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Coupling Electro-oxidation and UV Photolysis for Amoxicillin and Ibuprofen Synthetic Wastewater Treatment

Solomon Ali Yimam

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In partial fulfillment of the requirements for the attainment of the degree of Doctor of Philosophy in Chemical Engineering (Environmental Engineering)

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Declaration

This is to declare that this dissertation is submitted to the School of Graduate Studies of Addis Ababa University for the degree of Doctor of Philosophy (PhD) in Environmental Engineering. And I would like to corroborate that it is my own work and all other works used in the dissertation are well acknowledged.

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Abstract

The rapid growth and population increase globally have led to a worsening of water pollution. One of the significant environmental concerns is the presence of pharmaceutical pollutants in water. These pharmaceutical pollutants are recalcitrant and are typically found in low concentrations in water matrices. Conventional treatments used to treat municipal wastewaters are ineffective in removing pharmaceuticals. Recently, advanced oxidation processes (AOPs) such as EO and UV have shown promising results in effectively mineralizing pharmaceutical pollutants. The following studies examined the effectiveness operational parameters in active chlorine generation using EO and coupling EO and UV for the treatment of AMOX and IBU. The first study focused on investigating the combined effects of operational parameters on residual chlorine production using response surface methodology (RSM) and artificial neural network (ANN) techniques. The study evaluated the impact of sodium chloride concentration, electrical potential, electrolysis time, and electrode gap on residual chlorine production and energy consumption. The optimal conditions for residual chlorine production were found to be an electrical potential of 8.8 V, an electrolysis time of 25 minutes, a sodium chloride concentration of 25 g/L, and an electrode distance of 1 cm, resulting in a residual chlorine level of 2450 mg/L and an energy consumption of 21.76 kWh/L. It was revealed that electric potential, sodium chloride concentration, and electrolysis time had a positive influence on residual chlorine production. The study also indicated that the ANN models outperformed the RSM models in terms of prediction ability, suggesting the potential use of electrolysis for active chlorine production from saline solutions in industrial and water disinfection applications. In the subsequent experiment, electro-oxidation (EO) of synthetic wastewater containing amoxicillin (AMOX) and Ibuprofen (IBU) was carried out using Ti/IrO_2 electrodes in a batch reactor setup. The optimization of the EO process was performed using response surface methodology (RSM) to examine the effects of pH, current density, and initial concentrations of AMOX and IBU on the degradation of pollutants and Chemical Oxygen Demand (COD) removal. The optimal conditions for the degradation of AMOX and IBU were determined to be pH 3, current density of 10 mA cm^{-2}, and initial concentrations of AMOX and IBU at 276 µg/L,

resulting in COD removal efficiencies of 66.825% and 81.925%, respectively. Furthermore, combining EO with UV led to an increase in Total Organic Carbon (TOC) removal for IBU as the electrolysis time was extended, enhancing mineralization. The degradation rate of IBU was observed to be faster than that of AMOX when subjected to EO-UV processes over a specified duration, possibly due to the sensitivity of IBU's chemical structure to UV photolysis. For AMOX, at the initial stage of treatment the Total Organic Carbon (TOC) increased from 4.16 ppm to 6.81 ppm. Carbon cannot be generated during oxidation. This is an apparent increase in carbon is likely due to the release of previously undetected organic carbon into the dissolved phase. It may also be attributed to the formation of soluble oxidation intermediates and/or analytical uncertainty associated with TOC measurements at low concentration levels. Continued treatment resulted in a subsequent decrease in TOC. This decrease indicates progressive mineralization of the oxidation intermediates. After 30 minutes of treatment, there was the formation of transient intermediates prior to complete oxidation to carbon dioxide (CO_2). On the other hand, for IBU, the TOC values decreased from 6.07 ppm to 4.86 ppm after 30 minutes of treatment, demonstrating efficient mineralization. Extending the treatment duration to 40, 50, and 60 minutes has been found to result in a significant reduction in TOC levels, leading to improved mineralization. The presence of transformation products and reaction intermediates of AMOX and IBU has allowed for the proposal of a feasible degradation pathway for these compounds through anodic oxidation using Ti/IrO_2 electrodes. In this study, there is no identification of intermediates conducted using advanced analytical techniques but the term "partial oxidation products" was used to refer generally to transformation products formed during the degradation of AMOX and IBU before complete mineralization to CO_2 and H_2O. Studies showed that intermediates include penicilloic acid, penilloic acid, diketopiperazine derivatives, and products formed through β-lactam ring opening when AMOX undergoes oxidation. While IBU is oxidized the intermediates include hydroxy-ibuprofen, carboxy-ibuprofen, 4-isobutylacetophenone, and other hydroxylated or decarboxylated derivatives. Also the proposed degradation pathways was obtained from published literatures and it was hypothetical stemmed from based on known reaction mechanisms rather than direct identification of intermediates in this study. This study didn't include degradation

kinetics and mechanism. To assess the safety of the treated wastewater, toxicity tests were conducted using E.coli, revealing the formation of toxic intermediates and a significant inhibition of E. coli growth when electrolysis (EO) was used independently. When EO alone was used the percentages of colonies for the corresponding dilutions (25%, 50%, and 75%) are 16.2%, 24%, and 60%, which exceed the 10% threshold for IR that makes the wastewater unsafe. However, when EO-UV was applied the safety threshold for IR (< 10%) was achieved within the first 25 minutes for both AMOX and IBU, indicating a more rapid production of less toxic treated wastewater. The integration of EO and UV has shown advantages in terms of minimized treatment time and treatment efficiency compared to the use of EO alone processes. In addition, coupling EO with UV effectively reduced the toxicity associated with intermediates formed from AMOX and IBU treatment. The reduction in toxicity is attributed to the generation of chlorinated byproducts and short-chain carboxylic acids, such as acetic acid and oxalic acid, when EO is used in conjunction with UV, which exhibit lower toxicity towards E. coli growth.

Keywords: Pharmaceutical pollutants; Electro-oxidation ; UV photolysis; Electro-oxidation–UV; Active chlorine generation; Amoxicillin; Ibuprofen; Wastewater treatment.

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List of Abbreviations

ANOVA	Analysis of Variance
AO	Anodic Oxidation
AOPs	Advanced Oxidation Processes
BDD	Boron Doped Diamond
CCD	Central Composite Design
CE	Current Efficiency
CF	Carbon Felt
CNT	Carbon Nano Tubes
COD	Chemical Oxygen Demand
CRV	Central Rift Valley
CV	Coefficients of Variation
DBPs	Disinfection By Products
DPD	N, N-diethyl-p-phenylenediamine
DSA	Dimensionally Stable Anode
EDs	Endocrine Disruptors
GC-MS	Gas Chromatography-Mass Spectrometry
HPLC	High Performance Liquid Chromatography
PPCPS	Personal Care and Pharmaceutical

	Products
PEF	Photo-Electro-Fenton
PTFE	Poly Tetra Fluoro Ethylene
RSM	Response Surface Methodology
RVC	Reticulated Vitreous Carbon
SEC	Sono-Electro Chemistry
SEM	Scanning Electron Microscope
SHE	Standard Hydrogen Electrode
EO	Electro-oxidation
AMOX	Amoxicillin
IBU	Ibuprofen
GAC	Granular Activated Carbon
CAS	Conventional Activated Sludge

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1. Yimam, S. A., Kang, J. W., & Kassahun, S. K. (2024). *Applications of integrated response surface methodology statistical techniques and artificial neural network–based machine learning to optimize residual chlorine production and energy consumption*. *Water and Environment Journal*, *38*(3), 373-384. **https://doi.org/10.1111/wej.12922**

2. Yimam, S. A., Kassahun, S. K., & Kang, J. W. (2025). *Coupling electro-oxidation and UV for the treatment and mineralization of amoxicillin and ibuprofen from aqueous solution: RSM parametric optimization and toxicity assessment*. *Results in Engineering*, *26*, 106869. **https://doi.org/10.1016/j.rineng.2025.106869**

Chapter One

Introduction

1.1. Background

Over the past decades, the production and consumption of pharmaceuticals has increased tremendously which has brought worries about their intimidation to the ecosystem and the well-being of human beings, for these chemicals are discharged to the environment continuously and in uncontrolled manner [1]. These compounds are being detected in drinking water samples as well as in ground, and surface water [2]. Pharmaceuticals consumed by humans and animals undergo metabolism, and the remaining are excreted in feces and urine in their active forms. Thus, contaminating urban wastewater, manure, bio solids, and various other environmental matrices [3].

The presence of persistent pollutants, such as pharmaceuticals, in aquatic environments typically found in trace concentrations ranging from g L^{-1} and ng L^{-1} poses significant challenges for conventional treatment methods [4]. The existence of trace levels of pharmaceuticals could present toxicity risks and harm the ecosystem, potentially leading to the emergence of bacterial and antibiotic resistance genes in the environment [5]. It is therefore vital to remove pharmaceuticals from water sources and urban wastewater. To eliminate pharmaceuticals from wastewater several methods have been employed including biological processes, coagulation, sedimentation, adsorption, Ozonation, UV photo catalysis and photo electro catalysis [6] [7]

Generally, conventional treatment methods such as biological processes, coagulation, sedimentation and adsorption has low degradation efficiency over pharmaceuticals for these methods are not crafted for pharmaceutical removal and pharmaceuticals are non-biodegradable [8][9]. Of all these methods, great deal of attention has been given to the advanced oxidation processes (AOPs) for it uses an onsite produced reactive species such as hydroxyl radicals (.OH), hypochlorite ions ($HClO.,ClO^-$) [10]. Some of the most common AOPs are Ozonation, UV/hydrogen peroxide, electro-fenton, electrochemical oxidation and photo catalysis. Ozonation

is very effective over oxidizing pharmaceuticals yet because of the side reactions and unintended toxic chemical compounds formed it has deficiency in mineralizing the total organic carbon to carbon dioxide and water [9]. Photo catalysis demonstrates significant effectiveness in the removal of pharmaceuticals and total organic carbon (TOC); however, its high costs restrict its widespread use [6]. Also, UV photolysis consumes relatively much energy which increasing its operational cost [11].

Electro-Fenton reaction has gap of producing high ferric sludge and inefficient at neutral pH. Besides high concentration of electrolytes is required hence creating a saline final effluent [4][12]. In recent years, compared with these technologies notable progress has been achieved in electrochemical oxidation, attributed to its versatility, energy efficiency, potential for automation, environmental advantages, and cost-effectiveness, especially in the elimination of pharmaceuticals [13]. Even though, electrochemical oxidation has many advantages over many of the AOPs, single electro-oxidation for the treatment of pharmaceuticals has limitation of consuming high energy as a result of higher current densities required to achieve mineralization and the tendency of forming intermediate side products that may exhibit comparable and greater toxicity than the parent compound among others.

Combining electro-oxidation (EO) with UV irradiation not only requires electrical energy but also boosts energy efficiency through a synergistic effect. The treatment time needed for mineralization is reduced when these two methods are used together, leading to lower overall energy consumption.

In the EO process, oxidants such as HOCl/OCl^- and various reactive species are produced electrochemically. The application of UV irradiation subsequently photoactivates these oxidants, leading to the generation of highly reactive radicals (e.g. •OH and reactive chlorine species), which facilitate the degradation of pollutants. As a result, the desired removal efficiency can be attained in shorter treatment duration, with a lower current density, reduced electrode area, or decreased specific electrical energy consumption compared to using EO or UV independently. Thus, the advantage of the integrated EO/UV system is not in diminishing the total power input of

the UV lamp, but rather in enhancing the degradation rate and pollutant removal per unit of energy utilized.

Thus, hybridizing electrochemical oxidation with other methods enhances the efficacies obtained by the single electrolytic processes and minimizes the gap of electrochemical process [14].

UV irradiation is one of the most effective methods to increase the elimination of pharmaceutical compounds achieved by single EO process, involving activation by UV energy and the formation of active oxidant species from electrochemically produced oxidants. The following equations (Eqs 1-5) showed the formation of active oxidants when electro generated mediators are photo activated by UV irradiation energy (*hν*) which brings about chain reactions in the bulk solution [15].

$$HClO + h\nu \rightarrow \cdot OH + Cl\cdot \quad (1)$$

$$S_2O_8^{2-} + h\nu \rightarrow 2\,(SO_4^{-})\cdot \quad (2)$$

$$HBrO + h\nu \rightarrow \cdot OH + Br\cdot \quad (3)$$

$$H_2O_2 + h\nu \rightarrow 2\cdot OH \quad (4)$$

$$H_2O_2 + O_3 + h\nu \rightarrow 2\cdot OH + O_2 \quad (5)$$

Residual chlorine from electrolysis is a popular oxidant that is used in water disinfection and wastewater treatment. Indirect organic oxidation can occur through interactions with chlorine radicals (.Cl) adsorbed on the anode surface or by residual chlorine species in bulk solutions, such as Cl_2, HOCl, or OCl^- [17]. These oxidizing agents exist as gases (e.g. Cl_2 and ClO_2), liquids (e.g., NaClO, HClO, and NH_2Cl), and solids (e.g. $Ca(OCl)_2$).

Utilizing NaCl as the supporting electrolyte, it is scientifically justifiable to focus on reactive chlorine species (RCS) instead of hydroxyl radicals (•OH), due to the high concentration of chloride ions that are persistently transformed into active chlorine species during the electrolysis process. A high concentration of chloride ions promotes the formation of more reactive chlorine species than hydroxyl radical.

In electrolytes containing chloride, the initial anodic oxidation process converts chloride ions into chlorine gas and active chlorine species:

$$2Cl^- \rightarrow Cl_2 + 2e^-$$

$$Cl_2 + H_2O \rightleftharpoons HOCl + H^+ + Cl^-$$

$$HOCl \rightleftharpoons OCl^- + H^+$$

Under UV irradiation, these species are photolyzed:

$$HOCl + h\nu \rightarrow \cdot OH + Cl\cdot$$

$$OCl^- + h\nu \rightarrow O^-\cdot + Cl\cdot$$

The generated chlorine radicals can further react:

$$Cl\cdot + Cl^- \rightarrow Cl_2^-\cdot$$

Thus, in chloride-rich systems, reactive chlorine species such as Cl• and Cl_2 $\bullet^-$ are continuously produced and often become dominant oxidants. Electrolysis is the most common method for the production of residual chlorine. Electric current is passed by the electrodes into the electrolyte during electrolysis [18]. When sodium chloride (NaCl) is utilized as the electrolyte, chlorine evolves at the anode surface, and hydrolysis of the produced chlorine generates the residual chlorine species described in Equations (1)-(3) [18]. Residual chlorine primarily consists of hypochlorous acid (HOCl) and hypochlorite ions (OCl). The principal oxidants that can be employed for oxidation and disinfection operations include electro-generated residual chlorine species like Cl_2, OCl, and HOCl [19].

The decision to use chlorine-based electro-oxidation rather than hydroxyl radicals is tied to the electrolyte's composition in this study. NaCl was chosen as the electrolyte because it's readily available and cost-effective, making it an ideal option for developing countries like Ethiopia. At the anode, chloride ions undergo oxidation, resulting in the formation of reactive chlorine species

(RCS), such as Cl•, Cl_2•−, HOCl, and OCl^-. In conditions rich in chloride, these species can be generated in substantial amounts and significantly contribute to the degradation of pollutants.

Since this study utilizes active anodes, hydroxyl radicals (•OH) are not the predominant species. However, in chloride-containing electrolytes, •OH can quickly react with Cl^- to produce chlorine radicals. Thus, the preference for chlorine radicals in this study arises from the anticipated reaction pathways and the specific electrolyte composition employed.

This oxidant is based on the high occurrence of chloride ions in several effluents and natural waters, allowing residual chlorine to be involved during electrochemical treatment [20]. These species are formed at the anode surface and act as indirect oxidants [20]. Thus, the anode material is the most important component that influences residual chlorine production. Although numerous anode electrodes have been successfully employed to produce residual chlorine, dimensionally stable anode (DSA), such as ruthenium oxide (RuO_2) and iridium oxide (IrO_2), are the best anodes for producing residual chlorine [21]. Studies have shown that IrO_2/RuO_2 anodes have higher chlorine production efficiency than IrO_2 anodes [22]. In another study, RuO_2 and IrO_2 -based anodes were known for their electrocatalytic activity for oxygen and chlorine evolution [23]. RuO_2 and/or IrO_2 coated electrodes are best suited for disinfection based on hypochlorite generation. This is because of the high production efficiency of hypochlorite from water with a very low chloride content [24].

1.2. Significance of the Study

This study holds significant importance for public health, the environment, and wastewater treatment as it addresses the growing threat of pharmaceutical pollutants in a comprehensive manner, offering sustainable treatment solutions. One of the key contributions of this study is its ability to mitigate pharmaceutical pollution by providing a solution to eliminate persistent and toxic pharmaceutical residues, such as antibiotics and painkillers, from water bodies. This, in turn, reduces ecological harm to aquatic life. Furthermore, by effectively degrading antibiotics like amoxicillin, the study plays a crucial role in preventing antibiotic resistance, which is a major global health crisis caused by antimicrobial-resistant bacteria. In addition, unlike conventional

electro-oxidation methods that form harmful chlorinated intermediates, this study explores a hybrid EO-UV treatment approach to ensure complete mineralization with minimal toxicity, thus reducing toxic byproducts.

Moreover, this study contributes to the advancement of wastewater treatment technology by improving existing methods and developing an enhanced oxidation process that can be integrated into conventional wastewater treatment plants (WWTPs). The hybrid system innovation, which combines electro-oxidation (EO) with UV irradiation, enhances degradation efficiency while reducing hazardous byproducts, providing a scalable and sustainable alternative. The study also focuses on optimization for real-world use by utilizing Response Surface Methodology (RSM) to determine optimal conditions such as pH, current density, and time for maximum pollutant removal with minimal energy consumption. In terms of public health benefits, the study's findings contribute to safer drinking water by preventing pharmaceutical contaminants from entering water supplies, thus reducing long-term exposure risks to humans, such as endocrine disruption and antibiotic resistance. Additionally, the validation of treatment safety through E. coli bioassays ensures that discharged water is non-toxic to microorganisms and ecosystems, thereby reducing overall toxicity. From a policy and regulatory perspective, this study supports the implementation of stricter water quality standards by providing scientific evidence for regulators to enforce limits on pharmaceutical pollutants in wastewater. Furthermore, it aligns with global sustainability goals by contributing to UN SDG 6 (Clean Water and Sanitation) and SDG 3 (Good Health and Well-being) through the promotion of pollution-free water systems. Economically and industrially, the study offers a cost-effective solution by optimizing treatment parameters to balance efficiency and operational costs, making it feasible for municipal and industrial adoption. Additionally, it helps reduce long-term cleanup costs by preventing pharmaceutical accumulation in the environment, thereby avoiding expensive remediation efforts later on. In conclusion, this study not only represents a technical improvement in wastewater treatment, but also stands as a critical step towards safeguarding ecosystems, promoting human health, and ensuring future water security. By showcasing the efficiency and safety of EO-UV systems in degrading AMOX and IBU, it paves the way for real-world

implementation, with the potential to influence environmental policies, industrial practices, and public health strategies.

1.3. Statement of the Problem

The rise in pharmaceutical contaminants found in water resources has become a pressing concern, representing a serious risk to both ecosystems and human health. These harmful substances, which include antibiotics such as Amoxicillin and pain relievers like Ibuprofen, make their way into water bodies through various channels, including wastewater discharges, agricultural runoff, and improper disposal of waste. Their prevalence in aquatic environments highlights a significant shortcoming in conventional wastewater treatment facilities, which are often unable to effectively eliminate these emerging, persistent, and bioactive compounds. Consequently, this leads to their accumulation in rivers and a detrimental impact on water quality. Even though these contaminants are present in low concentrations, typically between nanograms per liter (ngL^{-1}) and micrograms per liter (μgL^{-1}), they have a significant impact on aquatic ecosystems. They contribute to the development of antibiotic resistance in bacterial populations and disrupt the endocrine systems of fish. This situation poses a threat to the food chain, which can ultimately have negative consequences for human health, including increased risks of cancer and the spread of antibiotic-resistant bacteria. Additionally, these contaminants are resistant to biodegradation, making them even more problematic. Current wastewater treatment methods, such as activated sludge processes and chlorination, are not effective in completely removing these pollutants. Instead, they often lead to the formation of more harmful intermediate compounds. Advanced oxidation processes (AOPs), which encompass techniques such as electro-oxidation and ultraviolet (UV) light-based treatments, show significant promise in the field of pharmaceutical wastewater treatment. Nevertheless, these methods encounter several obstacles, including substantial energy costs, complex operational requirements, and the potential production of hazardous byproducts, such as toxic intermediates like chlorinated organic compounds. Additionally, there are challenges related to the scalability of these processes for widespread practical use.

While both EO and UV processes have been extensively researched, they each exhibit limitations when applied independently, such as excessive energy usage, incomplete mineralization, and the generation of persistent transformation products. The innovation of this study is found in the integration and optimization of the EO/UV process to enhance the efficiency of oxidant utilization. Active chlorine species generated electrochemically ($HOCl/OCl^-$) are subsequently activated by UV light to yield highly reactive radicals, which significantly improve degradation and mineralization rates. This synergistic interaction can facilitate the desired removal of pollutants in a reduced treatment duration or at a lower current density, thus minimizing specific energy consumption. Additionally, the presence of these extra reactive species aids in the oxidation of intermediate products into smaller, less persistent compounds, which may lower their toxicity. Consequently, this study fills a research gap by advancing pharmaceutical degradation and mineralization while simultaneously enhancing energy efficiency through the coupling of EO and UV processes.

Consequently, there is a pressing need to develop a treatment approach that is not only efficient and sustainable but also economically feasible. This coupling should aim to achieve complete mineralization of pharmaceutical contaminants, minimize the generation of toxic intermediates, and be compatible with existing wastewater treatment plant (WWTP) technologies. This research investigates the application of hybrid (EO-UV) for the removal of AMOX and IBU, with a focus on reducing environmental risks

1.4. Research Questions

The following the research questions addressed in this study.

- ✓ What is the removal efficiency of conventional WWTPs for selected persistent pharmaceutical compounds present in wastewater?
- ✓ How does the performance of the EO–UV hybrid system compare with standalone EO and UV processes in terms of pharmaceutical degradation and mineralization?
- ✓ What effects does coupling EO with UV irradiation have on the generation and mitigation of toxic intermediates, including chlorinated disinfection by-products?

- ✓ Which operational conditions provide the best balance between pollutant removal efficiency, energy consumption, and treatment cost in the EO-UV hybrid process?
- ✓ How does the EO–UV treatment process alter the toxicity of wastewater, particularly with respect to *Escherichia coli* inhibition?

1.5. Objectives

1.5.1. General Objective:

The general objective of this study is to investigate the degradation of synthetic wastewater containing the pharmaceutical pollutants amoxicillin (AMOX) and ibuprofen (IBU) using a combined process of electrochemical oxidation and UV photolysis. The objective is to evaluate the effectiveness and efficiency of this hybrid advanced oxidation process (AOP) in removing these persistent contaminants. Key parameters such as pH, applied current, reaction time, and UV intensity will be varied to optimize treatment conditions. In addition to measuring degradation efficiency and reaction kinetics, the study will assess the potential synergistic effects of the combined process compared to individual treatments, and monitor the formation of intermediate by-products to evaluate the extent of mineralization. This research aims to contribute to the development of more effective treatment strategies for pharmaceutical-laden wastewater.

1.5.2. Specific Objectives:

- ✓ To examine the generation of residual chlorine and consumption of energy by applying integrated response surface methodology statistic techniques and artificial neural network-based machine learning
- ✓ To analyze the efficiency of electrochemical oxidation only process for the treatment of synthetic AMOX and IBU wastewater and analyzing the effect of initial AMOX and IBU concentration, current density, and pH
- ✓ To investigate effects of UV intensities and exposure time on the degradation of AMOX and IBU by UV photolysis process
- ✓ To evaluate the efficacy of Coupling electrochemical oxidation and UV photolysis for the removal of AMOX and IBU synthetic wastewater

- ✓ To analyze toxicity of by products from the degradation of AMOX and IBU in electro-oxidation only, UV photolysis only processes and combining electro-oxidation and UV photolysis

Chapter Two

Literature Review

2.1. Introduction

Recently, the intake of pharmaceuticals that are used in public healthcare, and medical treatments has been increasing and the ingested drugs are metabolized and excreted leading to a mixed discharge of active ingredients and metabolites that makes the pharmaceuticals to be found in the environment specially in wastewater and sewage systems of hospitals, clinics, pharmaceutical industries, and residential areas, due to their uncontrolled discharge, non-metabolized in humans and animals [25][26][27]. Also, in the past two decades the presence and toxicity of pharmaceuticals in surface water bodies and wastewater treatment facilities brings about concern to humans [28]. Pharmaceuticals are trace organics meaning occur in natural water bodies and wastewaters in small concentrations ranging from nanogram to microgram per litre (ngL^{-1} to μgL^{-1}) [29]. Pharmaceutical wastewater composition is complex, that has high concentration of organic matter, microbial toxicity, high salt, and difficult to biodegrade [30].

PPCPs are considered as emerging contaminants (ECs) as they have been analyzed recently and are believed to cause serious impacts on health or the environment [31]. Pharmaceuticals existence in the environment leads to the development of antibiotic resistance genes possibly resulting in antibiotic resistant bacteria or so-called superbugs [32].

To efficiently treat these pollutants different methods have been tried. One of these methods is conventional treatments for example activated sludge have demonstrated an incomplete success in eliminating pharmaceuticals such as diclofenac, ibuprofen, and triclosan. Nevertheless, this method takes time, which leads to low degradation rates. Accordingly, for the biodegradation of other PPCPs require larger treatment plants which leads to increased costs. Generally, both aerobic and anaerobic methods have limitations over treatment the removal of PPCPs by biodegradation. In addition to biological processes conventional wastewater treatment methods

include; physical, chemical or combination of physicochemical process, involving the following: filtration, flocculation, coagulation process, sedimentation, membrane filtration, chlorination, adsorption process via activated carbon (AC), carbon nanotubes (CNTs), graphene oxide (GO), ozonation, photo-catalysis ultraviolet irradiation, ultra-sonication, and others [33]. On other hand, emerging treatments such as EAOPs have shown promising results in degrading PPCPs [34]. This chapter reviews the prevalence of pharmaceuticals in wastewater, elimination of pharmaceuticals in wastewater treatment plants by conventional and advanced oxidation methods.

2.2 Prevalence and fate of pharmaceuticals in effluent of wastewater treatment Facilities

Approximately 3,000 compounds are used as medicine globally [35]. The most widely used therapeutic pharmaceutical groups that are detected in wastewater are: anti-inflammatory and analgesic agents; antidepressants; antiepileptic medications, lipid-lowering agents, beta-blockers; antiulcer medications and antihistamines; antibiotics and various other substances [36]

So far, pharmaceuticals entered into the environment has been posing concern regarding ecological risks. For human and livestock medicine, pharmaceuticals are widely used to protect against diseases. After taking pharmaceuticals it is converted in the human body into more polar and soluble forms as metabolites or as conjugates of glucuronic and sulphuric acid. Pharmaceuticals and metabolic products are usually excreted with feces and urine and join urban wastewater treatment facilities. In addition, the administered pharmaceuticals that go to metabolic process by biochemical reactions has two routines: the first routine is oxidation, reduction, hydrolysis, and alkylation reactions happen; and the second one is the way in which glucuronide or sulfate conjugates are formed and excreted by urine or bile in the form of more polar and hydrophilic derivatives, as a metabolite or as a mixture of multiple metabolites.[37]

The major sources of contamination of pharmaceuticals to enter into aquatic environment are excretion, bathing, direct disposal, and veterinary use. Conventional treatment methods have proven ineffective over pharmaceuticals, resulting in their presence not only in treated wastewater discharge but also in surface water, groundwater, and drinking water and since most

pharmaceuticals are polar and less volatile they are found in water bodies [38] [39] [40]. Many scholars pointed that conventional treatments that are utilized in WWTPs have limited capability in degrading pharmaceuticals. [41]

For example: studies have shown that wastewater treatment plants in USA and Europe were inefficient in pharmaceuticals removal [42].The typical features that separates pharmaceuticals from other known contaminants from industries are: i) pharmaceuticals are formed by complex and large molecules which differ in their molecular weight, structure, functionality and shape ii) since these compounds are polar molecules they consist of more than one ionizable group and level of ionization is highly dependent on the pH of the matrix, also they are lipophilic and some of them moderately dissolve in water iii) some pharmaceuticals stay in the environment for more than a year and others might persist for several years and through accumulation it gets biologically active iv) The chemical structure of pharmaceutical changes after their administration by which the molecules are absorbed, distributed, and lead to metabolic reactions [36].

A study demonstrated that WWTPs are unable to completely remove many of the pharmaceuticals which led to their occurrence in WWTP effluents, rivers, and lakes and sometimes in groundwater.[43] Similarly, another study showed that anti-inflammatory medications, anticonvulsant drugs, antiviral drugs, and antilipemic agents were found at the downstream of water treatment plants and could not be completely degraded by the integration of physical-chemical treatment with the activated sludge system.[44] Also, another studies confirmed that WWTPs that are utilized in municipal wastewater treatment plants are ineffective over removal of some pharmaceuticals. For example: carbamazepine, and clorofibric acid removal was less than 16% and less than 35 %, respectively after conventional biological treatment [45].

Other study also highlighted that in wastewater treatment plant effluents there is up to 90% of drug residues [40]. Also a study in India showed that pharmaceuticals concentrations as high as 31,000 μgL^{-1} was detected in a WWTP effluent, and these discharges caused nearby groundwater and surface water concentrations as high as 2,500 μgL^{-1}. In the same manner, in a Taiwan WWTP effluent diclofenac concentration exceeded 20 μgL^{-1} [46]. In addition, a recent study, indicated

that effluent from a local wastewater treatment plant has shown extremely high levels (mgL^{-1}) of some drugs [47].

