WATER SAVING AND REUSE STRATEGIES FOR A LOCAL SOUTH AFRICAN FRESH-PRODUCE BULK MARKET

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PREFACE

This research was conducted with the collaborative assistance of Durban University of Technology, particularly the Green Engineering Research Group under the Department of Chemical Engineering, REFFECT AFRICA, and Clairwood Fresh Produce Bulk Market. The project was overseen by Dr. Emmanuel Tetteh and Prof. Sudesh Rathilal, who had affiliations with the respective institutions. All samples were collected in the bulk market, whereas all experimental runs were conducted at the Durban University of Technology. This project was completed within 34 months, spanning from February 2022 to July 2025.

13 February 2026

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To God be the glory.

DEDICATION

This thesis is lovingly dedicated to my family, whose unwavering support and encouragement have been the foundation of my journey.

To my beloved grandmother, (Maduma, Lwandle Aluwelwa, Luwelwa Zinkonjana Kuphela), whose wisdom, prayers, and unconditional love have been the source of my immense strength and inspiration. Your enduring belief in me and your shared knowledge in faith and devotion to God have deeply shaped me to be the person I am today.

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ABSTRACT

This research addresses water conservation and wastewater reuse strategies at the Clairwood fresh produce bulk market in Durban, South Africa. The research aims to optimise wastewater treatment processes through a comparative evaluation focusing on three technologies: electrocoagulation (EC), dissolved air flotation (DAF), and slow sand filtration (SSF). Effluents from three sources – Trader's Hall, Distribution Centre, and Final Effluent were characterised based on physical, chemical, and biological parameters, including chemical oxygen demand (COD), turbidity, suspended solids, and microbial content. Results revealed high pollution levels, with COD concentrations ranging from 300 to 1200 mg/L, turbidity reaching 150 NTU, and suspended solids up to 500 mg/L.

To address these challenges, wastewater samples were collected from the Trader's Hall, Distribution centre, and Final Effluent and were analysed for key physical, chemical, and biological parameters. The samples were treated using EC, DAF, and SSF under controlled laboratory conditions. Optimisation of the processes was performed using Response Surface Methodology (RSM), considering operational parameters such as induced voltage, agitation speed, and retention time on treatment performance. The effectiveness of each technology was assessed based on pollutant removal efficiencies, including COD, turbidity, suspended solids, and microbial reduction. EC emerged as the most effective treatment, achieving 95% COD removal, 98% turbidity reduction, and significant decreases in conductivity. In comparison, DAF achieved 85% COD removal, while SSF demonstrated limited effectiveness, achieving only 60% COD removal. The optimised EC process showed scalability potential and was validated against the South African National Water Act and South African National Standards for applications such as cleaning the skip area, ablution systems, and process water.

The study concluded that up to 27% of the market's municipal water demand could be offset by reusing treated wastewater, based on a comparison between the volume of treated effluent meeting reuse standards and market's recorded municipal water consumption. This research underscores the viability of EC as a cost-effective and environmentally sustainable technology, particularly for regions grappling with water shortages. Although a detailed economic analysis was not conducted, the high treatment efficiency and relatively low operational requirements suggest potential cost benefits. Beyond its technical contributions, the study highlights broader implications for optimising wastewater treatment and reuse processes in the fruit and vegetable process industry (FVPI). These findings align with global sustainability goals (in particular, #6 Clean water and sanitation), offering a replicable model for integrating wastewater reuse into industrial systems to mitigate water scarcity and promote environmental stewardship.

AUTHOR'S PUBLICATIONS

The candidate (N Mjoli), under supervision, was responsible for conceptualisation, experimental designs and execution, data collection, interpretation and analysis of results, and preparation of manuscripts and presentations of the outputs listed below.

I. **Mjoli, N.S**., Mkhize, N., Chollom, M., Tetteh, E., Rathilal, S. Electrocoagulation treatment of a local South African bulk market wastewater: Effect of operating conditions. (under preparation for submission August 2025)

II. **Mjoli, N.S**., Ngema L., Jairajh S., Madondo N.I. Tetteh, E, Rathilal, S. Optimization of electrocoagulation on municipal wastewater using response surface methodology (under preparation for submission August 2025)

L**ist of conference presentations and proceedings (Main Presenter):**

I. **Mjoli, N.S**., Ngema L., Madondo N.I. Tetteh, E, Rathilal, S. Treatment of wastewater in the fruit and vegetable process industries (FVPI) through electrocoagulation process- Analysis using RSM, Proceeding of the 39th Johannesburg International Conference on "Chemical, Biological and Environmental Engineering" (JCBEE-23) Nov.16-17, 2023 Birchwood Hotel and OR Tambo Conference Centre, Johannesburg, International Institute of Chemical, Biological & Environmental Engineering.

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NOMENCLATURE

COD	Chemical Oxygen Demand
m^3/day	Cubic meters per day
DAF	Dissolved air floatation
EC	Electrocoagulation
FOG	Fats, oil and grease
HRT	Hydraulic Retention Time
kL	Kilolitre
µS/cm	Micro-Siemens per centimetre
mg/L	Milligrams per litre
NTU	Nephelometric Turbidity Unit
pH	Potential of Hydrogen
REFFECT AFRICA	Renewable Energy, Food and Water for Africa
RSM	Response Surface Methodology
SDG	Sustainable Development Goals
SSF	Slow Sand Filtration
TSS	Total Suspended Solids
WHO	World Health Organisation

CHAPTER 1 INTRODUCTION

1.1 Background

Human activities, such as industrial operations, domestic usage, and agricultural practices, have exacerbated the shortage of water (Zahoor and Mushtaq 2023). To maintain the long-term viability of this resource for a rapidly growing global population, it is imperative to investigate certain treatment techniques that may be used for the retrieval, reclamation, and reutilization of water (Mishra et al. 2021; Saravanan et al. 2021).

The investigation of efficacious methodologies for wastewater treatment has assumed an unmatched level of importance, considering that an estimated 80% of the world's total wastewater output is released into natural aquatic ecosystems and waterways (Kookana et al. 2020; Tarpeh and Chen 2021). This has the potential to do significant damage, not only to the environment but also to the welfare of many persons, especially those living in underdeveloped countries (Nie et al. 2020; Ahmad et al. 2022).

South Africa suffers from water shortages as a result of irregular rainfall patterns, an unpredictable climate, and the production of high-risk effluent by industry, ranking it as the 30th driest country in the world (Du Plessis 2023). Water safety has become a vital element in promoting public health and mitigating poverty in South Africa (Hove et al. 2021). The nation gets its water from several sources, mainly surface water (68%), groundwater (13%), return flows (13%), and other sources (6%), accumulating more than 15 billion cubic metres annually (Jain et al. 2020). Moreover, the agricultural sector constitutes around 61% of the total yearly consumption, while the municipality itself consumes 27%. The remaining share consists of other industrial and residential activities, as illustrated in Figure 1-1 (Loki, Aliber and Sikwela 2021). This distribution emphasises the need for efficient support and resource allocation in the agriculture sector to foster sustainable practices.

South Africa has to deploy modern treatment technologies and acquire new water resources to close the gap between water supply and demand by 2030 (Ijoma et al. 2022; Issaoui et al. 2022). In addition, the magnitude and quality of effluent can potentially influence the conceptualisation of a disposal process (Ijoma et al. 2022). Therefore, significant importance is placed on the physicochemical and biological characteristics of water and effluent management systems. The aforementioned variables depend on the population served, which includes industrial wastewater deposited into municipal sewers, as well as the effluent treatment performed before its disposal (Tchobanoglous, Burton and Stensel 2003; Salgot and Folch 2018). South Africa is undertaking projects to examine the practicality of recycling and reutilizing wastewater in the agro-food industry through the implementation of a comprehensive strategy that incorporates water footprint assessment and mass pinch analysis (Mabhaudhi *et al.* 2018; Bag, Gupta and Kumar 2021).

The Fruit and Vegetable Process Industry (FVPI) relies heavily on water as an essential resource, primarily using municipal drinking water as its main supply (Swartz et al. 2021). The wastewater that is produced as a result is commonly treated using traditional techniques (Tchobanoglous, Burton and Stensel 2003; Swartz et al. 2021). Although the industry does not frequently employ advanced wastewater treatment (WWT), the majority of them do perform basic treatment (Sun et al. 2019; Swartz et al. 2021). Approximately 16% of the facilities examined in the FVPI offer tertiary or advanced care (Swartz et al. 2021). The FVPI use many techniques for treating wastewater, including aerobic ponds, biological aeration, trickling filters, and anaerobic ponds (Tchobanoglous, Burton and Stensel 2003; Swartz *et al.* 2021). The constraints of these methods include the production of sludge, potential release of odour, land utilisation, potential groundwater pollution, and operational effects resulting from environmental variables (Swartz et al. 2021; Zahoor and Mushtaq 2023).

Recovering and recycling wastewater, selecting a suitable treatment approach, and identifying the proper usage of the reclaimed water are all crucial elements of water conservation. The project evaluated various treatment methods at a small-scale laboratory level to ascertain the most effective approach for eliminating pollutants. The selection was determined by criteria such as the efficacy, simplicity of execution, and user-friendliness of the technology. The techniques examined in this study include slow sand filtration (SSF), electrocoagulation (EC), and dissolved air flotation (DAF) for the treatment of wastewater originating from the bulk market.

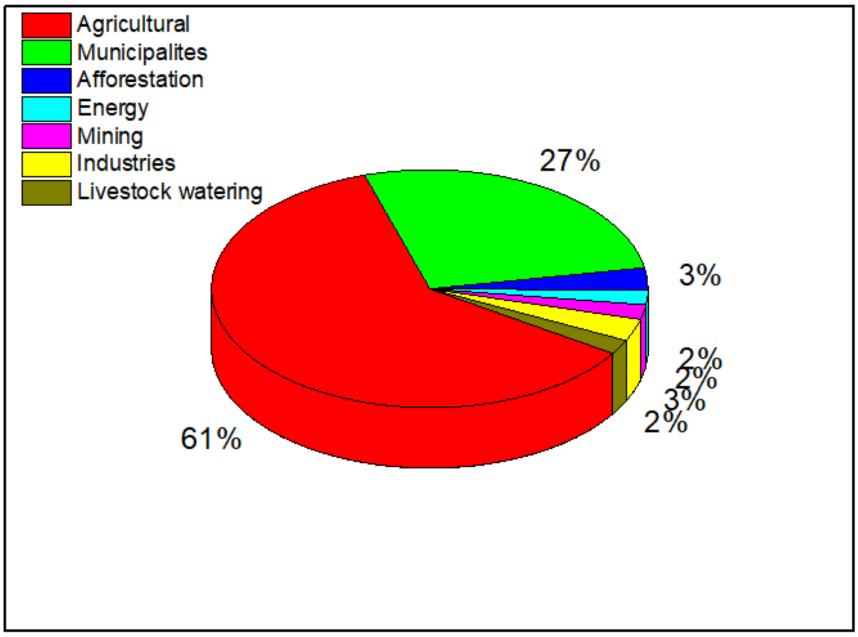


Figure 1-1 Water consumption in South Africa, adapted from (Loki, Aliber and Sikwela 2021)

1.2 Importance of Wastewater Management

Protection of natural ecosystems, public health, and the establishment of sustainable livelihoods are essential. Therefore, the efficient management of effluent is essential in South Africa due to the country's limited water supplies, pollution and inadequate treatment and distribution of water. The National Water Act No. 36 of 1998 establishes the framework for the sustainable management and protection of water resources, delineating the minimum standards for human and aquatic environmental water usage. The Act governs water utilisation and its disposal, emphasising the treatment of wastewater as essential for mitigating environmental damage. The Wastewater Management Act and the South African National Standards (SANS) stipulate requirements for wastewater treatment procedures, as well as discharge restrictions and the reuse of treated water.

The South African Department of Water and Sanitation has a certification and incentive-based program designed to assess and improve wastewater management practices called the Green Drop. The Green Drop program evaluates wastewater treatment plants against a range of standards and criteria, including compliance with various SANS regulations such as SANS 241: Drinking Water Quality; SANS 10400-Part Q: Non-Water-Borne Sanitary Disposal and SANS 11186: Symbolic Safety Signs. These regulations pertain to the water quality treatment process, and environmental impact.

Additionally, the World Health Organisation (WHO) sets global standards and guidelines regarding wastewater, influencing policy in South Africa, particularly in the prevention of waterborne diseases and the provision of safe water to communities (Omohwovo 2024). Adherence to its criteria protects public health and enhances the quality of life for South African citizens.

Table 1-1 presents the prescribed limit values for wastewater discharge into South African water resources, as outlined in the National Water Act. These limits are critical for protecting aquatic ecosystems, maintaining water quality, and ensuring sustainable water use. Each parameter reflects specific thresholds to prevent pollution and environmental harm. Adherence to these limits is legally required to safeguard both natural resources and public health. General limits apply to standard discharges into less sensitive water bodies, while special limits are stricter and apply to sensitive environments to offer greater protection.

Table 1-1 Limit values for wastewater that apply to discharge into a water resource, as per the South African Water Act (Gildenhuys 1998)

Parameter	General Limit	Special Limit
Faecal Coliforms (per 100 mL)	1000	0
Chemical Oxygen Demand (mg/L)	75	30
pH	5.5-9.5	5.5-7.5

Ammonia	6	2
Nitrate/Nitrite (mg/L)	15	1.5
Chlorine as Free Chlorine (mg/L)	0.25	0
Suspended Solids (mg/L)	25	10
Electrical Conductivity (µS/m)	70 µS/m above intake to a max of 150 µS/m	50 µS/m above intake to a max of 100 µS/m
Orthophosphate (mg/L)	10	1

Wastewater generated from the processing of fruits and vegetables in South Africa contains high organic and nutrient concentrations that, if not properly handled, may negatively affect the ecosystem (Mundi, Zytner and Warriner 2017; Swartz *et al.* 2021). This wastewater comprises solid materials, organic chemicals, carbohydrates, and acidic by-products, among others, resulting from cleaning and treatment processes that require specialised treatment before discharging into receiving water bodies (Swartz *et al.* 2021; Trajer, Winiczenko and Dróżdż 2021). The management of this industrial wastewater is governed by South African legislation, which aligns with WHO standards, thereby reducing the risks associated with high pollutant loads and enhancing sustainability to prevent environmental contamination and facilitate water recycling, especially in areas with limited water availability (Mundi *et al.* 2018). In industries of this nature, the efficient execution of wastewater treatment processes can result in the production of biogas from recovered resources, thereby enabling the industry to reduce its environmental impact.

The conventional treatment of wastewater is the process of cleaning municipal and industrial wastewater to satisfy regulatory standards, ensuring its safe discharge into the environment (Tchobanoglous, Burton and Stensel 2003; Gerba and Pepper 2019). As illustrated in Figure 1-2, this treatment predominantly takes place at wastewater treatment plants (WWTPS), which comprise of three principal stages (primary, secondary, and tertiary), and supplementary stages (pre-treatment and sludge treatment and disposal) (Tchobanoglous, Burton and Stensel 2003):

1. Pre-treatment: The elimination of coarse particles and debris via screening and grit removal prevents potential damage to subsequent treatment units.

2. Primary Treatment: The settling tanks facilitate the deposition of suspended solids, thereby reducing the organic load in the effluent. The purified effluent advance to future processes, while the settled sludge is extracted.

3. Secondary Treatment: Biodegradable organic waste undergoes decomposition through biological processes, which may involve activated sludge or biofilm reactors. The microbes degrade organic materials, hence reducing biochemical oxygen demand and suspended solids levels.

4. Tertiary Treatment: Sophisticated methods encompass filtering, disinfection via chlorination or ultraviolet treatment, and nutrient extraction, which refine the effluent to comply with discharge or reuse criteria. The tertiary stage is crucial, particularly in enterprises that release effluents into fragile aquatic ecosystems.

5. Sludge treatment and disposal: the sludge produced from both primary and secondary treatment must be processed to decrease volume, stabilise organic matter, and mitigate environmental impact during disposal or reuse.

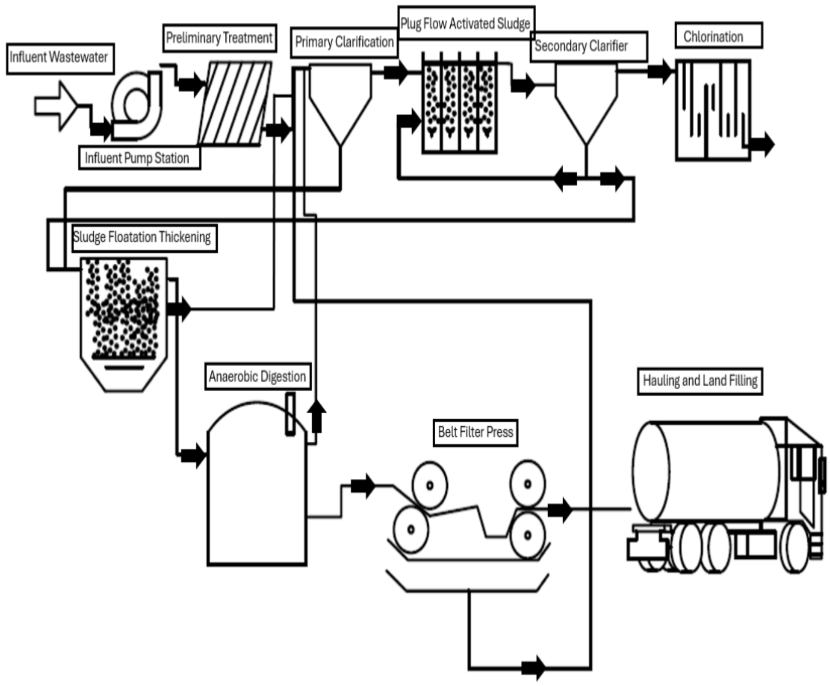


Figure 1-2 Layout of a conventional wastewater treatment plant.

This study concentrated on the primary phase of treatment, employing electrocoagulation (EC), dissolved air flotation (DAF), and slow sand filtration (SSF) for the treatment of wastewater generated by FVPI. In such a sector, these technologies have the potential to reduce pollutants, mitigate their adverse environmental impacts, and enhance water recycling.

1.3 Fruit and Vegetable Process Industry (FVPI)

The fruit and vegetable processing industries (FVPI) can be categorised into two segments: fresh pack and processing (Swartz et al. 2021). The farmer harvests the produce and then packages it into cartons or large containers for transportation to a processing facility. Produce is subjected to cooling techniques to preserve its integrity and undergo fumigation or treatment to manage insect infestation or the growth of microbiological diseases (Kookana *et al.* 2020; Bag, Gupta and Kumar 2021). The processing section, often known as packers, encompasses all the individual procedures that enhance the durability of processed food and enhance its value by modifying the produce to meet specific market demands. The fresh pack sector of the industry combines unit operations with the processing sector. The processes encompass sorting, washing, grading, and packaging lines. However, following the packaging process, supplementary unit operations may contribute to the waste generation process, specifically inside the processing segment (Kookana et al. 2020; Swartz et al. 2021).

High water usage, industrial wastewater production, problematic solid output, and high energy consumption for heating and cooling operations are some of the major environmental challenges specific to the food industry's processing of fruits and vegetables (IPCC 2006). The FVPI has been recognised as one of the food processing industry sub-industries in Australia that consumes the most water annually (Australian Government Department of Agriculture 2007). Water intensity has been previously established as a defining characteristic of the food industry (Australian Government Department of Agriculture 2007). According to the Integrated Pollution Prevention and Control (IPCC 2006) Directive of the European Union, water usage is a significant contributor to the adverse environmental consequences associated with the food sector. The food and beverage industry generates biodegradable effluents for the most part; however, certain sectors utilise salt or brine, which have demonstrated resistance to conventional treatment methods (IPCC 2006).

In South Africa (SA), the Water Research Commission (WRC) and the Department of Water and Sanitation initiated a series of nationwide surveys in the 1980s to examine the water and effluent management practices of various industries. The NATSURV reports, often known as such, have been extensively utilised for the previous thirty years. Since the 1980s, the South African economy and its industrial sectors have seen significant growth or contraction, resulting in a transformed economic environment.

The 1987 publication NATSURV 14: Water and Wastewater Management in the fruit and Vegetable Processing Industry has been updated and revised in November 2021. According to the survey, the average volumes of the effluent streams released from canning operations, juicing processes, freezing processes, and the industry were 298 m^3/d, 274 m^3/d, and 595 m^3/d, respectively, in the FVPI sector. The Clairwood bulk market generates 421.9 m^3/d of effluent. The industry's energy consumption ranged from approximately 2780 to 14,000 kWh/d. These values represent aggregate operational energy demand and are not normalised to water usage or quantities of produce processed, due to variability in processing activities and reporting limitations noted in the survey. Scheduled power outages have a significant impact on the Agri-processing sector, often necessitating extended operating hours or overtime to meet production demands (Swartz et al. 2021).

Most previous work focused on laboratory scale experiments or single technologies, with limited investigation into combined applications or their feasibility for water reuse in the South African FVPI context. This study addresses these gaps by evaluating the performance of EC, DAF, and SSF on effluents from Clairwood fresh-produce bulk market, assessing in both individual and sequential application for water reuse. The research aims to optimise treatment conditions using RSM, quantify potential water savings, and provide a locally relevant, practical strategy for enhancing resource sustainability in South African FVPI operations.

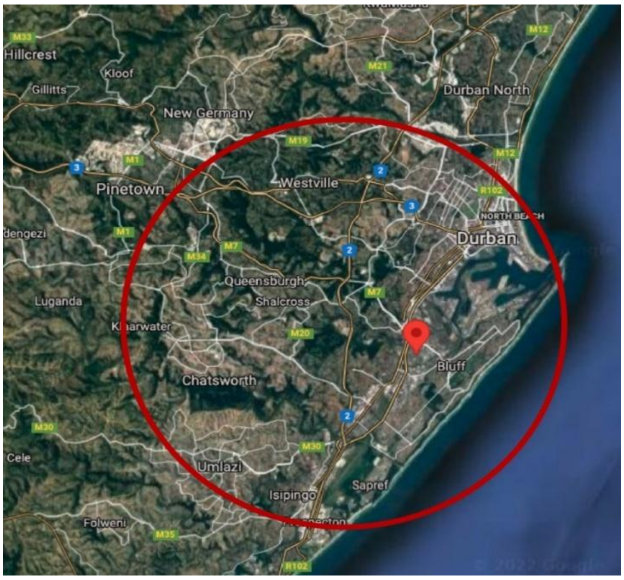


Figure 1-3 Map of Durban showing the Clairwood Market in the eThekwini Municipality, KwaZulu-Natal Province

The Durban Clairwood Market, formally known as the Durban Fresh Produce Market, is located in the suburb of Clairwood within the eThekwini Metropolitan Municipality of KwaZulu Natal Province, South Africa as demonstrated in Figure 1-3. Positioned southwest of Durban's central business district, the market is situated near major transport routes, making it easily accessible for both supplier and buyers. This facility serves as a major hub wholesale distribution of fresh produce in the region and is managed by the eThekwini Municipality's Business support, Tourism and Markets Unit. The market is located -29.911025° latitude and 30.986511° longitude, placing it within a well-connected industrial and commercial zone that supports the city's broader logistics and agricultural supply networks.

1.4 Problem Statement

Water shortage is a significant issue for several nations and the global community. Water consumption has grown at a pace more than double that of population growth over the last century, leading to a rise in chronic water shortages in many areas. By 2025, a significant portion of the global population may face water scarcity, with 1.8 billion individuals living in regions experiencing severe water shortages. South Africa enforces strict restrictions on access to clean water and sanitation to achieve the United Nations Sustainable Development Goals by 2030, despite facing a shortage of freshwater supplies. South Africa is projected to have a 17% water shortage by 2030 if present trends continue.

Achieving Sustainable Development Goal 6, target 6.1, which aims for universal and equitable access to safe and affordable drinking water, requires innovative solutions to optimise water use in all sectors. In urban markets such as Clairwood Bulk Market, high volumes of wastewater are generated from produce handling, cleaning, and washing activities. Currently, this wastewater is largely discharged without treatment or reuse, leading to unnecessary consumption of potable water in increased operational cost.

Wastewater reclamation, recycling, and reuse are essential in water resource management as they enable the production of high-quality source waters to fulfil worldwide agricultural, industrial, and urban needs (Tchobanoglous, Burton and Stensel 2003). Wastewater reclamation, recycling, and reuse represent a practical strategy to reduce potable water demand, lower operational costs, and support sustainable water management. However, certain treatment processes suitable for market wastewater, such as DAF, EC, and SSF remain underutilised. These processes can efficiently remove suspended solids, organic matter, oils, colloidal particles from wastewater, but their application is limited due to factors such as lack of technical knowledge, perceived operational complexity, high capital costs, and insufficient optimisation for variable market wastewater quality

This research is part of the REFFECT AFRICA project, which focuses on developing renewable energy sources for on-grid and off-grid communities, integrating them into existing systems, and improving water-energy-food links. It will generate biochar and improve local farmers' fertiliser and establish reliable water laboratories. The primary objective of the REFFECT AFRICA Project is to investigate the use of waste as a means of generating power to effectively augment the electricity supply to households. In addition, the procedure will generate a bio-fertiliser that may be utilised in agricultural practices and adopt a water conservation strategy within the market.

The specific problem addressed in this study is the evaluation and optimisation of these three treatment technologies to identify the most efficient and cost-effective approach for treating market wastewater. The study applies Response Surface Methodology (RSM) to systematically optimise operational parameters, enabling maximum contaminants removal with minimal energy and chemical input. By providing an evidence based comparison of DAF, EC, and SSF for market wastewater treatment, this research contributes to enhancing water use, reducing reliance on potable water, and supporting the achievement of SDG 6.1 ant the local market level.

1.4.1 Study Area

Water scarcity and rising operational costs are pressing issues for market's that rely heavily on municipal water. Achieving SDG 6, target 6.1, which aims to provide universal and equitable access to safe and affordable drinking water, requires innovative solutions such as wastewater reclamation and reuse to reduce potable water consumption and operational expenses.

The market facility is regulated by the National Water Act of 1998 (Act No. 36 of 1998) and the National Environmental Management: Waste Act of 2008 (Act No. 59 of 2008). These legislative frameworks prevent the facility from producing hazardous waste that might harm human health and the environment. Any waste produced is categorised according to SANS 10234 within 180 days. If there is a change in activity or production, it is categorised within 30 days. However, observations from the field visit and utility data analysis (Table A 1 in Appendix A) revealed that four primary activities contributed to the site's excessive water consumption and pollution, namely:

- Ablution facilities are toilets and showers used by employees and merchants.
- Washing and cleaning of fruits and vegetables before packaging.
- The market skip area involves cleaning skip bins, trash compartments, autos, and flooring.
- Chiller and boiler units' process water from the cooling towers.

The facility receives 165.06 kL/d of municipal water daily, which is divided into three streams: process (45%), Market area (40%), service (10%), and potable water (5%), as shown in Figure 1-4. Water contamination at this plant is caused by residential garbage, general waste, decaying fruits and vegetables, dissolved salts, trace elements, and water vapour from the cooling towers (Chiller and boiler units). Other environmental contamination issues are floods, domestic sewage, industrial water bodies, and nearby air pollution from the Wax industry.

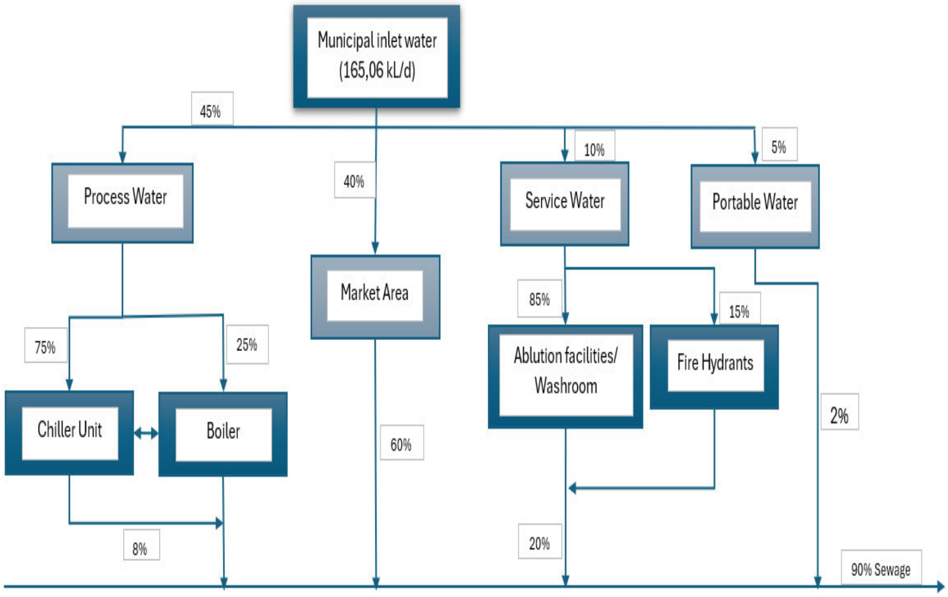


Figure 1-4 Block diagram of the water balance of water consumption and wastewater production.

Further evidence that wastewater abatement for reuse necessarily comes from data (Table A 1) collected between December 5, 2021, and February 4, 2022. The data obtained through an analysis of the plant's water accounts and onsite meter readings taken at various points in the system. This investigation revealed excessive water consumption and elevated costs. The plant was predicted to use 4687.6 kL/d of water over 426 days, with an average working day of 27 days per month, with 90% of that being discharged as wastewater. These findings highlight the scale of water loss and underscore the importance of implementing wastewater abatement and reuse strategies.

1.4.2 Electrocoagulation (EC)

Electrocoagulation (EC) is an electrochemical method that treats contaminated water with electricity as opposed to the use of costly chemical reagents (Islam 2019). Electrolysis is a process in which oxidation and reduction of an electrolytic solution occur as a result of the application of an electric current (Grigoriev et al. 2020). Electrochemical technology is a promising method for eliminating organic contaminants in many types of wastewaters without the need for extra chemicals. Moreover, the superior quality of the electrochemical process hinders the formation of undesired by-products (Grigoriev et al. 2020). It may also be utilised for metal recovery from many types of effluent.

Electrocoagulation has proven to be an effective method for treating soluble or colloidal pollutants found in a wide range of industrial effluents (Islam 2019). These effluents originate from food industries, tanneries, wastewater textile industries, mechanical workshops (soluble oil), polymerization manufacture, and contain heavy metals, suspended solids, emulsified organics, and numerous other contaminants (Butler et al. 2011). Electrocoagulation is more effective in removing very small colloidal particles compared to standard flocculation-coagulation due to the electric field that helps coagulate and destabilise charged particles. Electrocoagulation-flotation can decrease waste generation in wastewater treatment and shorten treatment duration (Bracher et al. 2022).

1.4.3 Dissolved Air Floatation (DAF)

The fundamental principle underlying flotation is the separation of solids from liquids via bubbles. The bubbles are created using a gas that has low solubility in the liquid (Fuerstenau and Urbina 2018). Air is the most often used gas in practice due to its accessibility, safety, and cost-effectiveness. Various flotation procedures are generated by the technique of bubble production, including electrolytic flotation, dispersed air flotation, and dissolved air flotation (Kyzas and Matis 2018).

DAF bubbles are generated by decreasing the pressure of a water flow that is saturated with air. There are three forms of Dissolved Air Flotation (DAF): vacuum flotation, micro-flotation, and pressure flotation (Tetteh 2018). Among them, pressure flotation is the most crucial and commonly employed in water and wastewater treatments. Pressure flotation involves dissolving air in water under high pressure and then releasing it at atmospheric pressure through a needle valve or nozzle to create little air bubbles (Rajapakse et al. 2022).

1.4.4 Slow Sand Filtration (SSF)

Filtration is a physical process whereby solid particles are extracted from a liquid or gaseous solution using a filter medium that permits the fluid to pass through while retaining the solid particles. It may imply the employment of a physical barrier, a chemical, or a biological process (Akers 2017; Wang et al. 2020). The essential components are a filter medium, a fluid containing suspended particles, a driving force to induce fluid flow, and a filter structure that houses the filter medium, retains the fluid, and allows the application of force (Akers 2017; Nath 2017).

Different systems employ different filters for wastewater purification. There are several compelling justifications for this. The quality of the incoming water might impact the specific filtering system used (Drechsel and Keraita 2019). Furthermore, the necessary level of purity for the water to be reused, following filtration, is a determining factor in the choice of filter type (Yin *et al.* 2019; Hube *et al.* 2020). Particle filtration and membrane filtration are the two fundamental filtration mechanisms employed in waste water treatment systems (Li et al. 2018; Wang et al. 2020).

