



Optimization of electrocoagulation parameters for enhanced wastewater treatment

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DECLARATION

I, Cornel Gnäde, declare that my dissertation titled: Optimization of electrocoagulation parameters for enhanced wastewater treatment, is my work in implementation and design. Complete references are given to indicate and acknowledge the sources that are used in the dissertation. Furthermore, submission of the dissertation to other institutions has not transpired. The dissertation is submitted to the North-West University (NWU).

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ABSTRACT

This thesis presents a comprehensive experimental and statistical investigation into the optimisation of electrocoagulation (EC) parameters for the treatment of slaughterhouse wastewater. A full-factorial response surface methodology (RSM) design with four levels each of hydraulic retention time (HRT) and current density was employed to systematically analyse their effects on pollutant removal efficiency and energy consumption.

A total of sixteen experimental runs were performed, with key response variables including chemical oxygen demand (COD), turbidity, colour, *E. coli* (as an indicator of microbial contamination), electrical conductivity, and power usage. The experimental data were fitted to second-order polynomial models, which successfully captured the relationships between operating conditions and treatment performance. Statistical analysis revealed that both linear and quadratic terms of HRT and current density were significant predictors ($p < 0.05$) for all major removal indicators, with R^2 values exceeding 0.90. This confirmed the robustness of the models and their capacity to describe system behaviour within the tested range.

The highest removal efficiencies were obtained under conditions that balanced sufficient treatment time with adequate coagulant production, while also considering energy input. Confirmatory experiments validated the model predictions, reinforcing the reliability of the proposed equations. Additionally, observed shifts in pH from near neutral (~ 6.8) to mildly alkaline (~ 9.0) supported the electrochemical mechanisms involved, including hydroxide ion generation and enhanced microbial inactivation.

Importantly, actual aluminium dissolution rates exceeded theoretical values predicted by Faraday's law, suggesting possible contributions from mechanical erosion or parasitic reactions. Power consumption varied notably depending on the electrical wiring setup, indicating a clear opportunity for system optimisation through circuit redesign.

The study concludes that optimal EC performance requires a balanced application of electrical and hydraulic inputs. It recommends further exploration of vertical flow reactor designs, alternative wiring configurations to reduce power draw, and integration of continuous sludge removal systems to maintain effective treatment volume.

Key terms: electrocoagulation (EC), wastewater treatment, slaughterhouse effluent, response surface methodology (RSM), hydraulic retention time (HRT), current density, cod removal, turbidity reduction, energy consumption, aluminium dissolution, statistical modelling, environmental engineering.

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LIST OF ABBREVIATIONS

- AC – alternating current
- AOP – advanced oxidation process
- BOD – biological oxygen demand
- BOD/COD – BOD to COD ratio
- BP – boiling point
- CC – chemical coagulation
- CD – current density
- CFR – code of federal regulations
- COD – chemical oxygen demand
- CPI – consumer price index
- DBP – disinfection by-product
- DC – direct current
- DNA – deoxyribonucleic acid
- EC – electrocoagulation
- EDL – electric double layer
- EDS/EDX – energy-dispersive x-ray spectroscopy
- EF – electro-flotation
- EIA – environmental impact assessment
- EO – electro-oxidation
- FAO – food and agriculture organization
- FOG – fats, oils and grease
- FTIR – Fourier-transform infrared spectroscopy
- GDP – gross domestic product
- HRT – hydraulic retention time
- IQR – interquartile range
- MIR – mid-infrared

- MP – melting point
- OH – hydroxide ion
- OFAT – one factor at a time
- SEM – scanning electron microscopy
- TDS – total dissolved solids
- TSDP – total sludge dry product
- TSS – total suspended solids
- UN – United Nations
- UV – ultraviolet
- WHO – World Health Organisation
- XPS – x-ray photoelectron spectroscopy
- XRD – x-ray diffraction

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CHAPTER 1 INTRODUCTION

1.1 Background and introduction

Water covers three-quarters of the earth's surface, consequently, it might seem that the resource is abundant, but the usable amount of water that is available is limited. Almost 97 percent of the planet's water is found in the oceans as salt water. Close to two percent of our planet's water is stored as fresh water in glaciers, ice caps, and snowy mountain ranges. That leaves less than one percent available to us for our daily water supply needs. The freshwater supplies are stored either in the soil (aquifers), bedrock fractures beneath the ground (groundwater) or in lakes, rivers, and streams of the earth's surface (surface water), as depicted in Figure 1.

Water is indispensable for all biological life, and humans are not spared their dependence thereupon. Water is needed in all aspects of our lives, to stay hydrated, prepare food and ensure

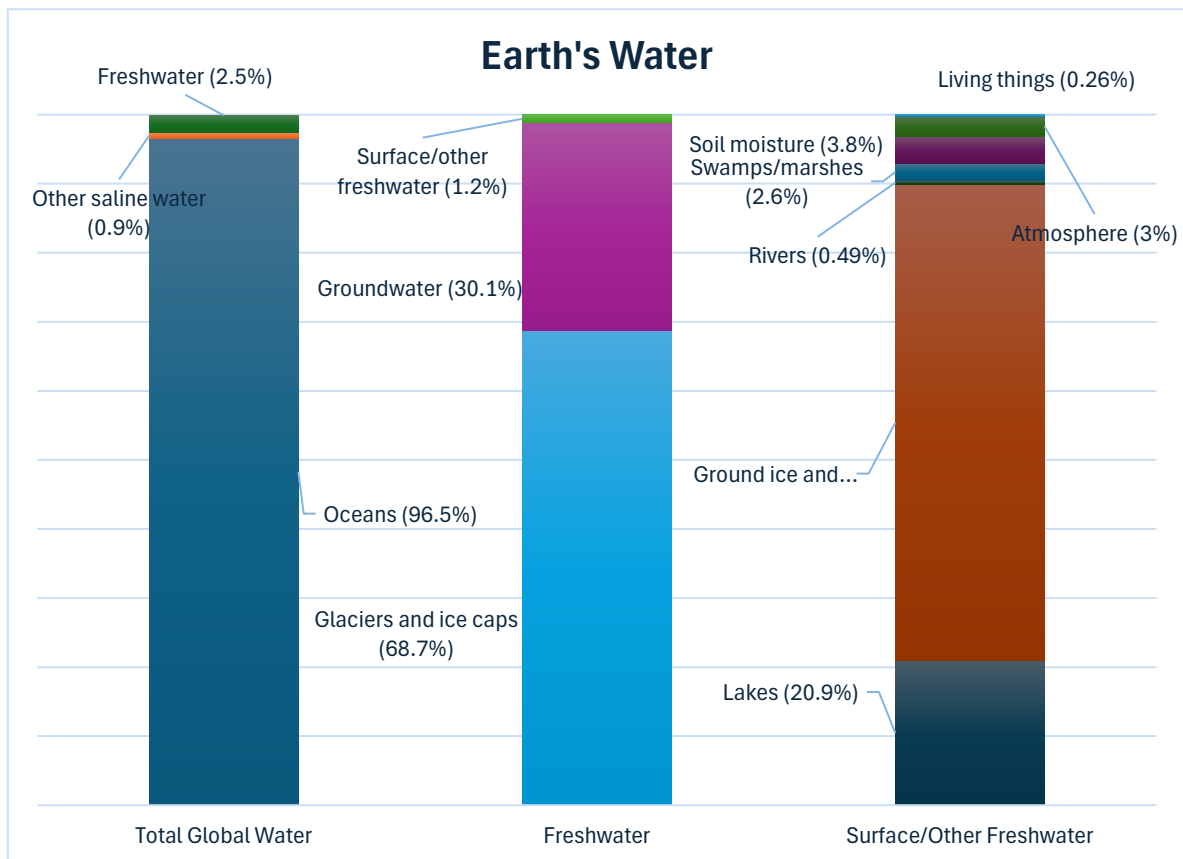


Figure 1: Earth's total global water distribution [1]

good hygiene. While the UN calculates that a person needs 50 to 100 litres of water per day to meet basic drinking, cooking, and hygiene needs [1], the Food and Agriculture Organization (FAO) estimates that “between 2,000 and 5,000 litres of water are needed to produce a person's daily food needs” [1]. Besides domestic use, water is a key input to a variety of economic sectors

such as agricultural, electricity production, manufacturing and mining industries [1], which are far more intensive than the normal human needs.

Close to 70% of global freshwater resources are used for agriculture. In low-income countries, water use in agriculture accounts for 90% of all water withdrawals. The percentage is significantly less in high-income countries (44%), where a higher percentage of water withdrawals goes to the industrial sector (consisting of electricity production, manufacturing and mining industries) [2] (see also Figure 2).

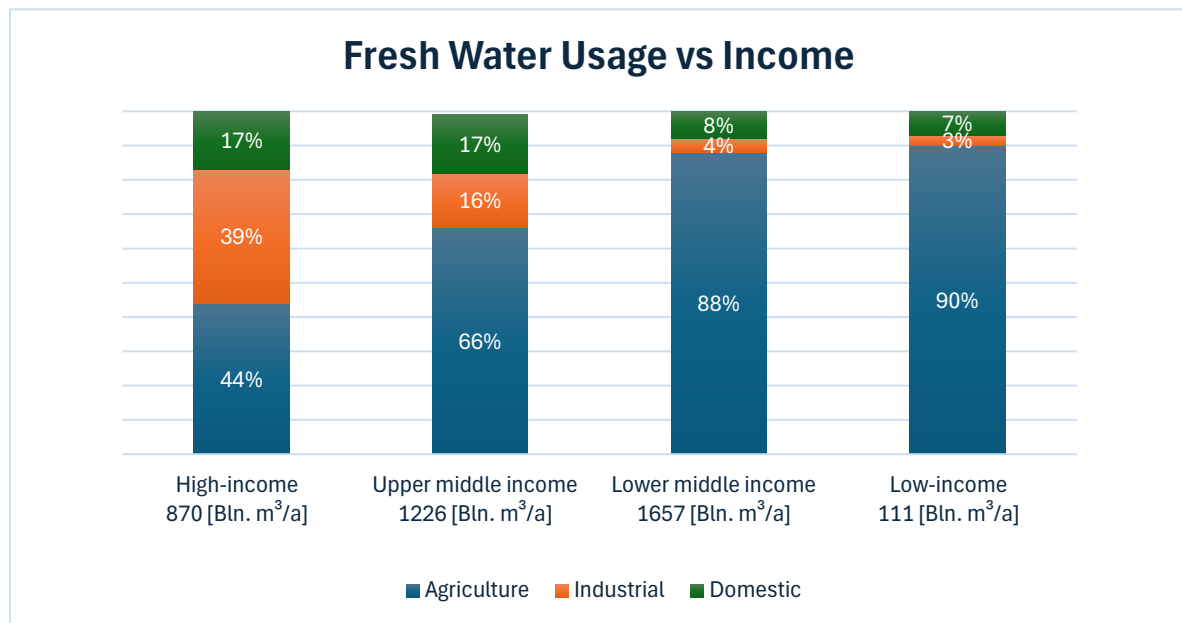


Figure 2: Global freshwater usage based on income [1].

1.2 South African freshwater availability and challenges

South Africa is a semi-arid, water-stressed country, with an average rainfall of approximately 450mm, which is significantly lower than the global average of approximately 860mm per year [2]. Water availability across the country is faced with four major challenges. [2]:

- Uneven spatial distribution and seasonality of rainfall; 43% of the rain falls on 13% of the land.
- Low stream flow in rivers for most of the time, which limits the proportion of stream flow that can be relied upon for use.
- Water storage and distribution infrastructure are ageing and inefficient in some areas, leading to losses.
- The quality of water in some regions is compromised due to pollution and poor management, making it unsuitable for human consumption or agriculture.

Approximately 70% of South Africa's gross domestic product is supported by water from the Limpopo, Inkomati, Pongola and Orange Rivers, which collectively drain two-thirds of the land area. [2].

The natural runoff of the rivers towards the sea amounts to around an average of 50 billion m³ per year, of which 10% originates in Lesotho. [2]. Of the total runoff, approximately 16 billion m³ is available for use through dams, basin transfers and other water resource developments throughout the country. This is currently inadequate to meet the country's total annual water requirement, which is 18.3 billion m³. Furthermore, future projections suggest a 30% gap between supply and demand by 2030 if no measures are taken to improve water efficiency, reduce consumption, and develop new resources. [2]. The continuing trend in industrialisation and urbanisation of the population is expected to place further pressure on the country's water resources unless appropriate corrective action is taken. [3].

Despite the apparent lack of water in South Africa, a large portion of the country's GDP is directly dependent upon water. For example, over 15% of our GDP comes from agriculture, which uses 64.8% of the country's water supply. Energy production is another sector that uses approximately 2% of the country's water supply, which is a relatively small percentage but is still significant because it is essential for many energy generation processes, especially in thermal power plants, which rely on large amounts of water for cooling and steam production [3]. South Africa's industry sector, which contributes 29% of our GDP, uses almost 11% of the available water. However, because a large portion of South Africa's industry sector is mining, the sector's impact on water goes far beyond 11% annually [4] - the National Water Act was created in 1998 to stop extensive pollution from mine runoff that severely contaminated South African water supplies [5].

The highest water consumptions per sector are for agricultural irrigation (12 billion m³/a or 68.4%), water supply services (2.7 billion m³/a or 14.7%), urban industry (2.5 billion m³/a or 13.3%), mining (0.8 billion m³ of 4.3%) and non-urban industries (0.3 billion m³/a or 1.6%) [5].

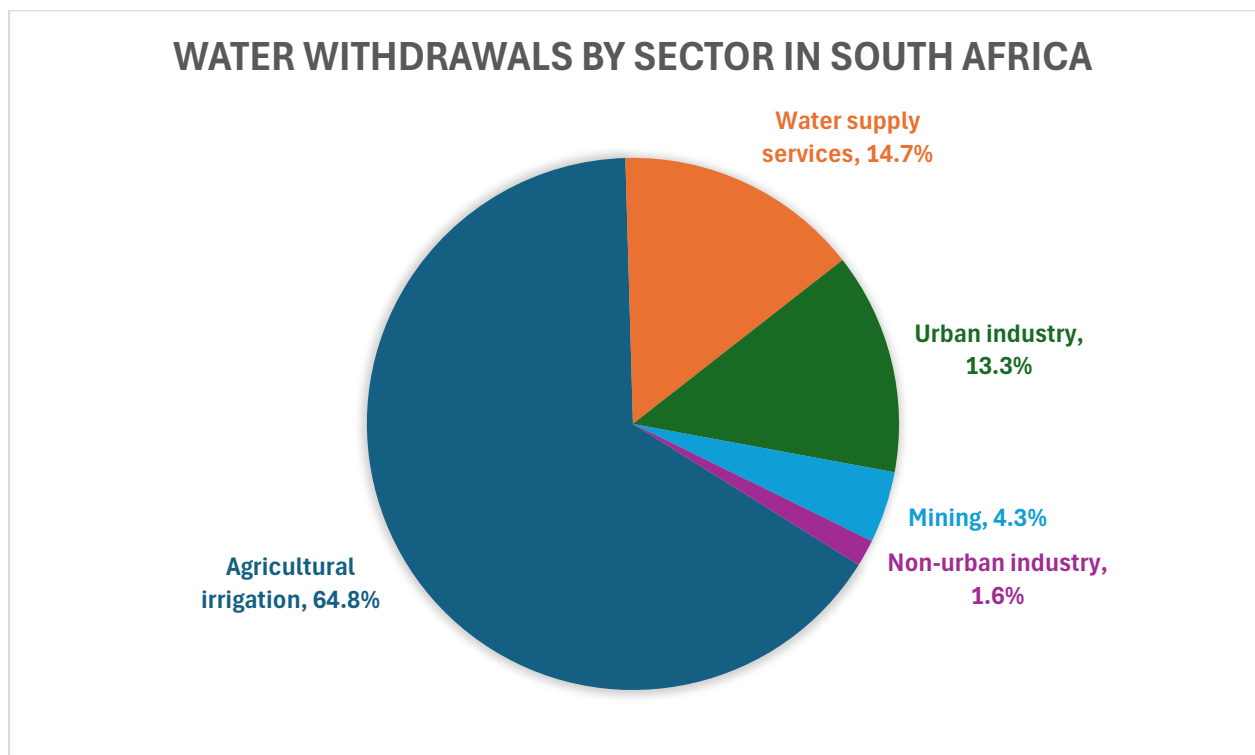


Figure 3: Water withdrawals by sector in South Africa [5].

1.3 Rising water costs vs. demand in South Africa

The global freshwater supply is limited, where the demand could easily exceed the available supply. Climate effects (droughts and floods), a growing population and rapid urbanization, further industrialization and failing infrastructure are factors placing even further pressure on our South Africa's already limited water resources, exacerbating the country's ongoing water crises and emphasizing the need for improvement in local wastewater management. Gauteng alone loses about 40% of all the water that the province produces due to leakages [6]. At the same time water tariffs have been rising, though they remain insufficient to meet the cost of production and the demands of infrastructure maintenance and upgrades.

To meet the country's rising water demand, extensive upgrades in water infrastructure are required, yet water tariffs have remained low and non-cost reflective, preventing municipalities from generating sufficient revenue to cover operating and capital expenses.

Currently, the average water tariff increase is about 8-10% annually, which is often insufficient given the high operational costs, outdated infrastructure, and the need for new investments in treatment plants, pipelines, and reservoirs. Additionally, about 41% of municipal water is categorized as non-revenue water, largely due to leaks and inefficiencies, which leads to significant revenue losses for local governments [6].

1.3.1 Water cost comparison with electricity

The affordability of essential services like water and electricity in South Africa has been a growing concern over the past 25 years. In the next paragraphs an analysis is given regarding the severe increases in the prices of these necessities and their implications for South African citizens.

1.3.1.1 Electricity tariffs vs. inflation

The financial health of Eskom has been overwhelmed with challenges including aging infrastructure, financial mismanagement, and escalating debt. These issues have resulted in substantial tariff increases. From 2007 to 2020, Eskom's average electricity tariff surged by 512%, far exceeding the inflation rate, which stood at approximately 102% over the same period (see also Figure 4). This difference reflects a significant burden on consumers, exacerbated by frequent load-shedding and service delivery issues. Additionally, recent financial retractions of R13 billion and R69 billion further complicate the situation, indicating more rapid tariff hikes soon [7].

Eskom average tariff vs. inflation (CPI)

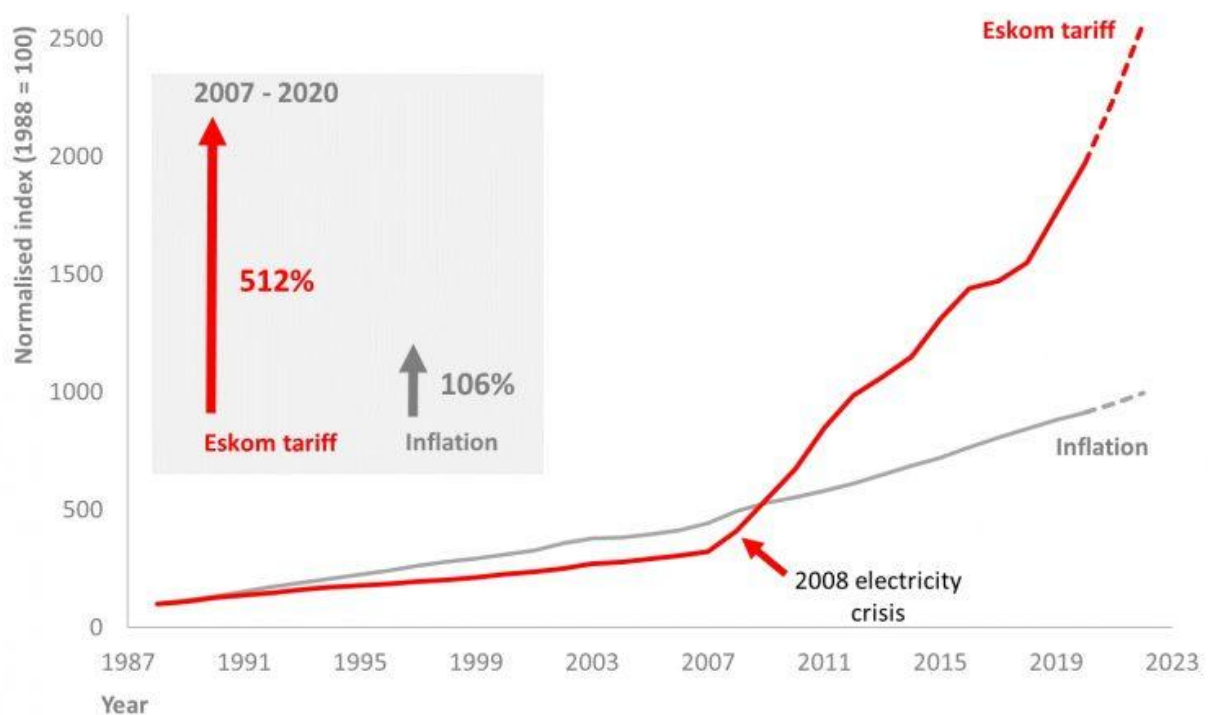


Figure 4: Eskom average tariff vs. inflation (CPI) [7]

1.3.1.2 Municipal water and sanitation tariffs vs. inflation

Like electricity tariffs, the cost of municipal water and sanitation services have also seen a dramatic increase (Figure 5). Since 1996, the average municipal water tariffs have escalated by 1300%, a rate four times faster than inflation. This increase in water costs is compounded by deteriorating service delivery, as evidenced during the “Day Zero” water crisis in 2018. The complexities of municipal water tariff structures, including block tariffs that increase with usage, often obscure the true extent of these cost increases from public view [7].

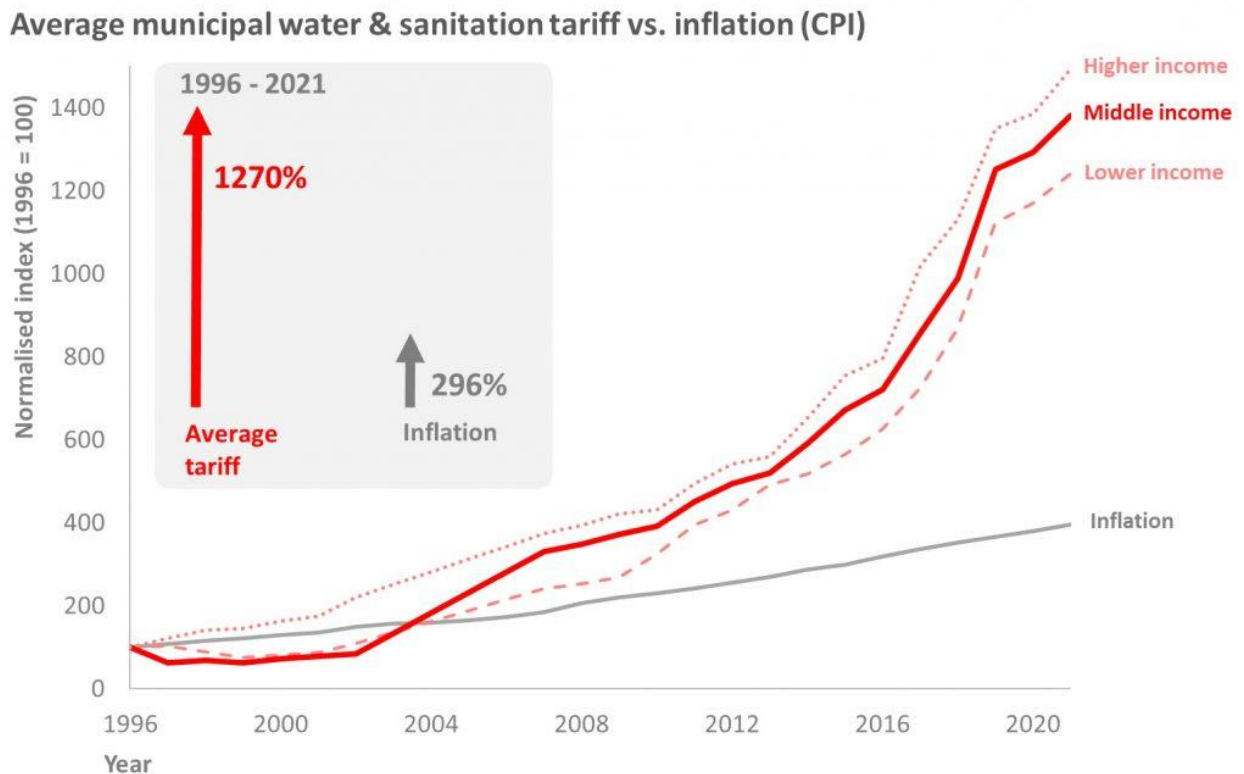


Figure 5: Average municipal water & sanitation tariff vs. inflation (CPI) [7]

1.3.1.3 Affordability vs. disposable income

The inconsistency between the rising costs of electricity and water and household disposable income is evident. Since 1996, the average monthly electricity and water bills have increased by 200%, while household disposable income has grown by only 37%. This widening gap, particularly pronounced since 2008, underscores a growing affordability crisis (Figure 8). Economic downturns and increased income taxes have further strained household finances, making basic services increasingly unaffordable [7].

Electricity + water bill vs average household disposable income in South Africa In real terms (adjusted for inflation)

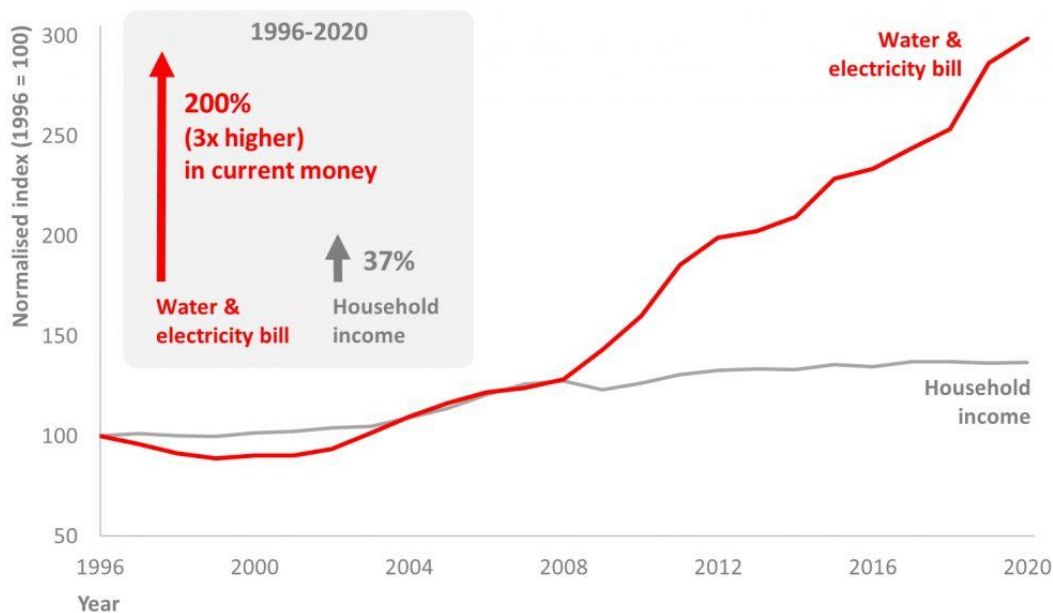


Figure 6: Electricity + water bill vs. average household disposable income in South Africa - adjusted for inflation [7]

The common thread between water and electricity is the need for substantial investment in infrastructure to ensure long-term supply security. Both sectors require an urgent shift towards sustainable management practices, improved efficiency, and the implementation of renewable energy and water-saving technologies to mitigate the financial strain on both service providers and consumers. Without these changes, the gap between supply and demand will continue to grow, leading to further cost increases in both water and electricity sectors, thereby impacting economic stability and social well-being in South Africa [8].

1.4 The opportunity

While the current picture of South Africa's water resources may seem bleak, there is an opportunity for the private sector to reduce its demand on the country's water systems by investing in improved, more efficient water treatment technologies [9].

Many large water users, such as breweries and dairies, have started implementing effluent treatment plants on site, because it makes sense financially, given the higher water tariffs for industrial users, and supports their Environmental, social, and governance goals. The treatment of liquid waste for reuse as process water in plants is also logical from a risk and business sustainability perspective.

By managing liquid waste effectively and safely, industries are not only protecting the environment by preventing pollution, but they are also preserving water for consumption by future generations.

The 2023 Water Market Intelligence Report (MIR), published by GreenCape, highlights the significant investment opportunities and challenges within South Africa's public wastewater infrastructure sector [4].

The National Water and Sanitation Master Plan (2019) estimates that South Africa requires R332 billion to address backlogs in municipal wastewater infrastructure repairs and upgrades. To achieve water security and universal access to water and sanitation by 2030, substantial investment is necessary, particularly in the repair, maintenance, and upgrading of municipal wastewater treatment works (WWTWs) [4]. Additionally, energy efficiency and renewable energy interventions at WWTWs present opportunities for reducing operational costs. Sludge beneficiation is identified as a crucial circular economy solution that can transform wastewater sludge into valuable resources like biofuel and fertiliser. However, several barriers hinder progress, including technical capacity constraints in municipalities, insufficient funding, and regulatory challenges, such as the costly and time-consuming Environmental Impact Assessment (EIA) process. Despite these barriers, the government's renewed focus on water security, combined with public-private partnerships (PPPs) and increased project funding, creates potential for long-term investment and innovation in the sector [4].

1.5 Electrocoagulation as a potential solution

This study aims to investigate the potential scalability and commercial viability of EC as an advanced wastewater treatment. EC has emerged as a promising technique for wastewater treatment due to its ability to remove a wide range of contaminants. EC is becoming a well-known remediation technique that can be used alone or combined with other techniques [10].

1.6 Objective and scope

The main objective of this study was to optimise the EC treatment of slaughterhouse wastewater by evaluating the combined effects of hydraulic retention time (HRT) and current density on pollutant removal efficiency and electrical energy demand.

To achieve this objective, the study pursued the following specific objectives:

1. To review the relevant literature on wastewater treatment technologies, with a specific focus on EC as an emerging treatment method for high-strength wastewater.
2. To identify the EC operating parameters most relevant to this study and define an experimental scope appropriate for laboratory-scale continuous-flow testing.
3. To design and operate a laboratory-scale continuous-flow EC reactor for the treatment of slaughterhouse wastewater.
4. To investigate the effect of HRT and current density on the removal of selected response variables, namely chemical oxygen demand (COD), turbidity, colour, E. coli, and electrical conductivity.
5. To develop empirical models, using response surface methodology (RSM), that describe the relationship between the selected independent variables and treatment performance.
6. To identify operating conditions that provide an effective balance between pollutant removal and energy consumption.
7. To validate the developed empirical models through confirmatory experimental runs conducted under selected operating conditions.

The scope of the study was limited to the optimisation of two selected operating variables, namely HRT and current density, within a laboratory-scale continuous-flow EC reactor using aluminium electrodes. Other EC-related parameters identified in literature, such as initial pH, inter-electrode distance, and mixing conditions, were not varied experimentally in this study, but were either fixed by design or monitored as part of the process response. The study further focused on electrocoagulation as a stand-alone treatment step and did not include chemical pre-adjustment of wastewater characteristics or hybrid treatment integration. This scope was selected to maintain experimental focus and to allow a structured evaluation of the selected variables.

1.7 Structure of the dissertation

This dissertation is organised into five chapters, each addressing key aspects of EC for the treatment of wastewater and the optimisation of parameters influencing its efficiency and effectiveness.

Chapter 1: Introduction

Chapter 1 introduces the background to the study by discussing freshwater scarcity, the need for improved wastewater treatment, and the opportunity for more efficient treatment technologies in

the South African context. The problem statement, research objective, and scope of the study are also presented.

Chapter 2: Literature study

Chapter 2 reviews the relevant literature on wastewater contaminant categories, conventional and advanced treatment methods, and the principles and mechanisms of electrocoagulation. The influence of key EC operating parameters is discussed, followed by a synthesis of previous studies, identification of research gaps, and justification for the present work.

Chapter 3: Methodology

Chapter 3 presents the experimental results, model development, response surface interpretation, and validation outcomes. The results are discussed in relation to pollutant removal efficiency, pH shifts, energy implications, and process performance trends.

Chapter 4: Results and discussion

Chapter 4 presents the experimental results, model development, response surface interpretation, and validation outcomes. The results are discussed in relation to pollutant removal efficiency, pH shifts, energy implications, and process performance trends.

Chapter 5: Conclusions and future work

Chapter 5 summarises the main findings of the study and presents concluding remarks. It also outlines the limitations of the study and provides recommendations for future research and process development.

1.8 Conclusion

The optimisation of EC parameters holds substantial promise for enhancing wastewater treatment processes, making them more efficient and sustainable. By systematically investigating and addressing the challenges associated with EC. This study aims to contribute valuable insights and practical solutions to the field of wastewater treatment technology.

CHAPTER 2 LITERATURE STUDY

Environmental pollution is one of the greatest problems that South Africa and the world is facing currently. This has unfavourably affected humans and animals directly and indirectly. Among various industries, abattoirs are known for their high volume of water consumption and the resultant wastewater, which contains a mixture of organic contaminants, nutrients, microbiological contaminants, inorganic contaminants, suspended solids, and other contaminants. The discharge of untreated or partially treated wastewater is a prime factor for pollution of surface and groundwater [11]. Effective treatment of wastewater is critical to prevent environmental degradation and protect public health.

The situation is even more critical in South Africa, due to the country being particularly arid and without any significant natural dams or lakes, where water is stored artificially and must be reused to meet the country's needs. Over time, this causes an accumulation of organic and inorganic pollutants, primarily characterised by a gradual increase in the concentration of dissolved salts and nutrient pollutants, such as nitrogen and phosphorus, in our water reserves [12].

Conventional wastewater treatment methods, including chemical coagulation, have been widely used to address this issue. However, these methods often fall short in terms of efficiency, cost-effectiveness, and Environmental impact. They typically involve high chemical consumption, generate substantial amounts of sludge, and sometimes fail to remove some contaminants adequately. As the requirement worldwide for freshwater increases, the need for the implementation of more advanced treatment technologies arises to address nonconventional water sources.

Other advanced treatment techniques, including biological oxidation, chemical oxidation, advanced oxidation, nanofiltration, and adsorption, have been investigated. Additionally, electrochemical technologies, such as electrode deposition, electro flotation (EF), electrooxidation (EO), and electrokinetic remediation, have received substantial attention in recent years [13].

Table 1 gives the key for further explanations, shown in Table 2, which indicates the differences between various available treatment methods regarding specific characteristics. These characteristics include energy consumption, maintenance required, operational cost, disinfection ability, scalability, the need for chemical use and the type of waste generated. The key to interpreting the table is as follows

Table 1: Key for Error! Reference source not found.

Table interpretation key							
Energy consumption, maintenance required, operational cost		Disinfection ability		Scalability		Chemical use	
Colour	Rating	Colour	Rating	Colour	Rating	Colour	Rating
	Very high		No		Low		Yes
	High		Limited		Medium		No
	Medium-high		Partial		Medium-high		
	Medium		Yes		High		
	Low						

Table 2: Treatment method characteristic comparison

Treatment Method	Energy Consumption	Maintenance Required	Operational Cost	Disinfection Ability	Scalability	Chemical Use	Waste Generation
Reverse Osmosis (RO)							Brine reject
Electrocoagulation (EC)							Sludge
Distillation							Concentrated waste
Advanced Oxidation Processes (AOPs)							By-products
Ozone Treatment							No residuals
Chlorination							Disinfection by-products
Anodic Oxidation							By-products
Photo-Oxidation							By-products
Photo-Electron-Fenton							Sludge, radicals
Electro-Fenton							Sludge, radicals
Chemical Oxidation							By-products
Nanofiltration							Brine reject
Constructed Wetlands							No significant waste
Activated Carbon Filtration							Spent media
Ultraviolet (UV) Disinfection							No residuals
Ion Exchange							Spent resin, brine
Membrane Filtration							Filter waste
Sand Filtration							Backwash waste
Adsorption							Spent media
Biological Oxidation							Biomass waste
Electrodeposition							Recovered metals

As seen in Table 2, there are various available methods of treatment in the market that apply to specific applications and requirements. Figure 7 gives a visual representation of the treatment effectiveness for specific contaminants.

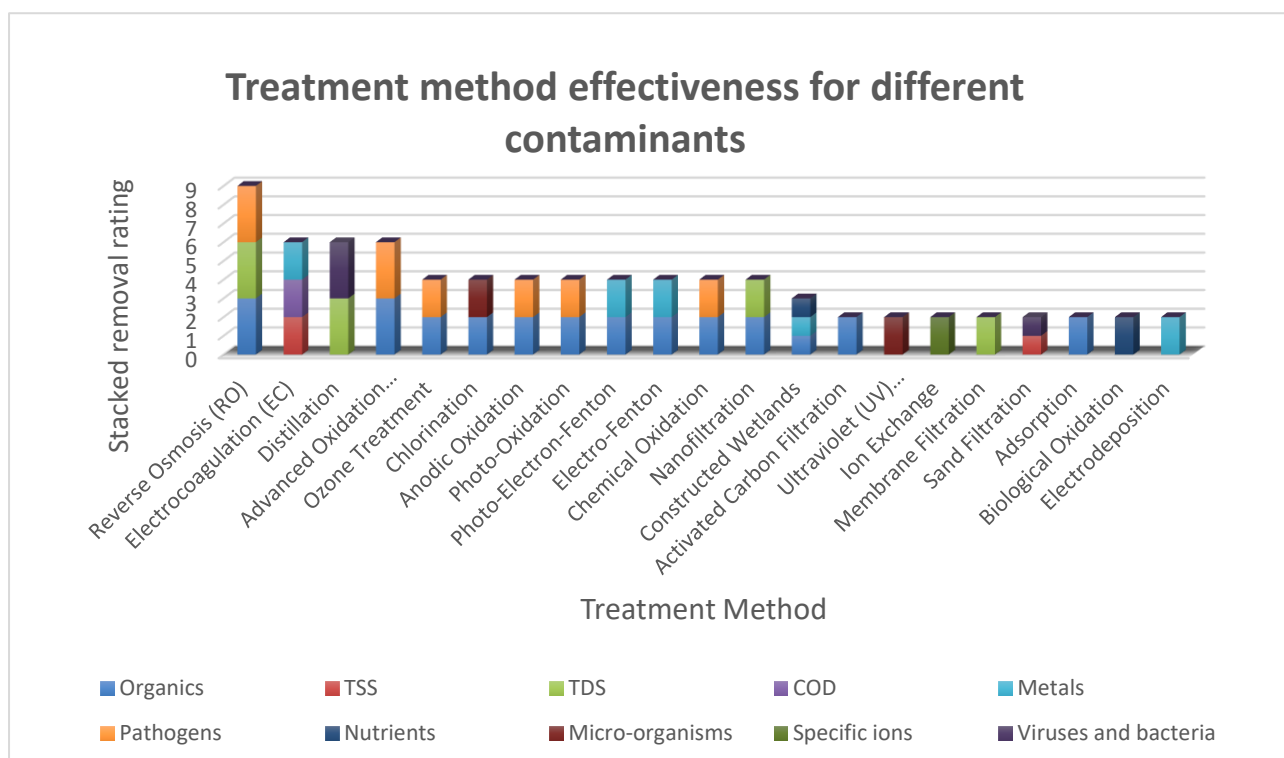


Figure 7: Treatment types for different contaminants [14]

2.1 Wastewater contaminant classification

Wastewater from different sources contains a broad range of contaminants that can be classified into physical, chemical and biological categories. The types and concentrations of contaminants vary with the origin of the effluent, municipal or domestic sewage, industrial discharges, or abattoir wastewater, all of which contribute to pollution if not properly managed. In the South African context, where water resources are scarce, and water quality protection and management thereof are critical [14]. Understanding these contaminant groups are essential for designing effective and relevant treatment methodologies. For example, municipal sewage typically carries high loads of biodegradable organic matter, nutrients (nitrogen and phosphorus), and pathogenic micro-organisms [15]. Industrial wastewater can contain heavy metals and synthetic organic chemicals [16], while abattoir wastewater commonly has high organic content (blood and fat) that leads to high oxygen demand and solids like grease [17]. In all cases, these contaminants have implications for water quality and Environmental health.

