.

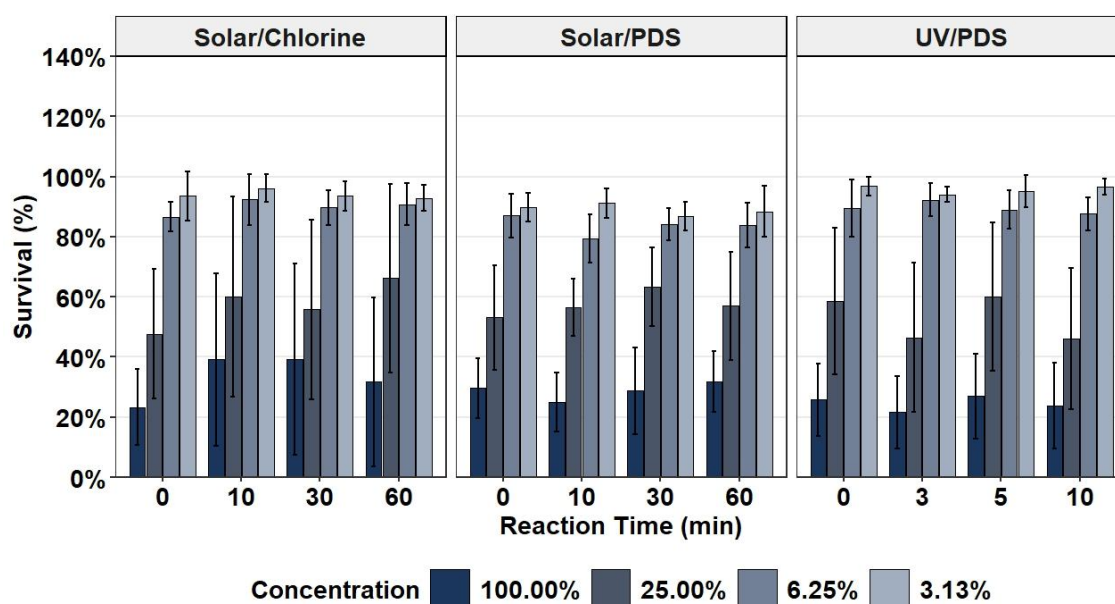


Figure 4.27 Survival of *E. coli* in each AOP system

([PPCP]₀=5 mg/L, [PDS]₀=[Chlorine]₀=50 μM, Solar simulator condition: 9 mW/cm², 500W, UV condition: 8 mW/cm², 14W, 254nm LPUV)

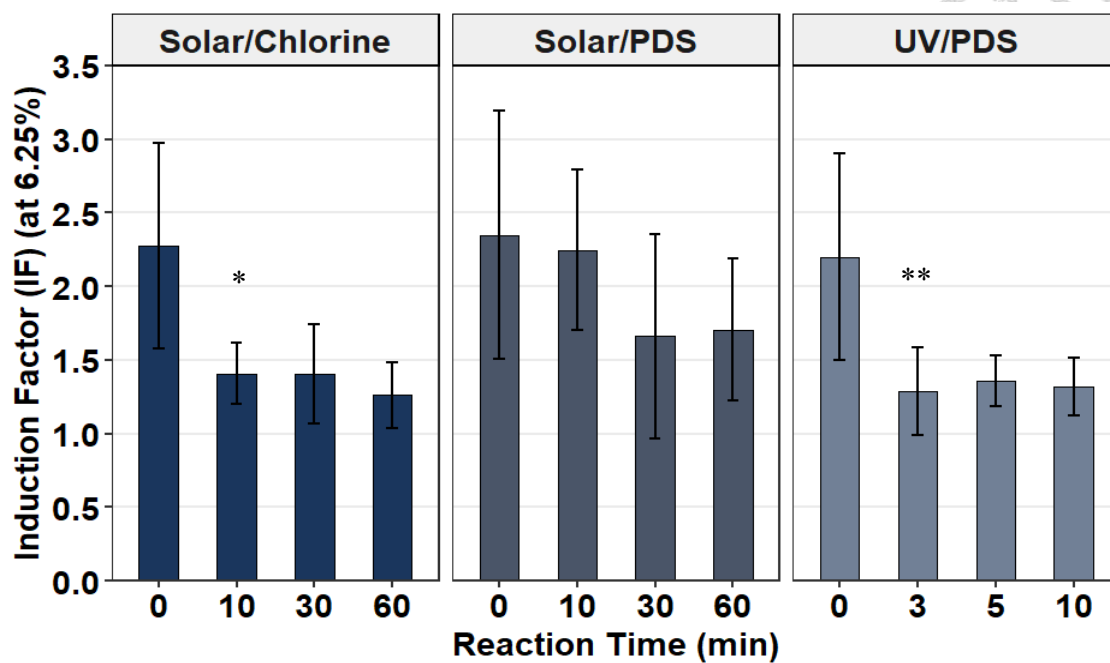
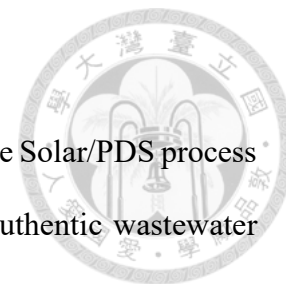


Figure 4.28 IF in each AOP system at 6.25% group

([PPCP]₀=5 mg/L, [PDS]₀=[Chlorine]₀=50 μM, Solar simulator condition: 9 mW/cm², 500W, UV condition: 8 mW/cm², 14W, 254nm LPUV, * p<0.05, ** p<0.01)

Chapter 5 Conclusions and Suggestions



This study primarily evaluates the feasibility and efficiency of the Solar/PDS process for removing trace-level antibiotics from both Milli-Q water and authentic wastewater matrices. To address the limitations of conventional wastewater treatment, where trace-level antibiotic residues may persist, and chlorination-based processes can lead to disinfection by-product formation, alternative advanced treatment strategies are needed. In this context, peroxydisulfate (PDS) activation can generate highly reactive sulfate radicals, which have a relatively long lifespan in aqueous environments, while solar irradiation serves as an environmentally friendly and sustainable energy source. To thoroughly assess whether this integrated system can function effectively and safely within aquatic environments, a comprehensive evaluation encompassing degradation kinetics and toxicological safety is necessary.

