TREATMENT OF WASTEWATER FROM SLAUGHTERHOUSES BY ELECTROCOAGULATION: CASE STUDY OF GACHORORO SLAUGHTERHOUSE, KIAMBU COUNTY, KENYA

LILIAN K. MULIMI

MASTER OF SCIENCE
(Environmental Legislation and Management)

JOMO KENYATTA UNIVERSITY OF
AGRICULTURE AND TECHNOLOGY

2020

Treatment of Wastewater from Slaughterhouses by Electrocoagulation: Case Study of Gachororo Slaughterhouse, Kiambu County, Kenya

Lilian K. Mulimi

A Thesis Submitted in Partial Fulfilment for the Degree of Master of Science in Environmental Legislation and Management in the Jomo Kenyatta University of Agriculture and Technology

2020

DECLARATION

This thesis is my original work and has not been presented for a degree in any other University.

Signature………………………………………….

Date…………………………

Lilian K. Mulimi

This thesis has been submitted with our approval as university Supervisors

Signature………………………………………….

Date…………………………

Prof. Patrick G. Home, PhD

JKUAT, Kenya

Signature………………………………………….

Date…………………………

Prof. Joseph Chacha, PhD

Maseno University, Kenya

Signature………………………………………….

Date…………………………

Dr Daniel O. Siringi, PhD

University of Eldoret, Kenya

DEDICATION

To my husband, Victor Abuka and my beloved parents, Mr. and Mrs. Jeremiah Mulimi. Thank you for being my pillar of support throughout this process.

ACKNOWLEDGEMENTS

First and foremost I want to thank the Almighty God for giving me the opportunity, finances and strength to advance my studies.

Special gratitude to my supervisors Professor Patrick Home, Professor Joseph Chacha and Dr. Daniel Siringi for their guidance and constructive critiques throughout this process with utmost diligence, expertise and inspiration in the process of preparing this project.

I thank my colleagues, family and friends for their continuous encouragement as well as moral and material support all through.

I would also like to thank the JKUAT fraternity for providing learning materials, laboratory equipment and the professionals required for my research project.

Last but not least, I give gratitude to the National Commission for Science, Technology and Innovation (NACOSTI) who funded this project. God bless them all.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ABBREVIATIONS, ACRONYMS AND SYMBOLS

NEMA	National Environment Management Authority, Kenya
EMCA	Environment Management and Co-ordination Act
WHO	World Health Organization
EC	Electro-coagulation
Al	Aluminium
Fe	Iron
BOD	Biological Oxygen Demand
COD	Chemical Oxygen Demand
DO	Dissolved Oxygen
DWAF	Department of Water Affairs and Forestry
FEPA	Federal Environmental Protection Agency
TSS	Total Suspended Solids
TDS	Total Dissolved Solids
EF	Electro-floatation

ABSTRACT

Wastewater from slaughterhouses is highly contaminated with high concentrations of organic materials, colloids such as proteins, cellulose, and fats, as well as suspended solids. Discharge of this water before treatment into the rivers affects the water quality and the fauna as well as microflora present in it, increases long term biological oxygen demand (BOD) and creates water treatment problems. The conventional methods including the aerobic and anaerobic biological treatment processes used to treat this water are characterized by many disadvantages including; production of high levels of sludge, high energy consumption required for aeration, sensitivity to high organic loading rates and long hydraulic retention times. It is therefore necessary to explore other alternative treatment methods such as electro-coagulation. This study therefore sought to explore slaughterhouse wastewater treatment using electro-coagulation method. The set-up included elctrocoagulation reactors using iron and aluminium electrodes of different surface areas under varying electrical voltages. pH, Electrical Conductivity, alkalinity, Chemical Oxygen Demand, Turbidity, Total Suspended Solids, and BOD of raw and treated wastewater were determined using standard methods. The results revealed that electro-coagulation method is able to reduce all these parameters in wastewater to levels that are acceptable by the Government of Kenya Standards for treated wastewater and effluent discharge into the environment. The highest removal efficiencies of 98.4% (BOD), 98.9% (COD), 53% (Total Alkalinity), 83% (conductivity), 99.9% (turbidity), and 99.4% (TSS) were obtained at voltage of 25V and surface area of 40 cm^2 with aluminum electrodes and 90 cm^2 with iron electrodes. ANOVA analyses demonstrated that electrocoagulation is very effective in treating slaughterhouse wastewaters at optimum voltage and surface area of electrode and that aluminium is the better option for electrode material. The study concludes that electrocoagulation is a very effective method for treating slaughterhouse wastewaters and that aluminium is the better electrode material. The treated water can be recovered and be re-used for activities such as cleaning and farming but requires further treatment for it to be used for drinking. From the findings, the study recommends further inquiry into the process through which electro-coagulation reduces total alkalinity of wastewater. It also recommends further studies to investigate other metals or materials that may be more effective and environmentally friendly as electrodes, and on use of solar or wind as sources of energy in the electrocoagulation process.

CHAPTER ONE

INTRODUCTION

1.1 Background of the Study

The demand for new and better technologies for treating wastewater continues to grow as the world's population grows and sources of freshwater get polluted. Developing countries continue to struggle with waterborne diseases due to the lack of appropriate knowledge and technology regarding water purification. The limited water sources are also used by industries which have to been forced to settle for using lower quality raw water as a higher proportion of the limited fresh water available is required for human consumption. Technologies for treating wastewater in municipal as well industrial applications have to be improved and further developed so as to reduce pollution of the water bodies that receive the wastewater.