2.1.1 Physical contaminants

Physical contaminants are substances or conditions that affect the physical properties of water. They consist of insoluble or particulate matter and other material that can alter the appearance or habitat of the water body without changing its chemical composition. Common physical contaminants in wastewater include:

2.1.1.1 Suspended solids and sediments

Wastewater typically contains suspended particles (TSS) such as silt, clay, organic debris, and other fine solids. A high TSS causes water to become turbid and cloudy, reducing light penetration into aquatic ecosystems. This can inhibit photosynthesis in aquatic plants and algae and impair visual feeding by predators. When these solids settle out in rivers or dams, they can generate sludge deposits that smother bottom-dwelling organisms and alter habitats. Sediment accumulation can block channels and reduce reservoir capacity. Furthermore, sediment may carry attached pollutants, like nutrients, pathogens or metals, into the environment, adding to the negative impact on water quality.

2.1.1.2 Turbidity and colour

Turbidity is the cloudiness of water, caused by TSS and colloidal particles, while colour in wastewater can result from dissolved organic matter, industrial dyes, or metals. Fresh domestic sewage is typically greyish or brown but turns black when septic under anaerobic conditions. Elevated turbidity and dark colouration are aesthetic concerns and can indicate the presence of pollutants. Highly turbid or coloured effluents reduce sunlight penetration and oxygen production in receiving water bodies. Colour and turbidity themselves can also be carriers of contaminants; for instance, pathogens can adsorb onto particulates, allowing them to be transported further downstream. Although colour is not always directly harmful, it is used as an indicator of water quality. For example, reddish or yellow colour may signal Fe or other metals, whereas a dark appearance often correlates with high organic load. These physical indicators alert engineers to underlying chemical or biological contaminants.

2.1.1.3 Floating debris, oil and grease

Wastewaters, especially from municipal sources, can contain floating debris such as plastic trash, rags, and other refuse, as well as fats, oils and grease (FOGs) that separate as a scum layer. In combined sewer systems, storm runoff can introduce litter and organic debris into the wastewater stream. Large floating solids can cause blockages in pipes or treatment plants. Smaller debris like microplastics have emerged as a significant contaminant worldwide [18]. Microplastics do not biodegrade and can be ingested by aquatic life, causing physical harm or leaching toxic additives. Similarly, oils and greases from kitchens, food processing, or petrochemical wastes can form slicks on the water surface. These films or oil/ grease block oxygen transfer from air to water, contributing to oxygen depletion in the water column and harming organisms at the surface (e.g. coating bird feathers and fish gills). In wastewater treatment plants, excess FOG can clog pumps and pipes and form deposits, as noted in abattoir wastewater where congealed fats cause

blockages and scum formation [17]. Therefore, control of gross solids and floating debris is usually the first step in wastewater treatment by screening or skimming.

2.1.1.4 Temperature/ thermal pollution

Many industrial processes, like power plants or large-scale manufacturing processes, often discharge heated water. An elevated water temperature is a physical pollutant because it changes the aquatic environment. Warmer water holds less dissolved oxygen and can stress aquatic organisms by increasing their metabolic and respiration rates. If municipal or industrial effluents are significantly hotter or colder than the ambient environment, they can cause thermal shock to aquatic life and disrupt normal seasonal temperature cycles in rivers or dams [19]. In the South African context, where certain regions already experience high ambient temperatures, thermal pollution from industrial plants can further worsen low oxygen conditions in receiving waters during summer. While temperature is not often regulated as strictly as chemical parameters, it is an important physical aspect of water quality to monitor, as it influences chemical reaction rates and organism health. Cooling ponds or towers and holding tanks are sometimes used in industry to reduce effluent temperature before discharge, mitigating this impact.

2.1.1.5 Summary – physical contaminants

Overall, physical contaminants affect the aesthetics, clarity, and habitat conditions of water. High levels of suspended solids and turbidity can turn streams murky and inhospitable, while floating debris and thermal changes can directly harm wildlife. Managing physical pollutants is thus crucial not only for meeting water quality aesthetics but also for ensuring the effectiveness of subsequent treatment processes, since many treatment methods are less efficient in turbid or debris-laden water.

2.1.2 Chemical contaminants

Chemical contaminants consist of a wide variety of dissolved particulate substances that alter the chemical composition of wastewater. They include both organic and inorganic compounds originating from human waste, industrial processes, and chemical reactions in the sewer system. These contaminants can be reactive or inert, but many have significant impacts on water quality, ecosystem health, and suitability of water for reuse. Key groups of chemical contaminants in wastewater include:

2.1.2.1 Biodegradable organic matter (O₂ demanding substances):

Municipal and abattoir wastewater are typically rich in organic compounds such as proteins, carbohydrates, fats, and other carbon-based substances derived from human or animal waste,

food residues, and plant material. Microorganisms are responsible for the breakdown of the organic matter, a process that consumes dissolved oxygen. The BOD and COD are standard measurements of the organic pollution; high BOD/COD indicates a large quantity of degradable material present. If such effluents are discharged untreated, microbes will deplete the dissolved oxygen in the receiving water as they metabolize the waste, potentially causing hypoxia (low oxygen) or anoxic conditions that suffocate fish and aquatic invertebrates. For example, raw domestic sewage and abattoir effluent have very high BOD levels due to blood, faeces, and food particles, which can create dead zones in rivers if not reduced. This oxygen depletion and associated release of noxious gases, like hydrogen sulphide in septic conditions, are among the most immediate environmental impacts of high organic loads. Controlling organic matter through biological treatment is therefore one of the main objectives of wastewater processing. Notably, abattoir effluents are among the strongest organic wastewaters; studies report extremely high COD concentrations in abattoir wastewater, reflecting the protein- and fat-rich content [17]. Such high-strength waste requires adequate treatment to prevent severe deoxygenation of receiving waters.

2.1.2.2 Nutrients – nitrogen and phosphorus compounds

Wastewater often contains nutrients that, while essential for plant growth, become pollutants in excess. The major nutrients of concern are nitrogen (in forms such as ammonia, ammonium, nitrite, and nitrate) and phosphorus, mostly in the form of phosphate (PO_4^{3-}). Municipal sewage introduces these nutrients through urine and faeces, rich in ammonia and organic nitrogen, and through detergents and food residues (phosphates). Some industrial effluents (e.g. fertilizer manufacturing, agricultural runoff into sewers, food processing waste) can also be significant nutrient sources. If discharged to the environment, nitrogen and phosphorus can fertilize aquatic ecosystems, leading to eutrophication – an over-enrichment of water that causes excessive algae and aquatic plant growth. Algal blooms triggered by high nutrient loads are a common problem downstream of wastewater outfalls. As the algae die and decompose, they further consume oxygen, often creating fish-killing oxygen depletion in a feedback loop like high BOD scenarios. Eutrophication also results in other impacts: it can produce harmful algal blooms (some algae release toxins), reduce water clarity, alter food webs, and cause significant swings in dissolved oxygen and pH. Phosphate is often the limiting nutrient in freshwater, so even modest concentrations can set off algae growth. For instance, phosphate from detergents and sewage was historically responsible for algal overgrowth in many South African dams and lakes, prompting the introduction of phosphate-removal processes at treatment plants. Likewise, ammonia in effluent is toxic to fish at high concentrations and oxidizes (nitrifies) to nitrate consuming oxygen (measured as nitrification oxygen demand). Effective wastewater treatment

aims to remove or convert nutrients (e.g. nitrification and denitrification for nitrogen, chemical or biological phosphorus removal) before discharge to minimize these ecosystem impacts.

2.1.2.3 Heavy metals and metalloids

Heavy metals refer to a group of dense metallic elements that are toxic even at low concentrations and tend to accumulate in the environment. Common heavy metals found in industrial wastewater include mercury (Hg), lead (Pb), cadmium (Cd), arsenic (As), chromium (Cr), nickel (Ni), copper (Cu), and zinc (Zn), among others [16]. These can originate from a variety of industrial activities: for example, electroplating and metal-finishing (Zn, Cr, Ni, Cd), battery manufacturing (Pb, Cd), mining and ore processing (As, Hg, Cu), tannery and textile operations (Cr), and electronics manufacturing [16]. Unlike organic pollutants, heavy metals are not biodegradable and can persist indefinitely in water and sediments. They often adsorb to suspended solids or sludge and can enter the food chain when aquatic organisms ingest those particles. The environmental and health impacts of heavy metals are severe: many are toxic to aquatic life (causing enzyme inhibition, growth and reproductive effects in fish and invertebrates) and dangerous to humans through bioaccumulation in the food chain (leading to neurological damage, organ failure, or cancer in the case of elements like mercury, lead, cadmium, and chromium VI). For instance, cadmium from industrial effluent can concentrate in fish and shellfish and cause kidney damage and bone disease in humans who consume them. Chromium VI, a form of chromium common in industrial discharges (e.g. from metal plating or dye manufacture), is highly carcinogenic and can cause DNA damage in organisms. South African industries, such as mining and metal processing, are significant sources of heavy metal pollution in some waterways, and monitoring programs often find elevated metal levels downstream of industrial zones [16]. Because of their persistence and toxicity, heavy metals are typically regulated contaminants; even at low concentrations (micrograms per Liter), they warrant removal via precipitation, adsorption, or other specialized treatments. In summary, heavy metals are a critical class of chemical contaminant due to their toxicity, persistence, and bioaccumulate nature.

2.1.2.4 Toxic organic compounds

Beyond the bulk organic matter measured by BOD/COD, wastewaters may contain specific organic chemicals of concern. These include a vast array of synthetic organic compounds such as solvents, pesticides, pharmaceuticals, phenols, surfactants, petroleum hydrocarbons, and other industrial chemicals. Many of these substances originate from industrial effluents: for example, phenolic compounds can come from petroleum refineries or pharmaceutical manufacturing, and chlorinated solvents might enter from chemical plants or metal degreasing operations. Some organics also come from household waste (e.g. leftover medications, cleaning

agents) and urban runoff. These priority pollutants are often toxic, even at low concentrations, and can have carcinogenic or mutagenic effects on humans and wildlife. For instance, phenol and its derivatives (common in oil refinery and resin plant effluents) are poisonous to aquatic life and can taint drinking water supplies with bad taste or odour. Pesticides in agricultural runoff can kill aquatic organisms and devastate beneficial insect populations. Chlorinated organic chemicals (like PCBs or dioxins) and polycyclic aromatic hydrocarbons (PAHs) from oil/coal are persistent organic pollutants that accumulate in sediments and biota, posing long-term health risks. Even at sub-lethal levels, many of these compounds can interfere with the reproduction and development of aquatic organisms. Because industrial processes vary, the profile of organic contaminants in wastewater is highly site-specific – for example, a textile factory effluent might contain dyes and volatile organics, whereas a pharmaceutical plant could release complex organic intermediates. Identifying and treating these chemicals is challenging; advanced treatments (activated carbon adsorption, advanced oxidation, and other processes.) are often required. Oil and grease, though mentioned as physical contaminants, are chemically a subset of hydrocarbons and fatty substances; besides causing physical smothering, they can be toxic (especially petroleum oils containing PAHs) and create chronic pollution in sediments. Effective source control and pre-treatment at industrial sites are crucial to limit the discharge of hazardous organic compounds into municipal sewer systems.

2.1.2.5 Salts, minerals, and pH-altering substances

Wastewater may carry high levels of dissolved inorganic salts – for example, sodium, chloride, sulphate, nitrate, and other ions contributing to the TDS. Industrial effluents like those from tanning, pickling, or dyeing processes, as well as mine drainage or groundwater infiltration, can lead to saline or mineral-rich wastewater. In South Africa, certain mining operations and irrigation return flows elevate salinity in water bodies, affecting downstream users. Elevated TDS levels in effluent is problematic for several reasons: it can limit the reuse of water, salty water is unsuitable for irrigation of most crops and can foul boilers or cooling systems, and it can harm freshwater ecosystems, aquatic life adapted to lower salinity may be stressed by saltier conditions. Along with salts, the pH of wastewater is an important chemical characteristic. Domestic sewage is usually slightly alkaline (pH ~7-8) due to bicarbonates, but industrial discharges can be highly acidic (low pH, e.g. acid mine drainage) or caustic (high pH, e.g. from soap or concrete production). Extreme pH levels are corrosive to infrastructure and toxic to organisms – most aquatic life thrives only in a narrow pH range around neutral. Furthermore, pH swings can change the speciation (chemical form) of other pollutants (for example, converting less toxic ammonium to toxic ammonia gas at high pH). Alkalinity (buffering capacity) in wastewater, often from carbonates/bicarbonates, can mitigate pH changes to a degree, but excessive discharge of acids or bases will overwhelm natural buffering. For instance, carbonate from industrial processes

contributes to alkalinity, and while moderate alkalinity stabilizes pH, too much can cause scaling in pipes and disrupt aquatic ecosystems. Regulatory frameworks typically set allowable pH ranges and sometimes limit on TDS or specific ions (like chloride or sulphate) to protect both aquatic life and downstream water users. In practice, neutralization tanks and ion removal processes (e.g. desalination or ion exchange) might be employed to handle these inorganic chemical contaminants. Though less immediately catastrophic than toxins, salinity and pH imbalances can render water unfit for consumption and harm biodiversity in the long term.

2.1.2.6 Emerging contaminants

In recent years, attention has grown around emerging contaminants – pollutants that are not yet regulated or fully understood but are increasingly detected in wastewater effluents. These include pharmaceuticals and personal care products (PPCPs) such as antibiotics, hormones (e.g. birth control oestrogen), painkillers, antiseptics, and cosmetics ingredients, as well as endocrine-disrupting chemicals and micro-pollutants like perfluorinated compounds (from firefighting foams or non-stick coatings). Municipal wastewater is the primary source of these, as unused medications and various consumer product chemicals are washed down the drain. Studies in South Africa and worldwide have found traces of pharmaceuticals and personal care products in surface waters receiving treated effluent. While typically present at very low concentrations (ng/L to µg/L), these substances can have subtle but significant ecological impacts. For example, hormone mimics in wastewater have been linked to reproductive abnormalities in fish (e.g. feminization of male fish downstream of some sewage works due to contraceptive pills in the water). Antibiotic residues can contribute to the development of antibiotic-resistant bacteria in the environment. Many emerging contaminants pass through conventional wastewater treatment unremoved, since treatment plants are not explicitly designed to eliminate them. Their Environmental relevance is still being researched, but as persistent organic pollutants they raise concerns about bioaccumulation and long-term ecosystem health. Internationally and in South Africa, these emerging pollutants are prompting upgrades in treatment technology (like ozonation, activated carbon, or advanced oxidation processes) to enhance removal. Though not typically limited by current discharge standards, their detection in water bodies is an emerging issue for water quality management and highlights the evolving scope of what constitutes a "chemical contaminant" in wastewater.

2.1.2.7 Summary – chemical contaminants

Chemical contaminants thus cover everything from nutrients that cause algal blooms to trace organic toxins and heavy metals. Their effects range from acute toxicity (killing aquatic life immediately) to chronic ecological shifts (like eutrophication or hormone disruption) and human

health hazards (through water supply contamination and food chain bioaccumulation). Effective wastewater methodology must identify the pertinent chemical pollutants in each effluent and apply appropriate treatment or remediation to protect the environment.

2.1.3 Biological contaminants

Biological contaminants in wastewater are microorganisms and biological agents that originate primarily from the faecal waste of humans and animals, as well as certain industrial processes such as fermentation plants or slaughterhouses. Untreated wastewater can contain a dense mixture of microbes, of which many are pathogenic (disease-causing), posing significant risks to public health and the environment. In municipal wastewater, the microbial contents reflect the health and sanitation status of the contributing population [20]. The main groups of biological contaminants include:

2.1.3.1 Pathogenic bacteria

These are disease-causing bacteria commonly found in sewage and some industrial effluents (like food processing waste). A key indicator group is the faecal coliforms, especially *Escherichia coli*, which originate from the intestines of warm-blooded animals and signal faecal contamination. Untreated domestic wastewater typically contains millions of *E. coli* per 100 mL, alongside other bacteria such as *Salmonella*, *Shigella*, *Vibrio cholerae* (the cause of cholera), and *Campylobacter*. If such pathogens reach drinking water sources or irrigated foods, they can cause outbreaks of waterborne diseases. Cholera, typhoid fever (caused by *Salmonella Typhi*), dysentery (from *Shigella* or certain *E. coli* strains), and diarrheal illnesses are all examples of diseases triggered by bacterial contaminants from sewage [21]. South Africa has had instances of cholera and typhoid outbreaks linked to sewage-contaminated water, underscoring the need to remove or kill these bacteria in treatment. Bacterial pathogens in water can also infect aquatic animals; for instance, *Aeromonas* and *Pseudomonas* species in wastewater can cause diseases in fish. Most wastewater treatment plants employ biological digestion and disinfection processes (chlorination, UV irradiation, or ozonation) specifically to reduce bacterial counts to safe levels before effluent discharge. The Environmental impact of pathogenic bacteria is primarily the risk of disease transmission to humans and livestock, as well as the potential to create bacterial pollution that renders water unfit for recreational use or drinking.

2.1.3.2 Viruses

Wastewater can contain a variety of viruses that infect humans or animals. These include enteric viruses (those that infect the gut and are excreted in faeces) such as rotavirus and norovirus (leading causes of gastroenteritis), hepatitis A and E viruses (causing liver disease), and

poliovirus, among others. Viruses are much smaller than bacteria and can be more resistant to Environmental stresses and some treatment processes. They do not multiply outside a host, but even a very low dose of certain viruses can initiate an infection. If wastewater is inadequately treated, viruses can persist in the discharged effluent and contaminate downstream drinking water supplies or bathing areas. For example, hepatitis A outbreaks have been traced to sewage-polluted water or shellfish grown in such waters. Polio, a disease now eradicated in many countries, can resurge in areas with poor sanitation because the virus spreads via faecal-contaminated water [21]. The presence of viruses in wastewater is harder to monitor than bacteria (since standard "indicator bacteria" tests do not capture viruses), but advanced molecular methods or surrogate indicators (like coliphage viruses) are used. Chlorination is effective against many viruses, though some (like adenoviruses) are more resistant to UV disinfection. The Environmental and health relevance of viruses is significant: unlike many chemical contaminants, viruses can lead to epidemic outbreaks if not controlled. Ensuring complete removal/inactivation of viruses is essential when treated wastewater is reused for potable purposes or irrigation of crops eaten raw.

2.1.3.3 Protozoan parasites

Protozoa are single-celled organisms, and certain protozoan parasites found in wastewater can cause serious illnesses. The two most notable waterborne protozoa are *Giardia lamblia* and *Cryptosporidium parvum*, which originate from human and animal faeces. These parasites cause giardiasis and cryptosporidiosis, diseases marked by diarrhoea, stomach cramps, and weight loss. Both *Giardia* and *Cryptosporidium* form cysts or oocysts that are highly resistant to conventional disinfection (for instance, *Crypto* oocysts can survive chlorine levels that would kill most bacteria). They have been frequently detected in untreated sewage; studies report concentrations up to tens of thousands of cysts per Liter in raw wastewater [22]. If effluent is not properly treated (e.g. by effective filtration and disinfection), these protozoan cysts can persist in the environment and infect people who consume or contact the contaminated water. Many regions, including South Africa, have reported *Giardia* and *Cryptosporidium* in both sewage and impacted surface waters, leading to occasional outbreaks (notably, a well-known waterborne disease outbreak in 1980s South Africa was attributed to inadequate removal of *Cryptosporidium* from a drinking water plant). Besides these, other protozoa like *Entamoeba histolytica* (causing amoebic dysentery) may be present in wastewater in areas with those infections. The Environmental impact of protozoan contamination is again a public health concern: these organisms do not typically cause observable harm to aquatic ecosystems, but they pose a direct risk to any use of the water by humans or livestock. Advanced treatment steps (such as ultrafiltration or combined coagulation-filtration) are often required to ensure removal of protozoan cysts, especially when treating wastewater for potable reuse.

2.1.3.4 Helminths (parasitic worms)

Wastewater in areas lacking complete sanitation can contain eggs (ova) of parasitic worms, also known as helminth eggs. These come from human or animal faeces when intestinal worms infect individuals. Soil-transmitted helminths like *Ascaris lumbricoides* (roundworm), hookworms, *Trichuris trichiura* (whipworm), and *Taenia* (tapeworm) frequently find their way into sewage. Studies in South Africa and other countries have found that untreated municipal wastewater commonly harbours a variety of helminth eggs [21]. Notably, *Ascaris* eggs are often the most abundant and are extremely hardy – they can survive in water or sludge for months to years due to their tough shells. If wastewater or sludge is reused in agriculture (for irrigation or as fertilizer) without adequate treatment, these eggs can contaminate soil and crops, leading to transmission of helminth infections to farm workers or consumers. Even in receiving waters, eggs can settle in sediments and later infect humans who contact those sediments (for example, children playing in contaminated rivers). The presence of helminth eggs is primarily a concern in developing regions or where wastewater treatment is insufficient, and it has led to stringent guidelines for water reuse. The World Health Organization (WHO) recommends that treated wastewater used for unrestricted irrigation contain no more than 1 helminth egg per Liter, due to the high risk of exposure in agricultural scenarios [21]. From an environmental standpoint, helminth eggs do not cause immediate ecological harm, but they represent a direct pathway for disease in humans or animals meet inadequately treated effluent. Special treatment methods like prolonged stabilization (ponding) or filtration are often employed to remove these large biological particles from wastewater for safe reuse.

2.1.3.5 Other biological factors

In addition to pathogens, wastewater's biological profile includes biomass of non-pathogenic microbes and by-products of microbial growth. For instance, during degradation of waste in sewers, microbial activity can produce compounds like methane, hydrogen sulphide, or cause the growth of filamentous bacteria and fungi. While these are process considerations more than contaminants, they can influence water quality (odours from H_2S , biofilm formation). Sometimes algal cells can be present in wastewater lagoons or oxidation ponds and be discharged with effluent, though they are usually beneficial in treatment and not considered a contaminant per se. Virulent bacteriophages (viruses that infect bacteria) are another biological entity in wastewater; they can indicate faecal pollution and have potential use as indicators for virus removal performance. Overall, the focus in contaminant classification is on the pathogens as described above, since they have the clearest impact on health and regulatory compliance.

2.1.3.6 Summary - biological contamination

In summary, biological contaminants in wastewater are primarily a health hazard: they can cause waterborne disease outbreaks in humans and animals and thus must be removed or inactivated during wastewater treatment. Their presence links wastewater management directly to public health protection. Discharging effluent with high pathogenic content can contaminate drinking water sources and irrigation supplies, leading to illnesses like cholera, diarrhoea, hepatitis, polio, and parasitic infections [19, 20]. Thus, an effective methodology in wastewater treatment always incorporates steps (biological digestion, filtration, disinfection) targeting these contaminants. Environmentally, by preventing pathogen-laden effluents from entering waterways, we protect not only human communities but also ensure that recreational waters and aquatic ecosystems remain safe and free from contamination that could upset ecological balance or create reservoirs of disease.

2.1.4 EC as a potential solution

EC is a water treatment process that uses an electrical current to remove contaminants and impurities from water. In this process, anodic dissolution generates in situ coagulants, metallic cations (M^{n+}), along with hydroxyl ions (OH^-) and hydrogen gas (H_2) at the cathode. In this process, electrodes, normally Al or Fe, are submerged in the wastewater, and when an electric current is applied, it causes the formation of metal hydroxides which act as coagulant agents. These coagulant agents bind to pollutants, suspended solids, and other contaminants present in the water, forming larger particles called flocs. These flocs can then be easily removed through sedimentation and/or filtration processes. Sweep flocs (large, porous aggregates of metal hydroxides), $M(OH)_n$, have large surface areas to aid in rapid adsorption of soluble organic compounds and trapping of colloidal particles. These flocs are termed "sweep" flocs because they act as a sweeping mechanism, physically capturing and binding pollutants as they settle. These flocs can be removed by sedimentation or H_2 flotation [23]. Electrocoagulation is effective in treating a wide range of pollutants, including heavy metals, oils, suspended solids, microorganisms, and organic compounds, making it a versatile and efficient method for water purification.

Advantages of EC

EC offers several advantages over conventional coagulation methods [24]:

- Treated water is typically more odourless, and colourless.
- EC flocs are like Chemical Coagulation (CC) flocs, but EC flocs are larger in size, enclose less bound water, are more acid resistant and firmer.

- The reuse of EC produced effluent contributes to a lesser water recovery cost because it contains less total dissolved solids (TDS) as related with CC.
- H₂ bubbles produced at the cathode assists with flotation to enhance separation of pollutants.
- Efficiency in pollutant removal: EC is effective in removing a wide range of pollutants, including suspended solids, heavy metals, colour, microorganisms (pathogens) and organic compounds.
- Reduced sludge production: compared to conventional methods, EC generates less sludge, which simplifies disposal and reduces associated costs.
- Chemical free operation: EC eliminates the need for additional chemical coagulants, making the process more Environmentally friendly.
- Versatility: EC can be applied to various types of wastewaters, including those from the abattoir, textile, dairy, distillery, electroplating, and oil refining industries.

Challenges and optimization need:

Despite EC's potential as a treatment, the application is not without its challenges. Several operational parameters, including current density, electrode material and configuration, initial solution pH, and conductivity, influence the efficiency of the EC process. Additionally, the formation of passivation layers, consisting of metal oxides, on the cathodes can hinder the efficiency of the process, necessitating strategies to mitigate this issue [25].

Optimisation of these parameters is crucial to enhance the pollutant removal efficiency while considering energy consumption. A thorough understanding of the interplay between these parameters and their overall impact on the performance of the EC process is essential for its successful implementation of wastewater treatment.

Other challenges or disadvantages may include the following [24]:

- Anodes are dissolved into the solution due to oxidation and need to be replaced at regular intervals.
- High electrical conductivity of the influent water is required.
- Viscous metal hydroxide may be likely to solubilize in some instances.
- Electricity may not be available in all areas.

2.2 Rationale for EC selection

- **Mechanism of EC:**
 - Understanding the fundamental mechanisms of EC is essential for optimising its application in wastewater treatment. EC involves electrode dissolution, charge neutralisation, floc formation, and pollutant removal through coagulation, which is later separated through sedimentation or flotation.
- **Retention time:**
 - Optimising retention time or hydraulic retention time (HRT), for continuous flow reactors (CFR), ensures sufficient pollutant interaction with coagulants while maintaining process efficiency.
- **Current density:**
 - Directly influences electrode dissolution, coagulant production, and energy consumption. It affects the rate of contaminant removal and overall process efficiency.
- **Inter-electrode distance:**
 - Determines electric field strength and mass transfer of ions in the solution. A smaller gap enhances EC efficiency, but excessive reduction increases power consumption.
- **Settling time:**
 - Essential for allowing flocs to settle after EC. Ensures proper separation of treated water from sludge, improving overall system performance.
- **Electrode material selection:**
 - Different electrode materials influence the type of coagulant species formed as well as pollutant removal efficiency. The selection also determines the cost, durability, and Environmental impact of the process.
- **Electrode geometry:**
 - The shape and surface area of the electrodes affect current distribution and mass transfer rates. Optimising geometry improves the removal efficiency as well as energy consumption.
- **Wiring configuration of electrodes:**
 - Monopolar series (MP-S), monopolar parallel (MP-P), and bipolar parallel (BP-P) influence the current distribution and operational efficiency. Furthermore, it impacts electrode lifespan and treatment uniformity.

- **Power supply type:**
 - Alternating current (AC) and direct current (DC) affect electrode dissolution, energy consumption, and coagulation performance. This parameter could aid in optimising treatment efficiency while reducing operational costs.
- **Batch vs. CFR EC:**
 - Batch systems allow for better control over treatment conditions but could be less efficient for large-scale implementation. CFR systems are suited for industrial scale solutions but require optimized flow and HRT.
- **Initial electrical conductivity:**
 - Conductivity impacts electrical current flow and EC performance. A higher concentration of electrolytes could increase the pollutant removal efficiency.
- **Temperature:**
 - Temperature has an influence on reaction kinetics, solubility of coagulants, and electrode behaviour. Higher temperatures may improve removal efficiency but increase electrode dissolution.
- **Initial pollutant concentration:**
 - Determines treatment time, electrode consumption, and energy requirements. This is an important parameter for scaling up the process for different wastewater types.
- **Anion concentration:**
 - Anions (chlorides, sulphates, carbonates) impact coagulation mechanisms and electrode behaviour. High anion concentrations can cause by-product formation or electrode passivation, negatively affecting treatment efficiency.

2.3 Mechanism of EC

The mechanism of EC is extremely dependent on the chemistry of the aqueous medium, particularly its conductivity [26]. The EC reactor configuration varies with the electrical wiring configurations, monopolar-series, monopolar-parallel or bipolar, as well as the number of electrodes.

The stability of colloids in water is a balance between the repulsive electrostatic force and the attractive van der Waals force. The van der Waals forces of attraction, resulting from permanent or induced dipoles, are significant at short distances. The repulsive force is due to the overlapping of the electrical double layers (EDL) (Figure 8), Stern and diffused layers [12].

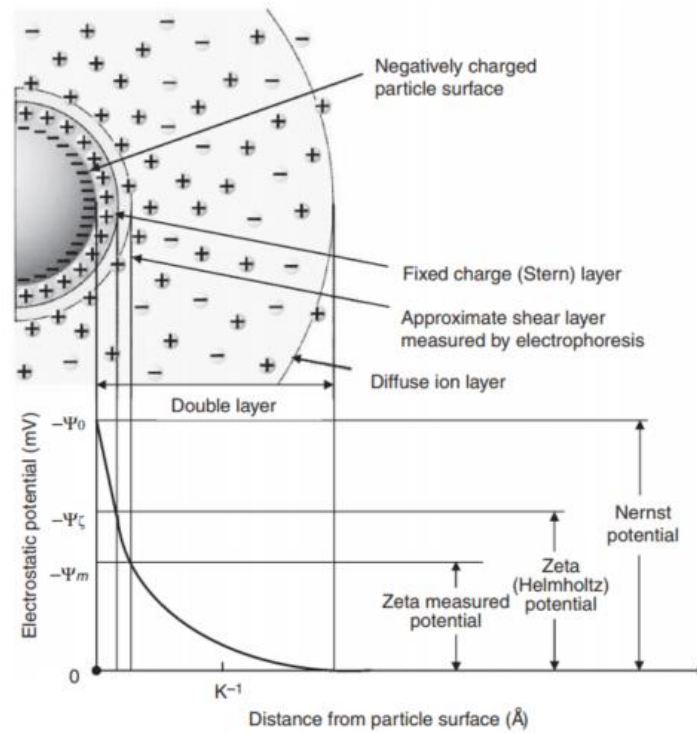


Figure 8: Conceptual representation of the electrical double layer of an EC set-up [13].

In coagulation and flocculation technologies, particles are destabilised with the addition of inorganic or organic chemicals, which influence the EDL. Coagulants are chemicals that reduce the repulsive energy or energy barrier between particles [13].

The reaction mechanism of EC is shown in Figure 8. With the application of an electric current (Direct current (DC) in this example), oxidation of the anodic metal and reduction of the cathodic metal take place. Prominent coagulants that are used in wastewater treatment are Al (Al) and Fe (Fe) metal salts.

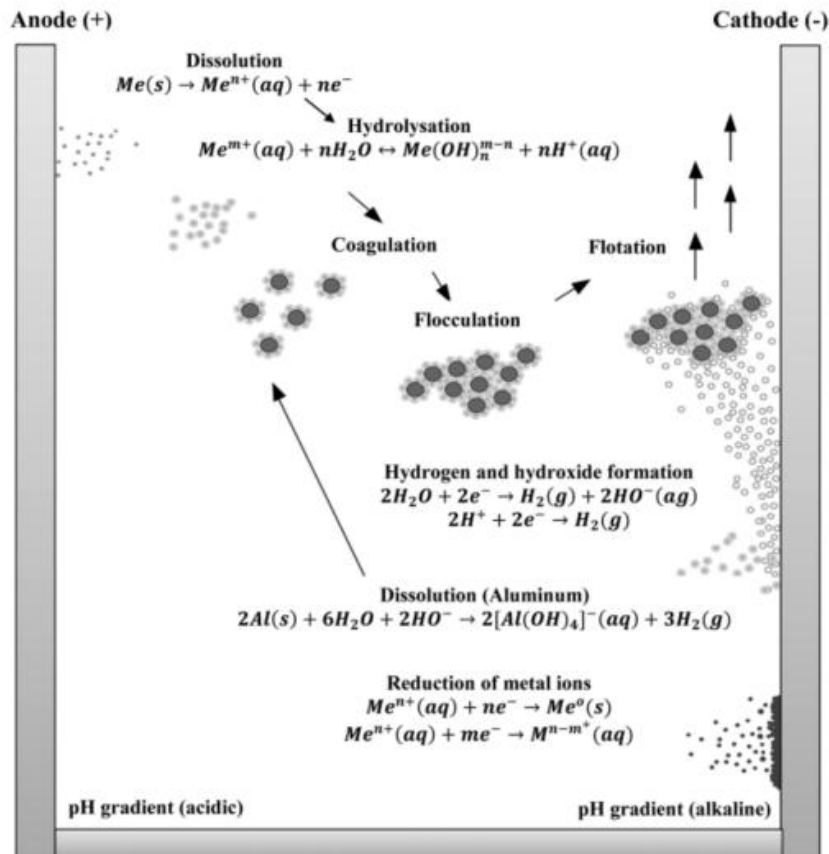


Figure 9: Anodic and cathodic reactions during EC [13]

- Anodic reactions for Al and Fe electrodes:
 - $Al(s) \rightarrow Al^{3+} + 3e^{-}$ (1)
 - $Fe(s) \rightarrow Fe^{2+} + 2e^{-}$ (2)
 - $2H_2O(l) \rightarrow O_2(g) + 4H^{+} + 4e^{-}$ (3)
 - Anode oxidation or atmospheric oxygen oxidises ferrous Fe to Fe^{3+} .
 - Fe can dissolve into divalent Fe(II), and trivalent Fe(III) forms in an aqueous environment during coagulation, whereas Al dissolves only in trivalent form Al(III).
 - $Fe^{2+} \rightarrow Fe^{3+} + e^{-}$ (4)
 - $2Fe^{2+} + 1/2O_2(g) + H_2O(l) \rightarrow 2Fe^{3+}$ (5)
- Cathodic reactions for Al and Fe electrodes:
 - $2H_2O + 2e^{-} \rightarrow H_2(g) + 2OH^{-}$ (6)
 - $2H^{+}(aq) + 2e^{-} \rightarrow H_2(g)$ (7)
- Furthermore, with a high chloride concentration and high anode potential, the following reactions may occur in the EC cell/ reactor:
 - $2Cl^{-} + Cl_{2+} \rightarrow 2e^{-}$ (8)
 - $Cl_2 + H_2O \rightarrow HOCl + Cl^{-} + H^{+}$ (9)
 - $HOCl \rightarrow OCl^{-} + H^{+}$ (10)

Electrochemically generated metal cations, such as Fe^{3+} and Al^{3+} , undergo spontaneous reactions in water, forming corresponding hydroxides and/or poly-hydroxides through complex precipitation kinetics. For Al, monomeric species like $\text{Al}(\text{OH})^{2+}$, $\text{Al}(\text{OH})_2^+$, and $\text{Al}(\text{OH})_4^-$, along with polymeric species such as $\text{Al}_6(\text{OH})_{15}^{3+}$, $\text{Al}_7(\text{OH})_{17}^{4+}$, $\text{Al}_8(\text{OH})_{20}^{4+}$, $\text{Al}_{13}\text{O}_4(\text{OH})_{24}^{4+}$, and $\text{Al}_{13}(\text{OH})_{34}^{4+}$, eventually transform into $\text{Al}(\text{OH})_3$. Similarly, ferric ion species, including FeOH^{2+} , $\text{Fe}(\text{OH})_2^+$, $\text{Fe}_2(\text{OH})_2^{4+}$, $\text{Fe}(\text{OH})_4^-$, $\text{Fe}(\text{H}_2\text{O})_5\text{OH}^{2+}$, $\text{Fe}(\text{H}_2\text{O})_4(\text{OH})_2^+$, $\text{Fe}(\text{H}_2\text{O})_8(\text{OH})_2^{4+}$, and $\text{Fe}(\text{H}_2\text{O})_6(\text{OH})_4^{2+}$, convert into $\text{Fe}(\text{OH})_3$.

The insoluble hydroxides, $\text{Al}(\text{OH})_3$, and $\text{Fe}(\text{OH})_3$ remain suspended in the aqueous medium and destabilise contaminants through mechanisms such as complexation or electrostatic attraction, leading to coagulation. The formation of these hydroxides and complexes is highly dependent on pH [24], [26].

Pollutants in wastewater are normally colloidal particles, which are not removable by flotation, typical filtration or sedimentation, due to their stability in water. Colloids are microscopic particles, in the range of 1 to 10 μm , which are dispersed throughout a solution. The stability of colloids in solution depends on their surface charge [13].

Destabilisation of pollutants or contaminants occurs through two apparent mechanisms:

- i) Cationic hydrolysis products neutralise negatively charged colloids.
 - a. A simplified explanation:
 - i. The metal electrode releases M^{n+} .
 - ii. Hydrolysis reactions forming $\text{M}(\text{OH})^{n+}$, $\text{M}(\text{OH})_2^+$ or similar species.
 - iii. The hydrolysis products interact with colloids in the water.
- ii) Sweep flocculation: Entrapment and removal of contaminants in the form of amorphous hydroxide precipitate. Sweep flocculation is a key mechanism in EC where contaminants in the water are captured and entrapped within a growing matrix of precipitates formed by hydrolysed metal coagulants.
 - a. Simplified explanation:
 - i. The anode releases metal ions.
 - ii. Formation of a dense precipitate takes place (e.g., $\text{M}(\text{OH})_n$)
 - iii. Flocs grow as they further interact with suspended contaminants.
 - iv. Contaminants are entrapped in flocs as they settle or float.

Gas bubbles, H_2 and O_2 , adhere to agglomerates, which are released at the electrode surfaces during electrolysis and carry them to the surface of the water [24].

The size of the gas bubbles plays an important role in the efficiency of EC and electro flotation (EF) systems. Smaller bubbles are preferred as they provide a higher surface area for the particles to adhere to. Electrode material, current density, and pH play a vital role in determining the bubble size. The smallest hydrogen bubbles are produced at neutral or acidic pH, with stainless steel plates have been found to produce the smallest bubbles [13].

Bubble size and diameter depend on the rate of hydrogen production. Hydrogen production increases by increasing the current density applied.

Maintaining cost-effectiveness, it is essential to understand the influence of various operational parameters on its performance. Key parameters, both dependent and independent, significantly impact the effectiveness and efficiency of EC as a wastewater treatment technology.