In the wastewater results, the overall treatment efficiency decreased due to the complex wastewater matrix, which reduced light penetration and scavenged reactive radicals. Solar/PDS demonstrated higher removal efficiency compared to Solar/Chlorine at equivalent oxidant dosages. For the removal of trace-level contaminants from wastewater, PDS achieved higher removal efficiency than chlorine for several target compounds, particularly levofloxacin. However, despite the sustainable energy aspect of Solar-driven PDS, ultraviolet (UV) irradiation remains essential for the removal of certain compounds. Only within a UV-based system can N-acetyl Sulfamethoxazole, the principal metabolite of Sulfamethoxazole, be effectively degraded during the treatment.

Beyond the targeted contaminants removal efficiencies, the analysis of residual oxidants revealed distinct chemical behaviors between the two systems. Specifically, the apparent PDS consumption was lower than the chlorine depletion, as indicated by the higher residual PDS concentrations throughout the reaction period. Conversely, the rapid

depletion of free chlorine was primarily driven by its high reactivity with the dissolved organic matter (DOM) naturally present in wastewater. This rapid interaction between chlorine and organic matter in wastewater was consistent with the observed formation of chlorinated DBPs.

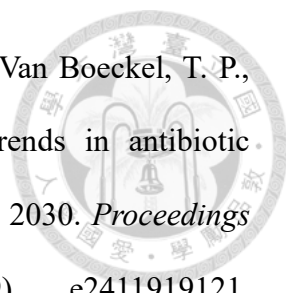
This study also evaluated cytotoxicity and genotoxicity during antibiotic degradation involving three advanced oxidation processes (AOPs): Solar/Chlorine, Solar/PDS, and UV/PDS. While none of these processes significantly influenced acute cytotoxicity, all three effectively reduced genotoxicity. Nonetheless, these findings are preliminary and require further investigation to support a comprehensive ecotoxicological risk assessment. To fully establish the environmental safety of these AOPs, additional studies focusing on chronic toxicity profiles and transformation product (TP) identification are needed.

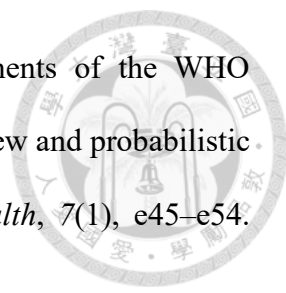
Overall, Solar-driven PDS oxidation represents a promising alternative strategy for reducing trace-level antibiotics in hospital wastewater. While Solar/PDS provides a sustainable treatment option with effective removal performance, UV/PDS remains necessary to degrade specific metabolites, such as NASMX. Therefore, understanding contaminant-specific reactivity and wastewater matrix effects is essential for optimizing Solar/PDS performance while identifying situations where complementary UV activation may be required. Future studies should focus on long-term ecotoxicological assessments and comprehensive identification of transformation products (TPs) to further evaluate the environmental safety and practical applicability of Solar/PDS systems.

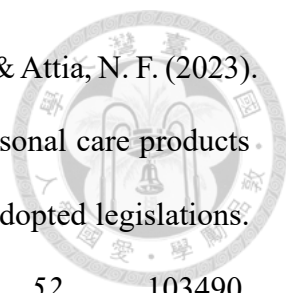
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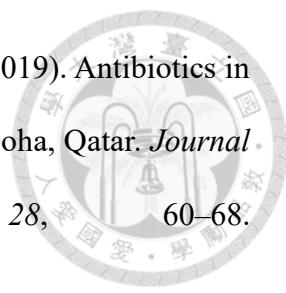
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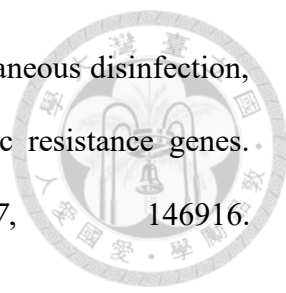
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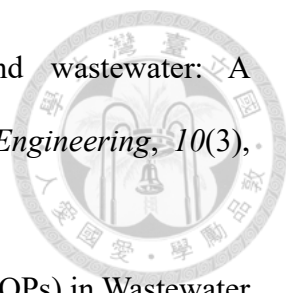
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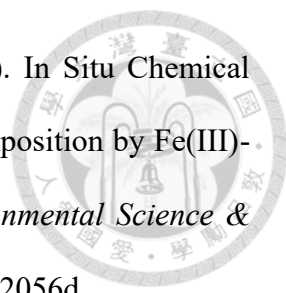
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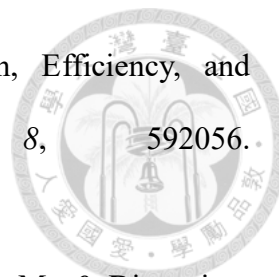
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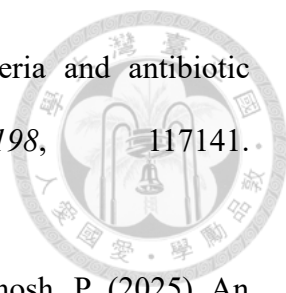
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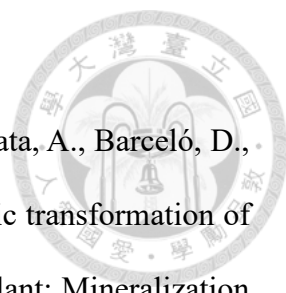
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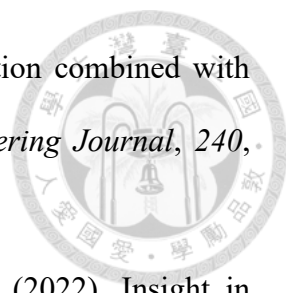