1.5 Aims and Objectives

Aim:

This research seeks to develop a water-conserving technology for the purifying effluent generated by the bulk market in South Africa, treating it to a standard that is suitable for reuse.

Objectives:

1. To characterize the wastewater from the market to establish their physical, chemical, and biological contaminants.
2. To evaluate and optimise the treatability efficiency of slow sand filter (SSF), electrocoagulation (EC) and dissolved air floatation (DAF) using response surface methodology (RSM) by accessing electrical conductivity, turbidity and COD to identify the most efficient approach for water use.
3. To compare the three technologies and recommend the best treatment technology at its optimised conditions.

1.6 Approach

The process of characterising wastewater involved the collection, preparation, and analysis of wastewater from three separate sources: the distribution centre, trader's hall, and final effluent. Effluent analysis criteria were employed to classify the effluent from the market.

The One-factor-a-time (OFAT) technique was employed to ascertain the threshold of the operating conditions for the three technologies: electrocoagulation (EC), dissolved air flotation (DAF), and filtration. After establishing the threshold, it was employed for optimising purposes.

Response surface methodology (RSM) was employed to optimise the process based on the collected data. The goal was to determine the most favourable operating conditions for the various technologies. The results comprised the COD removal %, turbidity removal %, and conductivity. Standard procedures were utilised to examine the presence of physical, chemical, and biological pollutants like faecal coliform and fungi.

1.7 Structure of Dissertation

This dissertation consists of five chapters as follows:

Chapter one: The first chapter serves as the introduction and provides a concise overview of the wastewater generated in the fruit and vegetable processing industry (FVPI), as well as a problem statement that specifies the intended outcomes and objectives. Additionally, a summary of the various technologies that will be implemented (filtration, electrocoagulation, and dissolved air flotation) is incorporated.

Chapter two provides a literature overview on the present and emerging treatment methods for wastewater treatment in the FVPI. The text discusses a few kinds of technologies, their mechanisms, limitations, and the necessity for improvements regarding FVPI wastewater characteristics. It also highlights the important water quality parameters that are regulated and explains the reasons for the development of these technologies. Furthermore, it also offers information on the design of experiments, which is a tool for optimising processes, commonly utilised.

Chapter three: outlines the research approach, including the materials, methodologies, and details of the equipment and processes utilised for characterising and treating the effluent. The text also outlines the analytical and data simulation approach for optimising the process.

Chapter four: details the experimental results and discussions. The following results are presented and discussed: the effects of voltage, agitation speed, and settling time using electrocoagulation; the effects of pressure, agitation speed, and wastewater level on treatment using dissolved air flotation; and the effects of flow and horizontal retention time on treatment through filtration. It also explores the process of optimising the operational parameters using response surface methodology (RSM).

Chapter five: The conclusions and recommendations for future research are concisely outlined in Chapter 5.

CHAPTER 2 LITERATURE REVIEW

2.1 Introduction

Wastewater management remains a critical challenge in South Africa, particularly in KwaZulu-Natal (KZN), where rapid industrialisation, population growth, and poor infrastructure exacerbate water pollution and resource depletion. The increasing water demand, combined with inconsistent wastewater treatment, has led to significant environmental concerns. Industries, particularly the fruit and vegetable processing industry (FVPI), contribute to wastewater generation, further straining existing treatment facilities. This literature review examines the key challenges associated with wastewater management in South Africa, with a specific focus on KZN and the industrial sector, particularly the FVPI.

Moreover, this chapter also provides a comprehensive review of three significant research areas: electrocoagulation, dissolved air flotation, and slow sand filtration. Electrocoagulation is a versatile method for treating water and wastewater that uses electrochemical processes to eliminate pollutants. DAF is a widely utilised method in industries and municipalities for the separation of suspended solids from water using small air bubbles. Slow sand filtration is a centuries-old technique that eliminates contaminants through physical and biological separation processes. The principles of the technologies, their challenges and operational conditions and techniques of optimising form the foundation of this chapter, explaining their relevance and capabilities to address modern wastewater treatment challenges.

2.2 Industrial wastewater challenges in South Africa

Industries contribute significantly to wastewater pollution due to high concentrations of organic and inorganic pollutants. Industrial wastewater often contains heavy metals, oils, greases, and other toxic compounds, posing a serious health risk to aquatic life and human health. Many industrial facilities lack dedicated wastewater treatment plants, relying on municipal WWTPS. However, municipal plants are often not equipped to handle industrial effluents, leading to poor treatment outcomes (Iloms *et al.* 2020). This is particularly concerning in KZN, where the province's industrial base includes chemical, textile, and food processing industries that produce complex waste streams (Steyn *et al.* 2021).

2.3 Challenges associated with wastewater in KwaZulu-Natal

South Africa is classified as a water-scarce country, with wastewater treatment playing a crucial role in ensuring water availability for reuse (Salgot and Folch 2018). However, untreated or poorly treated wastewater contributes to water pollution, making it difficult to meet the growing demand for clean water. In KZN, major rivers such as uMngeni and uThukela are severely affected by wastewater discharge, leading to eutrophication and loss of biodiversity (Wade, O'Brien and Jewitt 2024).

A significant challenge in wastewater management is the aging infrastructure of wastewater treatment plants (WWTPs). Many facilities in KZN were designed decades ago and have not been upgraded to meet the increasing demand. 56% of wastewater treatment plants in the province are either non-compliant or operating below optimal standards due to mechanical failures, lack of skilled personnel, and inadequate funding (Rajasakran 2019).

While South Africa has some robust policies such as the National Water Act (Act 36 of 1998) and National Environmental Management Act (NEMA, Act 107 of 1998), enforcement remains weak. Municipalities, particularly in KZN, struggle to regulate wastewater discharge from industries, leading to high pollutant loads in water bodies. Lack of compliance monitoring exacerbates the problem, allowing industries to discharge untreated or partially treated wastewater (Report 2021).

2.4 Challenges in Managing Wastewater from The Fruit and Vegetable Process Industry

The fruit and vegetable processing industry (FVPI) is highly water-intensive, using large quantities of water for washing, peeling, blanching, and packaging. South African FVPI plants generate approximately 250 million litres of wastewater daily, with KZN contributing a significant portion due to its agricultural processing hubs (Volschenk 2020; Liu *et al.* 2022)

Fruit and vegetable processing wastewater is characterized in high concentration of organic matter, suspended solids, and nutrients. High levels of chemical oxygen demand (COD) and biological oxygen demand (BOD) pose a significant treatment challenge. Furthermore, the seasonal nature of production leads to fluctuating wastewater volumes, making treatment consistency difficult (Volschenk 2020).

Conventional wastewater treatment methods such as activated sludge and trickling filters are often ineffective in handling the specific composition of fruit and vegetable processing wastewater. electrocoagulation (EC), dissolved air flotation (DAF) and slow sand filtration (SSF) have been explored as alternative methods, but high cost and operational complexities hinder their widespread adoption (Al-Juhaimi *et al.* 2018; Lehto 2019)

2.5 Electrocoagulation (EC)

Electrocoagulation (EC) is a comprehensive treatment technology that eliminates various contaminants from water, including total suspended solids (TSS), turbidity, metals, lubricants, and microbes. This is achieved through the application of direct current to sacrificial electrodes that are submerged in an aqueous solution.

2.5.1 Colloidal Particle Stability

Colloid particles can be found in a wide variety of systems, both naturally occurring and synthetic. The existence of repulsive electrical charges on the surface of colloidal particles is frequently used to explain their stability, and stability may be assessed by considering the forces of contact between the particles (Matusiak and Grządka 2017; Boinpally *et al.* 2023). The system will continue to be scattered if repulsive forces are in control. While the contact forces are under control, the particles will clump together, and the solutions may become less stable. It is necessary to minimise this repulsion if destabilisation is to be achieved, as particles with the same charge repel each other (Heurtault et al. 2003; Matusiak and Grządka 2017). Colloids are classified as microscopic particles, with an extremely low mass-to-surface area ratio, and typically range in size from 1 nm to 2 µm (Matusiak and Grządka 2017; Moussa et al. 2017). Due to the significant difference in surface area relative to their mass and size, the gravitational forces of colloids are generally disregarded in favour of investigating the surface phenomena that dominate colloidal suspensions (Moussa et al. 2017). To achieve charge neutralisation, counter charged particles are employed to be drawn towards the surface of the colloids, resulting in the formation of an electric double layer, as seen in Figure 2-1.

An exterior layer, known as the "Ion Diffuse Layer" or "Slip Plane," allows ions to move freely owing to diffusion, while an interior area, known as the "Stern Layer," contains oppositely charged ions that are firmly bonded to the surface of colloidal particles. The outside boundary of the stern layer is defined by the shear surface, which is the contact between the inner and outer layers (Moussa et al. 2017).

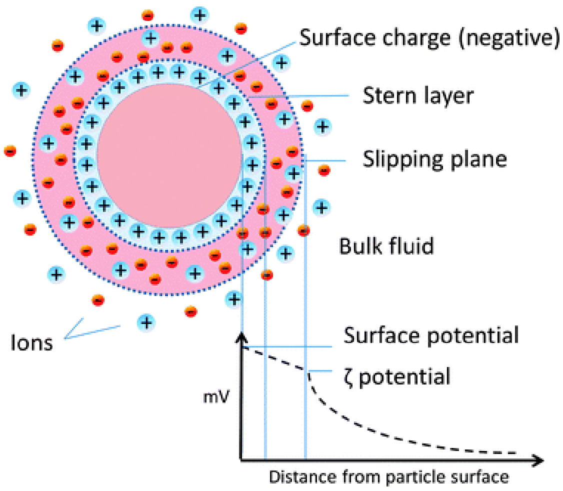


Figure 2-1 Zeta potential and electrical double layer schematic illustration (Herrada García et al. 2014).

The Nernst potential, also known as the maximum potential, is observed at the surface of a colloidal particle. As we go over the stern layer, the potential lowers because to the existence of particles with opposing charges. This decrease in potential is referred to as the Zeta potential, which is recorded at the surface of shear, see Figure 2-2 (Matusiak and Grządka 2017; Moussa et al. 2017). The primary factor determining the stability of a colloidal system is the zeta potential, which quantifies the electrical charge disparity between the first and second layers. It also provides insight into the level of repulsion between colloidal particles that possess the same charge. A larger Zeta potential value corresponds to a stronger repulsion force between particles, resulting in increased stability of the colloidal system (Heurtault et al. 2003; Matusiak and Grządka 2017). Typically, solutions containing colloidal particles with zeta potentials greater than +30 mV or less than -30 mV are regarded to be stable (Moussa et al. 2017).

The DLVO (Derjaguin-Landua- Verwey-Overbeek) theory states that the total interaction energy (Vt) between two particles may be calculated by adding the Van der Waals attraction energy (VA) and the electrostatic repulsion energy (VR) (Moussa et al. 2017). This relationship is illustrated in Figure 2-2, where the energy is plotted against the distance (X) between the particles. The potential energy of interaction (VA) exhibits an inverse relationship with X, resulting in a fast rise as the oil droplets approach each other. In contrast, the repulsive energy (VR) has an exponential dependence on X and changes at a slower rate. The net interaction curve (VA+ VR) depicted in Figure 2-2 exhibits three different characteristics: a main minimum, a secondary minimum, and a maximum (Matusiak and Grządka 2017; Moussa et al. 2017).

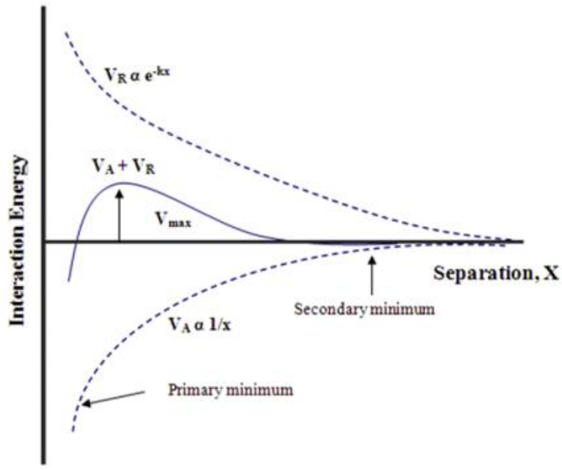


Figure 2-2 Attractive and repulsive energies in relation to particle distance, DLVO theory.(Moussa et al. 2017).

The primary function of coagulants/flocculants in both CC/CF or EC processes is to disrupt the stability of colloids by diminishing the repulsive interactions between particles, hence promoting their agglomeration for more efficient separation. Destabilising colloidal systems via coagulation/flocculation is accomplished through four distinct mechanisms, each of which is determined by a variety of variables, including the chemical and physical properties of the solution, the coagulant/flocculant, and the type of pollutant. The mechanisms involved in this process are commonly referred to as compression of the electrical double layer, adsorption/charge neutralisation, adsorption/inter-particle bridging, and trapping of particles in precipitate (Matusiak and Grządka 2017; Moussa et al. 2017).

2.5.2 Theory of Electrocoagulation

The theory of electrocoagulation, an advanced technology that combines coagulation, flotation, and electrochemistry, should be studied after grasping the stabilisation and destabilisation of colloidal systems. The three technologies that form the foundation of EC have each been the subject of in-depth research. The intricacy of EC technology is due to the use of many pollutant removal methods, including as sweep coagulation, flotation, and adsorption. The process for removing pollutants is governed by the characteristics of the coagulants formed in the solution, as well as the properties of the dissolved ions and contaminants in the wastewater (Barrera-Díaz, Balderas-Hernández and Bilyeu 2018; Ungureanu, Vlăduț and Paraschiv 2020). Despite several researchers documenting the efficacy of electrocoagulation (EC) in addressing various contaminants, they have not been able to provide a theoretical explanation for its mechanism (Moussa et al. 2017). Based on existing literature, EC has been utilised to remediate water contaminated with heavy metals, tannery, textile and coloured wastewater, pulp and paper industry wastewater, viscous wastewater, and food industry contaminants for the past century (Ungureanu, Vlăduț and Paraschiv 2020; Vepsäläinen and Sillanpää 2020).

As illustrated in Figure 2-3, the fundamental electrolytic cell unit generally comprises an electrolytic cell, which includes an anode and cathode metal electrodes that are immersed in the solution requiring treatment and connected externally to a DC power source. The most often utilised metals for EC cells are iron and aluminium electrodes because they are readily accessible, non-toxic, and have a track record of dependability (Moussa *et al.* 2017; Boinpally *et al.* 2023). While EC shares similarities with CC/CF in terms of the destabilisation process, it diverges from CC/CF in terms of other factors, such as the occurrence of synchronous side reactions at both electrodes. In an electrocoagulation (EC) cell, the anode acts as the coagulant by dissociating and releasing metal cations when an electric current flows through the cell (Moussa et al. 2017; Tegladza et al. 2021).

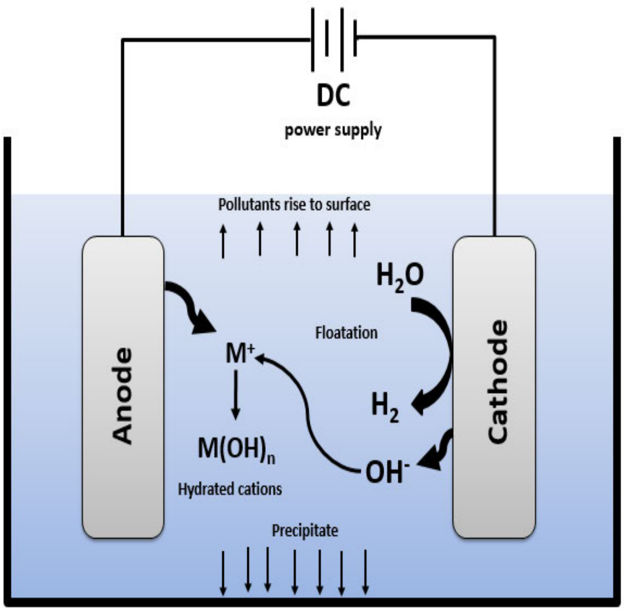


Figure 2-3 Electrocoagulation cell (Moussa et al. 2017).

Electrocoagulation entails the concurrent formation of hydroxyl ions and hydrogen gas at the cathode and the dissolution of metal cations from the reactor anode:

$M \rightarrow M^{+} + ne^{-}$

2-1

$2H_2O(l) \rightarrow 2OH^{-} + H_2(g)$

2-2

The metal (M) is oxidised to its cation (M+) when a current flows through a metal electrode, see Eq. *2-1*. Hydrogen gas and the hydroxyl ion (OH) are produced concurrently upon the reduction of water Eq. *2-2*. By utilising sacrificial anodes (usually iron, stainless steel, or aluminium) that must be replaced periodically (Mousazadeh et al. 2021), electrocoagulation thus introduces metal cations in place. A variety of elimination processes working in concert makes up the overall response mechanism. As the reaction advances, the prevailing mechanism may fluctuate and undoubtedly alter in response to modifications in treatment conditions, operational parameters, and specific pollutant categories (Ungureanu, Vlăduț and Paraschiv 2020; Othmani *et al.* 2022). By forming monomeric and polymeric hydroxo complex species, highly charged cations generated at the anode destabilise colloidal particles during electrocoagulation. These metal hydroxo complexes aggregate pollutants into robust matrices due to their high adsorption capacity (Ungureanu, Vlăduț and Paraschiv 2020). The degree of metal hydrolysis is contingent upon the pH, the concentration of total metal cations, and the nature and quantity of other species that are present in the solution (Lu, Zhang and Li 2021; Othmani *et al.* 2022).

2.5.3 Factors Affecting Electrocoagulation Efficiency

This section discusses the key factors that have a significant impact on the effectiveness of electrocoagulation and its capacity to eliminate contaminants from wastewater:

2.5.3.1 Electrode Arrangement

In the monopolar-parallel (MP-P) arrangement, all the anodes and cathodes are interconnected and coupled to the external DC power source. In this arrangement, the electric current is distributed amongst the electrodes, leading to a reduced potential difference as compared to electrodes linked in series. As an alternative, a monopolar-series (MP-S) connection is established by connecting the anode and cathode, which are the two outermost electrodes, to the external circuit while connecting each pair of inner electrodes without any interconnection to the outer electrodes. In this scenario, the cell voltage is increased, resulting in a greater potential differential (Othmani *et al.* 2022; Boinpally *et al.* 2023). Sacrificial electrodes refer to the interior electrodes, which may be composed of metals that are similar or different. Their function is to reduce the anode's consumption and the cathode's passivation (Moussa et al. 2017; Othmani et al. 2022). Alternative three is the bipolar-series (BP-S) configuration, which involves connecting the external power supply directly to the exterior electrodes while leaving the interior electrodes unconnected. When an electric current flows through the main electrodes, the inner electrodes positioned next to them become polarised and acquire an opposing charge to that of the adjoining electrode. In this arrangement, the two electrodes on the outermost sides are monopolar, whereas the inner sacrificial electrodes are bipolar. (Moussa et al. 2017; Othmani et al. 2022).

The selection of the optimal electrode connection mechanism is governed by the efficacy of pollutant removal and the cost of treatment, and has been thoroughly examined in several research. Demirci, Pekel and Alpbaz (2015) investigated the impact of several electrode connections (MP-P, MP-S, BP-P) on the colour, turbidity reduction, and overall treatment cost of EC in the treatment of textile wastewater. The findings demonstrated that the removal efficiencies were comparable across all three connections. However, it was determined that the MP-P configuration had the highest cost-effectiveness.

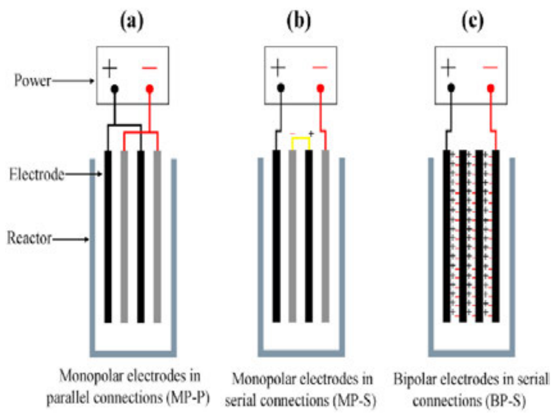


Figure 2-4 Configurations of electrode types (Othmani et al. 2022)

2.5.3.2 Type of Power Supply

In electrocoagulation cells, DC power supplies are commonly employed. Nevertheless, after a certain period of operation, DC power supply usage induces cathode passivation, which is the formation of an oxide layer on the cathode and results in oxidation or consumption of the anode. Passivation induces an elevation in the passive over potential, resulting in heightened power consumption. Furthermore, the formation of the passive layer reduces the current flow between the two electrodes and decreases the efficiency of the electrochemical process (Barrera-Díaz, Balderas-Hernández and Bilyeu 2018; Othmani *et al.* 2022). According to Yang et al. (2015), usage of alternating pulsed current (APC), which inhibits the formation of passive layer when Al or Fe electrodes are employed, or the addition of a significant number of chloride ions, which break down the passive layer, can both solve the passivation problem.

Electrode passivation can be a problem in the DC EC process. Some research suggest that using alternating current (AC) power might help achieve better pollutant removal efficiency, reduce sludge production, and lower energy consumption (Karamati-Niaragh *et al.* 2019; Mousazadeh *et al.* 2021; Boinpally *et al.* 2023). Karamati-Niaragh et al. (2019) conducted a study on the elimination of nitrate using a continuous EC method. They discovered that the average operating cost in the AC mode was more than 40% lower compared to the DC mode. This was mostly attributed to the reduction in Al electrode use. In their study,(Vasudevan and Lakshmi 2011) examined the impact of AC and DC on the elimination of cadmium from water using iron EC. They found that employing an AC power supply resulted in better efficiency in removing cadmium and reduced energy usage compared to a DC power supply. Additionally, some research revealed that the procedure was more cost-effective when employing EC with alternating pulsed current (APC) rather than a DC power source (Asaithambi et al. 2021; Xu et al. 2022). APC can be employed to decrease the passivation or polarisation of electrodes (Xu et al. 2022). With the application of electrochemical characteristics and polarity reversal in an electrical field, pulsed EC technology can cause dipole formation in some nonpolar wastewater particles, allowing insoluble materials to form micro-aggregates (Lu, Zhang and Li 2021).

2.5.3.3 Current Density

The current density, defined as the amount of current flowing through a given area of the electrode, directly influences the quantity of metal ions that are liberated from the electrodes. Dissociation of metal ions is, on average, proportional to the current density that is applied. A drop in current efficiency, which is measured as the ratio of the current required to generate a certain product to the overall current consumption, can also occur when an excessive amount of current is utilised, increasing the likelihood of losing electrical energy in the process of heating the water. The current density is a determining factor in the size of gas bubbles produced, whether from the anode or the cathode. This, in turn, has an impact on the efficiency of the process. Exceeding a certain threshold of current density results in no meaningful improvement in the quality of treated water. The choice of an optimal current density is further influenced by additional factors like pH, temperature, and water flowrate (Lu, Zhang and Li 2021; Othmani *et al.* 2022; Xu *et al.* 2022). A group of researchers conducted a study to examine how the current density affects the operation and efficiency of electrochemical cells under various operating circumstances. They discovered that with higher current densities generally increasing pollutant removal rates but also raising energy consumption and reducing electrode lifespan. (Demirci, Pekel and Alpbaz 2015; Moussa *et al.* 2017).

Conversely, in the case of aluminium and iron anodes, the dissolution reaction is the main reaction, whereas the contribution of other processes is minimal for normal current densities and electrode potentials when the pH is neutral or acidic (Demirci, Pekel and Alpbaz 2015). At alkaline pH, the rate at which iron anodes dissolve may be lower than what is predicted by Faraday's law. This indicates the potential for other reactions to occur at the anode, given the specified circumstances (Moussa *et al.* 2017; Boinpally *et al.* 2023). Therefore, when the current density increases, there is a corresponding increase in the number of dissolved metal ions at the anodes. This leads to the production of more monomeric and polymeric metal ion complexes (Lu, Zhang and Li 2021; Othmani *et al.* 2022). These complexes can efficiently adhere to or co-precipitate with pollutants. However, when an excessively high current is applied, the process of passivation of electrodes also accelerates. This can lead to an increase in the voltage of the electrochemical cell and a decrease in the efficiency of the Faradaic process (FE). The Faradaic efficiency represents the percentage of actual metal production compared to the theoretical maximum and can be calculated using the equation *2-3* as described by Chen et al. (2020). Ideally, the FE value of 1.0 signifies that the full charge flows via the anode for metallic production and transfer to the bulk suspension. The gradual decline in FE signifies a deterioration in the efficacy of the EC treatment. The presence of chloride ions in solutions can result in the occurrence of pitting corrosion, which can cause an increase in metallic loss and result in FE values above 1 (Chen *et al.* 2020; Lu, Zhang and Li 2021). The term used to describe this occurrence is super-faradaic disintegration. The formula for calculating the Faradaic efficiency (FE) is expressed in 2-3.

$$\mathrm{FE} = \frac{\mathrm{ZmF}}{\mathrm{MIt}} \times 100\%$$

2-3

Where: Z represents the number of electron transfers

m is the mass loss of the anodes in grams,

F is the Faraday constant (96485 C/mol),

M is the molar mass of Al or Fe in grams per mole,

I is the applied current in Amperes, and

t is the total time of the electrochemical reaction in seconds.

During an electrochemical process, the current density frequently reaches an optimal value. If the value beyond this threshold, the efficacy of EC therapy does not exhibit a substantial enhancement. Moreover, an excessively high current density results in a significant ohmic drop between the electrodes, leading to an elevated voltage (Lu, Zhang and Li 2021). This might potentially result in the inefficient use of electrical energy for heating water. The choice of the most effective current density is further influenced by additional factors like solution pH, temperature, and water flow (Barrera-Díaz, Balderas-Hernández and Bilyeu 2018; Chen *et al.* 2020; Lu, Zhang and Li 2021). Chen et al. (2020) suggested that for a maintenance-free long-term operation, the current density of the EC process should fall within the range of 20-25 A/m^2, provided that no effective measures are taken.

2.5.3.4 Concentration of Anions

The impact of various anions on the destabilisation characteristics of metal ions varies. By increasing the potential between the electrodes, sulphate ions reduce current efficiency and corrosion/metal dissolution from the electrodes, thereby preventing colloidal destabilisation. However, chloride and nitrate ions counteract the inhibition of sulphate ions by disrupting the passive layer (Matusiak and Grządka 2017; Barrera-Díaz, Balderas-Hernández and Bilyeu 2018).

A significant aspect influencing the effectiveness and power consumption of EC is the solution's conductivity, the greater the conductivity, the lower the EC's power consumption because of the better pollutant removal efficiency. Consequently, the addition of anions in the form of salts like NaCl enhances the conductivity of the solution. It was also discovered that the presence of chlorine ions aids in water disinfection (Matusiak and Grządka 2017).

2.5.3.5 Effects of Initial pH

The pH level is a crucial factor in electrocoagulation since it directly impacts the solution's conductivity, zeta potential, and electrode dissolution. Establishing a definitive link between the pH of the solution and the efficacy of electrocoagulation is challenging due to the fluctuating pH of the treated water during the EC process. As a result, it is common to refer to the initial pH of the solution.

The effluent pH increases when the initial pH value is less than 4 (acidic) and decreases when the original pH value is more than 8 (alkaline). When the initial pH value is in the neutral region (about 6-8), the pH of the effluent varies very minimally. This scenario suggests the presence of a pH buffering effect during electrocoagulation, which distinguishes it from conventional chemical coagulation methods. The pH buffering capacity of electrocoagulation is due to the equilibrium between the generation and utilisation of hydroxyl ions, as well as the requirement for charge neutralisation before the conversion of soluble aluminium compounds into aluminium hydroxides. $Al(OH)_3$ generated close to the anode may cause the pH to drop. The ultimate pH values obtained after electrocoagulation, however, have also been observed to rise quickly during the process, generally being higher than 8–9 (Othmani *et al.* 2022; Boinpally *et al.* 2023). This phenomenon is to be expected, as the cathode continuously generates hydroxyl ions. The primary cause of a reduction in pH in alkaline circumstances is the creation of $Al(OH)_4$ complexes. Furthermore, at neutral pH, hydrogen bubbles formed at the cathode are the smallest and finest, giving an adequate surface area for gas-liquid-solid interfaces and mixing efficiency to promote the aggregation of micro-destabilised particles and colloids (Boinpally *et al.* 2023).

2.5.3.6 Electrode Material

It is crucial to choose the appropriate electrode material because it dictates the chemical reactions that will occur. Aluminium (Al), mild steel, stainless steel and iron (Fe) electrodes are the most commonly used due to their availability and established dependability However, limited research has been conducted on copper and nickel as sacrificial electrodes (Gafoor et al. 2021). Nevertheless, research has shown that Fe (II) is a less effective coagulant than Fe (III) because of its reduced positive charge. A lower positive charge signifies a lower capacity of the ion to compress the electrical double layer or destabilise colloids. The efficacy of Al electrodes in pollutant removal surpasses that of Fe electrodes, according to the majority of studies (Moussa et al. 2017).

The electrode material significantly affects the properties of the sludge produced in the electrocoagulation process. Zodi et al. (2009) utilised the sludge volume index (SVI) to describe the settling properties of electrocoagulation sludge produced by aluminium and iron electrodes. The light and fluffy nature of the sludge generated on aluminium electrodes resulted in high SVI values, which hindered settleability. However, compared to the sludge created by aluminium electrodes, the SVI values found for the sludge formed by iron electrodes were significantly lower. The sludge created by the iron electrodes was heavy enough to create a compact layer, which was the reason for this (Vepsäläinen and Sillanpää 2020; Boinpally *et al.* 2023).

2.5.3.7 Induced Voltage

The induced voltage and applied current density are interconnected variables. In an electrochemical cell, higher voltages cause currents to rise and vice versa; but, under some circumstances, such as when the electrodes are passivated, high overpotentials cause currents to fall. The cell voltage can be influenced by the applied current density via different overpotentials (Vepsäläinen and Sillanpää 2020; Boinpally *et al.* 2023) . This current density governs the rate of coagulant administration and, consequently, the removal of contaminants. Proton reduction suppression can be achieved by using a cathode material with a high overpotential. The overpotentials associated with the oxidation or reduction of the solvent might exhibit significant variations depending on the electrode material.

Induced cell voltage (and, by extension energy utilisation), and the treatment efficacy also has a direct impact on the interelectrode distance In systems with low conductivity, the cell voltage (and hence the energy cost) will be comparatively high, perhaps resulting in the choice of a smaller electrode gap (Mousazadeh et al. 2021).

2.5.3.8 Mixing Speed

The agitation speed is a critical parameter in electrochemical (EC) procedures as it improves the kinetics of mass transfer by increasing the ion mobility in the solution. The rate at which particles collide and the discharge of metal ions and hydroxyl groups both increase in parallel with the mixing velocity. The rate at which metal hydroxides and flocs are formed is a factor in that formation. Using the EC method, an agitation speed of 80–300 rpm is generally regarded as adequate for WWT. In water treatment, agitation velocities ranging from 30 to 60 rpm are utilised; however, exceeding the optimal value of agitation speed results in a decrease in the efficiency of pollutant removal due to the degradation of flocs caused by collision (Bajpai *et al.* 2022; Boinpally *et al.* 2023). An investigation was undertaken to compare the effects of agitation speed on the EC process. It was discovered that almost no separation occurred in the absence of agitation; however, the treatment efficacy of the pollutant increases when the agitation speed is increased from 300 to 500 rpm in WWT (Boinpally *et al.* 2023). To improve the efficacy of COD removal from spent wash effluent from distilleries, an additional EC experiment was performed with the agitation speed varied between 200 and 600 rpm. The findings show that the effectiveness of therapy increases from 200 to 500 rpm and decreases from 500 to 600 rpm (Garcia-Segura *et al.* 2017; Boinpally *et al.* 2023).

The scientists also observed that the augmentation of ion mobility necessitated an increase in mixer speed to eliminate the colour. It was demonstrated that fast mixing will break the bridging between colloids and ultimately stabilise them, in addition to flooding the solution with supporting electrolytes (Bajpai et al. 2022). Increasing the current density leads to a slight increase in the final turbidity of the solution when mixing is present (Garcia-Segura *et al.* 2017; Lu, Zhang and Li 2021). However, in the absence of mixing, the process appears to be unjust. Therefore, this suggests that the method with mixing works better than the one without. The mixing method meets quality standards, but the non-mixing method does not.