The availability of water, its cost and the treatment of wastewater are a growing area of concern. Hence great care should be taken in conserving this important resource in Kenya which is regarded to be water scarce (Ngigi & Macharia, 2006). Slaughterhouses are amongst the animal agricultural industries that produce wastewater with high concentrations of soluble and insoluble organic species. Waste water and effluents from slaughterhouses are high in biodegradable organic compounds such as proteins, carbohydrates; nitrogen and the cleaning and washing reagents. These organic compounds when incorporated in water render it polluted (Eryuruk, Tezcan, & Ogutveren, 2011; Budiyono & Johari, 2010).

Ndarugu river (where the slaughterhouse being studied discharges its wastewater) serves as a source of freshwater to many of the villages located along its profile both within Kiambu county and parts of Nairobi County hence needs to be protect from pollution. Ndarugo is one of the tributaries of the Arthi River which is a very important river in

Kenya. This river however receives a lot of untreated agricultural and industrial waste discharge as it traverses along the industrial and agricultural areas of Gachororo, Gatundu and Juja resulting in high pollution and therefore low quality of its water as well as that of the receiving bodies downstream (Hadgu, Nyadawa, Mwangi, & Kibetu, 2014). Ensuring that water is free from both organic and inorganic contaminants before discharging it into the river is important.

Chemical flocculation and coagulation are typically used as methods of water purification process for the removal of soluble and suspended pollutants from raw wastewaters. The main function of these two processes is to boost particle separation in the succeeding processes of sedimentation, floatation, filtration and, sedimentation. Chemical coagulation and flocculation processes are used in industrial as well as municipal treatment systems for raw water and wastewater. These processes enhance the removal of various types of pollutants from water streams such as; heavy metals, natural organic matter and nutrients (Eryuruk *et al*., 2011). Aluminium salts and iron salts such as chlorides and sulphates are the most commonly used chemicals for the coagulation process. These salts work by forming various hydrolysis products in the water based on its chemistry such as concentration of anions and its pH. The formed metal cations and hydroxides then destabilise colloid pollutants present in the water by decreasing repulsion forces between the colloids and trapping particles present in the sludge.

Electrocoagulation (EC) method is one of the electro-chemical processes used in tertiary treatment of water and has been has been suggested as an advanced alternative to chemical coagulation in pollutant removal from raw waters and wastewaters. It has been found to be very effective in removing inorganic and organic contaminants as well as pathogens from wastewaters. Research conducted by Belkacem Khodir and Abdelkrim (2008); Saleem Bukhari and Akram (2011); and Butler Yung-Tse, Yeh and Al Ahmad (2011) indicates that EC has several advantages including: simple equipment which is easy to operate and automate, small area occupied by the plant, short retention time, high sedimentation velocity, low sludge production, no chemicals required, and processes multiple contaminants. This technology, which is fairly new, offers an alternative way of removing pollutants from wastewater, particularly that with high concentration of suspended solids such as slaughterhouse wastewaters. The EC process is currently attracting considerable attention in treatment of industrial wastewater for its environmental compatibility and versatile nature (Budiyono & Johari, 2010). It has been applied in treating water containing various contaminants including oil wastes, mine wastes, foodstuff waste, suspended particles, organic matter from landfill leacheates, dyes, synthetic detergent effluents, solutions containing heavy metals, deflourination of water, and chemical and mechanical polishing waste (Budiyono & Johari, 2010; Bayramoglu, Eyvaz, & Kobya, 2007). Several types of wastewater such as textiles industrial effluent, domestic wastewater; lecheate, and chemically industrial fibre have successfully been treated using the EC technology (Budiyono & Johari, 2010; Chen, 2004). According to the findings of the study by Adhoum and Monser (2004) and Chen, Chen, and Yue (2000), EC has also been found to be effective in treating wastewaters from various food industries such as restaurants and olive oil processing plants.

EC process has shown to be a very effective and reliable technology that provides a method of treating wastewater from various sources and is environmentally compatible (Chen, 2004; Bayramoglu *et al*., 2004; Mollah Schennach, Parga, & Cocke, 2004). Moreover, unlike the case of chemical treatment, the salt content of the liquid /electrolyte does not increase appreciably during the EC process (Budiyono & Johari, 2010).

Electrocoagulation is based on dissolution of the electrode material used as an anode. This so-called “sacrificial anode” produces metal ions which act as coagulant agents in the aqueous solution in situ (Emamjomeh & Sivakumar, 2009). At its simplest, an electrocoagulation system consists of an anode and a cathode made of metal plates, both sub-merged in the aqueous solution being treated (Kuokkanen, Kuokkanen, Rämö, & Lassi, 2013). The electrodes are usually made of aluminium, iron, or stainless steel (SS), because these metals are cheap, readily available, proven effective, and non-toxic. Thus they have been adopted as the main electrode materials used in EC systems (Akbal & Camci, 2010). The sacrificial anodes corrode to release active coagulant initiators or precursors into solution. These hydroxides/polyhydroxides/polyhydroxy-metallic compounds have a strong affinity for dissolved, suspended or any ions that might be dissolved in the water to cause coagulation which then removes impurities from the water (Kuokkanen *et al.,* 2013).