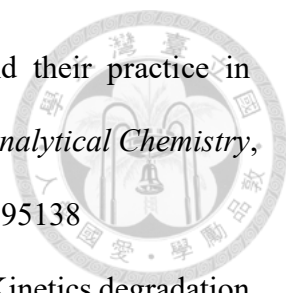
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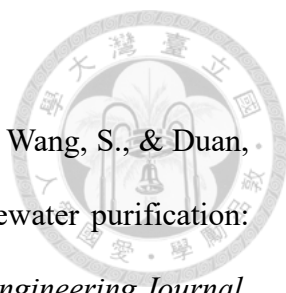
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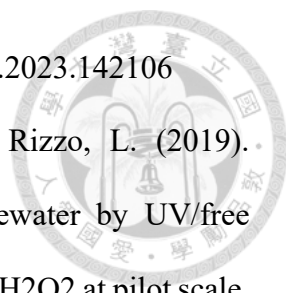
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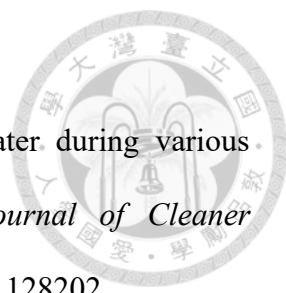
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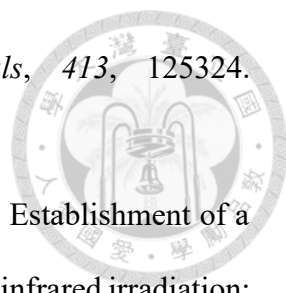
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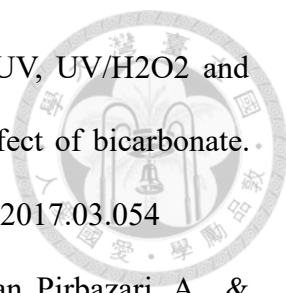
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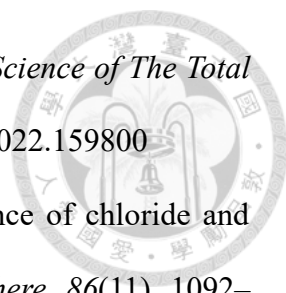
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APPENDICES



1. Preparing stock solutions of the standards

(1) Materials

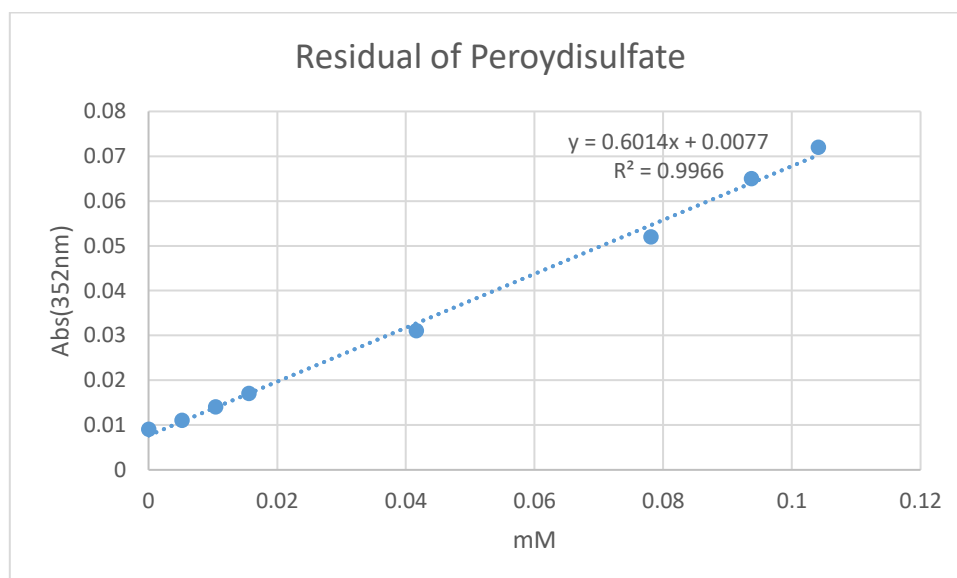
- A. Sulfamethoxazole, analytical graded, $\geq 99.0\%$, 1000 $\mu\text{g/mL}$ in methanol, Absolute Standards, Hamden, CT, USA
- B. Trimethoprim, analytical graded, $\geq 99.0\%$, 1000 $\mu\text{g/mL}$ in methanol, Absolute Standards, Hamden, CT, USA
- C. N-Acetyl Sulfamethoxazole, analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada
- D. Levofloxacin, HPLC graded, $\geq 98\%$, Sigma-Aldrich, St Louis, MO, USA
- E. Clarithromycin, analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada
- F. Azithromycin, analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada
- G. Sulfamethoxazole- d_4 , analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada
- H. Trimethoprim- d_9 , analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada
- I. Levofloxacin- d_8 , analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada
- J. Clarithromycin-N-methyl- ^{13}C , d_3 , analytical graded, $\geq 95.0\%$, Toronto Research Chemicals, Toronto, ON, Canada