When these biologically active pharmaceuticals entered into the environment they have a number of health impacts to human and animals. For example, at low concentration the antibiotic such as sulfamethoxazole causes mutations in genes. While there are analgesics such as paracetamol which causes kidney cancer, damage to liver and asthma. Generally, when exist in the environment there are adverse effects of pharmaceuticals such as disruption of the endocrine system, and induction of antimicrobial resistance in environmental systems [40]

2.3. Elimination of Pharmaceuticals in wastewater treatment plants

2.3.1. Conventional Treatment of pharmaceuticals

Numerous pharmaceutical substances have been identified as possible environmental pollutants, as indicated by the European Union Water Framework Directive. However, there remains a significant lack of understanding regarding their effects on the environment and human health. Various studies showed that conventional treatment systems have limited ability to remove pharmaceuticals from wastewater treatment plants. One of the most common conventional chemical treatment methods is chlorination. A study indicated the formation of unknown chlorination byproducts after observing unidentified peaks in a chromatogram for the treatment of nine pharmaceuticals (clofibric acid, diclofenac, fenoprofen, gemfibrozil, ibuprofen, indomethacin, ketoprofen, naproxen and propyphenazone) by chlorination process [48]. The amine-containing pharmaceuticals such as acetaminophen go through a rapid reaction with hypochlorous acid to form chlorinated amines which are more toxic than the parent compound [49]. Similarly, a study showed that the reaction of pain killer acetaminophen with hypochlorite produced two toxic products namely 1,4-benzoquinone and N-acetyl-p-benzoquinone imine (NAPQI), and chloro-4-acetamidophenol and dichloro-4-acetamidophenol [50].

Even though Conventional WWTPs has primary system consisting physicochemical treatment such as sedimentation and coagulation and secondary stage comprising of biological treatments

such as activated sludge, Conventional WWTPs are not designed to removal pharmaceuticals thus it removes pharmaceuticals inefficiently [6]. Research has demonstrated that anti-inflammatory medications, anticonvulsant drugs, antiviral drugs, and antilipemic agents have been detected in the water downstream of water treatment plants. These substances pose a challenge to eliminate completely through the activated sludge system [44].

A study has demonstrated that the activated sludge process was not able to fully eliminate Oseltamivir Carboxylate (OC) from wastewater, indicating that OC is highly stable during conventional treatment processes. In the same study, it was found that the efficiency of removing Cyclophosphamide and Carbamazepine using conventional treatment was very low, at 24% and -4% respectively. Other studies have shown that the efficiency of eliminating Indometacin, Ketoprofen, and Diclofenac from wastewater using conventional treatment was 50%, 40%, and 30% respectively [51]. A study also showed that using CAS treatment it is unable to remove carbamazepine from pharmaceutical wastewater [39]. Furthermore, other study found out that biological treatment processes have negative to low elimination efficiency for micro pollutants such as pharmaceuticals. Also, the study added besides to its low removal performance biological processes such as CAS has limitation of producing sludge, difficult to control the process, and consumes high energy when the removing micro pollutants such as pharmaceuticals [52]. Another study showed that applying conventional wastewater treatment process for the elimination of 20 antibiotics from different class of antibiotics was found non-biodegradable. This is a clue that this process is ineffective against removal of pharmaceutical wastewater [53]. Generally, applying conventional methods to eliminate pharmaceuticals from WWTPs is ineffective [54].

2.4. Removal of pharmaceuticals by Adsorption/Sorption Process

Adsorption is a physicochemical process in which atoms, ions, or molecules from a liquid phase accumulate on the surface of a solid phase, and it is widely applied in water and wastewater treatment for the removal of contaminants, including organic and inorganic pollutants. The process occurs through several sequential steps, beginning with the transport of solutes from the bulk liquid to the boundary layer surrounding the adsorbent, followed by diffusion through the

liquid film (film diffusion). Subsequently, the solute molecules migrate within the internal pore structure of the adsorbent (pore diffusion) until they reach available active sites, where adsorption takes place. This attachment onto the solid surface can occur through weak van der Waals forces, known as physisorption, or through stronger chemical bonds, referred to as chemisorption, as well as hydrogen bonding, depending on the properties of both the adsorbate and the adsorbent. The solid material that provides the adsorption surface is called the adsorbent, while the substance being removed from the liquid phase is referred to as the adsorbate. The efficiency of the adsorption process is strongly influenced by the characteristics of the adsorbent, including its porosity, pore size distribution, pore volume, and the presence of functional groups that enhance interactions with the adsorbate molecules. Furthermore, several operational and environmental factors affect adsorption performance, such as the nature and concentration of the adsorbate, solution pH, temperature, and contact time. Additional factors, including the particle size of the adsorbent and the presence of competing pollutants or co-existing ions in the solution, can either enhance or inhibit the adsorption process, thereby influencing overall efficiency [55]

2.4.1. Activated carbon adsorption

For the past years, adsorption using Activated Carbon has been regarded as a preferred traditional method for treatment of contaminants. This due to its high surface area, multilevel pore structure, very high adsorption capacity and invariable chemical property. Particular to the treatment of pharmaceutical effluents AC is used yet difficult to reach effluent discharge standards. AC adsorption is classified as physical and chemical adsorption. Physical adsorption is a physical process, reversible and has no selectivity to adsorbate. In physical adsorption it is not difficult to desorb. Contrary to this, chemical desorption adsorbs adsorbates chemically which is difficult to desorb and the binding is irreversible. AC can be used for advanced treatment since it can be recycled and higher treatment impact [56].

There are various studies on the treatment of pharmaceutical wastewater using adsorption process. A study showed that the common antibiotic tetracycline adsorption on swelling clays is higher than adsorption on kaolinite [57]. Other study on the removal tetracycline indicated that

palygorskite, a fibrous clay mineral, has a better performance on the removal of Tetracycline from wastewater for it has high sorptivity to organic compounds area and high surface [58]. Another study highlighted that using the natural clay, Na-montmorillonite an efficient elimination of paracetamol from effluent was achieved. Also for the removal of AAIDS from STP effluents montmorillonite has removal efficiency range between 82 % and 94% with 5 gL^{-1} adsorbent at pH 6.5 after 120 min of contact time [59]. A recent study showed that a natural clay mixture of smectite-kaolinite and quartz was used to eliminate ibuprofen, naproxen and carbamazepine. The results indicated that ibuprofen and naproxen were significantly removed since these drugs were found in a protonated form while carbamazepine showed lowest removal of 32 mg g^{-1} this is due the presence of carbemezepine in neutral form. Also, a study using Na-rich Mt for the treatment of sulfonamide and tetracycline antibiotics from synthetic effluent was carried out. The results showed negligible removal for sulfonamide and high removal for tetracycline [60]. Using GAC as adsorbent for treating pharmaceutical wastewater a maximum adsorption of 522.3 $mgCg^{-1}$ AC was achieved. In this study up to 52% removal efficiency was achieved for a concentrated pharmaceutical wastewater [61]. The process of adsorption in wastewater treatment has certain limitations that need to be considered. One limitation is the generation of secondary wastewater during the regeneration process, which has environmental implications. Another factor to consider is the relatively lower performance of the regeneration process and its complex nature. Despite notable achievements in adsorption to treat pharmaceuticals efficiently, there are challenges that remain regarding secondary waste management, adsorbent regeneration, operational cost, and long-term sustainability [62].

Table 1. 1 Removal of pharmaceuticals by using Adsorption

Adsorbent	Pharmaceutical	Experimental conditions	Removal efficiency or adsorption capacity	Ref
Aluminum-pillared clay	Amoxicillin and Imipramine	▪ $[C]_O$=0-100mg/L ▪ T= 298–318K ▪ Adsorbent dose = 50mg; ▪ V= 50 ml; pH = 2-12; ▪ Contact time = 0-180 min	• IMP by Al-PILC is higher (~59.8mgg^{-1}) as compared to AMOX (~7.7 mg g^{-1}) • Pillaring of MMT has enhanced its adsorption capacity for AMOX and IMP by 681% and 332% respectively; • Maximum adsorption capacity of MMT for AMOX and IMP were 0.98 and 12.06 mg g^{-1}, respectively	[63]
Activated carbon and sonicated activated carbon	Ibuprofen	▪ $[C]_O$ =100mg/L; ▪ pH= 2; ▪ T = 298 K; ▪ Adsorbent dosage of 0.5 g/L	• 84.5% and 92.5% of maximum adsorption was achieved for the AC • Decrease in solute adsorption was observed from 84.10 to 7.30 mg/g and from 106 to 15.73 mg/g for the AC and SAC respectively	[64]
AC	Amoxicillin	▪ $[C]_O$=300mg/L; ▪ V = 250 mL; ▪ pH = 2-7; ▪ T= 300C; ▪ Adsorbent dose = 0.1–1.5 g	• Adsorption Capacity= 221.8mg/g. • qe values also indicate that the amount adsorbed at equilibrium increases as the pH value decrease	[65]

AC (GAC and PAC)	Carbamazepine, Oxazepam, Sulfamethoxazole, Piroxicam, Cetirizine, Venlafaxine and Paroxetine	▪ $[C]_O$ = 5mg/L; ▪ Contact Time = 5 – 120min; ▪ Adsorbent conc = 0.02 - 4.0 g/l	▪ AC (PBFG4) has the best results with the highest adsorption coefficient (K_f) obtained for PIR (128 ± 2 mg g1 (mg L^{-1})N) and the lowest obtained for VEN (39.7 ± 0.6 mg g^{-1} (mg L^{-1}) N)	[66]
Pigeon Pea Hulls Ash (PPHA)	Sulfamethoxazole (SMX), Carbamazepine (CBZ), Ketoprofen (KP), Naproxen (NPX), Diclofenac (DCF), Ibuprofen (IBF)	▪ $[C]_O$ = 1–50 mg/ L; ▪ Contact Time = 1–120 min; ▪ Adsorbent dose = 0.1 g; ▪ pH = 6; ▪ V= 100ml	▪ Adsorption capacities were 17.193, 17.685, 19.265, 17.657, 21.116 and 23.332 mg g-1 for SMX, CBZ, KP, NPX, DCF and IBF, respectively	[67]
Multiwalled Carbon Nanotubes (MWCNTs)	Oxytetracycline (OTC) and Carbamazepine (CBZ)	▪ $[C]_O$ for CBZ=1-25 mg/L ▪ [C]O for OTC = 1.25-25 mg/L ▪ pH=7.0+0.2 ▪ Adsorbent dose = 1-2 mg	▪ Depending on the MWCNT outer diameter, 13.8-25.2% and 62.7-90.6% of initially adsorbed OTC and CBZ, respectively, were desorbed after 200 h	[68]

2.5. Advanced oxidation processes for the treatment of pharmaceutical pollutants

The review mentioned above demonstrates that conventional treatment methods, biological treatment technologies, and adsorption processes are effective in treating pharmaceutically contaminated wastewater. However, these approaches may not achieve the required efficiency to reduce pollutants to levels that comply with environmental regulations for reuse of effluents. Research has shown that Advanced Oxidation Processes (AOPs) are effective in tackling both organic and inorganic pollutants. These processes typically involve the in situ generation of oxidants, such as hydroxyl radicals (HO.), superoxide anions (O_2^-.), and hydroperoxyl radicals (HO_2.), with hydroxyl radicals being particularly significant due to their high reactivity. This radical can effectively interact with most organic molecules, exhibiting rate constants between 10^6 and 10^9 M^{-1} s^{-1}. There are various systems available that produce these free radicals for the treatment of organic pollutants, highlighting the multipurpose of AOPs [63].

2.5.1. AOPs based on electrochemical oxidation

For wastewater treatment the most noticeable technique is the electrochemical oxidation which is commonly called anodic oxidation (AO) [64]. There are two modes of electrochemical oxidation that take place within an electrolytic cell: i) direct oxidation at the anode surface via electron transfer, and ii) indirect oxidation that involves free radicals produced in the bulk solution from the discharge of water, aided by physisorbed hydroxyl (OH) or chemisorbed active oxygen. Voltages are applied to achieve the oxidation of pollutants while simultaneously maintaining the activity of the anode, which significantly affects the efficiency of the process and the yield, depending on the anode material. Furthermore, the study revealed that there are both active and non-active anodes, which exhibit differences in their behavior [65]. Active anodes comprise materials like platinum (Pt), iridium oxide (IrO_2), and ruthenium oxide (RuO_2), whereas inactive anodes are made up of lead dioxide (PbO_2), tin oxide (SnO_2), and boron-doped diamond (BDD). When water is oxidized at both categories of anodes, the physisorbed hydroxyl radical leads to the creation of M (OH). At the surface of active anodes, oxidants and radical species interact,

resulting in the conversion into chemisorbed "active oxygen" or superoxide, referred to as MO. The MO/M functions as a mediator, promoting the electrochemical transformation of organic compounds. Conversely, non-active anodes show limited reactivity with M(OH), which instead interacts directly with organic materials until complete mineralization is achieved. Additionally, various reactive oxygen species are generated during electrochemical oxidation, including OH, H_2O_2 from its dimerization and the discharge of water, and O_3 produced at the anode's surface. Among these, the physisorbed OH exhibits the highest oxidizing capability [36].

The most effective and well-known "non-active" anode is boron-doped diamond (BDD), which facilitates the removal of organic pollutants through anodic oxidation. In the process of electrochemical oxidation of electrolyte ions with reactive oxygen species (ROS) such as bisulfate, bicarbonate, and phosphate utilizing BDD, oxidants are generated alongside competitive and less potent oxidants like peroxodisulfate, peroxodicarbonate, and peroxodiphosphate, respectively [66]. In wastewater treatment, when chloride ions act as electrolytes, the active chlorine species produced such as Cl_2, $HClO/ClO^-$, and ClO_2^- often participate in competitive reactions with the organic compounds found in the wastewater, as well as with free radicals. Electro-oxidation is a process that involves the oxidation of chloride ions (Cl^-) at the anode. This oxidation leads to the production of soluble chlorine. The produced chlorine then diffuses away from the anode. Once diffused, it quickly undergoes hydrolysis and disproportionation, resulting in the formation of hydrochlorous acid and chloride ions [67].

Table 1. 2 Removal of different pharmaceuticals using electrochemical oxidation

Pharmaceutical	Anode type	Electrolyte system	%TOC Removal	Ref
Tetracycline (TC)	Ti/ SnO_2 –Sb	▪ Approximately 10 mmol L^{-1} Na_2SO_4 as a supporting electrolyte solution. ▪ The reaction was carried out with the distance between anode and cathode of 10 mm, Current density of 15 mA cm^{-2} and Initial TC concentration of 5 mg L^{-1}	52.51–81.69%	[74]
Sulfamethazine	BDD	▪ Reactor: Continuous stirred tank ▪ Current density of 21.43 mA Persulfate loading of 0.41 gL^{-1} ▪ pH=4	100%	[4]
Enrofloxacin	Conductive-diamond (p-Si-boron doped diamond)	▪ 0.05 M Na_2SO_4 as background electrolyte ▪ pH 3.0 + 0.1 and ▪ Temperature of 35^0C. ▪ Single-compartment electrochemical flow-cell working under a batch operation mode	< 40% for COD	[75]
Ibuprofen	Ti-IrO_2 mesh plate	▪ Continuous-flow reactor. ▪ Sodium sulfate or Sodium Chloride as electrolyte. ▪ Current density of 1 and 20 mA/cm^2	98% in 4 h	[76]

Clofibric acid	BDD and Pt	▪ 0.05 M Na_2SO_4. ▪ Constant apparent current density (j_{app}) of 33, 100 and 150 mA cm^{-2} at 35^0C	> 97% for TOC; 30% of mineralization by Pt at 4 h	[70]
Ketoprofen	BDD and Pt	▪ Na_2SO_4 (0.05 M) ▪ Volume of Cylinder is 250-mL ▪ The currents of 100–2,000 mA ▪ Magnetic stirrer with 800 rpm.	Using BDD reached 89 and 95 % TOC removal at 300 and 2,000 mA	[77]
Paracetamol	Stainless steel	▪ Fiberglass electrochemical oxidation cell with volume of 180 mL. ▪ Degradation of Paracetamol was conducted using a stainless steel electrode cell, a pH of 3, and ▪ Direct current densities of 5.7mA/cm^2 (6 V) and 7.6 mA/cm^2 (12V).	95% of Paracetamol was oxidized in only 15 min at the 7.6 mA/cm^2 current density	[78]
Amoxicillin	Pt-RuO_2-IrO_2 (PRI) electrode	▪ Current density (20 - 100 mA/cm^2) ▪ NaCl as electrolyte ▪ Flow rate was 2.08mLs^{-1}	< 10% of the COD removal in the absence of chloride; 100% removal of TOC (mineralized) in the presence of chloride	[79]

Acetaminophen	Stainless Steel	▪ Volume to 200 mL ▪ Electrochemical oxidation of samples of each solution was conducted at DC densities of 12.3 mA/cm^2 (8.5 V), 16.3 mA/cm^2 (10V) and 20.3 mA/cm^2 (12V), and different reaction times.	At moderate current density and pH values (16.3 mA/cm^2 and pH 5), good performance of the process was achieved and not only acetaminophen but also its transformation products were completely degraded in only 7.5 min.	[80]
Diclofenac	BDD and Ta/PbO_2	▪ A two-compartment cell (V=200 cm^3) with a lead nitrate solution (1 M) as the anolyte and a sulphuric acid solution (1M) as the catholyte. ▪ Electro-oxidation of diclofenac was performed at different current density values (10, 15 and 20 mA/cm^2)	TheCOD removal efficiency obtained on the Ta/PbO_2 anode (97%) was higher than that obtained on the BDD anode (71%) after 2.5 hours.	[81]

2.5.2. UV based AOPs for the treatment of pharmaceutical pollutants

Recently, there has been a notable increase in research interest in advanced oxidation processes (AOPs) due to their remarkable effectiveness in treating pharmaceutical contaminants. Within the AOP methodology, the hydroxyl free radicals generated are pivotal in the chemical oxidation of these pharmaceutical pollutants. Research indicates that combining UV photolysis with other advanced oxidation processes like hydrogen peroxide and ozone shows substantial effectiveness in treating pharmaceutical wastewater [68].

Studies have demonstrated that the use of UV processes alone for the removal of pharmaceuticals is not as effective as when they are combined with other advanced oxidation processes (AOPs). A study demonstrated that using UV irradiation alone resulted in a removal efficiency of only 58% for azithromycin. However, when sodium persulfate was added to the UV irradiation, the removal efficiency significantly increased to 98%. The study also found that specific conditions were crucial for optimal performance, including an initial azithromycin concentration of 5 mgL^{-1}, a persulfate concentration of 1 mmol, a contact time of 30 minutes, and a pH of 7 [69].
Also recent study found that many pharmaceuticals and personal care products (PPCPs), such as cyclophosphamide and 2-QCA, were not effectively eliminated when subjected to UV treatment alone, despite a substantial increase in UV dose from 38 mJ cm^{-2} to 5644 mJ cm^{-2}. However, when UV treatment was combined with hydrogen peroxide, there was a notable improvement in the complete removal of most PPCPs, with approximately 90% removal of seven specific PPCPs observed after 30 minutes of UV irradiation and a UV dose of 691 $mJcm^{-2}$. The study emphasized the significant enhancement in degradation efficiency when H_2O_2 was added during UV treatment, indicating that increasing the UV dose is crucial for the overall degradation efficiency [70].

A comprehensive study was undertaken to thoroughly investigate the removal of PRO (Propranolol) through direct photolysis using different types of reactors. The results of the study revealed a 71% removal of PRO after 240 minutes, but the achieved mineralization only reached a maximum of 7%. This indicates that while PRO was degraded, it was not fully converted into

CO_2 , H_2 O, and inorganic ions, suggesting the formation of resistant intermediate byproducts under direct photolysis. However, the addition of TiO_2 catalyst in the photocatalysis process enhanced the degradation of PRO. For example, at a concentration of $0.4gL^{-1}$, approximately 94% removal of PRO was achieved after 240 minutes, with the mineralization reaching 41%. The study also found that increasing the concentration of catalysts, such as TiO_2, resulted in higher reaction rates. This is likely attributed to the increased availability of active sites on the catalyst surface, facilitating the generation of reactive oxygen species (e.g., hydroxyl radicals) under light irradiation, which are responsible for the degradation of organic pollutants like PRO [71].
According to a study, the process of photolysis caused diclofenac to transform into byproducts, resulting in a maximum mineralization of 10%. This indicates that a significant portion of the diclofenac was converted into byproducts [72].

A study demonstrated that the removal of six pharmaceuticals (meprobamate, carbamazepine, dilantin, atenolol, primidone, and trimethoprim) by UV/H_2O_2 at UV-254 ranges between 0 and 90% [73]. A photolysis study using a low-pressure mercury UV lamp with a wavelength of 254nm and a combination of UV and H_2O_2 was conducted for the removal of diclofenac (DCF), sulfamethoxazole (SMX), carbamazepine (CBZ), and trimethoprim (TMP). The results indicated that after 8 minutes of UV irradiation, the degradation of DCF and SMX reached 1 μgL^{-1}, showing that DCF and SMX are affected by UV irradiation. However, TMP and CBZ showed no significant degradation with 58.2% and 25.2% respectively after 1 hour of UV exposure. Integrating UV with hydrogen peroxide (H_2O_2) at a concentration of 0.2 gL^{-1} significantly improved the degradation rate of TMP and CBZ, reaching 91.2% and 99.7% respectively. Toxicity testing of each pharmaceutical solution before and after treatment revealed that all parent compounds were not toxic, but after UV irradiation, the solution showed toxicity, indicating the presence of degradation products. Combining UV with hydrogen peroxide (UV/H_2O_2) decreased the toxicity level of all solutions [74]. According to a study, the use of photolysis and photocatalysis processes resulted in the complete removal of sulfamethoxazole (SMX) at a concentration of $12mgL^{-1}$ within 10 minutes of exposure to UVC irradiation (254nm) in the photolytic process. In comparison to TiO_2 photocatalysis, which required 30 minutes of exposure,

photolysis showed faster removal of SMX under UVC irradiation. Additionally, it was observed that photocatalysis resulted in a higher removal of chemical oxygen demand (COD) of SMX compared to photolysis. After a 6-hour exposure, photolysis and photocatalysis indicated a 24% and 87% removal of COD, respectively [75]. Also, another study demonstrated that a significant removal efficiencies of more than 90% was achieved for clofibric acid and more than 98% for carbamazepine and diclofenac [76]. In addition, a study was conducted to assess the treatment of forty-one pharmaceuticals, including 12 antibiotics and 10 analgesics, found in secondary effluent using UV-based processes (UV and UV/H_2O_2).The study revealed that the UV-only process effectively removed pharmaceuticals such as ketoprofen, diclofenac, and antipyrine. However, the elimination efficiency of UV alone was found to be very low for macrolide antibiotics like clarithromycin, erythromycin, and azithromycin, even with a raised UV dose of 2,768 $mJcm^{-2}$. Interestingly, when UV was combined with H_2O_2, the removal efficiency improved significantly, reaching 90% for 39 out of the 41 pharmaceuticals, with a reduced UV dose of 923 $mJcm^{-2}$.This indicates that integrating H_2O_2 with UV reduces the energy required for UV photolysis, as demonstrated in studies [77][78].

Furthermore, in a separate study, it was found that anti-inflammatories like metamizole, diclofenac, and ketoprofen were effectively eliminated through the process of photolysis. Moreover, when exposed to a UV dose of less than 100 $mJcm^{-2}$, ciprofloxacin, diclofenac, ketoprofen, and metamizole underwent degradation as a result of photolysis [79].

Table 1.3 *Elimination of pharmaceuticals by UV based advanced oxidation processes*

Drug	Class	Exp'tal Conditions	Removal (%)	Ref
Diclofenac	Non-Steroidal anti-inflammatory (NSAIDs)	▪ Mercury low-pressure 10 W was used as a UV source. ▪ The incident UV radiation photon flux at 254 nm was 3.59 ± 0.12 μ Einstein s^{-1}. ▪ The constant temperature (22±1°C). ▪ Volume of Cylindrical Glass Reactor is 1 L. ▪ Initial DCF concentration of 100 mg L^{-1} were prepared.	▪ 90% conversion time decreased from 48 to 23.5 min for UV photolysis	[94]
Venlafaxine	Antidepressant	▪ pH= 7± 0.2. ▪ $[C]_0$ = 5 mgL^{-1} Experiments were initialized by spiking of PAA (peracetic acid) or H_2O_2 under constant stirring using magnetic stirrer bars. ▪ Distance between UV lamps and liquid surface was 11cm irradiation range of 647 Wm^{-3} and 3,502 Wm^{-3}.	UV/H_2O_2 more efficient than UV/PAA for degradation of FLU, VEN and CBZ as an increase in the degradation rate of 33%, 55% and 74%, respectively for UV/H_2O_2	[95]

Sulfametho xazole Fluoxetine Carbamazepine Chloramphenicol	Sulfonamide antibiotic Antidepressant Anticonvulsant and Analgesic drug Antibiotic	▪ 150 mL of the chloramphenicol solution with a concentration of 20 mg L^{-1} was used to perform the experiments ▪ UVC (germicidal lamps) and solar radiation, with or without the presence of H_2O_2 ▪ The reactor with germicide lamps emitted 2,500 lux and radiation of 53μWcm^2 for the wavelength between 290 and 390 nm and 18.6 μW cm^2 for 254 nm	▪ 83% of the pharmaceutical chloramphenicol after 12 h of exhibition to UVC radiation was obtained. ▪ Chloramphenicol degradation of 100% after 2h of exposure to UV/H_2O_2	[96]
Ranitidine and Nizatidine	Histamine-2 blockers	▪ The average UV fluence rate was determined to be 2.1mWcm^{-2} by atrazine actinometer and the UV photolysis experiments were conducted in triplicate. ▪ The temperature of the UV system was controlled at 25^{0}C by circulating the water in the bath. RNTD and NZTD concentrations ranged from 0 to 10 mM	At a typical pH of 7.0, the removal efficiencies of RNTD and NZTD in these three waters during UV disinfection were estimated to be <70%	[97]
Propranolol	Beta blockers	▪ Petri dishes were filled with 150 mL of 50 or 100 mg L^{-1} MET or PRO solution and then irradiated for a period that varied between 30 min and 24 h. ▪ Initial concentration 50 mg L^{-1}. ▪ The UV lamps used in this work were a UV-254 mercury lamp (UV-C) and a UV-365 black light mercury lamp.	In 24 h of UV-C irradiation it is able to reach a total MET elimination and above 60% of PRO removal. UV-A irradiation could achieve MET removals near 22 and 37% after 8 and 24 h, respectively.	[85]

Metronidazole	Antibiotic	▪ pH= 8. ▪ Light intensity set at 765 W/m^2.	38 % for trimethoprim 36% for sulfamethoxazole	
Sulfamethoxazole	Antibiotics			[98]
Ketoprofen	NSAID	▪ UV absorbance of the samples was measured (between 200 and 400 nm). ▪ Secondary dilutions on which UV treatment was practiced were prepared with initial concentrations ranging from 1 to 3 mM. ▪ 25mg/L as $CaCO_3$ total alkalinity, pH 7, 4.2mg C/L dissolved organic carbon (DOC), and 74% UV transmission).	At 100mJcm^{-2}, naproxen's negligible removal by LP-UV photolysis increased to approximately 36% with MP lamps, while the removal of clofibric acid increased from approximately 19% using the LP system to 50% using MP lamps.	[99]

Naproxen	NSAID			
Carbamazepine	Anticonvulsant and Analgesic drug	▪ Lamotrigine was added to ultrapure unbuffered water at 5 M concentration. ▪ The final pH of the sample was approximately 6. ▪ Lamp irradiance was set at 1.5 mW cm^{-2}. Sample depth was 1.7 cm. ▪ Petri factors were 0.97 and 0.99 for low pressure and medium pressure lamps respectively.	After 500 mJ cm^{-2} dose approximately 20 -40% of lamotrigine was transformed. In all experiments at most 45% of the parent molecule was transformed	[100]
Ciprofloxacin	Antibiotic			
Clofibric acid	Anticholesteremic and antilipemic drug			
Iohexol	Contrast agent			
Lamotrigine	Anticonvulsant			

Although adsorption has demonstrated considerable effectiveness in removing various pharmaceutical compounds from wastewater, several limitations remain. The removal efficiency is highly dependent on the physicochemical properties of both the adsorbent and the pharmaceutical compounds, resulting in variable performance among different contaminants. For example, compounds such as tetracycline exhibit high adsorption capacities, whereas carbamazepine and sulfonamides often show lower removal efficiencies. Another significant limitation is that adsorption transfers pollutants from the aqueous phase to the adsorbent rather than destroying them, creating challenges related to adsorbent regeneration and secondary waste management. In addition, the long-term economic feasibility, regeneration efficiency, and environmental sustainability of adsorption processes remain insufficiently investigated. These limitations highlight the need for alternative or complementary treatment technologies capable of achieving complete degradation and mineralization of pharmaceutical pollutants.

UV-based advanced oxidation processes have shown promising results for pharmaceutical removal, several limitations remain. Most studies focus on the degradation of parent compounds rather than complete mineralization, with many reporting high removal efficiencies but low TOC and COD reduction, indicating the formation of persistent intermediates. The effectiveness of UV treatment also varies significantly among pharmaceutical classes, making it difficult to generalize treatment performance. Furthermore, experimental conditions differ widely across studies, limiting direct comparison and the identification of optimal operating parameters. Most investigations have been conducted using synthetic wastewater, which may not accurately represent the complexity of real wastewater matrices. Another major gap is the limited understanding of the formation and fate of toxic transformation products and chlorinated disinfection by-products. While some studies report increased toxicity after photolysis, comprehensive toxicity assessments remain scarce. In addition, research on EO-UV hybrid systems is relatively limited compared to UV/H_2 O_2 and photocatalytic processes, particularly regarding synergistic effects, energy efficiency, by-product control, and treatment cost.

Chapter Three
Materials and Methods

3.1. Study Design and Experimental Framework

3.1.1. Preparation of electrolyte solutions

Sodium chloride salt was used as the electrolyte, and distilled water was used to produce concentrations of 5, 15, and 25 g/L. These concentrations were chosen based on the NaCl concentration in the seawater. The pH of the electrolyte solution was maintained neutral (pH = 7).