1.2 Problem Statement

The Kenya Environment Management and Co-ordination Act (EMCA) 2015 (amended Act of EMCA (1999)), and the Environmental Impact Assessment and Audit Regulations, 2005 demand that all business establishments before this Law be audited to determine their effects to the environment and develop Environmental Management Plan and Monitoring guidelines. EMCA has mandated the Standards and Enforcement Committee of the National Environment Management Authority (NEMA) to recommend minimum effluent quality standards. This Act stipulates that all effluent from any industrial undertaking should be discharged into the local authority sewerage system where that is available and that the effluent should be treated before being discharged into the sewerage system or the environment. It also stipulates that standards for the waste and the disposal methods should be prescribed. Industrial enterprises are expected to comply with stipulated standards for effluents discharged into the environment. However, slaughterhouses in the country do not comply with the Act and most discharge their wastewater into the environment raw. Slaughterhouses have come into focus as key players in the meat industry which cause significant environmental impacts in both the biophysical and social dimensions.

The problem of wastewater treatment arises from the question of how to justify on economic grounds the treatment of wastewater including effluent. Though this may not be estimated in quantitative terms the gains from cleaner surface water, reduction in the risk of waterborne diseases and improved public health and usability of surface waters are real benefits even though they defy quantification. Most slaughterhouses in the country (including the case study located at Gachororo) do not have any form of effluent pre-treatment before the effluent is discharged into the environment, mainly streams and rivers, which in most rural areas are used downstream for domestic purposes, irrigation and aquaculture. Those that have any pre-treatment use mainly soak pits which also have an impact on the surface and ground water. There are a few slaughterhouses within the urban areas, which discharge their wastewater into the local authority sewerage systems. This main reason is that wastewater treatment is an expensive and complicated process that most small-scale industries including slaughterhouses avoid and often end up discharging their effluents into the environment untreated causing pollution.

The main pollutant in slaughterhouse effluents is soluble and insoluble organic matter which increases concentrations of biological oxygen demand (BOD) and chemical oxygen demand (COD) of the receiving water bodies hence lowering their quality. When micro-organisms degrade organic material in water, they utilise the dissolved oxygen. When the dissolved oxygen is depleted at rates higher than can be replenished by absorption on water surface and photosynthesis, there is interference with aquatic life. Slaughterhouse wastewaters contain high proportion of blood as well as excreta, indigested food, grease and hairs. All these contribute to very high BOD in the receiving water bodies (Eryuruk *et al.,* 2011; Budiyono & Johari, 2010). This implies that untreated slaughterhouse effluent discharged into municipal sewerage would have a far higher BOD than most sewage treatment plants can handle. Most sewage treatment plants are designed to handle wastewater of domestic sewage quality hence they are likely to fail when such wastewater is discharged into them without prior treatment. The polluting chemicals of most concern from slaughterhouse wastewaters are nitrates, nitrites and phosphates which bring about changes in the aquatic life of the fresh water bodies where they are discharged into (Kanu & Achi, 2011). These chemicals are potentially very destructive by interfering with the aquatic ecosystem. Pathogenic microorganisms from animal wastes can also get incorporated in the water and can be transmitted to humans upon contact with the water. It is important that slaughterhouse wastewater is treated before it is discharged into the environment. This study therefore proposes a wastewater treatment method that is more cost-effective and environmental friendly which can be adopted by slaughterhouses.

1.3 Significance and Justification of the Study

In Kenya, wastewaters from slaughterhouses are usually discharged into the environment (particularly water bodies) raw. NEMA has in the recent years closed several abattoirs within Nairobi and the surrounding areas either permanently or temporarily for not complying with various regulations, especially on the management of solid and liquid waste. According to NEMA, most of these slaughterhouses do not have pre-treatment facilities and quality analysis facilities hence cause a lot of pollution to the environment. Dagoretti abattoirs for example have previously found to be pollutants of the environment within Dagoretti Market and particularly the Kabuthi River by disposing sludge which promotes water pollution and eutrophication. Kiamaiko slaughterhouses have also been found to be a major source of pollutants to Nairobi River as it was established that they all discharge their waste directly into the River. This reduces the levels of oxygen concentrations thereby affecting aquatic life and also making the water unsafe for domestic use. Vegetation along the riparian environment also gets clogged with the indigested seeds from the intestines of animal carcasses leading to prolification of nuisance plants.

Slaughterhouses in the country discharge their wastewaters raw into the environment because of the limitations that characterise most of the commonly used pre-treatment facilities that make them costly and ineffective. Developing and adopting wastewater treatment techniques that are cheaper and more effective has therefore become necessary in Kenya.

Electrocoagulation seems to provide a solution to limitations that characterise the conventional water treatment methods used in Kenya because of its high effectiveness, rapid achievement of results, reduced labour needs and lower maintenance cost (Heidmann & Calmano, 2008; Belkacem *et al*., 2008). Previous research on electrocoagulation and its application in treatment of industrial wastewater owing to the increase in environmental restrictions on effluent wastewater and has found it to be viable and effective even for the smallest factories. Whereas this process has been proposed as a potential effective method to treat wastewater including that from slaughterhouses, it is currently not applied in Kenya. This is in spite of spite of the advantages it has been found to have over other methods and the fact that it is a simple and efficient method that can be used by both small and large industries.

Determining how effective electrocoagulation method is and comparing the performance of Aluminium and Iron as electrode material is important as it could give slaughterhouses a more sustainable treatment alternative to their waste water problem. This will not only save the money for the slaughterhouses but also contribute towards protection of the environment and water bodies in the country from pollution due to dumping of untreated waste water by slaughterhouses.

1.4 Research Objectives

The main objective was to determine the effectiveness of electrocoagulation as a method of treating wastewater from slaughterhouses in Kiambu County, Kenya.

1.4.1 Specific objectives:

1. To characterize wastewater from slaughterhouses in terms of COD, BOD_5, total alkalinity, turbidity, total suspended solids (TSS), electrical conductivity and pH.
2. To determine the effectiveness of electrocoagulation in treating slaughterhouse wastewater.