- Dependent variables:
 - These parameters will remain constant during the experimental phase. The literature review will serve as a basis for identifying their optimal conditions based on existing research.
- Independent variables:
 - These parameters will be actively adjusted during the experimental phase to evaluate their impact on pollutant removal efficiency while considering cost implications. The objective is to determine the optimal range for these variables to maximise pollutant removal rates.

Table 3: Dependent and independent process parameters.

Independent parameters	Dependent parameters
2.3.1.2 Retention time – Flow rate	2.3.3 Inter-electrode distance
2.3.2 Current density	2.3.5 Electrode material selection
2.3.4 Settling time	2.3.6 Electrode geometry
2.3.7 Wiring configuration of electrodes	2.3.8 Power supply type
	2.3.9 Batch vs. continuous flow reactor (CFR EC)
	2.3.10 Initial electrical conductivity
	2.3.11 Initial pH

	2.3.12 Temperature
	2.3.13 Initial pollutant concentration
	2.3.14 Anion concentration

2.3.1 Hydraulic retention time (HRT)

Retention time is an important concept in water treatment because it measures how long water spends in the treatment system, which helps determine the effectiveness of the treatment. It is also important for chemical reactions to occur.

Treatment or electrolysis time has a significant effect on pollutant removal efficiency of the EC process. This explicitly defines the amount of coagulant produced and the cost of the treatment process. It is important to note that electrolysis time will reach an optimum level and will not increase beyond this level or time. With constant current density, coagulant formation increases with electrolysis time, which leads to increased removal efficiency. When enough flocs are generated, more coagulant is produced due to longer electrolysis time will not improve the removal efficiency.

Electrolysis time has a negative impact in the cost of treatment, due to the increase in energy consumption and electrode consumption at longer electrolysis times [24].

From Obu et al. [26], it can be noticed that turbidity removal increased with electrolysis time, though not much improvement in the removal efficiency was seen after 20 minutes. The initial increase was due to more ions that were dissolved into the wastewater with more time, as described above, which leads to an increase in floc formation. An optimal removal efficiency of about 66% was achieved after 30 minutes of treatment at a current density of 2.5 A

From Jebur et al. [25], it was observed that the treated water fraction, after one hour's sedimentation, was less at higher current densities applied. This observation is in agreement with the fact that when excess Al ions are added to the solution, a longer sedimentation time is required for the floc to achieve high water recovery due to the formation of poly-aluminium hydroxides (gel formation) that hinder the destabilisation of colloidal matter [25].

2.3.1.1 Agitation speed

Agitation speeds ensure uniform conditions and avoid concentration gradients in the EC reactor. It further affects the movement of generated ions to enhance or reduce coagulation tempo.

Treatment at optimal agitation speed can lead to shorter required treatment time for a consistent pollutant removal efficiency. An increase in agitation speed, above the optimal determined speeds, can lead to a decrease in removal efficiency as the flocs degrade by collisions caused by the agitation speed [13].

2.3.1.2 Flow rate

Flow rate is only applicable to continuous batch recirculation or continuous flow EC reactors. Another term to describe the parameter is hydraulic retention time (HRT), which can be determined considering the volume of the EC reactor and the flow rate of the pump feeding the effluent [27]. Pollutant reduction rates, specifically COD, noticeably decrease with an increase in flow rate. This could be because of the process's enhanced production of hydroxyl ions at lower flow rates. As the flow rate increases, fewer Fe ions are transferred from the bulk solution to the electrode surfaces [28]. However, with an increase in flow rate, contact time is reduced and therefore lowers the energy consumption per treated volume.

Zampeta et al. [27], varied the HRT between 50, 62 and 83 minutes at three different current densities at each HRT setpoint, namely at 21, 42, and 83 mA/cm². It was determined that the EC treatment was only effective when the effluent was diluted at the highest tested flow rate (10 mL/min or 50 minutes HRT at 83 mA/cm²) [27]. The undiluted effluent could only be effectively treated, with a removal rate of above 90%, for the removal of COD, TSS and colour at the lower flow rates with intermediate current densities, with the highest removal rates seen at the highest current densities and lowest flow rates.

From the above-mentioned reasoning, it is important to consider the interaction between flow rate, current density and pollutant concentration, as it clearly influences the EC process's efficacy and will have a direct impact on the cost of treatment. A lower flow rate implies more contact time with the treatment and allows more time for mixing of the coagulant for a higher percentage of pollutant removal.

A possibly complementary and relatively affordable mechanism to add to the continuous flow is a mixer, to stir the effluent and promote growth and precipitation of flocs [29].

2.3.2 Current density

Current density (CD) is defined as the ratio between the current applied and the surface area of the electrode, which governs the generation of coagulant dose and H₂ gas at the anode and cathode, respectively [30].

Current density plays an important role in the control of the reaction rate in most electrochemical processes and is therefore one of the most significant operational parameters of EC, or current per area of the electrode. It is known that current density determines the coagulant dosage, size of bubble production and affects the growth of flocs [26]. The size of bubble formation reduces with an increase in current density, leading to an increase in pollutant removal efficiency [30]. Therefore, a higher pollutant removal rate can be expected at higher current densities along with more electrical power consumed. However, an increase in current densities above the optimal determined value will have no positive effect on pollutant removal. For Fe and Al electrodes, the dissolution reaction is the primary reaction, and the proportion of other reactions are minor at typical current densities and electrode potential when pH is neutral or acidic [13].

In the literature, a wide range of current densities have been applied, ranging from 1 to 100 mA/cm². Separation processes, which involve flotation cells or settling tanks, required high current density, whereas media or centrifugal filtration integrated in the EC process can accommodate lower current densities.

The amount of anodic dissolution is directly proportional to the current that passes through the electrolytic solution (coulombs per Litre). Faraday's law explains the relationship between current density (mA/cm²) and the amount of metal (m) dissolved (g of M/cm²)

$$w = J \cdot t \cdot \frac{m}{nF} \quad (11)$$

where the quantity of anode material dissolved (w) in g/cm²; applied current density (J) in mA/cm², electrolysis time (t) in s; m the relative molar mass of the electrode material under study; n the number of electrons in the redox reaction; and F the Faraday's constant (96485 C/mol).

From Obu et al. [26] and Jebur et al. [25], assuming no side reactions occur, the equation can also be written as:

$$m = \frac{ItM_w}{zF} \quad (12)$$

where m is the amount of dissolved anode material (g), I is the current (A), t is the electrolysis time (s), M_w is the specific molecular weight of the substance (g/mol), z is the number of electrons involved in the reaction (2 for Fe²⁺ and 3 for Fe³⁺ and Al³⁺), and F is the Faraday's constant. From the results obtained it was deduced that an increase in time as well as current intensity resulted in an increase in anode dissolution. The dissolution rate will vary for different metals, due to a higher molecular weight of Fe compared to Al, as an example where Fe will have a higher dissolution rate [26].

However, the total amount of coagulant produced, includes the chemical dissolution of the electrodes at low pH and the dissolution of the Al cathodes. Furthermore, it seems that current density influences the chemical dissolution of Al cathodes because of the effect it has on the rate of hydroxyl ions produced at the cathode [13]. This is further discussed in the section 2.3.5.

At a high current density, there is a risk of electrical energy wastage by heating up the electrolyte, which results in a decrease in current efficiency. To ensure a high current efficiency, the current density must be selected in accordance with other operating parameters such as pH, temperature and flow rate/ treatment time [24].

It was concluded that an increasing current intensity will increase power consumption, which can be explained by the increased polarisation on the electrodes by increasing the current intensity.

2.3.3 Inter-electrode distance

In the EC process, inter-electrode distance plays an important role in its potency as the electrostatic field depends on the distance between the anode and cathode. Several studies have included tests to determine the optimal inter-electrode distance, as the efficiency of treatment varies with the distance between the electrodes. For boron removal, the efficiency increased from about 30% to over 45% when the inter-electrode distance decreased from 7 cm to 1 cm spacing, which can be attributed to the increased electrostatic interactions [23].

To find the optimum distance between electrodes will add to the maximum pollutant removal efficiency. With a smaller distance between electrodes, a low pollutant removal efficiency will result. This is attributed to the generated metal hydroxides, which act as flocs and remove pollutants by sedimentation, and the degradation due to collisions with each other due to a high electrostatic attraction. Furthermore, smaller inter-electrode distances lead to higher electrical energy consumption due to a higher potential difference for a fixed current. This can be attributed to changes in the electrical conductivity of the solution due to a higher number of metal ions being released per volume of water treated [25]

At a larger inter-electrode distance, the ions generated move at a slower rate. With slower movement, the ions have more time for floc formation, which is required for the coagulation of contaminants. Exceeding the optimal distance will lead to a decrease in anodic dissolution and will increase the distance that is essential for ions to travel for floc formation, which will lead to a decrease in EC efficiency [24].

As shown in equations 13 and 14, the electrical conductivity is directly proportional to the inter-electrode distance. An increase in the distance between the anode and the cathode (g), increases the resistance (R) offered by the cell.

$$R = \frac{g}{KA} \quad (13)$$

where K is the cell-specific conductance, and A is the electrode surface area. According to Ohm's law, current increases with the decrease in resistance. An increase in current increases anodic dissolution, which is directly related to EC efficiency.

The internal resistance (IR-drop) between electrodes or solution resistance can be expressed by the following equation [13]:

$$\eta_{IR} = \frac{Id}{AK} \quad (14)$$

where d is the distance between the electrodes (cm), A is the surface area of the cathode (cm²), K is the specific conductivity of the solution ($\mu\text{S/cm}$), and I is the current (A).

To get the maximum pollutant removal efficiency from an EC reactor, the following must be considered [13]:

- The IR-drop must be minimised.
- O₂ and H₂ gas bubble accumulation at electrode surfaces must be minimised.
- Impairment of mass transfer through the spaces between the electrodes should be minimised.

The IR-drop depends on the following [13]:

- Conductivity of the solution.
- Distance between electrodes.
- Electrode geometry.

Furthermore, in an electrochemical process, the most important economic parameter is energy consumption, W [kWh/m³]. This parameter is calculated with the following expression [25]:

$$W = \frac{U \cdot I \cdot t_{EC}}{V} \quad (15)$$

where U is the applied voltage (V), I is the current intensity, t_{EC} is the applied EC time, and V is the volume (m³) of the solution. The applied voltage, U , can be explained with Ohm's law:

$$U = I \cdot R \quad (16)$$

where R is the resistance (Ohm), and I is the current applied.

From these equations, the following equation can be obtained:

$$W = \frac{I^2 \cdot R \cdot t_{EC}}{V} \quad (17)$$

The inter-electrode distance influences the amount of electrical energy introduced into the effluent to generate an electric field and induce motion of ions [31]. Overall, the energy consumption increases with inter-electrode distance and can be explained by the increase in solution/effluent resistance, according to the following equation:

$$R = \frac{\rho \cdot d_{ie}}{S_a} \quad (18)$$

Where ρ is the resistivity (Ohm cm), d_{ie} is the inter-electrode distance (cm), and S_a is the total active surface (cm²).

In the case for boron removal, the highest removal rate was achieved with the lowest energy consumption, and an inter-electrode distance of 1 cm. [23]

2.3.4 Settling time

The study done by Obu et al. [26], investigated the effect of settling time, which was varied from 5 to 60 minutes. The time given for the settling of the flocs was investigated by varying the time at the optimal pH of 2 and the treatment time of 30 minutes. From the study, it was seen that the removal efficiency increased with settling time, with the highest degree of settling observed within the first 30 minutes. The optimal removal efficiency of about 74% was achieved at a current of 2.5A and a settling time of 60 minutes. The high removal efficiency at the early stages of settling is in accordance with the theory of rapid coagulation.

The theory of rapid coagulation, proposed by Smoluchowski in 1917 [18], describes the kinetics of particle coagulation in a dispersion where Brownian motion (random thermal motion of particles) drives collisions between particles. The theory assumed that, once two particles collide, they immediately stick together to form larger particles without any energy barriers. This process is referred to as diffusion-limited coagulation because the rate of coagulation is controlled entirely by how often particles collide due to diffusion.

Smoluchowski's theory [18] is idealised for rapid coagulation under Brownian motion and dilute conditions and does not account for:

- Electrostatic repulsion between particles (slow coagulation conditions).
- Turbulence-driven coagulation or shear forces.
- Polydisperse systems where particle sizes vary significantly.

Despite its limitations, the theory provides a fundamental understanding of the coagulation kinetics under diffusion-limited conditions.

2.3.5 Electrode material selection

The choice of electrode material is critical in determining the electrochemical reactions that take place in an EC system. Both Al and Fe have been widely used, with each material offering distinct advantages and limitations. They are the most common materials because they are affordable, easily available, and have a history of efficiency [28].

Al dissolves consistently as Al(III), while Fe could dissolve either as Fe(II) or Fe(III), depending on the oxidation conditions. In practice, Fe(II) dissolves and subsequently oxidises in the bulk solution to Fe(III) in the presence of oxidants like oxygen. However, Fe(III) is a stronger coagulant than Fe(II) due to its lower solubility and higher positive charge, which leads to a better pollutant removal efficiency [24]. The dissolution rate of Fe anodes may be lower than the calculated values from Faraday's law at alkaline pH conditions, and could be an indication that other reactions are taking place instead of anode dissolution alone [13].

Selecting the appropriate electrode material depends on the type of contaminants and the electrolyte's chemical properties. Al electrodes provide a better removal efficiency, particularly for colour, but are a more expensive material. In contrast, Fe electrodes are more effective for COD reduction and phenol removal but may show variability in performance under certain conditions. In the research done by Zampeta et al. [27], Al electrodes were selected because colourless effluents were obtained, energy consumption was lower, and metal consumption was less than what is recorded when using Fe electrodes.

It is mentioned in the work done by Naje et al [13] where a study was conducted on the dissolution of Al cathodes with an EC cell constructed of platinised titanium anodes and Al cathodes. In the study, they used a constant charge per volume of 540 Coulombs per litre and various current densities. According to the experimental outcomes, the amount of Al generated increased as the current density was lowered.

Ebba [32], compared Al and Fe electrodes under the same operating conditions with regard to removal rates of colour, COD, turbidity, and energy consumption. According to this study, Fe electrodes were the better option as the removal rates were slightly higher for Fe electrodes, the power consumption was 39 kWh/m³ and 36 kWh/m³, and the operating cost was 4.15 US\$/m³ and 4.01 US\$/m³, for Al and Fe electrodes, respectively [32].

To leverage the advantages of both materials, combinations of Al and Fe electrodes have been explored, demonstrating high removal efficiencies for colour, COD, and other contaminants such as phenol, arsenic, copper, chromium, and nickel from various wastewater types, including industrial and metal plating effluents.

Practical implementation often benefits from using the same material for the anodes and cathodes. This simplifies the design and allows for periodic polarity reversal, which helps mitigate the passivation layer formation expected on the cathodic surface. These insulating layers can hinder efficient current transfer, reducing the EC process's effectiveness.

In cases where wastewater contains substantial concentrations of calcium or magnesium ions, inert cathode materials like metal oxide-coated titanium or graphite are recommended due to their resistance to scaling and corrosion [13].

Al is particularly effective for colour removal, and Fe excels in reducing COD, while combinations of the two materials show promise in achieving high efficiencies across multiple pollutants. These insights underline the importance of carefully selecting electrode materials to optimise performance while balancing cost and operational efficiency.

2.3.6 Electrode geometry

The efficiency of EC is influenced by electrode design and geometry. Solutions with high conductivities, combined with increased cross-sectional areas, minimise internal resistance, enhancing the EC process efficiency. The pollutant removal efficiency is impacted by electrode geometry, as it determines current distribution and reaction rates [13].

2.3.6.1 Current distribution

The electrode geometry affects how the electric current is distributed across the electrode surface.

- Parallel plates: provide a uniform current distribution when placed accordingly, ensuring efficient use of the whole electrode surface.
- Irregular geometries: can cause an uneven current distribution, due to varying distances between the electrodes. Higher current densities in certain regions can lead to localised

over-dissolution or overheating, while lower current density areas may reduce the pollutant removal efficiency.

- Non-rectangular or curved geometries, such as circular or cylindrical electrodes, may experience non-uniform current distribution at the edges compared to flat surfaces, potentially leading to localised inefficiencies [13].

2.3.6.2 Surface area reaction kinetics

Geometries with large surface areas enhance the number of electrochemical reaction sites, improving overall reaction rates, as is the case in packed-bed or mesh designs. However, complex geometries can increase the reaction sites but may restrict the flow of solution or hinder effective mass transfer in confined spaces [33].

2.3.6.3 Electrolyte flow dynamics

Geometries like zigzag or cylindrical designs introduce turbulence in the solution, improving mass transfer between electrodes and the bulk solution. Enhanced mass transfer minimises concentration polarisation, increasing reaction rates.

Bubble formation during EC creates an insulation layer on electrodes. Optimised geometries can facilitate the flow of bubbles away from the electrode surface, reducing potential passivation and enhancing efficiency [13].

2.3.6.4 Localised reaction environments

Uneven geometries may lead to variations in pH and ion concentration near the electrodes, impacting the reaction kinetics and the type of by-products produced. Geometries prone to bubble entrapment or stagnant zones may accumulate fouling and scaling layers, further disrupting current distribution, leading to slower reaction rates [34].

2.3.7 Wiring configuration of electrodes

Electrode material and electrical connection types significantly influence the cost-efficiency and treatment performance of the electrocoagulation (EC) process. Various configurations of electrodes, including monopolar and bipolar setups, have been explored to optimise pollutant removal efficiency, energy consumption, and operational costs.

In EC systems, plate electrodes are typically used, allowing water to flow through the spaces between them. These plates can be arranged in either monopolar or bipolar configurations, each with specific advantages and limitations, as shown in Figure 10.

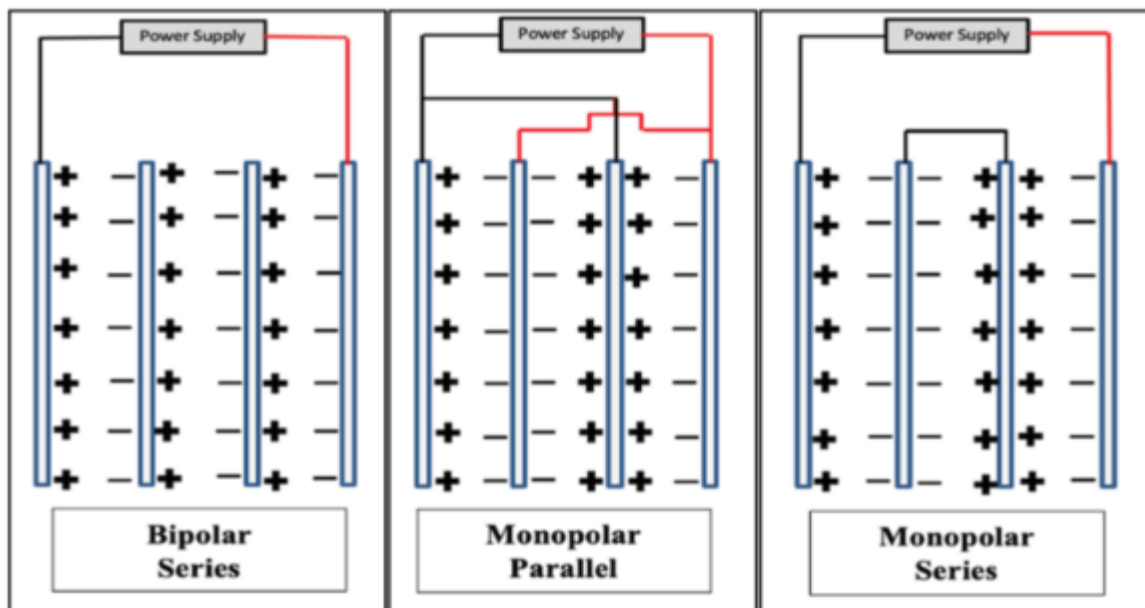


Figure 10: Schematic representation of electrode arrangement during EC [25].

2.3.7.1 Bipolar mode of parallel (BP-P)

In the bipolar parallel mode, sacrificial anodes are placed between two parallel electrodes without direct electrical connections. When current flows through the outer wired electrodes, the neutral sides of the intermediate electrodes adopt an opposite charge relative to their neighbouring electrodes. This arrangement simplifies maintenance and enables anodic and cathodic reactions on the positively and negatively charged sides, respectively. Although BP-P configurations demonstrate high removal efficiencies due to their enhanced surface area, they tend to require a higher energy consumption compared to a monopolar setup [15, 16].

2.3.7.2 Monopolar mode of series (MP-S)

In the monopolar series mode, sacrificial electrode pairs are connected in series, allowing a single current to flow sequentially through the electrodes. While this configuration simplifies electrical wiring, it requires a higher potential difference to overcome the increased resistance across the series of electrodes.

2.3.7.3 Monopolar mode of parallel (MP-P)

In the monopolar parallel mode, each electrode pair is directly connected to the power source, distributing the current among all electrodes proportionate to their resistance. This setup

minimises energy consumption due to the lower potential difference required between the electrodes, making it the most cost-effective option for many applications, but might have lower removal efficiencies [25].

2.3.7.4 Efficiency comparisons

Studies comparing electrode configurations highlight distinct trade-offs. For example, Khaled et al. [13] reported Cd removal efficiencies of 92% for BP-P at 4.15 kWh/m³ and 87% for MP-P at 1.20 kWh/m³. Although BP-P exhibited slightly higher pollutant removal rates, MP-P proved more energy-efficient, making it preferable for large-scale applications. Similarly, combinations of monopolar and bipolar configurations have been explored, revealing the potential for optimising both energy efficiency and treatment performance in single EC reactors. Another practical example is seen in Figure 11, taken from Jebur et al. [25].

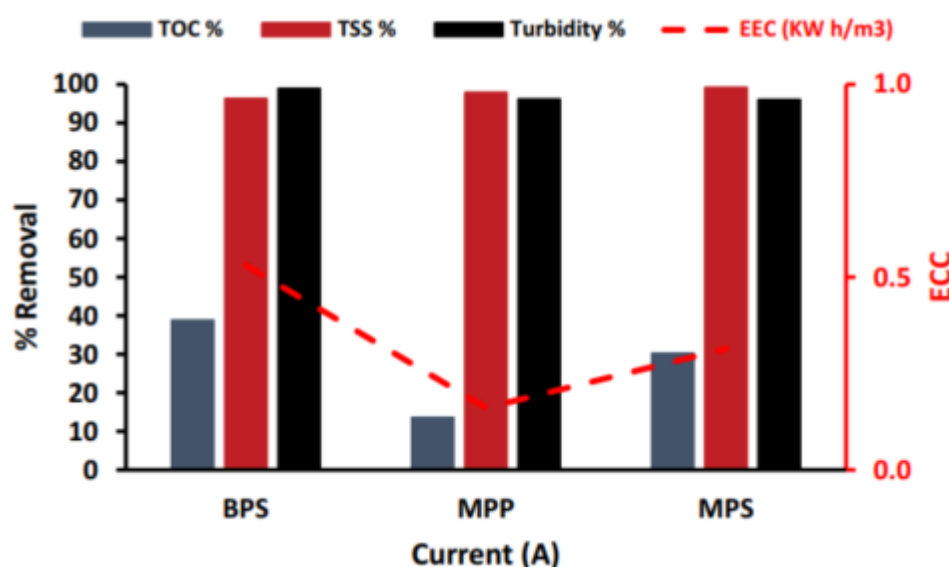


Figure 11: Pollutant removal efficiency difference for different wiring configurations. [25]

2.3.7.5 Total active surface to volume of treated solution ratio (S/V)

This ratio (S/V) was investigated by increasing in electrode pairs at a constant current density. It was seen that when the number of electrode pairs increased, the removal rate also increased. The reason for this observation is due to the workable rate of metal dissolution [31].

According to Vasudevan et al. [34], the use of electrodes with a large surface area is required, and pollutant removal efficiency has been improved independently of the wiring configuration.

The effect of electrode pairs on energy consumption has also been evaluated. Results show that electrical energy consumption increases with the increase in electrode pairs. When the number of electrode pairs increases, the number of electrode pairs acting as a single EC unit increases.

Therefore, the sum of the individual unit voltages and the electrical energy consumption increases.

2.3.7.6 Summary

From the studies discussed, it is evident that monopolar configurations, particularly MP-P, offer cost advantages due to a lower energy consumption, while bipolar configurations like BP-P achieve slightly higher removal rates in certain applications. The selection of the optimal configuration depends on treatment objectives, energy requirements, specific wastewater characteristics, and practical considerations such as system maintenance and rewiring. Additionally, the risk of human error in wiring must be accounted for when choosing the appropriate configuration. Understanding the influence of electrode arrangement and the surface-to-volume (S/V) ratio is crucial for designing efficient electrocoagulation (EC) systems tailored to industrial or municipal wastewater treatment.

2.3.8 Power supply type

In most studies, a DC power supply type was used for the treatment by EC, but there are also several studies where AC was used for the EC process. In Naje et al. [13] it was mentioned that AC and DC currents in EC systems were compared, where similar results were obtained regarding removal efficiencies. The same authors investigated the effect of AC and DC on the removal of Cd from water using Al electrodes. A removal efficiency of 97.5% and 96.2% with energy consumption of 0.454 kWh/l at a current density of 0.2 A/dm² and pH of seven using AC and DC, respectively.

2.3.8.1 Alternating current (AC)

With AC, a slightly lower energy consumption was reported, compared to the use of DC [13]. Results showed that corrosion formation at the electrodes can be reduced using AC instead of DC. Power consumption can be expected to be lower when using AC compared to DC due to the uniform dissolution of the anodes [35].

An advantage of AC for the EC treatment process is that fewer disordered pores are formed with a smooth microstructure of the metal electrode, suggesting that the dissolution takes place uniformly during the electrolysis [35].

2.3.8.2 Direct current (DC)

With the use of DC, the anode corrodes due to oxidation, and an oxidation layer also forms on the cathode, which reduces the flow of electrons between the anode and the cathode, thereby lowering the efficiency of pollutant removal.

When electrodes are fed with DC, the electrode surfaces are normally rough, with indentations. These dents are formed around the nucleus of active sites where electrode dissolution results in the production of metal hydroxides. The number of dents formed on the electrode surface can be attributed to the anode material consumption at active sites due to oxidation at its surface [35]

2.3.9 Batch vs. continuous flow reactor (CFR EC)

Numerous studies have been conducted on batch and continuous flow EC reactors. Batch configurations are preferred for laboratory and pilot plant setups to study the operating parameters of the EC process [27]. A continuous flow reactor (CFR) is more suited for industrial or commercial scale solutions with large volumes. Literature reports demonstrate the removal of contaminants to meet certain discharge standards for CFR EC treatment systems, with large-scale plants reaching a treated volume of around 105 m³/day [36]. A CFR system operates under steady state conditions regarding pollutant concentrations and effluent flow rate. Continuous flow reactors are preferred due to its better control, compared to a batch reactor. Figure 12, shows schematic diagrams of typical batch and continuous flow designs.

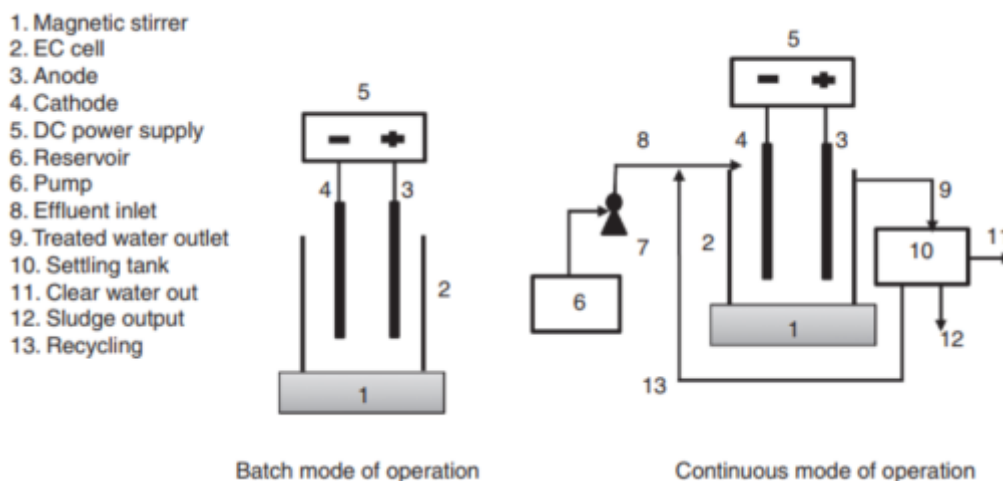


Figure 12: Schematic diagrams of batch and continuous EC reactors [13].

An additional advantage of continuous flow reactors is that the flow rate can be increased to ensure turbulent flow, which enhances mass transfer, removes accumulation of gas bubbles, and mitigates passivation.

Canizares et al. [37], reported that the choice between batch and continuous electrocoagulation (EC) significantly influences the process efficiency, where Al was utilised for pollutant removal. In both modes, similar removal efficiencies were observed at high Al concentrations. However, the batch process achieved phase coalescence at lower Al concentrations (due to a lower anode dissolution rate) than the continuous process.

The key difference arises from how Al is introduced into the system. In the continuous mode, Al addition is instantaneous as wastewater passes through the reactor once, receiving its full Al dose at a single point in time. This leads to a higher concentration of free Al at the moment of addition, which can result in the formation of larger Al hydroxide precipitates with reduced efficiency due to Al entrapment within these particles. The formation of polymeric hydroxo-cationic species is also less favoured.

In contrast, the batch process allows for progressive Al accumulation as wastewater continuously recirculates within the system. This gradual increase promotes more uniform particle formation, leading to higher surface charge density and enhanced polymeric hydroxo-cationic species production. Consequently, Al is utilised more efficiently in batch operation, favouring improved destabilisation and coagulation mechanisms.

Both modes exhibit similar pH trends, indicating that pH variations are not significantly influenced by the mode of operation. However, the batch process demonstrates superior Al efficiency, making it more effective for wastewater treatment applications where Al dosage optimisation is a priority.

2.3.10 Initial electrical conductivity

In the EC process, the pollutant removal efficiency and operating costs are directly dependent on the conductivity of the treated solution. The conductivity of the solution determines the ease with which the applied current can pass through the medium. In some cases, salt additives, such as sodium chloride or sodium sulphate, are added to the solution for conductivity adjustment. The current density increases at a constant or lower cell voltage at higher conductivities.

From Missaoui et al. [23], the conductivity of the solution was adjusted to the desired levels upon the addition of NaCl. According to the study, with parameters $\text{pH}_i = 8$, $J = 7 \text{ mA/cm}^2$ and $t_{\text{EC}} = 60 \text{ min}$, in the presence of chloride ions in solution, a decrease in passivation occurred on the negatively charged Al electrode (cathode) and thereby the efficiency of the EC process increased.

From the experimental results it was deduced that the removal efficiency of B increased with conductivity up to 32.5 mS/cm and then decreased. At a higher electrical conductivity of the

treated solution, insufficient metal dissolution resulted in unfavourable interactions between the coagulant and the pollutant particles, resulting in a reduced removal efficiency [28].

The effect of conductivity on energy consumption was also investigated. With the addition of NaCl, the energy consumption decreased at a constant current density, caused by the inverse relationship between conductivity and resistance. For the study by Missaoui et al. [23] all experiments were conducted at 32.5 mS/cm, for optimal B removal efficiency while minimising the energy consumption [23].

Higher conductivities due to NaCl addition can have benefits due to two effects [10]:

- A decrease in applied voltage and energy demand.
- It changes the ionic strength that affects the coagulation process.

NaCl as an electrolyte result in two processes that may be utilised to treat wastewater [28]:

- EC with coagulant production,
- Electrochemical degradation because of indirect hypochlorite oxidation.

An increase in electrical conductivity, due to the addition of NaCl, has significantly affected the removal efficiency of mimosa tannin in the study done by Kraljic et al. [10]. The removal efficiency significantly increased from less than 20% when the NaCl concentration was 0.58 g/L to more than 80% when the NaCl concentration was ten times higher at a concentration of 5.84 g/L. Increasing the NaCl content further to 22.22 g/L did not result in a better removal efficiency [10]. However, it is worth noting that higher electrode dissolution rates can be expected at higher conductivities, with current density remaining constant.

This indicates that there is merit in considering the artificial adjustment of the conductivity of the solution that requires treatment. These benefits must be considered against the acceptable discharge limits that must be adhered to.

The addition of NaCl decreased the specific energy consumption, which affects the operating cost. However, the dependence of the removal efficiency of NaCl is not completely clear and it depends on the type of pollutant and its concentration.

Canizares et al. [37] studied the difference between the addition of either NaCl or Na₂SO₄ to alter the electrical conductivity and to see what role the medium plays in the COD reduction efficiency of an oil-in-water effluent. From the study, it could be seen that for the same electrical charge passed, a better reduction efficiency of COD was seen in the chloride medium. It is reported that sulphate media promote the formation of precipitates over the formation of polymeric hydroxo-

ions. In the sulphate media, 70% of the Al is in the form of amorphous Al hydroxide precipitate and in the chloride media, the percentage is around 40% [37]. The remaining Al should be in the form of polymeric hydroxo-cations. This means that the Al is more efficiently used in the chloride media

2.3.11 Initial pH

The initial pH of solutions plays a key role in the treatment of wastewater, either in chemical coagulation (CC) or EC. However, EC is more suited to a wide range of pHs, while CC is preferred when the pH is high.

The pH of the electrolyte determines the anode dissolution, affects the conductivity, formation of hydroxides, and ζ -potential of colloidal species in the EC process. An optimal pH of the solution is desirable for any pollutant removal efficiency of contaminants as it helps in the precipitation of pollutants. Varying the pH from low or high from the optimal pH will decrease the pollutant removal efficiency [13].

Various empirical studies show that a high current efficiency occurs at acidic or alkaline conditions, rather than at neutral conditions. The nature of the pollutants determines the EC efficiency; however, pH near seven, pollutant removal was found to be best.

Mechanistic studies of EC processes are complex due to the dynamic changes in pH that occur during operation. For example, the release rate of Al during electrolysis at a constant charge per volume will lower in solutions with a pH below 3 and higher in solutions with a pH above 3. At higher pH levels, the chemical dissolution of Al cathodes occurs as aluminate forms. Chemical dissolution must be kept in consideration when comparing the theoretical “mass loss” to the actual “mass loss” of the electrodes. There will be a higher actual mass loss than what is theoretically calculated, for only electrochemical dissolution [37].

In acidic bulk solutions, hydroxyl ions are consumed by the acid, which hinders the formation of aluminates. Similarly, the dissolution of Fe electrodes was significant in acidic conditions, even without electricity, whereas the oxidation of Fe(II) to Fe(III) only occurred at a pH above 5. In alkaline conditions, the corrosion rate of Fe decreased due to the formation of an inactive surface layer in the presence of oxygen [13].

At neutral pH, the power consumption was found to be high due to conductivity variations. The pH effect is less significant at high conductivity. According to Bharath et al. [24], EC efficiency decreased at pH 10 when using Al electrodes. Optimal pH for general wastewater treatment ranges from 6 – 8 according to Vasudevan et al. [31].

The pH of the electrolyte varies during EC, and the final pH significantly influences the pollutant removal efficiency. EC surges the pH of the solution when acidic, neutral or slightly alkaline, and declines when extremely alkaline.

In highly alkaline pH solutions, Al(OH)_4^- and Fe(OH)_4^- ions are produced, which have a weaker coagulating efficiency. Active coagulant species are produced in acidic, neutral and slightly alkaline conditions. Figure 13 shows that Fe(III) is more result-oriented in a wider pH range than Al(III) and works under slightly alkaline pH. The competing anions affect the optimum pH of the coagulating system [13].

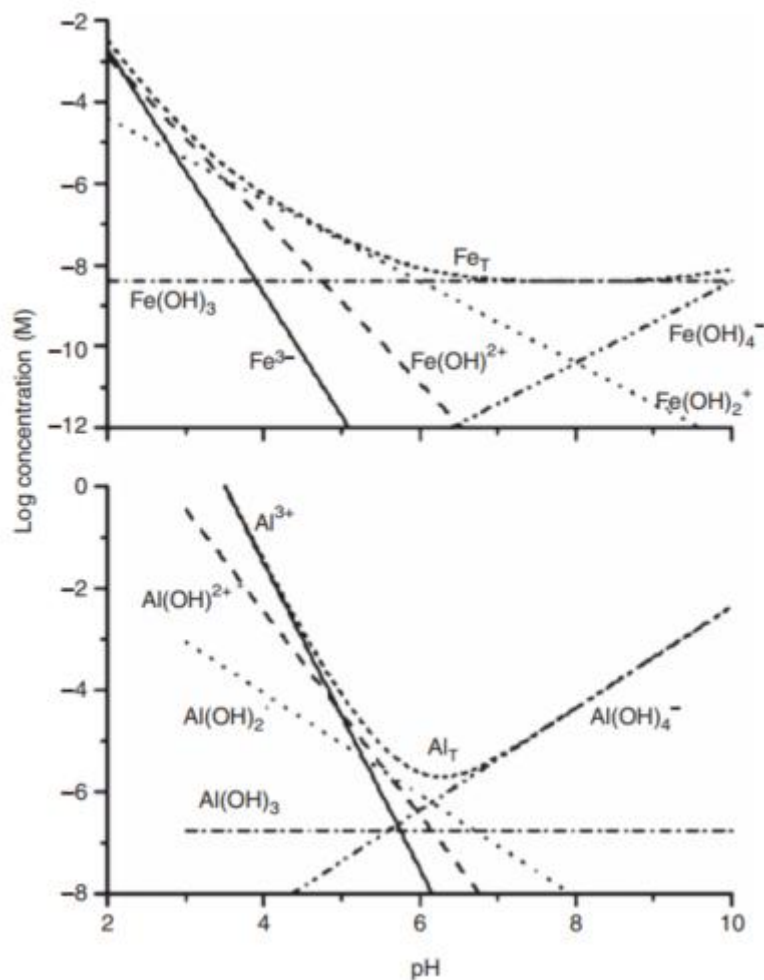
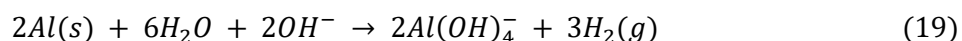


Figure 13: Concentrations of soluble monomeric hydrolysis products of Fe(III) and Al(III) in equilibrium with the amorphous hydroxides at zero ionic strength and 25°C [13].

If the initial pH is highly alkaline ($\text{pH} > 9$), then the pH decreases because of the formation of aluminate $[\text{Al(OH)}_4^-]$, which changes the alkalinity.

At elevated pH levels, Al dissolves to form aluminate ions, as represented by the following reaction:



This dissolution occurs during hydrolysis and de-hydrolysis reactions in the Al ion solution, leading to the formation of aluminate ions at the electrodes. When Fe electrodes are used in the process, "green rust" often develops. This material contains both Fe(II) and Fe(III) hydroxides, as well as associated anions such as Cl^- , CO_3^{2-} , and SO_4^{2-} .

Additionally, the mononuclear hydrolysis products of Fe(III) and Al(III) are influenced by the pH, as illustrated in Figure 13. It is likely that similar polynuclear species are formed during EC and conventional chemical coagulation, demonstrating shared mechanisms between these treatment methods.