(2) Methods

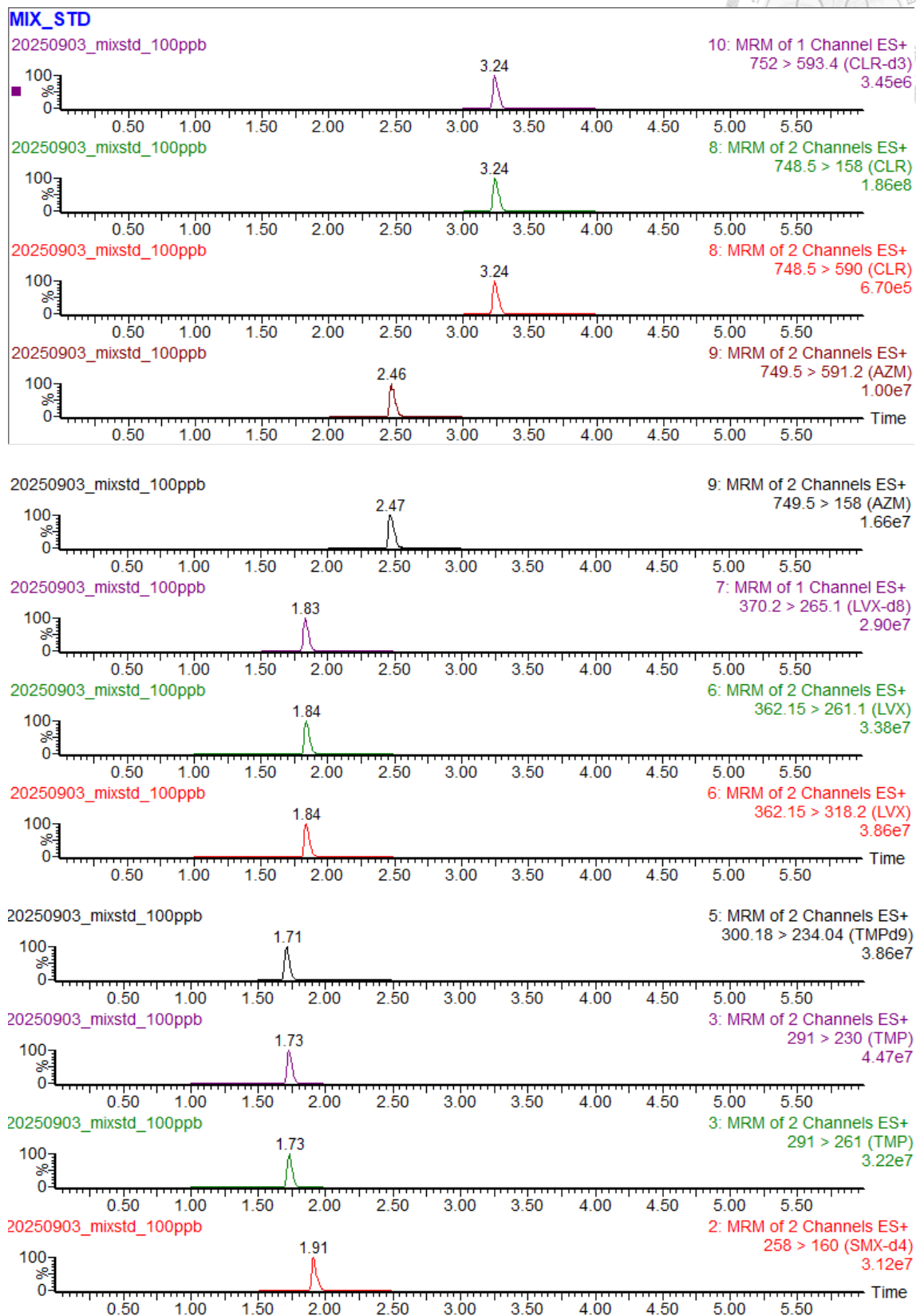
Except for Sulfamethoxazole and Trimethoprim, which already dissolve in methanol,

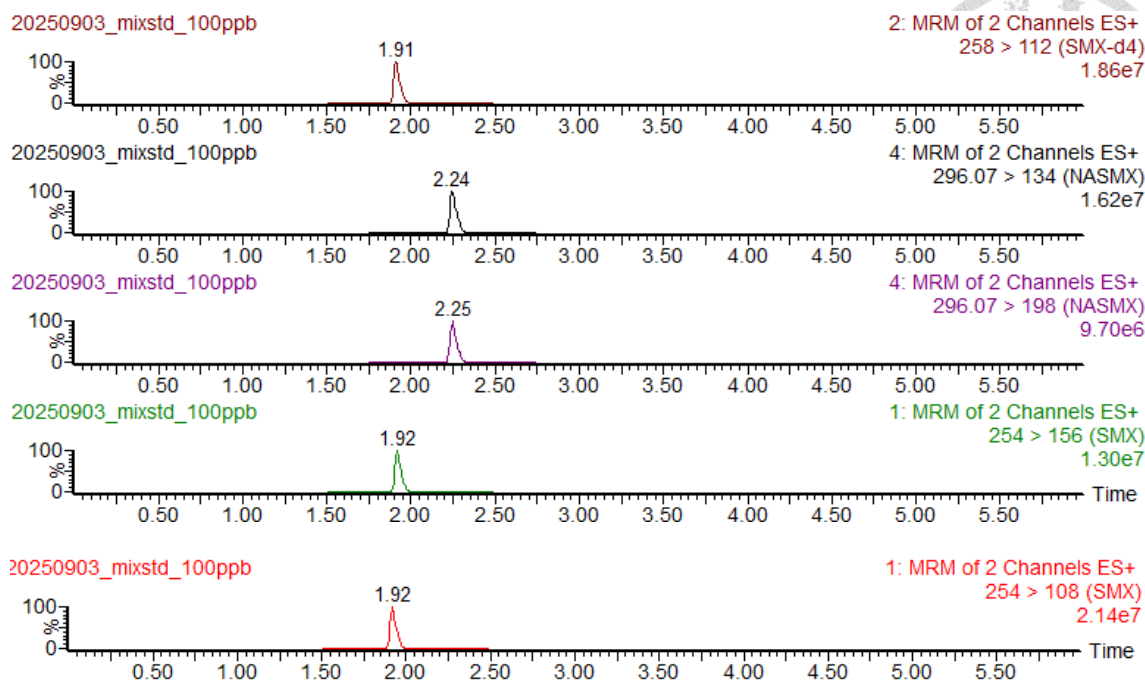
other standards are in the solid phase. N-Acetyl Sulfamethoxazole, Levofloxacin, Clarithromycin, and Azithromycin were prepared by dissolving 10 mg of each standard in 4 mL of methanol, yielding a 2500 µg/mL stock solution. As for internal standards, Sulfamethoxazole-d₄ was prepared by dissolving 5 mg in 4 mL of methanol, yielding a 1250 µg/mL stock solution. Trimethoprim-d₉, Levofloxacin-d₈, and Clarithromycin-N-methyl-¹³C, d₃ were prepared by dissolving 1 mg in 4 mL of methanol, yielding a 250 µg/mL stock solution.