3. To compare the performance of iron and aluminium as electrode materials in electrocoagulation treatment of slaughterhouse wastewater.

1.5 Research Questions

1. What are the characteristics of the slaughterhouse wastewater before and after treatment through electrocoagulation?
2. Is electro-coagulation an effective method for treating wastewaters and can the water be recovered and be re-used?
3. How does the performance of iron compare to that of aluminium as electrode material in ctrocoagulation process of treating slaughterhouse wastewater?

1.6 Limitations of Scope of Study

The main limitation of the scope of this study is that it is focused on only one case study in that water was sampled from only one slaughterhouse. This could introduce issues of generalizability of the findings. The water from the treatment process can also not be used for domestic purposes as the study did not conduct further treatment to achieve the recommended standards for domestic water which can also be consumed.

CHAPTER TWO

LITERATURE REVIEW

2.1 Electrochemical Technology

Various promising techniques that use electrochemical technology that do not require addition of chemicals are being developed and improved (Deniel, Bindu, Rao, & Anjaneyulu, 2008). Electrochemical technology can be used to remove ions, silicates, humus, dissolved oxygen (Chen, 2004), reduce copper (Comninellis and Pulgarin, 1993) and decolorize wastewater (Heidmann & Calmano, 2010). Electrochemical technology has also been applied successfully to treat potable water, protein wastewater, yeast wastewater, urban wastewater, restaurant wastewater, tar sand, and oil shale wastewater, nitrate containing wastewater, heavy metals, textile dyes, fluorine, polymeric wastes, aqueous suspensions of ultrafine particles, and phenolic waste (Nouri, Mahvi, & Bazrafshan, 2010). Recent research hints at electrochemical technologies being more efficient and more compact. The removal mechanism of the electrochemical process includes coagulation, adsorption, precipitation, and flotation (Heidmann & Calmano, 2010).

Chen (2004) describes lectrocoagulation (EC) as one of the most simple and efficient electrochemical methods for the purification of various types of water and wastewaters EC is characterized by the following advantages; simple equipment, easy to operate, and low amounts of sludge (Nouri *et al.*, 2010). This process involves electrolytic oxidation of a suitable anode made to generate the coagulant at an appropriate pH. It forms an insoluble metal hydroxide which then removes various pollutants from the wastewater (Chen, 2004; Budiyono & Johari, 2010). The metal hydroxide ions/species neutralize the electrostatic charges present on the suspended solids and oil drops facilitating coagulation and the resulting separation from the aqueous phase (Chen, 2004; Kanu, Achi, Ezeronye, & Anyanwu, 2006). Review of existing literature demonstrates a growing interest of using EC in the treatment of various types of wastewaters: metal processing wastewaters (Kanu & Achi, 2011), semiconductor production wastewater (Parawira, Kudita, Nyandoroh, & Zvauya, 2005), textile dyeing wastewaters (Rezaee, Hossini, Masoumbeigi, & Soltani, 2011), tannery wastewater pre-treatment (Rulkens, 2006), olive mill wastewater (Liu, 2003), urban wastewater (Cuetos, Gómez, Otero, & Morán, 2008), and organics removal from poultry slaughterhouse wastewaters (Eryuruk *et al*., 2011). EC has also been found to be effective in treating landfill leachate by some researchers (Budiyono & Johari, 2010).

In EC using iron electrodes, the generation of iron hydroxides $Fe(OH)_n$ is followed by an electrophoretic concentration of colloids which are usually negatively charged in the region close to the anode (Kurt, Gonullu, Ilhan, & Varinca, 2008). The produced ferrous ions hydrolyze to form polymeric hydroxide complexes and monomeric hydroxide ions that are dependent on the pH of the solution. The polymeric hydroxides, which are highly charged cations, destabilize the negatively charged colloidal particles allowing aggregation and formation of flocs. Generation of the hydroxides also depends on how soluble the metal hydroxide is. When the iron amount in the water being treated exceeds the solubility of the metal hydroxide, the amorphous metal hydroxide precipitate is formed, which causes sweep-floc coagulation (Saleem *et al*., 2011).

Coagulation in the EC process is generated in situ by electrolytic oxidation of an anode made from appropriate material (Shafaei, Rezayee, Arami, & Nikazar, 2010). Charged ionic species are removed from wastewater during this process through its reaction with an ion having the opposite charge, or by reacting with floc metallic hydroxides produced within the effluent. EC treatment methods present an alternative to the use of polymers, polyelectrolyte and metal salts that are usually added in the water being treated to break stable emulsions and suspensions (Kongjao, Damronglerd, & Hunsom, 2008). The EC treatment method generates polymeric metal hydroxides that are highly charged in the aqueous media. This neutralizes the electrostatic charges that exist on suspended solids and oil droplets to facilitate coagulation and the resulting separation from the aqueous phase. This triggers the precipitation of certain metals and salts. According to Chen (2004), treatment performance of EC system is studied by observing optimization of electrical current and of electrode types, the two most important parameters for the EC method.

2.2 Theory of Electrocoagulation

EC has a long history: the first plant was built in London in 1889 for the treatment of sewage (Moreno *et al*., 2009; Vik, Carlson, Eikum, & Gjessing, 1984). In spite of some promising results, the success of this technology has been limited. However, there has been renewed scientific, economic and environmental interest in this technology in recent years due to demand of alternative water treatment technologies. EC understandably has several similarities with the chemical coagulation but also significant differences, such as side reactions, which are discussed in this section.