2.5.3.9 Electrolysis Time

The efficacy of pollutant elimination is additionally influenced by the duration of electrolysis. As the duration of electrolysis increases, so does the efficacy of pollution removal. Nevertheless, once the optimal duration of electrolysis has been attained, there is no further enhancement in the efficacy of pollutant removal with subsequent increases in electrolysis time. Anode dissolution results in the formation of metal hydroxides (Gafoor *et al.* 2021; Bajpai *et al.* 2022; Boinpally *et al.* 2023). A steady current density and an extended duration of electrolysis result in a subsequent rise in the number of metal hydroxides produced. Pollutant removal effectiveness increases with an increase in floc production with a longer electrolysis time. When the electrolysis time surpasses the ideal electrolysis time, the efficacy of pollutant removal does not improve since there are enough flocs available for pollutant removal (Gafoor et al. 2021; Mousazadeh et al. 2021).

An investigation was conducted into the efficacy of DC (Direct Current), AC (Alternating Current), and APC (Alternating Pulse Current) in removing pollutants. With its reduced operating cost, shorter operation duration, and reduced electrode corrosion when utilising the Fe-Al combination, APC was determined to be the most appropriate option, even if the efficiency stayed the same in all three conditions (Boinpally *et al.* 2023). The impact of varying electrolysis time affected the effectiveness of removing colour and COD while keeping the pH constant at 8 and the current density constant at 15 A/m^2. As the electrolysis time goes up, the CRE value goes up until a certain point, at which point it stays the same. The primary factor contributing to the sustained CRE value was the development of nascent flocs of electro coagulant (Gafoor et al. 2021; Mousazadeh et al. 2021).

Another experiment was conducted using the EC method, with electrolysis periods ranging from 15 to 120 min, to examine the impact of electrolysis time on colour removal efficiency (CRE) from the effluent obtained from the textile industry. The results showed that increasing the EC duration from 15 to 45 minutes resulted in a substantial improvement in colour removal (30–61%), whereas increasing the EC period from 45 to 90 minutes only slightly improved CRE (61–68%). However, when the EC duration was prolonged from 90 to 120 minutes (68–98%), CRE rose significantly (Bajpai et al. 2022). In summary, time depends on the contaminant we want to remove, the desired efficiency, and mostly on process cost for residency, retention, or total operation.

2.5.4 Advantages and Disadvantages of Electrocoagulation

Table 2-1 provides an overview of the key advantages and disadvantages of electrocoagulation as a treatment method for wastewater. Electrocoagulation is a promising technology known for its ability to effectively remove a wide range of contaminants with relatively low chemical usage. However, like any treatment process, it comes with its challenges, including high initial capital costs and maintenance requirements. Table 2-1 highlights the main benefits and limitations, helping to assess the feasibility of electrocoagulation for various wastewater treatment applications.

Table 2-1 Summary of the advantages and disadvantages of electrocoagulation in wastewater treatment (Shahedi *et al.* 2020; Das, Sharma and Purkait 2022).

Aspect	Advantages	Disadvantages
Efficiency	High removal efficiency for contaminants such as heavy metals, dyes, and colloids	Generally poor treatment efficiency with high concentrations of any of the above contaminants in solutions or in waters of high conductivity
Water Quality	Production of high-quality effluent, with this commonly meeting extremely stringent rules relative to discharge	The treatment can result in secondary waste; this may be in the form of sludge that needs to be disposed of.
Footprint	Compact equipment space required is small	May need post treatment or filtration systems that increase the space needed for this technology
Operational Flexibility	Suitable for different types of wastewater to treat, such as municipal, industrial, mining, and textile, among others.	It was found to behave very well under a very controlled selection of current density and electrode material.

Chemical Usage	Reduces or eliminates the need for chemical coagulants; hence, environmentally friendly	Electrodes may dissolve over time and thus may require periodic replacement and monitoring
Settleability of Solids	Effective removal of particles that may not settle well by conventional methods	Sludge generated could be difficult to handle and therefore may require further processing or dewatering
Energy Consumption	Moderate energy consumption; usually lower than that for other electrochemical processes	Energy-intensive due to the pollutant load, solution conductivity, and necessary treatment level.
Maintenance	Low maintenance if well-monitored, with no moving parts in the primary system	Electrodes may be subject of corrosion and require therefore periodic cleaning or replacement
Process Complexity	Easy to operate without involving significant chemical handling	Requiring skill to maintain current density, pH, and other operating variables
Cost	Economical for high-strength or complex wastewaters, which reduces the chemical costs.	Initial capital costs are high; replacement of electrodes adds to operation costs.

2.6 Dissolved Air Flotation (DAF)

In order to achieve effective separation of solid and liquid in influents containing particles and natural colour using Dissolved Air Flotation (DAF), it is necessary to first carry out coagulation or flocculation. This process helps in the formation of bubble-floc aggregates when microbubbles are introduced. The main concept behind DAF is to float particles that have a specific gravity similar to that of water. This is done by using low-density gas bubbles, typically air. The air bubbles stick to the particles, lowering their specific gravity to less than 1.0. This causes the particles to stick together and float to the top of the flotation tank. Bubble generation, bubble–particle attachment, and rotation of the bubble–particle agglomeration are the three primary methods of particle removal in DAF (Palaniandy *et al.* 2017).

2.6.1 Bubble Formation

Bubble creation involves two distinct processes. The nucleation process begins immediately when the pressurised water containing air is discharged via the nozzle. In the second stage, the surplus air present in the water that is already saturated is transported to the flotation tank in the form of gas. During this stage, the bubbles undergo coalescence and grow while experiencing a drop in hydrostatic pressure as they ascend through the flotation tank. Nevertheless, in this stage, the amount of air stays unchanged (Rajapakse et al. 2022).

2.6.2 Bubble–Particle Attachment

Bubble-particle attachment in flotation processes occurs through several mechanisms. As air bubbles rise through the liquid medium, they may collide with suspended solids or flocs, resulting in attachment through direct contact. In addition, bubbles can become entrapped within the floc structure during their upward movement, enhancing the buoyancy of the flocs and promoting separation. Bubble may also be incorporated into the floc matrix during floc formation, where they assimilated as the flocs develop. Together, these mechanisms enhance floc buoyancy and improve the overall efficiency of the flotation process.

The process of bubble-particle attachment involves three sequential mechanisms: adhesion or collision, trapping, and absorption. To achieve effective agglomeration between the particle and the bubble, it is necessary to destabilise the particle. Nevertheless, the particle must satisfy two crucial requirements: having a neutral charge and a hydrophobic surface. Under these circumstances, the bond between the particle and the bubble is highly robust, leading to effective floating. The influent necessitates a coagulation process that involves precise dosing and pH control, leading to a reduction in particle charge and the creation of a hydrophobic particle surface (Dassey and Theegala 2011; Rajapakse et al. 2022).

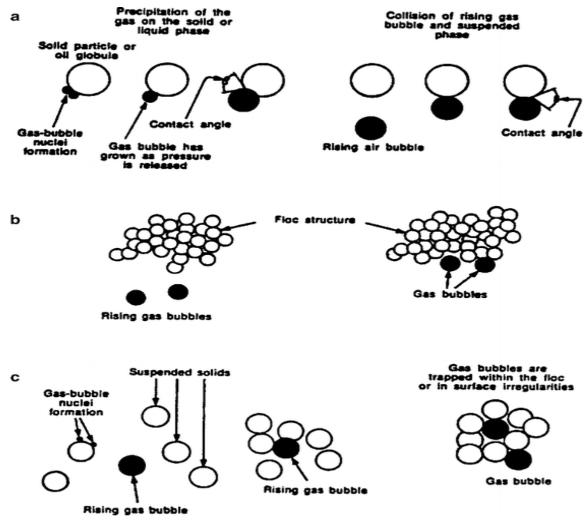


Figure 2-5 Illustration of Bubble-Particle Attachment in Dissolved Air Flotation (DAF) Process (Shammas and Bennett 2010).

This Figure 2-5 demonstrates the mechanism of bubble-particle attachment during the DAF process, where fine air bubbles are introduced to the water, facilitating the aggregation of particles, which then rises to the surface for removal.

2.6.3 Flotation of Bubble–Particle Agglomerate

During the Dissolved Air Flotation (DAF) process, the particles become buoyant because of bubbles that decrease the density of the agglomerates formed by the bubbles and particles. When bubble-particle agglomerates have a density lower than that of water (1.00 g/cm^3), they will ascend and remain buoyant on the surface. Smaller particles necessitate fewer bubbles to reduce density compared to larger particles, which require a greater number of bubbles. The agglomerates formed by the combination of bubbles and particles are expected to ascend to the top of the flotation tank. The agglomerates that fail to reach the surface are removed by the purified water. Stokes' law may be used to determine the increasing speed of the bubble-particle. The three primary theories in DAF demonstrate that certain elements influencing the DAF system must be carefully considered before to implementing it in water or wastewater treatment. Hence, it is important to consider the functioning of the system and all relevant elements while creating and using the DAF system (Zhang, Trompette and Guiraud 2017; Fanaie and Khiadani 2020).

2.6.4 Kinetics of Flotation

Harper (1988) stated that it is rare for tests to support the hypothesis that a bubble rising in a Newtonian liquid may be studied separately, unless contaminants are carefully eliminated. A bubble with a constant surface tension will ascend slowly under the influence of gravity if two conditions are met:

1. Its motion remains stable despite random tiny disturbances.

2. The time it takes for the bubble to reach its terminal velocity is far less than the time it takes for the bubble to undergo significant changes in size. At a low Reynolds number, the retarding or drag force acts in the same direction as the terminal velocity but with an equal and opposite magnitude (Palaniandy *et al.* 2017).

$$D = 6\pi a\mu U, \qquad 2\text{-}4$$

Where: D is the drag force (kN)

a is the radius of bubble (m)

μ is the dynamic viscosity (kg/m/s)

U is the terminal velocity (m/s)

The drag force in the equation 2-4 is used to predict how bubbles will move through the liquid. The bubble size (a), liquid viscosity (μ), and the bubble velocity (u) all influence the bubble's ascent. The equation indicates that as the bubble radius increases, the drag force increases, slowly raising the bubble's upward motion. These relationships are vital in the flotation systems, where control over the bubble's ascent is necessary for effective contaminant removal.

2.6.5 Factors Affecting Dissolved Air Flotation

This section examines the principal factors that significantly influence the efficacy of dissolved air flotation and its ability to remove pollutants from wastewater:

2.6.5.1 Solubility of Air

Flotation typically measures the amount of air utilised by the ratio of air volume to water volume treated. Henry's law is applied when considering saturated water as a solution of air in water that is not very concentrated. It is important to note that Henry's law was initially established by Henry's experiment with N_2, O_2, N_2O, H_2O, CO_2, and water at room temperature (25ºC). The notion that the law may be employed for universal applicability is baseless (Sander 2023). Nevertheless, research conducted on wastewater containing dissolved solids of up to 1000 mg/L and subjected to pressures of up to 500 kPa showed that the Henry's law constant may be employed to determine the quantity of dissolved air. In cases of ideal dilute solutions when the solute follows Henry's law (equation 2-5) but not Raoult's law, and the solvent follows Raoult's law , the application of Henry's law is appropriate (Palaniandy *et al.* 2017).

$PB = xBKB$ 2-5

Where: PB is the vapour pressure,

xB is the mole fraction of the solute,

KB is constant.

This equation applies to situations where the solute concentration is low enough for Henry's law to hold, which in case in the typical wastewater treatment systems. Thus, despite the law's formation at room temperature, it remains a useful tool for understanding and calculating the dissolved air in flotation processes under certain operational conditions, particularly in systems such as Dissolved Air Flotation (Kouhestani *et al.* 2020; Piaggio *et al.* 2024).

2.6.5.2 Bubble Generation

The geometric design and operating circumstances of the injection nozzles had a crucial role in defining the size of the bubbles (Palaniandy *et al.* 2017; Rajapakse *et al.* 2022). According to their findings, the pressure of the saturator does not consistently impact the effectiveness of the nozzle. Conflicting assertions were made on the relationship between increased pressure and the size of bubbles. Some studies suggested that higher pressure leads to smaller bubbles, while others argued that it results in larger bubbles Research demonstrated that factors such as the valve's shape and roughness, the level of turbulence and dilution in the saturator feed downstream of the valve, and the concentration of particulate nuclei in the dilution water had minimal impact on the precipitation of air from a solution (Palaniandy *et al.* 2017; Rajapakse *et al.* 2022). In other words, these factors had little effect on the amount of air that was precipitated per unit volume of saturator feed. Nevertheless, the results provided by other researchers presented conflicting evidence regarding the morphologies and roughness of the valves. More research demonstrated that when the saturator pressure was set at 500 kPa, a nozzle with a curved channel generated bubbles with a median diameter of 49.4 μm, whereas a nozzle with a narrowing exit produced bubbles with a median diameter of 29.5 μm (Palaniandy *et al.* 2017; Rajapakse *et al.* 2022). Decreasing the saturator pressure resulted in a reduction in bubble diameters.

2.6.5.3 Air-Water Ratio

The air-water ratio is a critical parameter in dissolved air flotation (DAF) systems for wastewater treatment, significantly influencing the efficiency of particle separation and removal (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). This ratio, which determines the amount of air introduced relative to the water volume, affects bubble formation and their attachment to suspended solids (Palaniandy *et al.* 2017). Optimal air-water ratios typically enhance flotation by producing sufficient microbubbles to lift contaminants to the surface without excessive energy consumption or bubble coalescence. However, improper ratios can lead to operational inefficiencies, such as poor separation performance or clogging (Palaniandy *et al.* 2017; Rajapakse *et al.* 2022). Careful adjustment and monitoring of the air-water ratio are essential for achieving high treatment efficacy in DAF systems.

The suggested air-water ratio in dissolved air flotation (DAF) systems for treating wastewater typically ranges between 0.02 to 0.06 by volume (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). This range ensures the production of sufficient microbubbles to effectively bind with suspended solids, oils, and other impurities without causing excessive energy consumption or operational issues like turbulence. The optimal ratio may vary depending on the wastewater characteristics, such as pollutant concentration, particle size, and water temperature, and is often fine-tuned during system optimisation (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). To quantify the air-water ratio, it can be expressed using the following equation (equation 2-6):

$$A/W = \frac{C_s - C_o}{\rho_w . V} \qquad 2\text{-}6$$

where: A/W is the Air-Water ratio.

C_s is the concentration of air dissolved after pressurisation,

C_o is the initial concentration of air dissolved in water before pressurisation,

ρ_w is the density of water,

V is the volume of water.

2.6.5.4 Impact of Pressure on Air Dissolution

Pressure is crucial in the dissolved air flotation process. Numerous studies indicate that increased pressure enhances the solubility of air in water. The solubility of air in water is enhanced under pressure, facilitating the formation of smaller bubbles upon the release of that pressure. This improves the flotation process by increasing the surface area of the bubbles available for attachment to the suspended particles.

Şengör *et al.* (2009) illustrated that elevating the saturation pressure from 4 to 6 bar significantly enhanced the elimination of suspended particles and oil droplets in industrial wastewater treatment. The elevated air solubility at increased pressure produced a greater number of microbubbles, which were more effective in separating contaminants from water.

While pressure is the foundation of most DAF systems, it is not obvious that the higher the pressure, the better the results. There exists an optimal range, often between 4 and 7 bar (Palaniandy *et al.* 2017;

Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021), where air dissolution is maximised, hence enhancing the efficiency of the flotation process. Moreover, the energy expended for maintaining such pressure could negate the benefits in efficiency.

Operating pressures beyond 6 bar resulted in diminishing results, highlighting the necessity for optimisation in real applications (Ward *et al.* 2008). It was advised that systems be engineered with pressure control mechanisms that balance air dissolution with the energy expenses associated with sustaining elevated pressures.

2.6.6 DAF Application in Wastewater Treatment

Dissolved air flotation has been widely investigated for various wastewater types and different operational scales. For municipal wastewater, pilot-scale studies have demonstrated DAF's ability to significantly reduce microbial loads (up to 99.8%), phosphorus (55-81%), COD (28-39%), and suspended solids (77%), particularly as tertiary following clarification (Palaniandy *et al.* 2017). in industrial applications, DAF has been shown to remove significant fractions of contaminants from food processing effluents; for example, vegetable oil wastewater achieved approximately 75.09% oil removal and 78.3% COD reduction with optimised coagulant dosing, while poultry slaughterhouse wastewater attained TSS, protein, and lipid reduction of approximately 91%, 79%, and 93%, respectively, with biologically supported flotation (Palaniandy *et al.* 2017). Dairy processing wastewater have also achieved up to 92% removal of solids and fats with coagulant assistance, indicating the method's applicability to organic-rich streams (Muniz, Borges and da Silva 2020). comprehensive industrial reviews report that DAF typically removes 70-80% of BOD and 30-90% of COD from high organics industrial wastewaters when combined with coagulation/flocculation.

Sector	Wastewater Type	Scale	Key Parameters	Removal Efficiency
Municipal	Primary/tertiary municipal wastewater	Pilot	Enteric microbes, TP, COD, SS	Enteric microbes: 98-99.8%, TP: 55-81%, COD: 28-39%, SS: 77%
Industrial	Vegetable oil manufacturing effluent	Pilot	Oil, COD, TS, SS	Oil-75.9%, COD-78.3%, SS-85.5
Industrial	Poultry slaughterhouse wastewater	Steady State	TSS, protein, lipids	TSS-91%, protein-79%, lipids-93%

Industrial	Dairy processing wastewater	Laboratory	Solids, fat	Up to 92% removal.

2.6.6.1 Agricultural Sector

Several studies have examined the efficacy of Dissolved Air Flotation (DAF) in treating wastewater from the dairy, the slaughterhouse, and poultry sectors. The breakdown of casein is responsible for the development of sludge with a distinct butyric acid odour in dairy wastewater, which is also characterized by high levels of FOG, COD, and BOD (Dlangamandla, Ntwampe and Basitere 2018). Due to their high levels of turbidity and total suspended solids (TSS), dairy effluents released into bodies of water without first being treated may negatively affect ecosystems.

Sewage from the slaughterhouse sector is known for having significant concentrations of FOG, detergents, pathogens, nutrients, Organic Matter (OM), and occasionally heavy metals and antibiotics (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021; Philipp *et al.* 2021). In the poultry industry, DAF has also been regarded as one of the most promising modern technologies for wastewater treatment. Effluents in this business exhibit elevated concentrations of proteins, lipids, organic and inorganic phosphates, and carbohydrates (Palaniandy *et al.* 2017; Philipp *et al.* 2021).

2.6.6.2 Industrial Sector

Wastewater treatment with DAF has been implemented in the paper industry, mining, water, and sanitation sectors. In addition, it has been employed in desalination, oil recovery, and the elimination of oil and grease (Palaniandy *et al.* 2017; Tetteh 2018).

Biological treatments are usually one of the additional approaches that must be used when DAF alone is not enough to fulfil the elimination limitations set by environmental regulations (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). Radioactive waste, OM, and heavy metals are additional crucial removal factors for industrial wastewater. The elimination of heavy metals and other chemical compounds from wastewater creates a challenge in the metallurgical and mining sectors. DAF has demonstrated efficient removal outcomes for several elements and other chemical constituents.

Dissolved air flotation is widely applied across agricultural, industrial, and municipal wastewater. In agriculture, DAF treats diary and slaughterhouse effluents, removing turbidity, fats, and organic matter effectively (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). In industry, DAF separates oil and grease from vegetable oil and refinery wastewaters, achieving 90% oil removal and 92% oily pollutant reduction when combined with coagulants and optimised operating conditions (Saththasivam *et al.* 2022). DAF has also been used to remove heavy metals (Cr, Cd, Ni, Pb, Cu) from electroplating wastewater, reaching 97% removal (Saththasivam *et al.* 2022). Its efficiency is enhanced when integrated with coagulation and flocculation, making it versatile across scales from laboratory to pilot and suitable for meeting regulatory limits.

2.6.6.3 Domestic and Municipal Sector

The dissolved air flotation (DAF) technique has been successfully applied in treating many types of wastewaters produced by the residential and municipal sectors. It has demonstrated effective outcomes when used in conjunction with other methods (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). These wastewaters mostly contain OM, nitrogen, organic carbon, phosphorus, and FOG (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021), along with microorganisms and biodegradable particles.

Typically, physicochemical removal criteria such as OM (organic matter), nitrogen, organic carbon, phosphorus, and FOG (fats, oils, and grease) that come from cleaning or medicinal products and other household activities are handled using bacterial cultures in activated sludge systems (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021). Nevertheless, a study was conducted to investigate the effectiveness of integrating high-rate activated sludge systems with dissolved air flotation (DAF) to assess their impact on solid-liquid separation. Under specified coagulation conditions, a significant reduction of 78% in Total Suspended Solids (TSS) and 68% in Chemical Oxygen Demand (COD) was achieved (Wang *et al.* 2018; Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021).

2.6.7 Advantages and Disadvantages of DAF Application in Wastewater Treatment

The flotation approach offers clear advantages and disadvantages compared to the usual gravity settling technique when it comes to removing low-density particles that prefer to float (Palaniandy *et al.* 2017; Tegladza *et al.* 2021). It is widely used for treating wastewater and removing suspended solids, oils, and other contaminants. Table 2-2 highlights the key benefits, such as its efficiency in particle removal and relatively low chemical usage, as well as its limitations, including operational costs and maintenance. This comparison provides valuable insight into the suitability of DAF for different applications and its potential challenges. The pros and cons of Dissolved Air Flotation (DAF) are tabulated as follows:

Table 2-2 Advantages and Disadvantages of Dissolved Air Flotation (DAF) System (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021; Kaltchev 2024).

Aspect	Advantages	Disadvantages
Efficiency	Highly effective in removing fine solids, oils, and greases.	Not very effective for dense or heavy particles.
Water Quality	Produces high-quality effluent with low turbidity.	Can require additional filtration for certain water quality standards.
Footprint	Compact footprint relative to sedimentation tanks.	However, it may still require appreciable space depending on flow rate and scale of treatment.

Operational Flexibility	This technology can be applied in several fields such as municipal, industrial, mining, and so on.	The application requires accurate tuning of air and chemical dosing for the process to be at its best.
Chemical Usage	In some cases, reduced consumption of chemicals because of efficient solid separation.	In most of operations, chemical coagulants/flocculants are applied which increase operational cost and handling quantity of chemicals.
Settleability of Solids	Effective in the case of materials that are not easily settled.	Less effective in dense particles, which normally can settle easily.
Energy Consumption	For some types of wastewaters, the energy requirement can be low.	Energy-intensive due to air saturation and pressurisation.
Maintenance	Generally easier to maintain than some alternatives.	Requires periodic cleaning, so clogging and fouling in the release and dissolution systems do not take place.
Process Complexity	Fast start-up, ease of operation with automated controls.	The setup of the process is complicated, particularly the balancing of pressures and air-to-solid ratios.
Cost	Economical in application for those waters requiring removal of oils, greases, and fine particles.	Higher treatment option cost compared to other options for some water types.

2.7 Slow Sand Filtration (SSF)

SSF has been thoroughly researched and utilised in water purification systems, specifically for drinking water treatment and, more recently, for managing wastewater. Filtration technologies, first introduced in Europe during the 19th century, operate with relative simplicity, consume minimal energy, and naturally purify through natural processes (Angelakis, Capodaglio and Dialynas 2022; Abdiyev *et al.* 2023). In recent years, SSF has attracted interest as an environmentally sustainable and economical approach to wastewater treatment, especially in rural or resource-limited settings.

2.7.1 Principles of Sand Filtration

Fundamentals of Slow Sand Filtration The primary purification mechanisms in slow sand filters include physical, biological, chemical, and adsorption processes. Water typically penetrates a medium, commonly sand, at a depth ranging from 0.5 to 1.5 meters, with a velocity of 0.1 to 0.3 meters per hour (Verma, Daverey and Sharma 2017). Filtration comprises multiple layers in the procedure:

1. **Schmutzdecke or biological layer:** this is the essential element in SSF, forming on the surface of the sand bed throughout time. It comprises various bacteria, algae, and protozoa, making it physiologically active. It creates a biologically active region that microbial groups, including bacteria, algae, and protozoa, can inhabit and grow in. These bacteria significantly contribute to the breakdown of organic pollutants and the reduction of harmful organisms, hence influencing water quality (Liu *et al.* 2019).
2. **Physical Filtration:** Water that penetrates through the sand becomes physically trapped within the sand matrix, collecting particles and microorganisms, hence facilitating turbidity removal and pathogen reduction.
3. **Chemical:** The minimal flow rate facilitates chemical adsorption processes, enabling dissolved contaminants, including heavy metals and nutrients, to adhere to sand particles. Ion exchange, catalysis, and various chemical reactions can enhance the elimination of specific contaminants.
4. **Adsorption:** Organic and inorganic compounds adhere to the sandy grains, hence diminishing the contaminants.

The effectiveness of slow sand filtration (SSF) is influenced by several factors, including the properties of the sand (typically with grain size ranging from 0.2 to 0.5 mm), temperature, water quality, and operational parameters such as flow rate and cleaning frequency. SSF has been shown to significantly reduce bacterial and viral pathogens, including coliform bacteria and enteric viruses, making it a valuable method for mitigating waterborne infections (Angelakis, Capodaglio and Dialynas 2022; Abdiyev *et al.* 2023).

2.7.2 Pathogen Removal

Slow sand filtration (SSF) was initially developed for drinking water treatment, however, increasing demand for decentralised and sustainable wastewater treatment solutions has led its growing application in wastewater management. Recent research indicates that SSF is highly effective in removing bacterial contaminants, achieving coliform bacteria removal efficiencies of up to 99%. This performance is largely attributed to physical straining within the sand bed and biological processes occurring in the schmutzdecke layer, including predation and biodegradation. Although SSF is less effective in removing viruses compared to bacteria, studies have shown that extended filtration durations and increased residence time can enhance viral retention. Prolonged contact time, together with adjustments in flow velocity and filter depth, may improve virus removal efficiency. In addition, SSF has demonstrated considerable effectiveness in removing protozoan parasites such as Cryptosporidium and Giardia, primarily due to their large size, which makes them susceptible to physical filtration and biological decomposition within the filter media.

2.7.3 Reduction of Organic and Nutrient Contaminants

Slow sand filtration (SSF) has demonstrated effective removal of organic pollutants, with reported biochemical oxygen demand (BOD) and chemical oxygen demand (COD) reductions typically ranging between 60 and 90%, depending on influent quality and operating temperature. The removal of organic matter primarily occurs through biological degradation processes within the schmutzdecke layer and the underlying sand bed, where microbial communities metabolise biodegradable compounds. Nutrient removal in SSF, particularly for nitrogen and phosphorus, has shown variable performance. Certain nitrogenous compounds may be reduced through microbial nitrification and denitrification processes occurring within the bioactive layers of the filter. However, phosphorus removal is generally limited due to low adsorption capacity of conventional sand media. As a result, recent research has focused on the development of low-cost alternative or modified filter media, such as zeolite and activated carbon, to enhance nutrient adsorption and improve overall nutrient efficiency.

2.7.4 Advancements and Prospective Routes in SSF for Wastewater Treatment

Recent studies have concentrated on enhancing the effectiveness of SSF for wastewater treatment by integrating multi-media filtration, amalgamating SSF with artificial wetlands, and using adsorptive materials like biochar. Recent studies on biochar-augmented solid-state fermentation indicate its significant potential to enhance the removal rates of dissolved nutrients, metals, and micro-pollutants (Peter-Varbanets *et al.* 2009).

Hybrid systems that integrate SSF with sophisticated technologies like membrane filtration or built wetlands may address certain limitations of SSF. The modelling of microbial communities within the schmutzdecke is increasingly significant as solid-state fermentation research advances in comprehending their function in the treatment of complicated wastewater elements, such as pharmaceuticals and endocrine-disrupting substances.

Future study will concentrate on optimising the composition of filter medium and biofilm, as well as evaluating the efficacy of SSF in eliminating new pollutants, including micro-plastics and pharmaceuticals. An increasing interest in utilising locally obtained materials for filter media should enhance the accessibility of SSF for rural and resource-constrained populations.

2.7.5 Factors Affecting Sand Filtration

Biological filtration is the most effective way for enhancing the physical, chemical, and microbiological quality of water. While traversing the filter, the water is exposed to mechanisms that have the potential to enhance the quality of the incoming water. Purification mechanisms, transport mechanisms, and attachment mechanisms are the internal mechanisms that enhance the water quality. The mechanisms are interconnected to enhance the effluent water quality (Verma, Daverey and Sharma 2017). Certain mechanisms rely on the flow rate passing through the filter. The cross-sectional area of the sand, the water loading head (hydraulic head) above the outflow of the sand filter, the length of the sand filter, the properties of the circulating fluid, and the properties of the sand all influence the flow rate through the filter. High flow rates through the filter bed can be achieved by utilising coarser sand particles or by lowering the sand height (Verma, Daverey and Sharma 2017). The flow resistance is increased by smaller sand particles because they have a larger surface area per unit volume than coarser ones. If the flow inside the filter becomes turbulent, the pressure loss will further rise. The presence of sand particulates within the filter will decrease the cross-sectional area available for water flow. By virtue of the decreased surface area, the fluid will be compelled to exert pressure on the sand particles, thereby augmenting the velocity within the sand bed (Cescon and Jiang 2020).

2.7.5.1 Transport mechanisms

A transportation mechanism facilitates the interaction between sand grains and contaminants in the water, such as microorganisms and particles. The physical characteristics of particles; namely their size, shape, and density; are the primary determinants of transport mechanisms (Verma, Daverey and Sharma 2017). Straining, sedimentation, inertial and centrifugal forces, diffusion, and electrometric attraction are a few of the transport processes (Cescon and Jiang 2020).

(a) Straining,

Straining is an effect that takes place when the particle size of the sand is greater than the pore aperture that separates the granules. It is not influenced by the filtration rate. The process occurs predominantly on the filter's surface (Cescon and Jiang 2020). The filtration process experiences an increase in strain and headloss across the upper sand layer due to the reduction in pore volume caused by particles in the influent water settling between the sand grains (Cescon and Jiang 2020). Consequently, to eliminate the larger particulates, straining in the sand filter should be prevented through preventative pre-treatment. Additionally, as the schmutzdecke, a purification mechanism within the filter that will be elaborated upon subsequently (Verma, Daverey and Sharma 2017), develops, the straining mechanism intensifies.

(b) Sedimentation

Gravitational forces are utilised by sedimentation to eliminate particulates from the influent water. The efficiency of particle removal through a filter is determined by both the settling velocity of the dispersed matter and the flow velocity of the fluid (Cescon and Jiang 2020). This is particularly evident when considering large and dense particles. Sedimentation within the filter utilises the entire upward-facing surface of the particle media, as opposed to solely the bottom, as compared to a conventional settling tank (Cescon and Jiang 2020).

(c) Inertial and centrifugal forces.

This phenomenon occurs when suspended particulates, which have a greater density than water, exit streamlines and make contact with sand grains (Cescon and Jiang 2020). The inertial mechanism is observed to be greater under higher surface loads, as it is non-existent at low Reynolds numbers and velocities (Cescon and Jiang 2020).

(d) Diffusion,

Brownian motion in fluids, also known as diffusion, takes place throughout the entire depth of the filter (Cescon and Jiang 2020). It primarily contacts the filtration media with extremely small suspended matter (Cescon and Jiang 2020) and is unaffected by the filtration rate, even when the water is not streaming through the filter (Cescon and Jiang 2020). Incoming water particles will move randomly between flows until they hit a media grain (Thames Water and University of Surrey, 2005). Diffusion is influenced by the temperature of the water, the size of the suspended matter, and the particle size of the media. Denser media and suspended matter, as well as smaller grain sizes, result in greater diffusion (Cescon and Jiang 2020).

(e) Electrostatic and electrometric attraction

Following contact, electrostatic and electrometric attraction maintain the adhesion of particulates to the granule of media.

(f) Hydrodynamic action

The hydrodynamic action in the vicinity of the media granules is determined by the velocity gradient of the suspended matter. The particles in the influent water tend to rotate because of the velocity gradient; this generates a pressure difference across the particles, which subsequently contacts the media grains. This does not constitute the filter's primary mechanism (Cescon and Jiang 2020).