2. Calibration curve of PDS in the study



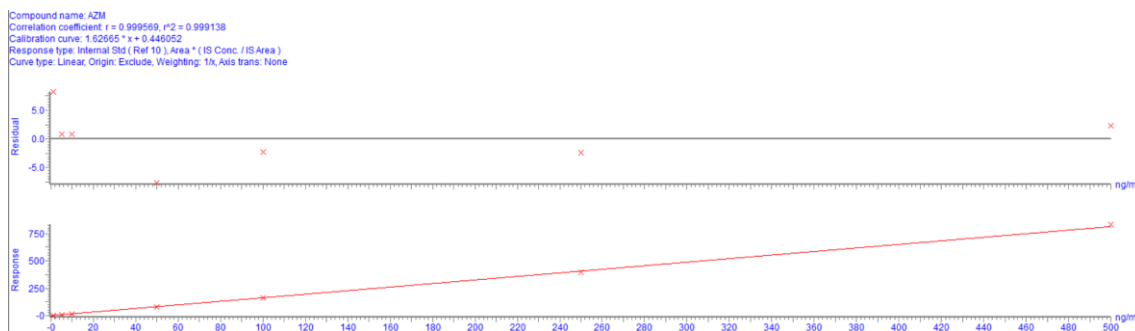
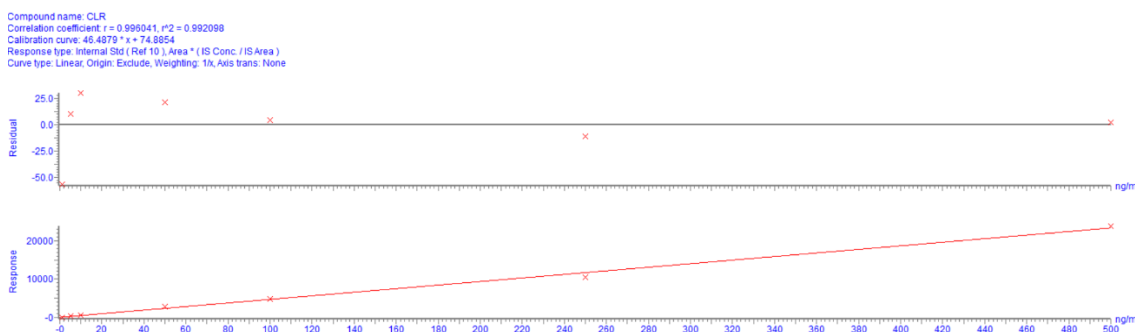
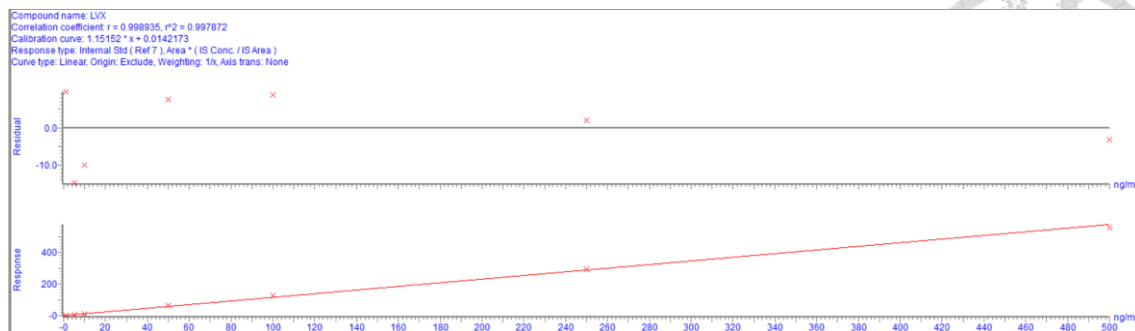
3. Peaks of selected Antibiotics with internal standard included





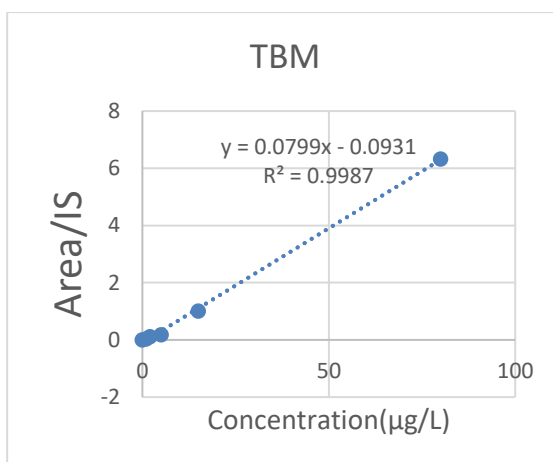
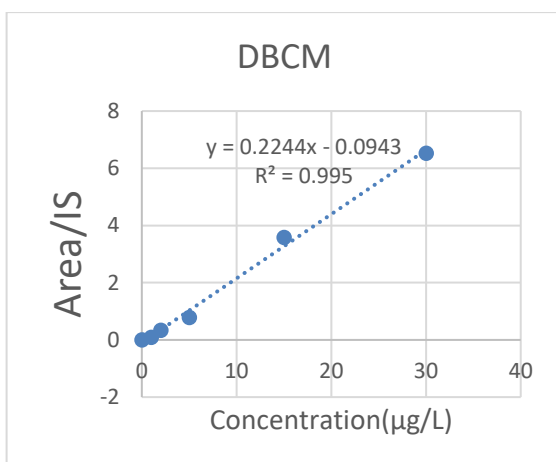
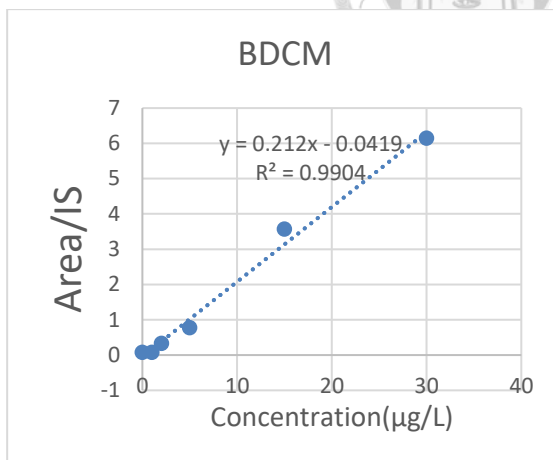
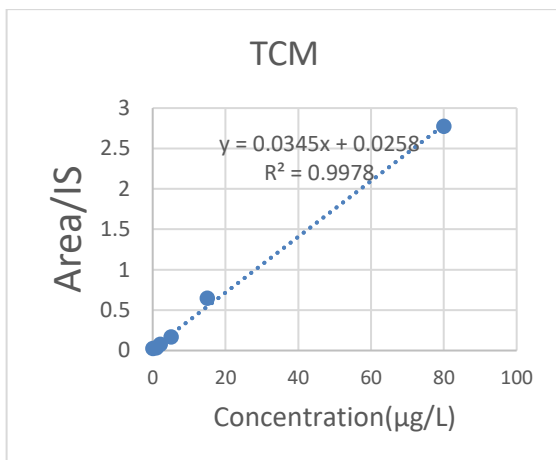
4. Calibration curve of selected antibiotics