In the EC system there are multiple electrochemical reactions occurring simultaneously at the anodes and cathodes. These mechanisms can be divided into the main mechanisms that cause destabilisation of pollutants, and side reactions, such as hydrogen formation.

2.2.1 Main Reactions

EC mechanism is extremely dependent on the chemistry of the aqueous medium, in particular its conductivity. The mechanism by which ions are generated by the EC process can be explained using iron and aluminium. These electrodes produce coagulants into water. In addition, there can be inert electrodes, typically cathodes, which are sometimes used as counter-electrodes in the system. Iron produces iron hydroxide in an electrolyte system. In the case of iron or steel and aluminium anodes, two mechanisms for the production of the metal hydroxide have been proposed (Kurt *et al*., 2008; Saleem *et al*., 2011). Iron and aluminium cations dissolve from the anodes according to Equations 1 and 2.

$Fe (s) \rightarrow Fe^{n+}(aq) + ne^-$……………………………...…………………………………(1)

$Al (s) \rightarrow Al^{3+}(aq) + 3e^-$…………………………………..……………………......(2)

In typical aqueous conditions and environment of the EC process, iron can dissolve in divalent Fe (II) and trivalent Fe (III) forms, whereas aluminium dissolves only in trivalent form Al (III). Fe (II) can further oxidise to Fe (III) (Eq.3) if oxidation-reduction potential (ORP) and pH conditions are suitable. Oxygen has to be present and pH has to be neutral or alkaline to achieve a reasonable reaction rate (Moreno *et al*., 2009).

$4Fe^{2+} (aq) + O_2(g) + 10H_2O(l) \rightarrow 4Fe(OH)_3(s) + 8H^+_{(aq)}$................................. (3)

Moreno *et al*. (2009) studied the electrochemical reactions of iron electrodes in EC system. According to their results and the thermodynamical data they presented, the region where Fe (III) is generated is in essence the potential of the iron anode in an EC system. Other researchers however argue that the potential of the cathode is in the region where Fe (III) is reduced to Fe (II) form and therefore, both forms exist in the EC system (Vepsäläine *et al*., 2012). Iron is produced on the anodes mainly in Fe (II) form (Morgan & Lahav, 2007). At low pH the chemical dissolution of iron can be significant and total iron concentration can be higher than would be theoretically expected. Sasson, Calmano, and Adin (2009) also studied oxidation of produced Fe (II). According to their results, no significant oxidation occurred at pH 5; oxidation rate was moderate at pH 6 and very rapid oxidation occurred at pH 7–9.

The amount of metal cations dissolved during the reactions at the anode can be calculated according to Faraday's law (Eq. 4).

$$m = \frac{ItMw}{zF} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 4$$

where I is the current (measured in amperes), t refers to the operation time (in seconds (s)), M_w refers to the molecular weight of the substance (g/mol), F is Faraday's constant (given as 96485 C/mol), while z refers to the number of electrons involved in the reaction (2 for Fe (III) and 3 for Fe (II) and Al (III)) and m is the quantity (mass) of metal dissolved (g). Several studies have reported current yields higher than 100% for the dissolving of aluminium electrodes (Picard, Cathalifaud-Feuillade, Mazet & Vandensteendam, 2000). It seems that aluminium also dissolves from the cathodes. This occurs when pH on the surface of the cathode decreases as a result of the formation of (HO^-) as described in Eq. 5 or as a result of the consumption of hydronium ions/protons as demonstrated in (Eq. 6).

$2H_2O$ (l) $+2e^- \rightarrow H_2$ (g) + 2 OH^- (aq).. (5)

$2H^+$ (aq) + 2 $e^- \rightarrow H_2$ (g).. (6)

At higher pH, aluminium dissolves as aluminate (Eq. 7)

2Al (s) + $6H_2O$ + $2HO^-$ (aq) $\rightarrow 2[Al(OH)_4]^-$ + $3H_2$ (g)…………………………..…. (7)

In their study, Picard *et al.* (2000) examined cathodic dissolution during the EC process. They compared hydrogen evolution from cathodes made from stainless steel with aluminium cathodes. They found that hydrogen produced at the stainless steel cathodes adhered to Faraday's law. In tests with the aluminium cathodes, hydrogen production was higher due to the chemical dissolution of aluminium according to the equation 8 below;

Fe(s) + $2H_2O$ (l) $\rightarrow Fe(OH)_2 + H_2$... (8)

Hydrogen formation and aluminium dissolved from the cathodes increased exponentially with the current intensities.

When aluminium ion, aluminate or iron ions are produced on the electrodes they experience hydrolysis or dehydrolysis reactions in the solution. Green rust is formed when iron electrodes are used. Green rust contains both Fe^{2+} and Fe^{3+} hydroxides and anions, such as Cl^-, CO_3^{2-} and SO_4^{2-}. Other metal cations, such as Cu (II) and Ni (II), can also substitute Fe (II) in green rust if they exist in the solution (Refait & Genin, 1998). In EC system, green rust is due production of Fe^{2+} ions.