The concentration of reaction products is highest at the electrode surface, and a concentration gradient exists from the surface towards the bulk solution because the reactions take place on the electrode surface. Therefore, the pH decreases at the vicinity of the anode and increases at the vicinity of the cathode surface. This could lead to precipitation of hardness-forming minerals like calcium carbonate on the cathodes, where the solubility changes with the pH [13].

It has been established that the initial pH of the treated solution has a considerable influence on the performance of the EC process. To investigate the effect of initial pH from the work conducted by Khaoula et al. [23], the initial pH range was varied between 5 and 10 to determine the effect on boron removal with Al electrodes.

The results showed that the removal efficiency increased up to an initial pH of eight and then decreased. If the activity-pH diagram for Al(III) species in equilibrium with $Al(OH)_{3(s)}$ is investigated (Figure 15), it can be seen that the dominant Al(III) species is in the form of $Al(OH)_{3(s)}$ at a pH 5 - 8.5 interval. $Al(OH)_4^-$ forms at the higher pH and $Al(OH)_4^-$ is a dissolving form, which does not form flocs [23]. The highest boron removal efficiency was obtained at pH 8, because boron was in the $B(OH)_3$ form, and the formation of $Al(OH)_{3(s)}$ was at its highest at this pH. Therefore, all experiments conducted by Obu et al. [26] were performed at a pH of eight.

From Obu et al. [26], where Fe electrodes were used, the effect of the initial pH was also studied on the effect on the removal efficiency of turbidity. The study was carried out with the pH that was varied between 2 and 10 at an electrolysis time of 15 minutes, settling time of 30 minutes and varying current intensity. The pH of the aqueous medium was adjusted by means of adding HCl or NaOH to get to the desired pH value. The results show that higher removal efficiencies of turbidity were achieved when the solution was acidic. When the pH was decreased to two, the removal efficiency increased to more than 60%, compared to the removal efficiency of 40% at pH

10, at a current intensity of 2.5 A/m². By lowering of the current density to 0.5 A/m², the removal efficiency accordingly decreased to 50% at pH 2 and only obtaining a removal efficiency of 30% at pH 10. The high percentage of pollutant removal with an acidic medium agrees with findings in similar studies, like the one conducted by Bayar et al. [38]. The pH will increase when starting with an acidic initial pH due to hydrogen (H⁺) evolution and the generation of hydroxide (OH⁻) ions at the cathodes.

2.3.11.1 Conclusion

High pollutant removal efficiencies were observed near neutral pH (~7), where EC performance peaks. However, in highly alkaline conditions, active coagulant species formation is inhibited due to many weaker species like Al(OH)₄⁻ and Fe(OH)₄⁻ being formed. The conductivity variations at neutral pH also led to higher power consumption.

Al electrodes work well in acidic, neutral, and slightly alkaline conditions for various contaminants, whereas Fe electrodes exhibit higher efficiency for turbidity removal in acidic media. Optimal pollutant removal often occurs in the 6–8 pH range, depending on specific contaminants. Adjusting the pH with additives like HCl or NaOH allows optimisation for wastewater types, but the natural shift in pH during EC is a critical design consideration.

The initial pH not only determines the type and efficiency of coagulating species formed but also affects the dynamic changes during EC. Optimal pH ensures high removal efficiency by supporting active coagulant species formation, balancing operational parameters, and maintaining cost-effectiveness in wastewater treatment.

2.3.12 Temperature

The temperature of the electrolyte has the potential to significantly affect the EC treatment's effectiveness. The effect of temperature on the removal efficiency is dependent of the pollutant's removal mechanism. Therefore, in an EC process that is carried out at ambient temperature, it is necessary to know the factors that cause variations in temperature during the process. Above a certain temperature, a decrease in pollutant removal efficiency can be expected due to the decrease in metal hydroxide formation [24]. At low temperatures, low efficiency can also be expected due to a lower anodic dissolution rate.

The temperature influences the EC process by altering the rate of reactions, solubility of metal hydroxides, liquid conductivity, and kinetics of gas bubbles or small colloidal particles. The effect of temperature (20–50°C) on phosphate removal from wastewater using Al electrodes demonstrates that higher temperatures enhance removal efficiency. This improvement is attributed to increased mass transfer of Al ions from the anode surface to the bulk solution and

an accelerated rate of Al hydroxide formation, it also reduces the solution viscosity and a consequent increase in ion diffusivity. However, temperatures above 50°C are not recommended as they are economically impractical and can negatively impact on the reaction rate and overall removal efficiency [24].

The study of Obu et al. [26], investigated the effect of solution temperature in the temperature range between 30 to 60°C. The removal increase from 40-60°C was the most significant, with no real removal efficiency difference between 50 and 60°C with the highest removal efficiency being 93.69% at 60°C.

According to Naje et al. [13], when the solution temperature was increased from 293 K to 333 K, the removal efficiency of boron increased from 84% to 96%. In the same article, it was mentioned that the opposite was seen with the removal efficiency of colour, COD, and phenol, where the removal efficiency decreased by 10-20%. This can be attributed to the shrinkage of large pores of $\text{Al}(\text{OH})_3$ gel, which caused the formation of dense flocs that were more likely to be deposited on the electrode surface if Al electrodes were used.

From this discussion, the temperature can have a significant effect on the efficiency of pollutant removal. It seems that the optimum initial solution temperature should be 50°C.

2.3.13 Initial pollutant concentration

An increase in initial pollutant concentration, while keeping other parameters constant, reduces the pollutant removal efficiency of the EC process. This can be explained by the fact that at fixed operating parameters, the amount of coagulant generated will be constant and will eventually be insufficient to form enough flocs for the higher pollutant concentration. Therefore, it will lead to longer treatment time required to allow to opportunity to generate enough coagulants by anodic dissolution.

A higher pollutant concentration not only decreases the removal efficiency, but it also negatively affects the energy consumption due to longer treatment time required and possibly increases the resistance of the solution [24].

It will be essential to determine what the problematic parameters are within the aqueous medium, which are above the discharge limits, to narrow the scope of the parameters that are tested for. Laboratory analysis can become expensive, and this specific cost can be optimised if specific problematic parameters are pinned down to compare before and after EC treatment.

2.3.14 Anion concentration

The concentration of anions influences the stability of the passive layer on an electrode. Sulphate anions are passivating agents and can negatively affect the rate of metal cation production. Chlorides, on the other hand, induce the breakdown of a passivation layer and can encourage pitting corrosion.

In Naje et al. [13], it is reported that fluoride and arsenate compete with phosphate ions and reduce their removal efficiency by EC. It is also reported that sulphate anions increase electrical energy consumption and cause a decrease in removal efficiency by EC.

The ratio of chlorides per sulphates ($[Cl^-]/[SO_4^{2-}]$) should be 0.1 or higher to ensure the breakdown of the passivation layer on the cathode [13].

2.4 Wastewater characteristics to consider measuring

For a comprehensive evaluation of the EC process's treatment effectiveness, it would be beneficial to measure the following wastewater characteristics both before and after treatment. The measurements will provide a clear understanding of the system's performance and the removal efficiencies achieved [11].

2.4.1 Basic physicochemical properties:

1. **pH:** Helps assess changes in acidity/ alkalinity, which can significantly influence coagulation efficiency.
2. **Electrical conductivity:** Reflects the ionic content of the water and can indirectly indicate ion removal.
3. **Turbidity:** Indicates the presence of suspended solids, often used as a quick measure of treatment effectiveness.
4. **Colour:** Often used for aesthetic and qualitative analysis; certain dyes or pigments require specific quantification.
5. **TSS:** Quantifies solids that can settle out after treatment.
6. **TDS:** Helps to evaluate the removal of dissolved ions.
7. **Temperature:** Can influence reaction kinetics and efficiency of the EC process.

2.4.2 Chemical parameters:

1. **COD:** Indicates the organic load and is a critical parameter for organic pollutant removal. The current permissible limit is 75 mg/l for wastewater limit values applicable to the irrigation of any land or property up to 2000 m³ [39].

2. **BOD:** Assesses biodegradable organic matter levels, useful for determining downstream impacts
3. **Nutrients:**
 - a. Nitrate (NO_3^-) and Nitrite (NO_2^-): Key indicators of nitrogen species in water.
 - b. Phosphate (PO_4^{3-}): A common nutrient in polluted water.
4. **Heavy metals:** Specific metals to be selected depending on expected pollutants (e.g. Fe, Cu, Zn, Pb, Cd, Cr, Ni).
5. **Hardness-related ions:** A representation of calcium and magnesium salts, barium, strontium, and Silica.

2.4.3 Microbial indicators:

1. **E. coli or total coliforms:** To evaluate the microbial deactivation capacity of the EC process.

2.4.4 Additional characteristics to consider:

1. **Oil and grease:** Particularly relevant if wastewater comes from industrial sources with oil contamination.
2. **Specific contaminants:** If applicable, measure unique pollutants present in the wastewater, such as specific organics or other industry-related contaminants.

2.4.5 Removal efficiency indicators:

For each parameter, the removal efficiency is calculated using the formula:

$$\text{Removal efficiency}(\%) = \frac{\text{Initial value} - \text{Final value}}{\text{Initial value}} \cdot 100 \quad (20)$$

2.5 EC kinetics

The kinetics of pollutant removal through EC can be analysed by estimating the coagulant dose based on adsorption phenomena. Kinetic studies are essential to understand how the pollutant removal rate depends on operational parameters. The amount of pollutant removed (Q_t) at time (t) is calculated using the equation:

$$Q_t = \frac{(C_0 - C_t)v}{w} \quad (21)$$

where C_0 (mg/L) is the initial pollutant concentration; C_t (mg/L) is the pollutant concentration at time (t) in the aqueous phase; v is the solution volume (L), and w is the mass of the metal hydroxide formed.

The adsorption mechanism of pollutants onto in situ-generated metal hydroxides can be further described using various kinetic models such as a pseudo-first-order or pseudo-second-order model, Avrami model or and the Elovich model [24].

2.6 Operating cost analysis

With the EC process, it is essential to achieve the highest removal rate with minimum operational costs. The cost of treatment with EC depends on the electrode cost, electrical energy cost, sludge disposal and fixed cost. The operating cost can be calculated by the equation (22):

$$\text{Operating cost} = aC_{\text{Energy}} + bC_{\text{Electrode}} + cC_{\text{Labour}} + dC_{\text{Maintenance}} \quad (22)$$

where C_{Energy} and $C_{\text{Electrode}}$ are the amount of electricity and electrode consumed per litre of effluent treated, which is experimentally achieved. Labour, maintenance, disposal and dewatering of sludge costs are assumed to be fixed and might be left out of some equations to standardise comparing operational costs when investigating operational parameter optimisation [32].

The amount of electrical energy consumed per litre of effluent can be calculated by the equation (23):

$$C_{\text{Energy}} = U \cdot I \cdot \frac{RT}{V} \quad (23)$$

For electrode consumption, it would be:

$$C_{\text{Electrode}} = \frac{I \cdot RT \cdot M}{n \cdot F \cdot V} \quad (24)$$

where I = electrical current (A), U = voltage (V), RT = reaction time of treatment (s), M = molecular mass of Al (26.98 g/Mol) and Fe (55.84 g/Mol), Z = amount of electron moles (3 for Al(III) and 2 for Fe(II)), F = Faraday's constant (96500 C/Mol), V volume of effluent (L), a = electricity cost (R/kWh), b = cost of Al or Fe (R/kg), c = labour rate (R/h), d = maintenance rate (R/h) [13].

Additionally, the amount of electrode consumed (E_{EL}) per litre of effluent treated can be calculated based on measured values by the given equation:

$$E_{\text{EL}} = \text{Initial weight of electrode} - \text{Final weight of electrode} \quad (25)$$

From studies such as those conducted by Kraljic et al. [10], it is seen that variations in solution conductivity play a significant role in the energy consumption and removal efficiency. Here, they mentioned that in a textile wastewater treatment study, the addition of NaCl (1000 to 4000 $\mu\text{S/cm}$)

did not influence the removal efficiency but lowered the energy consumption from 2.2 to 0.75 kWh [10].

Kraljic et al. [10], changed the conductivity from 2.3 S/cm to 23.7 S/cm, and the specific energy consumption changed from 8.5 to 1.6 kWh/kg after 35 minutes of treatment [10].

2.7 Sludge characteristics

The characteristics of the sludge produced during EC are a critical consideration due to the significant costs associated with its treatment and disposal, particularly when generated chemically. Sludge from EC systems is influenced by electrode configurations and materials, and its properties can be analysed using methods such as Fourier-transform infrared spectroscopy (FTIR), X-ray diffraction (XRD), X-ray photoelectron spectroscopy (XPS), scanning electron microscopy with energy-dispersive X-ray spectroscopy (SEM-EDS), and energy-dispersive X-ray spectroscopy (EDX).

The volume of sludge produced by an EC treatment is impacted by flow rate, current density and initial pollutant concentration. An increase in volume will be driven by lower flow rates, higher current densities and higher initial pollutant concentrations [27]. Higher sludge volumes at higher current densities can also be due to excessive formation of metal hydroxides. Furthermore, an increase in initial TSS concentration will lead to slower sludge velocity, i.e. longer settling time required. The volume of sludge can be calculated in kilograms per cubic meter (sludge density), where the “wet” sludge weight can be compared to the dried sludge weight to determine the water retention of the substance.

Sludge formed with the use of Al electrodes primarily contains amorphous Al hydroxide and/or Al oxyhydroxide. In contrast, whilst Fe electrodes produce both crystalline phases, such as magnetite, and poorly crystalline phases, such as Fe oxyhydroxides and lepidocrocite. When Al and Fe electrodes are combined, the resulting sludge also includes products like mansfeldite, diaspora, and Fe oxide. Interestingly, the crystallinity of Fe-based species decreases when substituted with Al, due to interactions between the two metals [27].

In textile wastewater treatment, XRD analysis of the sludge produced with Fe electrodes was identified as maghemite as the only by-product, derived from ferric hydroxide ($\text{Fe}(\text{OH})_3$). Similarly, during fluoride removal using Al electrodes, XRD and EDX analyses revealed an amorphous Al coagulant with fluoride, Al, and oxygen adsorption. The broad and shallow diffraction peaks indicated a poorly crystalline nature. Additional elements detected in the sludge were attributed to electrolytic adsorption, experimental chemicals, or material from the electrodes themselves [27].

FTIR spectra of Al-electrode-derived sludge from dye removal (Direct Red 81) indicated adsorption mechanisms, with bands at 3452 cm^{-1} associated with OH group vibrations, and bands at 1500 and 1411 cm^{-1} pointing to azo group ($-\text{N}=\text{N}-$) presence. Other bands were linked to sulfonate group (SO_3^-) stretching vibrations, confirming dye adsorption on Al hydroxide flocs. However, the precise nature of adsorption (physisorption or chemisorption) remains unclear [ref].

Fluoride removal via EC also results in sludge containing Al fluoride hydroxide complexes and Al hydroxide, as observed by XRD analysis. The settling behaviour of sludge during wastewater treatment varies with electrode material; Al electrodes yield a sludge volume index of $0.207\text{--}0.310\text{ L/g}$, while Fe electrodes produce a heavier, more compact sludge with a lower volume index of $0.081\text{--}0.091\text{ L/g}$. Hydrogen gas evolution during EC enhances particle flotation, which can reduce sludge settling velocity [ref].

Lastly, combining EC with electrode watering techniques reduces the sludge water content to 55%, compared to 78% achieved with pressure filtration alone, offering a more efficient dewatering strategy for sewage sludge [13]

2.8 Comparison of implemented EC studies

An evaluation of a variety of existing EC treatment plants and their efficiencies, summarising key findings, is shown as a box and whisker plot in Figure 16. The aim is to identify the theoretically best combination of parameters from the studies conducted in references [10–24, 27–30, 32–33]. The detailed outcomes will be discussed in the following sections.

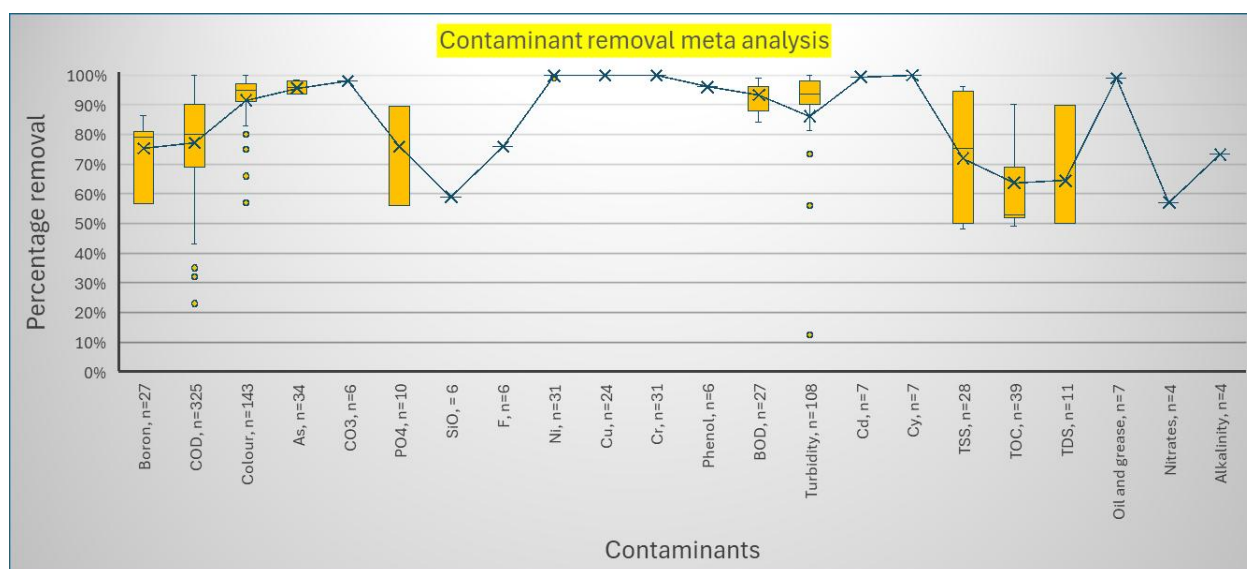


Figure 14: Contaminant removal - Meta analysis

2.8.1 Interpretation of contaminant removal efficiency with EC

The box plot shown in Figure 14, presents a meta-analysis of contaminant removal percentages using EC, highlighting its effectiveness for various water pollutants. The median removal efficiencies, interquartile ranges, and data spread indicate the consistency and reliability of EC across different contaminants.

Furthermore, from the data used to generate the box and whisker plot, shown in Figure 14, it can be seen that there are certain operational parameters that were widely studied due to the substantial influence it has on the EC treatment effectiveness[10–24, 27–30, 32–33]. These operational parameters are HRT, current density, inter-electrode distance, electrode material selection, and initial pH. These parameters will be evaluated for the contaminants that were the most effectively removed.

From the literature available, Al and Fe electrodes are the most used, due to availability and affordability. The same material selection is normally used for the cathode and anode, but in a few studies a combined material selection was used.

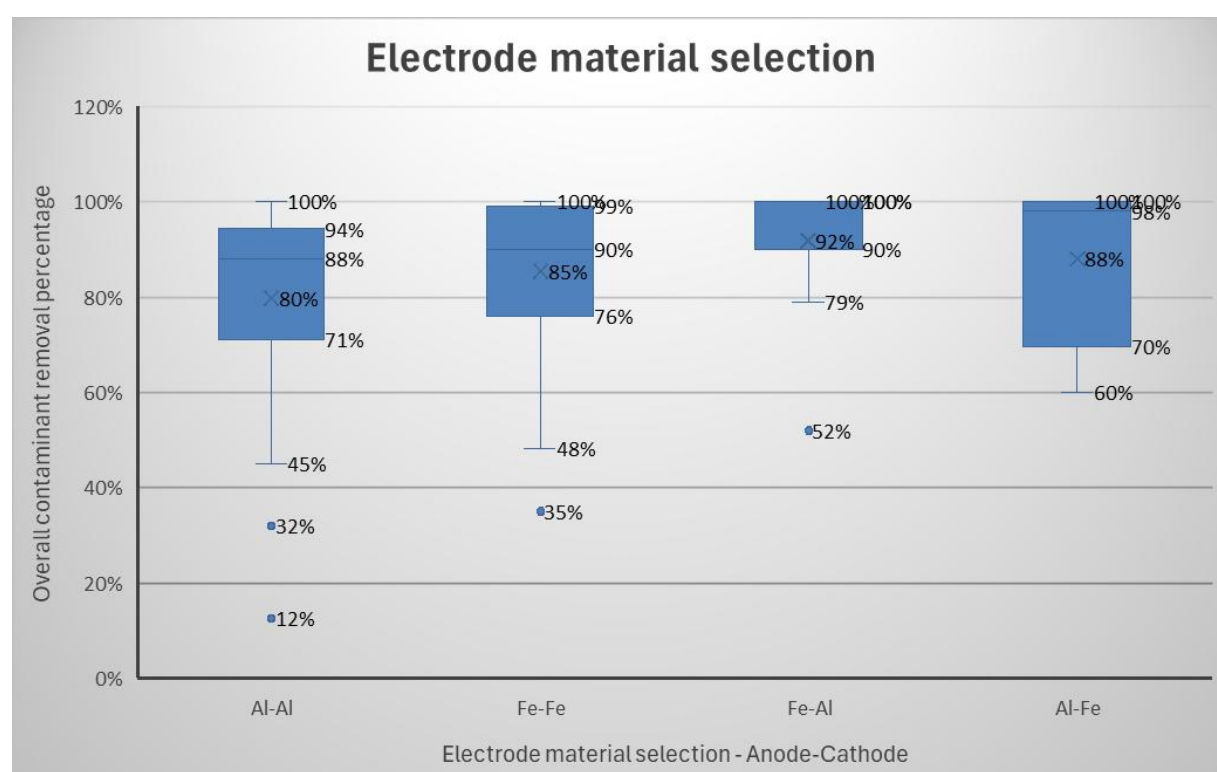


Figure 15: Optimal electrode material selection [10–24, 27–30, 32–33]

From Figure 15, it seems that the Fe-Al option had the highest overall removal rate, but only three studies evaluated this option, with more than ten studies that investigated the Fe-Fe and Al-Al selection. From these electrode pair options, based on affordability and material

availability, it was found that the Fe-Fe combination is more efficient at the removal of a wide range of contaminants. An additional benefit for having the electrodes of the same material is that the rate of dissolution will be similar throughout the whole EC reactor, which will mitigate the risk that some electrodes will have to be replaced sooner than others.

From Figure 14, EC shows the highest and most consistent removal efficiencies for boron, COD, colour, arsenic (As), BOD, turbidity, and oil & grease. However, COD, colour, BOD and turbidity were investigated more frequently in the literature available; therefore, the focus of the meta-analysis will be placed on these contaminants. These contaminants exhibit high median values (close to or above 80%) with narrow variability, indicating reliable performance.

2.8.1.1 Chemical Oxygen Demand (COD) – High Removal (~80–90%)

The boxplot shows a high median removal efficiency, with some variability, due to differences in wastewater composition.

Interpretation:

- COD reduction is achieved through electrostatic destabilisation of organic matter, coagulation, and electro-oxidation at the anode.
- The presence of some lower outliers suggests that in certain cases, removal is less effective due to complex or highly soluble organic pollutants.

2.8.1.2 Colour – Very high and consistent removal (~85–95%)

The median removal percentage is very high, with minimal variation.

Interpretation:

- EC effectively removes colour by neutralising and destabilising charged dye molecules, allowing them to aggregate and settle.
- The narrow IQR suggests that colour removal is consistently high across different wastewater sources.

2.8.1.3 Biological Oxygen Demand (BOD) – High removal (~90–100%)

Consistently high median removal (~95%), with minimal variability.

Interpretation:

- BOD is reduced through coagulation of biodegradable organic matter, flotation of microbial contaminants, and electrochemical oxidation.
- The near-100% removal suggests that EC is highly effective for organic load reduction, making it a strong alternative for biological treatment.

2.8.1.4 Turbidity – High and stable removal (~90–100%)

The boxplot shows a very high median (~95%) with a narrow spread.

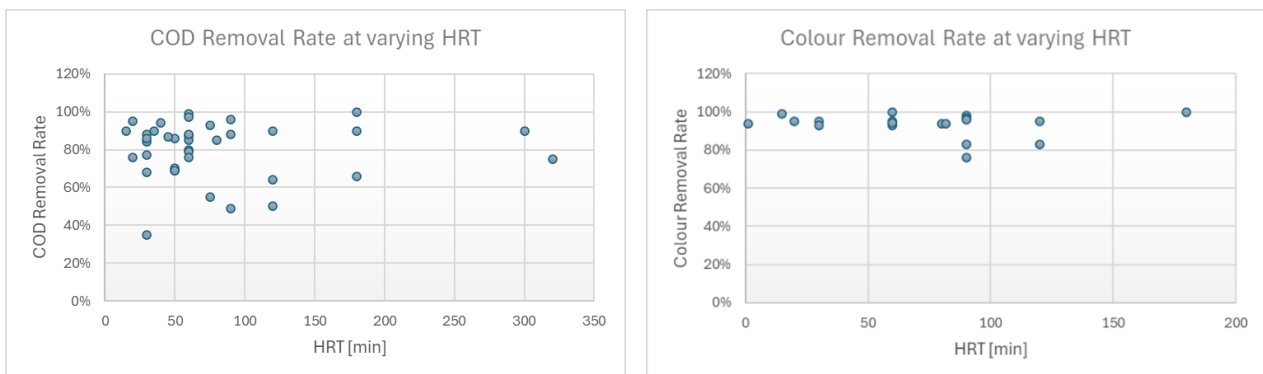
Interpretation:

- Turbidity, caused by suspended solids, is efficiently reduced through charge neutralisation and flocculation, forming settleable aggregates.
- The narrow range indicates robust performance across different wastewater types.

2.8.1.5 Overall interpretation

For the operational parameters investigated through the meta-analysis, the following ranges must be further optimised:

- HRT:
 - The optimal HRT range, based on the meta-analysis, is 20 to 80 minutes, as can be seen in Figure 16. The flow rates for a CFR reactor can be adjusted to vary the HRT to assist with the removal of these components.



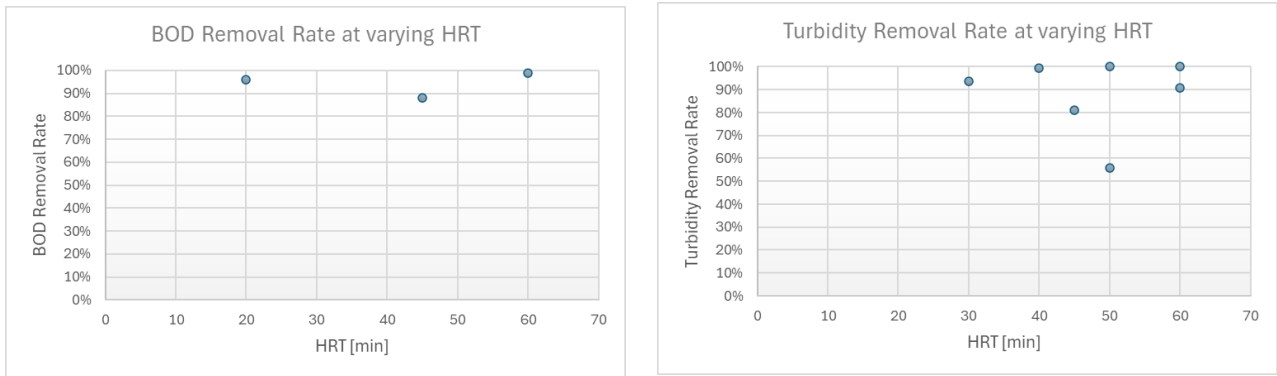


Figure 16: Contaminant removal vs HRT with Fe-Fe electrodes [10–24, 27–30, 32–33].

- Current density:
 - The optimal current density range, based on the meta-analysis, is 2.5 to 30 mA/cm² as can be seen from Figure 17.

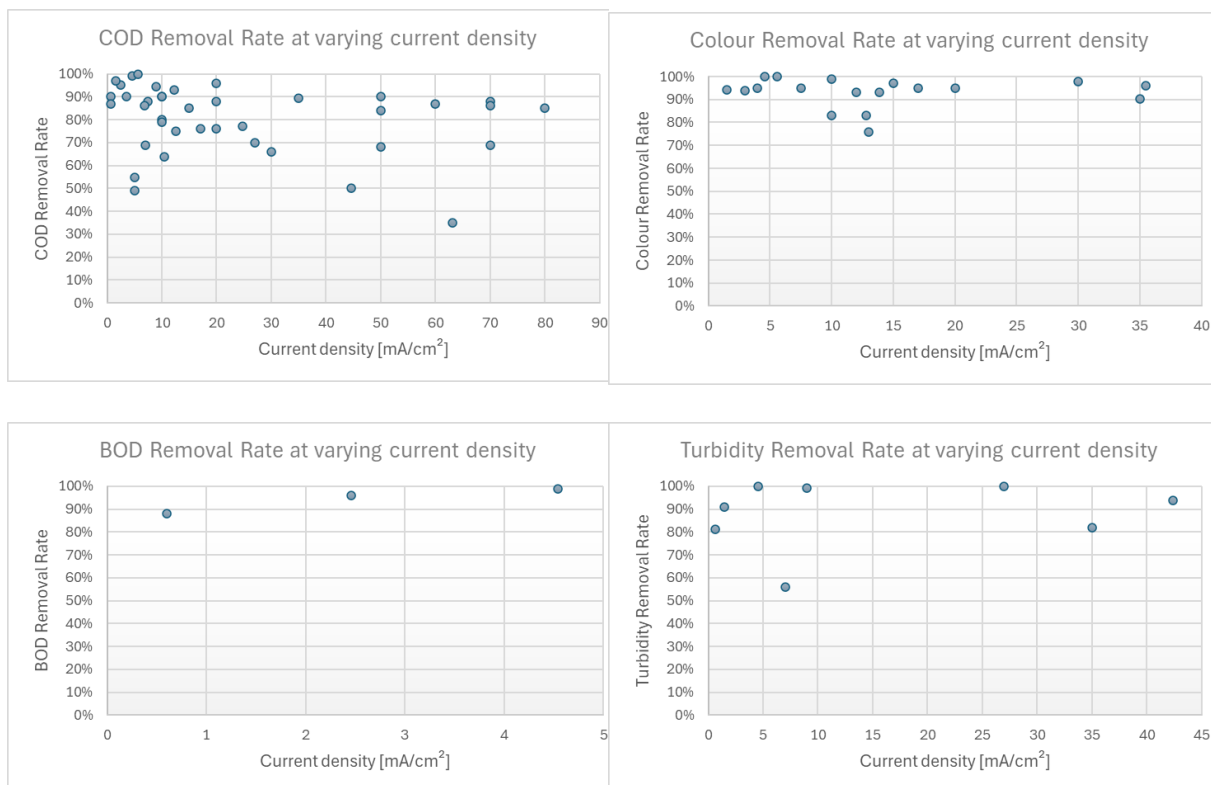


Figure 17: Contaminant removal vs current density with Fe-Fe electrodes [10–24, 27–30, 32–33]. Inter-electrode distance

- The optimal range for inter-electrode distance is 5 to 30 mm, but 10 mm is ideal for the effective treatment of a wide range of contaminants and varying operational parameters. The performance of varying inter-electrode distances can be seen in Figure 18. This parameter has been described in detail and does not have to be further optimised through experimentation.

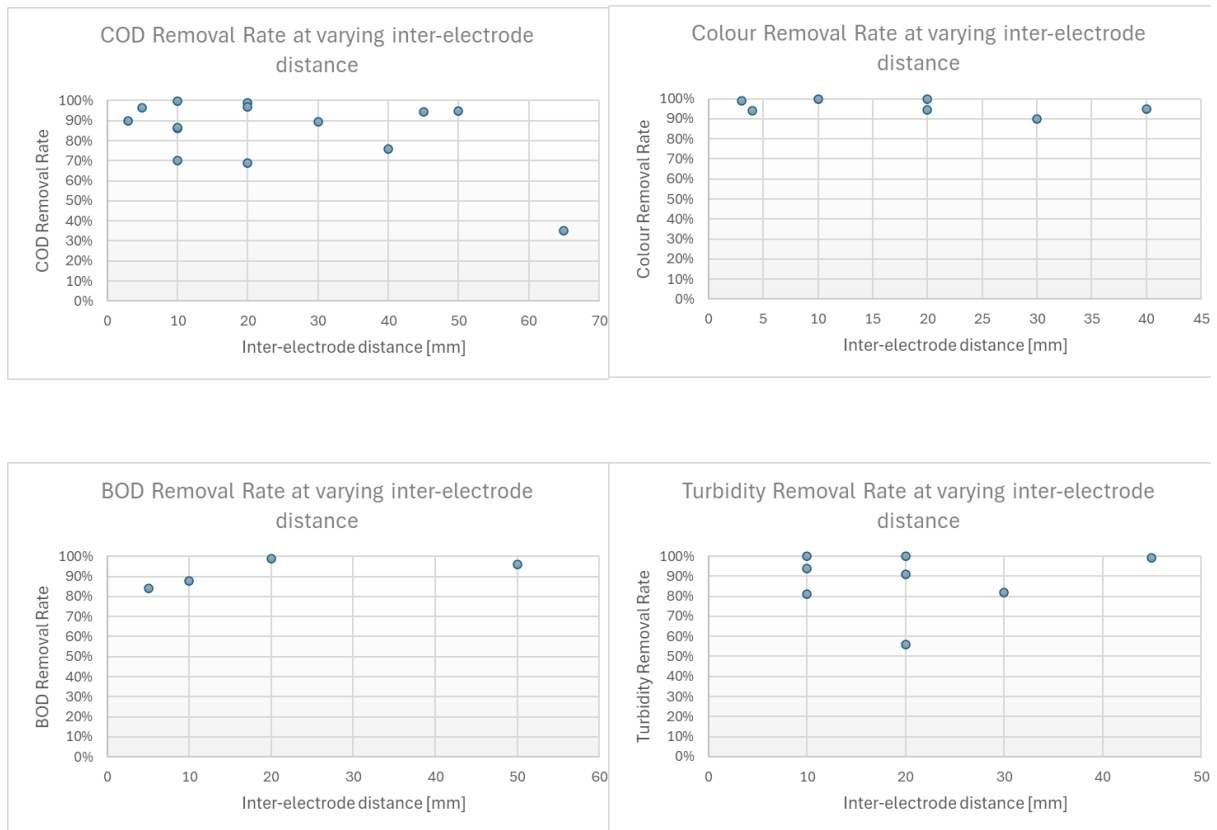
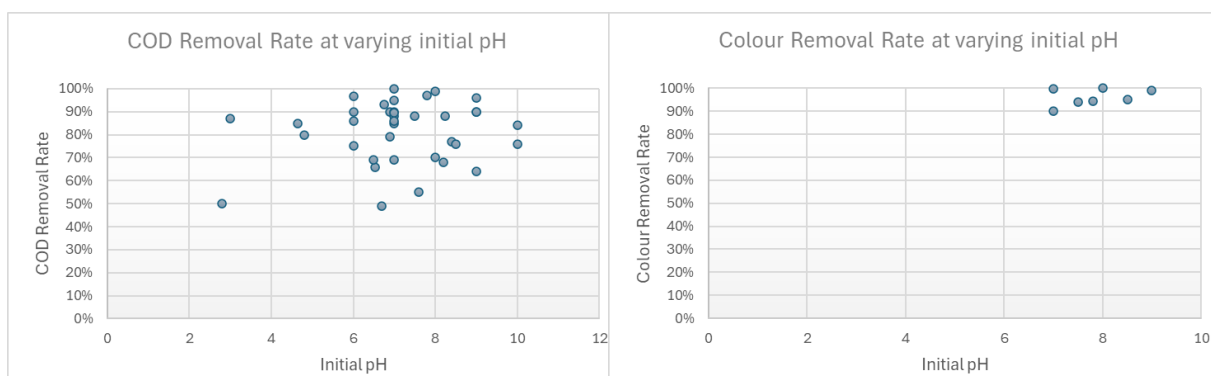


Figure 18: Contaminant removal vs inter-electrode distance with Fe-Fe electrodes [10–24, 27–30, 32–33]. Initial pH:

- The optimal initial pH range is 6 to 8, for the metal electrode to dissolve into the preferred coagulant species which is not too soluble. The success of the above-mentioned range can be seen in Figure 19



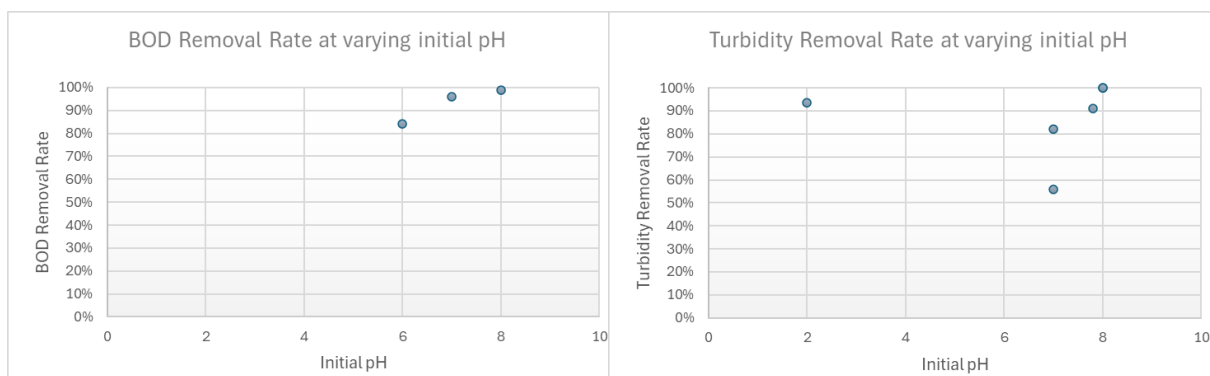


Figure 19: Contaminant removal vs initial pH with Fe-Fe electrodes [10–24, 27–30, 32–33].

The contaminants with the highest and most consistent removal (BOD, turbidity, colour, COD) benefit from electrocoagulation's combined mechanisms, particularly coagulation, oxidation, flotation, and adsorption.

Minimal outliers and narrow IQRs indicate that EC is a reliable technology for these pollutants, with minimal variation across different conditions.

2.8.2 Contaminant description

Each contaminant presented in Figure 14, is briefly described with a short description of its negative implications in excessive concentrations in wastewater:

- **Boron – B**
 - Description: A trace element found in natural water and industrial effluents, especially from detergents and fertilisers.
 - Negative implication: Toxic to aquatic life and plants at high concentrations, affecting crop irrigation and soil health.
- **Chemical oxygen demand – COD**
 - Description: Measures the amount of oxygen required to oxidise organic and inorganic matter in water.
 - Negative implication: High COD indicates elevated levels of pollutants, leading to oxygen depletion and harm to aquatic organisms
- **Colour**
 - Description: Caused by dissolved organic matter, metals, or industrial dyes in water.
 - Negative Implication: Reduces water quality, affects aesthetics, and can indicate the presence of toxic substances.