3.1.2. Experimental unit

The electrochemical reactor consists of three major components. A direct current (DC) power supply with an adjustable electrical potential of 0-30 V and current intensity of 0 -5 A was one component. The second component was a 500 mL electrolytic cell containing sodium chloride solutions. The cathode and anode electrodes were present in the electrolytic cell. Both electrodes were made of iridium oxide plated with titanium (IrO_2/Ti) with dimensions of 0.1 cm thickness 2 cm width 6 cm height and an effective total surface area of 48 cm^2. The Sungwon Forming Company (Korea) supplied the IrO_2/Ti electrodes. The distance between the electrodes was set to between 1.3 and 5 cm. The third component was a magnetic stirrer, which continually stirred all experimental runs at 150 rpm. The DPD (diethyl-p-phenylenediamine) technique was used to determine the concentration of residual chlorine. Figure 3.1 shows the experimental setup.

3.1.3. Experimental design and statistical analysis

When comparing the different design approaches of response surface methodology (RSM) tools, BBD is more efficient than central composite design (CCD) and three-level complete factorial design [80].

3.2 Materials and Chemicals

Sodium chloride (NaCl), sulfuric acid ($H_2 SO_4$), sodium hydroxide (NaOH), methanol, phosphate buffer, and distilled water were used as analytical-grade reagents. Amoxicillin

trihydrate and ibuprofen were obtained from commercial pharmaceutical sources. HPLC-grade water was used for chromatographic analysis.

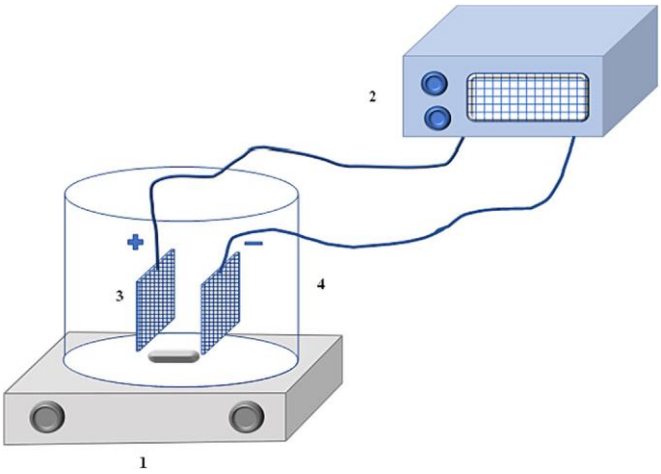


Figure 3- 1 *Laboratory scale schematic diagram of experimental setup: (1) magnetic stirrer, (2) DC source, (3) electrodes (IrO_2/Ti), (4) reactor*

BBD was selected from various experimental RSM models because it is an independent and rotatable quadratic model without embedded factorial points. BBD also requires few experiments, which makes it more effective and economically viable than other experimental design methods. Therefore, BBD is widely used to optimize research compared with other experimental design methods. The interaction effects of EO process operating parameters, such as NaCl concentration (A), electric potential (B), electrolysis time (C), and electrode gap (D), on residual chlorine production and energy consumption efficacy were examined using a four-level BBD experimental design under RSM. Under this condition, the BBD involved 29 runs of EO, including five replicates and centre points. Residual chlorine (Y1) and energy consumption (Y2) were considered as response variables.

ANOVA was used to investigate the interaction effects between the independent and dependent variables and to determine the optimal operating conditions. In this approach, a second-order polynomial model (regression equation) with relationship equation (6) was used to assess the

interresidual impacts of the four independent variables on the two dependent variables (responses).

$$Y = \beta_0 + \sum_{i=1}^{k} (\beta_i X_i + \beta_{ii} X_i^2) + \sum_{1 \le i < j}^{k} \beta_{ij} X_i X_j + \varepsilon \quad (6)$$

Where, Y is the response; β_0, βi (i = 1, 2, 3), and βij (i = 1, 2, 3; j = 1, 2, 3) are model coefficients for intercept, linear, quadratic, and interaction, respectively; Xi and Xj are the coded input factors; and ε is the residual. This mathematical correlation between the residual chlorine and energy consumption and the selected input variables can be used as a model equation to predict the optimum values. The value of the electrical energy consumption for the EO was calculated in kWh/L, as shown in Equation (7):

$$C\ energy\ (kWh\ L^{-1}) = (U \times I \times t) / V \quad (7)$$

where U is the cell potential (V) at the desired current I (A), t is the electrolysis time (h), and V is the treated sample volume (L). Table 3.1 shows the experimental region explored, the types of variables, the coded values, and their levels based on the Box–Behnken response surface design.

Table 3.1 Experimental range and levels of independent factors for the optimization of residual chlorine generation using EO process

		Range and levels		
Factor	**Unit**	**-1**	**0**	**+1**
NaCl concentration	g/L	5	15	25
Electric potential	V	3	9	15
Electrolysis time	min	5	15	25
Electrode gap	cm	1	3	5

3.1.4. Artificial neural network (ANN)

ANN is a subfield of machine learning inspired by the complex signal transduction process in the human nervous system. The ANN architecture was composed of different layers, and many neurons were allocated to each layer. The feed-forward network (multilayer perceptron) of the ANN was used in this study to create a nonlinear mapping between the independent and response

variables. All independent variables employed in this study, such as NaCl concentration, electric potential, electrolysis time, and electrode gap, are represented by four neurons in the input layer. Similarly, the response variables (residual chlorine concentration and energy consumption) were represented by two neurons in the output layer. Other characteristics such as the number of neurons and learning algorithms were optimized through trial and error. All ANN-based models were implemented in the neural network tool box (NNtool) in MATLAB (ver. R2018b, MathWorks, USA) software. The experimental combinations and responses used in the RSM-based optimization were normalized (parametric values of the variables were converted between 1 and 1 and then used for the construction of the ANN model). The values of the variables were normalized using the following expression:

$$X_{norm} = 2 * (X_i - X_{min}) / (X_{max} - X_{min}) - 1 \qquad (8)$$

Where, X_n represents the normalized data, X_i denotes the actual or observed data, and X_{min} and X_{max} indicates the upper and lower most values of the variables, respectively. The dataset used to build the ANN model is divided into three parts: training (70%), testing (15%), and validation (15%). The weights and biases for each linked neuron in the layers were assigned randomly by the ANN until the prediction value approximated the target dataset. The coefficients of determination (R2) obtained throughout the training, testing, and validation phases were utilized to identify the optimum ANN architecture. Six other sets of trial combinations (not used during the RSM-based optimization) were tested to validate the ANN model. The ANN- and RSM-predicted data for those combinations were compared with the actual data using the performance metrics (R2, root mean square error [RMSE], and absolute average deviation [AAD]) listed below (Equation 9-11).

$$RMSE = \sqrt{[(1/n)\,\Sigma(P_i - \hat{P}_i)^2]} \qquad (9)$$

$$R^2 = 1 - [\Sigma(P_i - \hat{P}_i)^2 / \Sigma(P_i - \bar{P})^2] \qquad (10)$$

$$AAD\% = (1/n) * \Sigma\,|(P_i - \hat{P}_i) / P_i| \times 100 \qquad (11)$$

Where n, P_i, $\hat{P}_i$ and $\bar{P}$ are number of data points, the data observed (by the ANN and RSM), predicted data and the, mean of observed data, respectively. The component n refers to the total number of observations.

3.2. Coupling Electro-oxidation and UV Irradiation for the Treatment, and Mineralization of Amoxicillin and Ibuprofen wastewater: RSM Parametric Optimization and Toxicity Assessment

3.2.1. Chemicals

Amoxicillin trihydratae (AMOX caps 500mg, Denk pharma) and Ibuprofen (Gofen 400, Mega pharma) capsules were purchased from a commercially available pharmaceutical store. Laboratory grade Sodium chloride, used as the electrolyte, was supplied by Sigma Aldrich. Sulfuric acid (5% V/V) and Sodium hydroxide (5%, m/m) were used to adjust the initial solution pH to acidic or alkaline conditions. Distilled water was used to prepare experimental solution except for HPLC experiments where HPLC grade water was used. Also in the entire HPLC experiments methanol was used as solvent.

3.2.2. Experimental set-up

3. 2.2.1 Electro oxidation experiments

NaCl with a concentration of 1.5 g/L was used as electrolyte. It was dissolved in 600 mL beaker with distilled water. A DC source power supply having adjustable electric potential of 0 to 30 V and electrical current of 0 to 5 A was used. IrO_2/Ti electrodes were used as Cathode and Anode with dimensions of 0.1 cm thickness, 2 cm width, 6 cm height. The surface area of the electrode is 48cm^2. The electrolysis time was set at 25 minutes and electrode gap was 1 cm. These values were chosen from the previous residual chlorine generation experiment [16]. For electrolysis 600mL beaker was used.

3.2.2.2. UV only and EO-UV experiments

A 23cm× 6cm× 8 cm rectangular glass reactor covered with Aluminum foil with a hole on its sides to hold the UV-C lamp, with 254nm wavelength was used to conduct the UV photolysis. The UV

irradiation intensity ($mWcm^{-2}$) at the three positions from the specimen was characterized using radiometer. For the integrated experiments (EO-UV) the rectangular glass reactor with UV-C lamp and suspended electrodes were used see Figure 6.

For EO-UV integrated experiments a rectangular glass reactor with dimensions of 23 cm× 6 cm× 8 cm was utilized as the experimental reactor. A UV-C lamp emitting light at a wavelength of 254 nm served as the source for UV photolysis. The UV irradiation intensity (mW/cm^2) was measured at two locations, resulting in UV intensities of UV_1 = 5.47 mW/cm^2 and UV_2 = 7.82 mW/cm^2, using a radiometer. IrO_2/Ti electrodes were suspended in the reactor during the electrooxidation process. Continuous mixing was maintained with the aid of a magnetic stirrer. Figure 2 provides a schematic illustration of the system.

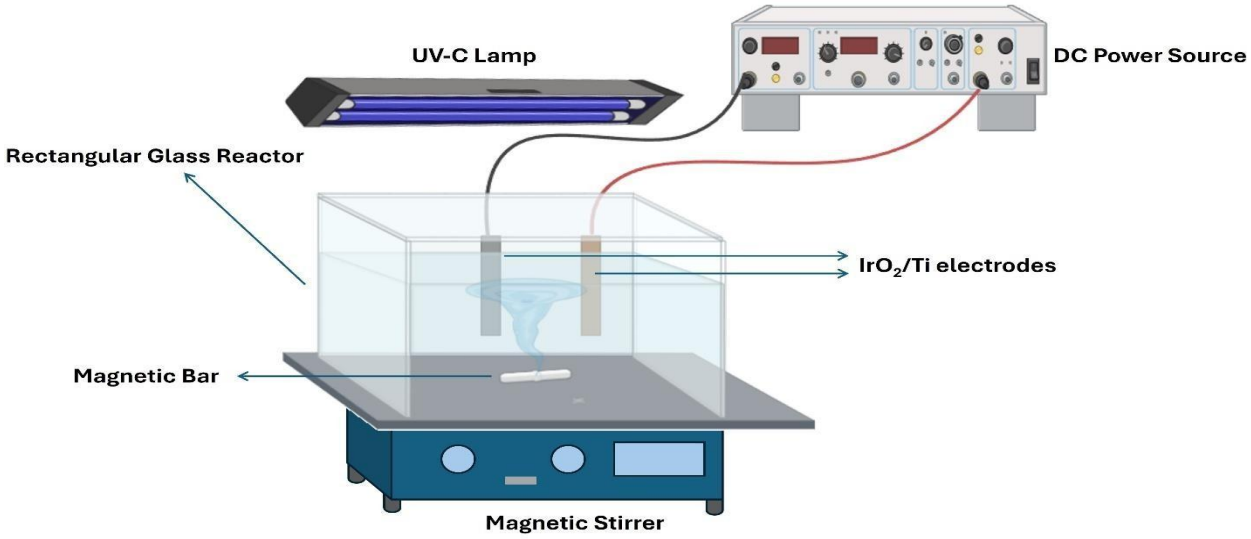


Figure 3- 2 Schematic diagram of integrated EO-UV reactor set up

3.2.3. Analytical methods

To monitor Amoxicillin and Ibuprofen drugs, degradation samples were taken for monitoring. The samples were filtered through a 0.45 μm membrane filter manufactured by Merck Millipore. The analysis of the samples was conducted using a high-performance liquid chromatography (HPLC) system, specifically the Agilent 1260 Infinity-II from the USA, which was equipped with

a C-18 column measuring 250 × 4.6 mm and having a particle size of 5 µm. The analysis was carried out using an isocratic mobile phase consisting of methanol and phosphate buffer in a 40:60 v/v ratio, delivered at a flow rate of 1.0 mL/min. The column temperature was maintained at 25℃ during the analysis.

Figure 3- 3 *HPLC system setup*

To analyze mineralization a Total Organic Carbon (TOC) analyzer was used (TOC-VCPH, PC controlled TOC Analyzer (Shimadzu, Japan).

TOC was determined using TOC removal efficiency (%), which was computed using following equation Equation 12.

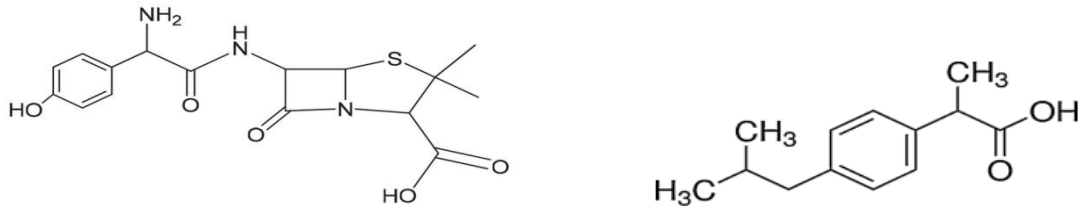
$$\textbf{Removal of TOC (\%)} = (\textbf{TOC}_0 - \textbf{TOC}_f) / \textbf{TOC}_0 \times 100 \quad (12)$$

Where TOC_0 and TOC_f denote the amount of TOC (mg L^{-1}) in synthetic wastewater before and after continuous EO treatment, respectively. In the same manner the COD removal (%) was calculated using Equation 13:

$$\textbf{COD removal efficiency (\%)} = (\textbf{COD}_0 - \textbf{COD}_f) / \textbf{COD}_0 \times 100 \quad (13)$$

Where COD_0 and COD_f denote the amount of COD (mg L^{-1}) in synthetic wastewater before and after continuous EO treatment, respectively. For this experiment two drugs were used Amoxicillin from antibiotics and Ibuprofen from NSAID therapeutic classes.

Amoxicillin (7-amino-3,3-dimethyl-6-oxo-2-thia-5-azabicycloheptane-4-car-boxylic acid) with the empirical formula $C_{16}H_{19}N_3O_5S$ shown in Figure 4(a) is a bactericidal β-lactam antibiotic of the aminopenicillin family and Ibuprofen (isobutylphenylpropionic acid) with molecular formula of $C_{13}H_{18}O_2$ is a non-steroidal anti-inflammatory drug that is used to relieve pain, fever, and inflammation. The molecular structure is depicted in Figure 3.4(b).



(a) (b)

Figure 3- 4 (a) The structure of Amoxicillin (b) The structure of Ibuprofen.

Figure 3- 5 Samples after EO (left) and EO-UV (right) experiments

Figure 3- 6 *EO-UV Experimental Setup*

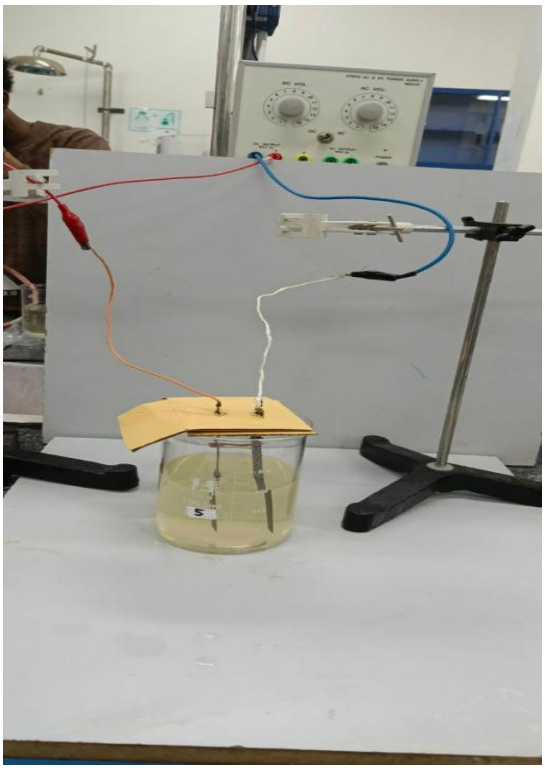


Figure 3- 7 *EO Experimental Set up*

3.2.4. Experimental design and statistical analysis

In a three level complete factorial design, Box Behenken Design (BBD) approach is efficient as response surface methodology tool than central composite design (CCD)[80]. BBD is also an independent model and needs few experiments that make it more economically feasible than other experimental design methods.

Thus, BBD is extensively used to optimize research studies compared with other experimental design models. The effects of current density (A), initial concentration (B) and pH (C) of pharmaceuticals (Amoxicillin and Ibuprofen) on COD removal efficiency were examined using a three level BBD experimental design and RSM. Based on this the BBD comprises of 17 runs of EO, including five replicates and centre points. COD removal efficiency (Y) (for AMOX and IBU) was considered as response variable.

To examine the interaction effects between the independent and dependent variables and to determine the optimal operating conditions ANOVA was used.

To evaluate the inter-residual effects of the three independent on the one dependent variable (response), a second order polynomial model (regression equation) with the relationship Equation 3 was used.

Table 3. 2 Experimental range and levels of independent factors for the optimization of AMOX and IBU degradation using EO process

		Range and Levels		
Factor	**Units**	**-1**	**0**	**+1**
Current density	$mAcm^{-2}$	10	27.5	45
Initial Pharm. Concentration	μgL^{-1}	276	288	300
pH	-	3	6	9

$$Y = \beta_0 + \sum \beta_i X_i + \sum \beta_{ii} X_i^2 + \sum\sum \beta_i \square X_i X \square + \epsilon \quad (14)$$

Where, Y is the response; β0, βi (i = 1, 2, 3), and βij (i = 1, 2, 3; j = 1, 2, 3) are model coefficients for intercept, linear, quadratic, and interaction, respectively; Xi and Xj are the coded input factors; and ε is the residual

Chapter Four
Results and Discussions

4.1. Applications of integrated response surface methodology statistic techniques and artificial neural network-based machine learning to optimize residual chlorine production and energy consumption

4.1.1.. RSM analysis

The experiments were carried out in a completely randomized design with four factors, namely, NaCl concentration, electric potential, electrolysis time, and electrode gap, at three levels and two experimental responses, namely, residual chlorine degradation and energy consumption, as shown in Table 3.2.

During the preliminary tests, the experimental domains were determined by considering the minimum and maximum sodium chloride concentrations (5 and 25 gL^{-1}), electrolysis period (5 and 25 min), electric potential (3 and 15 V), and electrode gap (1 and 5 cm). A total of 29 tests were conducted (including the number of blocks and centre points).

To describe the interaction between the specified study variables and the response, two-factorial interactions (2FI) and a quadratic equation were chosen for the two experimental responses of residual chlorine and energy consumption, respectively. The coefficients of determination (R^2) for the regression models of residual chlorine and energy consumption according to RSM were 0.9914 and 0.9993, respectively. This means that the models predicted 99.14% and 99.93% variability within the experimental range. The "pred R^2" values of 0.9754 and 0.9964 further imply that model equations can predict the responses even when the research factors out of the study experimental. The R^2 and predicted R^2 values for an acceptable model should typically be close to 1.

Table 4.1 Experimental design and dependent variables of residual chorine and energy consumption

Exp'tal run	Factor 1 A gL^{-1}	Factor 2 B V	Factor 3 C min	Factor 4 D cm	Obs. resp. Residual chlorine mgL^{-1}	Pred. resp.	Obs resp Energy consumption $kWhL^{-1}$	Pred resp
1	15	9	15	3	950	975.01	6.28	6.44
2	25	9	15	1	1971	2074.10	13.27	13.65
3	25	3	15	3	191.4	155.21	0.1	—0.5501
4	15	9	25	1	2137.6	2072.89	14.76	14.98
5	15	3	15	1	148.89	111.51	0.3	0.0232
6	15	3	5	3	33.7	71.38	0.072	0.4760
7	15	9	25	5	1203.5	1080.86	7.89	7.59
8	15	15	15	1	2610.89	2459.98	38.7	38.41
9	15	15	15	5	1234	1193.50	28.6	28.91
10	25	9	25	3	2407	2538.98	17.5	17.74
11	5	9	5	3	79.7	61.63	0.576	0.3773
12	5	9	25	3	519	614.76	3.69	3.97
13	25	9	5	3	666.5	684.65	3.2	2.96
14	15	3	15	5	62	135.03	0.15	0.4732
15	5	3	15	3	23	91.34	0.075	0.1842
16	15	3	25	3	177	175.16	0.369	0.4596
17	5	15	15	3	540.6	585.06	23.96	24.69

18	15	15	25	3	2843	2978.58	43.6	43.08
19	5	9	15	5	196.7	179.01	1.46	0.9563
20	15	9	15	3	1084.7	975.01	6.58	6.44
21	15	9	5	5	296	247.67	1.4	1.27
22	25	9	15	5	1098.9	1149.52	6.3	6.60
23	25	15	15	3	3128.5	3068.42	41.8	41.77
24	15	9	15	3	1063.5	975.01	6.79	6.44
25	15	15	5	3	499.8	674.90	24.9	24.69
26	15	9	5	1	489	498.60	2.55	2.93
27	15	9	15	3	1072	975.01	6.28	6.44
28	5	9	15	1	462.6	497.39	3.37	2.95
29	15	9	15	3	1084.7	975.01	6.28	6.44

Tables 4.4 and 4.5 show the analysis of variance (ANOVA) results for residual chlorine and energy usage, respectively. ANOVA is a set of statistical models and estimate processes used to examine differences between means. The ANOVA table allows us to make conclusions regarding the overall significance of our model. The F ratios and p values indicate whether or not each individual factor is connected to the response. The model F values of 206.69 and 1506.72, respectively, with corresponding p values less than 0.0001, indicate that the independent variables in the model have a significant impact on the dependent variable, in this case, residual chlorine and energy consumption. p values less than 0.05 in both ANOVA tables imply that the model terms are significant, and vice versa. In the present study, all the study variables were found to have a significant impact on the study response.

The signal-to-noise ratio is measured by the “adeq Precision” for the two responses, which were similarly discovered to be 46.46 and 126.11, and it is always preferred to be higher than 4. The

proposed 2FI and quadratic models can be used to explore the design space. Table 4.1 provides the model-predicted values for comparison with actual experimental data.

Equations (13) and (14) include the model equations that interact the responses to the process variables in terms of their actual value.

Table 4. 2 Model summary statistics for residual chlorine generation

Source	**Std. dev.**	**R^2**	**Adjusted R^2**	**Predicted R^2**	**PRESS**	
Linear	405.21	0.8288	0.8003	0.7326	6.15E+06	
2FI	105.07	0.9914	0.9866	0.9754	5.66E+05	Suggested
Quadratic	97.92	0.9942	0.9883	0.9688	7.18E+05	
Cubic	88.18	0.998	0.9905	0.789	4.86E+06	Aliased

Table 4. 3 Model summary statistics for Energy Consumption

Source	**Std. dev**	**R^2**	**Adjusted R^2**	**Predicted R^2**	**PRESS**	
Linear	6.51	0.7914	0.7567	0.6824	1550.7	
2FI	6.6	0.8395	0.7503	0.5093	2396.33	
Quadratic	0.481	0.9993	0.9987	0.9964	17.74	Suggested
Cubic	0.2096	0.9999	0.9997	0.9986	6.78	Aliased

Table 4. 4 Analysis of variance (ANOVA) of the residual chlorine response.

Source	**Sum of squares**	***df***	**Mean square**	***F* value**	***p* value**	
Model	2.282E+07	10	2.282E+06	206.69	<0.0001	significant
A-NaCl concentration	4.866E+06	1	4.866E+06	440.78	<0.0001	
B-Electric potential	8.705E+06	1	8.705E+06	788.52	<0.0001	
C-Electrolysis time	4.347E+06	1	4.347E+06	393.73	<0.0001	
D-Electrode gap	1.159E+06	1	1.159E+06	104.95	<0.0001	
AB	1.463E+06	1	1.463E+06	132.56	<0.0001	
AC	4.233E+05	1	4.233E+05	38.34	<0.0001	
AD	91 869.61	1	91 869.61	8.32	0.0099	
BC	1.210E+06	1	1.210E+06	109.59	<0.0001	
BD	4.160E+05	1	4.160E+05	37.68	<0.0001	
CD	1.373E+05	1	1.373E+05	12.44	0.0024	
Residual	1.987E+05	18	11 040.23			
Lack of fit	1.857E+05	14	13 261.04	4.06	0.0929	Not significant
Pure error	13 069.63	4	3267.41			
Cor total	2.302E+07	28				

Table 4. 5 Analysis of variance (ANOVA) of the energy consumption response.

Source	**Sum of squares**	***df***	**Mean square**	***F* value**	***p* value**	
Model	4880.01	14	348.57	1506.72	<0.0001	Significant
A-NaCl concentration	200.40	1	200.40	866.25	<0.0001	
B-Electric potential	3349.82	1	3349.82	14 479.76	<0.0001	
C-Electrolysis time	253.10	1	253.10	1094.04	<0.0001	
D-Electrode gap	61.43	1	61.43	265.52	<0.0001	
AB	79.34	1	79.34	342.97	<0.0001	
AC	31.28	1	31.28	135.22	<0.0001	
AD	6.40	1	6.40	27.67	0.0001	
BC	84.67	1	84.67	365.98	<0.0001	
BD	24.75	1	24.75	106.99	<0.0001	
CD	8.18	1	8.18	35.36	<0.0001	
A^2	1.12	1	1.12	4.86	0.0447	
B^2	715.03	1	715.03	3090.76	<0.0001	
C^2	0.3563	1	0.3563	1.54	0.2350	
D^2	0.0014	1	0.0014	0.0061	0.9389	
Residual	3.24	14	0.2313			
Lack of fit	3.02	10	0.3020	5.52	0.0571	Not significant
Pure error	0.2189	4	0.0547			
Cor total	4883.25	28				

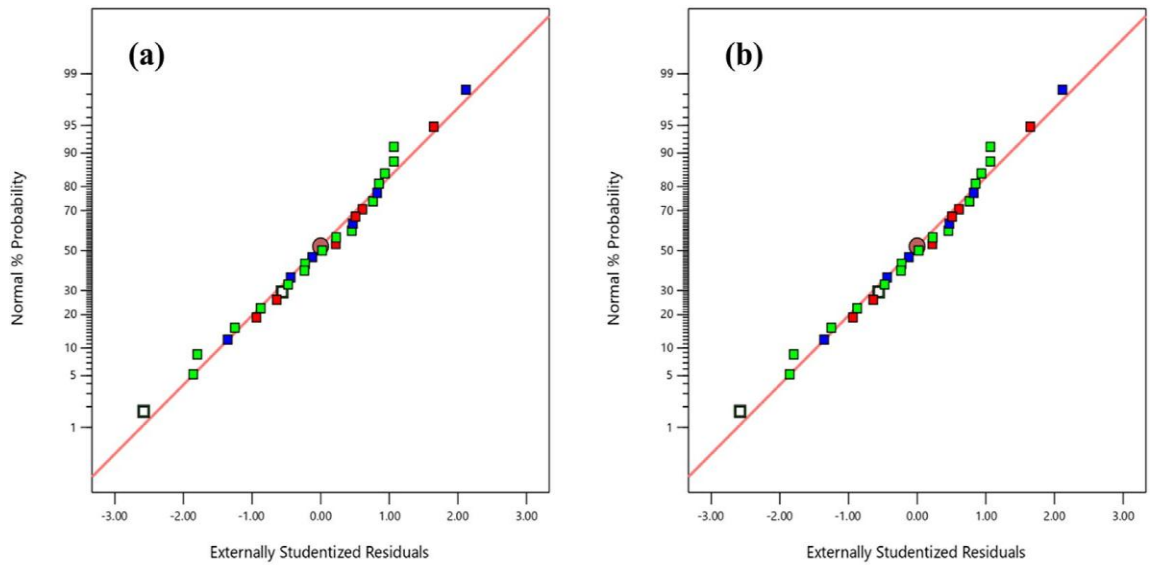


Figure 4.1 Normal plot of (a) residual chlorine and (b) energy consumption

$$\text{Residual Chlorine (mg L}^{-1}) = 152.359 - 53.113A - 66.132B - 43.312C + 339.124D + 10.081AB + 3.253AC - 7.577AD + 9.166BC - 26.875BD - 9.264CD \quad (15)$$

$$\text{Energy Consumption (kWh L}^{-1}) = 10.006 - 0.364A - 4.107B - 0.506C + 2.733D + 0.074AB + 0.028AC - 0.063AD + 0.077BC - 0.207BD - 0.072CD - 0.004A^2 + 0.291B^2 + 0.002C^2 + 0.004D^2 \quad (16)$$

The normal plot of internally studentized residuals for residual chlorine production and energy consumption is shown in Figure 4.8 (a) and (b). The correlation between the actual and predicted values of residual chlorine production and energy consumption is shown in Figure 4.2 (a) and (b). The response surface plots for the amount of residual chlorine produced as a function of different operational parameters are shown in Figure 4.3 a-f. The maximum residual chlorine generation was attained at an electric potential of 15 V during 25 min of electrolysis and 1-cm electrode gap in the presence of 25 g/L of sodium chloride. During electrolysis in the presence of NaCl electrolyte, chlorine gas is produced at the anode (Equation 15) and hydrogen gas. (Equation 16) The dissociation of chlorine produced in water results in hypochlorite and chloride ion (Equation

17) [81][67]. These reactions are as follows [82] [83].

$$2Cl^- \rightarrow Cl_2 + 2e^- \quad (17)$$

$$2H_2O + 2e^- \rightarrow 2OH^- + H_2 \quad (18)$$

$$Cl_2 + 2OH^- \rightarrow OCl^- + Cl^- + H_2O \quad (19)$$

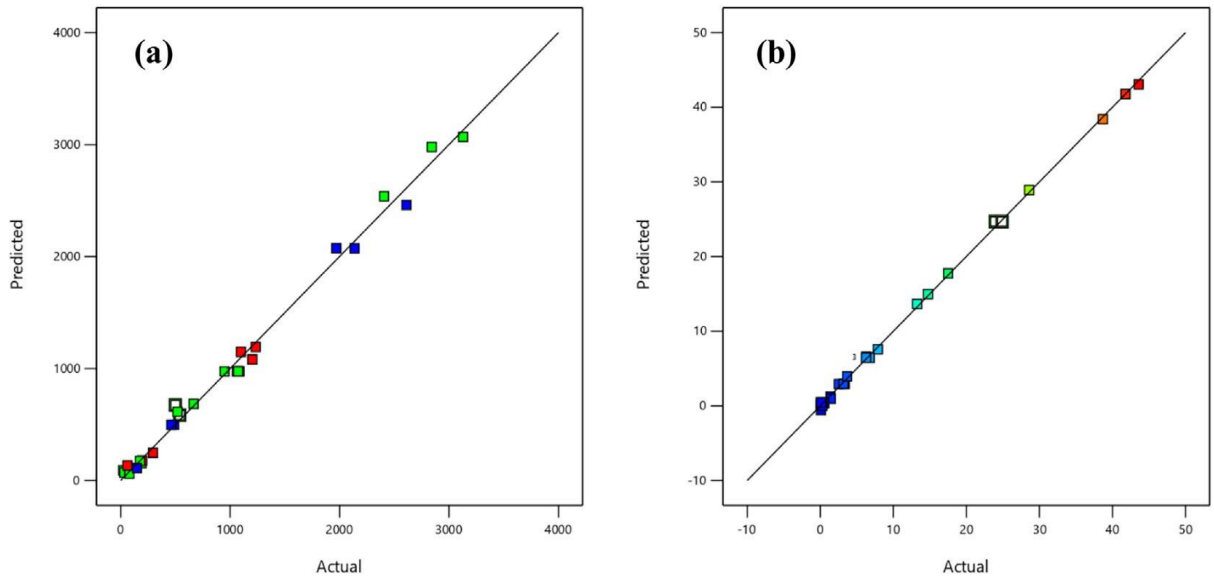


Figure 4.2 Predicted vs. actual plot for (a) residual chlorine and (b) energy consumption at the cathode

The electrode gap for residual chlorine has been demonstrated by the indirect oxidation of chlorine ions in the bulk solution and directly on the electrode surface [84]. According to the figures, as the independent variables sodium chloride concentration, electric potential, and electrolysis time increase, a large amount of residual chlorine (active chlorine) is produced. However, the amount of residual chlorine produced was not significantly affected by electrode spacing. The electric potential, sodium chloride concentration, and electrolysis time were the three elements that produced residual chlorine most effectively in that sequence. Additionally, it was demonstrated that increasing the electric potential causes an increase in charge, which speeds up the movement of electrons across the electrodes. An increased electric potential promotes the production of chlorine, which increases the current density passing through the electrodes. Although some studies have indicated that electrolysis for longer periods of time with higher electric potential

reduces the amount of residual chlorine produced, in general, the electric potential directly affects the production of residual chlorine [85].