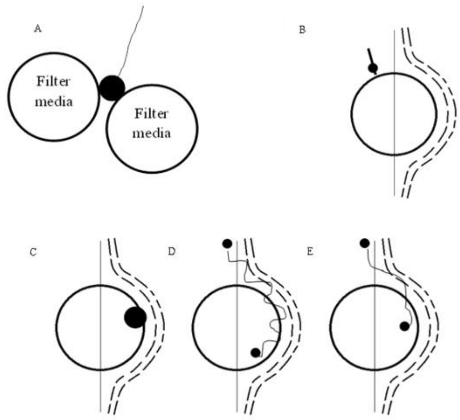


Figure 2-6 Illustration of water filtration transport mechanism: a) straining, b) sedimentation, c) interception, d) diffusion, e) hydrodynamic (Cescon and Jiang 2020).

2.7.5.2 Attachment mechanisms

Facilitates the adherence of particulates to the media grain subsequent to their contact. Adherence, Van der Waals force, and electrostatic attraction are the attachment mechanisms.

(a) Electrostatic attraction.

Depending on the electrostatic charge of the specific matter, the particles in the influent water may be repelled or attracted to the granule of media (Hube et al. 2020). Particles will adhere to the stream as it passes through the filter until it encounters an opposing charged particle of media.

(b) Van der Waals force

After particles have come into contact, Van der Waals force aids in maintaining them at the granule surface. Under some circumstances, particles may be attracted to the grain of a medium, despite the presence of a weak force.

(c) Adherence.

While water passes through the filter, organic debris is trapped on the surface of the sand grains. Subsequently, a viscous layer known as zoogloeal forms on top of the schmutzdecke, consisting of microorganisms and bacteria that attach to particles of organic and non-reactive substances in the incoming water (Liu et al. 2019). The organic stuff becomes incorporated into the zoogloeal, whereas the inert materials is eliminated when the sand is removed (Verma, Daverey and Sharma 2017; Hube *et al.* 2020).

2.7.5.3 Purification mechanisms

In contrast to the rapid sand filter, the slow sand filter is capable of generating a biological layer that enhances the chemical and biological quality of water (Verma, Daverey and Sharma 2017; Liu *et al.* 2019). A viscous film with a reddish-brown hue, it is composed of bacteria, protozoa, algae, and other decomposing organic matter. The biological layer enhances the water quality by eliminating and decomposing microorganisms and organic matter in the influent water. The biological layer absorbs, and digests organic matter and microorganisms present in the influent water for energy and to produce cell material (dissimilation and assimilation, respectively).

Bacterial activity persists for a distance of 30-40 cm and is influenced by the organic matter present in the influent water. Lower 30-40 cm, bacterial activity is relatively low in comparison to the upper stages because a significant portion of the material is consumed during earlier phases (Liu *et al.* 2019; Maiyo, Dasika and Jafvert 2023). To achieve an effective biochemical oxidation of organic matter, temperature, time, and oxygen concentration are crucial. Additionally, microorganisms require aerobic conditions within the filter. An anaerobic environment will promote the formation of substances that produce odour and flavour, such as hydrogen sulphide and ammonia. Insufficient levels of dissolved oxygen can also lead to the formation of dissolved metals, such as iron and manganese, rendering the water unfit for potable and cleansing purposes. To prevent anaerobic conditions, the average dissolved oxygen concentration must be a minimum of 3 mg/L (Verma, Daverey and Sharma 2017; Liu *et al.* 2019).

2.7.6 Comparison with Other Filters

Compared to rapid sand filtration (RSF), SSF does not use chemical coagulants and is simpler to manage, good for low-resource settings and rural areas. But RSF systems have faster filtration rate and require less land area. Hybrid systems combining SSF with RSF have demonstrated to work in some studies especially when paired with disinfection methods like chlorination for urban areas (Logsdon 2002).

2.7.6.1 Slow sand filters

A slow sand filter is often built within an exposed concrete enclosure, allowing water to pass through the filter from its upper to lower section. Water is sent downstream via the under-drainage system to either the customers or for further treatment. To prevent fine sand from being carried in effluent water, the sand is placed on top of a layer of gravel. Standard slow sand filters typically consist of a layer of fine sand ranging from 0.8 - 1.2 m in depth, with a grain size between 0.2 - 0.4 mm The sand bed must be underpinned by a layer of gravel measuring 0.3 meters in depth and composed of various sizes and depths. A stratum of fine gravel ought to underlie the fine sand, succeeded by medium gravel and coarse gravel. The water level over the sand surface typically ranges from 1.2 to 1.8 m (Maiyo, Dasika and Jafvert 2023). A slow sand filter should contain sand in the range of 0.6 to 1.2 m below the sand's surface and raw water that is between 1 and 1.5 meters below it (Liu *et al.* 2019; Maiyo, Dasika and Jafvert 2023)The recommended range for flow rates in slow sand filters is 0.1–0.4 $m^3/m^2/h$.

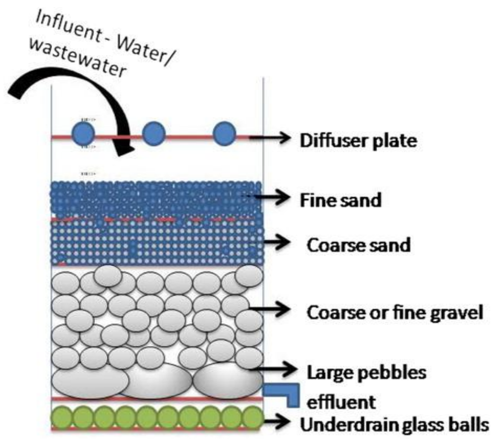


Figure 2-7 A diagram of a slow sand filter, illustrating structure of fine sand, gravel and underdrain system (Verma, Daverey and Sharma 2017)

2.7.6.2 Rapid sand filters

Rapid sand filters and pressure sand filters are similar, with the only distinction being that the pressure filter operates under pressure within a sealed container. While slow sand filters work at moderate loading rates, pressure and rapid sand filters both operate at high velocities through the filter (Bayo, López-Castellanos and Olmos 2020). The rapid sand filter is essentially an open pool containing a layer of sand, which is supported by larger grains at the bottom (Gude, Rietveld and Van Halem 2018; Bayo, López-Castellanos and Olmos 2020). Due to larger sand sizes, the filter has a better filtering rate than slow sand filters. The optimal sand size for a quick sand filter is typically in the range of 0.6-2.0 mm, resulting in flow rates of 5-15 $m^3/m^2/h$ (Vries et al. 2017). Cescon and Jiang (2020) state that the depth of sand in a rapid sand filter ranges from 0.5 to 0.75 meters, with a flow rate of 6 to 8 cubic meters per square meter per hour. The typical head loss in a well-maintained fast sand filter should be around 0.3 meters. Once the head loss in the filter reaches a range of 1.5-2 m as a result of blockage, it needs cleaning through the process of backwashing. Rapid sand filters work best when the turbidity of the influent water is 5 NTU or below, and the filtrated water should be 0.1 NTU (Vries *et al.* 2017; Cescon and Jiang 2020). In contrast to slow sand filters, rapid sand filters require just 2-5% of the area of slow sand filters and operate 20–50 times faster (Vries *et al.* 2017; Cescon and Jiang 2020).

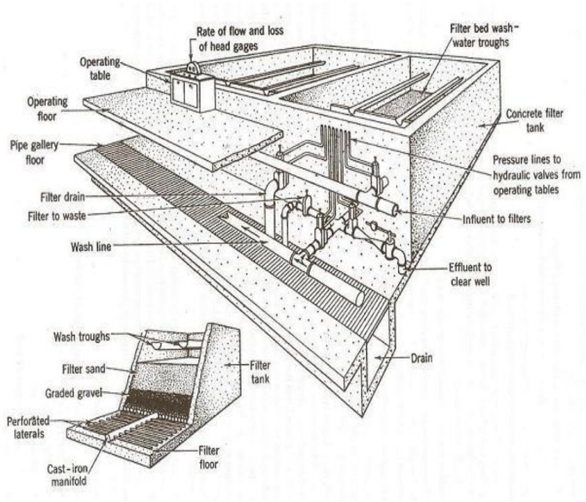


Figure 2-8 A schematic representation of a rapid sand filter, highlighting its key components (Ungureanu, Vlăduț and Paraschiv 2020).

2.7.6.3 Pressure sand filters

Pressure filters are enclosed filters that utilize overpressure to enhance the pace of filtering. The maximum head loss is greater. Pressure filters are commonly employed to reduce the turbidity of groundwater by removing iron and manganese (Binnie et al., 2002). Pressure sand filters use a bed of granular media (typically silica sand) to trap and remove impurities from water as it passes through under pressure. Pressure sand filters operate on the principle of mechanical straining and adsorption. In wastewater treatment they are typically used to remove suspended solids, turbidity, and particulates matter from water. They are especially effective as a pre-treatment step in both municipal and industrial wastewater treatment processes.

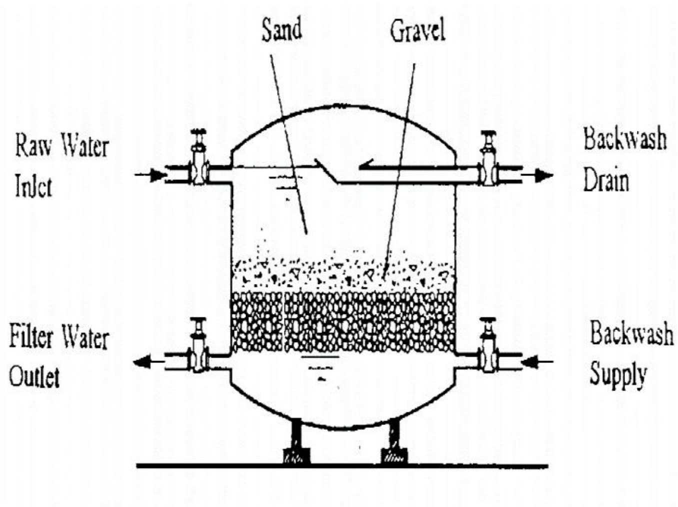


Figure 2-9 An illustration of a pressure sand filter, showcasing its enclosed cylindrical design with layers of sand and gravel (Ungureanu, Vlăduț and Paraschiv 2020).

2.7.7 Advantages and Disadvantages of Slow Sand Filtration in Wastewater Treatment

Table 2-3 summarises the key advantages and disadvantages of SSF, a traditional and effective method for wastewater treatment. Slow sand filtration relies on biological and physical processes to remove contaminants, making it a natural and cost-effective solution. However, its applicability may be limited by certain operational and design constraints, as discussed in the *Table 2-3*.

Table 2-3 Advantages and Disadvantages of Slow Sand Filtration in Wastewater Treatment (Gibbs 2018; Ntobela 2021)

Aspect	Advantages	Disadvantages
Efficiency	Effective in removing suspended solids, pathogens, and some organic matter.	Less effective for dissolved contaminants, heavy metals, and chemicals.
Water Quality	Produces of high quality, with biological stability, low turbidity, and low pathogen content	Requires regular maintenance to avoid clogging and to maintain water quality.
Footprint	Relatively small when considering most filtration systems	Requires land area, though this depends on the quantity of water to be treated
Operational Flexibility	Simple design with simple operation. Thus, appropriate for small-scale and decentralized systems	The treatment capacity is limited, and the technology is not best suited for high-turbidity or highly contaminated water.
Chemical Usage	Requires no chemical additives, making it an eco-friendly option	May still require chlorination or other disinfection for added pathogen control
Settleability of Solids	Handles low turbidity water satisfactorily; the biological layer captures fine particles well.	Higher turbidity waters will clog it; frequent cleaning is required.
Energy Consumption	Very low energy consumption since it is filtration by gravity.	Gravity flow dependent; in some configurations, energy input for pumping may be needed

Maintenance	Easy maintenance with rare but deep cleaning of the sand layer	Biological layer needs to be managed very precisely; cleaning is quite labour-intensive
Process Complexity	Straightforward operation with minimal technology or training required	The biological layer needs time to develop and mature for optimal filtration, which can delay the initial setup
Cost	- Low operational cost due to lack of chemicals and low energy requirements	Initial construction can be costly, and land acquisition may increase total costs

Slow Sand Filtration (SSF) is highly regarded for its affordability, simplicity, and capacity to generate high-quality water with minimal chemical inputs, particularly in rural and small-scale applications. Nonetheless, it is constrained by treatment capacity, the risk of obstruction from high-turbidity water, and the necessity for regular maintenance of the filter medium.

2.5 Summary of Literature

The literature review explores wastewater treatment challenges and technologies, focusing on the fruit and vegetable process industry (FVPI) in South Africa, particularly in KwaZulu-Natal (KZN). Wastewater management remains significant issue due to increasing water demand, aging infrastructure, and poor enforcement of environmental regulations. In KZN, major rivers such as uMngeni and uThukela are severely affected by industrial wastewater discharge, leading to eutrophication and loss of diversity. Many wastewater treatment plants (WWTPs) operate below standard due to mechanical failures, lack of skilled personnel, and inadequate funding. Additionally, industrial wastewater, particularly from FVPI, contains high concentrations of organic matter, suspended solids, and nutrients, making effective treatment a challenge.

To address these challenges, the study explores three wastewater treatment methods: Electrocoagulation (EC), Dissolved Air Flotation (DAF), and Slow Sand Filtration (SSF). Electrocoagulation is a process that applies electric current to destabilise pollutants, forming flocs that can be easily removed. It is effective in eliminating heavy metals, organic matter, and turbidity, but its efficiency depends on factors such as electrode material, current density, and pH. However, issues like electrode corrosion and high energy consumption pose challenges for large scale applications.

Dissolved Air Flotation, on the other hand, uses microbubbles to attach to suspended solids, allowing them to float and be removed. This method is highly effective for eliminating oils, grease, and fine solids, but has high operational and maintenance costs. Slow Sand Filtration relies on biological and physical filtration to remove contaminants and pathogens. While it is low-cost, chemical free treatment method, it requires significant land space and regular maintenance to prevent clogging.

The literature review concludes that while EC, DAF, and SSF are effective wastewater treatment methods, their efficiency varies depending on specific wastewater characteristics. The combination of these technologies may offer and optimised approach to wastewater treatment in the FVPI. Additionally, the study highlights the need for improved infrastructure, optimised operational conditions, and sustainable treatment methods to enhance wastewater management and water reuse in South Africa.

CHAPTER 3 METHODOLOGY

3.1 Introduction

This chapter presents the feasibility study of effluent treatment, water sample source, the chemicals used, the materials and equipment used, water quality parameters measured, and the respective standard analytical methods used. The three treatment methods were used to treat the wastewater collected from the bulk market, namely, electrocoagulation, dissolved air flotation, and slow sand filtration. These methods were selected based on their potential effectiveness in removing contaminants typically found in the market effluent, each were systematically evaluated to determine its suitability for water reuse application. Firstly, the one-factor-at-a-time (OFAT) approach was used to evaluate the key factors that affect the effectiveness of each technology in removing specific pollutants from wastewater. Response surface methodology (RSM) was the tool applied to design the experimental runs, analyse the data, and optimise the important interacting factors. Electrocoagulation (EC) and dissolved air flotation (DAF) were implemented using the Box-Behnken Design (BBD), while slow sand filtration was implemented using the Central Composite Design (CCD).

The BBD was used for the EC and DAF processes because these methods involved three input variables and did not require the inclusion of extreme values in the experimental range. BBD if efficient for RSM when working within a moderate, well-defined range of operating conditions. It requires fewer experimental runs than other designs such as CCD, which makes it cost effective and practical for preliminary optimisation studies (Myers, Montgomery and Anderson-Cook 2016).

In contrast, the CCD was used for SSF because this method involved continuous process variables where exploring a wide range, including extreme values, was important to understand the full response behaviour. CCD includes axial points outside the factorial design space, which provides more information to curvature in the response; which is essential for optimising biological or filtration based processes where non-linear effects are more likely to occur. CCD is especially suitable for modelling quadratic effects, which are commonly observed in natural filtration systems like SSF (Anderson and Whitcomb 2016; Myers, Montgomery and Anderson-Cook 2016).

3.2 Feasibility Study on Wastewater Treatment and Resource Optimisation in the FVPI

The FVPI plays a critical role in food supply chains but faces significant environmental and operational challenges, including high water and energy usage, wastewater production, and solid waste generation. The Clairwood Bulk Market, generating 4.219 m^3/d of effluent according to municipal water and effluent records, faces significant challenges in wastewater management due to its high water usage and the environmental impact of biodegradable and saline waste. This study evaluates the feasibility of using slow sand filtration (SSF), electrocoagulation (EC), and dissolved air flotation (DAF) to address these issues. While SSF offers a low-cost option for small-scale applications, it is less effective for high organic loads. EC and DAF are more efficient and scalable, with EC excelling in removing biodegradable matter and DAF effectively handling suspended solids and grease. Moreover, these technologies can mitigate wastewater challenges in the FVPI operations, offering a sustainable solution for wastewater treatment and resource optimisation.

3.3 Materials and analytical equipment

This section outlines the material and analytical equipment required for the experimental evaluation of wastewater treatment methods at the Clairwood Bulk Market. The selected tools and materials will support the implementation and assessment of SSF, EC and DAF processes, focusing on pollutant removal efficiency, scalability and operational feasibility.

3.3.1 Samples

The wastewater effluent samples were provided by the Clairwood fresh produce bulk market located in the eThekwini municipality of KwaZulu Natal, South Africa. The samples were obtained from three distinct locations, specifically the distribution centre (DC), trader's hall (TH), and the final effluent. The wastewater, originating from various operations such as fruit and vegetable washing, bathing and ablution systems, process water, and bund area washing, was collected and stored in an equalisation tank. To maintain the physicochemical characteristics of the samples, they were maintained in airtight containers made of polypropylene in the laboratory until being utilised. The samples employed in this study were acquired by the process of grab sampling, wherein 40 litres were collected every week from the equalisation tank using plastic scoops. Grab sampling was chosen for its simplicity and ability to provide immediate representative samples of the effluent at specific points in time. However, it may not capture temporal variations in wastewater characteristics, which could introduce some bias. To minimise this limitation, samples were collected consistently from three key locations (Distribution Centre, Trader’s Hall, and Final Effluent) and stored in airtight polypropylene containers to preserve their physiochemical properties until analysis.

For each objective, the samples were characterised during collection, and the performance criteria were precisely outlined. Samples were evaluated before and after treatment, as well as during optimisation, utilizing all three treatment methods (EC, DAF, and SSF treatment). *Table 4-1* contains the average wastewater sample information from the different sampling points, while Appendix B provides individual summaries of the findings sampling point for six months.

3.3.2 Analytical equipment

The efficacy of EC, DAF, and filtration in treating FVPI wastewater was evaluated by analysing the common critical pollutants from various technologies. The parameters were subjected to analysis according to their respective standards, utilizing the equipment outlined in Table 3-1.

Table 3-1 Compilation of Instruments and Methods Employed for Water Quality Assessment.

Water quality test	Instrument used	Method
pH	Hach DR890 potable colorimeter	Standard method
COD (mg/L)	Hanna HI 83099 COD and multi-parameter photometer	Standard method EPA 410.4 and USEPA reactor digestion method 8000
TDS (mg/L)	Hach DR890 potable colorimeter	Standard method
Turbidity (NTU)	Hach 2100N turbidimeter	Standard method
Ammonia (mg/L)	Multi-parameter photometer	Standard method
Nitrate (mg/L)	Multi-parameter photometer	Standard method
Nitrite (mg/L)	Multi-parameter photometer	Standard method
Ortho-Phosphate (mg/L)	Multi-parameter photometer	Standard method
Colour (Pt-Co)	Multi-parameter photometer	Standard method
Conductivity (mS/m)	Hach DR890 potable colorimeter	Standard method
Faecal Coliforms (CFU/100mL)	Dipslide Incubator	Standard microbiological method
Fungi (CFU/100mL)	Dipslide Incubator	Standard microbiological method
TSS (mg/L)	Filter, Drying oven	Standard gravimetric method (ALPHA 2540 D)

The selected parameters provide a comprehensive overview of wastewater quality. Parameters such as COD and TSS indicate organic and suspended load, nutrients (ammonia, nitrate, nitrite, ortho-phosphate) reflect potential eutrophication, turbidity and colour indicate visual water quality, and microbial indicators (faecal coliforms, fungi) assess health risk. Conductivity, pH, and TDS further characterise general water chemistry, which is critical for optimising treatment process performance.

3.4 Experimental setup

Experiments were setup to evaluate the performance of SSF, EC, and DAF in treating wastewater from the Clairwood Bulk Market. The setup includes detailed configurations, operational parameters, and control measures to ensure accurate, replicable results for assessing the effectiveness and feasibility of each treatment method.

3.4.1 Electrocoagulation setup

The experimental procedure involved conducting electrocoagulation treatments in glass beakers with a working capacity of 2 L. The experiment employed zinc electrodes as the anode and copper electrodes as the cathode. A direct-current power supply was used to connect the electrodes. During all tests, the electrodes were completely immersed in wastewater within a glass beaker and maintained at a consistent room temperature of 25 ± 0.5°C. A jar test flocculator JLT6 bench with blade stirrers was used to agitate the EC setup. Before conducting each experimental trial, the electrodes underwent mechanical polishing using abrasive paper and were thereafter washed with deionised water to eliminate any solid residue present on their surface. To eliminate any persistent contaminants on the surface, the electrodes were immersed in a 0.1 M HCl solution for 5 minutes. Following the completion of each experimental run, a sample of the supernatant was collected and subjected to analysis for chemical oxygen demand (COD) and physical properties to determine the effectiveness of the process.

(a) (b)

Figure 3-1 Demonstration of EC setup (a) inducing voltage in the wastewater (b) flocculation taking place.

The visual representation of the apparatus depicted in Figure 3-1(a) illustrates the ongoing process of coagulation, whereby voltage was applied to the wastewater through the DC supply. Conversely, (b) illustrates the occurrence of flocculation after the deactivation of both the agitator and the voltage supply. To ensure the uniform induction of voltage, the electrodes were linked in parallel.

3.4.2 Dissolved air flotation setup

The air saturator vessel was initially filled with deionised water, which accounted for about 80% of its total content, which is 1600 mL. The container was tightly sealed with an established a connection with the compressor. The water was pressurized by opening the valve that connects the air compressor and the saturator vessel and then activating the compressor. The system was infused with compressed air until the pressure rose to the designated range of 200-600 kPa. After reaching the desired pressure, the compressor was deactivated, and the valve was shut.

Four 2 L containers were used to contain the wastewater, each filled to approximately 60-80% of their total volume. The studies were carried out under controlled settings, which included rapid agitation at a speed of 100 to 300 revolutions per minute for 2 minutes, followed by moderate agitation at a speed of 30 revolutions per minute for 15 minutes. The outlet isolators of the distribution manifold and the saturator outlet were fully opened after the completion of the mixing procedure. The wastewater samples were subjected to the introduction of an air/water combination up to the 2 L brim.

The test jar was thereafter allowed to remain undisturbed for a period ranging for 30 minutes to facilitate the ascent of the sludge to the surface. Samples of 50 mL were collected from each output port using designated sample A bottles after a floating time of 10 minutes, in preparation for further analysis. The evaluation of the treatment process's efficiency was conducted with respect to Total Suspended Solids (TSS), COD, turbidity colour, pH and conductivity.

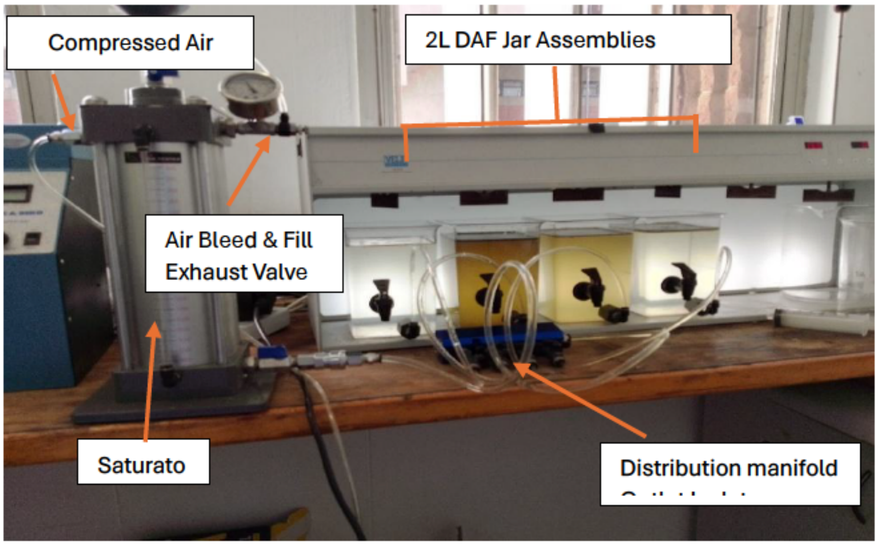


Figure 3-2 Demonstration of a dissolved air flotation (DAF) setup.

3.4.3 Filtration setup

The main part of the filter was made out of a 4 L water dispenser container. The bottom was secured with an outlet for the collection of filtered water. The base of the container was initially filled to a depth of 10 cm of coarse gravel. Above this layer, a gradual deposition of finer gravel of about 15 cm followed to reduce abrupt transitions and avoid clogs. Finally, the topmost layer consisted of 7 cm of fine sand, which served as the primary filtration medium by trapping smaller suspended particles. Figure 3-3(b) illustrates the particle distribution of the filter media. Each layer was compacted using a sieve shaker to achieve uniform compaction and reduce voids that could disrupt flow dynamics, as illustrated in Figure 3-3(a). A perforated pipe was positioned at the filter's top to ensure uniform dispersion of effluent across the sand layer. This was engineered to guarantee consistent flow and optimise interaction with the filtration medium. A pressure pump was integrated into the system to circulate wastewater through the filter. The pump's real flow rates, as seen in Table 3-2, established the pumping capacity, which varied from 40% to 80% of its maximum capacity. The pump facilitated a steady flow for different retention durations.

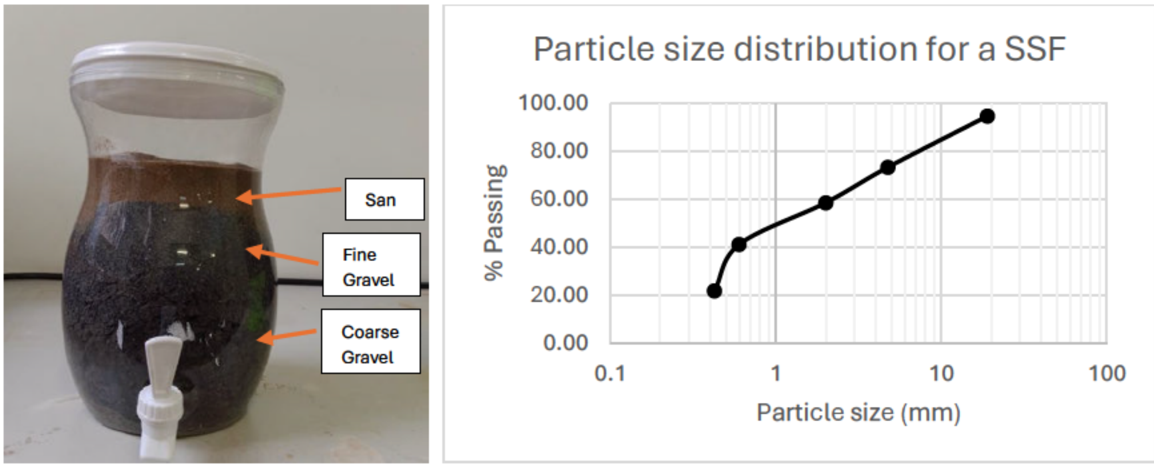


Figure 3-3 Illustration of (a) Slow Sand Filter with filter media, (b) particle size distribution of the filter.

Wastewater was collected in a 2 L glass beaker and fed into the filter by a pressure pump. Flow rates ranged from 40% to 90%, and retention times fluctuated between 10 and 50 minutes. The flow rate modifications were made according to the pump settings and flow curves to ensure regulated filtration conditions. Wastewater was transferred from the porous pipe to the layer of sand. The water flowed by gravity, initially percolating through the sand and subsequently through the gravel layers. As the water flowed towards the bottom of the container, contaminants and pathogens were caught along with suspended particles. Following every third run, the biofilm layer that formed on the surface of the sand was carefully scrapped off, without replacing the sand. This maintenance step helped to prevent clogging and maintain filtration efficiency. This mitigates obstruction and reinstates filtering efficacy. To prevent the accumulation of settled materials, the filter was backwashed after every third operation. This involved reversing the water flow through the filter to remove any accumulated material. During each run, flow rates and retention times were recorded. Flow modifications were implemented according to pump percentages: 40-80%. Correlations between flow rate, retention times, and filtration efficiency were tabulated. Samples were collected at the bottom of each run for analysis.

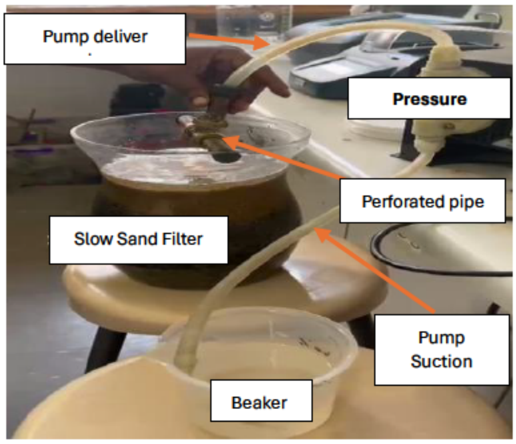


Figure 3-4 Illustration of a Slow Sand Filter setup.

Table 3-2 represents the flowrate of a pressure pump operating at a constant pressure. The pump's flow rate is adjustable, expressed in percentage of its maximum capacity. Each percentage value corresponds to a specific flow rate measured in millilitres per second (mL/s). This data serves to illustrate the relationship between the pump's adjustable flow rate setting and the actual flow delivered at 8 bar, providing a clear reference for operational adjustments and performance.

Table 3-2 Flow rate of a pressure pump at 8 bars with adjustable flow rate settings.

Flowrate (%)	Flowrate (mL/s)
100	2.17
80	1.82
60	1.43
40	0.83
20	0.33
10	0.11

The hydraulic retention time (HRT) was calculated to quantify the average time that wastewater remains in the treatment system, which is critical for assessing removal efficiency. HRT was determined using the following formula:

$$\text{HRT (min)} = \frac{V}{Q}$$

Where:

V - Volume of the treatment vessels or containers (L)

Q – Influent flowrate into the system (L/min)

For the slow sand filter, electrocoagulation, dissolved air flotation experiments, the volume of wastewater in the experimental container was measured (2 L), and the flowrate was controlled using the pump. The resulting HRT values correspond to 10, 20, 30, 40, and 50 minutes, which were tested to evaluate their effect on COD, turbidity, and colour removal.

This procedure ensures that the HRT is systematically varied and accurately known, allowing comparison of pollutant removal efficiencies at different contact times and supporting optimisation of treatment performance.

3.5 Design of Experiment

In this study, a systematic design of experiments (DoE) approach was implemented using Design Expert (version 13) software. The study utilised water from the FVPI to optimise the electrocoagulation (EC), dissolved air flotation (DAF) and slow sand filtration (SSF) processes. The DoE facilitated the collection of data necessary for developing empirical models to predict key responses, including the removal of COD, conductivity and the removal of turbidity. These models provided critical insights into the interactions and relationships between input variables and quality responses, enabling a comprehensive understanding of the dynamics of the processes and their performance under varying conditions.

The Design Expert software utilised in this study includes the following key features:

- **Construction of the Design Matrix**: The software produced a design matrix that delineated the experimental runs based on the chosen input elements. This established the foundation for the systematic collecting of response (output) data.
- **Data Analysis and Response Modelling**: Response data were inputted into the software for statistical analysis and modelling. Contour plots and 3D surface graphs have been created to illustrate the interaction effects between actual and forecasted values. ANOVA and regression coefficients (R^2) were computed to statistically characterise the polynomial correlations between input components and responses utilising actual or coded factors (-1, 0, 1).
- **Process Optimisation**: The software integrated input factor values to attain optimal conditions for the targeted outcomes. Multiple response optimisation was conducted using both numerical and graphical representations. This will guarantee that experimentation is efficient, and enlightening, and will result in the development of dependable models that can enhance DAF process performance.

3.5.1 Design of Experiment Procedure

Table 3-3 presents an overview of the input variables used in this research. The Box-Behnken Design (BBD) was employed for the DAF and EC experiments, while the Central Composite Design (CCD) was used for the SSF experiments. Both the DAF and EC process involved three input variables, whereas the SSF had two input variables. For all experiments, three response variables were measured: the percentage removal of COD, turbidity and conductivity.