In summary, when iron electrodes are used, the reactions occurring during the electrochemical treatment process are as follows:

Reactions in a cathode environment;

$2H_2O(l) + 2e^- \rightarrow H_2(g) + 2OH^- (aq)$.. (9)

$8H^+ (aq) + 8e^- \rightarrow 4H_2(g)$.. (10)

Reactions in an anode environment,

$2H_2O (l) \rightarrow O2 (g) + 4H^+ (aq) + 4e^-$.. (11)

$Fe(s) \rightarrow Fe^{2+}(aq) +2e^-$ .... (12)

Due to OH^- ion concentrations increasing near the cathode, the pH of the medium begins to rise. Meanwhile, the anode melts and dissolves ferrous ions into solution as shown in the following equations;

$Fe^{2+}(aq) + 2OH^-(aq) \rightarrow Fe(OH)_2(s)$... (13)

$2Fe^{2+} (aq) + 1/2\ O_2(g) + 5H_2O(l) \rightarrow 2Fe(OH)_3(s) + 4H^+(aq)$..................................... (14)

Overall;

$Fe(s) + 2H_2O (l) \rightarrow Fe (OH)_2 + H_2$... (15)

$Al^{3+} + 3H_2O \rightarrow Al (OH)_3 + 3H^+$ ……………………...…………………………..... (16)

2.2.2 Side Reactions

Other than the dissolving of aluminium or iron electrodes, other electrochemical reactions can also occur in the EC system. These include:

i. Formation of hydrogen at the cathodes due to Eqs. 5-7
ii. Increase of pH as a result of formation of hydroxyl ions or the consumption of hydronium ions or protons as in Eqs.5 and 6).
iii. Metal ions reduction on the cathodes.

Results from studies by Mollah *et al*. (2004) and Chen (2004) have also indicated that oxygen gas is also produced on the anodes. However, according to Sasson *et al*. (2009) and Mouedhen, Feki, Wery, and Ayedi (2008), this does not seem to take place in typical conditions (when there is electrochemical potential) of EC systems as dissolution of anodes has been shown to follow Faraday's law. Nevertheless, in basic conditions (at alkaline pH), dissolution of iron anodes has been found to be lower than calculated according to Faraday's law, which is an indication that other electrochemical reactions are taking place in these conditions (Matis & Peleka, 2010).

Matis and Peleka (2010) argue that bubbles that can be used to effectively separate particles within the solution can be produced by applying electroflotation (EF) technology. EF technology involves production of oxygen bubbles on the anodes whereas hydrogen bubbles are produced at the cathodes while in EC only hydrogen bubbles are produced at the cathodes. According to these authors, floatation efficiency in EC and EF technologies is determined by bubble size. Smaller bubbles are more effective as they provide a larger surface area for particle attachment.

Bubble size is influenced by pH, electrode material, and current density. The smallest hydrogen bubbles are produced when the solution is neutral or acidic (Sarkar, Evans, & Donne, 2010). Electrodes made from stainless steel plates have also been found to produce the smallest bubbles. Effect of current density on the bubble size is however still surrounded by controversy. Sarkar *et al*. (2010) investigated the effect of electrode and current density on bubble size produced at the cathodes and found that a significant amount of hydrogen produced on the cathodes can get dissolved in the solution. According to their results, bubble size diameter is a function of hydrogen production rate, bubble nucleation rate and dissolved gas concentration field.

In contrast to chemical coagulation, EC treatment increases the pH of the water being treated when it is initially in an acidic, neutral or slightly alkaline region and decreases it when it is highly alkaline. This change of pH during the EC treatment affects the speciation of aluminium and iron hydroxides. At highly acidic pH (pH 2) the alkalinity produced during the EC is not sufficient to increase the pH of the solution, whereas at pH 3 and higher, initial pH value rises during the treatment (Matis & Peleka, 2010). This is easily understandable, as concentration of hydronium ions increases by factor 10 when pH decreases from pH 3 to pH 2. When initial pH is significantly alkaline ($pH > 9$), pH probably decreases due to the formation of aluminate $[Al(OH)_4]^-$, which is an alkalinity consumer (Sarkar *et al*., 2010). It seems that the pH change rate and final steady state pH depends on the concentration of anions in the solution. According to Trompette and Vergnes (2009), pH increases more in sulphate solution than in chloride solution. This is when sulphates replace hydroxyl ions in the hydroxide precipitates and therefore less hydroxyl ions are bound to hydroxides.

Because electrochemical reactions occur at the electrodes surface, the reaction products' concentration is highest at the surface of the electrodes, and the concentration gradient exists from the surface towards the bulk solution. For that reason, pH of the solution decreases at the vicinity of the anodes and vice versa at the surface of the cathodes. This can result in precipitation of inorganic salts on the surface of electrodes if their solubility changes as a function of pH (Kobya, Hiz, Senturk, Aydiner, & Demirbas, 2006).

2.3 Important Parameters that affect Efficiency of Electrocoagulation

There are various parameters that determine the efficiency of EC in removing pollutants from wastewater water. Parameters which are known to have an important effect are discussed in the following sections.

2.3.1 Material of the Electrodes

Various materials can be used successfully in the EC process. They include iron, aluminium and/or inert material (typically cathodes) (Vepsäläinen *et al.,* 2012). However, it should be understood that iron and aluminium ions and hydroxides differ in their chemistries as well as applications.

Electrode material determines the electrochemical reactions that take place in the EC system. Aluminium and iron electrodes are the most typically used in successful EC systems. Aluminium dissolves in all situations as Al(III) whereas there is an ongoing debate as to whether iron dissolves as Fe(II) or Fe(III) (Sasson *et al*., 2009). Most studies such as that carried out by Bagga, Chellam, and Clifford, (2008); Moreno *et al*. (2009); and Sasson *et al*. (2009) demonstrate that iron dissolves as Fe(II) but is further oxidised in bulk solution to Fe(III) if the pH is alkaline and there is presence of oxidants such as oxygen in sufficient concentration. Fe(II) has lower positive charge and is a highly soluble hydroxide compared to Fe(III) hence a poor coagulant. This also explains why poor results are obtained when iron electrodes are used as opposed to aluminium (Bagga *et al.,* 2008). Optimal material selection is therefore dependent on the pollutants to be removed as well as the elecrolytes' chemical properties.