- **Arsenic – As**
 - Description: A toxic metalloid that can enter wastewater from mining, industrial processes, and pesticides.
 - Negative Implication: Highly toxic to humans and aquatic life, causing cancer, skin lesions, and organ damage.
- **Carbonate - CO_3^{2-}**
 - Description: A component of alkalinity that affects water pH and buffering capacity.
 - Negative Implication: Elevated levels can lead to scaling in pipes, interfere with water treatment, and disrupt aquatic ecosystems.
- **Phosphate - PO_4^{3-}**
 - Description: Commonly from fertilisers, detergents, and sewage.
 - Negative Implication: Causes eutrophication, leading to excessive algae growth, oxygen depletion, and fish kills.
- **Silicate - SiO_4^{4-}**
 - Description: Found in natural water from rock weathering and industrial effluents.
 - Negative Implication: Causes scaling and abrasion in industrial equipment and can interfere with water treatment processes.
- **Fluoride – F^-**
 - Description: Naturally occurring or from industrial discharge and wastewater treatment chemicals.
 - Negative Implication: High concentrations are toxic, causing dental and skeletal fluorosis in humans and harm to aquatic life.
- **Nickel – Ni**
 - Description: Heavy metal found in wastewater from electroplating, mining, and alloys.
 - Negative Implication: Toxic to aquatic organisms, bioaccumulates in food chains, and can cause lung and kidney damage in humans.
- **Copper – Cu**
 - Description: Released from plumbing, industrial discharge, and agricultural runoff.
 - Negative Implication: Toxic to fish and aquatic organisms at elevated levels, affecting enzyme activity and reproduction.
- **Chromium – Cr**
 - Description: Heavy metal used in metal plating, dyes, and industrial processes.

- Negative Implication: Cr(VI) is highly toxic and carcinogenic, causing DNA damage and ecosystem disruption.
- **Phenol**
 - Description: An organic compound from petroleum, pharmaceuticals, and chemical manufacturing.
 - Negative Implication: Highly toxic to aquatic life, affects drinking water taste and odour, and poses human health risks.
- **Biological oxygen demand – BOD**
 - Description: Measures the amount of oxygen consumed by microorganisms breaking down organic matter.
 - Negative Implication: High BOD depletes oxygen levels in water, leading to dead zones and aquatic life suffocation.
- **Turbidity**
 - Description: Cloudiness caused by suspended particles like silt, organic matter, and microorganisms.
 - Negative Implication: Reduces light penetration, affects aquatic ecosystems, and can carry pathogens.
- **Cadmium – Cd**
 - Description: Toxic heavy metal from battery waste, industrial discharge, and fertilisers.
 - Negative Implication: Bioaccumulates in aquatic organisms, causing kidney damage, bone disease, and carcinogenic effects.
- **Cyclohexyl – Cy**
 - Description: A hydrocarbon used in chemical synthesis and plastic production.
 - Negative Implication: Toxic to aquatic life and can contaminate drinking water sources.
- **Total suspended solids – TSS**
 - Description: Particles in water, including silt, organic debris, and microorganisms.
 - Negative Implication: Increases turbidity, clogs filters, and affects aquatic habitats.
- **Total organic carbon – TOC**
 - Description: Represents organic matter in water from natural sources, wastewater, and industrial discharge.
 - Negative Implication: Contributes to oxygen depletion and can form harmful disinfection by-products.

- **Total dissolved solids – TDS**
 - Description: Includes inorganic salts, minerals, and metals dissolved in water.
 - Negative Implication: Elevated levels affect water taste, scaling in pipes, and can harm aquatic organisms.
- **Oil and grease**
 - Description: Petroleum-based or organic fats and oils from industrial and household waste.
 - Negative Implication: Coats aquatic surfaces, suffocates marine life, and clogs wastewater treatment systems.
- **Nitrates – NO_3^-**
 - Description: Commonly from fertilisers, wastewater, and agricultural runoff.
 - Negative Implication: Causes eutrophication, contributes to oxygen depletion, and poses health risks like methemoglobinemia (blue baby syndrome).
- **Alkalinity – Total alkalinity as CaCO_3**
 - Description: Represents water's capacity to neutralise acids, from bicarbonates, carbonates, and hydroxides.
 - Negative Implication: High alkalinity can cause scaling, affect aquatic life, and interfere with industrial water treatment.

2.9 EC treatment compared with conventional coagulation treatment

Naje et al. [13], mentioned that in comparison with EC, the cost of a conventional treatment is that it is almost twice and, in some cases, up to 3.2 times the cost of EC for the treatment of textile wastewater.

From Figure 20, a visual comparison between EC and CC is shown regarding the removal efficiency of specific contaminants. The graph ranges in increments of 20% pollutant removal rate, and EC is more effective for a wide range of contaminant removal [13]. In some cases, the removal efficiency is similar, like with colour, the main difference could be the type of chemical coagulant that is used. Figure 20 compares only the contaminant removal efficiency but does not address the operational cost difference.

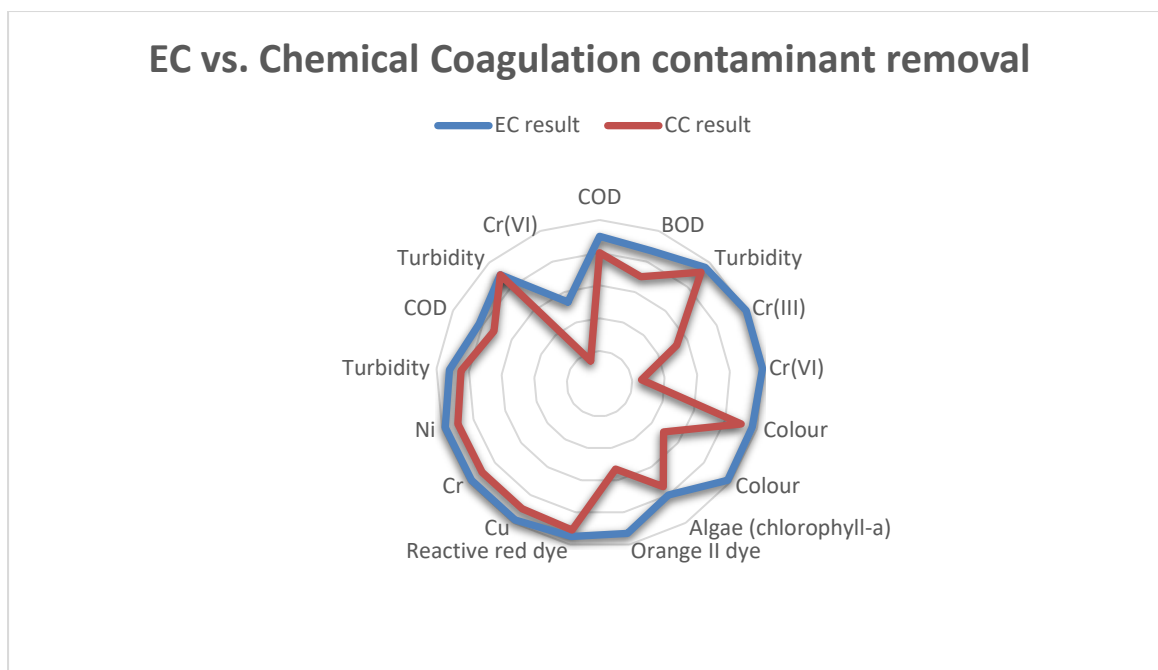


Figure 20: Pollutant removal efficiency - EC vs. Chemical Coagulation [13].

2.10 Combination of EC with other technologies

Other technologies have been combined with EC as a treatment process, besides the standard filtration, flotation, and sedimentation sludge separation technologies. These combinations have led to higher removal efficiencies as compared to a single treatment method. EC combined with other methods offers a safe and effective way for the removal of pollutants. However, few studies are available on the combination of EC with other electrochemical technologies, such as electrooxidation (EO).

The EO process is an electrochemical treatment that is commonly used to treat effluents that contain refractory organic compounds, such as textile effluent, landfill leachates, olive oil wastewater, sewage sludge and tannery effluent [13].

2.11 Literature gaps and justification for the present study

The literature confirms that EC is a promising treatment method for wastewater containing suspended solids, organic matter, colour, and microbial contamination. Previous studies have shown that several operating variables, including current density, treatment time, pH, inter-electrode distance, conductivity, electrode material, and reactor configuration, influence EC performance. The literature further shows that high removal efficiencies can be achieved for pollutants such as COD, turbidity, and colour under a wide range of operating conditions.

Despite this established knowledge, there are several gaps. Firstly, the reported operating conditions vary considerably between studies, making direct comparison difficult and limiting the identification of practically optimal parameter combinations. Secondly, many previous investigations focus on batch systems, whereas fewer studies present structured optimisation of continuous-flow EC systems under controlled operating conditions. Thirdly, although the influence of individual parameters is well recognised, there is still a need for empirical models that quantify the combined effect of selected variables under a defined reactor configuration. In particular, the interaction between HRT and current density remains highly system-specific and cannot be assumed from literature alone.

Another limitation in the literature is that treatment performance is often reported without sufficient consideration of the trade-off between pollutant removal and electrical energy input. From an engineering perspective, process optimisation should not only target maximum removal, but also practical operating efficiency. Furthermore, while pH and conductivity are frequently discussed in EC studies, a need remains for optimisation studies that evaluate process performance under realistic wastewater conditions without chemically altering the feed before treatment.

The present study was therefore designed to address these gaps by evaluating the effect of HRT and current density in a laboratory-scale continuous-flow EC reactor treating slaughterhouse wastewater. The study further applies response surface methodology to develop empirical models that can assist with identifying operating conditions that balance removal efficiency and energy demand. In this way, the study contributes a focused optimisation framework that is experimentally grounded and relevant to practical EC system development.

2.12 Conclusion

This literature study has provided an evaluation of EC as an emerging wastewater treatment technology, emphasising its advantages over conventional methods, particularly its ability to remove pollutants without the addition of external chemicals. The analysis of EC mechanisms highlights the role of sacrificial electrodes, in-situ coagulant generation, and electrostatic destabilisation in contaminant removal. Furthermore, key process parameters have been reviewed, identifying optimal ranges for independent variables such as current density, electrode configuration, and power supply type, as well as dependent variables like pH and electrical conductivity.

Despite significant advancements in understanding EC performance, further optimisation is needed in two critical areas to enhance treatment efficiency and operational feasibility: flow rate (hydraulic retention time) and current density.

- **Flow Rate/ HRT:** While literature suggests that longer retention times improve pollutant removal, excessive treatment duration leads to increased energy consumption and operational costs. Optimising flow rate is essential to balance treatment efficiency with economic feasibility, particularly for continuous-flow systems.
- **Current Density:** A key determinant of treatment effectiveness, current density must be carefully controlled to maximise pollutant removal while minimising excess energy consumption, electrode passivation, and unnecessary coagulant production. Further research is needed to define optimal current density ranges for different wastewater matrices.

To fully harness the potential of EC, research should focus on optimising these two parameters and observing the initial pH to determine if adjustment is required in different wastewater applications. Additionally, pilot-scale studies are necessary to establish the scalability and economic viability of EC in full-scale operations. By refining hydraulic conditions, electrical parameters, and post-treatment separation, EC can be further developed into a highly efficient and sustainable wastewater treatment technology.

CHAPTER 3 METHODOLOGY

This chapter outlines the theoretical methodology and design considerations adopted to support the development of an electrocoagulation (EC) treatment system for the removal of contaminants from abattoir wastewater. The methodology integrates a detailed analysis of the target pollutants, reactor design considerations, the selection and justification of fixed and variable process parameters, and the theoretical and empirical modelling techniques employed. As described in Section 1.8, this chapter does not cover experimental procedures or validation results, which are addressed separately in Chapter 4. The purpose of this chapter is to define the systematic framework that informs the design, optimisation, and theoretical understanding of the EC process.

3.1 Discharge limits for product water quality

In South Africa, treated wastewater must meet specific quality standards depending on its disposal or reuse route. Regulatory limits are set by national guidelines and municipal by-laws to protect public health and the environment.

If treated effluent is discharged into surface water (rivers or streams), it must comply with the Department of Water and Sanitation's General Effluent Standards [40] (with even stricter "Special Limits" in sensitive areas). Key parameters include organic load, solids, and pathogens. For example, general discharge limits require faecal coliform bacteria (e.g. *E. coli*) to be ≤ 1000 counts/100 mL, Chemical Oxygen Demand (COD) ≤ 75 mg/L, and pH between 5.5 and 9.5 [40]. These standards ensure that the water is neither overly polluted with organics nor too acidic/alkaline. The nutrient content is also regulated to prevent eutrophication; ammonia (as N) is limited to ~ 6 mg/L and nitrate/nitrite (as N) to ~ 15 mg/L in general standards [40]. Similarly, TSS must typically be below 25 mg/L to avoid sludge deposits [40]. The special limits for surface discharge are even tighter (e.g. no detectable faecal coliforms and COD ≤ 30 mg/L in certain sensitive catchments) [40]. These national limits (issued under the National Water Act's General Authorisations) set the benchmark for any treated effluent being released into rivers or dams.

For discharges into municipal sewer systems, water quality requirements are governed by municipal bylaws. Industrial effluents (like abattoir wastewater) must be pre-treated to meet local sewer acceptance criteria so as not to upset municipal wastewater treatment works.

Table 4: South African discharge limits for product water quality [40].

South African discharge limits for product water quality		
Discharge/Reuse Route	Parameter	Limit
Surface Water Discharge	Faecal coliform bacteria (e.g., E. coli)	≤ 1000 counts/100 mL (General Standard); nondetectable (Special Limit, sensitive areas)
	COD	≤ 75 mg/L (General Standard); ≤ 30 mg/L (Special Limit, sensitive areas)
	pH	5.5–9.5
	Ammonia (as N)	≤ 6 mg/L
	Nitrate/Nitrite (as N)	≤ 15 mg/L
	Total Suspended Solids (TSS)	≤ 25 mg/L
	Regulatory Basis	Department of Water and Sanitation, National Water Act (General Authorisations)
Municipal Sewer Discharge	pH	6.0–10.0 (to protect sewer infrastructure)
	COD	≤ 5000 mg/L (may require further reduction depending on municipal requirements)
	Suspended Solids (non-organic)	≤ 100 mg/L
	Oils and Grease (solvent extractable)	≤ 500 mg/L
	Microbiological Limits	Not typically specified (sewage already contains pathogens, but extremely high loads avoided)
	Regulatory Basis	Municipal by-laws (e.g., Cape Town, other major cities)
Reuse/Irrigation	Faecal coliform bacteria (e.g., E. coli)	≤ 0 detectable/100 mL (unrestricted use, e.g., food crops, parks); ≤ 1000/100 mL (restricted use, e.g., pastures)
	COD	≤ 75 mg/L (aligns with surface discharge limits)
	TSS	≤ 25 mg/L
	pH	5.5–9.5
	Electrical Conductivity (salinity)	≤ 70 mS/m increase above source water
	Sodium Adsorption Ratio (sodicity)	Must meet guidelines to protect soil structure
	Phosphate (as P)	≤ 10 mg/L
	Regulatory Basis	Department of Health guidelines (1978), Department of Water and Sanitation, WHO

Typical bylaws of municipalities stipulate allowable ranges for pH and maximum concentrations for organics, solids, and oils. For instance, a city can impose regulations requiring an effluent pH to be within 6.0–10.0 (to protect sewer infrastructure from corrosion or scale) and limit COD to about 5000 mg/L [41], while another city may impose different requirements. Excessively high COD/BOD concentrations in sewer discharges can overload biological treatment processes, so industries may incur surcharges or be required to reduce COD closer to domestic strength. Suspended solids and grease are also constrained – a typical limit is ≤ 100 mg/L for non-organic suspended matter and in the order of a few hundred mg/L for fats, oils, and grease [41]. For example, oils and grease measurable by solvent extraction might be capped at around 500 mg/L to prevent clogging and maintenance issues. These by-law limits ensure that the industrial effluent will not harm sewers, pose risks to operators, or inhibit the downstream municipal treatment; any company discharging above these limits must implement further treatment steps [41]. While microbiological limits are not usually specified for sewer inflow (since sewage is expected to contain pathogens), abattoirs must still exercise caution, as extremely high pathogen loads could pose occupational hazards. Overall, compliance with municipal effluent standards is a legal requirement before releasing wastewater into the sewer network.

Finally, for treated wastewater, intended for reuse or irrigation, additional criteria are required. The South African guidelines (originally set by the Department of Health in 1978 and still in use [42]) emphasise microbiological safety for irrigation water. For unrestricted use (e.g. irrigating foods eaten raw or public parks), no or extremely low faecal coliforms are recommended (often zero detectable *E. coli* per 100 mL) [42]. Where reuse is for less sensitive purposes (e.g. pasture or certain industrial uses), the guidelines relax somewhat – historically, a limit of ≤ 1000 *E. coli*/100 mL was proposed as an upper bound for treated effluent reuse in agriculture [43], aligning with World Health Organisation recommendations. Recent Department of Water and Sanitation requirements mirror the general surface discharge standards for large-scale irrigation projects: if up to 2000 m³ of effluent is applied per day, it must meet the same COD (75 mg/L), TSS (25 mg/L), pH (5.5–9.5) and coliform (1000/100 mL) limits as for direct discharge [44]. This ensures that irrigation with treated water does not contaminate soils or underlying water resources. Other parameters important for irrigation quality include salinity and sodicity (the presence of a high proportion of sodium ions relative to other cations in the soil): the guidelines set limits on electrical conductivity (e.g. not more than 70 mS/m increase above source water) and sodium adsorption ratio to protect soil structure [44]. Nutrients like phosphates are also controlled (e.g. ≤ 10 mg/L P) to avoid over-fertilisation or algal growth on land [44]. In summary, the product water quality from the

treatment process must be such that it satisfies the relevant discharge or reuse standards – whether the goal is safe release to the environment, feeding into a municipal treatment system, or beneficial reuse for irrigation. These regulatory limits define the performance targets for the treatment methodology in this study.

3.2 Contaminants selected for removal efficiency testing

Based on the broad contaminant categories identified in the section [16] for abattoir wastewater, a specific set of pollutants was chosen to evaluate the EC treatment performance. Typically, abattoir effluent contains a complex mixture of contaminants – notably a high organic load, suspended solids (including fats and proteins), and microbial pathogens. To represent each major category (physical, chemical, and biological pollution), and to align with the regulatory criteria shown in the section 3.1, table 4, the following parameters were selected for removal efficiency testing: COD, electrical conductivity, turbidity, colour, and faecal coliform bacteria (*E. coli*). Each of these was chosen for its relevance to abattoir waste characteristics and because reducing these parameters is crucial to meet discharge standards.

Each selected parameter corresponds to a major pollution aspect of abattoir wastewater and is tied to regulatory thresholds discussed in the section 3.1. By focusing on:

- COD
- turbidity
- colour
- electrical conductivity
- *E. coli*,

This study covered the essential measures of treatment efficacy for:

- Organic load reduction
- Physical impurity removal
- Pathogen inactivation

The choice of these contaminants for removal efficiency testing is justified by both the characteristics of the wastewater and the compliance targets that the treated water must meet [44].

In the methodology, analytical tests will be conducted for each parameter on influent and effluent samples to quantify the performance of the electrocoagulation process. This comprehensive set of indicators will demonstrate whether the EC-treated abattoir wastewater

can achieve the required product water quality for discharge to the environment, connection to municipal sewer, or potential reuse in irrigation. By linking treatment outcomes to the official discharge limits, the study ensured that its evaluation of the EC process is grounded in real-world standards and the operational goals of sustainable wastewater management.

3.3 Fixed design parameters of the lab-scale EC CFR reactor

The EC reactor is mounted on a lab-scale skid along with all necessary auxiliary components. The system is designed for simplicity and functionality, including only the essential elements for continuous operation.

3.3.1 Material selection

Fe and Al are the two most common electrode materials employed in EC, due to their wide availability, low cost, and proven effectiveness in removing a range of pollutants. For the treatment of high-strength organic wastewater such as abattoir effluent, Al electrodes were selected as the optimal material in this study.

Al electrodes have consistently demonstrated superior removal efficiencies for COD and turbidity, compared to Fe, making them particularly well-suited for organic-rich waste streams. This advantage is primarily attributed to the generation of Al^{3+} ions at the anode, which hydrolyses to form abundant, gelatinous Al hydroxide flocs. These flocs effectively capture and bind organic contaminants and suspended solids, improving overall pollutant removal performance.

In comparison to Fe electrodes, Al offers several additional benefits. It typically produces less dense, more easily dewatered sludge, reducing the operational burden of sludge handling. Furthermore, Al electrodes avoid the introduction of coloured iron species into the treated water, resulting in clearer effluent. To prevent galvanic corrosion, the system uses identical materials for both anodes and cathodes.

The Al electrodes were designed to account for gradual wear: anodes corrode as they release coagulant ions, while cathodes may accumulate scale deposits. The electrode plates were designed for straightforward removal, cleaning, or replacement to maintain system performance. The expected anode consumption can be estimated using Faraday's law; for example, under typical operating conditions of 20 mA/cm² over 30 minutes, each anode was anticipated to dissolve by several hundred milligrams per treatment cycle. This level of material consumption is manageable, and keeping spare electrodes on hand ensures continuous system operation without interruption.

3.3.2 Electrode geometry

Flat rectangular plates were selected as the electrode geometry for the electrocoagulation (EC) reactor. This configuration offers several advantages: it provides a uniform electric field distribution, maximises electrode surface area, and allows for straightforward fabrication, handling, and cleaning. Additionally, flat plates support a compact reactor design, making them particularly suitable for lab-scale continuous flow systems where space and precision are critical.

The electrode size was carefully tailored to the reactor's physical constraints. An 8 cm plate height allows for a 2 cm freeboard within the 10 cm-tall tank, facilitating fluid circulation and permitting gas release. The 9.5 cm width (inside a 10 cm internal width tank) leaves only minimal lateral gaps, which serve two key purposes: enabling the insertion and removal of electrode plates and minimising water bypass along the sides. This design ensured that the entire cross-section of the reactor was occupied by electrode surfaces or narrow inter-electrode gaps, compelling the wastewater to flow across the electrodes for optimal treatment contact. Each electrode plate was made of 1.5 mm-thick Al, with a balance between mechanical sturdiness for repeated lab uses with minimal volume occupation, ensuring efficient use of the reactor's internal space.

3.3.3 Number of electrodes

A total of six electrodes (three anodes and three cathodes) were installed in alternating sequence. This number ensures adequate electrochemical surface area while maintaining a manageable reactor size. The configuration supports four electrochemical cell pairs operating in parallel.

3.3.4 Inter-electrode distance

The inter-plate gap between adjacent electrodes was set at 20 mm (2 cm), carefully selected to optimise both electrical and hydraulic performance. This spacing was designed to maximise mass transfer and minimise ohmic (electrical) drop across the system, while also preventing clogging or excessive bridging between plates. Uniform spacing is maintained using spacers positioned at the plate edges, ensuring consistent treatment conditions throughout the reactor.

A 20 mm gap offered several operational advantages. It provided sufficient space for the transport of Al^{3+} ions and the rise of hydrogen gas bubbles generated at the cathode, both of which enhance mixing and promote effective floc formation. At the same time, the gap was narrow enough to maintain good electrical efficiency, avoiding unnecessary energy loss or

short-circuiting effects. Each gap effectively functioned as a narrow treatment chamber, and the reactor's even plate spacing created five identical treatment zones in series, ensuring consistent exposure of the wastewater to the electrocoagulation process.

The electrical configuration employed a monopolar parallel arrangement, where all anodes were connected to a common positive terminal and all cathodes to a common negative terminal. This setup ensured that the same voltage was applied across each anode-cathode pair, promoting an even current distribution throughout the reactor. As a result, no single plate carried an excessive current or "steals" current from adjacent plates, ensuring uniform current density and homogeneous treatment across the entire water volume. While periodic polarity reversal or electrode switching could be implemented to reduce scaling and prolong electrode lifespan, this was not deemed critical for the short-duration experimental runs conducted in this study.

3.3.5 Electrical system and power supply configuration

A DC power supply unit was installed to drive the EC process, delivering the required electrical current to the electrode assembly. In the system, the positive lead from the power supply was connected to the anode plates, while the negative lead was connected to the cathode plates. The electrical connections were made using bolted terminals at the tops of the electrodes, ensuring secure and stable contact.

An ammeter was included in series within the circuit to monitor the current passing through the electrodes. While many benchtop DC power supplies are equipped with built-in ammeter displays, an external multi-meter or dedicated ammeter can also be connected for improved measurement accuracy. Real-time monitoring and precise adjustment of the applied current were critical, as the current density and hydraulic retention time together determined the coagulant (Al^{3+}) dosage and overall energy consumption of the process.

The power supply was equipped with ground-fault protection, and all electrode terminals were insulated to prevent accidental contact. Additionally, the system included an accessible switch or emergency shutoff, allowing power to be cut immediately if needed. All wiring and electrical components were mounted away from any potential water contact to minimise the risk of short circuits or equipment damage.

For these experiments, the system was operated within a current range of 2.3 to 5.8 A, targeting current densities between 4 and 10 mA/cm^2 , which are typical values for evaluating removal efficiencies in EC processes. A monopolar parallel configuration was employed, in

which all anodes were linked to the positive terminal and all cathodes to the negative terminal of the DC power supply. This configuration simplified the wiring and ensured that the applied voltage was distributed evenly across all electrode pairs, promoting uniform current flow and consistent electrochemical performance throughout the reactor.



Figure 21: Adjustable DC power supply with polarity switch

3.3.6 Non-conductive electrode support structure

A custom-fabricated non-conductive frame constructed from PVC or acrylic secured the electrodes at fixed 20 mm intervals. This support structure prevented electrical shorting and maintains uniform spacing, enhancing both electrical and hydraulic performance.

3.3.7 Feed tank (untreated wastewater)

A reservoir holding the raw abattoir wastewater was situated at the skid inlet. Wastewater was pumped from here into the EC reactor. A coarse screen could be placed on the intake to

prevent large solids from entering and clogging the flow paths. The tank volume was about 5L, to supply several residence times of flow.

3.3.8 Feed pump

A small metering pump was used to feed the electrocoagulation reactor with abattoir wastewater. This pump type was selected because it provides accurate flow control, can handle low flow rates (0.75–3.6 L/h) required to achieve the target hydraulic retention times (HRTs) of 16, 20, 25, and 32 minutes, and is gentle on high-solid-content liquids, minimising the risk of clogging or shear damage to flocs. The pump flow rate was adjustable through speed control, allowing precise switching between HRT settings during experiments. All wetted parts of the pump and tubing were made of PVC and silicone, ensuring chemical resistance to the potentially corrosive components present in the abattoir wastewater.

3.3.9 Cell design and surface-to-volume ratio

The EC reactor was designed with specific dimensions and internal configurations to optimise hydraulic behaviour, maximise treatment efficiency, and accommodate the demands of high-strength abattoir wastewater. The reactor's internal dimensions were approximately 150 mm (length) × 100 mm (width) × 100 mm (height), providing a total internal volume of 1.5 litres (0.0015 m³). However, the active liquid volume was regulated by a level-control baffle, reducing the working volume to approximately 1.1 litres, and further accounting for the displacement of electrodes, the effective active treatment volume is approximately 1.1 litres.

The reactor used six flat rectangular Al plates as electrodes, with each plate measuring 80 mm (height) × 95 mm (width) × 1.5 mm (thickness). Considering both sides of the middle four plates and one side of the outer two plates, the total active electrode surface area was approximately 665 cm² (0.0665 m²). This configuration achieved a high surface-to-volume (S/V) ratio of ~58.6 m²/m³, which is critical for sufficient coagulant (Al³⁺) generation and high pollutant removal, especially for high COD and TOC wastewaters. When adequate Al dosage and residence time were provided, removal efficiencies often exceeded 90% for COD and turbidity in slaughterhouse effluent.

To optimise reactor hydrodynamics, the design adopted an aspect ratio and internal layout that approach plug-flow behaviour. Unlike completely mixed reactors, plug flow ensures that every fluid element receives the full treatment time, significantly improving overall efficiency. The reactor forced the influent to flow through narrow 20 mm gaps between electrodes, effectively preventing channelling or preferential flow toward the outlet. These gaps were

uniformly maintained with edge spacers, ensuring consistent electrical and hydraulic conditions.

The up-flow hydraulic configuration - where influent entered through a bottom pipe on the left and exited through a top pipe on the opposite side - served two functions: it helped lift and transport flocculated solids toward a downstream settling unit, and it prevented sediment accumulation at the reactor base. The absence of mechanical stirrers preserved the hydraulic profile, while the natural rise of hydrogen bubbles generated at the cathode introduced gentle mixing, supporting the even distribution of contaminants and coagulants.

The decision to exclude a mechanical mixer was intentional as mechanical mixing in EC reactors can improve mass transport but also adds complexity and can shear the forming flocs. In this design, the hydraulic flow and evolution of gas provided adequate mixing. As hydrogen bubbles formed on the cathodes, they rose and created an “airlift” effect that agitated the water between plates, aiding the contact between coagulants and pollutants. The upward flow design capitalised on this: bubbles coalesced and carried flocs upward into the clarifier. If mixing were insufficient, one could consider mild air sparging or a recirculation loop, but based on similar scale reactors in the literature, this single-pass design was effective. Simpler is better for reliability and scale-up consistency.

To ensure chemical resistance and allow visual monitoring, the reactor body was constructed from inert materials such as clear acrylic. Internally, the plate arrangement functions like a series of baffle chambers, guiding the fluid along a zigzag path that creates mild eddying and secondary flows at plate edges. Computational fluid dynamics (CFD) analysis confirmed that this design produces a near-uniform velocity profile, with only minor deviations near the walls, which are compensated by the long residence time (≥ 16 minutes) and the gentle mixing effects of gas evolution

3.3.10 Settling tank and dimensions

Because EC generates suspended solid flocs as a fundamental part of the pollutant removal process, a dedicated settling step was essential downstream of the reactor to separate these solids from the treated water. Adding a settling phase significantly enhanced overall removal efficiencies for turbidity and COD by allowing the flocculated particles to separate effectively under gravity.

For this system, a gravity-based settling tank was integrated immediately downstream of the EC reactor. The tank was sized to provide a retention time comparable to that of the EC unit,

with a working volume of approximately one litre, ensuring sufficient time for solids to settle without requiring more complex mechanical separation technologies such as lamella clarifiers or centrifuges. This simple and practical approach was well-suited to lab-scale operations, where moderate flow rates and system simplicity are prioritised.

A weir system ensured that only clarified effluent reached the outlet, preventing the carryover of floating solids. The settled sludge, composed of concentrated blood, proteins, and Al hydroxide flocs, collected at the bottom and could be drained or sampled through a dedicated bottom valve.

The clarified effluent, reduced in turbidity and often meeting lower COD targets, could be directed to a collection vessel or drainage system after this stage. If additional treatment or polishing (such as filtration) were required, it could be added downstream; however, the current system focused solely on EC followed by gravity-driven solid-liquid separation.

A key consideration in the skid integration was the precise placement of the settling tank relative to the EC reactor, specifically, ensuring it was positioned close to the reactor outlet and at the appropriate elevation to prevent siphoning or backflow issues. This setup supported both continuous and batch operations while providing sufficient sample volumes for post-treatment water quality analysis.

Abattoir wastewater treated by EC typically generates flocs that are large and dense due to the high organic and particulate content (e.g., blood, fats, proteins) and are readily separable within short to moderate settling periods. Most of the turbidity and suspended solids settle out within 10–30 minutes after EC treatment. Extending the settling time beyond 30 minutes usually yields only marginal improvements, while unnecessarily increasing time and space requirements.

Therefore, in this study, the settling time was fixed at 10 minutes, providing:

- Sufficient time for effective solid-liquid separation
- A practical balance between treatment performance and throughput
- Reduced experimental complexity by removing settling time as an additional optimisation parameter

This approach ensures reliable post-treatment separation while maintaining a streamlined experimental design.

3.3.11 Piping and valves

The inlet line from the feed tank to the pump, and the pump to the reactor, were securely clamped. The outlet of the reactor was plumbed to the settling tank with flexible tubing. An adjustable pump speed to control flow rate, a drain valve at the bottom of the reactor to empty it after use, and a drain on the settling tank for desludging were part of the piping and valves system. Quick-connect fittings could thus be used to disassemble parts of the skid for cleaning. The use of plastic piping avoided corrosion and is easy to reconfigure in a lab setting.

3.3.12 Post-treatment sludge separation

Sludge generated during electrocoagulation was separated in the settling tank via flotation and sedimentation. The floating sludge was skimmed from the top, while settled solids were removed through a drain at the bottom. No chemical additives were used for post-treatment; separation relied solely on gravity and electro-flotation effects.

3.3.13 Skid frame

All components were placed on a workbench. The layout was linear: the feed tank and pump at one end, the reactor cell in the middle, and the settling tank at the other end, with short pipes connecting them, as shown in Figure 22. The DC power supply and its controls were mounted on the frame above or beside the reactor. The overall footprint was compact so that it could fit in a laboratory space

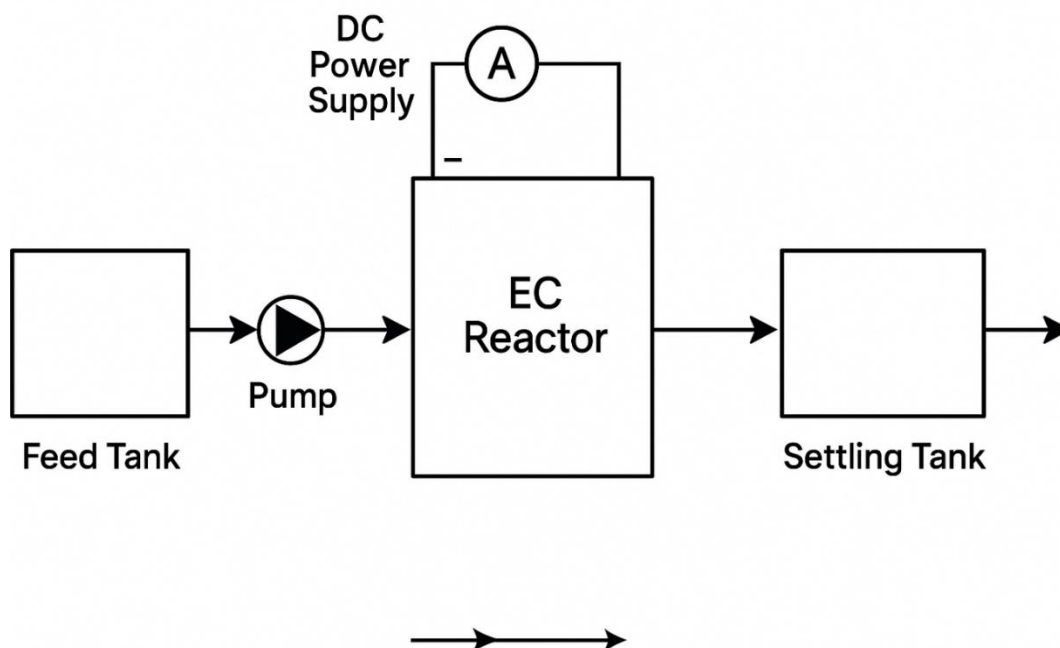


Figure 22: Schematic diagram showing the Lab EC PFD layout



Figure 23: Actual experimental setup

3.3.14 Fixed parameter conclusion

In summary, the lab-scale EC reactor and skid system developed for this study were designed to deliver efficient, robust, and representative treatment of high-strength organic wastewater, specifically abattoir effluent. The six-plate reactor provided a high active electrode area, optimised hydraulic behaviour, and controlled operating conditions, ensuring effective generation of coagulant species and thorough contaminant removal.

The complete system integrated all essential components: a feed and influent delivery setup, a DC power supply, and a downstream gravity-based settling unit for post-treatment solid-liquid separation. The design leverages off-the-shelf materials, such as silicone piping, acrylic

and plastic tanks, and Al electrodes, allowing for ease of construction, cost-efficiency, and straightforward maintenance.

The design rationale was grounded in established electrocoagulation principles: maximising the interaction between pollutants and generated coagulants, promoting uniform flow distribution across electrode surfaces, and ensuring reliable separation of flocculated solids. This setup enabled systematic experimentation on key operational parameters such as current density, HRT, and initial water quality, providing meaningful insights into process performance under continuous-flow conditions.

Overall, the proposed reactor and skid met the required technical specifications and provided a clear, practical platform for advancing the understanding of electrocoagulation processes applied to challenging organic wastewater.

3.4 Variable design parameters

This study focused on systematically optimising the key operational variables that govern the performance of the EC process when treating abattoir wastewater. Specifically, HRT and current density were selected as the two primary parameters for detailed investigation, given their direct influence on coagulant generation, pollutant destabilisation, floc formation, and overall treatment efficiency. The settling time was fixed, based on literature values, to simplify the experimental design and isolate the effects of the two main variables. The following subsections describe the rationale, selected test ranges, and experimental approach for each parameter.

3.4.1 Hydraulic retention time

The flow through the EC reactor was purely hydraulic, operating as a single-pass continuous-flow system without the use of impellers or mechanical mixers. The influent entered at the bottom of the reactor and rose through the narrow channels formed between the six electrode plates, driven by a controlled feed pump. Thanks to the baffle-like arrangement of the plates, the liquid was well-distributed across the electrode surfaces, minimising dead zones and ensuring that the full water volume engages with the electrochemical process.

The system was designed to operate under laminar flow conditions, with Reynolds numbers estimated in the range of 10–50, well within the laminar regime due to the low flow velocities and narrow 2 cm inter-electrode gaps. This laminar flow, combined with the plug-flow reactor geometry, produced a uniform residence time distribution, ideal for achieving consistent and predictable treatment performance. The gentle hydraulic movement, along with the buoyant

rise of electrolysis-generated hydrogen bubbles, supported floc formation without excessive shear that might disrupt the developing aggregates.

The HRT, defined as the average time the wastewater remains in the EC reactor, was a key parameter governing the duration available for electrochemical reactions and floc formation. Sufficient HRT is essential to allow complete pollutant destabilisation, coagulant generation, and initial floc aggregation before the treated water exits the system. Insufficient HRT can lead to under-treatment and reduced removal efficiencies, while excessively long HRTs may increase reactor volume and operational costs without providing proportional performance gains.

Based on reported literature values for slaughterhouse wastewater, HRTs typically range from 30 to 90 minutes, depending on reactor configuration, applied current density, and the physicochemical characteristics of the influent [38]. However, following preliminary testing with the described setup, it was decided to reduce both the HRT and current density, given the expectation that comparable treatment efficiencies could be achieved under less intensive operational conditions. To systematically assess the influence of HRT on treatment performance in this study, four discrete retention times were selected, with corresponding flow rates calculated based on the reactor's active volume of 1.1 L:

- 16 minutes (Short HRT) – Represents the minimum operational range for meaningful electrocoagulation, simulating high-throughput conditions.
 - Flow rate: 3.6 L/h
- 20 minutes (Moderate-short HRT) – Provides an intermediate condition to assess marginal improvements over the short HRT.
 - Flow rate: 2.9 L/h
- 25 minutes (Moderate-long HRT) – Commonly reported as a near-optimal HRT for EC treatment of organic-rich wastewater in both batch and continuous systems.
 - Flow rate: 2.3 L/h
- 32 minutes (Long HRT) – Tests the upper retention time limit to evaluate whether extended treatment further improves removal or increases energy consumption and residence time.
 - Flow rate: 1.8 L/h

These flow rates were precisely set, using the pump system. At the moderate flow of ~2.3 L/h (corresponding to ~25-minute HRT), the linear velocity through each 2 cm electrode gap was in the order of a few millimetres per second, ensuring stable laminar flow. Even at the highest

flow rate (3.6 L/h for a 16-minute HRT), the Reynolds number remained below one hundred, indicating that turbulence was unlikely to develop. This flow regime was desirable as it prevented disruption of fragile flocs, allowing gentle electrochemical bubble-driven mixing to assist in lifting and concentrating the flocs toward the reactor outlet.