The sequence of procedures employed in this research utilising Design Expert software is detailed below.

I. **Problem Identification:** The study aimed to optimise treatment processes—DAF, electrocoagulation, and slow sand filtration—by examining the influence of input variables on the removal of critical water quality indicators such as COD, conductivity, and turbidity.

II. **Selection of Experimental Design:** DAF and electrocoagulation necessitated the Box-Behnken Design (BBD) as it is optimal for studies involving three input variables. This design facilitated the execution of 17 experiments for DAF, and 15 for EC incorporating three levels and three centre points for each phase. For SSF, which includes two input variables, CCD was selected. CCD will enable the identification of optimal settings with a 13 number of experimental trials.

III. **Definition of Input Variables:** Three input variables were established for DAF and electrocoagulation. Two input variables were delineated in the context of SSF. These considerations included parameters relevant to the operational settings for each specific process.

IV. **Identification of Response Variables:** In all experimental designs, three response variables—COD removal, conductivity decrease, and turbidity removal—were designated as indicators of process performance.

V. **Development of Design Matrix:** The Design Expert program was utilised to create the suitable design matrix for BBD and CCD pertaining to DAF, electrocoagulation, and SSF, respectively. This design matrix defined the experimental runs to be conducted, along with the diverse combinations of input factor levels.

VI. **Execution of Experimental Trials:** In accordance with the experimental matrix, the necessary trials for DAF, electrocoagulation, and SSF were conducted based on the established input components and levels.

VII. **Data Entry and Statistical Analysis:** The acquired response data for COD removal, conductivity reduction, and turbidity removal were inputted into Design Expert software, which was then subjected to statistical analysis encompassing Analysis of Variation (ANOVA), regression analysis, and the formulation of model equations delineating the relationship between input factors and responses.

VIII. **Model validation and verification:** The models generated by Design Expert were validated by a comparison of anticipated values with experimental outcomes. The projected optimal conditions were validated through supplementary experimental trials if required.

IX. **Optimisation and Recommendations:** Additional optimisation and recommendations were derived from the data analysis conducted with Design Expert for all processes, including DAF, electrocoagulation, and SSF, aimed at maximising the removal of COD, conductivity, and turbidity. Conclusions and recommendations derived from these optimised conditions are presented.

This strategy facilitated a systematic and effective methodology for process optimisation utilising Design Experts.

Table 3-3 Experimental design parameters and factor values utilising BBD and CCD tailored from RSM.

Case	Process Investigated	Input Variables		Coded levels (X)		
				-1	0	1
1	Electrocoagulation	X_1	Voltage (V)	1	15.5	30
		X_2	Speed (rpm)	90	120	150
		X_3	Settling Time (min)	20	40	60
2	Dissolved Air Flotation	X_1	Pressure (kPa)	200	400	600
		X_2	Speed (rpm)	90	120	150
		X_3	Ratio (%)	60	70	80
3	Slow Sand Filtration	X_1	Flow rate (%)	40	65	90
		X_2	HRT	10	25	40

CHAPTER 4 RESULTS AND DISCUSSION

This section presents a comprehensive analysis of experimental findings from the characterisation and treatment of wastewater samples collected from the three sample points: Trader's Hall, Distribution Centre and Final Effluent. The study explored the optimisation application of the three treatment technologies namely: electrocoagulation (EC), Dissolved Air Flotation (DAF) and Slow Sand Filtration (SSF).

The one-factor-at-a-time (OFAT) approach was employed to establish baseline performance parameters for each treatment method. Subsequently, experimental designs were developed using Design Expert Version 13, with Box-Behnken Design (BBD) applied for EC and DAF; and Central Composite Design (CCD) for SSF. The experiments were conducted using the final effluent as a primary sample to identify the optimum treatment conditions.

Upon identifying the optimal operating parameters, these conditions were validated by applying the treatment methods to all three sample sources. The results provided insights into the efficiency of each treatment method across varying water qualities. The following sections detail the experimental outcomes, key findings, and implications of the findings.

4.1 Characterisation of Wastewater

The characterisation results for water samples from the Trader's Hall, the Distribution Centre, and final effluent were evaluated per the South African National Water Act (NWA) No. 36 of 1998 and the South African National Standard: SANS 241:2015 of the wastewater disposal and reuse.

The pH values of the three samples ranged from 6.91 to 6.94, which is within the acceptable limits of 5.5 to 9.5 for waste disposal, thus complying to the criteria for acidity and alkalinity. The conductivity and total dissolved solids (TDS) concentrations in the trader's hall (511.61mS/m, 340.83 mg/L); at the distribution centre (1316.72 mS/m, 845.89 mg/L) and final effluent (914.17 mS/m, 593.36 mg/L) exceed the permissible limits for wastewater discharge, indicating a significant ionic load that poses dangers to receiving water bodies.

The TSS concentrations are significantly elevated, particularly at the Distribution Centre with 2853.3 mg/L, although permitted discharge limits typically range between 25 to 50 mg/L. This requires the effective elimination of solids before discharge or reuse. The turbidity and colour of the distribution centre were 942.28 NTU and 1326.11 Pt-Co, respectively, signifying extreme contamination with both particulates and dissolved organic matter, significantly above permissible limits for discharge.

The ammonia contents are significantly elevated, particularly in the Distribution Centre at 65.37 mg/L and Final Effluent at 40.41 mg/L, exceeding the standard discharge limit of less than 10 mg/L. Elevated concentrations of nitrate and nitrite augment nutrient levels, hence fostering a risk of eutrophication in receiving water bodies. Orthophosphate concentrations in all samples were significantly elevated, ranging from 47.25 to 101.03 mg/L, above the discharge limit of < 1 mg/L, hence necessitating enhanced nutrient removal.

The COD concentrations in the distribution centre and final effluent are 464.56 mg/L and 236.44 mg/L, respectively, beyond the allowable maximum of 75 mg/L. This indicates high levels of organic pollutants, which can deplete oxygen in the receiving waters and harm aquatic life. The quantities of faecal coliforms were exceeding the permissible limits (<1000) for both the distribution centre and final effluent (refer to Table 4-1). This infers an impending risk of pathogenic contamination and poses a great health risk. No fungi were detected in any sample, potentially suggesting conditions were unfavourable for fungal growth in the wastewater.

Table 4-1 Characterisation of water samples from the Trader's Hall, Distribution Centre, and Final Effluent, highlighting physical, chemical, and microbiological parameters relevant to the water quality assessment.

Parameter	Unit	SANS Guideline Limits	Trader's Hall	Distribution Centre	Final Effluent
pH (at 25°C)	**pH**	5.5-9.5	6.94	6.91	6.92
Conductivity	**mS/m**	≤170	511.61	1316.72	914.17
Temperature	**^{0}C**	≤35	23.10	23.26	23.18
Total Dissolved Solids	**mg/L**	≤1200	340.83	845.89	593.36
Total Suspended Solids	**mg/L**	≤25 (discharge)	566.67	2583.30	1574.98
Turbidity	**NTU**	≤5 (reuse, non-potable)	18.24	942.28	480.26
Colour	**Pt-Co**	≤300	97.47	1326.11	711.79
Chemical Oxygen Demand	**mg/L**	≤75	8.33	464.56	236.44
Ammonia	**mg/L**	≤ 6	15.44	65.37	40.41
Nitrate	**mg/L**	≤ 15	3.67	9.85	6.76
Nitrite	**mg/L**	≤ 0.9	3.48	8.91	6.20
Ortho-Phosphate	**mg/L**	≤ 1	47.25	101.03	74.14
Faecal Coliforms	**CFU/100mL**	≤1000(discharge); ≤0-10 (reuse)	250.00	1000000.00	500125.00
Fungi	**CFU/100mL**	0	0.00	0.00	0.00

4.2 Treatment of Wastewater Through Electrocoagulation

This section explores the utilisation of electrocoagulation in wastewater treatment, emphasising on the optimisation of key operating conditions. The OFAT was employed to evaluate the individual effects of settling time, varying current. Moreover, RSM was utilised to evaluate the interactive effects of these characteristics. The findings provided insights into optimising EC for effective wastewater treatment in a laboratory-scale.

4.2.1 OFAT Method in Wastewater Treatment Using EC

One-Factor-at-a-Time (OFAT) approach was employed to examine the influence of individual parameters on the electrocoagulation process. Each experiment was conducted in triplicates under identical conditions to ensure repeatability and reliability of the results. This method allowed for the identification of optimal conditions for effective wastewater treatment.

4.2.1.1 Effect of Settling Time on Electrocoagulation

The influence of settling time (10, 20, 30 and 40 minutes) on turbidity, colour, and COD removal was evaluated. For electrocoagulation (EC) process, a constant current of 2.2 A was applied for a charge time of 30 minutes, after which the treated wastewater was allowed to settle. These conditions yielded the highest removal efficiencies for the targeted contaminants. The charge of 30 minutes was maintained throughout the study, consistent with Dolati *et al.* (2017), who recommended an EC charge time of 20-40 minutes for effective treatment; the average of the recommended charge time range from previous research was adopted to provide a representative and practical duration for electrocoagulation experiments..

Therefore, for our study, we decided to use the average of the recommended charge time. For each experiment, the COD, colour, and turbidity were analysed by standard methods. Figure 4-1 shows the removal efficiency for the turbidity, COD, and colour while Figure 4-2 shows the variation in pH and conductivity as measured during the experiment. Observable from Figure 4-1, all the contaminants removal were increasing as the settling time increased too.

The results indicated that removal efficiencies increased with increasing settling time within the investigated range. Moreover, the results showed that the optimum removal of COD, colour and turbidity was obtained at 40 min with removal efficiencies of 39.7, 57.5 and 62.8%, respectively. Electrolysis duration influences EC efficiency as well. Longer electrolysis times result in greater elimination (Dolati et al. 2017; Vepsäläinen and Sillanpää 2020).

Although Dolati *et al.* (2017) investigated a different type of wastewater, their recommended settling time range was used as a guideline for this study even though no plateau or reverse trend was observed within this range, 40 minutes was considered optimal under the present experimental conditions. It is acknowledged that further increases in settling time may lead to additional improvements in removal efficiency. This suggests that extended settling periods should be considered in future studies to determine the true optimal conditions.

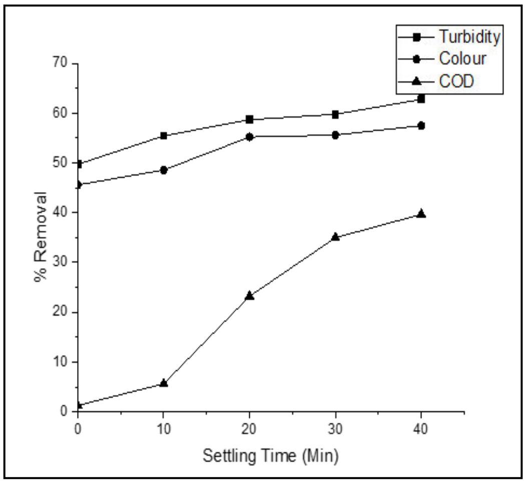


Figure 4-1 The effects of settling time on percentage removal on colour, COD and Turbidity.

Figure 4-2 shows the variation in pH and conductivity as measured during the experiment. From *Figure 4-2*, the effects of settling time on pH and conductivity were also observed. It was noticed that as the settling time increases, both pH and conductivity gradually decrease.

The quantity of dissolved materials, chemicals, and elements in the water is measured by conductivity. Measurements of conductivity are used to determine the best treatment method for removing contaminants and impurities (Butler et al. 2011; Garcia-Segura et al. 2017). In coagulation, the initial pH has an attribute in the removal efficiency of contaminants, where wastewater in alkaline and acidic conditions display low adsorption capacity for pollutants. In the neutral range of pH (6.5-8.5), the predominant species have high adsorption capacity for pollutants (Periyasamy and Muthuchamy 2018; Vepsäläinen and Sillanpää 2020).

Since neutral pH conditions (6.5-8.5) favour high adsorption capacity for pollutants, it is likely that coagulation is more effective within this range. A drop in pH and conductivity over time indicates that the system is reaching a more stable treated state (Bharti, Das and Purkait 2023). The decrease in conductivity implies that dissolved ions are being removed from the solution. As the particles settle, they can absorb and carry away dissolved ions, leading to a reduction in overall conductivity (Boinpally *et al.* 2023; Sadaf *et al.* 2024). The observed trend reinforces the importance of adequate settling time in water treatment. A longer duration allows more suspended particles to settle, reducing dissolved solids and improving overall water quality (Boinpally *et al.* 2023).

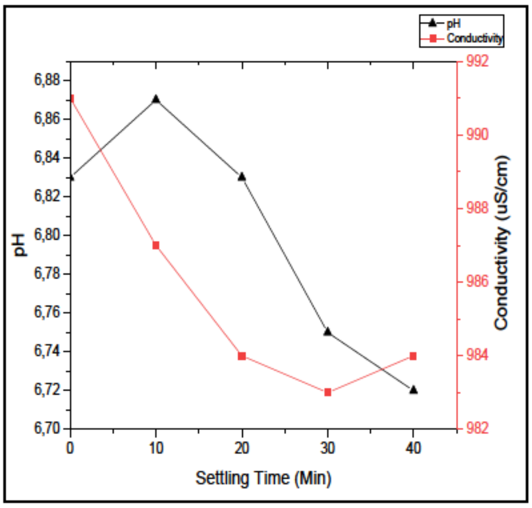


Figure 4-2 The effects of settling time on pH and conductivity.

4.2.1.2 Effect of Varying Current on Electrocoagulation

The effect of current density on the reduction of metal ions from wastewater in the batch reactor was studied with different current densities (CD) that was varied in the range of 0.88 – 4.43 A, at constant charging time (30 min), mixing speed (150 rpm) and settling time (40 min). From *Figure 4-3*, the COD removal efficiencies increase as the current density was increased. This is due to the increase of oxidizing radicals' formation. As a result, higher current densities produced better removal efficiency (Periyasamy and Muthuchamy 2018). For the current density of 0.88 and 1.77 A, the COD removal percentage was 35.8 and 40.8%, respectively at the same conditions of over a duration of 30 minutes. Furthermore, the result showed that the COD removal efficiency of electrocoagulation (EC) at a current density of 2.66 A was 53%, while for the current density of 4.43 A, it increased to 62%. From the above results, the most favourable percentage removal of COD was at 4.43 A.

To ensure reliability, all experiments were conducted in triplicates under identical conditions, and the individual results were closely examined before calculating the average values. The three independent tests produced very similar results, with only minor deviation observed, confirming a high level consistency and reproducibility. The variation between replicates were minimal, with results showing tight clustering around the mean, indicating a high level of consistency and reproducibility. From literature, it has been observed that when the electrolysis time increases, the concentration of metal ions and their hydroxide flocs increase; thus, the COD removal efficiencies increase (Vepsäläinen and Sillanpää 2020). This behaviour is due to the applied current density that determines the coagulant dosage rate, the bubble production rate and size of flocs growth resulting in a faster removal of pollutants (Periyasamy and Muthuchamy 2018; Vepsäläinen and Sillanpää 2020). In other words, by increasing the current of the cell, the amount of hydrogen bubbles at the cathode increases, resulting in a greater upwards flux and a faster removal of the pollutant and sludge flotation. Both colour and turbidity removal efficiencies readily decrease as the current is increased (from 0.89 A to 1.77 A and from 1.77 to 2.66 A), which later gradually increases when the current increases from 2.66 A to 4.43 A(Periyasamy and Muthuchamy 2018; Boinpally *et al.* 2023).

This trend contradicts findings in literature, which generally report that increasing current density improves removal performance due to higher coagulant production and gas bubble formation (Periyasamy and Muthuchamy 2018; Vepsäläinen and Sillanpää 2020). Several factors may account for the initial decline. These include excessive gas bubble generation, which may interfere with effective flocculation and settling; or the destabilisation of formed flocs due to turbulence at intermediate current levels(Boinpally *et al.* 2023).

When the current is increased, both the efficiency of removing colour and turbidity decline significantly from 58.3% to 52.3%, and from 60.6% to 48.9%, respectively (at 0.89 A to 2.66 A), but gradually increases when the current is increased from 2.66 A to 4.43 A. Literature indicates that an increase in current leads to an enhanced colour and turbidity removal (Crini and Lichtfouse 2019; Ismail and Go 2021). Therefore, the sudden decline in efficiency may be attributed to transient pH shifts, which could inhibit removal efficiency at certain current densities.

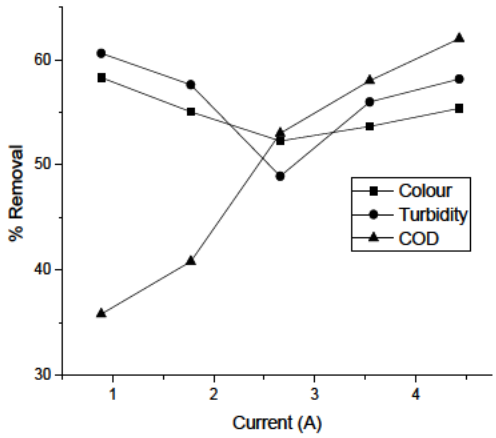


Figure 4-3. The effects of current density on percentage removal of colour, COD and Turbidity.

Conductivity represents the concentration of dissolved ions in water, which can be affected by electrochemical reactions. As current density increases, ions may be removed through coagulation, precipitation, or electrochemical reactions, leading to a decrease in conductivity. The application of current density in an electrochemical treatment process can influence pH levels depending on the dominant reactions at the electrodes. At the anode, oxidation reactions can generate acidic species, potentially decreasing pH. At the cathode, hydrogen evolution and hydroxide ion production can increase pH, leading to a more alkaline environment. The overall pH depends on the balance of these reactions and buffering capacities.

The trend observed in Figure 4-4 indicates that increasing the current density from 0.89 A to 4.43 A resulted in an overall improvement in COD removal from approximately 36% to 62%. Colour removal initially decreased from about 58% to 52% as the current increased to 2.66 A, before rising to nearly 60% at 4.43 A. Similarly, turbidity removal declined from approximately 61% to 49% and increased to about 58% at higher current densities. These results suggest that moderate current densities may cause temporary instability in treatment performance, while higher current levels promote improved contaminant removal. However, excessively high current densities may lead to undesirable pH shifts and increased energy consumption without proportional improvements in efficiency (Shahedi et al. 2020). Therefore, an optimal current density should be selected to balance effective pollutant removal and stable water quality parameters.

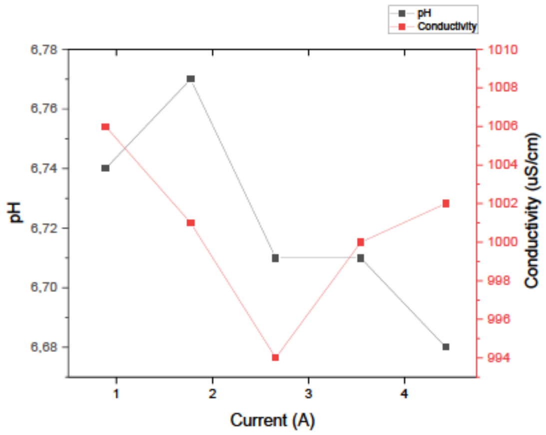


Figure 4-4. The effects of current density on pH and conductivity.

4.2.2 Evaluation of key interactive factors using of EC on a lab-scale plant using RSM

This subsection investigates the interactive effects of critical operating parameters on electrocoagulation performance, using RSM to optimise the process on a lab-scale.

4.2.2.1 Analysis of Variance (ANOVA)

The RSM was employed to evaluate the statistical parameters from ANOVA tables with Design Expert Version 13, focusing on key metrics that assist in analysing the regression model's fit to the experimental data. These encompass coefficient of determination (R^2), lack-of-fit, p-value, F-value, and additional essential values. This discussion pertains to three distinct responses: conductivity, turbidity removal, and COD removal. The F-value was used to evaluate the impact of the means among process factors. The F-values for electrical conductivity, turbidity removal, and COD removal are 36.91, 100.99, and 55.58, respectively, signifying the significance of their models. Voltage ($F = 34.16$) and running speed ($F = 31.74$) significantly influence conductivity, while voltage and settling time are the predominant variables in turbidity reduction. In contrast, voltage was the most significant variable in this model, with an F-value of 434.32, surpassing other variables in its effect on the reduction of COD. The p-value assessed the relevance of each term in the model.

At 95% confidence level, a p-value below 0.05 typically signifies that the model term is statistically significant (Karamati-Niaragh *et al.* 2019). The p-values for voltage at 0.0004, the running speed at 0.0005, and the quadratic term C^2 at less than 0.0001 indicate that these factors are highly significant in conductivity. The p-values for turbidity removal indicate that voltage at 0.0001, the running speed at 0.0011, and the settling time at 0.0003 are very significant parameters. For COD Removal, the voltage p-value is <0.0001, indicating a highly significant factor; both running speed and settling time were significant.

The analysis of the lack-of-fit values and model adequacy indicated that both models for electrical conductivity and COD removal provided a good fit to the experimental data. For electrical conductivity, the p-value of 0.3670 was greater than 0.05, which meant lack-of-fit was not statistically significant. This suggested that the model accurately represented the relationship between variables, with minimal disparity between the projected and observed values. Similarly, for COD removal, the p-value of 0.0742 also indicated a lack-of-fit that was not significant, implying that the model adequately captured the general trends of COD removal. Overall, both models demonstrated non-significant lack-of-fit values, implying that they were reliable in predicting the outcomes of the treatment processes. These results suggested that the experimental data was accurately represented by the models, and the treatment processes being modelled were well-understood.

Table 4-2 ANOVA analysis on the effects of induced voltage, mixing speed, and settling time on conductivity in electrocoagulation.

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	5628.45	6	938.08	36.91	< 0.0001	significant
A-Voltage	868.06	1	868.06	34.16	0.0004	
B-Running Speed	806.68	1	806.68	31.74	0.0005	
C-Settling Time	3.13	1	3.13	0.1230	0.7349	
BC	277.78	1	277.78	10.93	0.0108	
B^2	485.21	1	485.21	19.09	0.0024	
C^2	2996.61	1	2996.61	117.92	< 0.0001	
Residual	203.30	8	25.41			
Lack of Fit	174.56	6	29.09	2.02	0.3670	not significant
Pure Error	28.74	2	14.37			
Cor Total	5831.75	14				

Table 4-3 ANOVA analysis on the effects of induced voltage, mixing speed, and settling time on turbidity removal in electrocoagulation.

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	3111.28	10	311.13	100.99	0.0002	significant
A-Voltage	657.25	1	657.25	213.35	0.0001	
B-Running Speed	222.27	1	222.27	72.15	0.0011	
C-Settling Time	461.45	1	461.45	149.79	0.0003	
AB	267.04	1	267.04	86.68	0.0007	
AC	0.0029	1	0.0029	0.0010	0.9768	
BC	0.2938	1	0.2938	0.0954	0.7729	
A^2	15.66	1	15.66	5.08	0.0872	

B^2	177.72	1	177.72	57.69	0.0016	
C^2	13.29	1	13.29	4.31	0.1064	
AB^2	53.03	1	53.03	17.21	0.0143	
Residual	12.32	4	3.08			
Lack of Fit	10.31	2	5.16	5.13	0.1632	not significant
Pure Error	2.01	2	1.01			
Cor Total	3123.60	14				

Table 4-4 ANOVA analysis on the effects of induced voltage, mixing speed, and settling time on COD removal in electrocoagulation

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	7017.74	9	779.75	55.58	0.0002	significant
A-Voltage	6093.51	1	6093.51	434.32	< 0.0001	
B-Running Speed	650.95	1	650.95	46.40	0.0010	
C-Settling Time	199.00	1	199.00	14.18	0.0131	
AB	14.91	1	14.91	1.06	0.3499	
AC	11.51	1	11.51	0.8202	0.4066	
BC	10.99	1	10.99	0.7836	0.4166	
A^2	3.31	1	3.31	0.2356	0.6479	
B^2	35.00	1	35.00	2.49	0.1751	
C^2	0.3885	1	0.3885	0.0277	0.8744	
Residual	70.15	5	14.03			
Lack of Fit	66.64	3	22.21	12.64	0.0742	not significant
Pure Error	3.51	2	1.76			
Cor Total	7087.89	14				

4.2.2.2 Fit Statistics

The statistical table that is of importance is the fit statistics table, as illustrated in Table 4-5. Key statistical coefficients present in the fit statistics table are Adequate Precision, Standard Deviation (SD), Coefficient of determination (R^2), Predicted R^2, Adjusted R^2, Coefficient of Variation, and Mean.

Standard deviation quantifies the dispersion or variability of values relative to the mean. The standard deviations for electrical conductivity, turbidity removal, and COD removal are 5.04, 1.76, and 3.75, respectively. A small standard deviation indicates that the data points are closely clustered around the mean, whereas a large standard deviation signifies increased dispersion. Consequently, electrical conductivity exhibits larger variability than turbidity and COD removal. The mean represents the average of the response values. The mean values for Electrical Conductivity, Turbidity Removal, and COD Removal are 975, 40.85, and 53.42, respectively. These represent the average performance of each parameter in the system under the specified experimental conditions. The coefficient of determination, R^2, is a significant statistical metric that indicates the extent to which a predicted regression model aligns with actual experimental data points. The R^2 term quantifies the proportion of variation in the response variable, y, within its immediate vicinity, as elucidated by the suggested model. The value of R^2 varies from 0 to 1 (Myers, Montgomery and Anderson-Cook 2016; Montgomery 2017). A regression model that is consistently provides the best fit is characterized by an R^2 value that is close to 1.

A high R^2 value not only indicates a good fit but also implies that the chosen model captures the key trends and interactions in the experimental data, allowing meaningful interpretation of process behaviour. In this study, the turbidity removal regression model was more effective than the regression models for electrical conductivity and COD removal, with a significantly higher R^2 of 0.9961. In comparison, the model for electrical conductivity and COD removal had a lower R^2 values, highlighting that the turbidity removal process was more effectively captured by the model. This comparison underscores the relative influence and predictability of each process under the studied operational conditions, providing insight into which treatment response are most sensitive to the input variables and modelled interactions.

Consequently, numerous scholars favour the adjusted R^2 over the traditional coefficient of determination, R^2, (Othmani *et al.* 2022). A primary benefit of utilizing the modified R^2 is that the inclusion of an additional input variable does not elevate its value. Like R^2, turbidity exhibited the highest R^2 of 0.9961, demonstrating that the turbidity model was more robust. The anticipated R^2 is another significant parameter of the fit statistics, serving as an indicator for the estimated value of R^2 in the projected regression model. The predicted R^2 values for electrical conductivity, turbidity removal, and COD removal were 0.9651, 0.9961, and 0.9901, respectively, while the adjusted R^2 values were 0.9390; 0.9862, and 0.9723, respectively. The differences between the predicted and adjusted R^2 values were 0.0731, 0.1197 and 0.1238 for electrical conductivity, turbidity removal, and COD removal, respectively. The variation between the expected R^2 and predicted R^2 must be less than 0.2, as larger differences may indicate overfitting or poor predictive ability of the model. In this investigation, all measured values were below 0.2, indicating reasonable agreement between model predictions and experimental observations and confirming the reliability of the regression models. Consequently, there was no issue with either the model equation or the regression data points. The statistical parameter COV computes the ratio of standard deviation to mean expressed as a percentage (Stat-Ease, 2021).

The amount of the coefficient of variation (COV) indicates the extent of deviation of experimental data points from the mean. Design Expert software essentially suggests a low COV value since it develops a more precise regression model. The corresponding coefficients of variation for electrical conductivity, turbidity removal, and COD removal were 0.5168%, 4.30%, and 7.01%, respectively. A minimum of 0.5168% was observed in the electrical conductivity model, which suggests that it is more robust than the turbidity removal, and COD removal. The turbidity regression model is very reliable, since the standard deviation of 1.76 is minimal compared to the mean value of 40.85, evidenced by a coefficient of variation of 4.30% for turbidity. Adequacy precision (AP) quantifies the signal-to-noise ratio by establishing the limits of the projected response with the estimated error. AP allocates a greater value when the model's predicted response or signal is nearer to its corresponding error. The AP value exceeding 4.0 indicates that the regression model is indeed adequate. From Table 4-5, the AP values for Electrical Conductivity, Turbidity Removal, and COD Removal are 20.71, 37.85, and 23.95, respectively. All AP values above 4.0, signify appropriate regression models in every case. As no predicted response variables are influenced by model error, response design spaces can be defined using the projected model equations. Of all the response variables examined, the AP of 37.85 derived from the turbidity regression model was the highest. The turbidity regression model exhibits the highest signal-to-noise ratio, making it the superior model.

Table 4-5 Fit statistics for conductivity, turbidity removal and COD removal in electrocoagulation treatment.

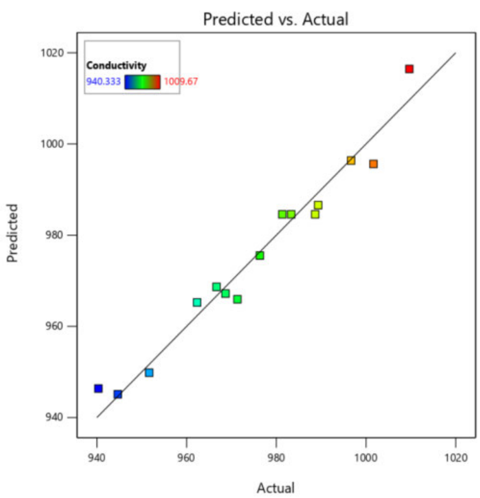
Statistical parameter	Electrical conductivity	Turbidity Removal	% COD % Removal
Standard deviation	5.04	1.76	3.75
Mean	975.51	40.85	53.42
Coefficient of Variation (COV -%)	0.5168	4.30	7.01
R^2	0.9651	0.9961	0.9901
Adjusted R^2	0.9390	0.9862	0.9723
Predicted R^2	0.8659	0.8665	0.8485
AP	20.7142	37.8501	23.9472

4.2.2.3 Model Validation

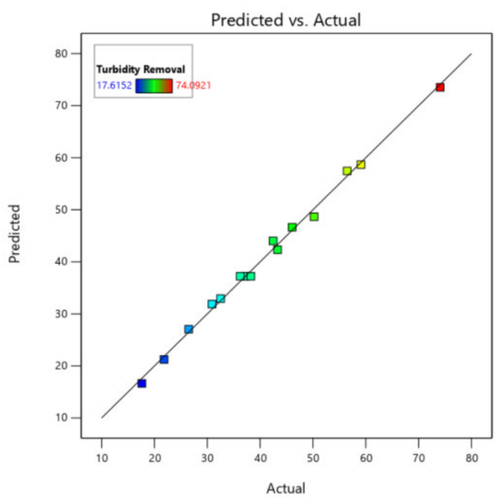
Response Surface Methodology (RSM) is an effective technique employed for modelling and analysing problems when several input variables may affect various performance metrics or responses. Validation of the responses entails verifying that the predictions generated by the RSM model are precise and congruent with the actual system being represented. The validation process typically entails a comparison of the model-predicted values with the experimental results, in addition to a verification of the model's fit to the data. One of the valuable graphs for model validation is the Predicted Vs Actual graph. Figure 4-5 presents the Predicted versus Actual graphs for all three responses: electrical conductivity, turbidity removal, and COD removal. Hypothetically, the predicted and actual response values should be randomly distributed along the 45-degree line.

The graphs indicate that the data points for electrical conductivity (Figure 4-5a), turbidity removal (Figure 4-5b), and COD removal (Figure 4-5c) are dispersed along the 45-degree line, signifying that all three regression model equations have effectively approximated the actual data points derived from the experimental work. A decent fit is indicated when residuals are randomly distributed without a clear trend. If patterns show up, it means that the model is missing some details of how the answer behaves and needs to be improved. Similarly, to the Fit Statistics section, the turbidity model exhibited significantly superior performance compared to the other models; the turbidity data points were more closely aligned with the 45-degree line than those of the other replies.

(a)



(b)



(c)

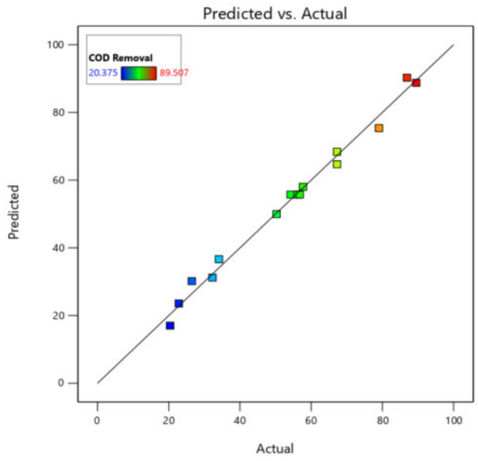


Figure 4-5 Predicted vs actual graphs for a) conductivity, b) turbidity removal, and c) COD removal through electrocoagulation treatment.