Generally, aluminium seems to be a superior electrode material compared to iron in most cases but only when the efficiency of the treatment is being considered. It should however be noted that aluminium is very expensive when compared to iron (Bagga *et al.,* 2008).

Inert electrodes, such as metal oxide coated titanium, have also been used as cathodes in some constructions. When water has significant amounts of calcium or magnesium ions, the inert cathode material is recommended (Chen, 2004). There are also some studies where combinations of aluminium and iron electrodes have been used. Linares-Hernández, Barrera-Díaz, Roa-Morales, Bilyeu & Ureña-Núñez (2009) obtained high removal of colour with aluminium electrodes, while iron was more effective than aluminium in reducing COD from industrial wastewater. A combination of iron and aluminium removes both colour (71%) and COD (69%) with high efficiency. Similar results were obtained when paper mill wastewaters were treated with various aluminium and iron electrode combinations (Katal & Pahlavanzadeh, 2011). Katal and Pahlavanzadeh (2011) established in their study that Aluminium electrodes were most effective in removing colour of the wastewater, whereas iron electrodes removed COD and phenol from the wastewater more effectively than aluminium electrodes. A combination of aluminium and iron electrodes removed colour, COD and phenol with high efficiency. Combination electrodes have been studied for arsenic removal from groundwater (Gomes *et al.*, 2007). Iron electrodes and a combination of iron and aluminium electrodes gave the highest arsenic removal efficiencies. Fe-Al pair has been most effective in removing indium from water (Chou, Wang, & Huang, 2009).

2.3.2 pH of the Solution

pH of the solution being treated is considered to be one of the main parameters of EC treatment. This is because besides affecting dissolution of the electrodes being used, it also affects speciation of hydroxides, conductivity of the solution, and potential of colloidal matter (Chou *et al.*, 2009). Aluminium and iron cations and hydroxides cause destabilization of colloids. Effective coagulant species are formed in acidic, neutral and slightly alkaline pH. In highly alkaline pH, $Al(OH)_4^-$ and $Fe(OH)_4^-$ ions are formed and these ions have poor coagulation performance. Figure 2.1 which presents concentrations of soluble monomeric hydrolysis products of Fe(III) and Al(III) in equilibrium with the amorphous hydroxides at zero ionic strength and 25 °C indicates that Fe(III) is effective in a wider pH area than Al(III) and also works in slightly alkaline pH. It is also known that competing anions have an effect on the optimum pH of the coagulation.

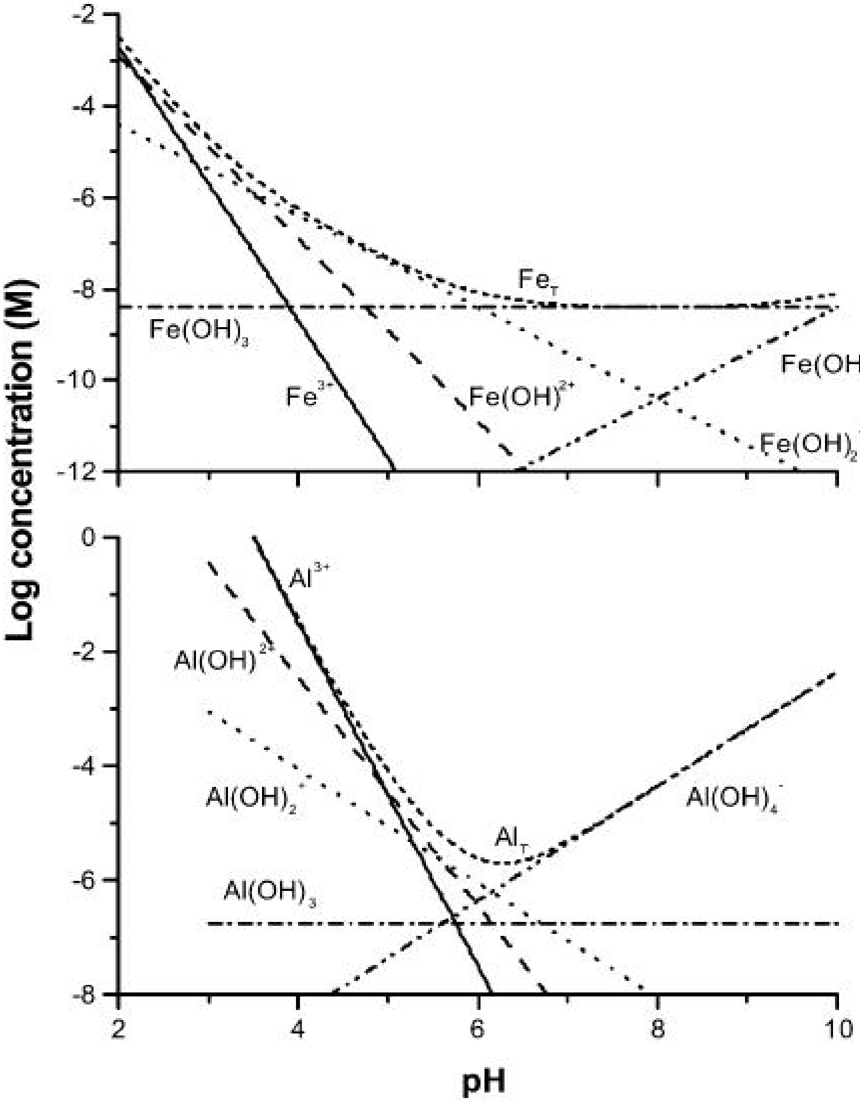

Figure 2.1: Concentrations of soluble monomeric hydrolysis products of Fe(III) and Al(III) in equilibrium with the amorphous hydroxides at zero ionic strength and 25°C

The effect of pH of water on the efficiency of pollutant removal can mostly be explained by the aforementioned mechanisms. However, pH increases during the EC treatment, making it a constantly changing parameter; therefore mechanistic studies of EC systems are difficult to conduct.