Additionally, as the electric potential increased, more current flowed through the electrolysis apparatus, increasing the current density, which in turn accelerated the transfer of electrons and produced more chlorine gas. The concentration of sodium chloride is another factor that directly affects the formation of residual chlorine. The findings showed that when the NaCl concentration increased, more residual chlorine was formed. This is because more chloride ions are produced, leading to an increase in the number of chlorine molecules produced [86]. The duration of electrolysis is another important aspect; as it continues, more chloride ions are formed and more chlorine molecules are produced in the electrolyte solution [87] [88]. Another factor that has little impact on the formation of residual chlorine is the electrode gap. However, it increased the quantity of residual chlorine when it interacted with the other three components. A greater electric potential and smaller electrode gap resulted in relatively larger residual chlorine. This is because of the fact that more chlorine will be created the closer the electrode distance is to zero [89]. According to the findings, the sodium chloride concentration and electrolysis time had less of an impact on residual chlorine production than the electrode gap interaction with electric potential.

The impact of the selected operational parameters on the energy usage of the system is shown in Figure 4.4. According to other researchers' findings [90], there is a clear correlation between electric potential and energy consumption: as electric potential increases (Figure 4.4 a), so does current density, and as a result, energy consumption. Energy use also increases as sodium chloride concentration increases [88]. The concentration of sodium chloride has less impact on energy consumption than the electric potential. Because the system's primary driving force is electrical potential, a higher electric potential will result in greater current flow through the bulk solution. Additionally, the electrode gap had less of an impact on the electrolysis time and any interactions between the other parameters shown in Figure 4.3 and 4.4 (c),(e),(f)

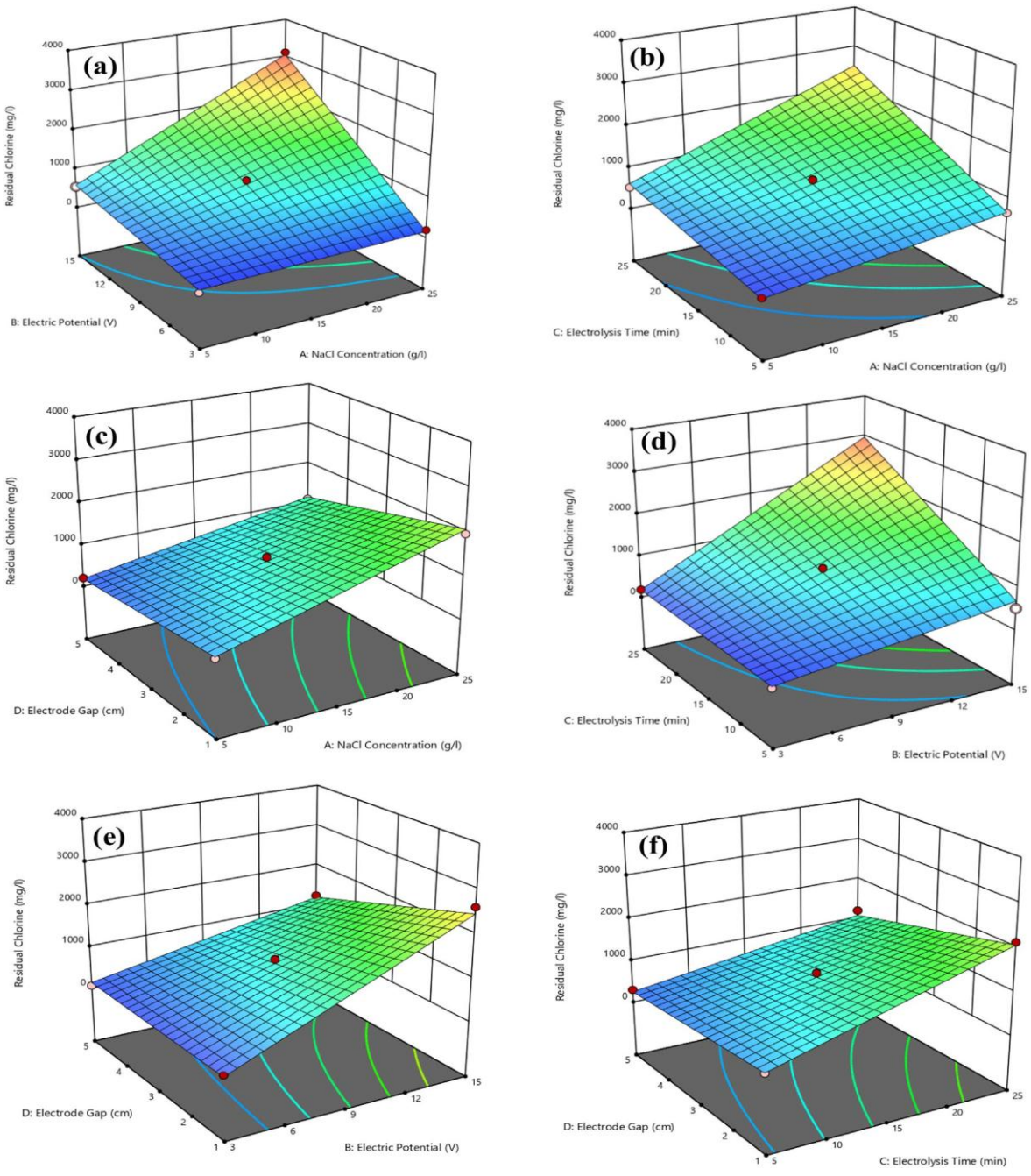


Figure 4.3 Surface response plot of chlorine production as a function (a) NaCl concentration and electric potential, (b) NaCl concentration and electrolysis time, (c) NaCl concentration and electrode gap, (d) electrolysis time and electric potential, (e) electric potential and electrode gap, and (f) electrolysis time and electrode gap.

4.1.2. ANN analysis

The correlation of the relationship between input and output by the ANN is also successful. For residual chlorine and energy consumption, three-layer (input-hidden-output) perceptron networks with topologies of 4-4-1 for training, test, and validation errors and 4-11-1 for training, test, and validation errors were determined to be the best. The outputs from the ANN were compared with the actual experimental values, as shown in Figure 4.6 a and b to assess these mistakes. Figure 4.5a shows that for residual chlorine, the training data, test data, and validation data R^2 values were discovered to be 0.9974, 0.9999, and 0.9983, respectively. Similar results for energy consumption were determined to be 0.9998, 0.9999, and 0.9999 (Figure 4.5b). The outcomes demonstrate that the chosen ANN topology provides very accurate approximations and has the necessary generalization capacity to forecast the residual chlorine of the electro-oxidation process under various operating conditions. The two projections are quite similar to one another when ANN and RSM predictions for residual chlorine and energy consumption.

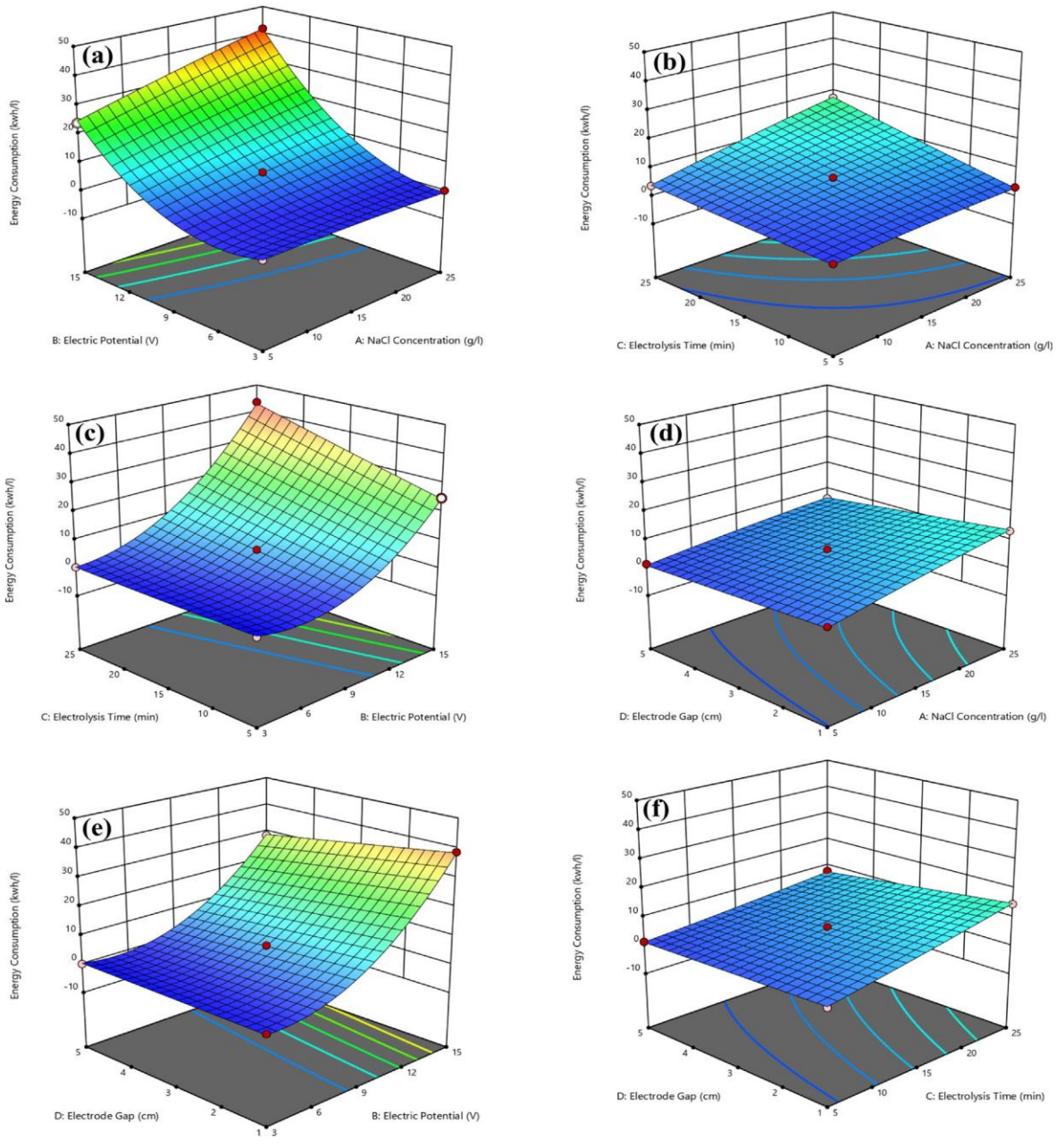


Figure 4.4 Surface response plot of energy consumption as a function (a) NaCl concentration and electric potential, (b) NaCl concentration and electrolysis time, (c) NaCl concentration and electrode gap, (d) electrolysis time and electric potential, (e) electric potential and electrode gap, and (f) electrolysis time and electrode gap.

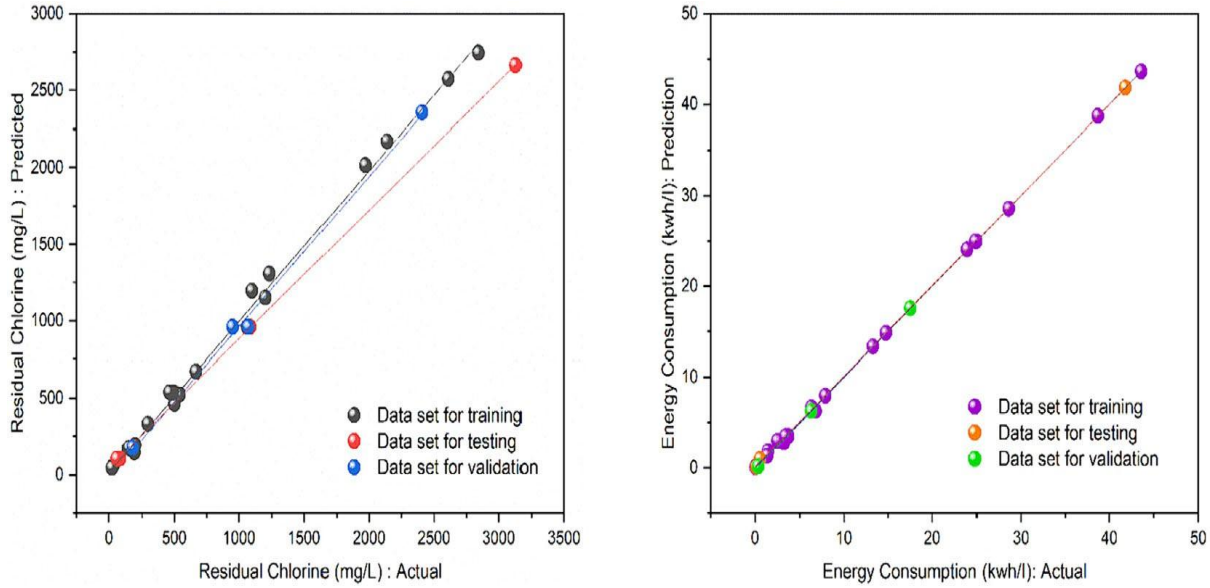


Figure 4.5 Correlation plots of actual and predicted for training, testing and validation data set for (a) residual chlorine and (b) energy consumption at the best fitted ANN topology.

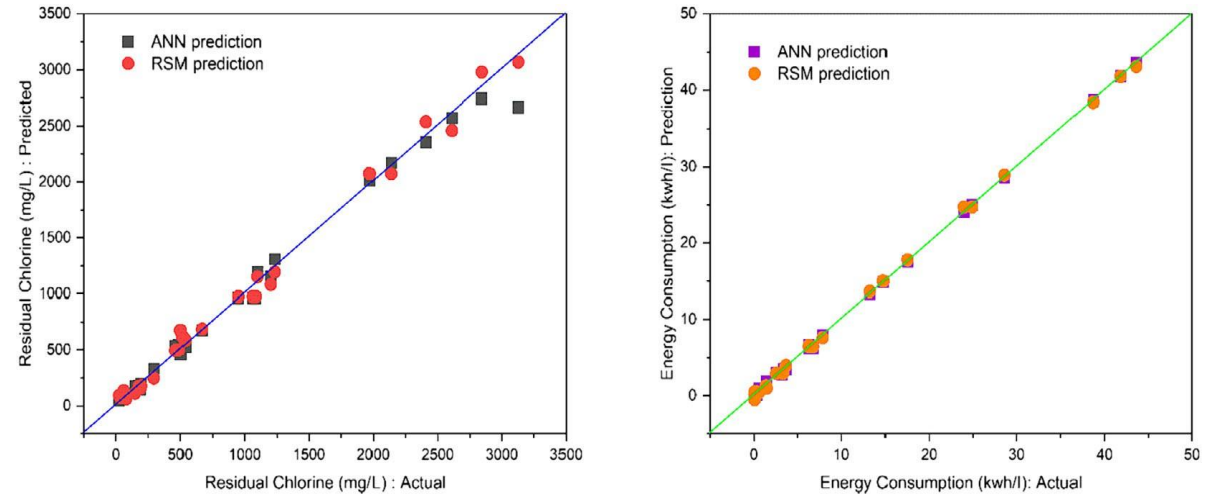


Figure 4.6 Comparison of predicted and actual values between RSM and ANN for (a) residual chlorine and (b) energy consumption.

Table 4. 6 Performance of ANN- and RSM-based predictions

	Residual chlorine production		Energy consumption	
Statistical parameters	RSM	ANN	RSM	ANN
RMSE	105.50	82.77	0.33	0.21
R^2	0.9861	0.9897	0.9993	0.9999
ADD (%)	16.15	8.91	4.72	0.74

AAD were low, but the values for ANN models are significantly lower than those for RSM models (Table 4.6). This further demonstrated that for the electro-oxidation process, ANN models have superior prediction ability over RSM models described in Figure 4.6.

4.1.3. Optimization with RSM

The desired set of electro-oxidation process parameters was established by optimization of the responses from the two model equations (Equations 13 and 14), which was based on Derringer's desirability function technique for multiple response processes. To maximizes residual chlorine while minimizing energy consumption, the four process variables-NaCl concentration, electric potential, electrolysis time, and electrode gap-were bound between the lower and higher levels (NaCl concentration of 5–25 mg/L, electric potential of 3–15 V, electrolysis time of 5–25 min, and electrode gap of 1–5 cm). This is because all study parameters were equally important.

As a result, the model's predictions for these circumstances were 3,106.96 mg/L of residual chlorine and 21.756 kWh/L of energy consumption at the best operating conditions of 25 mg/L of NaCl, 8.8 V of electric potential, 25 min of electrolysis, and 1 cm of electrode gap, with a desirability value of 0.769. On samples prepared under optimum circumstances, the expected values were also verified. The outcomes demonstrate that the model equations might perhaps be used to predict the electro-oxidation process parameters given the pred R^2 for each response.

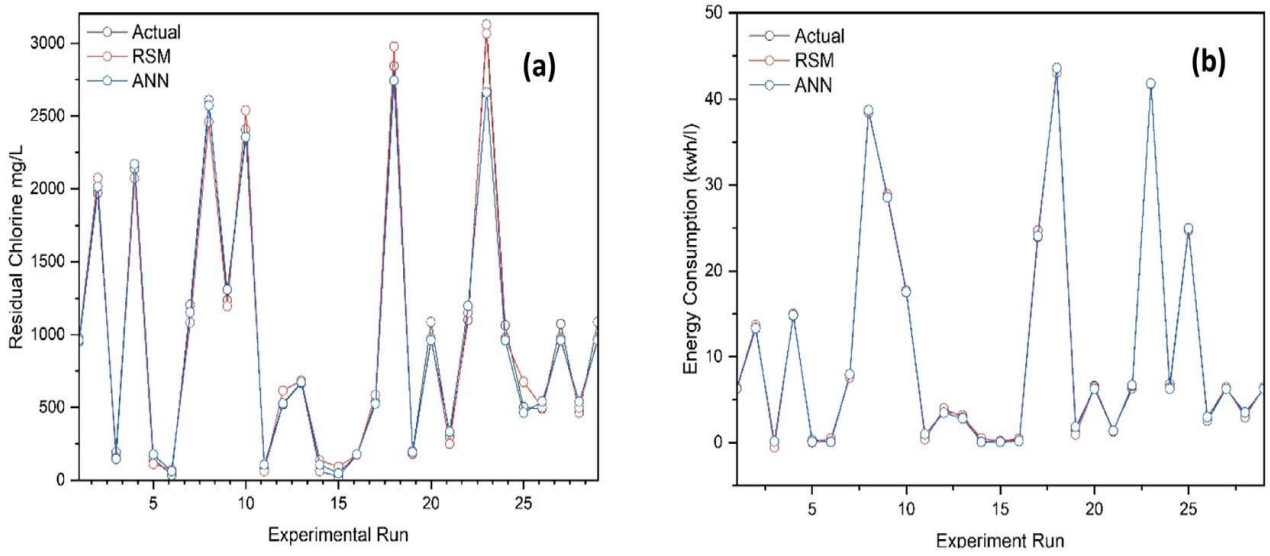


Figure 4.7 Comparison of actual experimental values with RSM and ANN for all experimental run (a) residual chlorine and (b) energy consumption.

4.1.4. Multi-objective genetic algorithm

The second-order polynomial equation obtained from RSM-based optimization experiments was used as fitness function to run the multi-objective genetic algorithm. The aim of this study was to maximize the production of residual chloride at minimum energy consumption. The desirability function of GA was used to optimize multiple responses simultaneously. After running the multi-objective GA, the maximum residual chloride that can be produced using electro oxidation process was estimated to be 4,283.01 mg/L under the following conditions: NaCl concentration of 23.38 g/L, electric potential of 14.55 V, electrolysis time of 21.83 min, and electrode gap of 1.67 cm. Similarly, the optimum energy consumption was estimated to be 55.62 kWh/L. The GA-estimated optimum values were greater than the values predicted by the RSM. A confirmatory experiment performed over the GA-estimated conditions showed the maximum production of residual chloride of 4125 mg/L with the energy consumption of 54.82 kWh/L.

4.2. Electro-oxidation process parameter optimization for the treatment of AMOX and IBU from Synthetic wastewater

4.2.1. Model fitting and analysis of variance (ANOVA)

The experiments utilized a random design featuring three factors: the initial concentrations of pharmaceuticals (amoxicillin and ibuprofen), current density, and pH, each evaluated at three different levels. The response measured in the experiment was the removal of AMOX and IBU, indicated by chemical oxygen demand (COD), as outlined in Table 4.10.

During initial testing, the experimental domains were established based on the minimum and maximum initial concentrations of pharmaceuticals (amoxicillin and ibuprofen) at 276 µg/L and 300 µg/L, current densities of 10 mA/cm^2 and 45 mA/cm^2, and pH levels ranging from 3 to 9.

Table 4.10 presents the experimental region that was investigated, detailing the types of variables, their coded values, and corresponding levels as per the Box-Behnken response surface design. Furthermore, the table indicates that the maximum removal rates for AMOX and IBU, expressed as COD, were 80 % and 95 %, respectively, both achieved at a current density of $45mA/cm^2$

The model statistics are the first statistical metrics and information that describe the performance and fit of a response surface model. It comprises R^2, adjusted R^2, p values for the model and individual terms, and other statistics that help to determine the quality of the model.

Also, the model summary statistics describe the correlation of the three independent study variables (Initial pharmaceutical concentration, current density and pH) and one dependent variable (removal of pharmaceuticals (amoxicillin and ibuprofen) are presented in Table 4.7.

To define the interaction between the experimental factors and the response, two-factorial interactions (2FI) and a quadratic equation were selected for the one response variable of removal of amoxicillin and ibuprofen as COD. Accordingly, the quadratic

model equation for the two responses is presented with Eqs. (18) and (19).

The coefficients of determination (R^2) for the regression models assessing the removal of amoxicillin and Ibuprofen were determined to be 0.9791 and 0.9807, respectively. This indicates that the models predicted 97.91 % and 98 % of the variability within the experimental range.

The predicted R^2 values of 0.9077 for amoxicillin and 0.7782 for Ibuprofen suggest that the model equations can predict with good accuracy even when the study parameters fall outside the experimental range. A suitable model should have R^2 and predicted R^2 values that are close to 1, or generally above 0.7.

Tables 4.8 and 4.9 present the results of the Analysis of Variance (ANOVA) for the removal of amoxicillin and Ibuprofen. ANOVA encompasses a variety of statistical models and estimation processes that help identify differences between means. The ANOVA table facilitates conclusions about the overall significance of our model. The F values for the two responses, AMOX and IBU, indicate that the lack of fitness is not significant when compared to the pure error. The relationship of each individual factor with the response is assessed through the F ratios and p values. The model F values of 36.41 for amoxicillin and 39.45 for Ibuprofen demonstrate that the independent factors in the model have a significant effect on the dependent variable, specifically the removal percentage of amoxicillin and ibuprofen as Chemical Oxygen Demand (COD). This is the F value indicating the lack of fit of the model. A higher value suggests a greater likelihood that the model does not sufficiently fit the data.

The p-values below 0.05 in the ANOVA for AMOX and IBU removal suggest that the model terms are significant, and the opposite holds true as well. In this study, A, B, C, and A^2 were identified as significant factors affecting the AMOX response. Likewise, A, B, C, A^2, and B^2 were also determined to be significant model terms influencing the IBU response. The Adeq precision is used to measure the signal to noise ratio for the one response, and it is recommended to be greater than 4. For the removal of amoxicillin and ibuprofen the adeq precisions are 15.6984 and 19.7123, respectively. The summary of ANOVA

adequacy tests is given in Table 4.7.

$$\text{Removal of AMOX (\%)} = 1152.3 - 0.79A - 7.16B - 9.70C + 0.002AB - 0.002AC + 0.02BC + 0.01A^2 + 0.01B^2 + 0.15C^2 \quad (20)$$

$$\text{Removal of IBU (\%)} = 1089.99 + 0.96A + 6.86B + 1.40C + 0.004AB - 0.005AC - 0.007BC + 0.007A^2 + 0.01B^2 + 0.02C^2 \quad (21)$$

The normal plots of internally studentized residuals, as well as the actual and predicted values for the removal of amoxicillin and Ibuprofen, are displayed in Figures 4.8 b & e and a&f, respectively. The normal probability plot illustrates whether the residuals adhere to a normal distribution, indicated by their alignment along a straight line. Thus, the analysis confirms that the residuals follow a normal distribution. Additionally, the actual and predicted values exhibit a straight line with minimal deviation between them. This plot represents the residuals against the ascending predicted response values, testing the assumption of constant variance. Ideally, the plot should display a random scatter, with a consistent range of residuals throughout the graph.

Table 4. 7 Summary of fit statistics for the responses

Test Parameters	Removal of COD(AMOX)	Removal of COD(IBU)
Std. Dev.	1.84	1.21
Mean	65.53	84.29
C.V. %	2.80	1.43
R^2	0.9669	0.9807
Adjusted R^2	0.9244	0.9558
Predicted R^2	0.7375	0.7782
Adeq Precision	15.6984	19.7123

Table 4. 8 ANOVA for amoxicillin

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	691.10	9	76.79	22.74	0.0002	significant
A-Current density	520.03	1	520.03	154.00	< 0.0001	
B-Initial concentration of pharmaceuticals	26.28	1	26.28	7.78	0.0269	
C-pH	72.00	1	72.00	21.32	0.0024	
AB	1.0000	1	1.0000	0.2961	0.6032	
AC	0.0625	1	0.0625	0.0185	0.8956	
BC	3.06	1	3.06	0.9069	0.3726	
A^2	43.12	1	43.12	12.77	0.0091	
B^2	12.17	1	12.17	3.60	0.0995	
C^2	7.39	1	7.39	2.19	0.1825	
Residual	23.64	7	3.38			
Lack of Fit	10.44	3	3.48	1.05	0.4608	not significant
Pure Error	13.20	4	3.30			
Cor Total	714.74	16				

Table 4. 9 ANOVA for Ibuprofen

Source	Sum of Squares	df	Mean Square	F-value	p-value	Significance
Model	517.33	9	57.48	39.45	< 0.0001	significant
A-Current density	450.00	1	450.00	308.82	< 0.0001	
B-Initial IBU concentration	12.50	1	12.50	8.58	0.0221	
C-pH	18.00	1	18.00	12.35	0.0098	
AB	2.25	1	2.25	1.54	0.2540	
AC	0.2500	1	0.2500	0.1716	0.6911	
BC	0.2500	1	0.2500	0.1716	0.6911	
A^2	19.92	1	19.92	13.67	0.0077	
B^2	11.81	1	11.81	8.11	0.0248	
C^2	0.1289	1	0.1289	0.0885	0.7747	
Residual	10.20	7	1.46			
Lack of Fit	7.00	3	2.33	2.92	0.1639	not significant
Pure Error	3.20	4	0.8000			
Cor Total	527.53	16				

The predicted R^2 values of 0.9077 and 0.7782 for Amoxicillin and Ibuprofen, respectively indicates that the model equations can predict about the response. The R^2 and predicted R^2 values for an acceptable model should be close to 1 or generally above 0.7.