4.2.2.4 Three-dimensional surface graphs

The correlation between multiple input variables and one or more response variables is mostly depicted using three-dimensional graphics of the RSM. This surface plot provides a clear representation of how the response fluctuates with alterations in the input variables. It aids in identifying the ideal operating conditions, analysing the interaction effects of factors, and examining the configuration of the response surface. A typical 3D graph in RSM would normally contain two independent variables plotted on the X and Y axes, which in this case are running speed and induced voltage, while a dependent or response variable is plotted on the Z-axis, in this case, conductivity, turbidity removal, and COD removal, displaying the height or depth of the surface to the inputs.

Figure 4-6(a), Figure 4-6(b), and Figure 4-6(c) illustrate the three-dimensional surface graphs depicting conductivity, turbidity removal efficiency, and chemical oxygen demand (COD) removal efficiency, respectively. These pots enable a visual interpretation of the interactions between input factors and their impact on treatment performance, providing a foundation for subsequent numerical optimisation. Figure 4-6(a) illustrates the induced voltage ranging from 1 to 30 V plotted against operation speed (90-150 rpm) and conductivity. The graph indicates that an increase in induced voltage corresponds with a decrease in conductivity, indicating that higher coagulant generation rates and faster mixing speeds enhance ion removal. It has higher conductivity at lower operating speeds and decreases with increasing operating speed for induced voltage, highlighting the necessity of balancing both parameters for optimal conductivity reduction.

In Figure 4-6(b), the removal percentage decreases as operating speed increases, declining from 27% at 90 rpm to 22% at 150 rpm. Nonetheless, the percentage elimination of turbidity increased from 27% to 46%, corresponding with a voltage rise from 1 V to 30 V. The reduction of turbidity is also affected by the settling time, which increases from 20 to 46 minutes, achieving its peak at 46 minutes before seeing a slight decrease thereafter. The optimal turbidity removal is achieved at the peak induced voltage and velocity with a settling duration of 46 minutes, this is due to higher operating speed initially reduce efficiency due to reduced particle retention but interact with voltage to enhance coagulation at optimal conditions.

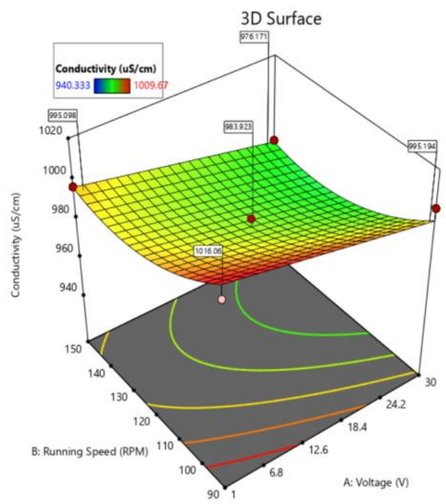
Figure 4-6 illustrates the correlation between COD removal and input variables (Figure 4-6c). The percentage of removed COD rises progressively with an increase in operating speed. The least COD elimination is achieved at the lowest voltage of 1 V and a speed of 90 rpm. An increase in voltage 1-30 V significantly enhances COD removal, peaking at 46 minutes, followed by a slight reduction from 46 to 60 minutes. Increasing voltage significantly improves COD removal, while higher speeds further enhance due to improved mass transfer and floc formation.

The 3D surface plots reveal the most significant interactions for each response:

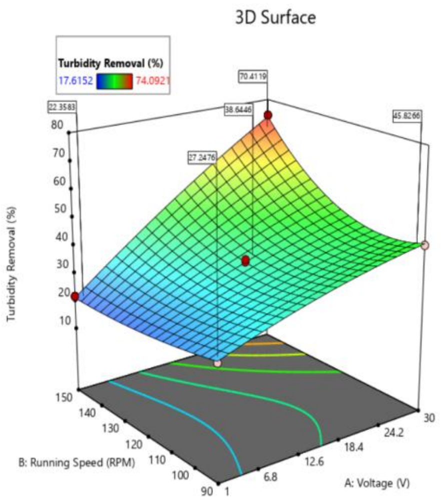
1. Viltage Vs Operating speed: strongly affects all three responses, particularly conductivity and COD removal.
2. Voltage Vs Settling time: Crucial for turbidity removal, highlighting the optimal combination for maximal particle settling.
3. Operating Speed Vs Settling time: Moderately influences turbidity and COD removal, affecting floc formation and residence time.

These interactions guide the numerical optimisation stage, helping identify the optimal combination of input factors to simultaneously maximise COD and turbidity removal while minimising conductivity.

(a)



(b)



(c)

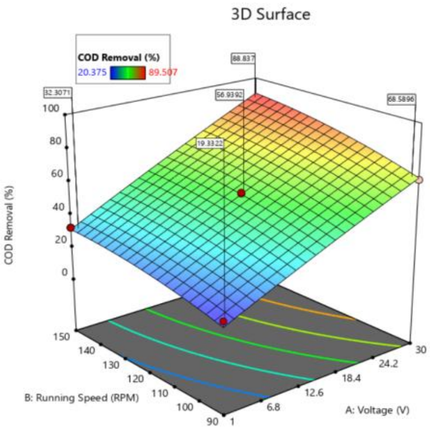


Figure 4-6: Three-dimensional (3D) response surface plots for (a) conductivity, (b) turbidity removal, and (c) COD removal through electrocoagulation treatment.

4.2.2.5 Numerical optimization

Numerical optimization employing RSM is concerned with the establishment of an optimal process by evaluating the correlation between the response and multiple input variables. It works well because it combines experimental design, statistical modelling, and optimization methods to solve difficult problems with more than one input variable. It is utilized to deliver optimal solutions with a minimal number of experiments in comparison to traditional procedures.

To numerically optimize the response outputs—electrical conductivity, turbidity removal, and COD removal—ramps were utilized to achieve the induced voltage, mixing speed, and settling time (Figure 4-7). Figure 4-7 clearly indicates that the optimal output variables for electrical conductivity, turbidity removal, and COD removal were 971.67 µS/cm, 75.28%, and 91.04%, respectively. These values were achieved at an induced voltage of 29.67 V, a mixing speed of 150 rpm, and a settling period of 46 minutes. A desirability of 76.4% was demonstrated by the study's results.

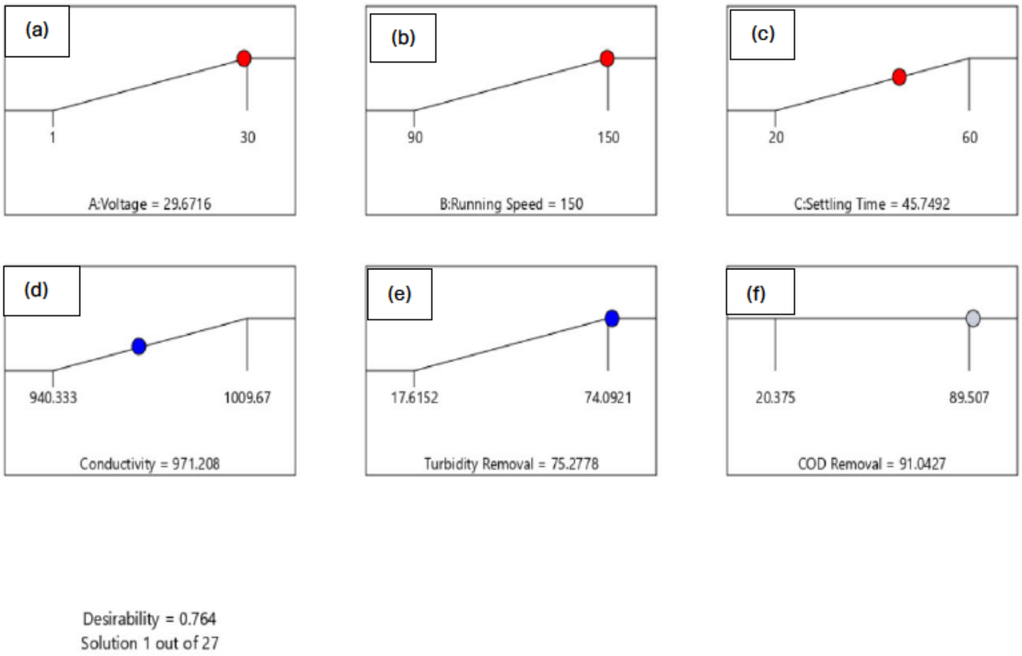


Figure 4-7: Numerical ramp plot of optimum conditions voltage (a), running speed (b), settling time (c), conductivity (d), turbidity removal (e), and COD removal (f)) with desirability at 76.4% confidence level in the treatment of wastewater through electrocoagulation.

The optimisation objective was to maximise turbidity and COD removal while minimising electrical conductivity. Each response was transformed into an individual desirability function ranging from 0 to 1, where 0 represents an undesirable response and 1 represents the most desirable response. The overall desirability was obtained by combining the individual desirability values using geometric mean.

4.2.2.6 Validation of optimum conditions

The experimental validation of the RSM model involved comparing predicted and actual values of key response variables, namely conductivity, turbidity, and COD, under the optimum conditions as determined by the model. The optimal parameters for the input variables were as follows: voltage of 29.67 V, mixing speed of 150 rpm, and settling time of 46 minutes. Predicted values for conductivity (971.21 µS/cm), turbidity (75.28%), and COD (91.04%) were estimated.

The conductivity measured 967.00 µS/cm, turbidity was 74.32% removal, and COD was 88.90% removal during the actual execution at specified conditions. The percentage discrepancies between the predicted and actual values were computed to assess the model's accuracy, revealing marginal differences: 0.43% for conductivity, 1.28% for turbidity, and 2.35% for COD as shown in *Table 4-6*.

The minor percentage discrepancies suggest that the RSM model has accurately predicted the system's behaviour under the investigated conditions. The close agreement between the predicted and experimental values confirms reliability in the use of the model for process parameter optimization. Nevertheless, minor discrepancies, particularly in COD, may suggest the necessity for additional refinement of either the model or the experimental configuration to achieve an enhanced level of precision.

The experimental results validate the robustness of the RSM model in predicting optimal conditions for the specified response variables. The minimal percentage errors indicate the model's efficacy in directing experimental settings to achieve intended outcomes with a high level of precision.

Table 4-6 Validation of optimum conditions on conductivity, turbidity removal, and COD removal in electrocoagulation treatment.

Response	RSM predicted results	Experimental results	Difference in results (%)
Conductivity (µS/cm)	971.21	967.00	0.43
Turbidity (%)	75.28	74.32	1.28
COD (%)	91.04	88.90	2.35

4.3 Treatment of Wastewater through Dissolved Air Flotation

This section investigates the application of DAF for wastewater treatment, focusing on the optimisation of key operational parameters. The OFAT method was employed to assess the individual effects of factors. Furthermore, RSM was utilised to examine the interactive effects of these variables on the treatment performance. The results offer valuable insights for optimising the DAF to achieve efficient wastewater treatment in a lab-scale environment.

4.3.1 OFAT Method in Wastewater Treatment Using DAF

The OFAT method was used to evaluate the impact of individual operational parameters on a DAF process. This approach helped identify optimal conditions for achieving effective wastewater treatment.

4.3.1.1 Level variation

The impact of different wastewater levels (50%, 60%, 70%, 80%, and 90% full) in 2 L containers was assessed in terms of their effect on the efficiency of removing turbidity, colour, and COD. The experiment utilized the ideal parameters, including a mixing speed of 200 rpm and a pressure of 400 kPa. The COD, colour, and turbidity of each experiment were analysed using established procedures. Figure 4-8 displays the effectiveness of removing turbidity, COD, and colour. Figure 4-9 illustrates the changes in pH and conductivity observed throughout the experiment. As shown in Figure 4-8, the elimination of COD and colour decreases steadily as the amount increases. The findings indicate a little increase in colour removal, from 50% to 60%, followed by a progressive decline, reaching an optimal removal rate of 46.7% at the 60% level. In contrast, the elimination of COD was 85.4% at the 50% level. The turbidity removal exhibited a positive correlation with the level rise, reaching an optimal value of 67.9% at the 70% level. However, there was a minor drop in turbidity removal from the 70%

level to the 90% level. These trends are driven by changes in organic load, bubble-particle interactions, and mixing efficiency. Higher wastewater volumes increase pollutant concentrations, reducing removal efficiency, while optimal bubble contact sufficient mixing at intermediate levels enhance removal. COD and colour removal declined with increasing wastewater volume due to higher organic loads, while removal initially improved from better bubble-particle contact but dropped slightly at the highest volumes as mixing and contact efficiency decreased.

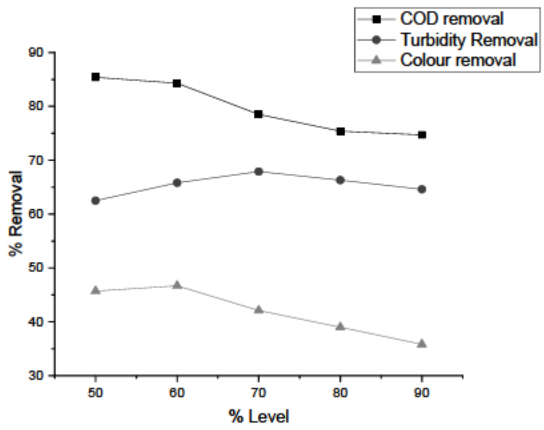


Figure 4-8 The effects of wastewater level on percentage removal of colour, COD and Turbidity.

The initial pH of a coagulation process significantly affects the efficacy of contaminant removal. Wastewater with either alkaline or acidic pH levels has a reduced ability to adsorb pollutants. At a pH that is neither acidic nor basic, the main species present have a significant ability to adsorb contaminants (Periyasamy and Muthuchamy 2018; Vepsäläinen and Sillanpää 2020). Figure 4-9 further demonstrated the correlation between wastewater level and pH and conductivity. An observation was made that as the level increased, the conductivity also increased. Additionally, there was a slight rise in pH from the 50-60% level, followed by a fall at the 70% level, when the pH slightly increased again.

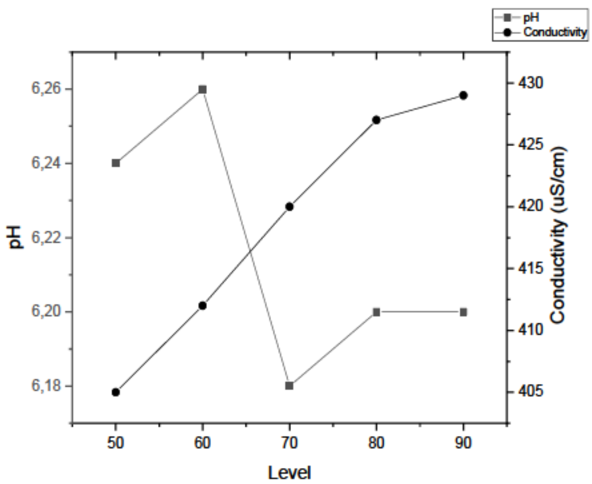


Figure 4-9 The effects of wastewater level on pH and conductivity.

4.3.1.2 Pressure variation

Figure 4-10 demonstrates the impact of pressure on the elimination of COD, turbidity, and colour using a DAF system. The COD and turbidity removal rates were both highest at the lowest pressure (200 kPa), with a percentage removal of 82.8% and 56.4%, respectively. The elimination of COD diminishes steadily with increasing pressure. However, the turbidity reduces until it reaches 400 kPa, after which it gradually increases between 400 and 600 kPa. The degree of colour removal is directly proportional to the rise in pressure, reaching its maximum removal rate of 41.9% at the greatest pressure. The treatment efficacy of the DAF system was influenced by the saturation pressure as it facilitated the size of bubbles formed. Maximum efficiency is achieved when the size of the floc exceeds the size of the bubble (Wang *et al.* 2021). The collision efficiency was maximized when the particles and bubbles had similar sizes and follow the trajectory model rather than the white-water collector model. This suggested that bigger particles result in better collision efficiency (Muñoz-Alegría, Muñoz-España and Flórez-Marulanda 2021; Wang *et al.* 2021).

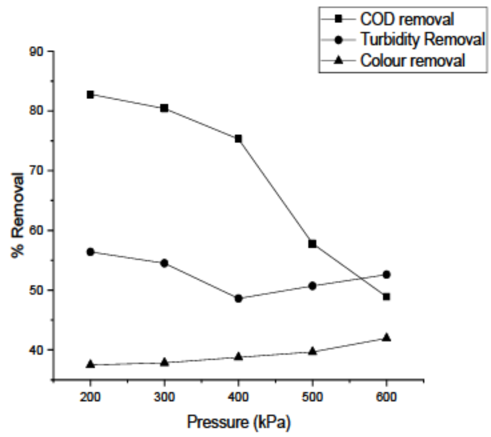


Figure 4-10 The effects of pressure on percentage removal on colour, COD and Turbidity.

Figure 4-11 demonstrates the impact of pressure on both pH and conductivity. The pH slightly drops from 6.24 at 200 kPa to 6.16 at300 kPa, and then progressively lowers from 300 to 600 kPa. Conversely, the conductivity first rises from 200-300 kPa and then steadily declines from 300-600 kPa. These trends suggest that increasing pressure enhances bubble generation in the DAF system, promoting floc-bubble interactions and improved particle removal. The initial rise in conductivity indicates release of ions from disrupted flocs at lower pressures, while the subsequent decline reflects effective removal of suspended solids and associated ionic species, resulting in improved water quality.

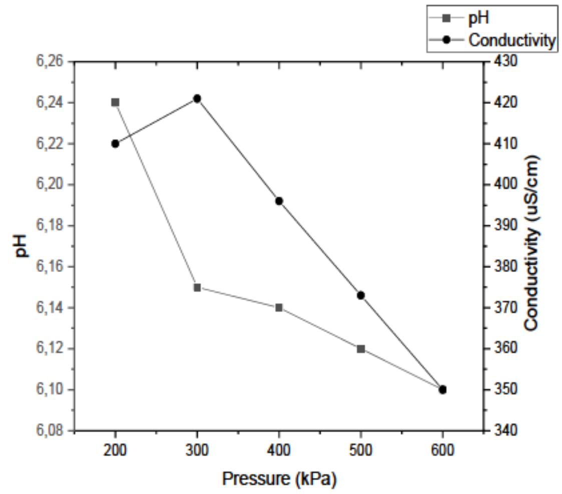


Figure 4-11 The effects of pressure on pH and conductivity.

4.3.1.3 Mixing speed variation

The agitation speed is a crucial factor in electrochemical (EC) processes as it enhances the rate of mass transfer by increasing the mobility of ions in the solution. Both the collision rate of particles and the release of metal ions and hydroxyl groups increase proportionally with the mixing velocity. As seen in Figure 4-12, the removal efficiency demonstrates an upward trend as the mixing speed escalates, reaching its peak at 250 rpm. The COD, turbidity, and colour have optimal efficiency of 77.8%, 54.3%, and 47.2%, respectively. The decline in removal beyond 250 rpm indicates the excessive agitation disrupts flocs formation, reducing particle aggregation and impairing pollutant removal. This highlights the importance of controlling agitation to balance mass transfer enhancement with floc stability.

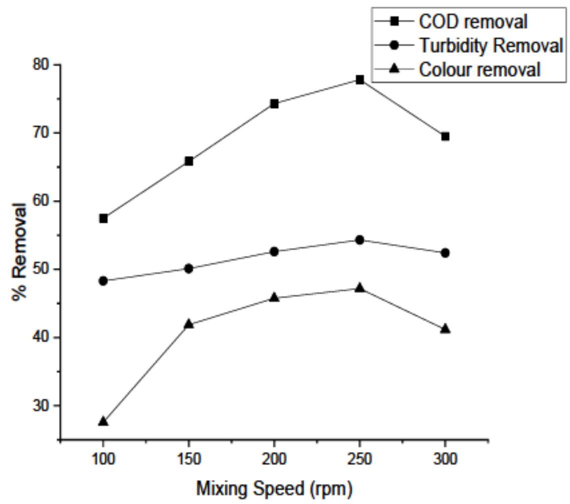


Figure 4-12 The effects of mixing speed on percentage removal on colour, COD and Turbidity.

Conductivity measurements are utilized to ascertain the most effective approach for eliminating pollutants and impurities. Conductivity is used to determine the concentration of dissolved compounds, chemicals, and elements in water. In Figure 4-13, the conductivity exhibits a progressive decline as the mixing speed rises, reaching its minimum value at 250 rpm. This trend suggests that increasing agitation improves mass transfer and promotes floc formation, enhancing the removal of dissolved ions and contaminants. Beyond 250 rpm, the conductivity stabilizes, indicating that the removal process has reached its practical limit. The ph changes reflect the balance between ion release from floc formation at low speeds and stabilisation of thee solution chemistry at higher mixing speeds.

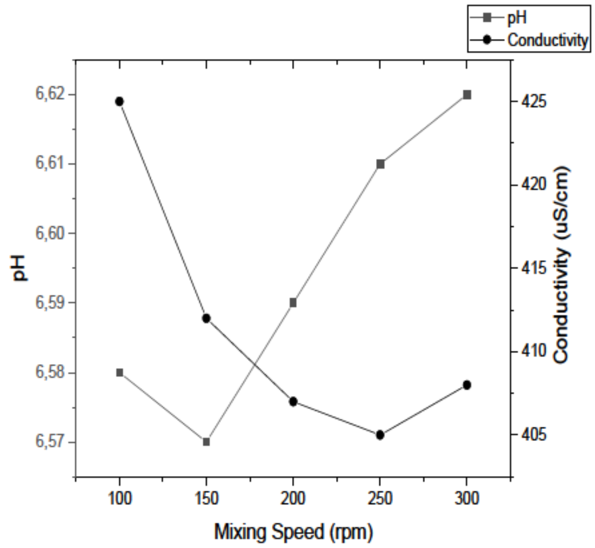


Figure 4-13 The effects of mixing speed on pH and conductivity.

4.3.2 Evaluation of key interactive factors using of DAF on a lab-scale plant using RSM

This section uses RSM to analyse the interactive effects of key operational parameters, such as air-water ratio, pressure and agitation speed on the performance of a lab-scale DAF setup.

4.3.2.1 Analysis of Variance (ANOVA)

To examine the effectiveness of dissolved air flotation as a treatment method, the following ANOVA analysis used response surface methodology with Box-Behnken Design. The statistical analysis involved determining the p-value, Lack of Fit (LOF) test, and significance of the model and terms for some response variables, including conductivity, turbidity removal, and COD removal.

The model has high significance, with a p-value below 0.0001, indicating that pressure, mixing speed, and air-water ratio all exhibit significant effects on conductivity in DAF treatment. The individual parameters of pressure A, speed B, and ratio C have highly insignificant p-values, indicating that each factor independently influences conductivity. Quadratic factors A^2 and C^2 are significant, suggesting that precise variations in pressure and ratio are crucial for attaining optimal conductivity decrease via DAF. The interactions of BC and AC exhibit exceptionally high F-values of 430.58 and 357.83, respectively, signifying that these interaction effects are crucial for reducing conductivity in the DAF process. Correspondingly, the LOF has a p-value of 0.1253, indicating that the model adequately fits the conductivity data.

The turbidity removal model is significant at p = 0.0001, even though not all the model terms were found to be significant. Thus, the model terms such as Pressure-A and Speed-B yielded p-values exceeding 0.05, signifying a minimal individual impact of these parameters on turbidity reduction. Significant interactions, such as AB^2, during turbidity reduction, indicated that a combination of pressure and velocity contributes to the optimisation of DAF treatment methods of turbidity removal. The C factor (Ratio) has an F-value of 338.30, the highest recorded, indicating a significant impact of Ratio variation on turbidity reduction. Therefore, adjusting the Ratio would substantially enhance the removal effectiveness of the DAF process. The p-value of LOF is 0.5757, indicating a strong fit, therefore suggesting that the model accurately predicts turbidity removal behaviour in the DAF process.

The factors identified in the analysis of this model, including Pressure (A), Speed (B), and Ratio (C), as well as their interactions, are essential for the substantial removal of COD. Also, the model is significant at $p = 0.0141$. Thus, elevated F-values for Pressure (A) and Speed (B) indicate that these variables provide reasonable adjustment during DAF, resulting in significantly enhanced efficiencies of organic matter removal. The F-value is highest for Pressure, indicating its significant impact on the removal of organic content in DAF, a common focus for optimisation in wastewater treatment. This model has a robust fit to the data, evidenced by a high LOF p-value of 0.8999, indicating minimal unexplained variation between model predictions and actual COD removal values.

Table 4-7 ANOVA analysis on the effects of induced voltage, mixing speed, and settling time on conductivity in dissolved air flotation.

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	1.383E+05	10	13825.96	114.35	< 0.0001	significant
A-Pressure	10658.00	1	10658.00	88.15	< 0.0001	
B-Speed	2826.69	1	2826.69	23.38	0.0029	
C-Ratio	4433.68	1	4433.68	36.67	0.0009	
AB	7396.00	1	7396.00	61.17	0.0002	
AC	43264.00	1	43264.00	357.83	< 0.0001	
BC	52060.03	1	52060.03	430.58	< 0.0001	
A^2	11749.39	1	11749.39	97.18	< 0.0001	
B^2	131.65	1	131.65	1.09	0.3369	
C^2	777.82	1	777.82	6.43	0.0443	
A^2B	196.68	1	196.68	1.63	0.2493	
Residual	725.44	6	120.91			
Lack of Fit	468.68	2	234.34	3.65	0.1253	not significant
Pure Error	256.76	4	64.19			
Cor Total	1.390E+05	16				

Table 4-8 ANOVA analysis on the effects of induced voltage, mixing speed, and settling time on turbidity removal in dissolved air flotation.

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	1434.27	10	143.43	39.09	0.0001	significant
A-Pressure	17.87	1	17.87	4.87	0.0695	
B-Speed	6.76	1	6.76	1.84	0.2234	
C-Ratio	1241.41	1	1241.41	338.30	< 0.0001	
AB	0.2500	1	0.2500	0.0681	0.8028	
AC	10.45	1	10.45	2.85	0.1424	
BC	21.17	1	21.17	5.77	0.0532	
A^2	1.78	1	1.78	0.4858	0.5119	
B^2	0.1167	1	0.1167	0.0318	0.8644	
A^2B	0.6061	1	0.6061	0.1652	0.6986	
AB^2	26.45	1	26.45	7.21	0.0363	
Residual	22.02	6	3.67			
Lack of Fit	5.31	2	2.66	0.6359	0.5757	not significant
Pure Error	16.71	4	4.18			
Cor Total	1456.29	16				

Table 4-9 ANOVA analysis on the effects of induced voltage, mixing speed, and settling time on COD removal in dissolved air flotation.

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	4632.29	11	421.12	8.53	0.0141	significant
A-Pressure	1283.30	1	1283.30	26.01	0.0038	
B-Speed	933.77	1	933.77	18.92	0.0074	
C-Ratio	322.66	1	322.66	6.54	0.0508	
AB	91.03	1	91.03	1.84	0.2325	
AC	562.71	1	562.71	11.40	0.0197	
BC	79.82	1	79.82	1.62	0.2594	
A^2	676.64	1	676.64	13.71	0.0140	
B^2	167.64	1	167.64	3.40	0.1246	
A^2B	184.24	1	184.24	3.73	0.1112	
A^2C	667.59	1	667.59	13.53	0.0143	

AB^2	662.89	1	662.89	13.43	0.0145	
Residual	246.72	5	49.34			
Lack of Fit	1.10	1	1.10	0.0180	0.8999	not significant
Pure Error	245.62	4	61.41			
Cor Total	4879.02	16				

4.3.2.2 Fit Statistics

The fit statistics for the three parameters; Electrical Conductivity, Turbidity Removal, and COD Removal; are shown in the *Table 4-10*. Electrical conductivity exhibits the largest standard deviation of 11.00, indicating greater variability in its data points compared to turbidity percentage removal and COD percentage removal, which had standard deviations of 1.92 and 7.02, respectively.

The mean values for Conductivity, Turbidity Removal, and COD Removal are 406.9, 47.48, and 68.52, respectively. The COD Removal exhibits the greatest coefficient of variation at 10.25%, indicating the biggest relative variability in relation to its mean. The turbidity percentage removal and electrical conductivity exhibit comparatively low coefficients of variation, at 4.0% and 2.7%, respectively, indicating that their data points fluctuate near their mean values.

Among all the responses, Electrical Conductivity exhibits the highest R^2 value of 0.9948, indicating an exceptionally robust model fit. The Adjusted R^2 is also high at 0.9861, while the Predicted R^2 is at 0.8622, demonstrating a robust consistency in their ability to predict. The next highest is Turbidity % Removal, which exhibits a good model fit with R^2, adjusted, and projected values of 0.9849, 0.9597, and 0.8221, respectively. The COD % Removal exhibits the lowest R^2 at 0.9494 and an Adjusted R^2 of 0.8382; however, its Predicted R^2 is somewhat superior at 0.8877, suggesting robust predictive capability.

The greatest AP for Electrical Conductivity is 33.0697, followed by Turbidity Removal at 19.7674 and COD Removal at 11.0495, all significantly exceeding the criteria (AP above 4). Therefore, each model will possess a sufficient signal for precise prediction, with Electrical Conductivity exhibiting the strongest signal.

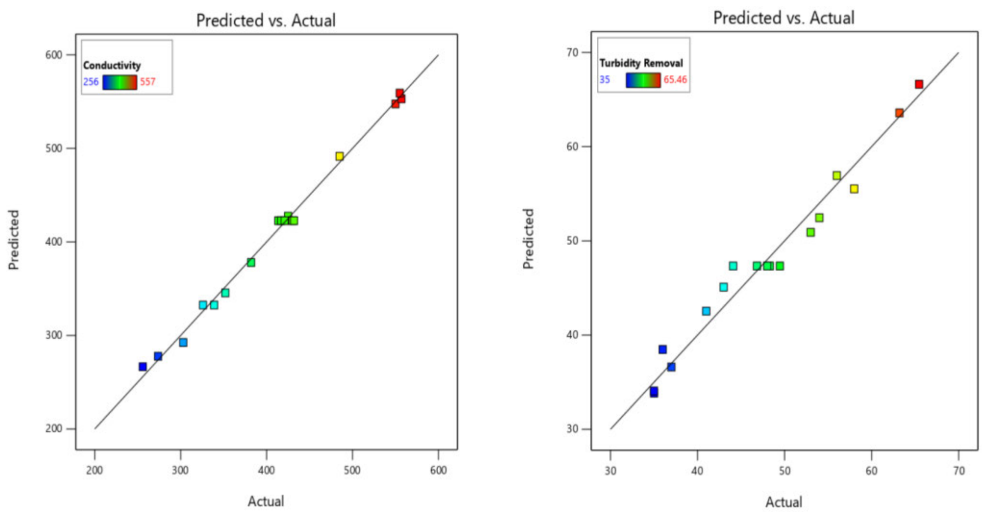
Table 4-10 Fit statistic for conductivity, turbidity removal and COD removal in DAF wastewater treatment.

Statistical parameter	Electrical conductivity	Turbidity Removal	%COD % Removal
Standard deviation	11.00	1.92	7.02
Mean	406.90	47.48	68.52
Coefficient of Variation (COV -%)	2.7	4.00	10.25
R^2	0.9948	0.9849	0.9494
Adjusted R^2	0.9861	0.9597	0.8382
Predicted R^2	0.8622	0.8221	0.8877
AP	33.0697	19.7674	11.0495

4.3.2.3 Model Validation

The following graphs indicate that the data points for electrical conductivity (Figure 4-14a), turbidity removal (Figure 4-14b), and COD removal (Figure 4-14c) are dispersed over the 45-degree line. All three regression model equations have accurately predicted the actual data points derived from the experimental study. The conductivity model had a significantly superior fit compared to the other models. The conductivity data points were nearer to the 45-degree line compared to the other responses. Since there is no apparent pattern, residuals are randomly distributed, an optimal fit is achieved.