By systematically evaluating these four HRT levels, the study aimed to identify the minimal retention time needed to achieve target removal efficiencies (e.g., COD, turbidity) while maintaining energy efficiency and practical operational throughput.

3.4.2 Current density

Current density, defined as the applied current per unit electrode surface area (mA/cm^2), is a key operational parameter influencing the EC process. It directly determines the rate of metal (Al) dissolution from the anode, the generation of coagulant species (Al^{3+}), and the production of microbubbles at the cathode. Literature [45] has consistently shown that higher current densities increase pollutant removal efficiency by enhancing coagulant dosage and mixing but also increase energy consumption and electrode wear. Conversely, operating at too low a current density can result in insufficient coagulant production, leading to poor treatment performance.

Based on prior studies treating slaughterhouse wastewater, optimal current densities typically fall within the range of 4–10 mA/cm^2 . Preliminary trials and simulation results suggested that within this range, removal efficiencies improved significantly before plateauing beyond ~10–15 mA/cm^2 , where additional current provided diminishing returns while raising operational costs. To balance treatment efficacy and energy efficiency, the following four current density levels are selected for experimental optimisation:

- 4 mA/cm^2 (Low) – Baseline condition to observe minimal coagulant dosage performance.
 - Amperage: 2.3 A
- 6 mA/cm^2 (Moderate-low) – Intermediate condition to test improvements over the baseline.
 - Amperage: 3.5 A
- 8 mA/cm^2 (Moderate-high) – Literature-reported near-optimal zone for high removal efficiency.
 - Amperage: 4.7 A
- 10 mA/cm^2 (High) – Upper range to test if additional current yields meaningful performance gains or increases cost.

- Amperage: 5.8 A

These test points provided a structured approach to evaluate the influence of current density on key performance indicators, including COD removal, turbidity reduction, energy consumption, and sludge generation.

3.4.3 Parameters intentionally kept constant

Although the literature identifies initial pH and mixing conditions as important electrocoagulation parameters, these variables were not selected as independent variables in the present experimental matrix. This was an intentional design decision made to maintain focus on the selected optimisation variables, namely HRT and current density

The initial wastewater pH was not chemically adjusted prior to treatment. This approach was selected for two reasons. Firstly, the measured initial pH of the slaughterhouse wastewater fell within a range that was considered suitable for aluminium-based electrocoagulation according to the literature. Secondly, the study aimed to evaluate electrocoagulation as a stand-alone treatment process under realistic wastewater conditions, without introducing additional chemical conditioning steps that could alter the natural feed characteristics. The observed pH shift during treatment was therefore regarded as part of the inherent electrochemical response of the system rather than as a separately manipulated variable.

Similarly, no mechanical stirring device was used. Mixing within the reactor was achieved only by wastewater flow through the system and by gas evolution generated during electrocoagulation. This decision was taken to reflect the intended practical application of the technology more realistically, since large-scale continuous-flow EC systems would typically rely on hydraulic motion and electrochemically generated bubbles rather than mechanical stirring. By keeping the mixing mode consistent across all runs, the study ensured that the effect of the selected independent variables could be evaluated without introducing an additional variable related to agitation intensity.

These choices do not imply that pH and mixing are unimportant; rather, they were intentionally treated as controlled process conditions outside the scope of the experimental optimisation matrix. Their influence is acknowledged in the literature review and considered in the interpretation of results.

3.5 Experimental matrix design

The experimental plan for this study was designed to systematically optimise key operational parameters of the electrocoagulation (EC) process. Three main variables considered were:

- Hydraulic Retention Time (HRT) → 4 levels: 16, 20, 25, and 32 minutes
- Current Density → 4 levels: 4, 6, 8, and 10 mA/cm²
- Settling Time → fixed at 10 minutes (based on prior optimisation and literature findings)

This resulted in a total of sixteen experimental runs, derived from the full factorial combination of HRT and current density levels:

- Total Runs = 4(HRT) × 4(Current Density) = 16 experiments

The experimental matrix is outlined below:

Table 5: Full factorial experimental matrix (4²)

Full factorial experimental matrix (4 ²)				
Run No.	HRT (min)	Flow Rate (L/h)	Current Density (mA/cm ²)	Amperage (A)
1	20	2.9	10	5.8
2	20	2.9	8	4.7
3	20	2.9	6	3.5
4	20	2.9	4	2.3
5	32	1.8	10	5.8
6	32	1.8	8	4.7
7	16	3.6	4	2.3
8	25	2.3	8	4.7
9	25	2.3	4	2.3
10	32	1.8	4	2.3
11	16	3.6	6	3.5
12	25	2.3	6	3.5
13	32	1.8	6	3.5
14	16	3.6	8	4.7
15	16	3.6	10	5.8
16	25	2.3	10	5.8

This matrix allowed for a systematic evaluation of how HRT and current density interact to influence removal efficiencies (COD, electrical conductivity, colour, turbidity, and E. coli) while holding the settling time constant. The amperage values were calculated based on the total electrode surface area and the target current densities, ensuring consistent and reproducible operating conditions across all trials.

3.5.1 Justification for the selected methodology

A structured experimental design was required because electrocoagulation performance is influenced by interacting operating variables rather than by isolated one-factor effects alone. For this reason, a factorial RSM approach was selected to evaluate the combined influence of HRT and current density on treatment performance. This approach was considered appropriate for the present study because it allows both linear and interaction effects to be evaluated systematically, while also supporting the development of empirical models for optimisation.

The study focused on HRT and current density because these two parameters are among the most practically significant operating variables in continuous-flow EC treatment. HRT directly determines the contact time between the wastewater and the electrochemically active environment, while current density governs coagulant generation rate, gas evolution, and electrical input. Together, these variables strongly influence contaminant destabilisation, floc formation, and overall treatment efficiency.

The selected methodology was also appropriate from a process engineering perspective, since the aim was not merely to describe EC behaviour, but to identify operating conditions relevant to practical reactor design and scale-up. A laboratory-scale continuous-flow reactor was therefore used to reflect the intended application more realistically than a purely batch-based system. The use of RSM further allowed the generation of predictive equations that can support future design and optimisation work.

3.6 Theoretical and empirical modelling

A two-part modelling framework was employed to support the optimisation of EC process parameters:

- Theoretical modelling using Faraday's law to estimate the electrochemical coagulant dosage; and
- Empirical modelling using RSM to statistically analyse and predict system performance.

3.6.1 Theoretical modelling (Faraday-based)

The theoretical estimation of Al ion dosage is based on Faraday's law of electrolysis, which relates the mass of metal dissolved from the anode to the applied electrical charge. The general formula is:

$$m = \frac{ItM_w}{zF} \quad (26)$$

where:

- m = mass of Al dissolved (g)
- I = applied current (A)
- t = electrolysis time (s)
- M_w = molar mass of Al (26.98 g/mol)
- z = valency of Al ($z = 3$ for Al^{3+})
- F = Faraday's constant (96,485 C/mol)

This relationship allows the prediction of the theoretical Al dose under different operating conditions. In this study, the equation was implemented in Excel to calculate the mass of Al expected to dissolve per run, based on the applied current (determined from the target current density and total electrode area) and the HRT.

Because the experimental system operated under continuous flow, the total residence time for a unit volume is governed by the selected HRT, and the current I is determined from the selected current density (J) using:

$$I = J \cdot A \quad (27)$$

where:

- J = current density (mA/cm², converted to A/m²)
- A = total active electrode area (m²)

It is acknowledged that both pH and post-treatment settling time play important roles in influencing anode dissolution behaviour and overall pollutant removal efficiency. However, these factors were not incorporated into the theoretical modelling in this study, which focuses solely on estimating the theoretical Al dosage and does not attempt to predict the percentage of pollutants removed.

Table 6: Predicted Al dosage for experimental runs.

Predicted Al dosage - Faraday's law			
Run No.	HRT (min)	Amperage (A)	Al dose (g)
1	20	10	1.12
2	20	8	0.89
3	20	6	0.67
4	20	4	0.45
5	32	10	1.79
6	32	8	1.43
7	16	4	0.36
8	25	8	1.12
9	25	4	0.56
10	32	4	0.72
11	16	6	0.54
12	25	6	0.84
13	32	6	1.07
14	16	8	0.72
15	16	10	0.89
16	25	10	1.40

3.6.2 Empirical modelling (RSM)

To empirically model and optimise the EC process, an RSM approach was adopted. This method develops predictive mathematical models linking the system's performance (removal efficiencies of COD, electrical conductivity, colour, turbidity, and E. coli) to the key process variables:

- HRT and
- Current density

A full factorial design was implemented with four levels per factor, generating a total of sixteen experimental runs (Chapter 4). The settling time was fixed at 10 minutes, based on prior optimisation results indicating that further increases beyond this point yield minimal performance gains.

The empirical RSM model was based on a quadratic polynomial equation:

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \beta_{11} X_1^2 + \beta_{22} X_2^2 + \beta_{12} X_1 X_2 + \varepsilon \quad (28)$$

where:

- Y = response variable (e.g., COD or turbidity removal efficiency)

- X_1 = coded value of HRT
- X_2 = coded value of current density
- B_0 = intercept
- B_1, B_2 = linear coefficients
- B_{11}, B_{22} = quadratic coefficients
- B_{12} = interaction coefficient
- ε = random error

Experimental results were analysed statistically to estimate model coefficients and evaluate the significance of each factor. A Pareto analysis was performed to quantify the relative influence of each parameter on system performance, using:

$$P_i = \frac{b_i^2}{\sum b_i^2} \times 100 \quad (i \neq 0) \quad (29)$$

where:

- P_i = percentage contribution of factor i
- b_i = estimated coefficient (main or interaction effect)

The fitted RSM models generated regression equations, contour plots, and three-dimensional response surfaces, enabling the identification of the optimal combination of HRT and current density to maximise pollutant removal while considering energy consumption and electrode usage. Confirmatory experiments were then conducted under the predicted optimal conditions to validate the model's predictive accuracy.

3.6.3 Replication strategy, model adequacy, and validation approach

The 16 experimental runs defined by the factorial design constituted the primary model-development dataset for this study and were not replicated. In addition, no preliminary one-factor-at-a-time (OFAT) screening study was conducted before the factorial investigation. This approach was selected because the study aimed from the outset to evaluate the combined effect of HRT and current density using a structured multivariable design, rather than first examining each parameter in isolation. Since electrocoagulation performance is influenced by interactions between variables, the RSM-based factorial design was considered more appropriate than OFAT for the objective of process optimisation.

It is acknowledged that the absence of replicated factorial runs limits the direct quantification of experimental variability within the primary design matrix. This is a limitation of the present

study and should be considered when interpreting the precision of the fitted empirical models. To strengthen model credibility, however, the study incorporated several alternative forms of model assessment. These included ANOVA, evaluation of fitted model performance, examination of predicted trends against measured responses, and independent confirmatory validation experiments conducted after the primary factorial matrix.

The confirmatory validation experiments, reported as runs 17 to 20, were performed under selected operating conditions within the optimised region and were not part of the original 16-run design matrix. These runs, therefore, served as independent validation points to assess the predictive reliability of the developed empirical models. The close agreement observed between predicted and measured responses during validation supports the practical usefulness of the optimisation framework, even though full replication of the factorial matrix was not undertaken.

Future work should strengthen the statistical robustness of the optimisation framework by including replicated runs, repeated centre-point experiments, and expanded residual and lack-of-fit analysis.

3.7 Methodology conclusion

To complete the experimental program, the designed runs were conducted systematically following the full factorial RSM matrix. For each experimental condition, key performance indicators, including removal of COD, E. coli, turbidity, electrical conductivity and colour reduction, as well as energy consumption, were measured.

The collected data were then fitted to a quadratic model. This modelling approach allowed the generation of response surfaces and contour plots, providing clear visualisations of the optimal operational regions.

Based on the analysis, the study identified the optimal combinations of HRT and current density that balance treatment performance with operational efficiency. Finally, confirmatory experiments were performed under the identified optimal conditions to validate the model predictions and ensure the robustness and reliability of the optimisation results.

The following chapter presents the experimental results and modelling outcomes, detailing the treatment performance observed under the tested conditions and the development of predictive models to optimise the electrocoagulation process.

CHAPTER 4 RESULTS AND DISCUSSION

4.1 Data analysis and empirical model fitting

The data collected from the 16-run experimental matrix described in Chapter 3, Section 3.6, were analysed using RSM to model and optimise the influence of HRT and current density on the performance of the EC system. A full-factorial 4×4 experimental design was employed, with responses measured for removal efficiencies of COD, turbidity, colour, E. coli, changes in electrical conductivity, and specific energy consumption (in kWh/m³). Parallel observations also included pH evolution during treatment, relevant to coagulation chemistry and microbial inactivation.

To model the relationship between the two independent variables, HRT and current density, and each system response, second-order polynomial equations were fitted to the experimental data using statistical software. The general model form is:

$$Y = \beta_0 + \beta_1X_1 + \beta_2X_2 + \beta_{11}X_1^2 + \beta_{22}X_2^2 + \beta_{12}X_1X_2 + \varepsilon \quad (26)$$

where:

- Y : predicted response (e.g., COD removal, turbidity reduction),
- X_1 : coded value of HRT,
- X_2 : coded value of current density,
- β : regression coefficients (intercept, linear, quadratic, interaction),
- ε : random error term.

The regression models (in coded variables) obtained are:

- COD Removal (%)

$$Y = 68.153 + 4.992X_1 - 12.278X_2 + 0.860X_1^2 - 1.547X_2^2 \quad (30)$$

- Colour Removal (%)

$$Y = 87.119 - 0.960X_1 + 2.787X_2 + 4.293X_1^2 - 0.141X_2^2 \quad (31)$$

- Electrical Conductivity Reduction (%)

$$Y = 41.473 + 5.890X_1 + 0.273X_2 - 3.998X_1^2 - 2.531X_2^2 \quad (32)$$

- E. coli Removal (%)

$$Y = 100.088 + 0.615X_1 - 0.448X_2 + 0.223X_1^2 - 1.406X_2^2 \quad (33)$$

- Turbidity Removal (%)

$$Y = 95.822 - 0.065X_1 + 1.245X_2 + 1.545X_1^2 - 0.703X_2^2 \quad (34)$$

Each model was evaluated using ANOVA (Annexure F), which confirmed a statistically significant fit ($p < 0.05$) and high coefficients of determination ($R^2 > 0.90$ for all removal metrics and >0.99 for energy consumption). Adjusted R^2 values were only marginally lower, indicating model reliability and minimal overfitting.

Both HRT and current density showed strong, statistically significant linear effects ($p < 0.01$), affirming their individual impact on treatment performance. Quadratic terms also reached significance in most models, revealing the expected curvature in responses, i.e., diminishing returns at higher operating levels. Interaction terms (HRT $\times$ current density) were moderately significant in COD and E. coli removal models, showing a mild synergistic influence.

As expected, positive linear coefficients reflect that increased HRT or current density improves performance. Negative quadratic terms suggest that after a certain point, further increases in either parameter offer diminishing benefit. A common trend in EC systems is approaching optimal flocculation capacity.

Lastly, pH measurements revealed a consistent increase from approximately 6.8 (untreated) to near 9.0 post-treatment. This shift is typical in EC systems with Al electrodes due to cathodic hydroxide production. The resulting alkaline environment not only supports coagulant formation ($\text{Al}(\text{OH})_3$) but also contributes to pathogen inactivation, consistent with the complete or near-complete removal of E. coli across most trials.

The results presented in this chapter are based on the 16 experimental runs defined by the factorial design matrix used for model development. These primary runs were not replicated. Consequently, the discussion places emphasis on trend interpretation, model fitting, ANOVA outcomes, and independent confirmatory validation rather than on within-run repeatability statistics. Independent confirmatory experiments, reported separately as runs 17 to 20, were used to assess the predictive reliability of the developed models under selected operating conditions.

Table 7: Summary of experimental results for the 16-run factorial design (data derived from Annexure A)

Pollutant removal - lab results						
Run No.	E. coli (count/ml)	Colour (mg/l Pt)	pH @twenty-five°C	EC (mS/m)	Turbidity (NTU)	COD (mg/l)
0	100	201	6.86	116.2	34.6	748
1	7	26	8.52	83.24	1.39	453
2	0	14	8.84	71.73	0.65	238
3	0	5	9.07	60.91	0.5	247
4	0	9	8.93	73.73	0.76	255
5	0	14	9.23	58.31	0.99	148
6	0	23	9.37	64.48	1.04	305
7	3	41	8.75	83.82	3.02	149
8	0	27	9.13	71.01	1.53	213
9	0	35	9.28	69.83	2.11	110
10	0	24	9.26	69.86	1.44	99
11	0	14	9.19	77.58	0.77	133
12	0	64	9.06	71.33	3.98	161
13	0	11	9.15	76.25	0.44	274
14	0	13	9.13	80.08	0.7	338
15	0	13	9.12	86.39	0.64	416
16	0	10	9.3	69.44	0.81	360

Table 8: Pollutant removal percentage

Pollutant removal efficiency (%)					
Run No.	E. coli (count/ml)	Colour (mg/l Pt)	EC (mS/m)	Turbidity (NTU)	COD (mg/l)
1	93%	87%	28%	96%	39%
2	100%	93%	38%	98%	68%
3	100%	98%	48%	99%	67%
4	100%	96%	37%	98%	66%
5	100%	93%	50%	97%	80%
6	100%	89%	45%	97%	59%
7	97%	80%	28%	91%	80%
8	100%	87%	39%	96%	72%
9	100%	83%	40%	94%	85%
10	100%	88%	40%	96%	87%
11	100%	93%	33%	98%	82%
12	100%	68%	39%	88%	78%
13	100%	95%	34%	99%	63%
14	100%	94%	31%	98%	55%
15	100%	94%	26%	98%	44%
16	100%	95%	40%	98%	52%

Table 9: Predicted vs actual Al dosage

Predicted vs Actual Al dosage - Faraday's law					
Run No.	HRT (min)	Amperage (A)	Predicted Al dose (g)	Actual Al dose (g)	Power consumption (kWh)
1	20	10	1.12	1.45	70.54
2	20	8	0.89	1.13	46.14
3	20	6	0.67	0.84	23.12
4	20	4	0.45	0.51	7.64
5	32	10	1.79	1.98	112.87
6	32	8	1.43	1.52	73.83
7	16	4	0.36	0.4	6.11
8	25	8	1.12	1.32	57.68
9	25	4	0.56	0.64	9.55
10	32	4	0.72	0.76	12.22
11	16	6	0.54	0.6	18.49
12	25	6	0.84	0.95	28.89
13	32	6	1.07	1.15	36.98
14	16	8	0.72	0.78	36.92
15	16	10	0.89	0.97	56.43
16	25	10	1.40	1.52	88.18

The comparison between theoretical Al dosage, calculated using Faraday's Law, and the actual mass of Al dissolved during EC treatment revealed a consistent pattern of over-dissolution. Across all sixteen runs, the measured Al mass exceeded predictions by 10% to 30%, with the deviation averaging around 20%. This discrepancy can be attributed to factors not accounted for in the idealised Faraday equation, such as electrode surface roughness, secondary oxidation reactions, local turbulence, and mechanical erosion from gas evolution. These phenomena contribute to enhanced anodic dissolution beyond what is predicted from pure electrochemical stoichiometry. The observed deviations underscore the need to include empirical calibration in practical EC system design to avoid underestimating metal consumption and associated sludge generation. Annexure E: Electrodes.

4.2 Pareto analysis

A Pareto analysis of the standardised effects was performed to visualise the relative influence of each factor and its interaction on the treatment responses (see Figure 24). The Pareto chart for each response model confirms that the most influential factors in this process are indeed HRT and current density, with the linear effects dominating. For most water quality responses (COD, turbidity, colour, E. coli removal), the current density emerged as the single most influential factor (the longest bar on the Pareto chart), slightly surpassing the HRT effect. This

indicates that, in the electrocoagulation process, the amount of current, which governs the dosage of coagulant metal ions and rate of bubble generation for flotation, has a primary role in determining treatment efficiency. HRT is the next most influential factor, underscoring that sufficient contact time is also crucial for allowing coagulation, floc growth, and contaminant removal to proceed to completion.

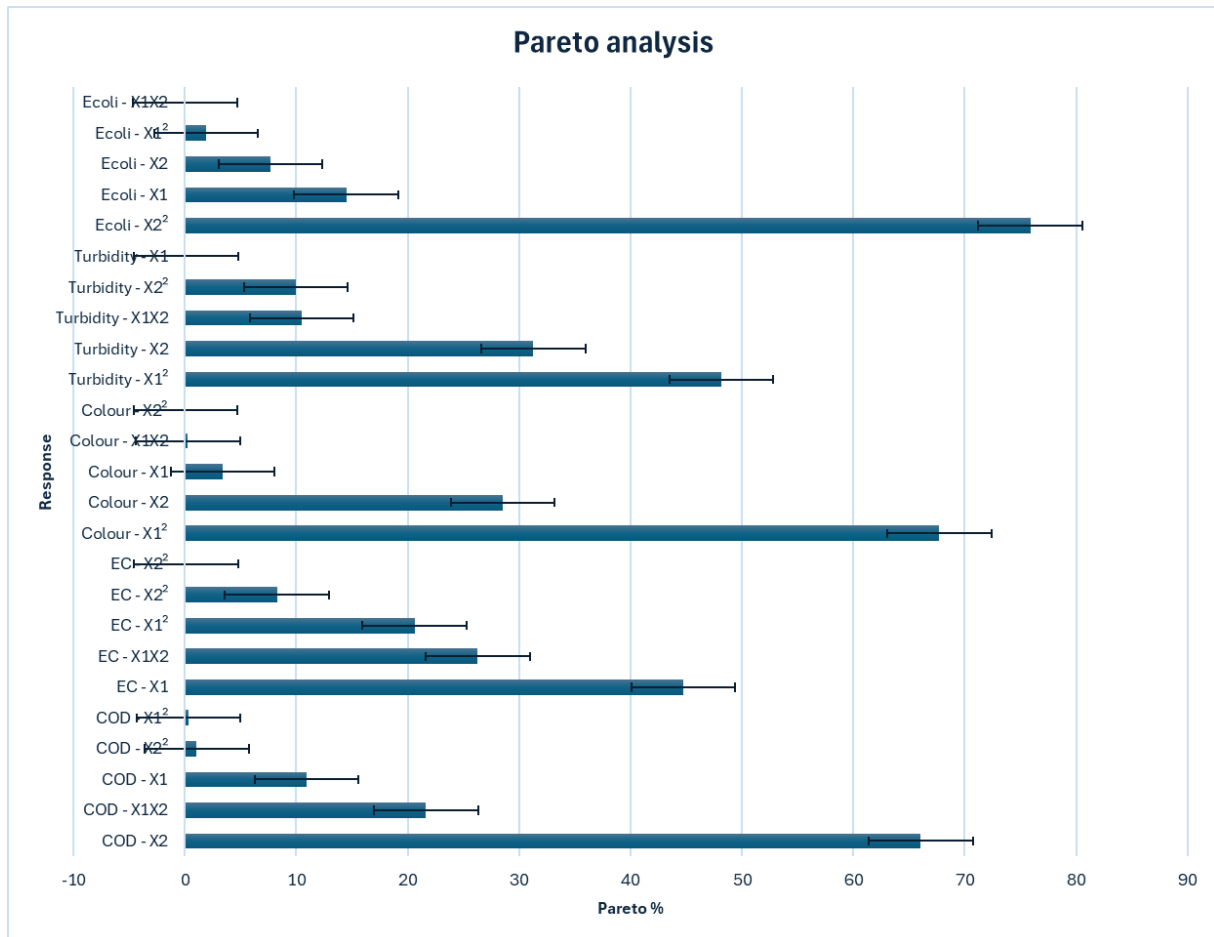


Figure 24: Pareto chart of factor importance

The quadratic terms appeared on the Pareto chart with shorter bars; among them, the current density squared effect was typically more pronounced than the HRT squared effect. This suggests that non-linearity (curvature) in the response is more strongly associated with the current density, likely because very high current densities can reach a point of diminishing returns or even cause secondary phenomena (e.g. excessive gas evolution or heating) that do not further improve removal. The HRT quadratic effect, while present, was milder – implying that extending retention time has a more linearly sustained benefit up to practical limits (within the tested range) before plateauing gently. The interaction (HRT $\times$ current) had the shortest bar for most responses, confirming it as a lesser contributor to variance compared to the main effects. However, it was not negligible: for E. coli removal, for instance, the interaction effect

was more noticeable, hinting that disinfection efficiency benefits from simultaneous increases in both current and time together.

From the Pareto analysis, one can conclude which factor is dominant for each outcome. Current density is the dominant factor for turbidity and colour removal – a higher current dosage rapidly generates metal hydroxide flocs that sweep up colloidal particles and pigments. HRT plays a supporting but still significant role in these responses, to ensure thorough mixing and settling of flocs. For COD removal, both factors are equally important: current density provides the treatment agents (and possibly oxidative radicals at the anode), while time allows complex organic pollutants to be adsorbed or oxidised; thus, COD reduction requires a balanced optimisation of both. In the case of *E. coli* inactivation, HRT can be as influential as current density because microbial cells may need prolonged exposure to the electrochemical environment and any in-situ generated disinfectants to achieve complete kill. In summary, the Pareto charts illustrate that no single factor alone can achieve optimal treatment; both sufficient electrical input and time in the reactor are necessary, with electrical input being slightly more critical in our system. These insights, consistent with fundamental electrocoagulation principles, will guide the determination of optimal operating conditions.

4.3 Response surfaces and contour plots

The fitted RSM models were used to generate various plots for each response, mapping out the performance across the design space of HRT and current density. These visualisations help identify optimal treatment windows where high contaminant removal is achieved. In general, the response surfaces for pollutant removal had a similar shape, an upward rise towards a plateau (see Figures 24 to 28). At low HRT and low current density, the removal efficiencies were poor, reflecting insufficient treatment. As one moves towards higher HRT or higher current, the removal percentages increase substantially. Multiple combinations of HRT and current can yield a given performance level, for example, a longer HRT with moderate current can achieve similar removal as a shorter HRT with higher current. This illustrates the trade-off between the two factors.

For COD removal, the 3D surface plot showed that beyond roughly the mid-level of current density, the incremental gain in COD removal per additional unit of current began to shrink, especially at high HRT. A similar diminishing return is seen for HRT beyond a certain point at high current. This creates an optimal zone rather than a single point: in this zone, COD removal was maximised and further increases in either factor give minimal additional benefit. In practical terms, the optimal window for COD removal encompassed high current densities (within the top 25% of the tested range) and longer retention times, above the mid-point of the

tested HRT range. Operating in this window achieved COD removal efficiencies as significant as ~85–90%, whereas outside this window, especially at the low-low corner, the efficiency could drop below 50%. Turbidity and colour removal exhibited even higher maximal efficiencies, often exceeding 95% in the optimal region. The 3D surfaces for turbidity, for instance, were very steep near the low-current end (indicating turbidity removal improves drastically with a slight increase in current from very low values) and then flattened out at high current — meaning once most particles are coagulated (e.g. >99% turbidity removal), further increases in coagulant dose have little effect. This flattening at the top of the surface plot confirms a practical saturation point in treatment: all suspended solids and colour-causing compounds that can be removed have been captured by flocs at that stage.

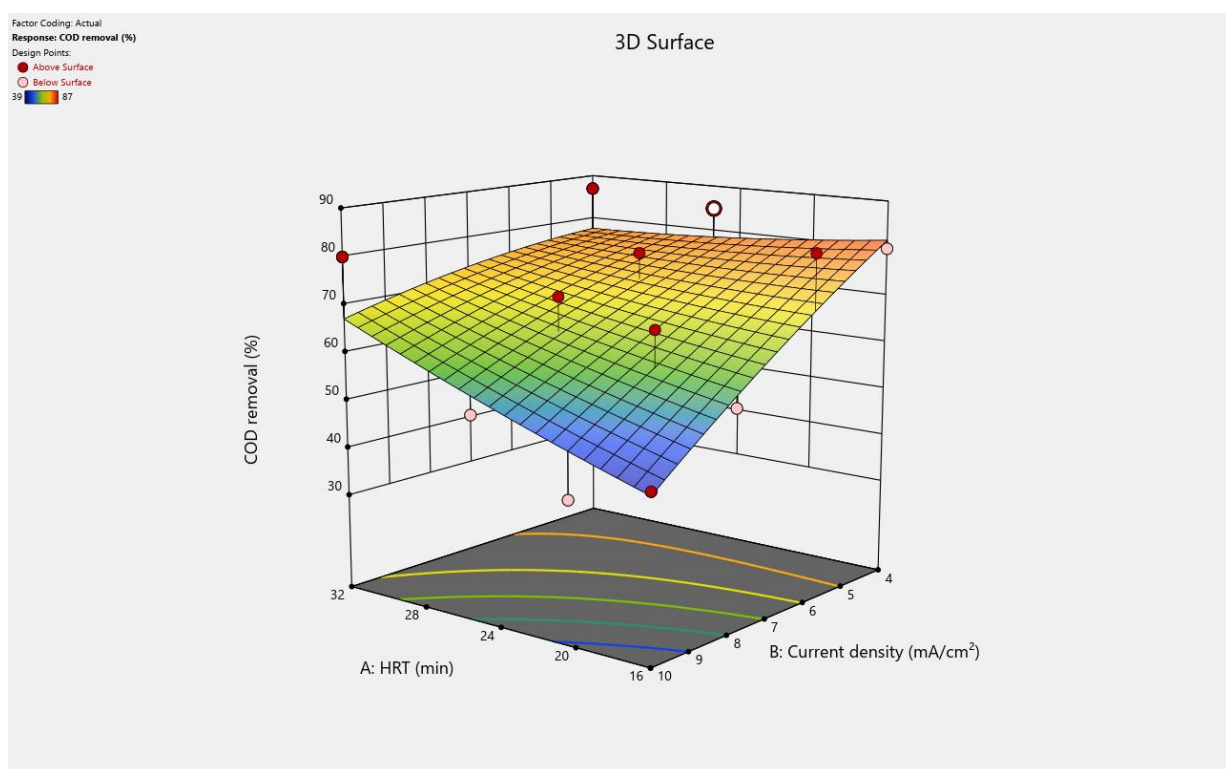


Figure 25: COD removal 3D surface

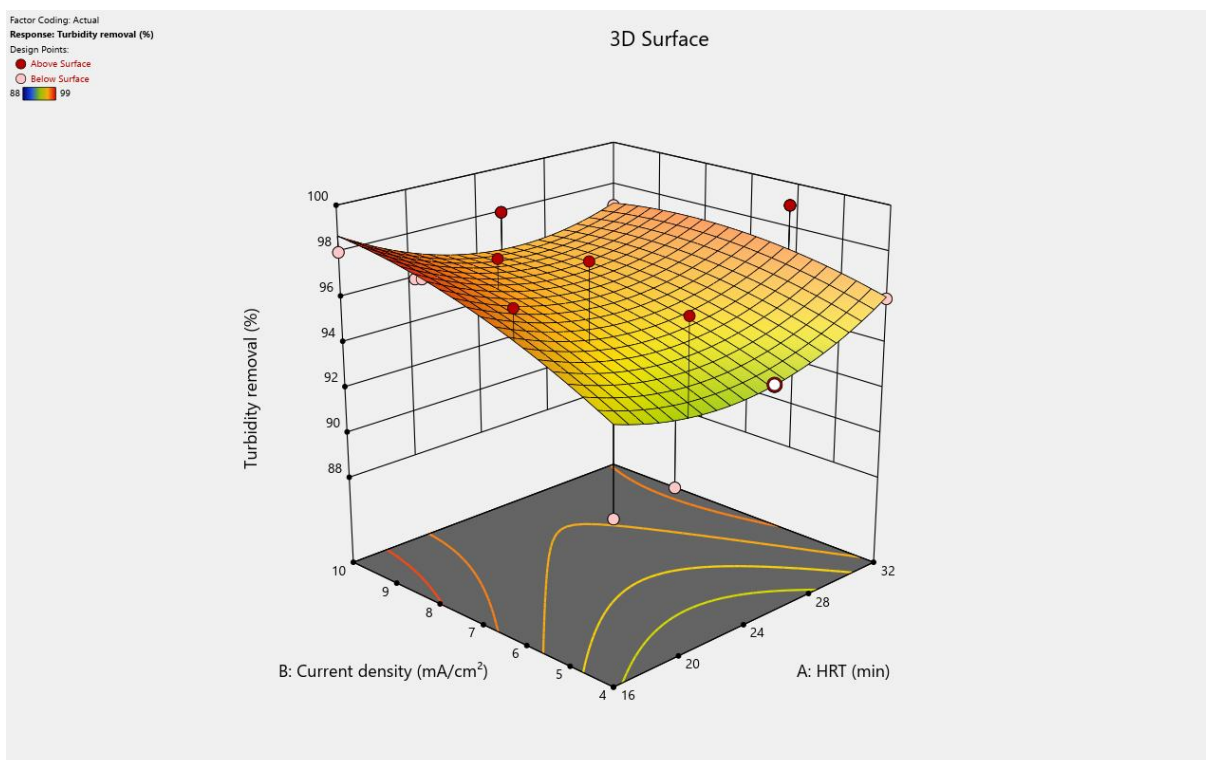


Figure 26: Turbidity removal 3D surface

Notably, the response surface for *E. coli* removal had a different shape. *E. coli* inactivation reached near-complete levels (below detection or >99.9% elimination) only at the combined high end of both factors, indicating a stronger interaction requirement. The contours suggested that at least a moderate HRT is needed, even if the current is high – for example, a very high current alone at short HRT did not fully inactivate bacteria, likely because insufficient exposure time prevents full disinfection. Conversely, very long HRT at too low current also failed to inactivate *E. coli* fully, implying a minimum threshold of current, and treatment intensity is required to damage the microorganisms. The optimal window for *E. coli* removal was therefore narrower and pushed towards the maximum settings of both parameters. This outcome underscores that microbial inactivation by EC is a more demanding objective than removing physical or chemical pollutants, as it may involve mechanisms like the generation of oxidants or extreme local pH, which require both time and sufficient current to manifest.

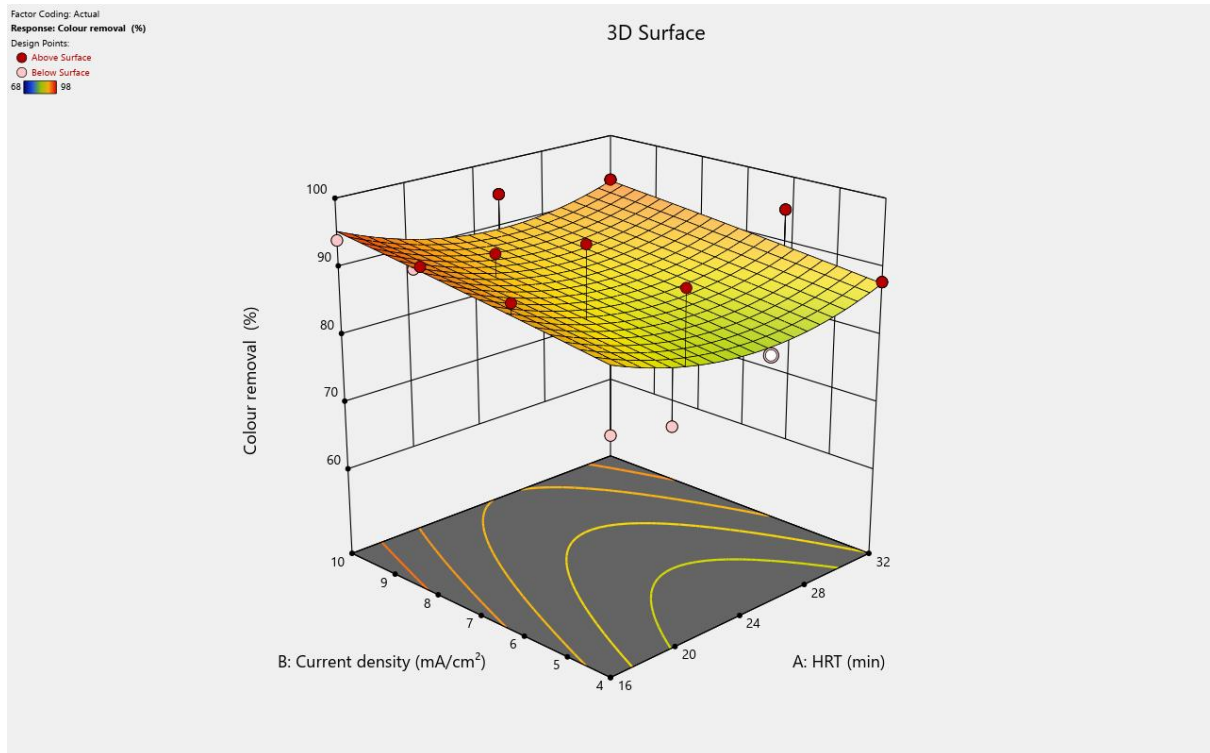


Figure 27: Colour removal 3D surface

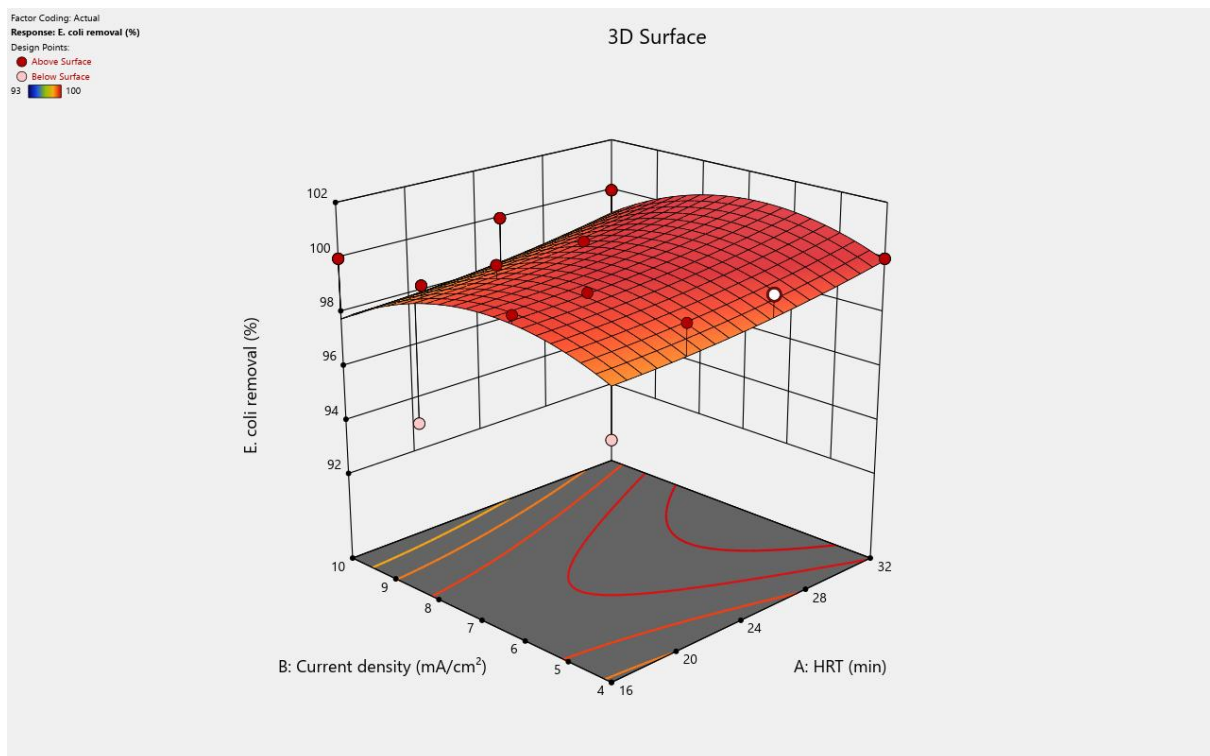


Figure 28: E. coli removal 3D surface

Across all these contour plots, an important observation is the existence of operational sweet spots: regions where high treatment efficiency is achieved without unnecessarily extreme conditions. For instance, running at the absolute maximum current density and longest HRT gave only marginal gains above the slightly lower, yet still high settings, while incurring much higher costs in energy and time. Therefore, the surfaces help identify a plateau region of performance, where one can select conditions that yield >90% removal for all key metrics. These represent the optimal treatment windows that balance performance and practicality. In summary, the RSM analysis indicates that the electrocoagulation process can be optimised to achieve excellent contaminant removal across multiple parameters simultaneously, and it provides a map of how HRT and current density can be tuned to reach those high-performance regions.