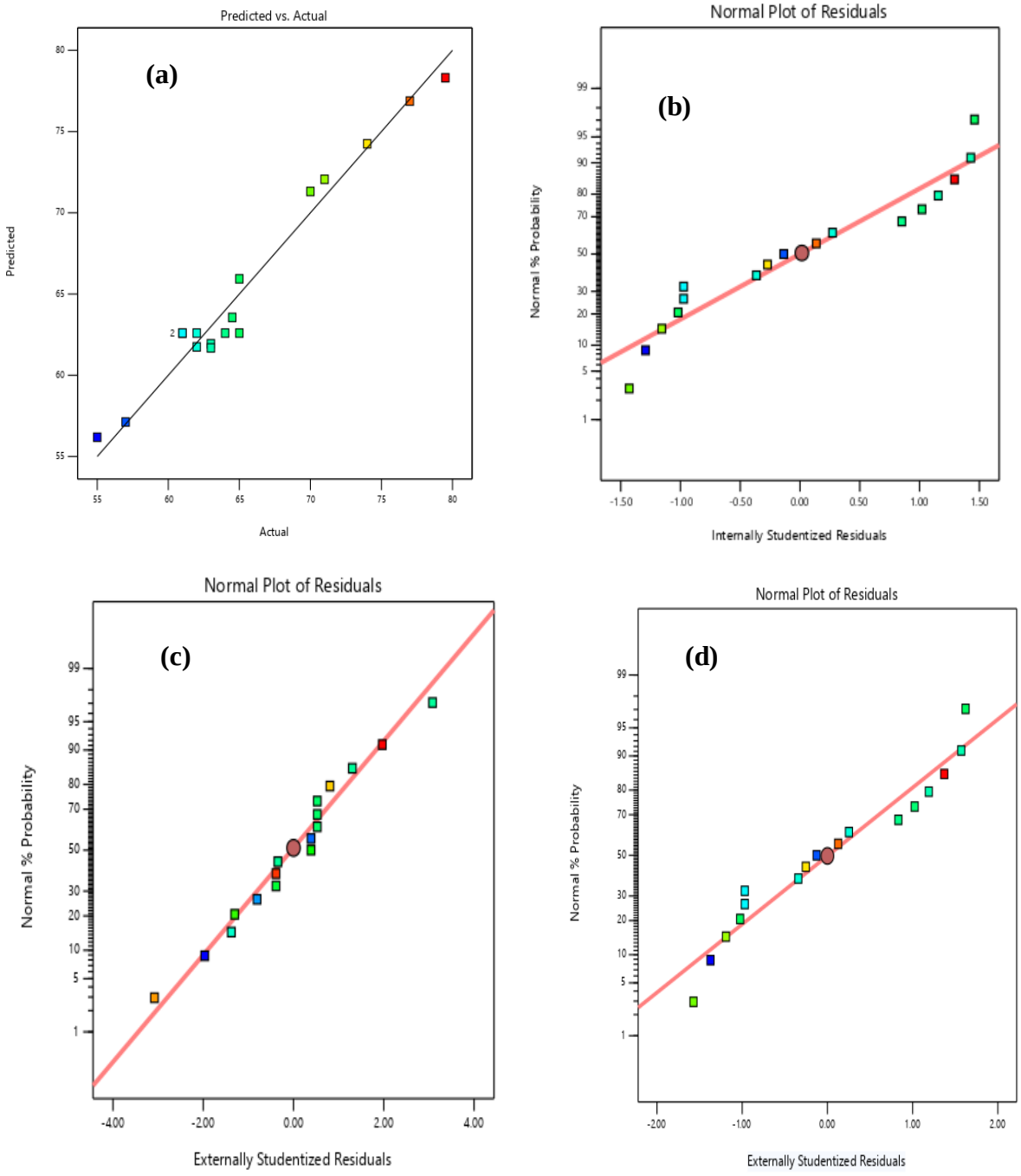
Tables 4.8 and 4.9 showed the Analysis of variance (ANOVA) results for removal of Amoxicillin and Ibuprofen. ANOVA is a set of statistical models and estimate processes used to scan differences between means. The ANOVA table allows us to make conclusions regarding the overall significance of our model.

Each individual factor's connection with the response is determined by the F ratios and p values. The model F values of 36.41 and 39.45 for Amoxicillin and Ibuprofen, respectively shows that the independent factors in the model have significant effect on the dependent variable, that is removal percentage of Amoxicillin and Ibuprofen as COD. The p values less than 0.05 in the ANOVA of AMOX and Ibuprofen removal indicate that the model terms are significant and vice versa. In this study, all the study factors have significant effect on the response. The adeq precision is used to measure the signal to noise ratio for the one response, and it is recommended to be greater than 4. For the removal Amoxicillin and Ibuprofen the adeq precisions are 15.6984 and 19.7123, respectively.

Table 4. 10 Experimental design and dependent variables for AMOX and IBU degradation by EO process

Run	Study factors			Response			
	A (mA/cm^2)	B (µg/L)	C (pH)	Removal of COD (AMOX) (%)		Removal of COD (IBU) (%)	
				Actual resp	Predicted resp	Actual resp	Predicted resp
1	27.5 (0)	276 (-1)	3 (-1)	70	69.8	86	82
2	45 (+1)	276 (-1)	6 (0)	77	75	94	91
3	45 (+1)	300 (+1)	6 (0)	74	75	92	93
4	27.5(0)	288 (0)	6 (0)	65	69	83	79
5	27.5(0)	300 (+1)	9 (+1)	63	56	82	85
6	10 (-1)	300 (+1)	6 (0)	57	63	77	72
7	27.5(0)	288 (0)	6 (0)	64	61	83	86
8	45 (+1)	288 (0)	3 (-1)	79.5	82	95	91
9	27.5(0)	288 (0)	6 (0)	62	54	82	86
10	27.5(0)	288 (0)	6 (0)	61	64	81	83
11	27.5(0)	300 (+1)	3 (-1)	65	61	85	82
12	10 (-1)	276 (-1)	6 (0)	62	61	82	78
13	10 (-1)	288 (0)	3 (-1)	63	68	78	73
14	27.5(0)	288 (0)	6 (0)	61	58	83	87
15	45 (+1)	288 (0)	9 (+1)	71	70	91	85
16	27.5(0)	276 (-1)	9 (+1)	64.5	67	84	87
17	10 (-1)	288 (0)	9 (+1)	55	57	75	70

The normal plot of internally studentized residuals for removal of Amoxicillin and Ibuprofen and the correlation between the actual and predicted values for removal of Amoxicillin and Ibuprofen is shown in Figure 4.8(a-f)



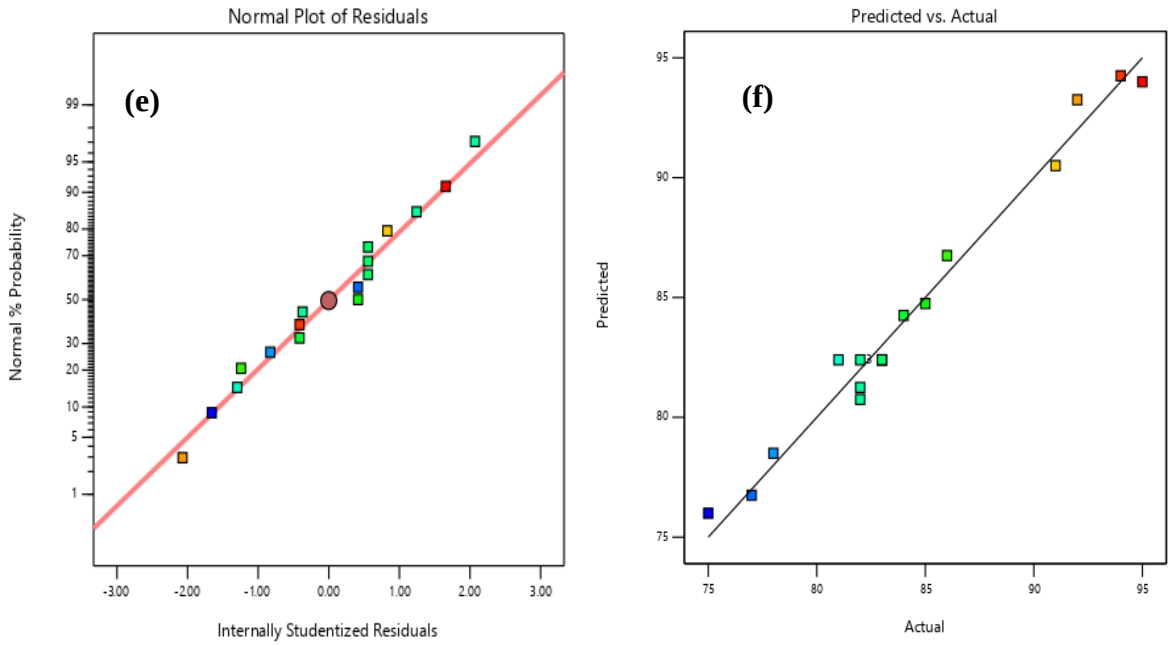


Figure 4.8 a) Predicted vs actual values for removal of Amoxicillin b) Normal plot of internally studentized residuals for removal of Amoxicillin c) Normal plot of externally studentized residuals for removal of Amoxicillin d) Normal plot of externally studentized residuals for removal of Ibuprofen e) Normal plot of internally studentized residuals for Ibuprofen f) Predicted vs actual values for removal of Ibuprofen

4.2.2. Effects of Parameters on Amoxicillin and Ibuprofen Removal by EO process

The response surface plots for the removal of Amoxicillin and Ibuprofen at different operational parameters are shown in Figure 4.9 and 4.10 (a-c). The maximum removal efficiency of Amoxicillin and Ibuprofen was attained at the same values current density of 45mAcm^{-2}, and initial concentration of 288 µgL^{-1} and at pH 3.

A. Current Density

Table 4.10 showed the actual and predicted responses as COD for the removal of Ibuprofen and Amoxicillin. For both Amoxicillin and Ibuprofen that increasing the current density from 10 to 27.5 and to 45 mAcm^{-2}. The lowest removal of COD for Ibuprofen was 75% which was achieved at 10mAcm^{-2} current density, 288 µgL^{-1} initial concentration and pH of 9. Which indicates at the

lower current density the lowest performance was attained. Also the lowest removal of Amoxicillin as COD was 55% achieved at the same values of current density, 288 μgL^{-1} initial concentration and pH of 9.

In both Amoxicillin and ibuprofen the lowest performance was achieved the lowest current density this is due to the fact that increasing current density increases the movement of electrons toward the electrodes so that direct oxidation occurs to produce reactive chlorine species (HClO, ClO^-) that degrade the pharmaceuticals. Even though current density increases if the pH is basic there are less reactive chlorine species formed. The maximum removal performance of ibuprofen and Amoxicillin was 95% and 82% respectively which was attained the same values of current density of 45$mAcm^{-2}$, 288 μgL^{-1} initial concentration and pH of 3. Increased current density at the same time acidic environment favors the formation of the reactive chlorine species thus enhancing the elimination of pharmaceuticals.

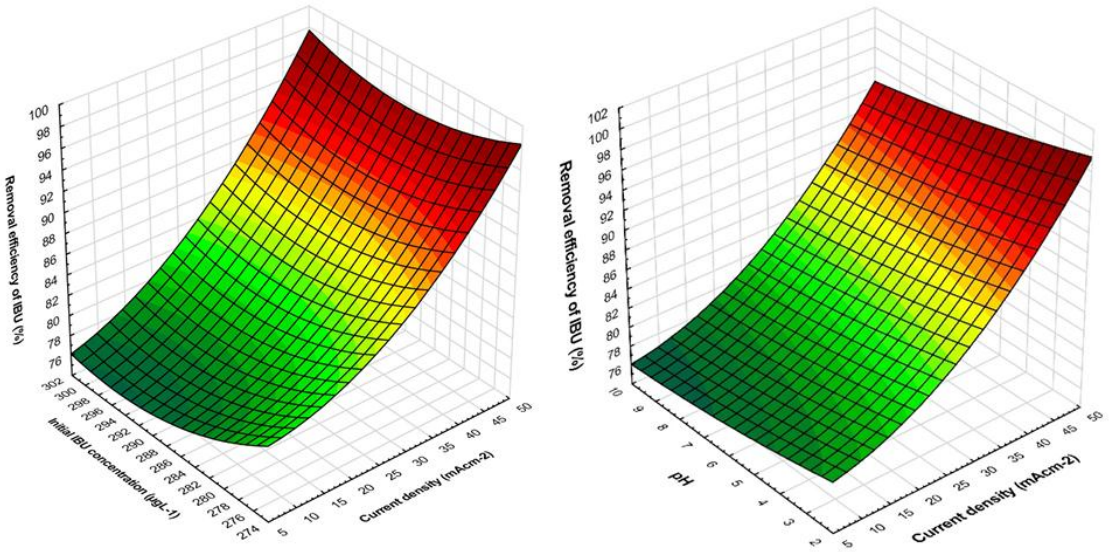
Increasing the current density within a certain range leads to increased formation of reactive species for instance hydroxyl radicals (·OH) and active chlorine species, which are important for oxidizing and degrading the pharmaceutical molecules. A study showed that the maximum Cephalexine COD removal (70.27%) using Ti/TiO_2-PbO_2 electrode is attained after prolonged electrolysis at a current density of 20 mA cm^{-2}[91]. Another study also confirms this result that pharmaceutical removal increases at increasing current density with a maximum removal attained at 1.8 mA cm^{-2} for the removal of Diclofenac, Carbamazepine and Amoxicillin using aluminum and stainless steel electrodes [92].

A study showed that H_2O_2 generation for electro fenton process was increased significantly by increasing the current density up to 60 $mAcm^{-2}$ using carbon felt cathode and boron-doped diamond (BDD) anode for the removal of chloroquine [93].

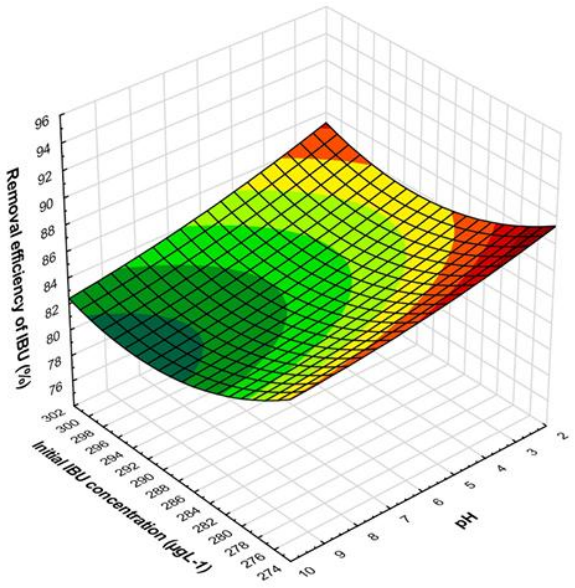
Also, increasing the current density stimulates the formation of oxidants at the electrode surfaces that leads to enhanced degradation of pharmaceuticals.

This is confirmed by a study which showed that acetaminophen was oxidized effectively at high current density and the activity of the anode plays a significant role on the selectivity and degradation efficiency resulting in electrochemical conversion and oxidation [94] [95]

Even though higher current densities bring about effective degradation, there is a possibility of leading to a negative effect if the amount of current density is very high. Very high current densities enhance the formation of side reactions intermediates, undesired products and consumes high energy [96] [97]. The effect of current density on the removal of Amoxicillin and Ibuprofen as COD is shown in the response surfaces as shown in the Figure 4.9 and 4.10 (a-c). The interaction of current density with initial concentration and pH is also shown in Figure 4.9 and 4.10 (a and b).

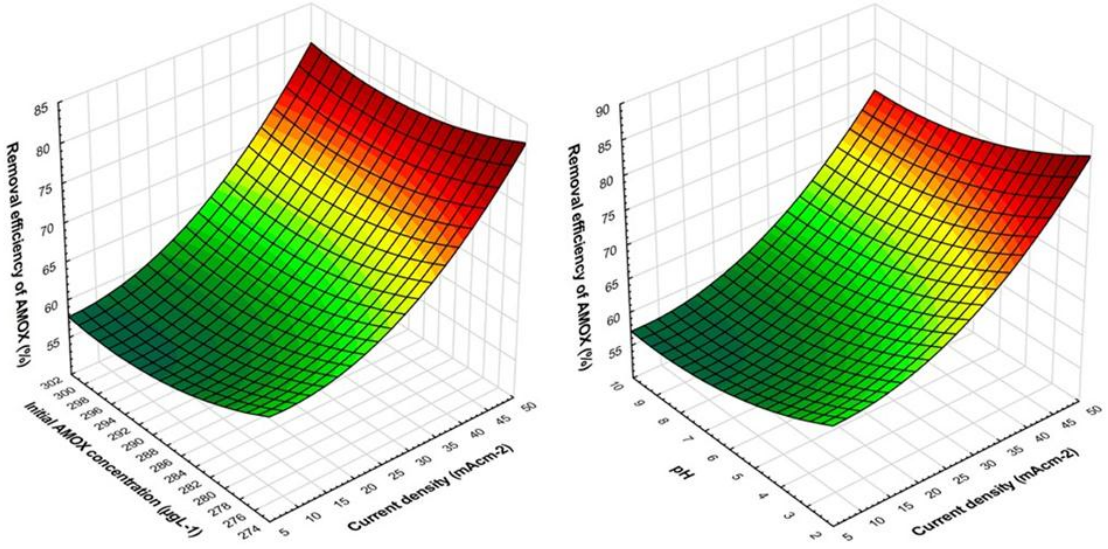


(a) **(b)**



(c)

Figure 4.9 Surface response plot parameter interaction effects of a) Effect of initial concentration and current density b) Effect of pH and current density c) Effect of initial concentration and pH on IBU removal.



(a) **(b)**

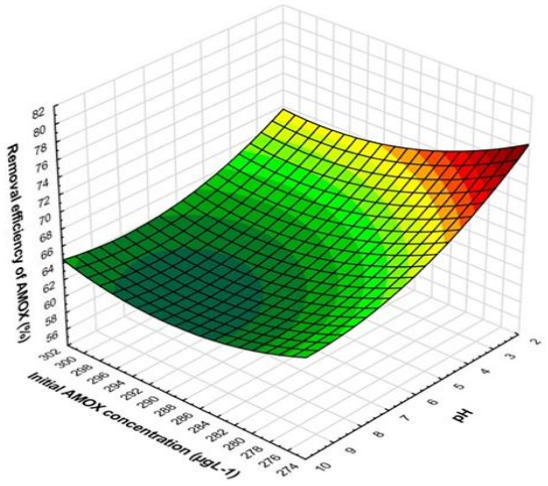


Figure 4.10 Surface response plot parameter interaction effects of a) Effect of initial concentration and current density b) Effect of pH and current density c) Effect of initial concentration and pH on AMOX removal.

B. pH

One of the main factors that affect the degradation of pharmaceuticals is pH. The formation of reactive chlorine species such as HClO, ClO^-, Cl_2 is pH dependent. The reactive species have different oxidation potentials that is Cl_2 and HClO have higher standard potentials (1.36 V and 1.49 V vs SHE, respectively) compared to ClO^- (0.89 V vs SHE), indicating they are more powerful oxidizing agents [98] [99].

Thus, the pH of the solution is highly dependent on the presence of the predominant active chlorine species. In acidic media (pH < 3), Cl_2 is the dominant active chlorine species and in slightly acidic media (pH = 3-8) HClO is the dominant active chlorine species in this pH range and hypochlorite ion (ClO^-) is the prevailing chlorine species at pH > 8. In this study it is shown that the maximum degradation efficiency of both AMOX and IBU was achieved at pH 3 that is 82% and 95%, respectively. Indicating the HClO is the dominant chlorine species in the bulk solution. While at pH 9 the lowest removal performance of AMOX and IBU was seen which agrees with the

prevalence of lowest oxidation potential hypochlorite (ClO^-) species at alkaline solution that is 55% and 75% for AMOX and IBU, respectively. At pH 6 the removal of AMOX and IBU is higher than at pH 9. For instance from table it shown that at 45 $mAcm^{-2}$ current density and pH 6 the removal efficiency of AMOX and IBU was 77% and 94% respectively while at the same current density removal performance for AMOX and IBU decreases to 71% and 91%,respectively at pH 9. Which concludes that by keeping the current density the same the variation in pH significantly affects the removal performance of electro-oxidation process. In this experiment, equation (6) Cl_2 reacts with water in the solution or evolved as gas. Cl_2 is found in different forms depending on the pH of the solution [100].

$$2Cl^- \rightarrow Cl_2 + 2e^- \quad (22)$$

$$Cl_2 + H_2O \rightarrow HClO + Cl^- + H_2 \quad (23)$$

$$HClO \rightarrow ClO^- + H^+ \quad (24)$$

The RSM analysis showed that the optimum pH is 3 meaning acid environmental is highly favored over neutral and alkaline medium in the degradation of both Amoxicillin and Ibuprofen. Studies have shown that pharmaceuticals like acetaminophen is faster in acidic surface water solutions compared to synthetic solutions, possibly due to the higher chloride concentration and enhanced oxidation of Cl^- ions at the anode [94]. Also a study showed that pH influences the degradation efficiencies of micro pollutants including pharmaceuticals [101].

C. Initial Concentration of AMOX and IBU

The other factor that affects degradation of pharmaceuticals using electro-oxidation is initial concentration of pharmaceuticals. Table 4.10 showed that even though initial concentration has effect on elimination of pharmaceuticals such as Amoxicillin and Ibuprofen, it is less than current density and pH, yet increasing initial concentration slightly diminishes the degradation efficiency. At the same current density and pH it is observable that decreasing initial concentration enhances performance of removal efficiency for both AMOX and IBU. The optimum initial concentration

for the removal of AMOX and IBU is 276μgL^{-1}. Meaning, 276μgL^{-1} requires less amount of current density to degrade efficiently than 288 μgL^{-1} and 300μgL^{-1}.

This is shown in Table 4.10 for instance at current density of 10mAcm^{-2}, initial concentration of 288 μgL^{-1} and pH of 9 the percentage removal of Amoxicillin and Ibuprofen is 55% and 75%,respectively. But increasing the current density to 45μgL^{-1} enhances the degradation efficiency of Amoxicillin and Ibuprofen to 71% and 91% respectively. To support this, a study indicated that initial concentration of a pharmaceutical compound impacts its removal efficiency during electro-oxidation, that increased current density and treatment time enhances the degradation efficiency diclofenac [102]. Many studies have shown that higher initial concentration of pharmaceuticals diminishes removal efficiency [103]

In another study it is confirmed that in electro-oxidation process when the concentration of Amoxicillin was 1mgL^{-1}, 5mgL^{-1},10 mgL^{-1} and 25mgL^{-1} the removal efficiencies were 98.7%, 99.1%, 94.3% and 79% respectively. Suggesting that AMOX elimination is more efficient at lower concentrations than at higher concentrations [104].

In this study, the decrease in treatment efficiency as initial concentration of AMOX and IBU increases is attributed to when pharmaceutical concentration increases the available active chlorine species is consumed to degrade the pharmaceutical and as well as in the formation and degradation of intermediates thus further increasing concentration plummets the degradation efficiency as there are no active chlorine species to degrade the pharmaceuticals. In the contrary at lower concentration the amount of reactive chlorine species is enough for pharmaceutical degradation and intermediate removal.

4.2.3 The Interaction effect of UV Irradiation Intensity, and Irradiation time on the removal of AMOX and IBU by UV Photolysis process

UV photolysis has been one of the most widely studied advanced chemical transformation processes for the destruction of micropollutants such as pharmaceuticals. Mostly UV photolysis has been used as pretreatment before the biological treatment which minimizes the

pharmaceuticals concentration for the efficacy of biological process [105]. The UV photolysis functions by breaking down the molecules of pharmaceutical chemical bonds bringing about their degradation, transformation and potentially reducing their toxicity [106]. Study shows that a UV photolysis degrades many antibiotics efficiently, such as fluorine substituents (like florfenicol and ofloxacin) and β-lactam antibiotics (like cefalexin, Amoxicillin, and ampicillin)[105]

In this study a UVC (254nm) was used for it is promising to eliminate and mineralize pharmaceutical compounds and UVC is effective over many compounds degradation [107]. In this study UV photolysis was used to degrade AMOX and IBU. Figure 19 showed by varying the UV Irradiation intensity ($mAcm^{-2}$) and irradiation time (min) the degradation efficiency of AMOX and IBU was observed. Studies showed that UV intensity and irradiation time directly affects degradation efficiency of pharmaceuticals. When UV intensity increase there are more photons bombarded on the pharmaceutical molecules that brings about fast degradation rate [108].

Also extending the irradiation exposure time gives prolonged time for the photochemical reactions to happen so that there will be more degradation [106].

For this UV photolysis experiment the concentration of pharmaceutical (276 µg/L) was taken from the optimum values from EO optimization experiment. Also the two UV Intensity are UV1 ($5.47mWcm^{-2}$), and UV2 ($7.82mWcm^{-2}$). Also the exposure times were taken from 0-60minutes

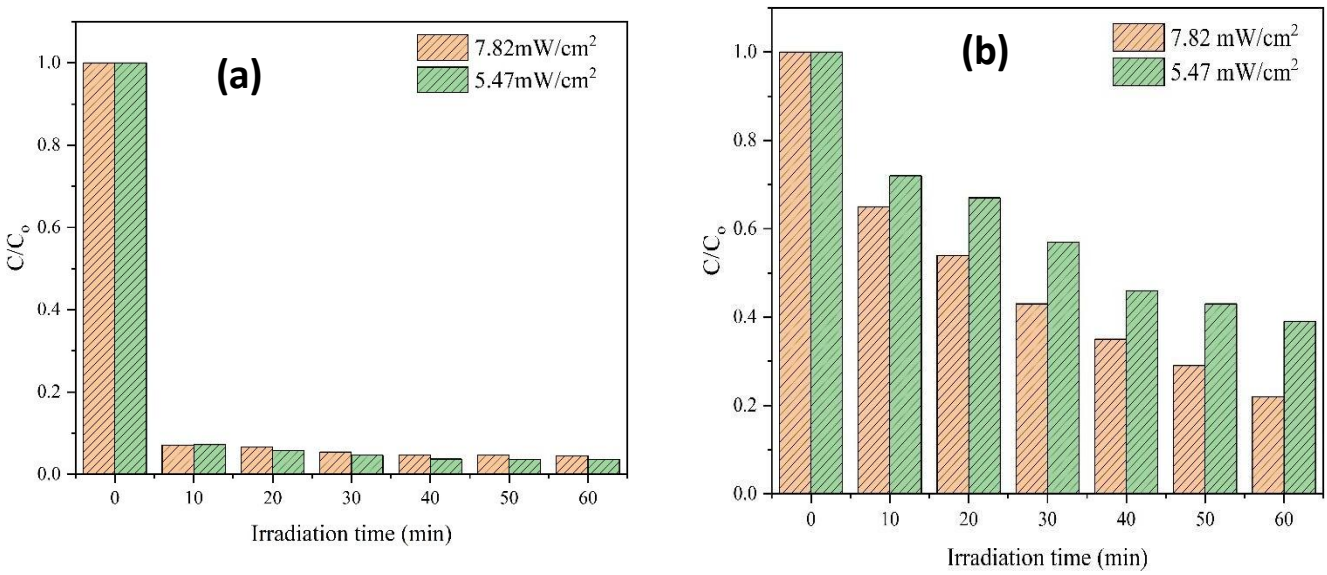


Figure 4.11 Degradation of a) IBU and b) AMOX degradation by UV irradiation

In some cases, the addition of oxidants (like hydrogen peroxide) can enhance the degradation of pharmaceuticals and reduce the toxicity of transformation products [109] [110]

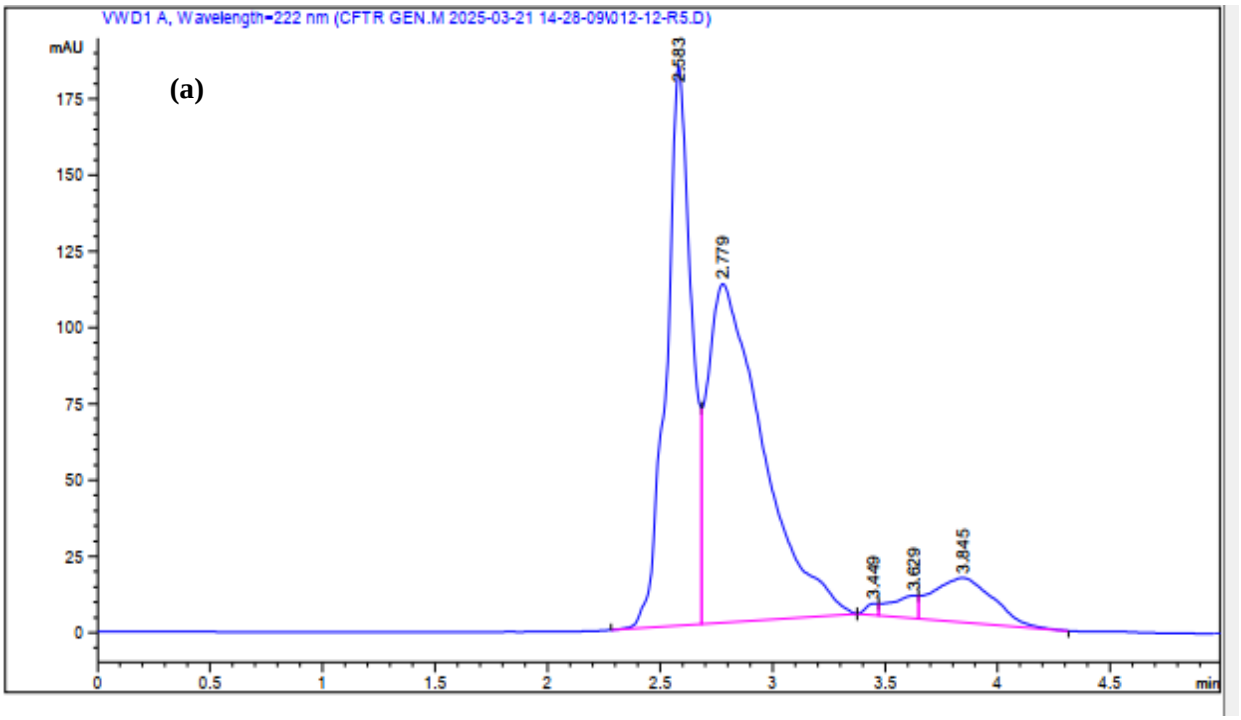
4.2.4. Combination of Electro-oxidation and UV Irradiation (EO-UV) for the elimination and mineralization of Amoxicillin and Ibuprofen

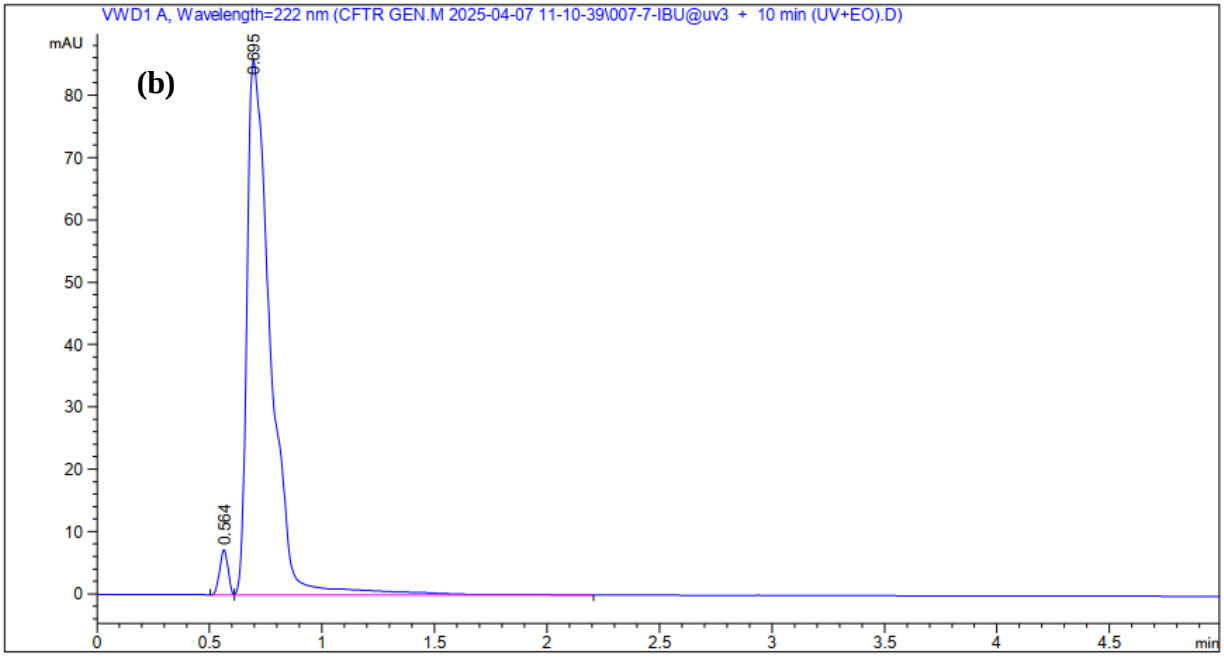
Though EO has good performance in degrading AMOX and IBU as shown in Table 15, it has challenges for the fact that it produces undesirable side products that has higher toxicity than the parent compounds. Since the oxidant used in this study is active chlorine species the formed side products are chlorine derivatives. Also, using EO the energy consumption for the degradation of AMOX and IBU enhanced as current density and electrolysis time increased. Thus, to enhance the degradation rate of AMOX and IBU, to reduce the toxicity (by minimizing the formed intermediates) and to decrease the energy consumption (per g COD removed) combining with the most efficient method is mandatory.