(a) (b)



(c)

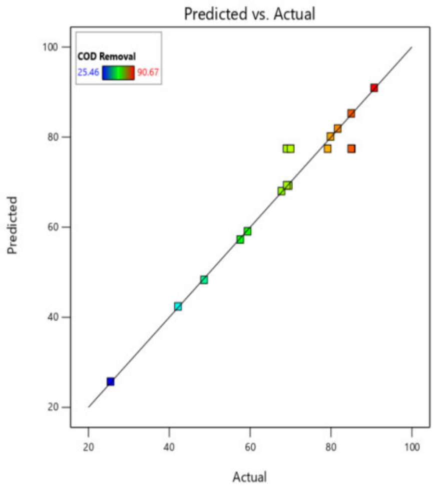
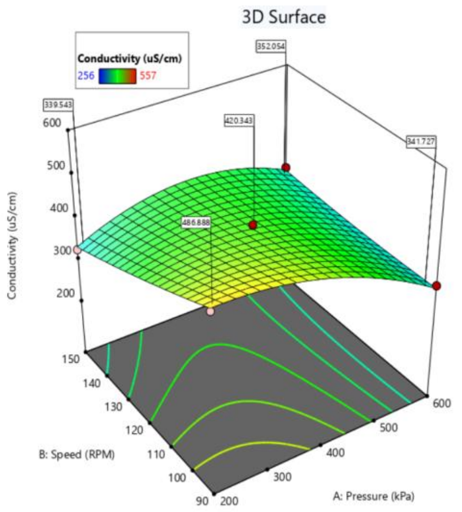


Figure 4-14 Predicted vs actual graphs for a) conductivity, b) turbidity removal, and c) COD removal through DAF treatment.

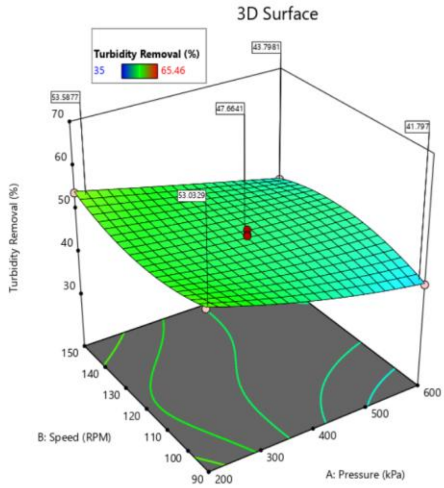
4.3.2.4 Three-dimensional surface graphs

Figure 4-15a, Figure 4-15b, and Figure 4-15c present three-dimensional surface graphs representing conductivity, turbidity removal efficiency, and chemical oxygen demand (COD) removal efficiency, respectively. The induced pressure, which ranges from 200 to 600 kPa, is displayed versus conductivity and operation speed in Figure 4-15a. The graph demonstrates a gradual decline in conductivity with increasing speed and pressure. Optimal conductivity is attained under conditions of elevated pressure and reduced mixing rate. In Figure 4-15b, the mixing velocity has minimal effect on turbidity reduction. The elimination percentage is insignificant. The adjustment of pressure has a minor effect on the turbidity reduction percentage. The percentage of turbidity reduction reduces moderately with an increase in pressure. The relationship between COD elimination and input factors is depicted in this figure (Figure 4-15c). The percentage of removed COD increases progressively with operating speed, achieving an optimum removal rate of roughly 83% at around 280 rpm, before gradually declining to approximately 64% at maximum speed and minimum pressure. A significant COD removal is also attained at maximal velocity and a pressure of around 400 kPa, resulting in a reduction of roughly 86%.

(a)



(b)



(c)

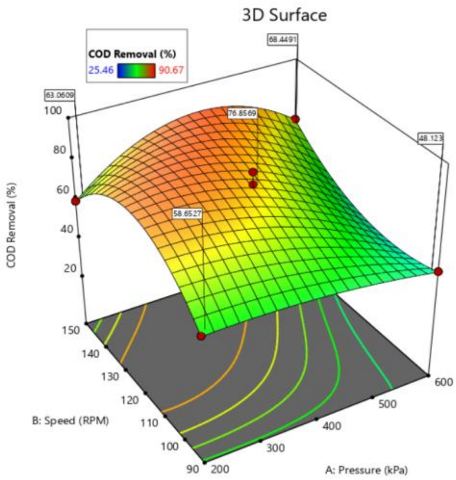


Figure 4-15 Three-dimensional (3D) response surface plots for (a) conductivity, (b) turbidity removal, and (c) COD removal through DAF.

4.3.2.5 Numerical optimisation

To achieve the most optimal results for electric conductivity, turbidity removal, and COD removal, the RSM-coupled numerical optimisation method was implemented. To maximise the performance of electrical conductivity, turbidity removal, and COD removal for each response, certain values of control variables have been attained using ramps. The following parameters were met: an air-to-water ratio of 0.43, an induced pressure of 363.11 kPa, and a mixing speed of 106 rpm. Optimal process performance through RSM means achieving electrical conductivity of 443.75 µS/cm, turbidity removal of 45.74%, and COD removal of 69.77%.

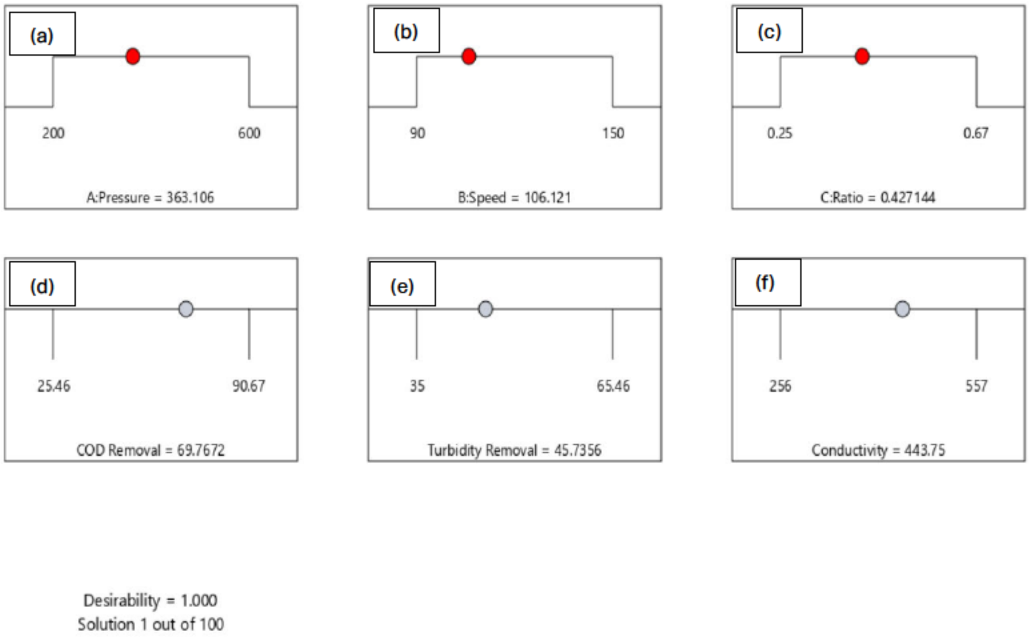


Figure 4-16: Numerical ramp plot of optimum conditions (pressure (a), running speed (b), ratio (c), COD removal (e), turbidity removal (f), and conductivity (g)) with desirability at 100% confidence level in the treatment of wastewater through DAF.

4.3.2.6 Validation of optimum conditions

The RSM model predicted a conductivity of 1255.91 µS/cm, however, experimental validation produced 1149.67 µS/cm. This indicates an 8.46% difference between predicted and experimental values, demonstrating a rather close correlation and suggesting that the RSM offers a reasonably accurate estimation of conductivity under these conditions. The RSM prediction for turbidity was 63.45%, whereas the experimental result was 62.17%, indicating a mere 2.02% discrepancy. This slight deviation illustrates significant predicted accuracy in turbidity levels, confirming the model's dependability in forecasting this parameter under the defined conditions. The COD forecast by RSM was 7.15%, while the empirically measured value was 6.96%, yielding a discrepancy of 2.62%. This close correlation suggests that the RSM is good in predicting COD levels in the optimised conditions. The minimal percentage variations across all parameters, particularly for turbidity and COD, demonstrate that the RSM model effectively reflects the experimental results, hence confirming the applicability of RSM for optimising these conditions. These findings emphasise the relevance of RSM in predicting treatment processes.

Table 4-11 Validation of optimum conditions on conductivity, turbidity removal, and COD removal in DAF treatment.

Response	RSM predicted results	Experimental results	Difference in results (%)
Conductivity (µS/cm)	1255.91	1149.67	8.46
Turbidity (%)	63.45	62.17	2.02
COD (%)	7.15	6.96	2.62

4.4 Wastewater Treatment Through Slow Sand Filtration

This section examines SSF for wastewater treatment. The OFAT method assessed the effects of HRT and flow variation, while RSM explored their interactive effects. The results offer insights for optimising SSF.

4.4.1 OFAT Method in Wastewater Treatment Using SSF

The OFAT method was used to assess the impact of individual parameters on slow sand filtration. This approach helped identify optimal conditions for effective treatment.

4.4.1.1 Flow variation

The efficacy of slow sand filters has been proven by the reduction of dissolved or suspended solids. This reduction led to a decrease in turbidity and colour, as seen in Figure 4-17. The optimal removal efficiency for both turbidity and colour occurs at the lowest flow rate, with values of 69.5% and 65.3%, respectively. The nature and content of turbidity differ from place to place, and as it is an optical feature of a suspension, factors like particle size, shape, and experimental characteristics can affect how it is measured (Verma, Daverey and Sharma 2019). Conversely, the efficacy of COD elimination is exceedingly poor, yet it reaches its peak at the most minimal flowrate, functioning at 12.5%.

Increasing the pace at which fluid is passed through the filter results in a reduction in the filter's efficiency. The reason for this is that low filtration rates are essential for retaining particles that have settled on the medium surface due to gravity. In order to optimize the collection of suspended particles by the SSF, it is advisable to maintain a low filtration rate (Maiyo, Dasika and Jafvert 2023). The pace at which filtration occurs is primarily influenced by the filter type, the properties of the water, and the desired level of turbidity reduction. The primary determinant of turbidity removal effectiveness is the particle sizes and distribution of the raw water (Verma, Daverey and Sharma 2019; Maiyo, Dasika and Jafvert 2023).

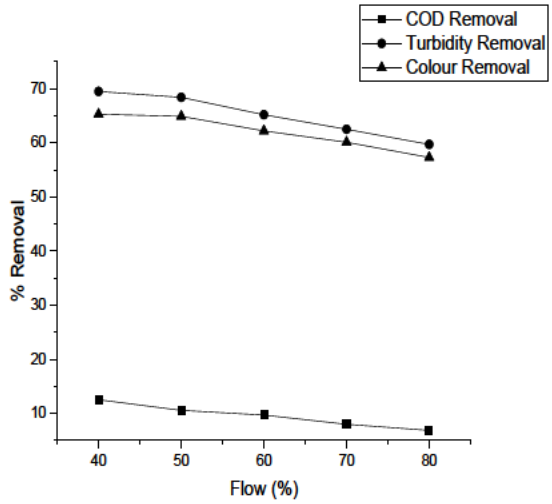


Figure 4-17 The effects of flow on percentage removal on colour, COD and Turbidity.

Electric current passing through a solution is measured by its conductivity. This signifies the level of electrolyte ions that are dissolved in water (Amankwah *et al.* 2025). Nevertheless, substantial reductions in conductivity also suggest that contaminants may have been eliminated from the water. Increasing the flowrate results in a reduction in conductivity, as seen in Figure 4-18. This phenomenon occurs because of a decrease of particles within the filter, leading to a decrease in conductivity.

pH is a measure of the acidity or alkalinity of a solution, reflecting the concentration of hydrogen ions (H^+) resulting from the dissociation of water molecules (Amankwah *et al.* 2025). Additional chemicals that could have been introduced into the water will undergo a chemical reaction with these ions, resulting in an uneven distribution of hydrogen. Figure 4-18 demonstrates the inverse relationship between pH and flowrate. The conductivity and pH levels reach their lowest point when the flow is at 70%.

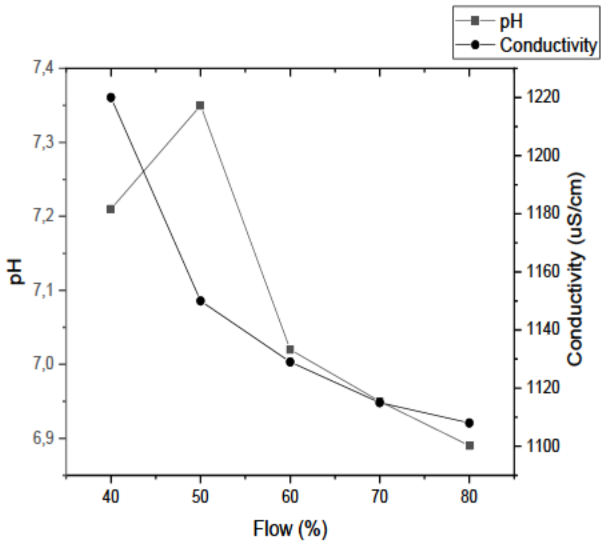


Figure 4-18 The effects of flow on pH and conductivity.

4.4.1.2 HRT variation

The Hydraulic retention time (HRT) is a metric that quantifies the average duration during which a soluble chemical persists within an integrated bioreactor (Maiyo, Dasika and Jafvert 2023; Pérez-Rodríguez *et al.* 2025). The removal efficiency % rises proportionally with an increase in hydraulic retention time (HRT). The removal efficiency reaches its maximum at the maximum time (50 min) for all pollutants, as shown in Figure 4-19. The levels of COD, turbidity, and colour removal are 10.1, 71.4, and 65%, respectively.

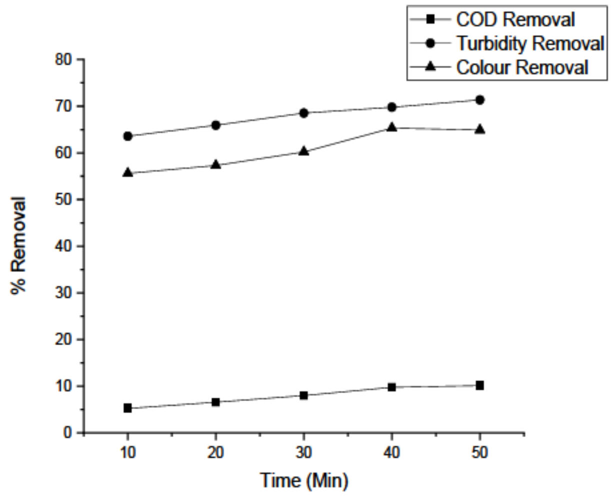


Figure 4-19 The effects of HRT on percentage removal of colour, COD and Turbidity.

The pH and conductivity exhibit an initial increase from 10 minutes to 20 minutes, followed by a progressive decline until 40 minutes. Between 40 and 50 minutes, there is a small rise in both conductivity and pH. The decline in conductivity demonstrates the decrease in the concentration of electrolyte ions that are dissolved in water (Khudair and Jasim 2018).

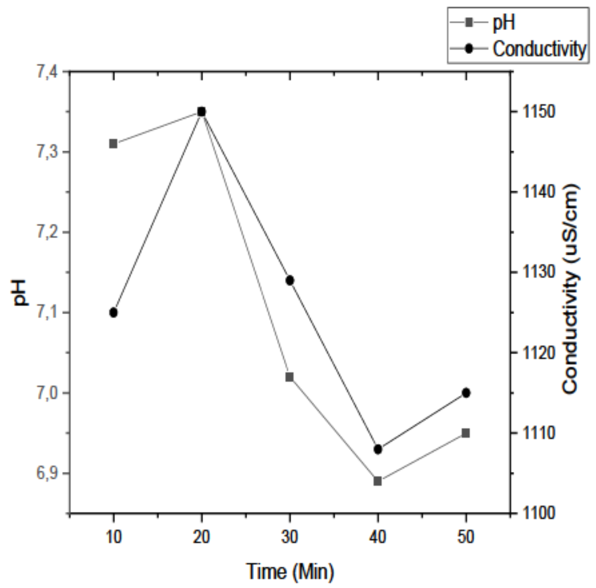


Figure 4-20 The effects of HRT on pH and conductivity.

4.4.2 Evaluation of key interactive factors using of SSF on a lab-scale plant using RSM

This section uses RSM to analyse the interactive effects of key operating parameters, on the performance of lab-scale SSF.

4.4.2.1 Analysis of Variance (ANOVA)

The high F-values and low p-values for most of the variables indicate that the model is effective when Conductivity is the response variable (Table 4-12). The Model F-value of 86.81 is highly significant, reflecting the model's overall significance, whereas the major effect of Flow (A) yields an F-value of 226.06. This indicates that variations in flow significantly influence conductivity. The elevated F-values of 55.66, 86.96, 10.64, and 60.19 correspond to the interaction variables HRT (B), AB, A^2, and B^2, respectively. This indicates that the exponents exert a highly significant individual and interaction influence on conductivity. The complete model's p-value is below 0.0001, indicating its statistical significance in completeness. All terms, except for residual error, exhibit a p-value less than 0.05, signifying robust evidence of the significance of flow, hydraulic retention time (HRT), their interaction, and quadratic terms as substantial factors influencing conductivity. The lack-of-fit p-value for Conductivity is 0.0022, signifying a significant lack of fit.

The analysis of Turbidity Removal, as illustrated in Table 4-13, reveals significantly high F-values and significantly low p-values. This signifies an excellent design that has accounted for a substantial share of the variations. The model's F-value for Turbidity Removal appears very high, at 1386.21. A high F-value suggests that the model's predictive ability is strong, and the factors included in the model are highly influential in explaining turbidity removal (Perez *et al.* 2017). The primary effects, Flow (A) and HRT (B), exhibit F-values of 177.50 and 511.29, respectively. The interaction term AB yields an F-value of 616.89, but the quadratic terms A^2 and B^2 produce F-values of 5435.69 and 2241.73, respectively. This evidence indicates that the components and their interactions are crucial for turbidity reduction. All terms, except AB^2, have yielded an exceedingly small p-value (< 0.0001), indicating a highly significant statistical level. The one exception, AB^2, exceeds the 0.05 threshold with a p-value of 0.2106, indicating that this specific interaction has minimal impact on turbidity reduction. The lack of fit p-value for Turbidity Removal is 0.0004, indicating statistical significance. This indicates that while the model is highly predictive, it does not fully account for all variability in turbidity removal; additional features or a more complex model may enhance the fit.

The model F-value and associated p-value for COD removal are moderate, suggesting that parameters selectively influence COD removal efficiency. Refer to Table 4-14. The model F-value is 15.86, noteworthy but considerably lower than the F-value determined for conductivity and turbidity removal. In terms of F-values, HRT (B) is predominant at 47.13, followed by A^2 at 19.89 and B^2 at 19.26. The interaction term AB exhibits an F-value of 0.1425, indicating it does not significantly contribute to COD removal. The model's p-value is 0.0039, which falls under the 5% significance level; hence, the model is deemed valid. Nonetheless, the Flow term A possesses a p-value of 0.1148, indicating that it is not a significant individual contribution to COD elimination. Terms including B, A^2, B^2, and A^2B exhibit p values <0.05, indicating that these parameters significantly affect the elimination of COD. Although interaction AB's p value is significantly larger at 0.7213, it has no bearing on the mentioned response. The p-value for the lack of fit indicates a non-significant lack of fit for COD Removal at 0.6486. The model's lack of significance suggested that it was well-suited for prediction within the experimental range and well represented the variations without added complexity.

Table 4-12 ANOVA analysis on the effects of HRT and flow variation on conductivity through slow sand filter wastewater treatment.

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	18067.97	5	3613.59	86.81	< 0.0001	significant
A-Flow	9410.53	1	9410.53	226.06	< 0.0001	
B-HRT	2317.25	1	2317.25	55.66	0.0001	
AB	3620.03	1	3620.03	86.96	< 0.0001	
A^2	442.90	1	442.90	10.64	0.0138	
B^2	2505.80	1	2505.80	60.19	0.0001	
Residual	291.40	7	41.63			
Lack of Fit	281.40	3	93.80	37.52	0.0022	
Pure Error	10.00	4	2.50			
Cor Total	18359.37	12				

Table 4-13 ANOVA analysis on the effects of HRT and flow variation on turbidity removal through slow sand filter wastewater treatment.

Source	Sum of Squares	df	Mean Square	F-value	p-value	
Model	2221.66	7	317.38	1386.21	< 0.0001	significant
A-Flow	40.64	1	40.64	177.50	< 0.0001	
B-HRT	117.06	1	117.06	511.29	< 0.0001	
AB	141.24	1	141.24	616.89	< 0.0001	
A^2	1244.53	1	1244.53	5435.69	< 0.0001	
B^2	513.25	1	513.25	2241.73	< 0.0001	
A^2B	24.48	1	24.48	106.90	0.0001	
AB^2	0.4719	1	0.4719	2.06	0.2106	
Residual	1.14	5	0.2290			
Lack of Fit	1.11	1	1.11	114.88	0.0004	
Pure Error	0.0385	4	0.0096			
Cor Total	2222.80	12				

Table 4-14 ANOVA analysis on the effects of HRT and flow variation on COD removal through slow sand filter wastewater treatment.

Source	**Sum of Squares**	**df**	**Mean Square**	**F-value**	**p-value**	
Model	6.70	7	0.9570	15.86	0.0039	significant
A-Flow	0.2194	1	0.2194	3.64	0.1148	
B-HRT	2.84	1	2.84	47.13	0.0010	
AB	0.0086	1	0.0086	0.1425	0.7213	
A^2	1.20	1	1.20	19.89	0.0066	
B^2	1.16	1	1.16	19.26	0.0071	
A^2B	4.11	1	4.11	68.04	0.0004	
AB^2	0.3635	1	0.3635	6.02	0.0576	
Residual	0.3017	5	0.0603			
Lack of Fit	0.0172	1	0.0172	0.2419	0.6486	not significant
Pure Error	0.2845	4	0.0711			
Cor Total	7.00	12				

4.4.2.2 Fit Statistics

The standard deviation of electrical conductivity is 6.45, indicating a moderate level of variability; this signifies that conductivity values range from the mean or expected value by approximately 6.45 units, exhibiting an acceptable level of precision. With a low standard deviation of 0.4785, the % Removal of Turbidity indicates that there is not much of a deviation from the pattern that the model captured. COD Percentage Reduction: The standard deviation was a relatively small value of 0.2456, indicating that the COD values closely correspond to the value that this model predicts. The average conductivity is 1260.41, indicating a significantly elevated level of conductivity across the completed studies. The filter exhibits a mean efficiency of 49.92%, effectively reducing approximately half of the turbidity, with potential for improvement. Nevertheless, the average COD removal is merely 6.72%, significantly lower than the turbidity removal under same experimental conditions. This indicates that the conditions for facilitating COD removal will be challenging.

The latter exhibits a coefficient of variation of 0.5119%, signifying exceptional precision, as the variation constitutes a negligible fraction of the mean in Electrical Conductivity. The turbidity removal rate has a coefficient of variation (COV) of 0.9586%, demonstrating exceptional precision and suggesting minimal relative variability around the mean. The COD model yields a COV of 3.66%, surpassing the COVs associated with the other responses. This indicates that the model for COD removal is highly accurate, although its consistency is somewhat inferior to that of the models for conductivity and turbidity removal.

R^2 score of 0.9841 indicates that about 98.4% of the variability in conductivity is accounted for by this model, demonstrating a strong fit. The R^2 score for turbidity removal is 0.9995, indicating that the model accounts for nearly all the variance and demonstrates a strong match. It can be seen from the R^2 value of 0.9569 that 95.7% of the variation in COD removal can be explained by the model. This means that the model does provide a good fit, even if it is a little lower than other responses.

The model has a decent distribution on Electrical Conductivity without overfitting, with an R^2 of 0.9728 which is close to the adjusted R^2. The model for turbidity had a strong fit, with an adjusted R^2 of 0.9988, showing minimal adjustment for additional terms. The adjusted R^2 of 0.8966 for COD is a significant decrease from R^2, which would suggest that some of the model's variables have negligible contributions. This may suggest that additional simplification or refinement is required for COD.

Despite being lower than the actual R^2 and the adjusted R^2, the predicted R^2 of 0.8902 still shows that conductivity can be predicted well. Conversely, a predicted R^2 of 0.9681 indicates that the model possesses strong predictive potential for additional data points, hence demonstrating great robustness in turbidity reduction predictions. The predicted R^2 of 0.7793 is reasonable, however, much lower than the adjusted R^2. This would then imply a lesser predictive accuracy of the model in the case of COD removal, probably due to some hidden variability or factors missing in the model.

Adequate precision is a measure of signal-to-noise ratio; a ratio exceeding 4 is typically preferred and signifies adequate model precision. The AP values of 32.1457, 90.8819, and 12.3757 indicate that all models provide a robust signal for predicting conductivity, turbidity removal, and COD removal, respectively. The AP for turbidity removal indicates that this model is extraordinarily robust with a high signal-to-noise ratio, making it highly dependable for predicting turbidity removal. Furthermore, although COD removal surpasses the threshold, it remains relatively lower than other responses, indicating that the model developed is adequate for COD prediction but might enhance its predictive capacity with further development.

These fit statistics indicate exceptional precision, predictive capacity, and model fitness for Turbidity Removal and Electrical Conductivity. This compares with the COD Removal model, which, while sufficient, demonstrates decreased consistency and predictive accuracy, seen by lower adjusted and predicted R^2-values. The COD model requires further refining and potentially new variables to enhance its robustness and forecast reliability.

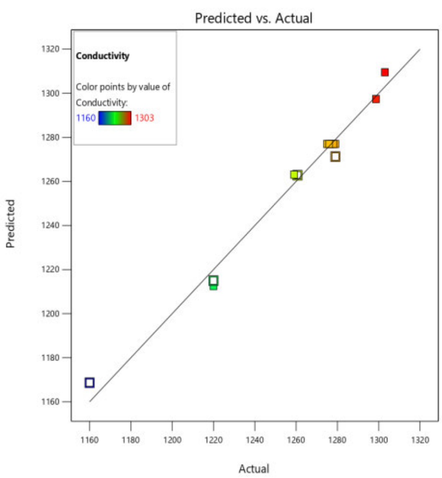
Table 4-15 Fit statistic for conductivity, turbidity removal and COD removal in SSF treatment.

Statistical parameter	Electrical conductivity	Turbidity Removal	% COD % Removal
Standard deviation	6.45	0.4785	0.2456
Mean	1260.41	49.92	6.72
Coefficient of Variation (COV -%)	0.5119	0.9586	3.66
R^2	0.9841	0.9995	0.9569
Adjusted R^2	0.9728	0.9988	0.8966
Predicted R^2	0.8902	0.9681	0.7793
AP	32.1457	90.8819	12.3757

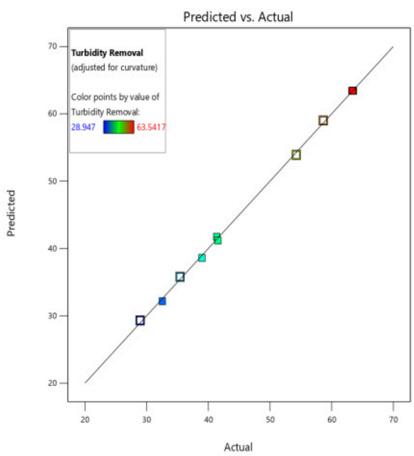
4.4.2.3 Model Validation

The description suggests that regression models for electrical conductivity, turbidity removal, and COD removal typically exhibit a strong correlation with the experimental data, as evidenced by the alignment of data points along the 45-degree line in each graph. This indicates that the model's predicted values closely align with the actual observed values. However, residuals that are dispersed randomly and show no obvious trend attest to the model's ability to accurately represent the data without systematic discrepancies. If a pattern exists in the residuals, it suggests that the model may require refinement to accurately represent the data's behaviour. Based on model performance, the turbidity removal model has proven to be the most effective of these three models. Proximity of data points to the 45-degree line denotes greater data fit; hence, this suggests that the turbidity model will more accurately reflect the experimental results than the conductivity and COD removal models.

(a)



(b)



(c)

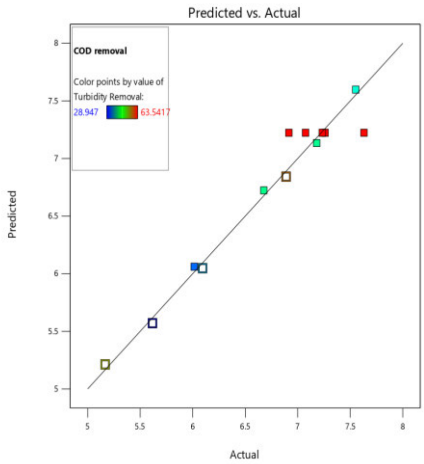


Figure 4-21 Predicted vs actual graphs for a) conductivity, b) turbidity removal, and c) COD removal through SSF treatment.

4.4.2.4 Three-dimensional surface graphs

Figure 4-22a, Figure 4-22b, and Figure 4-22c present three-dimensional surface graphs representing conductivity, turbidity removal efficiency, and chemical oxygen demand (COD) removal efficiency, respectively. Figure 4-22a depicts the flow between 40-90% plotted versus hydraulic retention time (10-40 min) and conductivity. The graph demonstrates that conductivity rises with an increase in hydraulic retention time (HRT) and decreases with increased flow. At high flow and high HRT, conductivity is at its lowest. At elevated hydraulic retention time and minimal flow, the conductivity is maximised. Figure 4-22b illustrates a three-dimensional graph depicting a concave downward surface like an inverted bowl. The maximum turbidity removal occurs at the centre of the graph, decreasing as one moves outward along both the x and y axes. Figure 4-22c illustrates the correlation between input variables and COD removal. This figure depicts a "wave-like" three-dimensional surface. This usually signifies a complex, nonlinear association between the independent factors and their responses (Ebba, Asaithambi and Alemayehu 2022). This indicates that the reactions do not merely rise or fall consistently with the factors but rather exhibit fluctuations with peaks and troughs. The maximum COD elimination occurs at a flow rate of 70-80% and an HRT of approximately 28 minutes.

(a) (b)

(c)

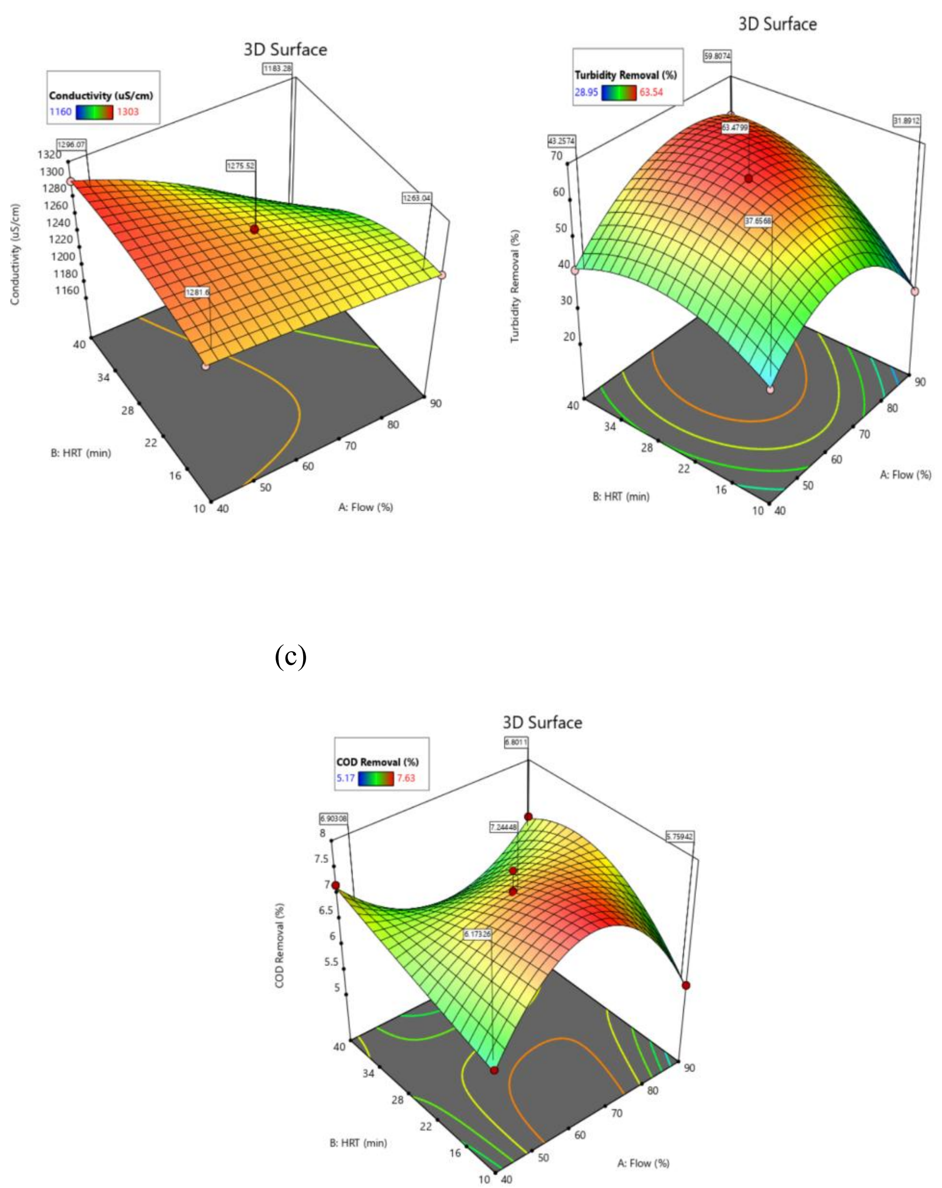


Figure 4-22: Three-dimensional (3D) response surface plots for (a) conductivity, (b) turbidity removal, and (c) COD removal through SSF treatment.