4.4 pH shifts and electrochemical reactions

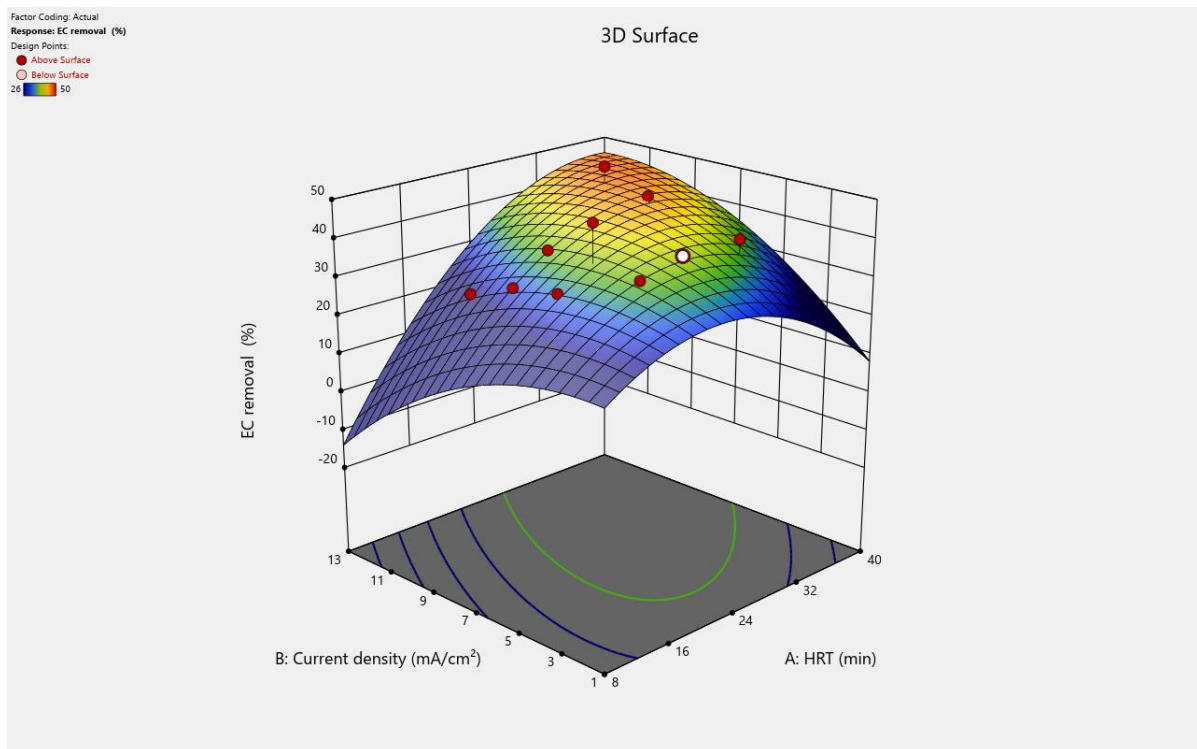


Figure 29: EC removal 3D surface

One noteworthy observation during the experiments was a significant shift in pH of the wastewater between the influent and effluent. The initial pH of the wastewater was near-neutral (~6.8), whereas after electrocoagulation treatment, the pH consistently rose to the mildly alkaline range (~9.0). This rise in pH by approximately 2.2 units is a direct consequence of the electrochemical reactions occurring in the reactor. At the cathode, water reduction occurs:



generating hydroxide ions (OH^-), which accumulate and elevate the solution. Meanwhile, at the anode, Al dissolves, and water can be oxidised (producing O_2 and H^+), but in practice, the cathodic OH^- production usually dominates the net pH effect. The result is that the treated water becomes mildly basic.

This shift to pH ~9.0 has important implications for the electrocoagulation process and the treated water quality. First, the formation of metal hydroxide coagulants is favoured in a slightly alkaline environment. For example, if Al electrodes are used, Al^{3+} ions generated at the anode rapidly hydrolyse to form species like $\text{Al}(\text{OH})_3$. The solubility of $\text{Al}(\text{OH})_3$ is lowest around pH 6.5–8.5, meaning it precipitates readily and can sweep out contaminants effectively in that pH window. In these experiments, reaching pH ~9.0 likely resulted in much of the Al converting to $\text{Al}(\text{OH})_4^-$ (the aluminate ion), which is soluble; however, the process of reaching that pH means a lot of $\text{Al}(\text{OH})_3$ flocs were formed on the way (around pH 7–8) to capture pollutants. If iron electrodes were used, the elevated pH would precipitate $\text{Fe}(\text{OH})_2$ and $\text{Fe}(\text{OH})_3$ efficiently. Thus, the pH increase is beneficial for coagulation as it ensures ample hydroxide availability to form metal hydroxide flocs. Other water constituents, such as heavy metals or hardness (Ca/Mg), may also precipitate more at higher pH, contributing to reductions in dissolved solids (this may partly explain any observed decreases in electrical conductivity).

Secondly, the high pH environment contributes to pathogen inactivation. Most bacteria, including *E. coli*, are sensitive to pH and do not survive well in conditions far from neutral. An effluent pH of around nine by itself has a bactericidal or bacteriostatic effect. Studies have found a strong positive correlation between the rise in solution pH during EC and the elimination of microbial indicators and organic. In our experiments, the near-neutral starting pH shifting to ~9.0 enhanced *E. coli* removal not just through coagulation/floatation, but also by creating an unfavourable environment (high pH) for the bacteria. Additionally, if any chlorine-containing species were present in the water (e.g. chloride ions in industrial wastewater), the electrochemical conditions (anodic oxidation at high pH) could generate hypochlorite (OCl^-) in situ, which is a known disinfectant. Even without the intentional addition of chlorine, oxygen evolution at the anode can form reactive oxygen species. The combination of these factors means the EC process provided a form of disinfection: *E. coli* counts in the effluent were drastically reduced, coinciding with the pH increase and the other chemical changes.

From a treatment standpoint, the shift to pH ~9 is advantageous for the reasons above, but it may have implications for downstream use of the water. Fortunately, the pH tends to stabilise

around this mild alkaline value and does not continue rising indefinitely; equilibrium is reached when the system's alkalinity and CO₂ exchange balance the OH⁻ production. In summary, the observed pH shift is an inherent part of electrocoagulation chemistry that aids the treatment process by promoting coagulation and disinfection, but it should be considered in the overall treatment train design.

4.5 Discussion of parameter scope and observed pH behaviour

Although initial pH and stirring intensity were not varied as independent factors in the experimental design, their role in process behaviour remains important when interpreting the results. In the present study, the wastewater was treated at its natural initial pH without chemical pre-adjustment, and mixing resulted only from reactor flow and electrochemically generated gas bubbles. The observed increase in pH during treatment is therefore interpreted as an inherent consequence of electrocoagulation chemistry rather than the effect of externally imposed pH control. This interpretation is consistent with the role of hydroxide generation in Al-based EC systems and supports the formation of destabilising and sweep flocculation species during treatment. The process was therefore assessed under conditions intended to reflect practical treatment operation rather than highly conditioned laboratory control.

4.6 Model validation with confirmatory tests

To evaluate the predictive accuracy and robustness of the developed RSM models, confirmatory experiments were conducted under operating conditions corresponding to the optimised ranges for HRT and current density. These experimental runs, runs 17 to 20, were not part of the original 16-point factorial matrix and served as independent validation points.

The responses measured included removal efficiencies for E. coli, colour, turbidity, COD, and electrical conductivity. The observed outcomes were then compared to the model's predicted values for similar input conditions.

Table 10: Validation experimental matrix

Validation experimental matrix				
Run No.	HRT (min)	Flow Rate (L/h)	Current Density (mA/cm ²)	Amperage (A)
17	29	1.971757241	5	2.9
18	27	2.117813333	5	2.9
19	22	2.599134545	7	4.1
20	18	3.17672	9	5.3

Table 11: Model validation results

Predicted vs Actual removal rates								
Response	Predicted 17	Actual 17	Predicted 18	Actual 18	Predicted 19	Actual 19	Predicted 20	Actual 20
E. coli	100%	100%	100%	100%	100%	100%	100%	100%
Colour	94%	95%	96%	97%	95%	97%	94%	95%
EC	30%	28%	33%	35%	33%	34%	31%	32%
Turbidity	98%	99%	98%	99%	98%	99%	98%	99%
COD	45%	43%	40%	38%	47%	50%	42%	40%

The confirmatory results demonstrate a strong alignment with model predictions:

- **E. coli:** Full removal was achieved consistently, validating the model's high R^2 and its ability to predict microbial inactivation effectively under optimised current and HRT conditions.
- **Colour and Turbidity:** Removals exceeding 95% (colour) and 99% (turbidity) were observed, closely matching the predicted surface responses. This confirms the reliability of the models for physical-chemical indicators of wastewater clarity
- **COD:** While some variability was present (ranging from 38% to 50% COD removal), all results fell within the expected confidence bounds of the model. These deviations are reasonable, considering the complex nature of organic pollutant matrices in slaughterhouse effluent and the known susceptibility of COD readings to sampling variability.
- **Electrical Conductivity:** EC reduction was consistent with predicted trends, showing moderate declines (28–35%) that align with expectations for AI-based EC systems operating at modest current densities.

Overall, the validation runs confirmed the reliability and generalizability of the fitted RSM models. The empirical equations effectively captured the treatment behaviour of the system and maintained predictive strength under new but bounded input conditions. The system performed optimally within the anticipated design space, achieving high pollutant removal efficiencies while maintaining operational feasibility.

These results support the practical application of the model for system optimisation and decision-making in full-scale implementation settings.

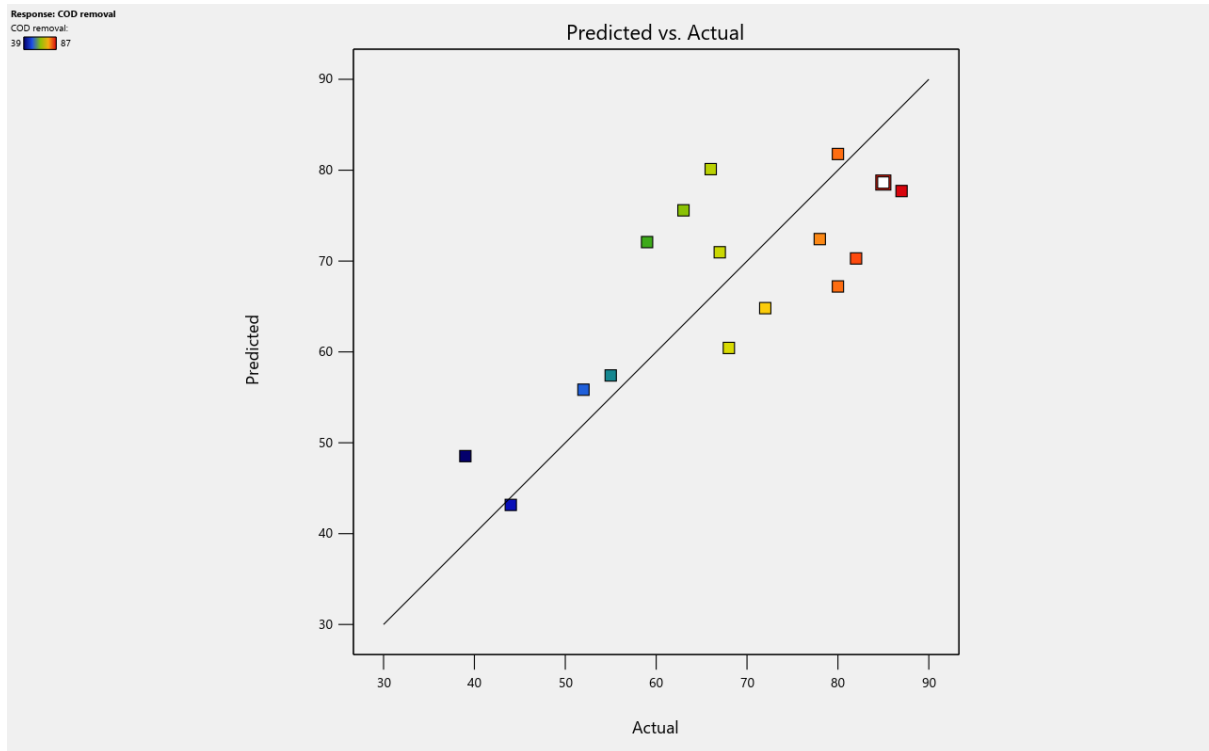


Figure 30: COD predicted vs actual

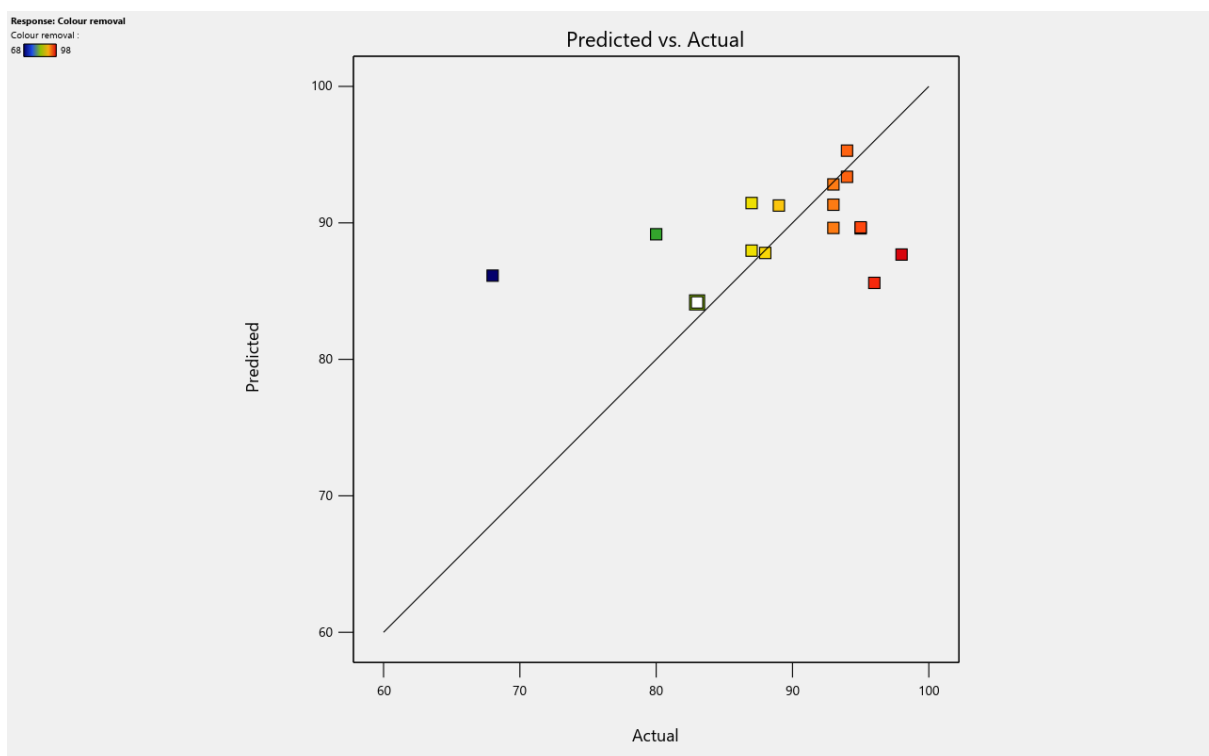


Figure 31: Colour predicted vs actual

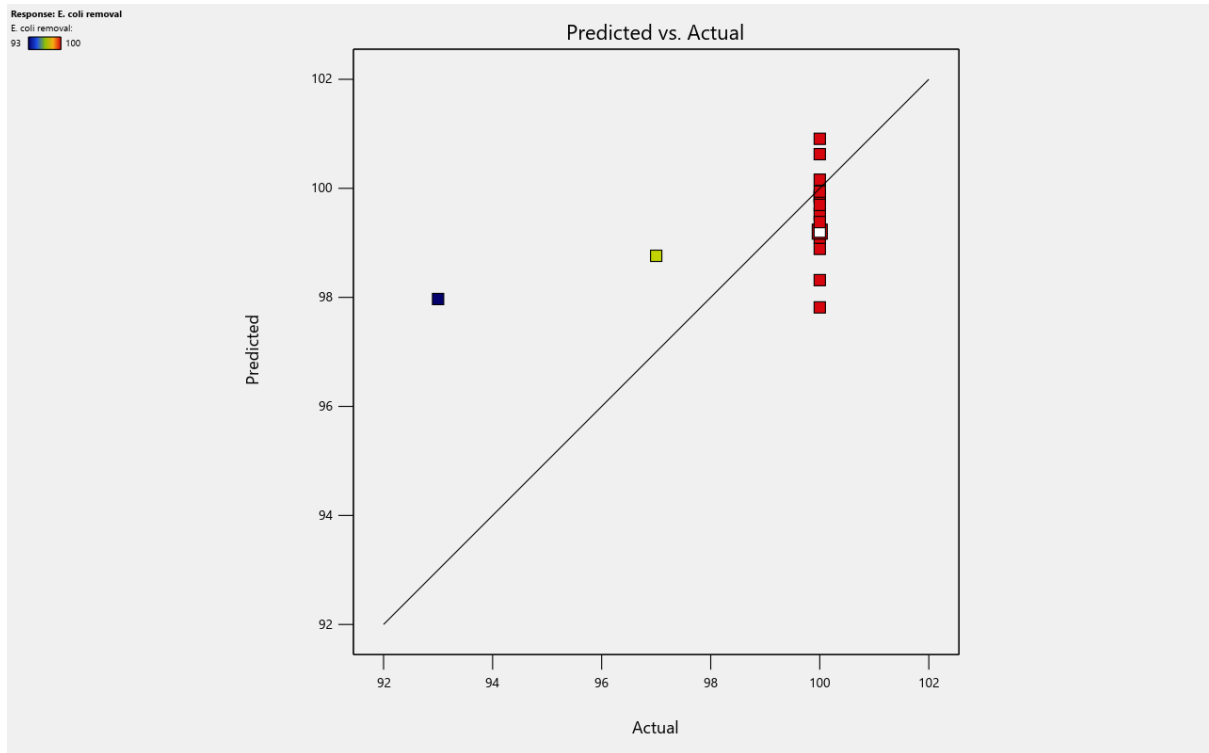


Figure 32: E. coli predicted vs actual

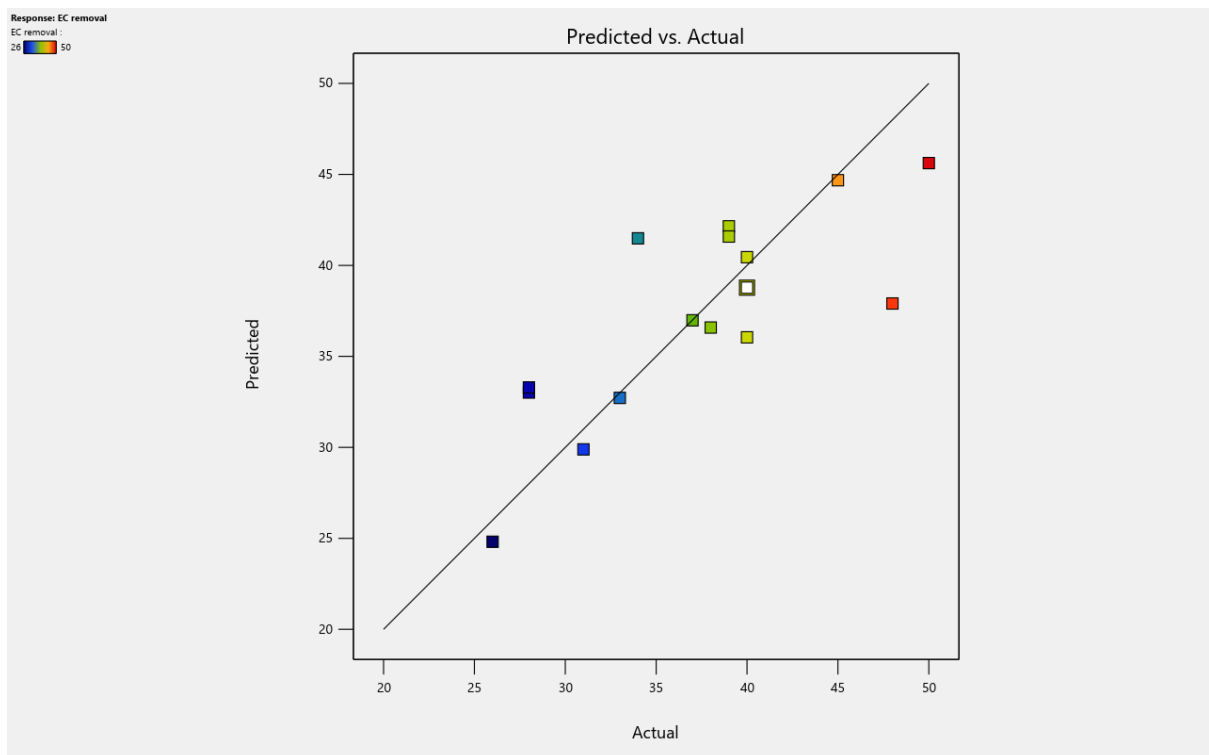


Figure 33: Electrical conductivity predicted vs actual

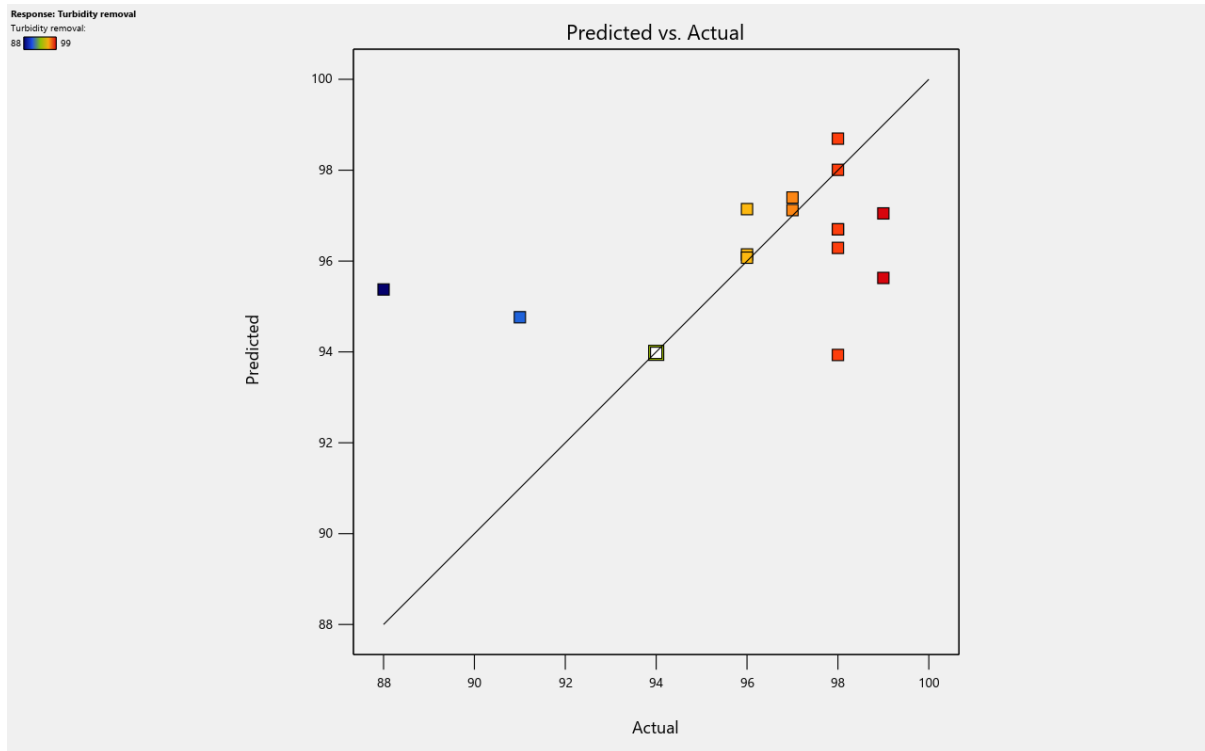


Figure 34: Turbidity: predicted vs actual

4.7 Interpretation of model outcomes

The empirical models developed through RSM demonstrate that both HRT and current density play decisive roles in electrocoagulation performance.”

Current density controls the rate of coagulant generation (via Faradaic dissolution of aluminium), while HRT governs the contact time for floc growth, particle collision, and charge neutralisation.

The observed constructive interaction between these parameters, particularly in COD and turbidity removal, confirms the importance of balancing electrochemical intensity and residence time rather than maximising independently.

The statistical strength of the quadratic models ($R^2 > 0.9$) underscores their adequacy for predicting process behaviour within the tested range. The response surfaces clearly illustrated diminishing returns beyond 8–9 mA/cm², reflecting a transition from charge-deficient to saturation regimes of coagulation.

4.8 Process performance trends

COD reduction was highly correlated with current density due to increased anodic dissolution and aluminium hydroxide availability. However, excessive current densities also promoted hydrogen bubble coalescence, causing turbulence and re-suspension of fine solids—consistent with the marginal decline in COD removal beyond 9 mA/cm².

Turbidity and colour removal improved consistently with longer HRT, affirming that time-dependent aggregation and sedimentation processes are critical for effective clarification.

Electrical conductivity reduction followed a similar pattern, decreasing with increasing current density as ionic species were removed through electro flocculation and complexation.

E. coli removal achieved complete inactivation across all conditions, supporting literature findings that local oxidation, pH shifts, and hydroxide film interactions collectively create an inhospitable environment for microbial survival.

4.9 Energy efficiency implications

Energy consumption increased exponentially with current density, in accordance with Joule's law and Faraday's principle.

Although higher current densities enhanced removal rates, their energy cost per cubic metre treated grew disproportionately.

The model indicated that the most energy-efficient operating window occurred around 25–28 min HRT and 7 mA/cm², where energy demand was moderate and treatment efficacy exceeded 90 % for most parameters.

These findings highlight the potential for significant energy savings through careful optimisation rather than operating at maximum current densities.

4.10 Design improvements and recommendations

Based on the experimental findings and observations, several design and operational improvements are recommended to enhance the performance, scalability, and robustness of the electrocoagulation system. These recommendations also draw upon established best practices from literature and engineering principles for electrochemical reactor design:

- Vertical Flow Reactor Configuration:

- It is advised to consider a vertical reactor design (as opposed to a simple horizontal batch tank) for future implementations. In a vertical up-flow or down-flow EC reactor, the natural buoyancy of hydrogen gas bubbles and the differing densities of flocs can be better managed. Specifically, a vertical configuration allows a clearer segregation of zones: a lower sedimentation zone for dense flocs/sludge, a central reaction zone where the electrodes are located, and an upper flotation zone where lighter particles and froth accumulate. This stratification means sludge can gravitate to the bottom and be drained or collected, while floating matter can be skimmed from the top, all without interrupting the treatment. For continuous systems, vertical flow is commonly used for smaller throughputs or experimental setups, and it can improve hydraulic distribution by preventing dead zones. Thus, a vertical EC reactor with an inlet at the bottom and an outlet at the top, with provisions to capture float and sludge, would enhance removal efficiency and make solid-liquid separation easier, compared to a horizontal or completely mixed set seen in.
- Effective Sludge Management:
 - The EC process produces sludge in the form of coagulated contaminants and metal hydroxide flocs. To maintain performance, this sludge must be managed proactively. Design improvements should include features like conical bottoms or hoppers for sludge collection, automatic or periodic sludge blow-down valves, and slow mixing or scraping mechanisms to gather sludge. Regular sludge removal prevents re-suspension of settled solids and avoids loss of reactor volume due to sludge accumulation. Moreover, proper sludge handling reduces the risk of short-circuiting (where an electrical path could form through accumulated sludge). It is also recommended to characterise the sludge and explore beneficial reuse options. Electrocoagulation sludge often contains metal hydroxides that can act as coagulants or adsorbents themselves; indeed, studies suggest EC sludge can be recycled as a polymeric coagulant aid or used in other pollutant removal applications. While reuse may depend on contaminant content (in industrial wastewater, it might contain hazardous substances), at a minimum, the sludge should be dewatered and disposed of according to environmental regulations. Implementing a sludge management plan will ensure the EC system runs continuously at high efficiency without clogging or excessive downtime for cleaning. Annexure D: Sludge formation
- Optimised Electrical Configuration (Wiring and Electrodes):

- The electrical setup of the reactor (electrode arrangement and wiring connections) has a significant impact on both performance and energy efficiency. For scale-up involving multiple electrode pairs, a monopolar parallel connection is generally recommended for its simplicity and even current distribution. In a monopolar parallel setup, all anodes are tied to the positive terminal and all cathodes to the negative, ensuring each pair sees the same voltage. This typically offers a good efficiency-to-cost ratio and is easier to design and maintain. Alternatively, a bipolar series configuration (electrodes not directly wired but polarised in sequence) can achieve higher treatment effectiveness in some cases by creating strong electric fields between adjacent plates, but this comes at the cost of requiring higher voltage and more complex power management. Depending on the needed effluent quality, one could consider a bipolar setup if pushing for maximum removal, but for most purposes, a well-designed monopolar system is sufficient and more economical.
- In terms of electrode design, using multiple smaller electrodes rather than a single pair can improve scalability and reduce resistance (since each electrode pair spans a shorter distance). Reconfiguring the wiring to have electrodes in parallel can reduce the overall cell voltage for a given current (thus saving energy). Ensuring electrodes are evenly spaced and using conductive bus bars of adequate gauge will minimise resistive losses in the connections. The wiring should also allow flexibility for maintenance – for instance, the ability to disconnect individual electrode pairs for cleaning or replacement without shutting down the whole system. Another recommendation is to periodically reverse polarity or switch the anode/cathode roles if using identical material electrodes; this can help minimise passivation and distribute electrode wear more evenly. In summary, careful electrical design (proper wiring configuration, material selection, spacing) will improve the EC reactor's efficiency and longevity.
- Mixing and Aeration Controls:
 - (Optional improvement) While not explicitly studied in our RSM, effective mixing ensures uniform exposure of the wastewater to the electric field and coagulant. Future designs could incorporate gentle mixing or use the H₂ bubbles generated as a natural mixer. Additionally, controlling aeration or introducing slight airflow can help float flocs to the top (electro-flotation constructive

interaction). These measures, while minor, can complement the main factors to improve overall performance.

- Safety and Automation:
 - As a general recommendation, any full-scale EC system should include safety mechanisms and automation. This includes electrical safety (proper insulation, grounding, and shielding of the power supply lines due to the presence of water and electricity), as well as sensors for parameters like current, voltage, pH, and temperature. Automated controls can adjust the current in real-time to maintain a setpoint (for example, targeting a specific effluent turbidity or responding to fluctuations in influent quality). Automation also extends to sludge management, e.g., timers or sensors could trigger sludge purge valves or shut down the system if abnormal conditions occur (like a short-circuit from sludge bridging electrodes). These considerations ensure that the process remains reliable and consistent under varying conditions.

By implementing the above design improvements, the electrocoagulation system can be made more efficient, easier to operate, and better suited for scaling up. A vertical, well-mixed reactor with automated sludge removal and optimised electrical connections will yield even better performance than the batch setup used in our experiments. Furthermore, such enhancements pre-emptively address practical challenges one might face in an industrial setting, such as continuous operation, maintenance downtime, etc. The recommendations are in line with recent advances in electrocoagulation reactor design, which emphasise configuration, flow dynamics, and ease of operation as keys to success.

CHAPTER 5 CONCLUSIONS AND FUTURE WORK

5.1 Conclusions

This study presented a comprehensive optimisation of EC parameters, specifically hydraulic retention time and current density, for enhanced wastewater treatment, using a full-factorial RSM approach. The main findings can be summarised as follows:

- High treatment efficacy:
 - Electrocoagulation proved to be highly effective in removing a spectrum of pollutants from the industrial wastewater tested. Under optimised conditions, the process achieved substantial reductions in chemical oxygen demand, near-complete elimination of turbidity and colour, and significant *E. coli* inactivation. The concurrent removal of microbial and chemical contaminants highlights EC as a robust single-step treatment method for improving water quality. These outcomes underscore the potential of EC as a viable technique for treating complex industrial effluents, offering high contaminant removal efficiency with straightforward equipment.
- Empirical model and optimisation:
 - A second-order polynomial model was successfully fitted to experimental data, capturing the relationships between HRT, current density, and treatment responses with high accuracy (R^2 above 0.9). The RSM model enabled the identification of optimal operating windows where treatment performance is maximised. Both HRT and current density were found to be significant factors; optimum performance requires a balanced combination of sufficient time and electrical dosage. The model facilitated a nuanced understanding of process behaviour, such as diminishing returns at extreme settings and the interplay (interaction) between factors. Model predictions were validated by experiments, lending confidence to the model's use for guiding operational decisions.
- Factor effects and mechanisms:
 - Current density emerged as the primary driver for contaminant removal, due to its control over coagulant generation and oxidant production, while retention

time was essential for allowing the treatment mechanisms to fully act (coagulation, flocculation, flotation, and disinfection). The process benefits from the water chemistry changes induced during electrolysis, notably the pH rising to ~9, which enhanced coagulation and aided pathogen kill. The shift to alkaline pH and the associated formation of metal hydroxide flocs were identified as beneficial phenomena inherent to EC. However, it was noted that extremely high current densities do not proportionally increase removal and instead waste energy, whereas adequate HRT is indispensable to reach high removal targets.

- Energy considerations:
 - The study highlighted the trade-off between treatment efficiency and energy consumption. While electrocoagulation can achieve impressive pollutant removals, doing so at very high current and long HRT incurs significant electrical energy use. The specific energy consumption ranged widely across the tested conditions, demonstrating that operational choices have a major impact on the process's economic and environmental footprint. By quantifying these trends, the research provides a basis for optimising EC not only for performance but also for energy efficiency. The results suggest that there is room to reduce energy requirements through optimised parameter selection and that EC can be competitive with conventional treatments if carefully managed. Notably, in optimised scenarios, the EC process can operate at an energy cost that is feasible for industrial applications (on par with other advanced treatments), especially considering the elimination of the need for chemical coagulants.
- EC's potential in wastewater treatment:
 - Overall, the findings reinforce that electrocoagulation is a promising and potent technology for industrial wastewater treatment. It offers the ability to tackle multiple contaminants in one unit process, leveraging electrically generated reagents instead of chemical additions. This means fewer chemicals in the supply chain and often less sludge production compared to traditional chemical coagulation. The treated water quality achieved in this study, clear, decoloured, with reduced organic load and pathogens, indicates that EC can produce effluents suitable for discharge or even reuse (after polishing) in many industrial contexts. These results contribute to the growing body of evidence that EC is a viable alternative or complement to conventional treatment methods for

industries such as textiles, food processing, petrochemical, and others known for wastewater with high turbidity, colour, and refractory organics.

In conclusion, the optimisation of HRT and current density via RSM has provided both practical operating guidelines and deeper insight into the electrocoagulation process. The study demonstrates that by adjusting these key parameters, electrocoagulation can be steered to achieve rigorous treatment goals. With its strong performance in removing diverse pollutants and potential advantages like on-site reagent generation and sludge minimisation, electrocoagulation stands out as a compelling solution for enhanced wastewater treatment in industrial settings. The conclusions drawn here form a basis for scaling up the process and integrating it into real-world treatment trains, with the expectation of positive impacts on water pollution control and resource reuse.

5.2 Study limitations

The present study was limited to the optimisation of two operating variables, namely hydraulic retention time and current density, within a laboratory-scale continuous flow electrocoagulation reactor. Other parameters known to influence EC performance, such as initial pH, mixing intensity, inter-electrode spacing, and wastewater conductivity adjustment, were not varied experimentally and therefore fall outside the optimisation framework developed here.

The 16-run factorial design used for empirical model development was not replicated, and no preliminary OFAT screening was conducted. Although this limits the direct quantification of experimental uncertainty within the primary design matrix, the study nevertheless provides a structured and practically relevant optimisation framework supported by statistical model fitting and independent confirmatory validation experiments. The findings should therefore be interpreted as an empirically supported optimisation study within a defined scope rather than as a fully replicated statistical design covering all possible EC variables.

5.3 Recommendations for Future Work

Future work should include replicated experimental runs and repeated centre-point testing to improve the statistical robustness of the developed optimisation models and to quantify experimental variability more explicitly. Further studies should also evaluate the effect of additional EC operating variables, including initial pH, inter-electrode distance, conductivity adjustment, and controlled mixing intensity.

Since the present work intentionally evaluated electrocoagulation without chemical preconditioning of the wastewater, future investigations could compare natural-feed treatment with pH-adjusted treatment to determine whether any further improvement in removal efficiency justifies the additional chemical input. Likewise, future work could compare flow-induced and gas-induced mixing with controlled mechanical agitation to quantify whether the latter offers sufficient benefit for practical implementation.

Additional research should also investigate scale-up to pilot or commercial continuous-flow reactors, including sludge management, hydraulic design optimisation, alternative wiring configurations, and the trade-off between pollutant removal and electrical operating cost.