In this study EO was combined with UV photolysis for the elimination and mineralization of AMOX and IBU. The optimum values from EO were used to integrate it with UV with higher irradiation intensity ($7.82 mWcm^{-2}$). Table 15 showed the maximum degradation of COD for

Amoxicillin and Ibuprofen is 79.5% and 95%, respectively. Here IBU is more degraded than Amoxicillin by EO process. But by combining EO with UV as the chromatogram graph showed the COD removal for Amoxicillin reached approximately 94% with the same electrolysis time combining with UV increased the efficiency by approximately by 15%. However, for IBU combining the EO with UV has reduced from the COD removal of 95% to approximately 93%. This may be related to the chemical structure of IBU unlike to AMOX that is sensitive to UV light.

The chromatogram in Figure 4.12(a) for IBU indicated that when EO was used the intermediate compounds' peaks were observed. However when EO was combined with UV (Figure 4.12(b)) most of the intermediates' peaks vanished which is an indication of IBU mineralization. Also for Amoxicillin as shown in Figure 4.12 (c) and (d) even though the intermediate peaks are not observed in the chromatogram it is certain that the AMOX concentration significantly reduced when seeing the area of the chromatogram when EO combined with UV.


VWD1 A, Wavelength=222 nm (CFTR GEN.M 2025-03-21 14-28-09\012-12-R5.D)
mAU
(a)
2.583
2.779
3.449
3.629
3.845
min


VWD1 A, Wavelength=222 nm (CFTR GEN.M 2025-04-07 11-10-39\007-7-IBU@uv3 + 10 min (UV+EO).D)
mAU
(b)
0.695
0.564
min

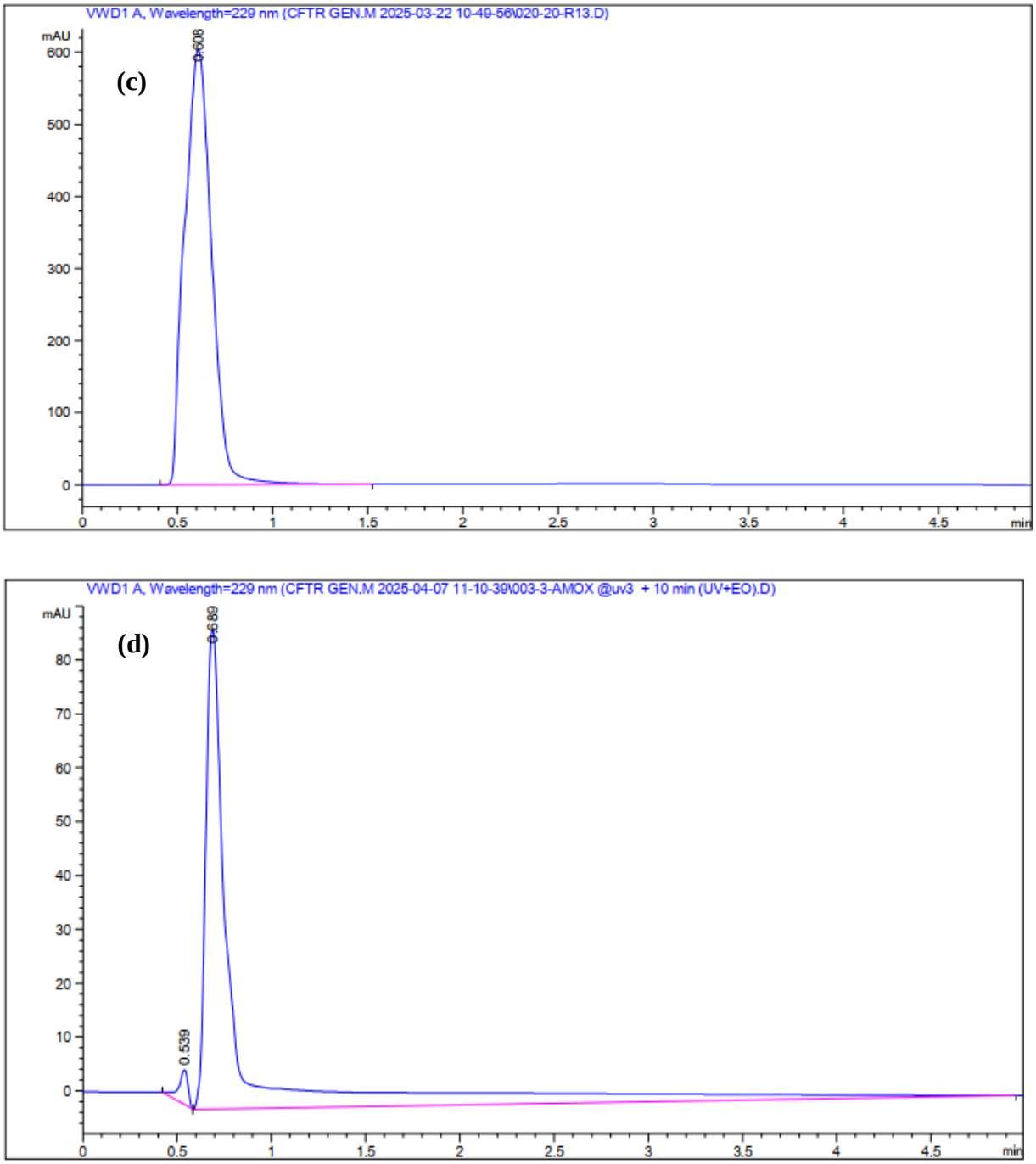


Figure 4.12 a) HPLC chromatogram for intermediates formed during removal of Ibuprofen by EO process b) Chromatogram of Ibuprofen after combining with EO with UV c) Chromatogram of AMOX after treatment by EO d) Chromatogram of AMOX after treatment using EO-UV

4.2.5 Identification of Amoxicillin and Ibuprofen reaction intermediates, mechanism and reaction pathways

When degrading and mineralizing pollutants using electro-oxidation process there are a number of reactions taking place at the anode surface and in the bulk solution that leads to the formation of undesirable intermediates products. Using IrO_2/Ti electrodes degradation of Amoxicillin and Ibuprofen formed intermediate compounds. Consequently, the degradation pathways for AMOX and IBU was proposed. The retention time for AMOX was 0.640 min. But after EO of AMOX for 25 min the retention time was 0.608 because of loss of amine group and opening of β-lactam ring of AMOX molecule.

The proposed degradation routes of AMOX by EO process using Ti/IrO_2 anodes are chlorination and hydroxylation. In both the physisorbed reactive chlorine species (Cl_2, HOCl, $\cdot OCl^-$) are the main player of the degradation and mineralization of AMOX. Even though the generation of .OH radicals is less by IrO_2/Ti, it has its own contribution in the degradation of AMOX.

A3 path shows the direct anodic oxidation of AMOX at the anode surface that forms carbon dioxide, water and oxidation products. There are two proposed pathways for AMOX degradation that is A1 and A2. In the A1 path, the β-lactam ring which is responsible for AMOX antibacterial nature breaks as a result of attack by Cl^- and even if no radical scavenging experiments done a possible formation of .OH radical because of water oxidation.

The (propane1,2,3, triol) or three carbon atom , eight hydrogen atoms and three oxygen atoms are removed. The A2 path is attacked primarily by active chlorine and as a result ammonia and carboxylic acid are removed. Then by C1 path by addition of active chlorine the benzene ring with oxygen eliminated. Also adding hydroxyl radical (using C3 path) dehalogenation and it mineralization occurs. B1 showed the elimination of carbonate by adding active chlorine and hydroxyl radical. Then by C4 path it mineralized by the addition of hydroxyl radical. While the B2 path demonstrates the elimination of the chlorinated benzene ring. This is attained by the addition of hydroxyl radical. Then after dehalogenation (elimination of chlorine) by adding .OH

in C2 path the compound is mineralized to carbon dioxide and water [111]. Figure 4.13 and Figure 4.14 showed the degradation reaction pathways for AMOX and IBU, respectively

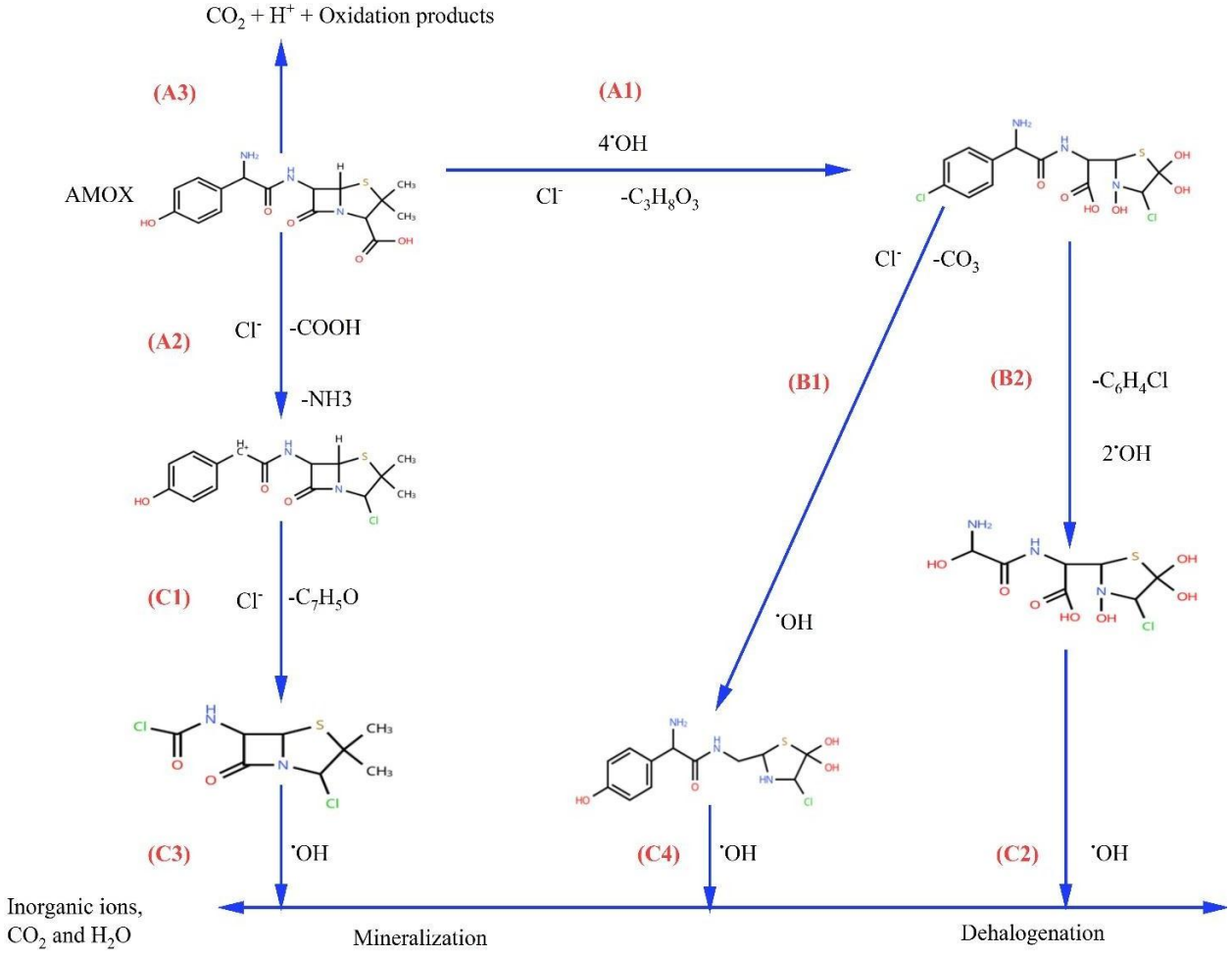


Figure 4.13 Proposed degradation pathways for AMOX by EO treatment using Ti/IrO_2 electrodes

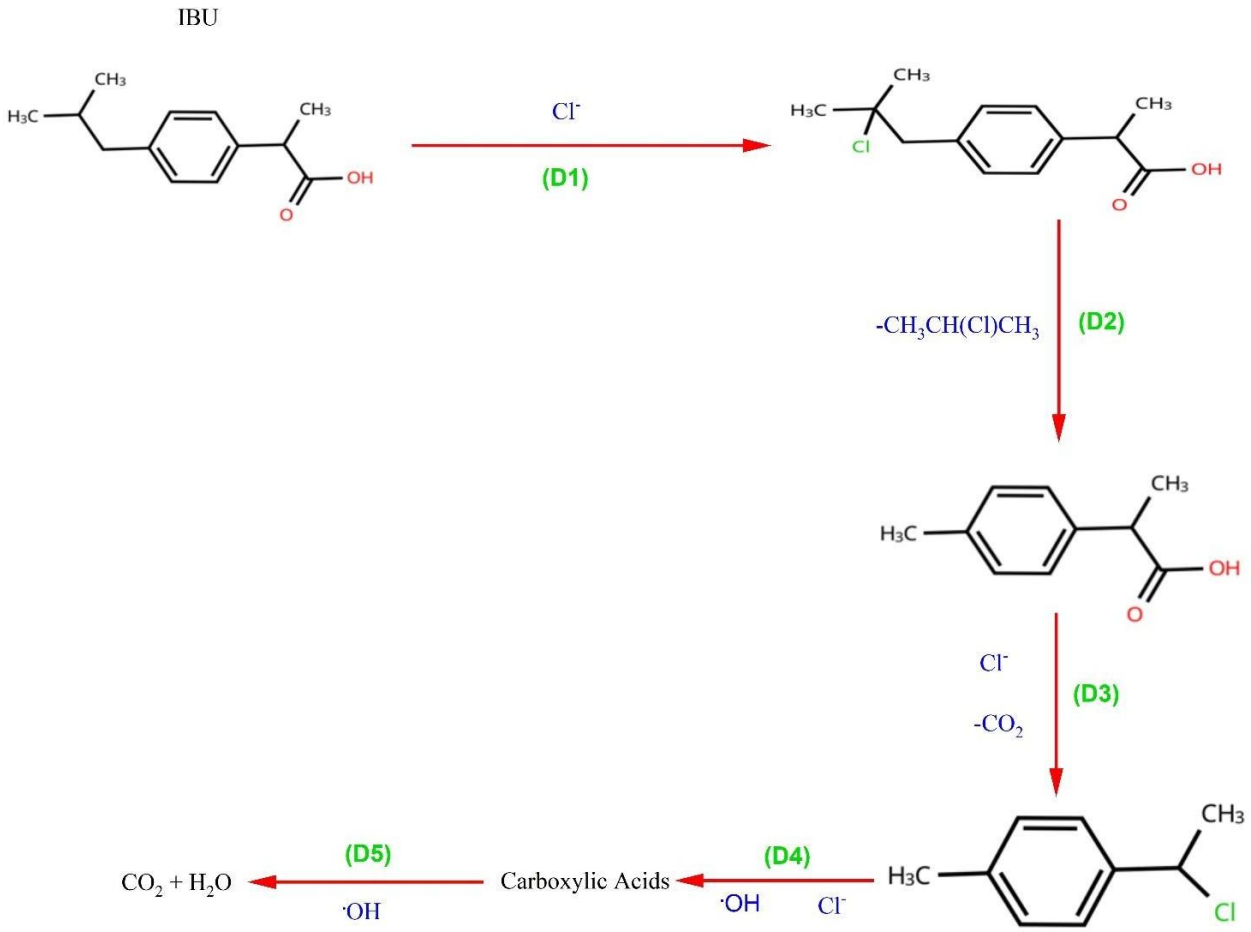


Figure 4.14 Proposed degradation pathway for IBU using EO process by Ti/IrO_2 electrodes

By using Ti/IrO_2 electrodes and NaCl electrolyte in electrooxidation process IBU degrades in two ways first through direct oxidation via electron transfer at the electrode surface and second through indirect oxidation via the generated reactive chlorine species. Yet this process produces intermediates that are more toxic than the parent compound. The following figure depicts the degradation pathways and the main intermediates pop up during the electrooxidation process. In direct oxidation, the IBU adsorbs onto the anode surface to transfer electrons, the anode (Unlike to BDD anode) Ti/IrO_2 limits the formation of hydroxyl (.OH) radical. Thus, the direct oxidation plays a secondary role in the degradation of IBU.

In the indirect oxidation NaCl dissociates into Cl^- ions that is oxidized t the anode surface to from active chlorine species (Cl_2,HClO, ClO^-). In general the electrooxidation IBU involves decarboxylation (loss of the -COOH group), hydroxylation (addition of -OH groups), Cleavage of the alkyl chain breakdown aromatic ring opening (formation of smaller organic acids).

Considering the detected intermediates the possible degradation pathway was proposed. During EO IrO_2/Ti produces small amount of hydroxyl radical and generates more active chlorine species, the degradation paths use chlorine species and hydroxyl radical. The produced chlorine undergoes substitution with hydrogen atom in the D1 pathway. Then a 2-chlorobutane is eliminated by D2 path. In D3 the presence of chlorine and elimination of carbondioxide. Then a chlorinated benzene ringed compound formed. The presence of both chlorine and hydroxyl radical produced various aliphatic carboxylic acids. These acids could be finally mineralized to CO_2 and H_2O via successive hydroxylation reactions [112].

4.2.6 TOC removal analysis

Various studies showed that EO only process achieves lower TOC removal for both Amoxicillin and Ibuprofen. A study indicated that TOC removal for Amoxicillin was not more than 40% [113]. Another study also supports this using Electrooxidation TOC removal of 37% to 46% were achieved for Amoxicillin. But increasing treatment time besides its side reactions formation and current densities might increase the TOC values [114]. However, a study demonstrated that using UV photolysis the TOC values for Amoxicillin were between 48 to 71% [115]. The reason for lower removal of Amoxicillin using EO than UV is due to the incomplete mineralization (partially oxidize) which is as a result of the formation of more reactive intermediates in EO which still contribute to increase the TOC. Also the available reactive oxidant species are low in the beginning of the treatment process compared to the formed intermediates during EO than UV.

Unlike to Amoxicillin with a more complex structure consisting of a β-lactam ring and additional functional groups, which are more resistant to oxidative cleavage and can hamper complete mineralization Ibuprofen has relatively simple aromatic structure with a single benzene ring and a

carboxylic acid group, making it more susceptible to attack reactive chlorine species and hydroxyl radical generated during electrooxidation [116][117]. Unlike EO, study showed UV photolysis for Amoxicillin resulted in low TOC removal (less than 5%). For Ibuprofen a UV based photocatalytic process remove TOC up to 78% [118][119][120].

The degree of mineralization of treated synthetic wastewater by EO was analyzed interms of the Total Organic Carbon (TOC) removal. TOC removal efficiency was computed using optimum initial concentration of pharmaceuticals was used as the initial concentration for TOC (276 μgL^{-1}).

In this study at 10, 20, 30, 40, 50 and 60 minutes the final TOC was determined. In the same manner using EO-UV the TOC removal was evaluated. The results indicated that EO-UV performed well for the removal of TOC more than EO only process. This is due to the fact that UV generates reactive chlorine species (e.g., -Cl, -ClO) and hydroxyl radicals (-OH) that complement electrochemical oxidation, leading to faster degradation of AMOX and IBU. Besides, the mineralization is faster in EO-UV since intermediates resistant to EO can be degraded by UV and vice versa. Studies showed that combining UV and EO enhances the mineralization (TOC removal) by 30-40% than EO alone. Figure showed that increasing the treatment time enhances the removal of TOC for IBU and AMOX.

IBU is degraded fast in a given time than AMOX using a combination of EO and UV processes. A study showed that complete elimination of ibuprofen has been achieved through electrolysis, attributed to the generation of hydroxyl radicals in the bulk via the Fenton reaction, either through in situ electro-generated Fenton's reagent or at the anode surface resulting from water oxidation [121]. Also another study supports this findings that the remarkable capacity for high mineralization efficiency exhibited by the electrochemical oxidation process renders it a compelling alternative for the remediation of Ibuprofen in aqueous solutions [112]. Using EO-UV process the TOC at 10 minutes for AMOX was 4.16 ppm and at 30 minutes of treatment the TOC for AMOX increased to 6.81 ppm.

Figure depicts the increase, decrease and finally increase trend of TOC removal for AMOX. Initially AMOX molecules are oxidized fast that bring about TOC reduction as the parent compound broken down into smaller molecules by the UV and into oxidized intermediates by the reactive chlorine and oxygen species [122]. The decrease in TOC removal of AMOX after 10 minute is as the oxidation process goes on there is buildup of intermediates that increase the TOC for instance carboxylic acids, aromatic compounds, trihalomethanes and other partially oxidized species. Most of these species are resistant to oxidation which slows down the TOC elimination rate [123]. After 30 minutes an exponential increase in TOC removal for AMOX was observed this is due to the fact that these intermediates are beginning to be mineralized since several reactive oxidative species from EO and UV are generated in the solution [124].

There are causes for the increase in TOC for AMOX when treatment time increased. This is due to the formation of transient intermediates before complete oxidation to CO_2 or here might be interference of chlorinated intermediates with the detection of TOC.

In this study DBP formation was not studied. However, previous studies showed that the application of active chlorine can cause the formation intermediate products. Amoxicillin is a comparatively large and intricate molecule, and it does not exhibit the same level of volatility or solubility as certain smaller organic compounds [125].

A study support this finding that large pharmaceutical molecules such as AMOX may not be fully detected as TOC in their original form due to their low volatility or solubility [126]. A study confirmed that combining EO with UV or persulfate significantly improved the TOC removal. For Amoxicillin, coupling EO with persulfate and UV attained 97% removal efficiency of TOC which outwork the single methods [127]. Also another study demonstrated that integrating UV with hydrogen peroxide or photocatalytic process increased the TOC level for both Amoxicillin and Ibuprofen that reach 50-85% under optimal conditions [128][129].

Studies have showed that further increasing the treatment time diminishes the TOC. Figure 4.15 showed that for IBU at 10 and 30 minutes of treatment, the TOC decreased from 6.07 ppm to 4.86

ppm, respectively. Further, increasing the treatment time to 40, 50 and 60 minutes the TOC level decreased significantly for both AMOX and IBU. Figure 4.15 demonstrated that further increasing the treatment time almost completely eliminated the TOC for both drugs and achieve mineralization within short period of time.

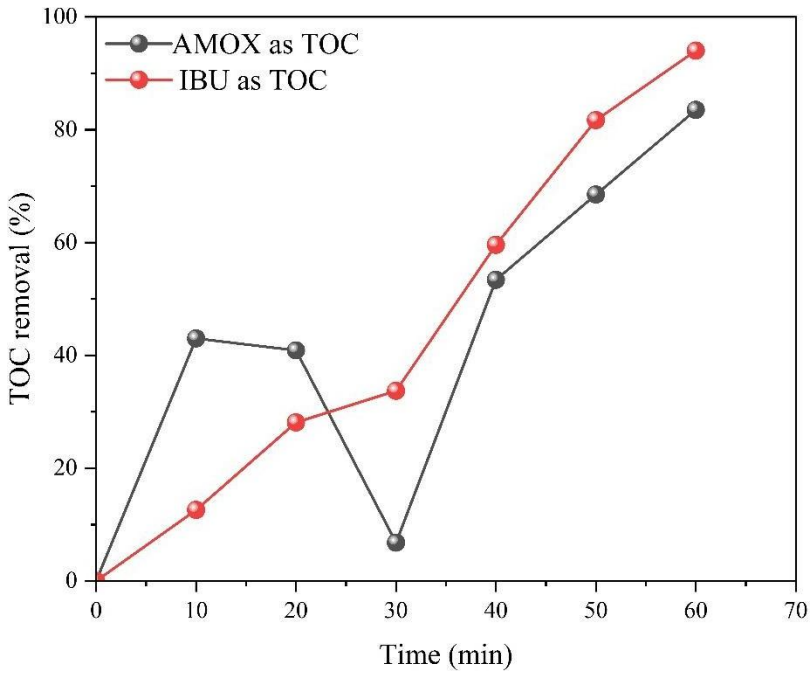


Figure 4.15 TOC removals for AMOX and IBU using EO-UV process

The proposed study on coupling electro-oxidation and UV was also compared with various literature reports based on different techniques involving several pharmaceutical and personal care products (Table 4.11). This study demonstrated comparable results in terms of degradation efficiency and mineralization.

Table 4. 11 Performance comparison of the present study with literature under different model pollutants

Type of AOPs	**Pollutant**	**Treatment efficiencies and conditions**	**Ref**
Electrochemical oxidation	Amoxicillin	99% within 3 h	[104]
Photo-based advanced oxidation processes	Ibuprofen	82% was higher than the effects of ultraviolet/hydrogen peroxide oxidation	[130]
Electro-oxidation	Amoxicillin	51.64% COD and 37.82% TOC removal was achieved	[123]
UV Direct Photolysis and UV/H_2O_2	Triclosan and Ibuprofen	UV/H_2O_2 97.39% COD and 47.27% COD removal in direct UV photolysis	[131]
Electrocatalytic oxidation	Amoxicillin	The optimum current density and initial pH were found to be 5.88 mA cm2 and 7.0, at which, 60% COD and 48% TOC were achieved in 60 and 240 min of electrolysis, respectively	[123]
UV/UV H_2O_2	Ibuprofen	Ibuprofen degraded by 8% and 3% under UV or H_2O_2 only Ibuprofen degraded by 78% within 4 min in UV/H_2O_2 process	[132]
Electrochemical mineralization	Ibuprofen	Ibuprofen mineralization (91.6%)	[133]

Ozone under ultraviolet irradiation	Caffeine (CAF), Atenolol (ATL), Carbamazepine (CBZ), Sulfamethoxazole (SMX), Trimethoprim (TMP), and Acetaminophen (ACT)	95 % with ozone only at 20 min 3 %–11 % with UV light alone 97 % removal with ozone-UV combined process within 7 min	[134]
Electrochemical degradation	Ibuprofen	COD removals between 60 and 95% and TOC removals varying from 48 to 92%, in 6 h	[135]
Electro-oxidation with boron doped diamond	Diclofenac	Mineralization degree of 72% after a 4 h	[136]
Low Pressure UV Photolysis (UVC)	Acetaminophen, Atenolol, Bezafibrate, Diclofenac and Ibuprofen	32% to 99% removal efficiencies after 60 min,	[137]
UVC and VUV photolysis	Diclofenac, Erythromycin, Ibuprofen and Phenacetin	Mineralization of PhCs (<10% TOC removal at 30 min reaction) with UVC photolysis 90% mineralization of PhCs with VUV photolysis at 60 min.	[107]
Electro-oxidation on metal-oxide-coated Ti anodes	Ibuprofen	93.2% of TOC removal after 60 minutes	[112]
UV photolysis	Florfenicol, Ofloxacin, Cefalexin, Amoxicillin and Ampicillin	Florfenicol =100% Ofloxacin >90 Cefalexin =100% Amoxicillin >90% and Ampicillin >90% at UV intensity of 5.6 mW/cm²	[105]

Electro-oxidation using Titanium Electrodes	Amoxicillin	99.23 % removal efficiency (81.13% COD removal) at 15 mA for 6 h.	[138]
UV Irradiation Using Hydrogen Peroxide	Amoxicillin	Complete degradation using 3 mL of H_2O_2 under UV irradiation (50 mW/cm²)	[139]
UV/H_2O_2 system	Ibuprofen	97% Ibuprofen degradation	[140]
Photochemical mineralization (by means of UV, hydrogen peroxide, titanium dioxide and iron)	Ibuprofen	81% of TOC removal at 39.2 mmol/L of H_2O_2 over 120 minutes	[141]
Coupling Electro-oxidation and UV	Amoxicillin and Ibuprofen	94% of COD removal (83.3% of TOC removal) for Amoxicillin 93% of COD removal (94% of TOC removal) for Ibuprofen	This study

4.2.7. Toxicity Assessment

In the electro-oxidation process, the generation of intermediate products requires an assessment of the potential environmental hazards linked to their existence.
During electrooxidation, pharmaceuticals are not instantly mineralized; instead, they are transformed into various intermediate products, some of which may possess higher toxicity than the original substance. This research evaluated the toxicity of AMOX and IBU, along with their intermediate products, by analyzing the growth rate or viability of E. coli cultures.