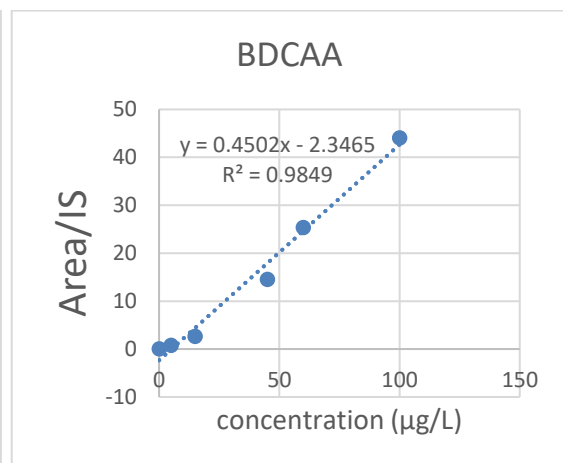
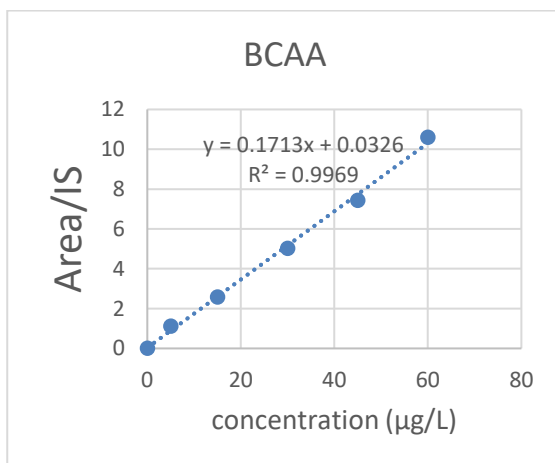
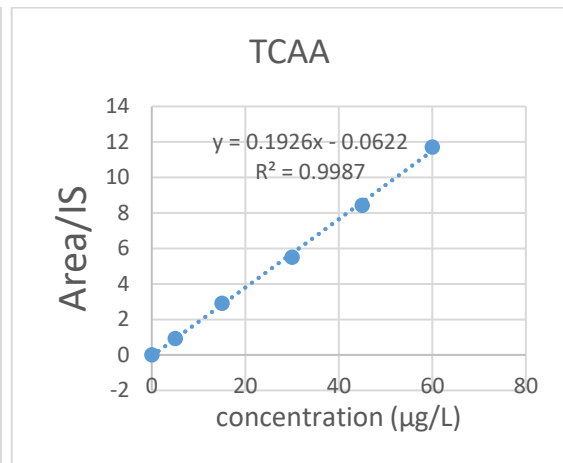
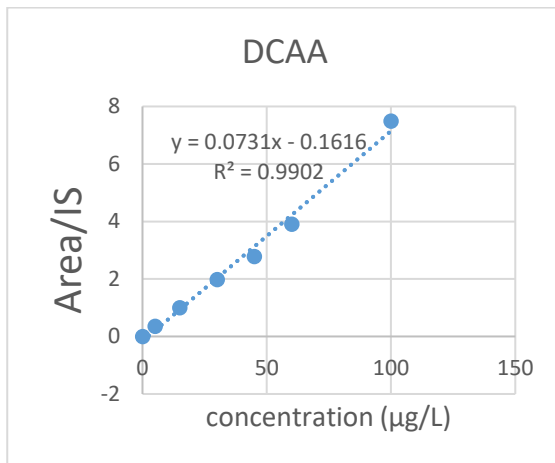
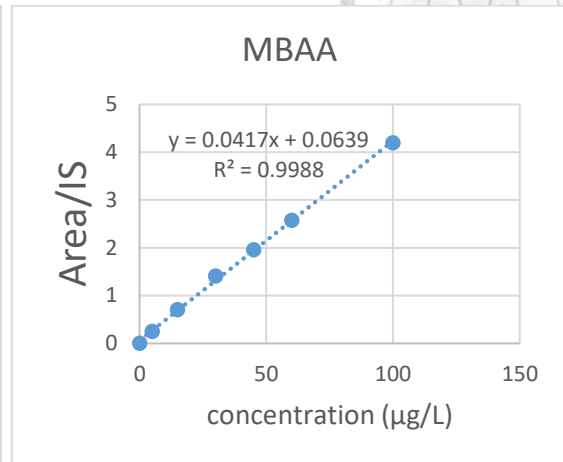
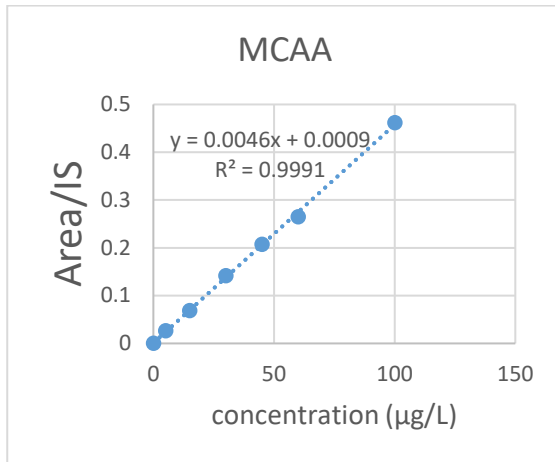