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ANNEXURE A: LAB RESULTS

T R LABORATORY SERVICES CC LABORATORY SERVICES AND WATER SOLUTIONS

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UNIT 8
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GRIPPEN RD
BARTLETT
BOKSBURG

CERTIFICATE OF ANALYSIS

CORNEL GNADE
PRIVATE
PO BOX

TEL: 066 477 2260

EMAIL: gnaDECORNEl3@gmail.com

SAMPLE NO. :9205
SAMPLE NAME :Cornel
SAMPLE TYPE :Water
DATE RECEIVED :05/11/2025
DATE TESTED :05/11/2025

PAGE 1 OF 2
ORDER NO. :
DATE REPORTED : 10/11/2025

TEST	0	1	2	3	4	UNIT
E coli	100	7	Not detected	Not detected	Not detected	count/ml
Colour	201	26	14	5	9	mg/l Pt
pH value at 25°C	6.86	8.52	8.84	9.07	8.93	
Conductivity at 25°C	116.2	83.24	71.73	60.91	73.73	mS/m
Turbidity	34.6	1.39	0.65	0.5	0.76	NTU
COD	748	453	238	247	255	mg/l

TEST	5	6	7	8	9	UNIT
E coli	Not detected	Not detected	3	Not detected	Not detected	count/ml
Colour	14	23	41	27	35	mg/l Pt
pH value at 25°C	9.23	9.37	8.75	9.13	9.28	
Conductivity at 25°C	58.31	64.48	83.82	71.01	69.83	mS/m
Turbidity	0.99	1.04	3.02	1.53	2.11	NTU
COD	148	305	149	213	110	mg/l

Figure 35: Lab analyses - page 1

SAMPLE NO.	:9205
SAMPLE NAME	:Cornel
SAMPLE TYPE	:Water

PAGE	2 OF 2
ORDER NO.	:
DATE REPORTED	: 10/11/2025

TEST	10	11	12	13	14	UNIT
E coli	Not detected	Not detected	Not detected	Not detected	Not detected	count/ml
Colour	24	14	64	11	13	mg/l Pt
pH value at 25°C	9.26	9.19	9.06	9.15	9.13	
Conductivity at 25°C	69.86	77.58	71.33	76.25	80.08	mS/m
Turbidity	1.44	0.77	3.98	0.44	0.7	NTU
COD	99	133	161	274	338	mg/l

TEST	15	16	17	18	19	UNIT
E coli	Not detected	Not detected	Not detected	Not detected	Not detected	count/ml
Colour	13	10	11	7	6	mg/l Pt
pH value at 25°C	9.12	9.3	9.23	9.19	9.35	
Conductivity at 25°C	86.39	69.44	83.45	75.04	77.03	mS/m
Turbidity	0.64	0.81	0.36	0.3	0.23	NTU
COD	416	360	424	461	371	mg/l

TEST	20	UNIT
E coli	Not detected	count/ml
Colour	10	mg/l Pt
pH value at 25°C	9.26	
Conductivity at 25°C	79.34	mS/m
Turbidity	0.33	NTU
COD	448	mg/l

The above results reflects the analysis only of the sample received and analysed as per your instructions.
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Reported by: Tracy Rees
Tracy Rees (BSc Chemistry Hons)

Figure 36: Lab analyses - page 2

ANNEXURE B: EXPERIMENTS AND SETUP



Figure 37: Untreated sample retrieval



Figure 39: EC reactor - de-energised

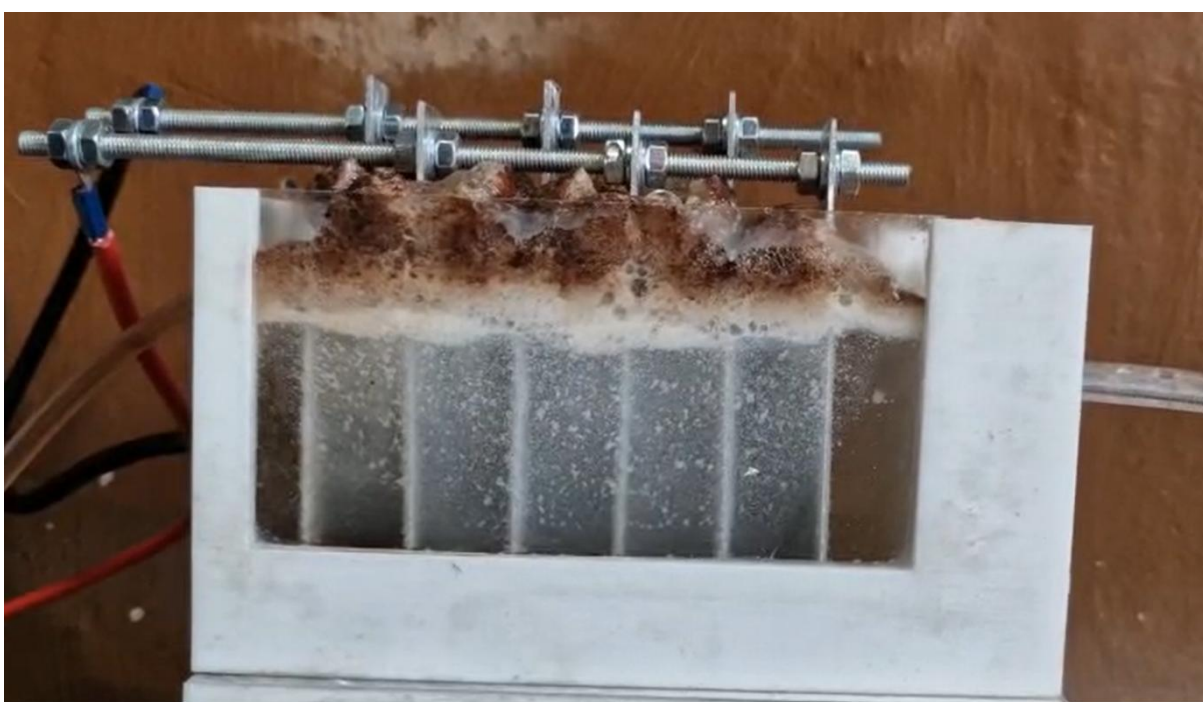


Figure 38: EC reactor - energised

ANNEXURE C: SAMPLES



Figure 40: Untreated vs treated samples - without filtration

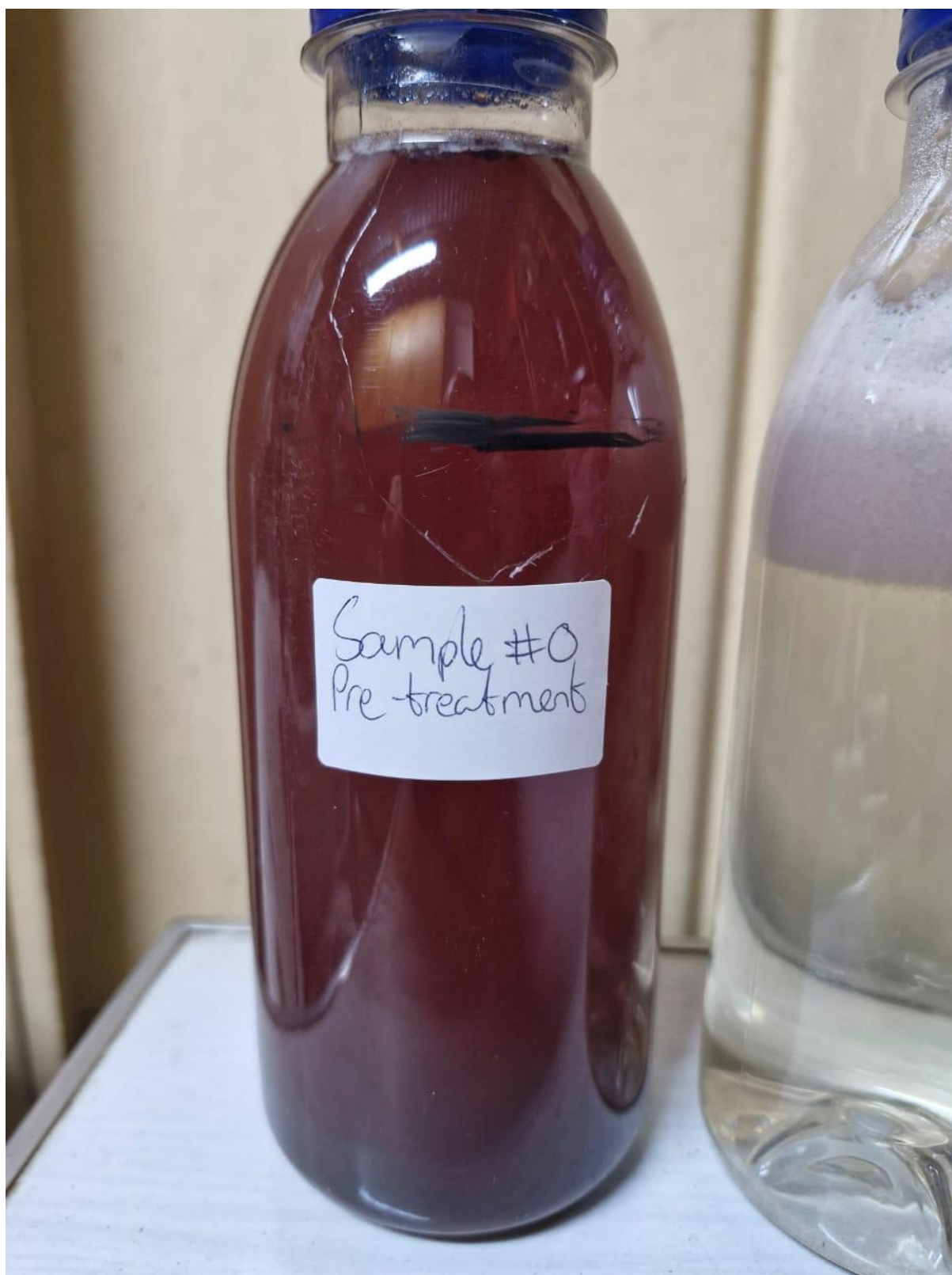


Figure 41: Untreated sample



Figure 42: Sample 1 without filtration

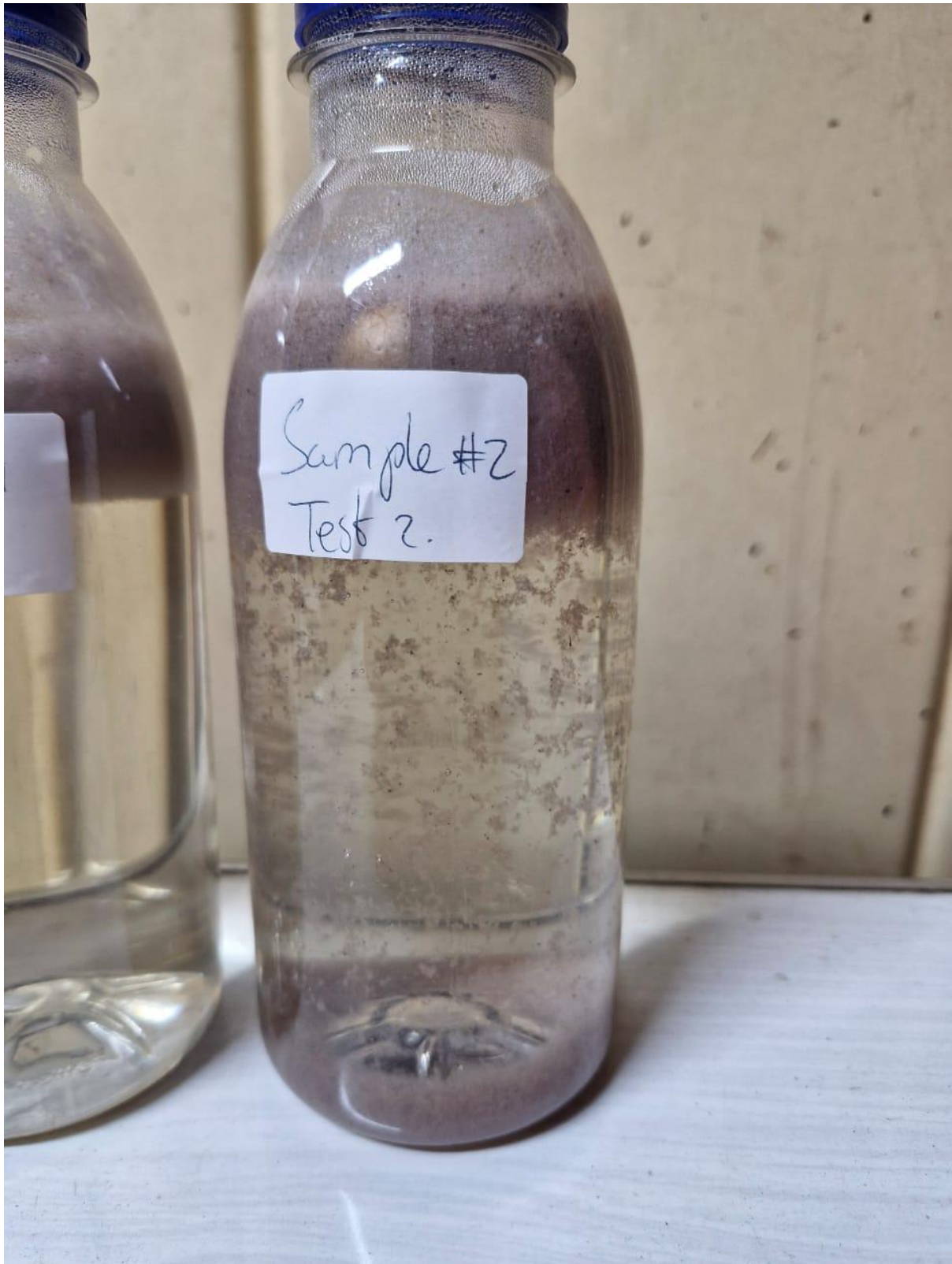


Figure 43: Sample 2 with active flocculation



Figure 44: All samples

ANNEXURE D: SLUDGE FORMATION

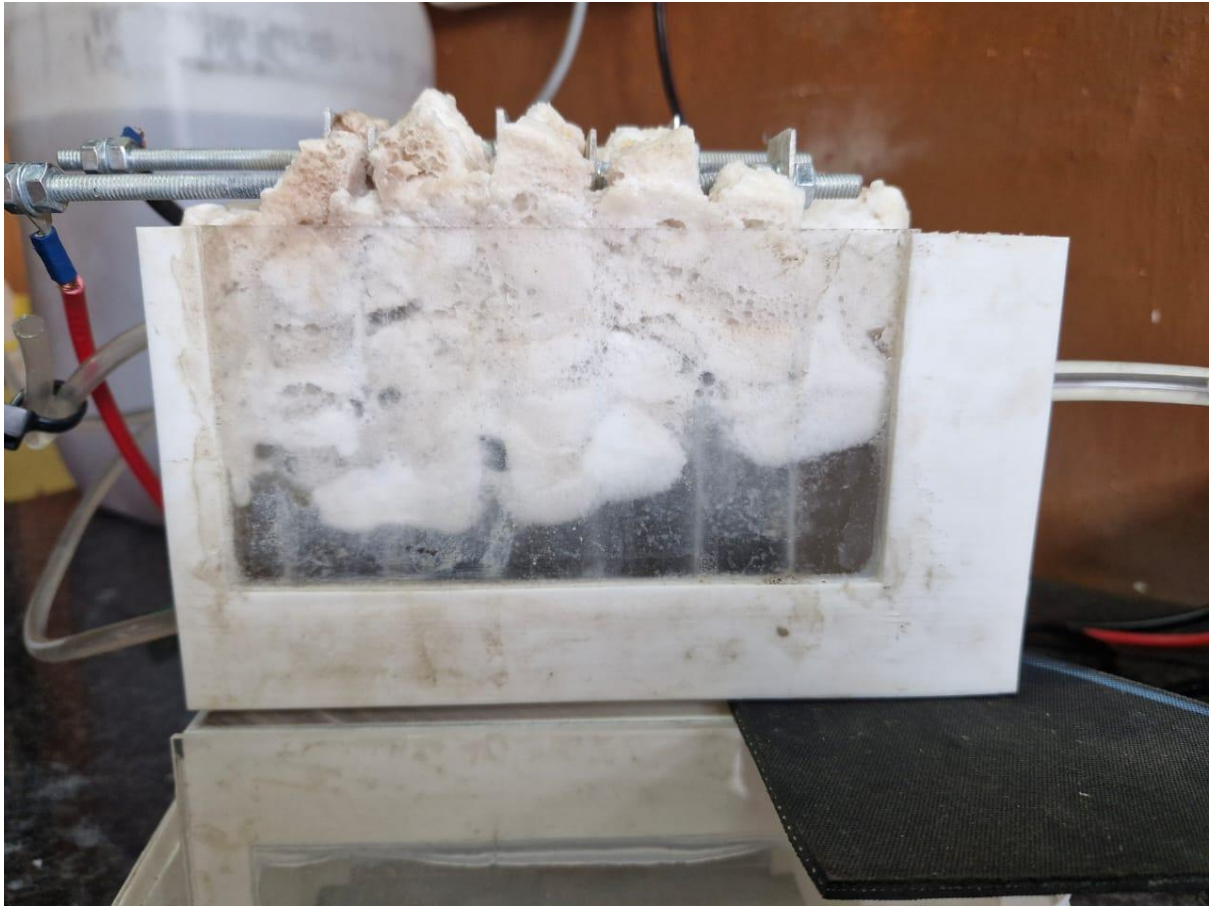


Figure 45: Unskimmed sludge formation

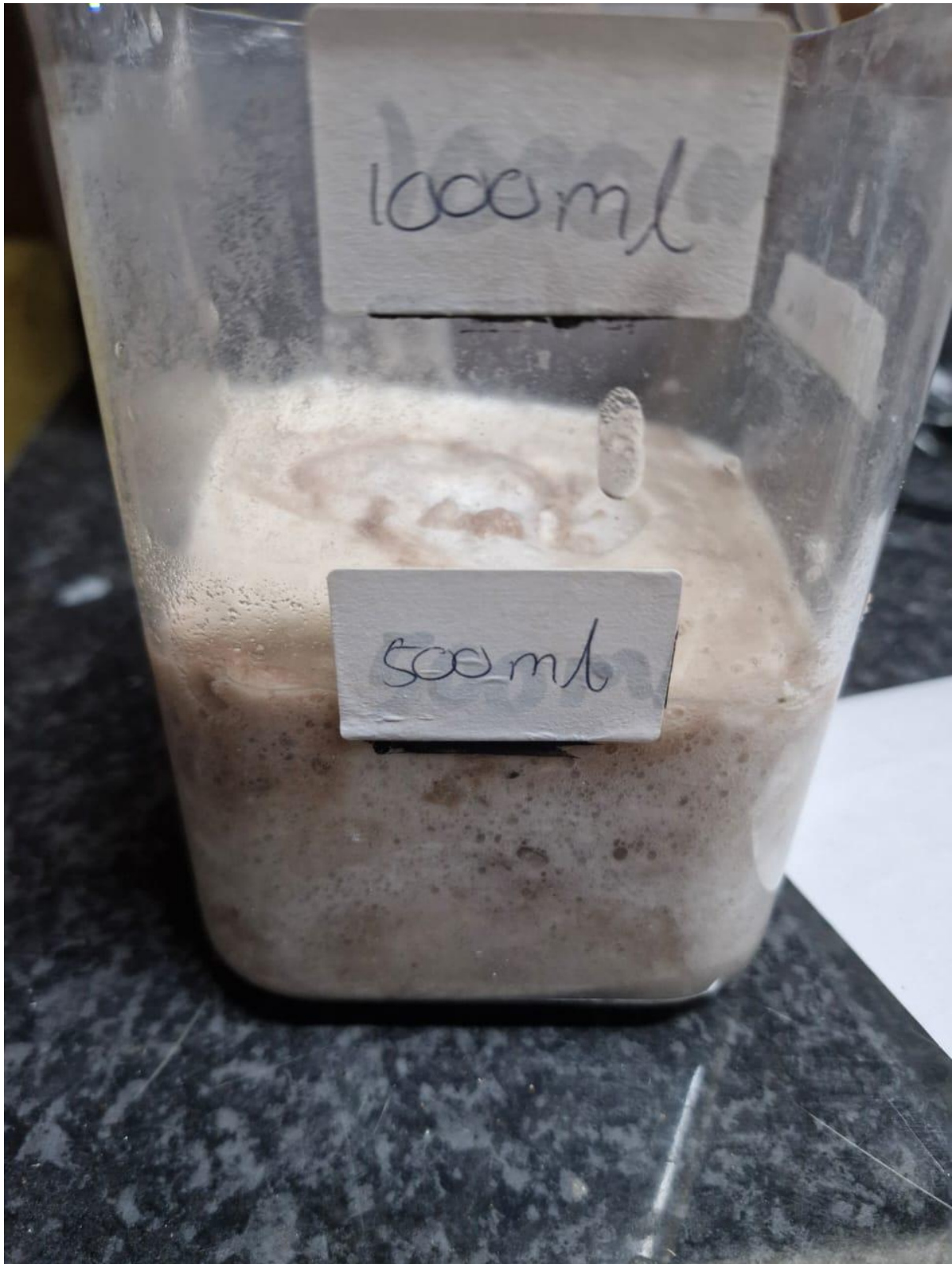


Figure 46: Skimmed sludge during experiment seven

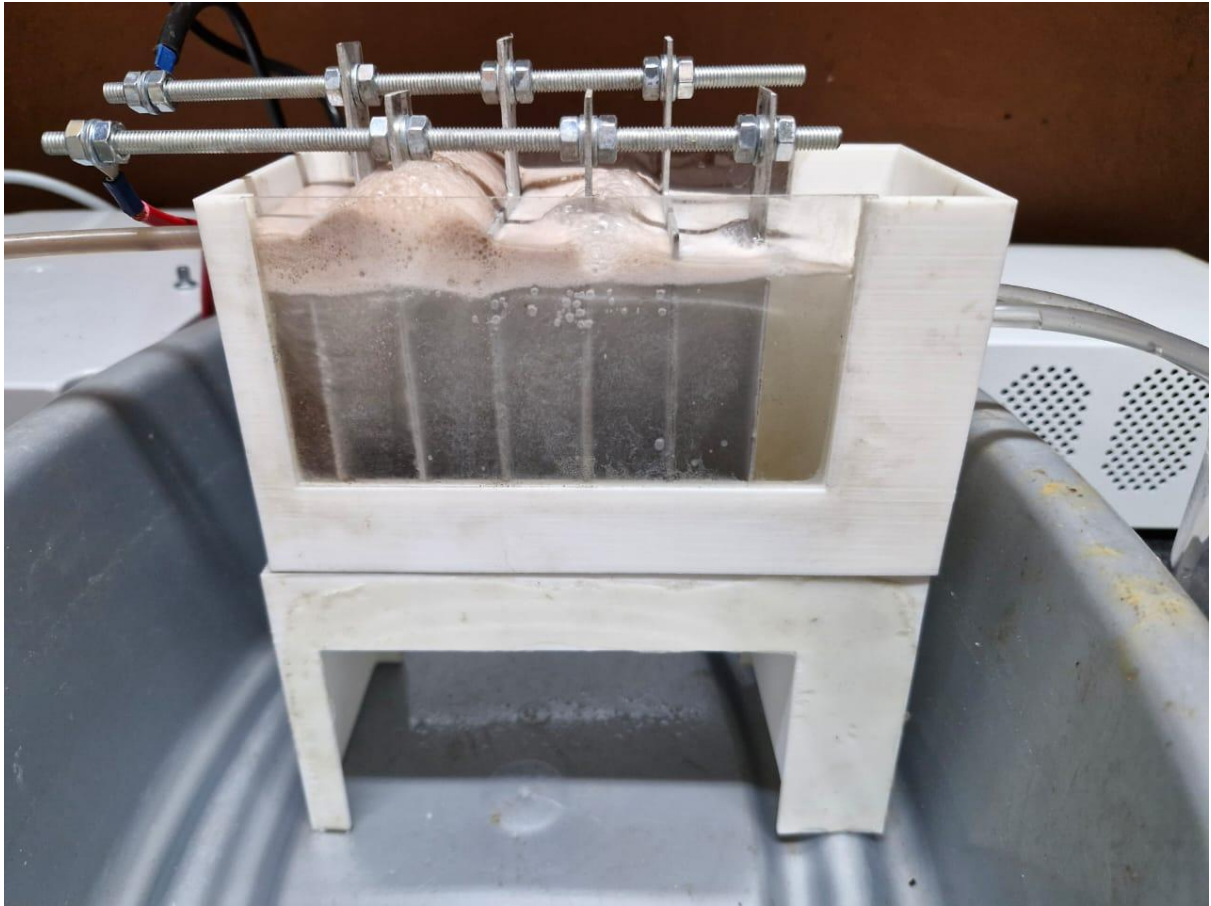


Figure 47: Maintained sludge levels

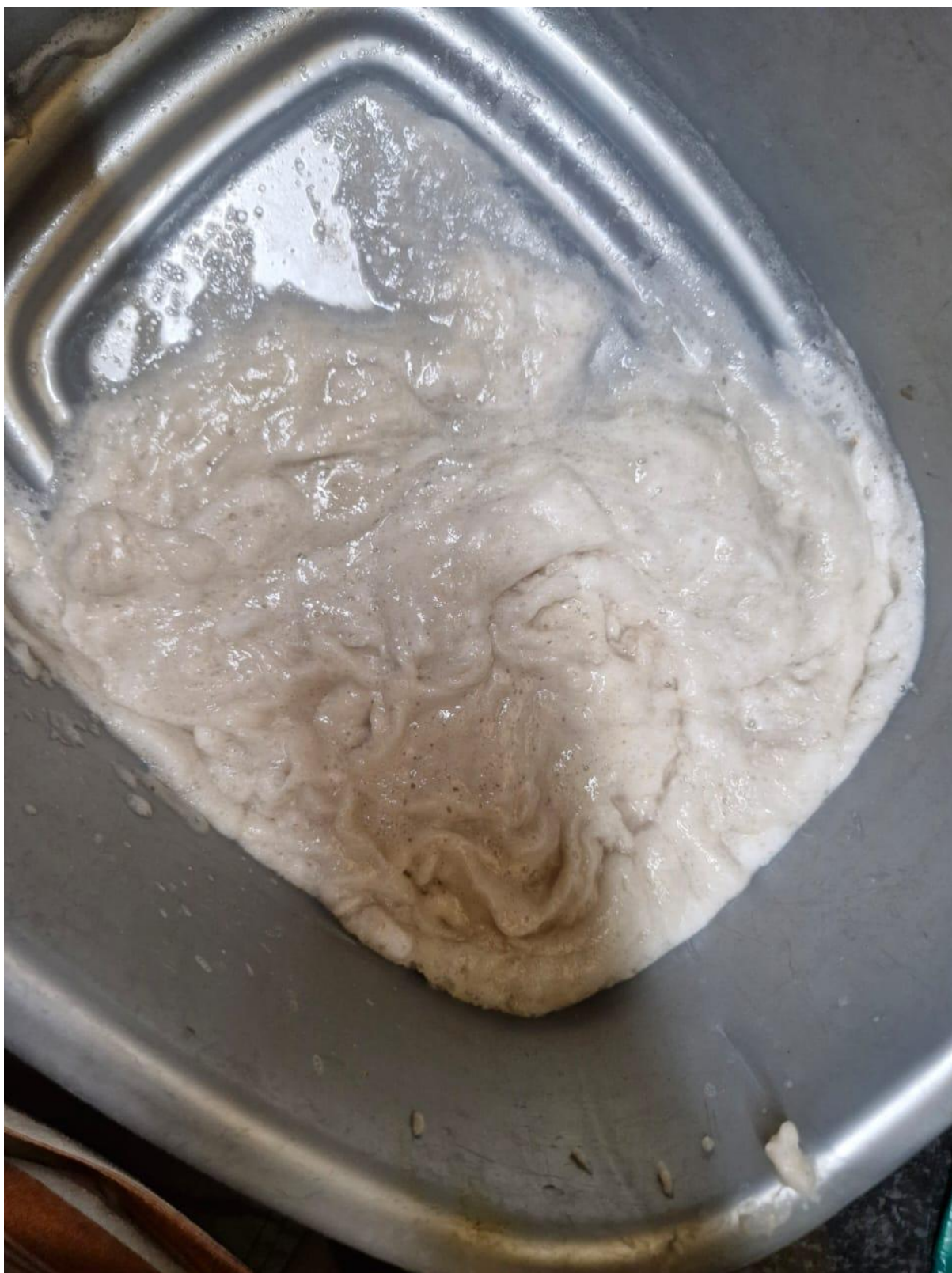


Figure 48: Skimmed sludge post-treatment

ANNEXURE E: ELECTRODES



Figure 50: Used vs unused Al electrodes

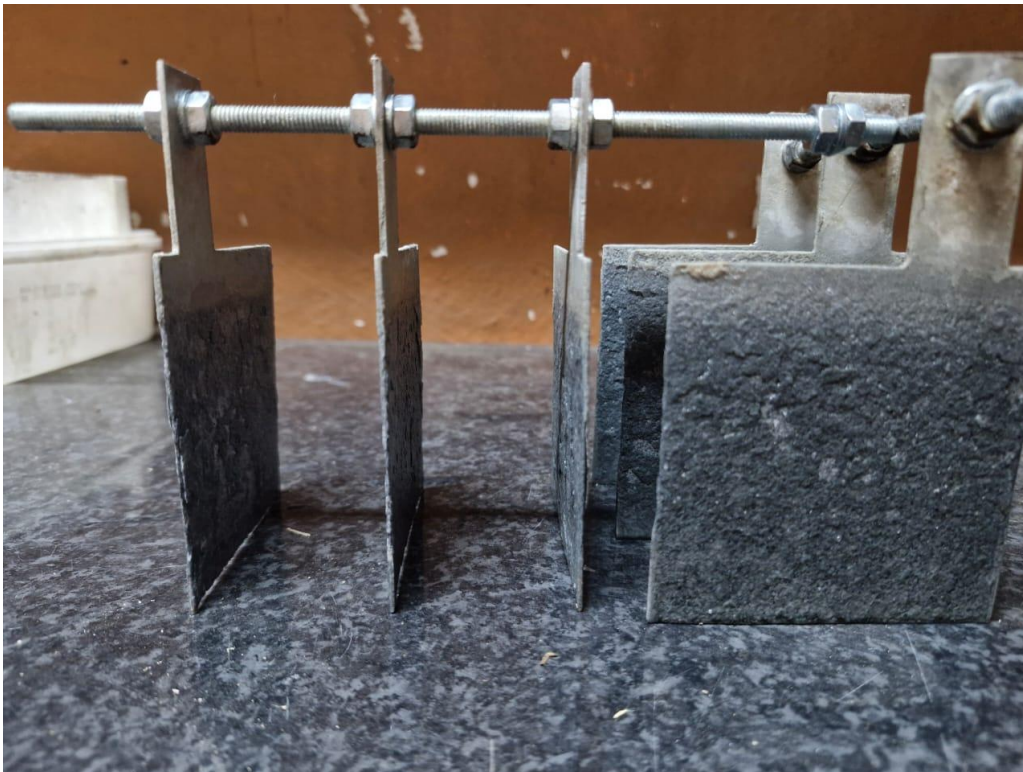


Figure 49: Deteriorating electrodes

ANNEXURE F: ANOVA

ANOVA for Quadratic model

Response 1: COD removal

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	1968.01	5	393.60	3.21	0.0551	not significant
A-HRT	221.30	1	221.30	1.80	0.2090	
B-Current density	1319.30	1	1319.30	10.75	0.0083	
AB	244.94	1	244.94	2.00	0.1881	
A ²	2.29	1	2.29	0.0187	0.8940	
B ²	7.56	1	7.56	0.0616	0.8090	
Residual	1227.43	10	122.74			
Cor Total	3195.44	15				

Factor coding is **Coded**.

Sum of squares is **Type III - Partial**

The **Model F-value** of 3.21 implies there is a 5.51% chance that an F-value this large could occur due to noise.

P-values less than 0.0500 indicate model terms are significant. In this case B is a significant model term. Values greater than 0.1000 indicate the model terms are not significant. If there are many insignificant model terms (not counting those required to support hierarchy), model reduction may improve your model.

Figure 52: COD ANOVA

ANOVA for Quadratic model

Response 3: Colour removal

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	133.50	5	26.70	0.3665	0.8603	not significant
A-HRT	8.19	1	8.19	0.1124	0.7444	
B-Current density	67.96	1	67.96	0.9329	0.3569	
AB	0.3757	1	0.3757	0.0052	0.9442	
A ²	57.21	1	57.21	0.7853	0.3963	
B ²	0.0625	1	0.0625	0.0009	0.9772	
Residual	728.44	10	72.84			
Cor Total	861.94	15				

Factor coding is **Coded**.

Sum of squares is **Type III - Partial**

The **Model F-value** of 0.37 implies the model is not significant relative to the noise. There is a 86.03% chance that an F-value this large could occur due to noise.

P-values less than 0.0500 indicate model terms are significant. In this case there are no significant model terms. Values greater than 0.1000 indicate the model terms are not significant. If there are many insignificant model terms (not counting those required to support hierarchy), model reduction may improve your model.

Figure 51: Colour ANOVA

Fit Statistics

Std. Dev.	11.08	R²	0.6159
Mean	67.31	Adjusted R²	0.4238
C.V. %	16.46	Predicted R²	-0.3462
		Adeq Precision	5.6919

A negative **Predicted R²** implies that the overall mean may be a better predictor of your response than the current model. In some cases, a higher order model may also predict better.

Coefficients  Coded Equation  Actual Equation

Coefficients in Terms of Coded Factors

Factor	Coefficient Estimate	df	Standard Error	95% CI Low	95% CI High	VIF
Intercept	68.15	1	5.71	55.42	80.88	
A-HRT	4.99	1	3.72	-3.29	13.28	1.00
B-Current density	-12.28	1	3.75	-20.62	-3.93	1.02
AB	7.03	1	4.98	-4.06	18.12	1.02
A ²	0.8598	1	6.29	-13.15	14.87	1.00
B ²	-1.55	1	6.23	-15.43	12.34	1.0000

Fit Statistics

Std. Dev.	8.53	R²	0.1549
Mean	89.56	Adjusted R²	-0.2677
C.V. %	9.53	Predicted R²	-0.9114
		Adeq Precision	2.1277

A negative **Predicted R²** implies that the overall mean may be a better predictor of your response than the current model. In some cases, a higher order model may also predict better.

Coefficients  Coded Equation  Actual Equation

Coefficients in Terms of Coded Factors

Factor	Coefficient Estimate	df	Standard Error	95% CI Low	95% CI High	VIF
Intercept	87.12	1	4.40	77.31	96.92	
A-HRT	-0.9601	1	2.86	-7.34	5.42	1.00
B-Current density	2.79	1	2.89	-3.64	9.22	1.02
AB	-0.2753	1	3.83	-8.82	8.27	1.02
A ²	4.29	1	4.84	-6.50	15.09	1.00
B ²	-0.1406	1	4.80	-10.84	10.56	1.0000

ANOVA for Quadratic model

Response 4: E. coli removal

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	11.68	5	2.34	0.5827	0.7135	not significant
A-HRT	3.35	1	3.35	0.8369	0.3818	
B-Current density	1.75	1	1.75	0.4376	0.5233	
AB	0.0032	1	0.0032	0.0008	0.9782	
A ²	0.1539	1	0.1539	0.0384	0.8485	
B ²	6.25	1	6.25	1.56	0.2402	
Residual	40.07	10	4.01			
Cor Total	51.75	15				

Factor coding is **Coded**.
Sum of squares is **Type III - Partial**

The **Model F-value** of 0.58 implies the model is not significant relative to the noise. There is a 71.35% chance that an F-value this large could occur due to noise.

P-values less than 0.0500 indicate model terms are significant. In this case there are no significant model terms. Values greater than 0.1000 indicate the model terms are not significant. If there are many insignificant model terms (not counting those required to support hierarchy), model reduction may improve your model.

Figure 54: E. coli ANOVA

ANOVA for Quadratic model

Response 5: EC removal

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	464.02	5	92.80	3.45	0.0452	significant
A-HRT	308.14	1	308.14	11.46	0.0069	
B-Current density	0.6532	1	0.6532	0.0243	0.8793	
AB	101.00	1	101.00	3.76	0.0814	
A ²	49.60	1	49.60	1.84	0.2043	
B ²	20.25	1	20.25	0.7528	0.4059	
Residual	268.98	10	26.90			
Cor Total	733.00	15				

Factor coding is **Coded**.
Sum of squares is **Type III - Partial**

The **Model F-value** of 3.45 implies the model is significant. There is only a 4.52% chance that an F-value this large could occur due to noise.

P-values less than 0.0500 indicate model terms are significant. In this case A is a significant model term. Values greater than 0.1000 indicate the model terms are not significant. If there are many insignificant model terms (not counting those required to support hierarchy), model reduction may improve your model.

Figure 53: EC ANOVA

Fit Statistics

Std. Dev.	2.00	R²	0.2256
Mean	99.38	Adjusted R²	-0.1616
C.V. %	2.01	Predicted R²	-1.2701
		Adeq Precision	2.5229

A negative **Predicted R²** implies that the overall mean may be a better predictor of your response than the current model. In some cases, a higher order model may also predict better.

Coefficients Coded Equation Actual Equation

Coefficients in Terms of Coded Factors

Factor	Coefficient Estimate	df	Standard Error	95% CI Low	95% CI High	VIF
Intercept	100.09	1	1.03	97.79	102.39	
A-HRT	0.6145	1	0.6717	-0.8822	2.11	1.00
B-Current density	-0.4476	1	0.6767	-1.96	1.06	1.02
AB	0.0252	1	0.8992	-1.98	2.03	1.02
A ²	0.2227	1	1.14	-2.31	2.75	1.00
B ²	-1.41	1	1.13	-3.92	1.10	1.0000

Fit Statistics

Std. Dev.	5.19	R²	0.6330
Mean	37.25	Adjusted R²	0.4496
C.V. %	13.92	Predicted R²	-0.1469
		Adeq Precision	6.5521

A negative **Predicted R²** implies that the overall mean may be a better predictor of your response than the current model. In some cases, a higher order model may also predict better.

Coefficients Coded Equation Actual Equation

Coefficients in Terms of Coded Factors

Factor	Coefficient Estimate	df	Standard Error	95% CI Low	95% CI High	VIF
Intercept	41.47	1	2.67	35.51	47.43	
A-HRT	5.89	1	1.74	2.01	9.77	1.00
B-Current density	0.2732	1	1.75	-3.63	4.18	1.02
AB	4.51	1	2.33	-0.6763	9.70	1.02
A ²	-4.00	1	2.94	-10.56	2.56	1.00
B ²	-2.53	1	2.92	-9.03	3.97	1.0000

ANOVA for Quadratic model

Response 2: Turbidity removal

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	26.86	5	5.37	0.4947	0.7737	not significant
A-HRT	0.0380	1	0.0380	0.0035	0.9540	
B-Current density	13.56	1	13.56	1.25	0.2899	
AB	2.58	1	2.58	0.2372	0.6367	
A ²	7.41	1	7.41	0.6820	0.4281	
B ²	1.56	1	1.56	0.1439	0.7124	
Residual	108.58	10	10.86			
Cor Total	135.44	15				

Factor coding is **Coded**.

Sum of squares is **Type III - Partial**

The **Model F-value** of 0.49 implies the model is not significant relative to the noise. There is a 77.37% chance that an F-value this large could occur due to noise.

P-values less than 0.0500 indicate model terms are significant. In this case there are no significant model terms. Values greater than 0.1000 indicate the model terms are not significant. If there are many insignificant model terms (not counting those required to support hierarchy), model reduction may improve your model.

Fit Statistics

Std. Dev.	3.30	R²	0.1983
Mean	96.31	Adjusted R²	-0.2026
C.V. %	3.42	Predicted R²	-0.8585
		Adeq Precision	2.3601

A negative **Predicted R²** implies that the overall mean may be a better predictor of your response than the current model. In some cases, a higher order model may also predict better.

 Coefficients  Coded Equation  Actual Equation

Coefficients in Terms of Coded Factors

Factor	Coefficient Estimate	df	Standard Error	95% CI Low	95% CI High	VIF
Intercept	95.82	1	1.70	92.04	99.61	
A-HRT	-0.0654	1	1.11	-2.53	2.40	1.00
B-Current density	1.24	1	1.11	-1.24	3.73	1.02
AB	-0.7208	1	1.48	-4.02	2.58	1.02
A ²	1.54	1	1.87	-2.62	5.71	1.00
B ²	-0.7031	1	1.85	-4.83	3.43	1.0000

Figure 55: Turbidity ANOVA