The results indicated that prolonged electrolysis results in further breakdown of these intermediates, leading to a decrease in overall toxicity, as demonstrated in Figures 4.16-4.19 through E. coli inhibition. Furthermore, it was noted that the use of electrooxidation alone produces more toxic intermediates and suppresses E. coli growth due to the generation of harmful chlorinated by-products from the NaCl electrolyte utilized in the electrooxidation of AMOX and IBU.
In contrast, when electrooxidation is paired with UV treatment, the toxicity is reduced, as UV-induced reactions promote the degradation of chlorinated compounds by activating chlorine species and further decomposing intermediates. Overall, the toxicity at the onset of the electrooxidation process is elevated; however, as the duration of electrooxidation extends, the toxicity diminishes. This decrease is attributed to the lack of reactive oxidants essential for the formation of intermediates.

In the electrochemical oxidation (EO) process, disinfection by-products (DBPs) are generated due to the application of active chlorine, and the incorporation of ultraviolet (UV) light transforms these DBPs into less harmful compounds. UV light alters reactive species from chlorine-centric pathways (in EO) to hydroxyl radical (.OH) driven oxidation, thereby diminishing the formation of DBPs [142].

EO processes that utilize chlorine-based electrolytes, such as Ti/RuO_2 -IrO_2 anodes, yield free chlorine, which has the potential to create detrimental DBPs like trihalomethanes. The application of UV irradiation facilitates the breakdown of residual chlorine and the conversion of DBPs into less toxic by-products. The Inhibition Ratio (IR) represents the percentage decrease in E. coli growth attributable to the toxicity of intermediates. The suggested safety limits for the IR are defined as (-10% < IR < 10%), which indicates the toxicological impact of the sample.

Utilizing EO at each dilution ratio of 25%, 50%, and 75%, the inhibition ratios for both AMOX and IBU fall outside the safe limits. For AMOX, at the dilution levels of 25%, 50%, and 75%, the colony counts were reduced to 31, 19, and 4 CFU/ml, respectively. The percentages of colonies for the corresponding dilutions (25%, 50%, and 75%) are 16.2%, 24%, and 60%, which exceed the 10% threshold and are therefore considered unsafe.

The integrated EO-UV process was evaluated for 25 %, 50 %, and 75 % dilutions, with the inhibition ratios for the treatment of IBU and AMOX analyzed as illustrated in Figures 25 and 27. Prior to treatment, the colony counts for the 25 % and 50 % dilutions of the AMOX sample were recorded at 37 and 25 CFU/ml, respectively (Table 4.12). However, following treatment with EO-UV, a reduction in colony counts of 5.4 % and 8 % was observed for the 25 % and 50 % dilution ratios, respectively, decreasing the colony count from 37 to 35 CFU/ml for the 25 % dilution and from 25 to 23 CFU/ml for the 50 % dilution (Figures 4.16 and 4.18, (a),(b), and (c)). This indicates that the inhibition ratio for AMOX treatment using EO-UV at these dilution ratios remained within the safety boundary of -10 % < IR < 10 %. Additionally, for the 75 % dilution, the inhibition ratio for AMOX also increased post-treatment with EO-UV, as the colony count decreased from 10 to 7 CFU/ml, representing a 30 % reduction in colony count, which exceeds the safety boundary. This phenomenon can be attributed to the higher concentration of the sample likely generating more toxic intermediates that inhibit E. coli growth. The control samples with IBU and AMOX exhibited no E. coli growth in both scenarios. In contrast to AMOX, the toxicity assessment revealed that IBU produces fewer toxic intermediates that affect the E. coli colony. For the 25 %, 50 %, and 75 % dilutions, the reduction in colony counts due to intermediates was 4

%, 6.7 %, and 11.1 %, respectively. The elevated 75 % dilution indicated a slight exceedance of the safe boundary. This suggests that after EO-UV treatment, the toxicity of IBU is nearly negligible across all three dilutions when compared to AMOX. In summary, the combination of EO with UV effectively reduced the toxicity associated with intermediates from AMOX and IBU relative to EO alone, maintaining the inhibition ratio within the safety boundaries of < 10 % during the initial 25 minutes for IBU and 35 minutes for AMOX. This reduction in toxicity is likely due to the formation of short-chain carboxylic acids, such as acetic acid and oxalic acid, when EO is combined with UV, which exhibit lower toxicity to E. coli growth.

Table 4. 12 The number of bacterial colony in CFU/mL before and after EO-UV treatment

S.No	Pollutants	% Dilution of pollutants	Before	After EO only	% Colony difference/ (EO only)	After/ (EO-UV)	% Colony difference/ (EO-UV)
1.	AMOX	25%	37	31	16.2	35	5.4
		50%	25	19	24	23	8
		75%	10	4	60	7	30
2.	IBU	25%	25	20	20	24	4
		50%	15	11	26.7	14	6.7
		75%	9	4	55.6	8	11

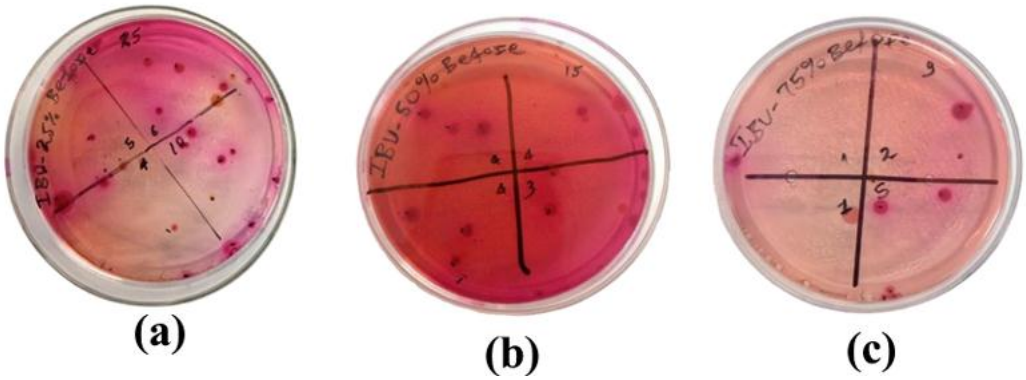


Figure 4.16 E.coli colonies for IBU before treatment at different dilutions (a) 25 %, (b) 50 % and (c) 75 %.

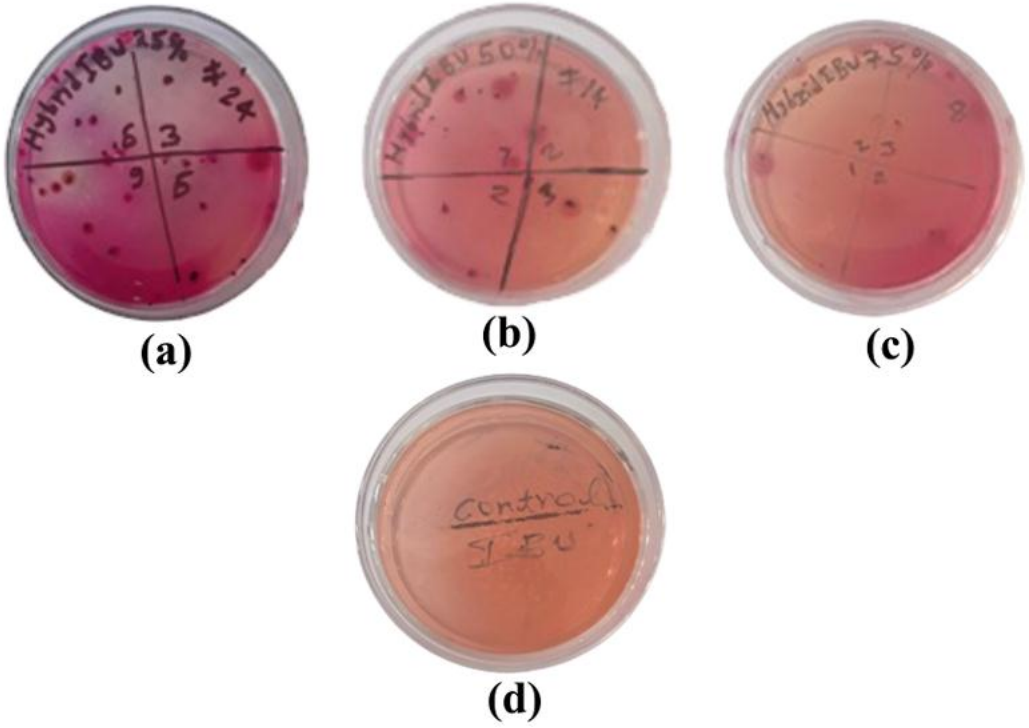


Figure 4.17 E.coli colonies for IBU EO-UV treatment at different dilutions

(a) 25 %, (b) 50 %, (c) 75 % and (d) control

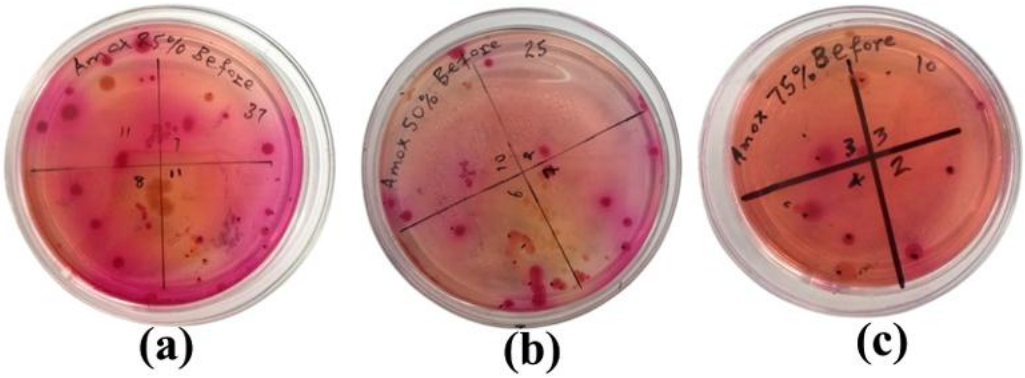


Figure 4.18 E.coli colonies for AMOX before treatment at different dilutions (a) 25 %, (b) 50 % and (c) 75%.

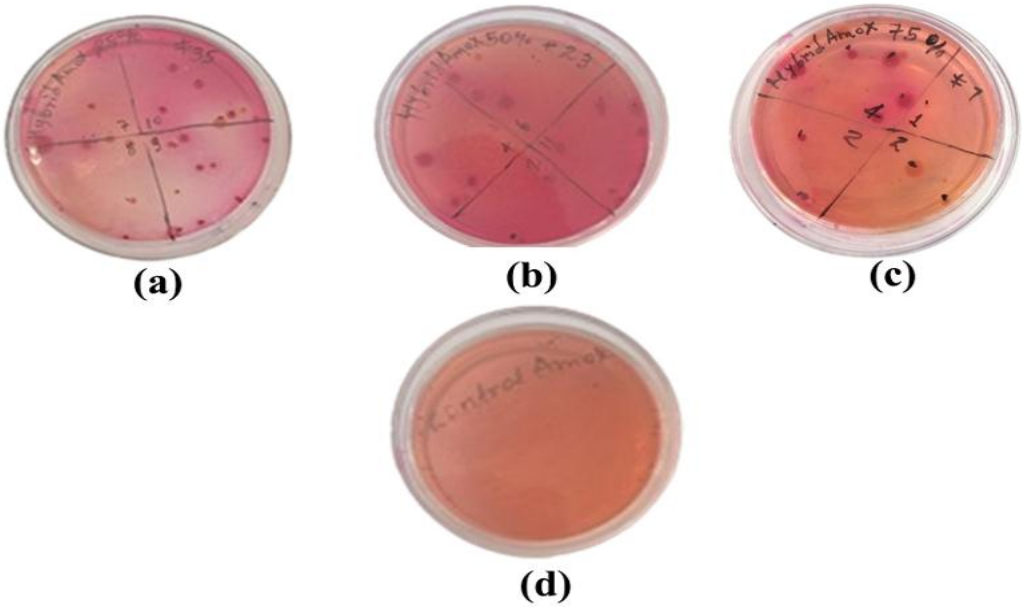


Figure 4.19 E.coli colonies for AMOX after EO-UV treatment at different dilutions (a) 25 %, (b) 50 %, (c) 75 % and (d) control

Consequently, the combination of EO and UV not only reduces the degradation time required for Total Organic Carbon (TOC) removal and improves mineralization but also eliminates toxic intermediates, transforming them into less harmful compounds. Research supports this assertion, indicating that the integration of EO and UV irradiation enhances the breakdown of pharmaceuticals, resulting in quicker mineralization and a more rapid reduction of harmful intermediates compared to the EO-only process. For example, one of the common intermediates produced during the oxidation of pharmaceuticals is oxalic acid; when subjected to photo-assisted EO, its concentration was significantly lower (2.5 mgC/L) post-treatment compared to the EO-only treatment (20.5 mgC/L), demonstrating a more effective removal of toxic intermediates. Table 4.12 presents a summary of the percentage differences between the samples before and after treatment with EO alone and EO-UV. Overall, as shown in the table, with the exception of the 75% dilution, all samples displayed less than 10% inhibition relative to the control (prior to treatment), which is regarded as non-toxic or having a negligible effect.

Chapter Five

Conclusion and Recommendations

5.1. Conclusion

The results of this study indicate that electro-oxidation (EO) is an effective method for generating active chlorine species from saline solutions, which can be applied in disinfection and pollutant degradation processes. The optimization of operational conditions-particularly electric potential, NaCl concentration, and electrolysis time-plays a central role in maximizing residual chlorine production while managing energy demand. The relatively minor influence of electrode spacing suggests that reactor performance is more strongly governed by electrochemical operating parameters than by physical configuration within the tested range.

This study further demonstrates that combining EO with UV irradiation improves the degradation and mineralization of pharmaceuticals such as amoxicillin (AMOX) and ibuprofen (IBU) compared to EO alone. This improvement is associated with enhanced breakdown of intermediate compounds, which contributes to higher total organic carbon (TOC) removal. However, the degree of improvement varies between compounds, indicating that treatment efficiency depends on the chemical structure and reactivity of the target pollutants.

In addition, the observed reduction in toxicity when EO is coupled with UV suggests that the combined process may reduce the persistence of intermediate transformation products compared to EO alone. This highlights the importance of integrating complementary advanced oxidation processes to improve treatment completeness rather than relying on a single oxidation pathway.

Finally, the comparison of modeling approaches indicates that artificial neural networks (ANNs) provide a more accurate representation of process behavior than response surface methodology (RSM) for the conditions studied.

5.2. Recommendation and future works

Considering the very promising results obtained in this thesis, the following recommendations are suggested for future works:

- This study used direct UV photolysis not solar light. Since it is relatively higher in cost and consumes energy future research could focus to use solar energy along with electro-oxidation for pharmaceutical removal.
- Under take kinetic study based on the intermediate formed to elucidate the mechanism of the degradation.
- Also besides to DSA electrodes such as IrO_2/Ti which are expensive future research could check efficiencies of other anode types which are less costly and available in developing countries like Ethiopia. Additionally, it would be beneficial to compare alternative electrodes for cost-toxicity trade-offs.
- Only Anti-inflammatory and antibiotic drugs (Ibuprofen and Amoxicillin) are considered for this study but the method effectiveness was not checked for other therapeutic class of drugs
- Also, future work should include testing the proposed system on hospital or industrial wastewater with higher pharmaceutical loads.

References

[1] García-Espinoza, J. D., & Nacheva, P. M. (2019). Degradation of pharmaceutical compounds in water by oxygenated electrochemical oxidation: Parametric optimization, kinetic studies and toxicity assessment. *Science of the Total Environment, 691*, 417–429. https://doi.org/10.1016/j.scitotenv.2019.07.118

[2] Lin, A. Y. C., & Tsai, Y. T. (2009). Occurrence of pharmaceuticals in Taiwan's surface waters: Impact of waste streams from hospitals and pharmaceutical production facilities. *Science of the Total Environment, 407*(12), 3793–3802. https://doi.org/10.1016/j.scitotenv.2009.03.009

[3]. Serwecińska, L. (2020). Antimicrobials and antibiotic-resistant bacteria. *Water, 12*(12), 3313. https://doi.org/10.3390/w12123313

[4] Nashat, M., Mossad, M., El-Etriby, H. K., & Gar Alalm, M. (2022). Optimization of electrochemical activation of persulfate by BDD electrodes for rapid removal of sulfamethazine. *Chemosphere, 286*(Part 1), 131579. https://doi.org/10.1016/j.chemosphere.2021.131579

[5] Ekpeghere, K. I., Lee, J. W., Kim, H. Y., Shin, S. K., & Oh, J. E. (2017). Determination and characterization of pharmaceuticals in sludge from municipal and livestock wastewater treatment plants. Chemosphere, 168, 1211–1221. https://doi.org/10.1016/j.chemosphere.2016.10.077

[6] Murgolo, S., Franz, S., Arab, H., Bestetti, M., Falletta, E., & Mascolo, G. (2019). Degradation of emerging organic pollutants in wastewater effluents by electrochemical photocatalysis on nanostructured TiO_2 meshes. *Water Research, 164*, 114920. https://doi.org/10.1016/j.watres.2019.114920

[7] Wang, J., Zhi, D., Zhou, H., He, X., & Zhang, D. (2018). Evaluating tetracycline degradation pathway and intermediate toxicity during the electrochemical oxidation over a Ti/Ti_4 O_7

anode. *Water Research, 137*, 324–334. https://doi.org/10.1016/j.watres.2018.03.030

[8] González, T., Domínguez, J. R., Palo, P., & Sánchez-Martín, J. (2011). Conductive-diamond electrochemical advanced oxidation of naproxen in aqueous solution: Optimizing the process. *Journal of Chemical Technology & Biotechnology, 86*(1), 121–127. https://doi.org/10.1002/jctb.2494

[9] Kharel, S., et al. (2021). Removal of pharmaceutical metabolites in wastewater ozonation including their fate in different post-treatments. *Science of the Total Environment, 759*, 143989. https://doi.org/10.1016/j.scitotenv.2020.143989

[10] Sopaj, F., Rodrigo, M. A., Oturan, N., Podvorica, F. I., Pinson, J., & Oturan, M. A. (2015). Influence of the anode materials on the electrochemical oxidation efficiency: Application to oxidative degradation of the pharmaceutical amoxicillin. *Chemical Engineering Journal, 262*, 286–294. https://doi.org/10.1016/j.cej.2014.09.100

[11] Buthiyappan, A., Abdul Aziz, A. R., & Wan Daud, W. M. A. (2015). Degradation performance and cost implication of UV-integrated advanced oxidation processes for wastewater treatments. *Reviews in Chemical Engineering, 31*(3), 263–302. https://doi.org/10.1515/revce-2014-0039

[12] Ameta, R., Chohadia, A. K., Jain, A., & Punjabi, P. B. (2018). *Fenton and photo-Fenton processes*. https://doi.org/10.1016/B978-0-12-810499-6.00003-6

[13] González Peña, O. I., López Zavala, M. Á., & Cabral Ruelas, H. (2021). Pharmaceuticals market, consumption trends and disease incidence are not driving the pharmaceutical research on water and wastewater. International Journal of Environmental Research and Public Health, 18*(5), 2532.* https://doi.org/10.3390/ijerph18052532

[14] Brito, N., Medeiros, D., Araújo, D., Martínez-Huitle, C. A., & Andrés, M. (2015). Understanding active chlorine species production using boron doped diamond films with lower and higher sp^3/sp^2 ratio. *Electrochemistry Communications, 55*, 34–38.

https://doi.org/10.1016/j.elecom.2015.03.013

[15] García-Espinoza, J. D. (2019). Synergistic effect between ultraviolet irradiation and electrochemical oxidation for removal of humic acids and pharmaceuticals. *Water Environment Journal, 0*, 1–15. https://doi.org/10.1111/wej.12456

[16] Yimam, S. A., Kang, J. W., & Kassahun, S. K. (2024). Applications of integrated response surface methodology statistic techniques and artificial neural network-based machine learning to optimize residual chlorine production and energy consumption. *Water Environment Journal, 38*(3), 373–384. https://doi.org/10.1111/wej.12922

[17] Hurwitz, G., Pornwongthong, P., Mahendra, S., & Hoek, E. M. V. (2014). Degradation of phenol by synergistic chlorine-enhanced photo-assisted electrochemical oxidation. *Chemical Engineering Journal, 240*, 235–243. https://doi.org/10.1016/j.cej.2013.11.087

[18] Zaviska, F., Drogui, P., & Pablo, G. (2012). Statistical optimization of active chlorine production from a synthetic saline effluent by electrolysis. *Desalination, 296*, 16–23. https://doi.org/10.1016/j.desal.2012.03.02

[19] Wang, Y., Xue, Y., & Zhang, C. (2020). Generation and application of reactive chlorine species by electrochemical process combined with UV irradiation: Synergistic mechanism for enhanced degradation performance. *Science of the Total Environment, 712*, 136501. https://doi.org/10.1016/j.scitotenv.2020.136501

[20] Park, H., Vecitis, C. D., & Hoffmann, M. R. (2009). Electrochemical water splitting coupled with organic compound oxidation: The role of active chlorine species. *Environmental Science & Technology, 43*(20), 7935–7945. https://doi.org/10.1021/es901573n

[21] Chlorine, A. (2018). *Indirect electrochemical oxidation using hydroxyl*. Elsevier Inc. https://doi.org/10.1016/B978-0-12-813160-2.00006-7

[22] Bergmann, H., & Koparal, S. (2005). The formation of chlorine dioxide in the electrochemical treatment of drinking water for disinfection. *Electrochimica Acta, 50*(25–26), 5218–5228. https://doi.org/10.1016/j.electacta.2005.01.061

[23] Chianca, D., et al. (2014). Active chlorine species electrogenerated on $Ti/Ru_{0.3}Ti_{0.7}O_2$ surface: Electrochemical behavior, concentration determination and their application. *Journal of Electroanalytical Chemistry, 731*, 145–152. https://doi.org/10.1016/j.jelechem.2014.08.008

[24] Kraft, A. (2008). Electrochemical water disinfection: A short review. *Platinum Metals Review, 52*(3), 177–185. https://doi.org/10.1595/147106708X329273

[25] Roslan, N. N., et al. (2024). Recent advances in advanced oxidation processes for degrading pharmaceuticals in wastewater: A review. *Catalysts, 14*(3), 1–25. https://doi.org/10.3390/catal14030189

[26] Li, W. C. (2014). Occurrence, sources, and fate of pharmaceuticals in aquatic environment and soil. *Environmental Pollution, 187*, 193–201. https://doi.org/10.1016/j.envpol.2014.01.015

[27] Rodríguez-Serin, H., et al. (2022). Literature review: Evaluation of drug removal techniques in municipal and hospital wastewater. *International Journal of Environmental Research and Public Health, 19*(20), 13105. https://doi.org/10.3390/ijerph192013105

[28] Arslan-Alaton, I., & Caglayan, A. E. (2005). Ozonation of Procaine Penicillin G formulation effluent Part I: Process optimization and kinetics. *Chemosphere, 59*(1), 31–39. https://doi.org/10.1016/j.chemosphere.2004.10.014

[29] Loganathan, P., Vigneswaran, S., Kandasamy, J., Cuprys, A. K., Maletskyi, Z., & Ratnaweera, H. (2023). Treatment trends and combined methods in removing pharmaceuticals and personal care products from wastewater: A review. *Membranes, 13*(2), 158. https://doi.org/10.3390/membranes13020158

[30] Guo, Y., Qi, P. S., & Liu, Y. Z. (2017). A review on advanced treatment of pharmaceutical wastewater. *IOP Conference Series: Earth and Environmental Science, 63*(1), 012025. https://doi.org/10.1088/1755-1315/63/1/012025

[31] Murray, K. E., Thomas, S. M., & Bodour, A. A. (2010). Prioritizing research for trace pollutants and emerging contaminants in the freshwater environment. *Environmental Pollution, 158*(12), 3462–3471. https://doi.org/10.1016/j.envpol.2010.08.009

[32] Alfonso-Muniozguren, P., Serna-Galvis, E. A., Bussemaker, M., Torres-Palma, R. A., & Lee, J. (2021). A review on pharmaceuticals removal from waters by single and combined biological, membrane filtration and ultrasound systems. *Ultrasonics Sonochemistry, 76*, 105656. https://doi.org/10.1016/j.ultsonch.2021.105656

[33] Oluwole, A. O., Omotola, E. O., & Olatunji, O. S. (2020). Pharmaceuticals and personal care products in water and wastewater: A review of treatment processes and use of photocatalyst immobilized on functionalized carbon in AOP degradation. *BMC Chemistry, 14*(1), 1–29. https://doi.org/10.1186/s13065-020-00714-1

[34] Lozano, I., Cervantes-Aviles, P., Keller, A., & López Aguilar, C. (2023). Removal of pharmaceuticals and personal care products from wastewater via anodic oxidation and electro-Fenton processes: Current status and needs regarding their application. *Water Science and Technology, 88*(5), 1143–1154. https://doi.org/10.2166/wst.2023.266

[35] Sim, W., Lee, J., & Oh, J. (2010). Occurrence and fate of pharmaceuticals in wastewater treatment plants and rivers in Korea. *Environmental Pollution, 158*(5), 1938–1947. https://doi.org/10.1016/j.envpol.2009.10.03

[36] Rivera-Utrilla, J., Sánchez-Polo, M., Ferro-García, M. Á., Prados-Joya, G., & Ocampo-Pérez, R. (2013). Pharmaceuticals as emerging contaminants and their removal from water: A review. *Chemosphere, 93*(7), 1268–1287. https://doi.org/10.1016/j.chemosphere.2013.07.059

[37] Gracia-Lor, E., Sancho, J. V., Serrano, R., & Hernández, F. (2012). Occurrence and removal of pharmaceuticals in wastewater treatment plants at the Spanish Mediterranean area of Valencia. *Chemosphere, 87*(5), 453–462. https://doi.org/10.1016/j.chemosphere.2011.12.025

[38] Drewes, E., Grassmann, J., Stadlmair, L. F., & Letzel, T. (2018). Enzymes in removal of pharmaceuticals from wastewater: A critical review of challenges, applications and screening methods for their selection. *Chemosphere, 205*, 649–661. https://doi.org/10.1016/j.chemosphere.2018.04.142

[39] Radjenović, J., Petrović, M., & Barceló, D. (2007). Analysis of pharmaceuticals in wastewater and removal using a membrane bioreactor. *Analytical and Bioanalytical Chemistry, 387*(4), 1365–1377. https://doi.org/10.1007/s00216-006-0883-6

[40] Ruziwa, D. T., et al. (2023). Pharmaceuticals in wastewater and their photocatalytic degradation using nano-enabled photocatalysts. *Journal of Water Process Engineering, 54*, 103880. https://doi.org/10.1016/j.jwpe.2023.103880

[41] Gros, M., Petrović, M., Ginebreda, A., & Barceló, D. (2010). Removal of pharmaceuticals during wastewater treatment and environmental risk assessment using hazard indexes. *Environment International, 36*, 15–26. https://doi.org/10.1016/j.envint.2009.09.002

[42] Kumar, S., Woo, H., Oh, J., & Park, H. (2011). Occurrence and removal of antibiotics, hormones and several other pharmaceuticals in wastewater treatment plants of the largest industrial city of Korea. *Science of the Total Environment, 409*(20), 4351–4360. https://doi.org/10.1016/j.scitotenv.2011.07.015

[43] Roberts, P. H., & Thomas, K. V. (2006). The occurrence of selected pharmaceuticals in wastewater effluent and surface waters of the lower Tyne catchment. *Science of the Total Environment, 356*, 143–153. https://doi.org/10.1016/j.scitotenv.2005.04.031

[44] Kimura, A., Osawa, M., & Taguchi, M. (2012). Decomposition of persistent

pharmaceuticals in wastewater by ionizing radiation. *Radiation Physics and Chemistry*, *81*(9), 1508–1512. https://doi.org/10.1016/j.radphyschem.2011.11.032

[45] Kompare, B., Zupanc, M., Kosjek, T., Petkovšek, M., Heath, E., & Širok, B. (2013). Removal of pharmaceuticals from wastewater by biological processes, hydrodynamic cavitation and UV treatment. *Ultrasonics Sonochemistry*, *20*, 1104–1112. https://doi.org/10.1016/j.ultsonch.2012.12.003

[46] Zaugg, S. D., Buxton, H. T., Furlong, E. T., Esposito, K., & Stinson, B. (2010). Pharmaceutical formulation facilities as sources of opioids and other pharmaceuticals to wastewater treatment plant effluents. *Environmental Science & Technology*, *44*(13), 4910–4916.