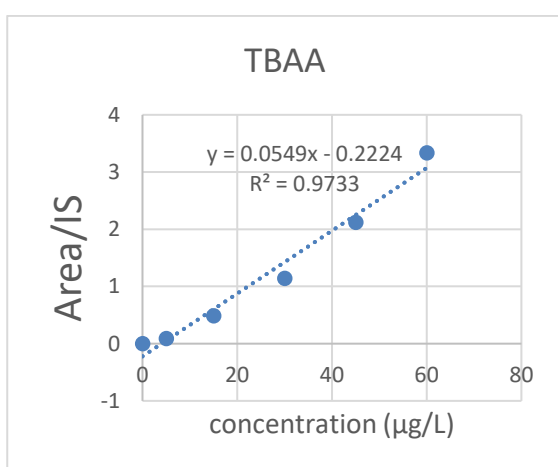
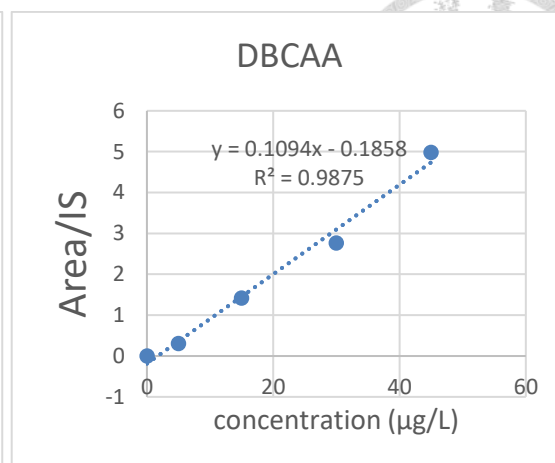
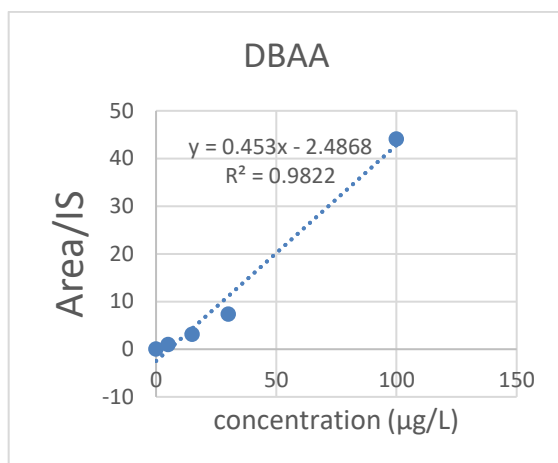
5. Calibration curves of DBPs

(1) THMs

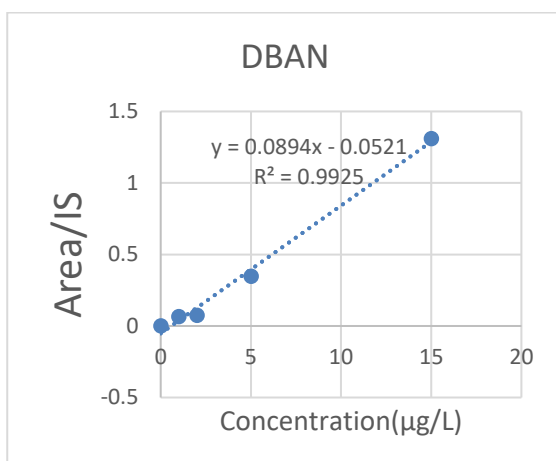
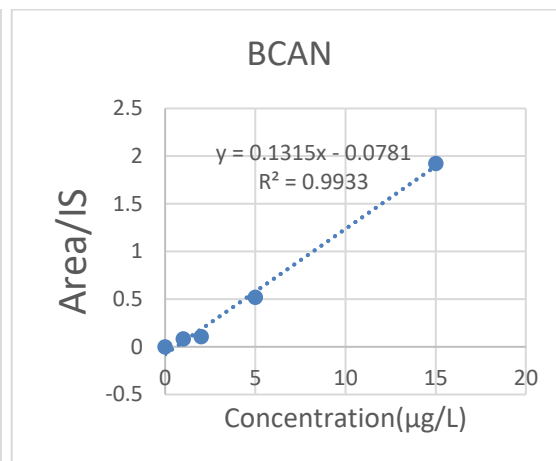
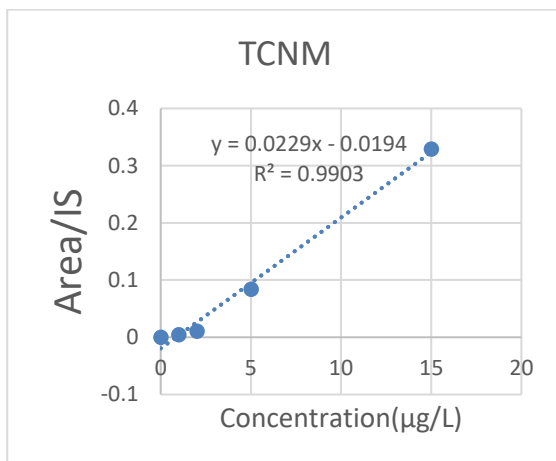
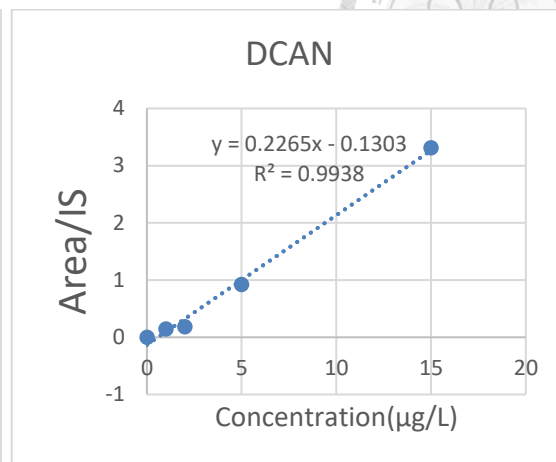
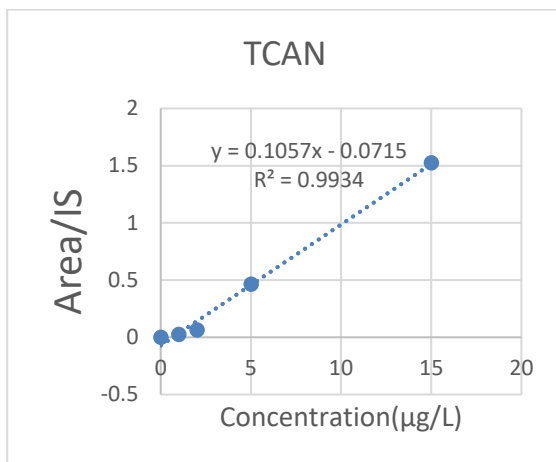