4.4.2.5 Numerical optimization

The RSM model's experimental validation entailed a comparison of the predicted and actual values of critical response variables, including conductivity, turbidity, and COD, under the optimal conditions as determined by the model. The optimum settings for the input variables were a flow rate of 76% and a hydraulic retention time of 27 minutes. The predicted values for conductivity (1255.91 µS/cm), turbidity (63.45%), and COD (7.15%) were determined.

During the actual experiment under the mentioned conditions, the conductivity was measured at 1149.76 µS/cm, the turbidity was 62.17% removed, and the COD was 6.96% removed. The percentage variances between the predicted and actual values were calculated to evaluate the model's accuracy, indicating slight variations: 8.46% for conductivity, 2.02% for turbidity, and 2.62% for COD.

The slight percentage deviations indicate that the RSM model has effectively anticipated the system's behaviour under the examined conditions. The strong correlation between the anticipated and experimental results validates the model's dependability for optimising process parameters. However, modest deviations, especially in COD, may indicate the need for further modification of either the model or the experimental setup to get a higher level of precision.

The experimental findings confirm the reliability of the RSM model in forecasting optimal circumstances for the designated response variables. The minimum percentage of mistakes demonstrates the model's effectiveness in guiding experimental conditions to attain desired results with high precision.

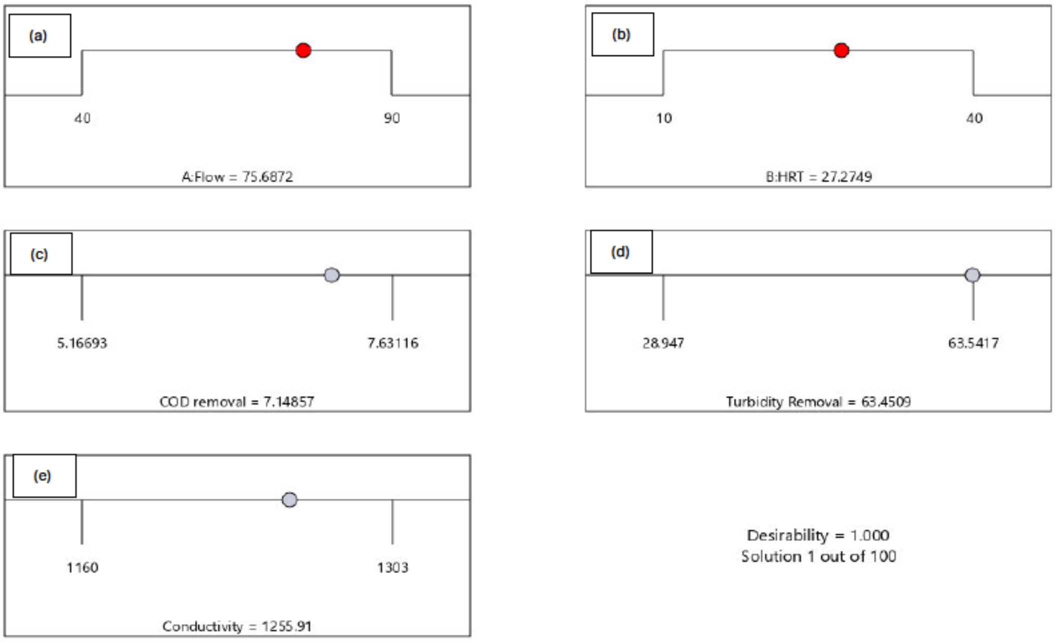


Figure 4-23: Numerical ramp plot of optimum conditions (Flow (a), HRT (b), COD removal (c), turbidity removal (d), and conductivity (e)) with desirability at 100% confidence level in the treatment of wastewater through SSF

4.4.2.6 Validation of optimum conditions

The RSM model predicted a conductivity of 1255.91 µS/cm, however experimental validation produced 1149.67 µS/cm. This indicates an 8.46% difference between predicted and experimental values, demonstrating a rather close correlation and suggesting that the RSM offers a reasonably accurate estimation of conductivity under these conditions. The RSM prediction for turbidity was 63.45%, whereas the experimental result was 62.17%, indicating a mere 2.02% discrepancy. This slight deviation illustrates significant predicted accuracy in turbidity levels, confirming the model's dependability in forecasting this parameter under the defined conditions. The COD forecast by RSM was 7.15%, while the empirically measured value was 6.96%, yielding a discrepancy of 2.62%. This close correlation suggests that the RSM is good in predicting COD levels in the optimised conditions. The minimal percentage variations across all parameters, particularly for turbidity and COD, demonstrate that the RSM model effectively reflects the experimental results, hence confirming the applicability of RSM for optimising these conditions. These findings emphasise the relevance of RSM in predicting treatment processes.

Table 4-16 Validation of optimum conditions on conductivity, turbidity removal, and COD removal in SSF treatment.

Response	RSM predicted results	Experimental results	Difference in results (%)
Conductivity (µS/cm)	1255.91	1149.67	8.46
Turbidity (%)	63.45	62.17	2.02
COD (%)	7.15	6.96	2.62

4.5 Optimisation of Treatment Technologies

Optimisation of the treatment technologies, namely, electrocoagulation (EC), Dissolved Air Flotation (DAF), and Slow Sand Filtration (SSF) was done to analyse the efficacy of each technique in wastewater treatment from the diverse sources within the bulk market. The optimisation considered varying operational conditions that were obtained through RSM, to achieve optimal treatment efficiency and ensure adherence to applicable water quality standards.

4.5.1 Optimisation of The Trader's Hall Wastewater Treatment Using EC, DAF and SSF

The table summarises water quality results from the Trader's Hall section in Clairwood Bulk Market, comparing raw water parameters with treated water using EC, DAF and SSF. The results are assessed against South African Standards.

Table 4-17 Comparison of water quality from the raw water and treated water through EC, DAF and SSF treatment methods from the Trader's Hall section.

Parameter	Unit	General Limits	Raw water	Treatment Technology		
				EC	DAF	SSF
pH (at 25°C)	pH	5.5 -≤ 9.5	6.94	7.32	7.14	7.01
Conductivity	mS/m	70-150	511.61	500	368	27.8
Temperature	^{0}C	25.00	23.10	23.5	24.1	25.5
TDS	mg/L	≤ 5	34.08	341	250	191
TSS	mg/L	25.00	566.67	5.74	3.53	1.94
Turbidity	NTU	5.00	18.24	17.00	4.36	7.22
Colour	Pt-Co	15.00	97.47	88.00	38.00	30.00
COD	mg/L	75.00	8.33	2.00	0.00	5.00
Ammonia	mg/L	15.00	15.44	14.22	13.30	6.31
Nitrate	mg/L	15.00	3.67	0.03	0.08	1.98
Nitrite	mg/L	15.00	3.48	0.02	0.05	1.77
Ortho-Phosphate	mg/L	10.00	47.25	0.22	2.88	1.20
Faecal Coliforms	CFU/100mL	1000	250.00	0.00	0.00	0.00
Fungi	CFU/100mL	1000	0.00	0.00	0.00	0.00

The results presented here are for water quality were collected from the Traders Hall section of the Bulk Market. Samples generated from showers and washing of fruit and vegetable. These results were evaluated against South African regulatory frameworks, including National Water Act (NWA), SANS 241: Drinking Water Standards, and National Environmental Management Act (NEMA), which collectively govern water use and environmental sustainability and public health standards.

The conductivity was 511.61 mS/m, which is above the limit of 70-150 mS/m given by SANS 241. Among the three methods, SSF reduced it to the largest extent to 27.8 mS/m, fully within the standard, while EC and DAF reduced it to 500 mS/m and 368 mS/m, respectively, still above the acceptable range. All three technologies were effective in reducing TSS, meeting the general limit of less than 25 mg/L. SSF had the best performance with the lower residual TSS, 1.94 mg/L, followed by DAF, 3.53 mg/L, and lastly EC, 5.74 mg/L.

In turbidity, DAF was the most effective, reducing turbidity to 4.36 NTU, below the limit of 5 NTU. In contrast, the EC had a turbidity of 17.70 NTU, while SSF was able to reduce it to only 7.22 NTU. For colour, both SSF and DAF significantly improved the raw water colour but still had residual values of 30 Pt-Co and 38 Pt-Co above the limit of 15 Pt-Co. On the other hand, colour was 88 Pt-Co with the use of EC.

All methods lowered the COD levels well below the general limit of 75 mg/L, with the best performance given by DAF at 0.00 mg/L. Similarly, EC performed very well in nutrient removal as manifested in the nitrate, nitrite, and orthophosphate reductions to almost or below the allowable limits. However, ammonia concentrations were alarming but still below the permissible limit of 15 mg/L for EC (14.22 mg/L) and DAF (13.30 mg/L), with only SSF showing significant improvement at 6.31 mg/L.

All the methods resulted in complete removal of faecal coliforms and fungi, hence ensuring microbial safety according to the SANS 241. The SSF gave the best in conductivity and TSS removal and therefore is suitable in water reuse applications. DAF is an ideal choice for turbidity reduction. Nutrient removal and COD are most appropriately done with EC.

4.5.2 Optimisation of The Distribution Centre Wastewater Treatment Using EC, DAF and SSF

Table 4-8 shows the results of the water quality analysis for the Clairwood Bulk Market distribution centre, focusing on the wastewater produced by the ablution systems and process water from the chiller and boiler units linked to the cooling towers. These results are compared against regulatory limits set by the National Water Act (NWA), South African National Standards (SANS), and the Water Management Act. It gives information on a series of parameters concerning water quality (turbidity, total suspended solids, chemical oxygen demand, ammonia levels, microbial contamination) as a characteristic of raw water quality and effectiveness in dealing with problems using different technologies of water treatment, such as electrocoagulation, dissolved air flotation, and slow sand filtration.

Table 4-18 Water quality analysis results from the Distribution Centre within the market using EC, DAF and SSF compared to regulatory limits.

Parameter	Unit	General Limits	Raw water	Treatment Technology		
				EC	DAF	SSF
pH (at 25°C)	pH	5.5 - ≤ 9.5	6.91	7.49	7.1	6.89
Conductivity	mS/m	70-150	1316.72	1043	751	930
Temperature	^{0}C	25.00	23.26	24.5	24.1	25.5
TDS	mg/L	≤ 5	845.89	708	510	635
TSS	mg/L	25.00	2583.30	7.23	4.90	3.60
Turbidity	NTU	5.00	942.28	113.00	159.00	16.50
Colour	Pt-Co	15.00	1326.11	625.00	395.00	127.00
COD	mg/L	75.00	464.56	198.00	231.00	307.00
Ammonia	mg/L	15.00	65.37	19.42	33.11	28.99
Nitrate	mg/L	15.00	9.85	0.10	0.04	3.04
Nitrite	mg/L	15.00	8.91	0.06	0.02	2.33
Ortho-Phosphate	mg/L	10.00	101.03	6.14	7.17	5.08

Faecal Coliforms	CFU/100mL	1000	1000000	100	1000	10000
Fungi	CFU/100mL	1000	0.00	0.00	0.00	0.00

Table 4-18 results of water quality from the distribution centre of Clairwood Bulk Market, which was sourced from ablution systems and process water from chiller and boiler units, have various exceedances against regulatory standards as expected characteristics for this type of wastewater. Turbidity was 942.28 NTU, while the limit value was 5 NTU, and TSS was 2583.30 mg/L against the limit of 25 mg/L. The COD was 464.56 mg/L, well over the 75 mg/L standard, indicating a high organic load. Ammonia levels were also elevated at 65.37 mg/L, significantly above the 15 mg/l limit. Microbial contamination was severe, with faecal coliforms reaching 1 000 000 CFU/100 mL, much higher than the 1 000 CFU/100 mL limit.

Electrocoagulation, dissolved air flotation, and slow sand filtration, resulted in different levels of treatment, reducing turbidity, TSS, and microbial contamination. Turbidity was reduced to 113 NTU and TSS to 7.23 mg/L with electrocoagulation. DAF and SSF improved the effluent turbidity to 159 NTU and 16.50 NTU, respectively. The values of COD and ammonia were still above the acceptable limits, which means that other treatment processes are required for full compliance with the National Water Act and SANS standards. These results emphasize the need for ongoing optimization of treatment technologies to meet regulatory requirements and ensure environmental protection.

4.5.3 Optimisation of The Final Effluent Wastewater Treatment Using EC, DAF and SSF

Table 4-9 presents the water quality results from the market's final effluent. The effluent was treated using EC, DAF and SSF and the results were analysed against the regulatory limits set by NWA, SANS, NEMA and Waste Management Act.

Table 4-19 Comparison of water quality parameters in the final effluent from the bulk market, treated by EC, DAF and SSF.

Parameter	Unit	General Limits	Raw water	Treatment Technology		
				EC	DAF	SSF
pH (at 25°C)	pH	5.5 - ≤ 9.5	6.92	6.96	6.98	7.02
Conductivity	mS/m	70-150	914.17	642	492	652
Temperature	^{0}C	25.00	23.18	24	24	25.3
TDS	mg/L	≤ 5	593.36	443	335	627
TSS	mg/L	25.00	1574.98	5.85	5.10	2.70
Turbidity	NTU	5.00	480.26	54.1	94.20	5.75
Colour	Pt-Co	15.00	711.79	515	390.00	39.00

COD	mg/L	75.00	236.44	137	83.00	203.00
Ammonia	mg/L	15.00	40.41	6.73	4.28	14.22
Nitrate	mg/L	15.00	6.76	0.021	0.068	4.75
Nitrite	mg/L	15.00	6.20	0.017	0.045	3.16
Ortho-Phosphate	mg/L	10.00	74.14	3.43	7.27	2.23
Faecal Coliforms	CFU/100mL	1000	500125	100	100	1000
Fungi	CFU/100mL	1000	0.00	0.00	0.00	0.00

The results from the final effluent at the Clairwood Bulk Market which is at the skid area (where waste management takes place) were evaluated to determine compliance with regulatory water quality standards. The wastewater is generated from three different sources that include the Trader's Hall (showers and vegetable washing), the Distribution Centre (ablution systems, chiller and boiler unit process water from cooling towers), and the skid area (which is used for washing skids that compact fruit and vegetable waste), were evaluated against the discharge limits prescribed by the National Water Act (NWA), relevant SANS standards, and the Water Services Act to determine its suitability for discharge or potential reuse.

Final effluent analysis disclosed some points of concern, and among these were high turbidity levels above the regulatory limit of 5 NTU; raw water turbidity showed 480.26 NTU, while treatment methods, mainly electrocoagulation, reduced this to 54.1 NTU. The TSS was also very high, with 1574.98 mg/L in raw water, reduced to 5.85 mg/L post-treatment with electrocoagulation, which is within the limit for discharge at 25 mg/L. COD, indicative of organic pollution, was high at 236.44 mg/L in the raw effluent, well over the 75 mg/L limit, though it was reduced to 137 mg/L after treatment by electrocoagulation, yet still above the threshold of acceptability.

The ammonia level was 40.41 mg/L in the raw effluent, which is over double the acceptable limit of 15 mg/L, though it was brought down to 6.73 mg/L by electrocoagulation. Other parameters like nitrates, nitrites, and ortho-phosphate were also present in higher concentrations than their limits in the raw effluent but showed significant reductions after treatment, especially with electrocoagulation and dissolved air flotation. The observed reductions in nitrite, nitrate, and ortho-phosphate are mainly due to the dominant removal mechanism of each treatment process. Electrocoagulation promotes phosphate removal through adsorption and precipitation onto the metal hydroxide flocs, while partial electrochemical reduction contributes to nitrogen species removal. Dissolved air flotation removes nutrients indirectly by separation nutrients bound suspended solids, and slow sand filtration achieves further reduction through microbial uptake and adsorption within the schmutzdecke layer. These combined mechanisms explain the significant decrease in nutrient loads after treatment.The faecal coliforms were a major indicator of microbial contamination and were very high at 500 125 CFU/100 mL in the raw effluent, well above the 1 000 CFU/100 mL limit. These have been effectively reduced to 100 CFU/100 mL using electrocoagulation.

DAF was moderately successful in bringing down turbidity, TSS, and microbial load. It reduced the turbidity to 94.2 NTU and TSS to 5.10 mg/l, which was within the limit for TSS, given to be 25 mg/l by standard. The application of DAF also resulted in lower ammonia levels of 4.28 mg/l well below the acceptable limit and brought COD to 83 mg/L. However, it was not as effective in the removal of COD as was electrocoagulation. Microbial contamination was also reduced: faecal coliforms were reduced to 100 CFU/100 mL, within the regulatory threshold.

Slow sand filtration was the weakest among the three for the final effluent in terms of treatment, even though it reduced turbidity to 5.75 NTU and TSS to 2.70 mg/L. In some instances, it did not deliver the effluent to the required limit of 25 mg/L. The COD reduction was minimal, resulting in a final effluent COD of 203 mg/L, which is still above the acceptable 75 mg/L standard. Moreover, ammonia was reduced to 14.22 mg/L, which, though an improvement, is still above the regulatory limit. However, SSF was very effective in reducing microbial contamination to bring faecal coliforms down to 1,000 CFU/100 mL, meeting the regulatory standard. Overall, though SSF indicated partial success in reducing turbidity and microbial contamination, its performance regarding TSS, COD, and ammonia concentrations eliminates it as a suitable model for full compliance on a stand-alone basis. In general, all three technologies showed their strengths and weaknesses: electrocoagulation was the most effective in reducing turbidity, TSS, and microbial contamination, though it still struggled with COD and ammonia; Dissolved Air Flotation provided moderate improvements in turbidity and TSS, while its performance on COD and ammonia was less impressive; Slow Sand Filtration showed some promise in reducing turbidity and microbial contamination but was less effective for other parameters.

4.5.4 Final Effluent Wastewater Treatment Using EC, DAF and SSF with Coagulant aid

Table 4-20 summarises the final effluent quality results after treatment with a coagulant (aluminium sulphate) dosage of 20 mg/L using electrocoagulation, dissolved air flotation and slow sand filtration compared against regulatory limits.

Table 4-20 Final effluent quality results after treatment with coagulant using EC, DAF, and SSF.

Parameter	Unit	General Limits	Raw water	Treatment Technology		
				EC	DAF	SSF
pH (at 25°C)	pH	5.5 -≤ 9.5	6.92	6.09	6.05	6.51
Conductivity	mS/m	70-150	914.17	701	532	892
Temperature	^{0}C	25.00	23.18	24.5	24	25.2
TDS	mg/L	≤ 5	593.36	477	361	627
TSS	mg/L	25.00	1574.98	6.59	5.02	2.20
Turbidity	NTU	5.00	480.26	10.01	27.1	12.5
Colour	Pt-Co	15.00	711.79	53.00	120	88
COD	mg/L	75.00	236.44	14.00	7	21.5

Ammonia	mg/L	15.00	40.41	6.32	5.37	5.14
Nitrate	mg/L	15.00	6.76	0.029	0.089	3.45
Nitrite	mg/L	15.00	6.20	0.021	0.067	2.93
Ortho-Phosphate	mg/L	10.00	74.14	0.012	0.18	4.21
Faecal Coliforms	CFU/100mL	1000	500125	0	0	100
Fungi	CFU/100mL	1000	0.00	0.00	0	0

The raw mixed wastewater effluent from Clairwood Bulk Market, after treatment with coagulant at a dosage of 20 mg/L, shows a notable improvement in several water quality parameters, although some still do not meet regulatory standards. This investigation was not part of the main research but was conducted out of curiosity to observe the coagulant's impact in water quality.

Turbidity experienced a significant drop from 480.26 NTU in the raw effluent to 10.01 NTU with electrocoagulation, which is well above the regulatory requirements of 5 NTU. This indicates that the coagulant effectively flocculated suspended solids, enhancing clarity. Likewise, TSS decreased drastically from 1574.98 mg/L to 6.59 mg/L with electrocoagulation, far below the 25 mg/L threshold.

Chemical oxygen demand, a measure of organic pollution was reduced from 236.44 mg/L in the raw effluent to 14 mg/L after aiding with a coagulant in EC treatment. This reduction indicated that the coagulant played an effective role in organic matter from the effluent. Ammonia levels, which initially were 40.41 mg/L, dropped to 6.32 mg/L, meeting the 15 mg/L limit. These results suggest the coagulant effectively aided the removal of organic matter and suspended solids.

Dissolved air flotation also showed significant improvement, with COD reduced to 7 mg/L and ammonia to 5.37 mg/L. however, it left faecal coliforms at 100 CFU/100 mL, indicating less effective microbial removal compared to EC. Slow sand filtration demonstrated a good reduction in turbidity (12 NTU) and TSS (to 2.2 mg/L), but COD (21 mg/L) and faecal coliforms (100 CFU/100 mL) were within acceptable limits.

In conclusion, the addition of a coagulant significantly enhanced the performance of all the treatment technologies, especially EC, which achieved the best results in reducing turbidity, TSS, COD and ammonia. While DAF and SSF showed improvement, they did not fully meet the regulatory standards, particularly in terms of microbial contamination. The improved performance is attributed to the coagulants ability to destabilise and aggregate colloidal and fine suspended particles, thereby enhancing floc formation and removal. In electrocoagulation, this effect is amplified by in situ coagulant generation and electrostatic attraction, leading to more effective removal of organic matter and nutrients, whereas in DAF and SSF the coagulant mainly aids physical separation processes, which are less effective for pathogen removal without additional disinfection.

CHAPTER 5 CONCLUSION AND RECOMMENDATIONS

This project aimed to develop water-conserving technology for the purification of locally produced fresh bulk market wastewater in South Africa to a usable grade. The analysed techniques comprised slow sand filtration (SSF), electrocoagulation (EC), and dissolved air flotation (DAF) for the treatment of wastewater derived from the bulk market. This chapter outlines the principal findings and proposes recommendations for subsequent study and decision-making.

The precise objectives were:

1. To analyse market effluent to determine the presence of physical, chemical, and biological contaminants.
2. Evaluate and enhance the efficiency of slow sand filter (SSF), electrocoagulation (EC), and dissolved air flotation (DAF) treatment technologies, and compare them at their optimal levels.
3. To optimise the most effective treatment technology using a response surface approach.

5.1 Conclusion

The results revealed varying strengths and limitations for each technology about the contaminants present and the objectives set for the study. Firstly, the study identified that the wastewater from the distribution centre wastewater was anaerobically decomposing, followed by the final effluent, and lastly, the wastewater from the trader's hall. This order highlighted the varying levels of contamination in the different wastewater sources. As expected, EC emerged as the most effective treatment option due to its ability to handle a wide range of contaminants efficiently. The optimum conditions for EC were achieved at an induced voltage of 29.67 V, an agitation speed of 150 rpm, and a settling time of 45.75 minutes, resulting in desired outputs of 971.21 μS/m conductivity and removal of 75% turbidity and 91% of COD at a desirability of 76.4%. This demonstrated EC's effectiveness in addressing chemical contaminants and achieving high levels of COD and turbidity removal, aligning well with the objectives to treat and reduce contaminants to acceptable levels for reuse.

SSF was effective in removing large particles and biological contaminants, whereas DAF excelled in removing suspended solids. These technologies, in combination with EC, played critical roles in the overall treatment process. By utilising EC, the study successfully demonstrated the highest contamination removal efficiency, particularly for chemical oxygen demand, turbidity, and conductivity, aligning the water quality with standards suitable for reuse.

The study estimated that up to 27% of the market's wastewater, approximately 45.06m^3/day, could be reclaimed and reused, significantly reducing dependence on municipal water supplies. This reclamation effort supports water conservation goals and contributes to the sustainability of the market's operations.

Specifically, the recycled water could be used in the skip section for washing of the waste bins or waste compactor, providing a practical example of waste reuse within the facility.

By adopting EC as the primary treatment technology, the market can ensure efficient wastewater management, promoting the reuse of treated water and providing a model for other industries facing similar challenges. This study highlights the importance of optimised treatment technologies, like EC, in resource efficiency, and sustainable water management techniques.

5.2 Recommendations

It is recommended that the Clairwood Fresh Produce Bulk Market install an electrocoagulation system to handle the market's daily wastewater volume, ensuring compliance with water reuse standards and optimising operational performance. To enhance treatment efficiency, a pre-treatment technology should be considered before EC, or a slow sand filtration system should be added post EC treatment. Additionally, EC performance can further improve by introducing a coagulant to enhance contaminant removal.

The treated water should be strategically reused within the market, including for ablution facilities, maintaining the skip area, and supplying process water for chiller units and boilers. Where necessary, additional disinfection measures should be implemented, particularly for applications involving produce washing, to ensure the treated water meets required safety standards.

To support sustainable operation, the market should develop a comprehensive water management plan, including regular quality assessments of treated effluent and a robust maintenance schedule for the EC system to prevent operational challenges such as electrode passivation. Staff training and awareness programs should be conducted to educate employees and maintenance personnel on proper system operation and the benefits of water reuse, fostering a culture of sustainability within the market.

Furthermore, integrating renewable energy sources, such as solar power or biogas derived from the market waste, to power the EC system will enhance the environmental benefits, aligning with the REFFECT AFRICA initiative and promoting sustainable water management practices.

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APPENDIX A: MONTLY WATER AND UTILITY COST

Table A 1 Monthly water and sewage utility costs of the Clairwood Bulk Market.

Date	Cumulative days	Monthly meter reading (days)	Flow meter (/d)	Monthly influent		Monthly wastewater		**Overall cost (R/kL/)**
	0	0	8757	Influent (kL/d)	Influent cost (R/kL/)	90% influent	Sewage cost (R/kL/)	
08/01/2021	34	34	12418	3661	135615.6	3294.9	33900.94	**169516.6**
04/02/2021	61	27	15999	3581	132299.6	3222.9	32930.38	**165229.9**
05/03/2021	90	29	19795	3796	140264.1	3416.4	34921.4	**175185.5**
07/04/2021	123	33	23900	4105	151774.2	3694.5	37824.26	**189598.4**
06/05/2021	152	29	27044	3144	116453	2829.6	29106.22	**145559.2**
02/06/2021	180	28	30919	3875	143092.8	3487.5	35589.29	**178682.1**
01/07/2021	207	27	35843.63	4924.63	181368.9	4432.167	44914.21	**226283.2**
06/07/2021	213	6	36937.99	1094.36	43758.21	984.924	10776.89	**54535.1**
03/08/2021	241	28	41834	4896.01	195838.7	4406.409	48260.18	**244098.9**
03/09/2021	272	31	47259	5425	216996.6	4882.5	53473.45	**270470**
06/10/2021	305	33	53125	5866	234604.5	5279.4	57799.68	**292404.2**
05/11/2021	335	30	58793	5668	226570.4	5101.2	55773.84	**282344.2**

03/12/2021	363	28	64376	5583	223077.9	5024.7	54875.89	**277953.8**
07/01/2022	398	35	71309	6933	277033.4	6239.7	68154.29	**345187.6**
04/02/2022	426	28	79071	7762	309475.2	6985.8	75859.66	**385334.9**
TOTAL		426	627380.6	70314	2728223	63282.6	674160.6	3402384
Average	212.5	26.625	39211.29	4687.6	181881.5	4218.84	44944.04	226825.6

APPENDIX B: RAW WATER SAMPLES

Table B 1 Raw wastewater data from the Trader's Hall.

Parameter	Unit	1	2	3	4	5	6	Average
pH (at 25°C)	pH	6.51	7.23	6.19	7.23	7.09	7.37	6.94
Conductivity	mS/m	501	501.67	583	502	482	500	511.61
Temperature	^{0}C	20.1	24.87	20.5	24.8	23.9	24.4	23.10
Total Dissolved Solids	mg/L	358	342	311	341	348	345	340.83
Total Suspended Solids	mg/L	478	452	645	635	601	589	566.67
Turbidity	NTU	15.2	12.13	15.83	12.5	27.5	26.3	18.24
Colour	Pt-Co	77.5	79.33	127	84	104	113	97.47
Chemical Oxygen Demand	mg/L	6	8	12	6	8	10	8.33
Ammonia (as N)	mg/L	16	15.5	5.05	18.56	17.89	19.65	15.44
Nitrate (as N)	mg/L	8.47	2.1	4.58	1.85	2.56	2.45	3.67
Nitrite (as N)	mg/L	7.5	2.15	4.43	2.01	2.47	2.32	3.48
Ortho-Phosphate (as P)	mg/L	48.5	43.9	45.8	49.6	47.2	48.5	47.25
Faecal Coliforms	CFU/100mL	100	100	1000	100	100	100	250.00
Fungi	CFU/100mL	0	0	0	0	0	0	0.00

Table B 2 Raw wastewater data from the Distribution Centre.

Parameter	Unit	1	2	3	4	5	6	Average
pH (at 25°C)	pH	6.31	7.1	6.67	7.15	7.11	7.09	6.91
Conductivity	mS/m	1749	1046.33	1458	1048	1548	1051	1316.72
Temperature	^{0}C	20.1	24.73	20.5	24.8	24.7	24.7	23.26
Total Dissolved Solids	mg/L	1230	706.33	1020	711	703	705	845.89
Total Suspended Solids	mg/L	2579.3	1905.8	3427.7	3045.8	2486.78	2054.4	2583.30
Turbidity	NTU	2940	497.67	723	484	507	502	942.28
Colour	Pt-Co	1258	1337.67	1229	1455	1329	1348	1326.11
Chemical Oxygen Demand	mg/L	358	330.33	1108	338	330	323	464.56
Ammonia (as N)	mg/L	68	53	47	78	75	71.22	65.37
Nitrate (as N)	mg/L	10.5	5.98	15.7	7.7	8.7	10.5	9.85
Nitrite (as N)	mg/L	9.8	5.89	11.6	7.54	8.65	9.98	8.91
Ortho-Phosphate (as P)	mg/L	85.23	115	95.34	98.54	102.56	109.5	101.03
Faecal Coliforms	CFU/100mL	1000000	1000000	1000000	1000000	1000000	1000000	1000000.00
Fungi	CFU/100mL	0	0	0	0	0	0	0.00

Table B 3 Raw wastewater data from the Final Effluent.

Parameter	Unit	1	2	3	4	5	6	Average
pH (at 25°C)	pH	6.73	7.02	5.48	7.32	7.54	7.29	6.90
Conductivity	mS/m	984	984	1220	1203	897	990	1046.33
Temperature	^{0}C	20.4	24.4	20.1	24.7	24.1	24.44	23.02
Total Dissolved Solids	mg/L	699	438	943	438	436	441	565.83
Total Suspended Solids	mg/L	1534.35	1758.87	979.54	1011.23	1257.63	1105.46	1274.51
Turbidity	NTU	487	583	482	485	473	476	497.67
Colour	Pt-Co	682	640.67	637	643	642	675	653.28
Chemical Oxygen Demand	mg/L	368	277	297	358	381	301	330.33
Ammonia (as N)	mg/L	24.18	27.95	28.17	31.1	24.16	12.84	24.73
Nitrate (as N)	mg/L	4.26	2.1	4.49	4.35	4.28	4.79	4.05
Nitrite (as N)	mg/L	3.87	1.98	3.89	4.02	3.95	4.23	3.66
Ortho-Phosphate (as P)	mg/L	72.1	97.5	77.5	93.5	88.7	75.3	84.10
Faecal Coliforms	CFU/100mL	1000	1000	10000	1000	10000	100000	20500.00
Fungi	CFU/100mL	0	0	0	0	0	0	0.00