[47] Fick, J., Söderström, H., Lindberg, R. H., Phan, C., Tysklind, M., & Larsson, D. G. J. (2009). Pharmaceuticals and personal care products in the environment: Contamination of surface, ground, and drinking water from pharmaceutical production. *Environmental Toxicology and Chemistry*, *28*(12), 2522–2527

[48] Simazaki, D., Fujiwara, J., Manabe, S., Matsuda, M., Asami, M., & Kunikane, S. (2008). Removal of selected pharmaceuticals by chlorination, coagulation-sedimentation and powdered activated carbon treatment. *Water Science and Technology*, *58*(5), 1129–1135. https://doi.org/10.2166/wst.2008.472

[49] Sedlak, D. L. (2004). *Transformation of aromatic ether- and amine-containing pharmaceuticals during chlorine disinfection*. Environmental Science & Technology, **38**(14), 4019–4025. https://doi.org/10.1021/es035368l

[50] MacCrehan, W. A. (2006). *Transformation of acetaminophen by chlorination produces the toxicant N-acetyl-p-benzoquinone imine*. Environmental Science & Technology, **40**(2), 516–522. https://doi.org/10.1021/es0509073

[51] Matsuo, H., Sakamoto, H., Arizono, K., & Shinohara, R. (2011). Behavior of

pharmaceuticals in wastewater treatment plant in Japan. *Bulletin of Environmental Contamination and Toxicology, 87,* 31–35. https://doi.org/10.1007/s00128-011-0299-7

[52] Matesun, J., Petrik, L., Musvoto, E., Ayinde, W., & Ikumi, D. (2024). Limitations of wastewater treatment plants in removing trace anthropogenic biomarkers and future directions: A review. *Ecotoxicology and Environmental Safety, 281*, 116610. https://doi.org/10.1016/j.ecoenv.2024.116610

[53] Kümmerer, K. (2009). The presence of pharmaceuticals in the environment due to human use: Present knowledge and future challenges. *Journal of Environmental Management, 90*(8), 2354–2366. https://doi.org/10.1016/j.jenvman.2009.01.023

[54] Bijlsma, L., et al. (2021). Investigation of pharmaceuticals in a conventional wastewater treatment plant: Removal efficiency, seasonal variation and impact of a nearby hospital. *Journal of Environmental Chemical Engineering, 9*(4), 105548. https://doi.org/10.1016/j.jece.2021.105548

[55] Fraiha, O., et al. (2024). Comprehensive review on the adsorption of pharmaceutical products from wastewater by clay materials. *Desalination and Water Treatment, 317*, 100114. https://doi.org/10.1016/j.dwt.2024.100114

[56] Bhatti, S., Godheja, J., Panda, S., & Haq, I. (2023). Treatment of pharmaceutical pollutants from industrial wastewater. In *[Book title not specified]*. https://doi.org/10.1016/B978-0-323-91902-9.00007-9

[57] Li, Z., Schulz, L., Ackley, C., & Fenske, N. (2010). Adsorption of tetracycline on kaolinite with pH-dependent surface charges. *Journal of Colloid and Interface Science, 351*(1), 254–260. https://doi.org/10.1016/j.jcis.2010.07.034

[58] Chang, P. H., et al. (2009). Sorptive removal of tetracycline from water by palygorskite. *Journal of Hazardous Materials, 165*(1–3), 148–155. https://doi.org/10.1016/j.jhazmat.2008.09.113

[59] Aydin, S., Celik Karakaya, M., Karakaya, N., & Aydin, M. E. (2023). Effective removal of selected pharmaceuticals from sewerage treatment plant effluent using natural clay (Na-montmorillonite). *Applied Water Science, 13*(6), 1–14. https://doi.org/10.1007/s13201-023-01930-5

[60] De Andrade, J. R., Oliveira, M. F., Da Silva, M. G. C., & Vieira, M. G. A. (2018). Adsorption of pharmaceuticals from water and wastewater using nonconventional low-cost materials: A review. *Industrial & Engineering Chemistry Research, 57*(9), 3103–3127. https://doi.org/10.1021/acs.iecr.7b05137

[61] Asheghmoalla, M., & Mehrvar, M. (2024). Adsorption of a multicomponent pharmaceutical wastewater on charcoal-based activated carbon: Equilibrium and kinetics. *Water (Switzerland), 16*(15). https://doi.org/10.3390/w16152086

[62] Satyam, S., & Patra, S. (2024). Innovations and challenges in adsorption-based wastewater remediation: A comprehensive review. *Heliyon, 10*(9), e29573. https://doi.org/10.1016/j.heliyon.2024.e29573

[63] Von Sonntag, C. (2008). Advanced oxidation processes: Mechanistic aspects. *Water Science and Technology, 58*(5), 1015–1021. https://doi.org/10.2166/wst.2008.467

[64] Sirés, I., & Brillas, E. (2012). Remediation of water pollution caused by pharmaceutical residues based on electrochemical separation and degradation technologies: A review. *Environment International, 40*, 212–229. https://doi.org/10.1016/j.envint.2011.07.012

[65] Comninellis, C., Jüttner, K., Galla, U., & Schmieder, H. (1994). Electrocatalysis in the electrochemical conversion/combustion of organic pollutants for wastewater treatment. *Electrochimica Acta, 39*(11–12), 1857–1862. https://doi.org/10.1016/0013-4686(94)85175-1

[66] Marselli, B., Garcia-Gomez, J., Michaud, P.-A., Rodrigo, M. A., & Comninellis, C. (2003). Electrogeneration of hydroxyl radicals on boron-doped diamond electrodes.

Journal of The Electrochemical Society, *150*(3), D79. https://doi.org/10.1149/1.1553790

[67] Panizza, M., & Cerisola, G. (2009). Direct and mediated anodic oxidation of organic pollutants. *Chemical Reviews*, *109*(12), 6541–6569. https://doi.org/10.1021/cr9001319

[68] Lester, Y., Avisar, D., Gozlan, I., & Mamane, H. (2011). Removal of pharmaceuticals using combination of UV/H_2O_2/O_3 advanced oxidation process. *Water Science and Technology*, *64*(11), 2230–2238. https://doi.org/10.2166/wst.2011.079

[69] Sadeghi, M., Sadeghi, R., Ghasemi, B. G., Mardani, G., & Ahmadi, A. (2018). Removal of azithromycin from aqueous solution using UV-light alone and UV plus persulfate (UV/$Na_2S_2O_8$) processes. *Iranian Journal of Pharmaceutical Research*, *17*(Special Issue 2), 54–64. https://doi.org/10.22037/ijpr.2018.2371

[70] Kim, I., & Tanaka, H. (2009). Photodegradation characteristics of PPCPs in water with UV treatment. *Environment International*, *35*(5), 793–802. https://doi.org/10.1016/j.envint.2009.01.003

[71] Dantas, R. F., Rossiter, O., Teixeira, A. K. R., Simões, A. S. M., & da Silva, V. L. (2010). Direct UV photolysis of propranolol and metronidazole in aqueous solution. *Chemical Engineering Journal*, *158*(2), 143–147. https://doi.org/10.1016/j.cej.2009.12.017

[72] Kovacic, M., Juretic Perisic, D., Biosic, M., Kusic, H., Babic, S., & Loncaric Bozic, A. (2016). UV photolysis of diclofenac in water; kinetics, degradation pathway and environmental aspects. *Environmental Science and Pollution Research*, *23*(15), 14908–14917. https://doi.org/10.1007/s11356-016-6580-x

[73] Rosario-Ortiz, F. L., Wert, E. C., & Snyder, S. A. (2010). Evaluation of UV/H_2O_2 treatment for the oxidation of pharmaceuticals in wastewater. *Water Research*, *44*(5), 1440–1448. https://doi.org/10.1016/j.watres.2009.10.031

[74] Alharbi, S. K., Kang, J., Nghiem, L. D., van de Merwe, J. P., Leusch, F. D. L., & Price,

W. E. (2017). Photolysis and UV/H_2 O_2 of diclofenac, sulfamethoxazole, carbamazepine, and trimethoprim: Identification of their major degradation products by ESI–LC–MS and assessment of the toxicity of reaction mixtures. *Process Safety and Environmental Protection, 112*, 222–234. https://doi.org/10.1016/j.psep.2017.07.015

[75] Nasuhoglu, D., Yargeau, V., & Berk, D. (2011). Photo-removal of sulfamethoxazole (SMX) by photolytic and photocatalytic processes in a batch reactor under UV-C radiation (λmax = 254 nm). *Journal of Hazardous Materials*, *186*(1), 67–75. https://doi.org/10.1016/j.jhazmat.2010.10.080

[76] Donner, E., et al. (2013). Ecotoxicity of carbamazepine and its UV photolysis transformation products. *Science of The Total Environment*, *443*, 870–876. https://doi.org/10.1016/j.scitotenv.2012.11.059

[77] Kim, I., Yamashita, N., & Tanaka, H. (2009). Photodegradation of pharmaceuticals and personal care products during UV and UV/H_2 O_2 treatments. *Chemosphere*, *77*(4), 518–525. https://doi.org/10.1016/j.chemosphere.2009.07.041

[78] Kim, I., Yamashita, N., & Tanaka, H. (2009). Performance of UV and UV/H_2 O_2 processes for the removal of pharmaceuticals detected in secondary effluent of a sewage treatment plant in Japan. *Journal of Hazardous Materials*, *166*(2–3), 1134–1140. https://doi.org/10.1016/j.jhazmat.2008.12.020

[79] Afonso-Olivares, C., Fernández-Rodríguez, C., Ojeda-González, R. J., Sosa-Ferrera, Z., Santana-Rodríguez, J. J., & Rodríguez, J. M. D. (2016). Estimation of kinetic parameters and UV doses necessary to remove twenty-three pharmaceuticals from pre-treated urban wastewater by UV/H_2 O_2 . *Journal of Photochemistry and Photobiology A: Chemistry*, *329*, 130–138. https://doi.org/10.1016/j.jphotochem.2016.06.018

[80] Solomon, D., Kiflie, Z., & Van Hulle, S. (2020). Using Box–Behnken experimental design to optimize the degradation of Basic Blue 41 dye by Fenton reaction. *International*

Journal of Industrial Chemistry, *11*(1), 43–53. https://doi.org/10.1007/s40090-020-00201-5

[81] Tang, J., Yao, S., Xiao, F., Xia, J., & Xing, X. (2022). *Electrochemical oxidation of landfill leachate after biological treatment by electro-Fenton system with corroding electrode of iron*. *International Journal of Environmental Research and Public Health, 19*(13), 7745. https://doi.org/10.3390/ijerph19137745

[82] Duan, P., Gao, S., Lei, J., Li, X., & Hu, X. (2020). Electrochemical oxidation of ceftazidime with graphite/CNT-Ce/PbO_2 –Ce anode: Parameter optimization, toxicity analysis and degradation pathway. *Environmental Pollution, 263*, 114436. https://doi.org/10.1016/j.envpol.2020.114436

[83] Naderi, M., & Nasseri, S. (2020). Optimization of free chlorine, electric and current efficiency in an electrochemical reactor for water disinfection purposes by RSM. *Journal of Environmental Health Science and Engineering, 18*(2), 1343–1350. https://doi.org/10.1007/s40201-020-00551-3

[84] Rajkumar, K., & Muthukumar, M. (2012). Optimization of electro-oxidation process for the treatment of Reactive Orange 107 using response surface methodology. *Environmental Science and Pollution Research, 19*(1), 148–160. https://doi.org/10.1007/s11356-011-0532-2

[85] Hsu, G. W., Lu, Y., & Hsu, S. (2016). Effects of electrolysis time and electric potential on chlorine generation of electrolyzed deep ocean water. *Journal of Food and Drug Analysis, 24*(1), 1–7. https://doi.org/10.1016/j.jfda.2016.07.001

[86] Whangchai, K., Uthaibutra, J., & Phiyanalinmat, S. (2013). Effects of NaCl concentration, electrolysis time, and electric potential on efficiency of electrolyzed oxidizing water on the mortality of *Penicillium digitatum* in suspension. *Acta Horticulturae, 973*, 193–198. https://doi.org/10.17660/ActaHortic.2013.973.26

[87] Wang, Y., Liu, Y., Wiley, D., Zhao, S., & Tang, Z. (2021). *Recent advances in electrocatalytic chloride oxidation for chlorine gas production*. **Journal of Materials Chemistry A, 9**, 18974–18993. https://doi.org/10.1039/D1TA02745J

[88] Bahadır, K. K. (2007). *Response surface optimization of electrochemical treatment of textile dye wastewater*. **Journal of Hazardous Materials, 145**, 277–286. **https://doi.org/10.1016/j.jhazmat.2006.11.031**

[89] Hsu, G. W., & Hsu, S. (2016). *Effects of electrode gap and electric potential on chlorine generation of electrolyzed deep ocean water*. *Journal of Food and Drug Analysis, 24*(1), 4–7. https://doi.org/10.1016/j.jfda.2016.01.006

[90] Rodziewicz, J., Mielcarek, A., Janczukowicz, W., & Bryszewski, K. (2020). *Electric power consumption and current efficiency of electrochemical and electrobiological rotating disk contactors removing nutrients from wastewater generated in soil-less plant cultivation systems*. *Water, 12*(1), 213. https://doi.org/10.3390/w12010213

[91] Hadavand, N., Khazalpour, S., Fotouhi, L., & Nematollahi, D. (2025). *Electrochemical degradation of cephalexin on Ti/TiO_2 /β-PbO_2 anode modified by sodium dodecyl sulfate*. *Scientific Reports, 15*(1), 1–14. https://doi.org/10.1038/s41598-025-94543-8

[92] Ensano, B. M. B., Borea, L., Naddeo, V., Belgiorno, V., de Luna, M. D. G., & Ballesteros, F. C. (2017). Removal of pharmaceuticals from wastewater by intermittent electrocoagulation. *Water, 9*(2), 1–15. https://doi.org/10.3390/w9020085

[93] Midassi, S., Bedoui, A., & Bensalah, N. (2020). Efficient degradation of chloroquine drug by electro-Fenton oxidation: Effects of operating conditions and degradation mechanism. *Chemosphere, 260*, 127558. https://doi.org/10.1016/j.chemosphere.2020.127558

[94] López Zavala, M. Á., Vega, D. A., Álvarez Vega, J. M., Castillo Jerez, O. F., & Cantú Hernández, R. A. (2020). Electrochemical oxidation of acetaminophen and its transformation products in surface water: Effect of pH and current density. *Heliyon, 6*(2),

e03394. https://doi.org/10.1016/j.heliyon.2020.e03394

[95] Perea, L. A., Palma-Goyes, R. E., Vazquez-Arenas, J., Romero-Ibarra, I., Ostos, C., & Torres-Palma, R. A. (2019). Efficient cephalexin degradation using active chlorine produced on ruthenium and iridium oxide anodes: Role of bath composition, analysis of degradation pathways and degradation extent. *Science of the Total Environment, 648*, 377–387. https://doi.org/10.1016/j.scitotenv.2018.08.148

[96] **Bampos, G., & Petala, A. (2021).** Recent trends in pharmaceuticals removal from water using electrochemical oxidation processes. *Environments, 8*(8), 85. https://doi.org/10.3390/environments8080085

[97] Balseviciute, A., et al. (2025). Degradation of multicomponent pharmaceutical mixtures by electrochemical oxidation: Process evolution at varying currents and electrolyte concentrations. *Separation and Purification Technology, 362*, 131697. https://doi.org/10.1016/j.seppur.2025.131697

[98] Murrieta, M. F., & Nava, J. L. (2021). Electrosynthesis of hypochlorous acid in a filter-press electrolyzer and its modeling in dilute chloride solutions. *Journal of Electroanalytical Chemistry, 892*, 115286. https://doi.org/10.1016/j.jelechem.2021.115286

[99] Dang, Q., et al. (2023). Bias-free driven ion-assisted photoelectrochemical system for sustainable wastewater treatment. *Nature Communications, 14*(1), 1–13. https://doi.org/10.1038/s41467-023-44155-5-44155-5.

[100] Fierro, S., Ouattara, L., Calderon, E. H., Passas-Lagos, E., Baltruschat, H., & Comninellis, C. (2009). Investigation of formic acid oxidation on Ti/IrO_2 electrodes. *Electrochimica Acta, 54*(7), 2053–2061. https://doi.org/10.1016/j.electacta.2008.06.060

[101] Huang, S., Yu, J., Li, C., Zhu, Q., Zhang, Y., Lichtfouse, E., & Marmier, N. (2022). The effect review of various biological, physical and chemical methods on the removal of antibiotics. *Water, 14*(19), 3138. https://doi.org/10.3390/w14193138

[102] Coria, G., Nava, J. L., & Carreño, G. (2014). Electrooxidation of diclofenac in synthetic pharmaceutical wastewater using an electrochemical reactor equipped with a boron-doped diamond electrode. *Journal of the Mexican Chemical Society, 58*(3), 303–308.

[103] Nava-Andrade, K., Carbajal-Arízaga, G. G., Obregón, S., & Rodríguez-González, V. (2021). Layered double hydroxides and related hybrid materials for removal of pharmaceutical pollutants from water. *Journal of Environmental Management, 288*, 112399. https://doi.org/10.1016/j.jenvman.2021.112399

[104] Kanwar, B., Koli, M., & Singh, S. P. (2024). Antibiotic amoxicillin degradation by electrochemical oxidation process: Effects of process parameters and degradation pathway at environmentally relevant concentrations. *Environmental Science and Pollution Research*, 1–23. Advance online publication. https://doi.org/10.1007/s11356-024-35780-8

[105] Ding, Y., Zhang, Y., Li, B., Liu, L., Wang, X., & Chen, Z. (2020). UV photolysis as an efficient pretreatment method for antibiotics decomposition and their antibacterial activity elimination. *Journal of Hazardous Materials, 392*, 122321. https://doi.org/10.1016/j.jhazmat.2020.122321

[106] Kawabata, K., Sugihara, K. S., Sanoh, S., Kitamura, S., & Ohta, S. (2013). Degradation of pharmaceutical in UVA, UVB, UVC irradiation. *Journal of Toxicological Sciences, 38*(2), 215–223. https://doi.org/10.2131/jts.38.215

[107] Giri, R. R., Ozaki, H., Guo, X., Takanami, R., & Taniguchi, S. (2015). Efficacies of UVC and VUV photolysis for mineralization of pharmaceutical compounds in mixed aqueous solution. *Desalination and Water Treatment, 54*(13), 3625–3631. https://doi.org/10.1080/19443994.2014.923204

[108] Ahmad, I., Fasihullah, Q., & Vaid, F. H. M. (2006). Effect of light intensity and wavelengths on photodegradation reactions of riboflavin in aqueous solution. *Journal of Photochemistry and Photobiology B: Biology, 82*(1), 21–27.

https://doi.org/10.1016/j.jphotobiol.2005.08.00

[109] Katsumata, C. P., Parizi, M. P. S., Lastre-Acosta, A. M., & Teixeira, A. C. S. C. (2022). Low pressure UV photolysis of the pharmaceutical compounds acetaminophen, atenolol, bezafibrate, diclofenac and ibuprofen. *Water, 14*(19), 3165. https://doi.org/10.3390/w14193165

[110] Hollman, J., Dominic, J. A., Achari, G., Langford, C. H., & Tay, J. H. (2020). Effect of UV dose on degradation of venlafaxine using UV/H_2 O_2 : Perspective of augmenting UV units in wastewater treatment. *Environmental Technology, 41*(9), 1107–1116. https://doi.org/10.1080/09593330.2018.1521475

[111] Nägele, E., & Moritz, R. (2005). Structure elucidation of degradation products of the antibiotic amoxicillin with ion trap MS^n and accurate mass determination by ESI TOF. *Journal of the American Society for Mass Spectrometry, 16*(10), 1670–1676. https://doi.org/10.1016/j.jasms.2005.06.002

[112] Wang, C., Yu, Y., Yin, L., Niu, J., & Hou, L. A. (2016). Insights of ibuprofen electro-oxidation on metal-oxide-coated Ti anodes: Kinetics, energy consumption and reaction mechanisms. *Chemosphere, 163*, 584–591. https://doi.org/10.1016/j.chemosphere.2016.08.057

[113] Shi, H., Wang, Q., Ni, J., Xu, Y., Song, N., & Gao, M. (2020). Highly efficient removal of amoxicillin from water by three-dimensional electrode system within granular activated carbon as particle electrode. *Journal of Water Process Engineering, 38*, 101656. https://doi.org/10.1016/j.jwpe.2020.101656

[114] Wakejo, A. G. D. W. K., Meshesha, B. T., Kang, J. W., & Dessalegn, E. E. (2024). Integrated electrochemical-adsorption for simultaneous removal of pharmaceuticals from water: Process optimization and synergistic insights. *Chemosphere, 344*, 143402. https://doi.org/10.1016/j.chemosphere.2024.143402

[115] Olama, N., Dehghani, M., & Malakootian, M. (2018). The removal of amoxicillin from aquatic solutions using the TiO_2 /UV-C nanophotocatalytic method doped with trivalent iron. *Applied Water Science, 8*(4), 1–12. https://doi.org/10.1007/s13201-018-0733-7

[116] Zhang, Y., Yang, T., Li, R., Cao, X., Kan, Y., & Wei, B. (2022). Efficient degradation of ibuprofen by Co/Fe@CNFs catalyst in the presence of peroxymonosulfate and persulfate: Characterization, performance, and mechanism comparison. *Journal of the Taiwan Institute of Chemical Engineers, 131,* 104161. https://doi.org/10.1016/j.jtice.2021.104161

[117] Firdaus, I., Yaqub, A., Ajab, H., Khan, I., Zafar Amin, B., Baig, A., & Isa, M. H. (2021). Electrochemical oxidation of amoxicillin, ciprofloxacin and erythromycin in water: Effect of experimental factors on COD removal. *Pakistan Journal of Pharmaceutical Sciences, 34*(1), 119–128. https://doi.org/10.36721/PJPS.2021.34.1.REG.119-128.1

[118] Jung, Y. J., Kim, W. G., Yoon, Y., Kang, J. W., Hong, Y. M., & Kim, H. W. (2012). Removal of amoxicillin by UV and UV/H_2O_2 processes. *Science of the Total Environment, 420*, 160–167. https://doi.org/10.1016/j.scitotenv.2011.12.011

[119] Poulopoulos, S. G., Ulykbanova, G., & Constantine, J. (2020). Photochemical mineralization of amoxicillin medicinal product by means of UV, hydrogen peroxide, titanium dioxide and iron. *Environmental Technology*, 41(23), 2977–3014. https://doi.org/10.1080/09593330.2020.1720300

[120] Candido, J. P., Andrade, S. J., Fonseca, A. L., Silva, F. S., Silva, M. R. A., & Kondo, M. M. (2016). Ibuprofen removal by heterogeneous photocatalysis and ecotoxicological evaluation of the treated solutions. *Environmental Science and Pollution Research, 23*, 23470–23483. https://doi.org/10.1007/s11356-016-6947-z

[121] Loaiza-Ambuludi, S., Panizza, M., Oturan, N., Özcan, A., & Oturan, M. A. (2013). Electro-Fenton degradation of anti-inflammatory drug ibuprofen in hydroorganic medium. *Journal of Electroanalytical Chemistry, 702*, 31–36.

https://doi.org/10.1016/j.jelechem.2013.05.006

[122] Farissi, S., Zakkariya, S., Akhilghosh, K. A., Prasanthi, T., Muthukumar, A., & Muthuchamy, M. (2023). Electrooxidation of amoxicillin in aqueous solution with graphite electrodes: Optimization of degradation and deciphering of byproducts using HRMS. *Chemosphere, 330,* 140415. https://doi.org/10.1016/j.chemosphere.2023.140415

[123] Kaur, R., Kushwaha, J. P., & Singh, N. (2019). Electro-oxidation of amoxicillin trihydrate in continuous reactor by Ti/RuO_2 anode. *Science of the Total Environment, 677,* 84–97. https://doi.org/10.1016/j.scitotenv.2019.04.339

[124] Ganiyu, S. O., Oladipo, A. A., Martínez-Huitle, C. A., & Rodrigo, M. A. (2016). Sub-stoichiometric titanium oxide (Ti_4 O_7) as a suitable ceramic anode for electrooxidation of organic pollutants: A case study of kinetics, mineralization and toxicity assessment of amoxicillin. *Water Research, 104,* 99–109. https://doi.org/10.1016/j.watres.2016.09.056

[125] Bergkessel, M., Forte, B., & Gilbert, I. H. (2023). Small-molecule antibiotic drug development: Need and challenges. *ACS Infectious Diseases, 9*(11), 2062–2071. https://doi.org/10.1021/acsinfecdis.3c00189

[126] Aranda, F. L., & Rivas, B. L. (2022). Removal of amoxicillin through different methods, emphasizing removal by biopolymers and its derivatives: An overview. *Journal of the Chilean Chemical Society, 67*(3), 5643–5655. https://doi.org/10.4067/S0717-97072022000305643

[127] Hakizimana, I., Pan, S., Huang, W., Zhao, X., & Wang, C. (2025). *Application of electro-activated persulfate oxidation enhanced with an ultraviolet process in a multi-stage flow-through filtration system for antibiotics abatement.* Separation and Purification Technology. Advance online publication. https://doi.org/10.1016/j.seppur.2024.130090

[128] Chinnaiyan, M. K., & Thampi, S. G. (2018). Photocatalytic degradation of metformin and

amoxicillin in synthetic hospital wastewater: Effect of classical parameters. *International Journal of Environmental Science and Technology, 15*(12), 2681–2692. https://doi.org/10.1007/s13762-018-1725-1

[129] Malefane, M. E., Feleni, U., Mafa, P. J., & Kuvarega, A. T. (2020). Fabrication of direct Z-scheme Co_3O_4 /BiOI for ibuprofen and trimethoprim degradation under visible light irradiation. Applied Surface Science, 529, 145940. https://doi.org/10.1016/j.apsusc.2020.145940

[130] Adityosulindro, S., Julcour, C., Riboul, D., & Barthe, L. (2022). Degradation of ibuprofen by photo-based advanced oxidation processes: Exploring methods of activation and related reaction routes. *International Journal of Environmental Science and Technology, 19*(4), 3247–3260. https://doi.org/10.1007/s13762-021-03372-5

[131] da Luz, V. C., Bazoti, S. F., Behling, L., Dalla Rosa, C., & Pasquali, G. D. L. (2022). Enhanced UV direct photolysis and UV/H_2O_2 for oxidation of triclosan and ibuprofen in synthetic effluent: An experimental study. *Water, Air, & Soil Pollution, 233*(4), Article 55 83. https://doi.org/10.1007/s11270-022-05583-z

[132] Wang, P., Bu, L., Wu, Y., Ma, W., Zhu, S., & Zhou, S. (2021). Mechanistic insight into the degradation of ibuprofen in UV/H_2O_2 process via a combined experimental and DFT study. *Chemosphere, 267*, 128883. https://doi.org/10.1016/j.chemosphere.2020.128883

[133] Regalado-Méndez, A., Ruiz, M., Hernández-Servín, J. A., Natividad, R., Romero, R., Cordero, M. E., Estrada-Vázquez, C., & Peralta-Reyes, E. (2020). Electrochemical mineralization of ibuprofen on BDD electrodes in an electrochemical flow reactor: Numerical optimization approach. *Processes, 8*(12), 1666. https://doi.org/10.3390/pr8121666

[134] Kim, J., Khan, I. A., Lee, J. W., Kim, Y., Jeon, S., Chae, S. R., & Kim, J.-O. (2024). Simultaneous degradation of pharmaceuticals and personal care products in hospital wastewater using ozone under ultraviolet irradiation. *Journal of Water Process*

Engineering, 68, 106366. https://doi.org/10.1016/j.jwpe.2024.106366

[135] Ciríaco, L., Anjo, C. M. S., Correia, J., Pacheco, M. J., & Lopes, A. (2009). Electrochemical degradation of ibuprofen on Ti/Pt/PbO_2 and Si/BDD electrodes. *Electrochimica Acta, 54*(5), 1464–1472. https://doi.org/10.1016/j.electacta.2008.09.022

[136] Zhao, X., Hou, Y., Liu, H., Qiang, Z., & Qu, J. (2009). Electro-oxidation of diclofenac at boron doped diamond: Kinetics and mechanism. *Electrochimica Acta, 54*(17), 4172–4179. https://doi.org/10.1016/j.electacta.2009.02.059

[137] Nguyen, T. P., Tran, Q. B., Ly, Q. V., Hai, L. T., Le, D. T., Tran, M. B., Ho, T. T., Nguyen, X. C., Shokouhimehr, M., Vo, D.-V. N., Lam, S. S., Do, H.-T., Kim, S. Y., Tung, T. V., & Le, Q. V. (2020). Enhanced visible photocatalytic degradation of diclofenac over N-doped TiO_2 assisted with H_2 O_2 : A kinetic and pathway study. *Arabian Journal of Chemistry, 13*(11), 8361–8371. https://doi.org/10.1016/j.arabjc.2020.05.023

[138] León, I., Gomes, H. I., Sepúlveda-Guzmán, S., Cárdenas, J., Rivera, F., Manríquez, J., & Bustos, E. (2021). Electro-oxidation of amoxicillin using titanium electrodes electrophoretically coated by iridium or ruthenium with tantalum oxides. *Journal of Chemical Technology and Biotechnology, 96*(3), 622–629. https://doi.org/10.1002/jctb.6575

[139] Qudsieh, I. Y., Ali, M. A., & Maafa, I. M. (2025). Photodegradation of amoxicillin under UV irradiation using hydrogen peroxide. *Crystals, 15*(2), 167. https://doi.org/10.3390/cryst15020167

[140] Su, R., Chai, L., Tang, C., Li, B., & Yang, Z. (2018). Comparison of the degradation of molecular and ionic ibuprofen in a UV/H_2 O_2 system. *Water Science and Technology, 77*(9), 2174–2183. https://doi.org/10.2166/wst.2018.129

[141] Tetorou, A., Makhatova, A., & Poulopoulos, S. G. (2019). Photochemical mineralization of ibuprofen medicinal product by means of UV, hydrogen peroxide, titanium dioxide and

iron. *Water Science and Technology, 80*(11), 2200–2205. https://doi.org/10.2166/wst.2020.041

[142] Jiang, X., Yuan, J., Zheng, Z., Tao, Y., & Wu, X. (2023). Degradation of sulfonamides by UV/electrochemical oxidation and the effects of treated effluents on the bacterial community in surface water. *ACS Omega, 8*(31), 28409–28418. https://doi.org/10.1021/acsomega.3c02637