(2) HAAs



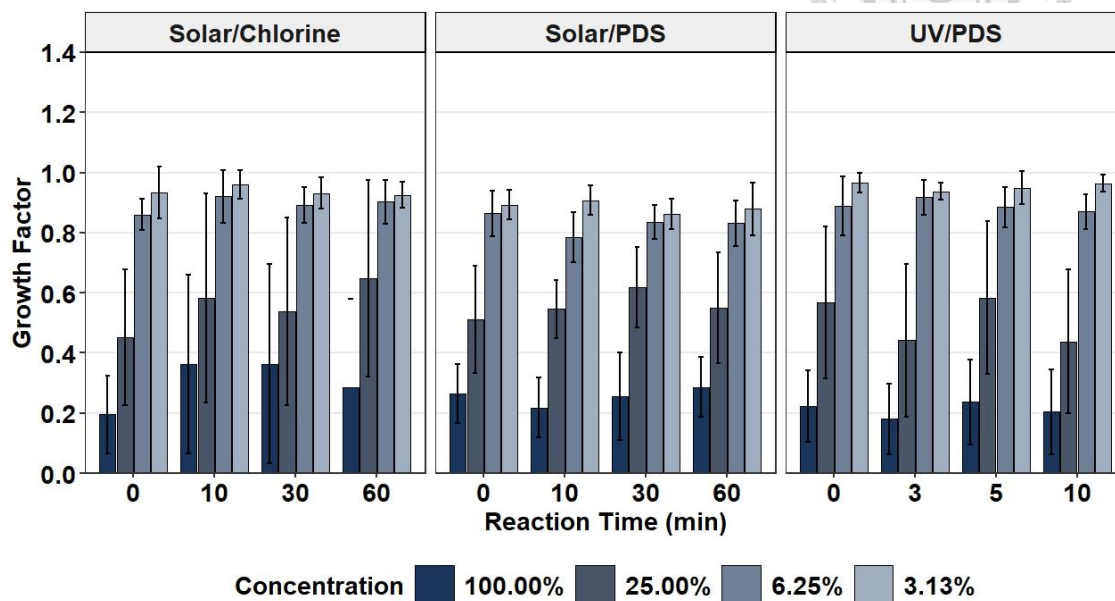


(3) HANs

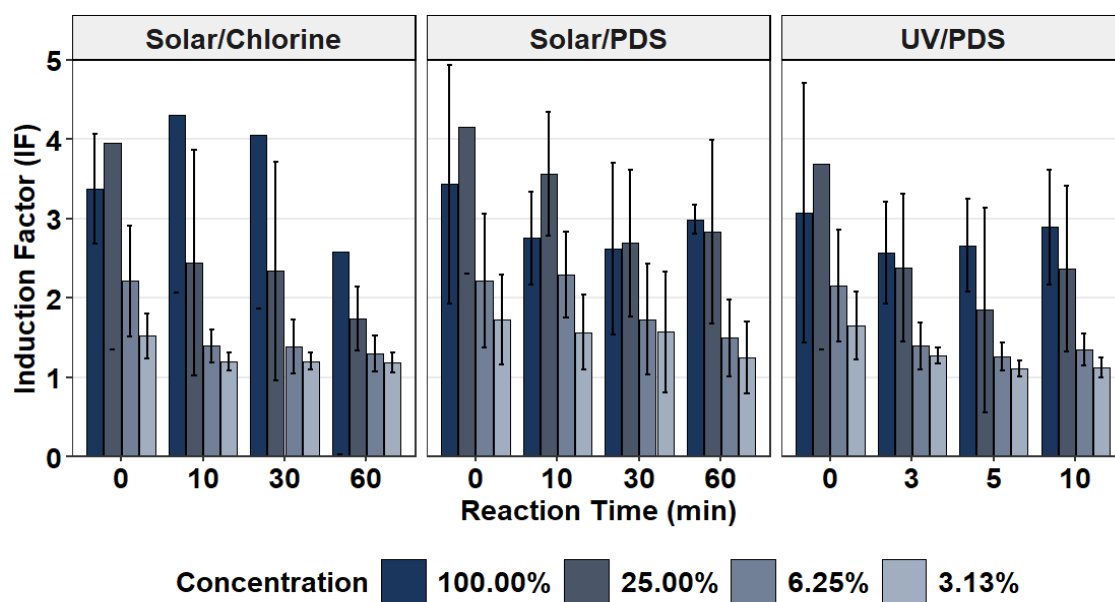


6. Toxicity

(1) Growth Factor of each group

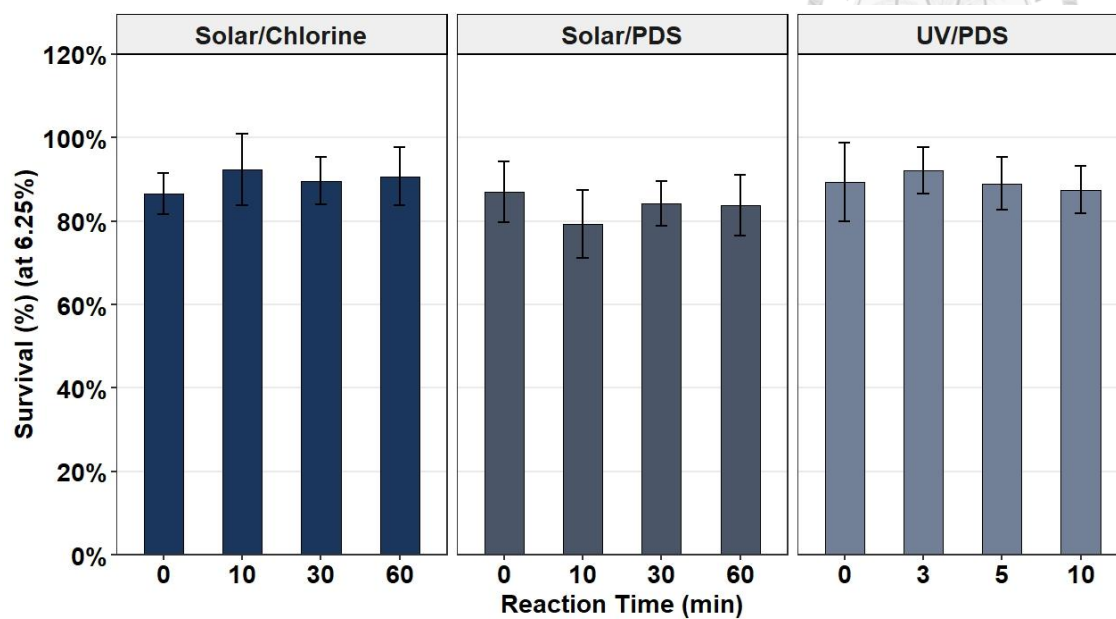


(2) IF of each group





(3) Survival rate of 6.25% group



(4) Growth Factor of 6.25% group