In pH lower than 3, the release rate of aluminium during electrolysis with a constant charge per volume was found to be lower than in pH above 3 (Mouedhen *et al*., 2008). Chemical dissolution of aluminium cathodes occurs because pH increases to a level where aluminate is formed. It is probable that acidic bulk solution inhibits this reaction because produced hydroxyl ions are consumed by the acid in the solution. Sasson *et al*. (2009) observed that in acidic pH, the dissolution of iron electrodes was significant even in the absence of electricity, whereas oxidation of Fe(II) to Fe(III) occurred only at pH above 5. The dissolution rate decreases at high pH, which is understandable as the corrosion rate of iron decreases in alkaline pH in the presence of oxygen because a passive layer forms on the surface.

It has been suggested that initial pH of sample being treated is a key parameter in either chemical coagulation or EC treatment method (Cañizares, Jiménez, Martínez, Rodrigo, & Sáez, 2009). EC has demonstrated to be more effective and suitable at higher pH is desired, whereas chemical coagulation is preferred when pH is low.

There are also some pollutants which have specific optimum pH treatment range (value) such as phosphorus and metal cations (Janpoor, Torabian, & Khatibikamal, 2011). For example the removal of phosphate from wastewater by EC with iron electrodes has been has been found to be most effective at low pH (pH 3).

Hanay and Hasar (2011) also studied removal of Copper II, Manganese (II) and Zinc (II) by aluminium electrodes and found that removal efficiency increased with increase of wastewater pH. Similar results have been obtained for the removal of Co (II) and As(V)

(Hanay & Hasar, 2011), Cu, Cr, Ni, Cu(II), Pb(II), Cd (II) (Al Aji, Yavuz, & Koparal, 2012).

2.3.3 Current Density and Treatment Time

Current density is directly proportional to the rate of the electrochemical reactions occurring on the electrode surface. This influences the electrode potential, which characterizes the reactions that occur on the electrode surface (Vepsäläinen *et al.,* 2012). It seems that on iron and aluminium anodes, dissolution reaction is the primary reaction, and the proportion of other reactions is insignificant at the typical current densities and electrode potentials when pH is neutral or acidic (Sasson *et al*., 2009; Mouedhen *et al*., 2008). In a situation of alkaline pH, the dissolution rate of iron anodes can be lower than the value calculated by Faraday's law, which implies that there can be other reactions on the anode at these conditions (Sasson *et al*., 2009).

Coagulant produced by electrolysis can usually be calculated according to Faraday's law (Eq. 4) when current and treatment times are known. Coagulant concentration produced by electrolysis on anodes is typically directly proportional to the electric charge added per volume or coulombs per litre. However, the total amount of coagulant dissolved also includes chemical dissolution of the electrodes in low pH and the dissolution of aluminium cathodes. It is possible that current density has some influence on aluminium cathode chemical dissolution as it affects the rate of hydroxyl ion production at the cathodes. Mouedhen *et al*. (2008) studied aluminium dissolution from the cathodes with an electrochemical cell constructed of platinised titanium anode and aluminium cathode. They used constant charge per volume of 540 C/L and various current densities. According to their results, as the current density decreases the amount of aluminium generated increases. These results indicate that even low current density of less than 1 A/dm^2 initiates the dissolving reaction on the cathodes if solution pH is not highly acidic and therefore the produced amount of aluminium on the cathodes depends more on the treatment time than it does on the electric charge added per volume. Treatment time and/or electric charge added per volume is directly proportional to the amount of coagulants produced in the EC system.

2.3.4 Concentration of Anions

Concentration of anions in the wastewater affects the stability of passive layer of the aluminium electrodes. Sulphate anions are passivating agents and therefore reduce the production of metal cations (Kolics, Polkinghorne, & Wieckowski, 1998). Chloride, on the other hand, induces the breakdown of the passive layer and pitting corrosion. How supporting electrolytes affect the efficiency of an EC system has been studied by treating unskimmed milk samples and a cutting oil emulsion by aluminium electrodes in the presence of NaCl, Na_2SO_4, NH_4Cl and $(NH_4)_2SO_4$ (Trompette & Vergnes., 2009). According to results, sulphate anions increase electrical consumption and have a negative effect on EC efficiency. Ratio of $[Cl^-]/[SO_4^{2-}]$ should be 0.1 or higher to ensure breakdown of the passive film. Mouedhen *et al*. (2008) reported that optimum concentration of Cl^- ions was 61 mg/l.

Besides the effect on the passive layer, some salts can precipitate on the cathodes if the concentration of salts in water is sufficiently high (Hasson, Lumelsky, Greenberg, Pinhas, & Semiat, 2008). This insulating (passive) layer increases power consumption significantly (Chen, 2004). Competing anions can replace hydroxyl anions in the precipitate, which has an effect on the properties of hydroxide and therefore, also on efficiency and optimum conditions of coagulation processes. Competing anions can also have a direct influence if the pollutant is an anion e.g. fluoride or phosphate. Hu, Lo, & Kuan (2003) studied fluoride removal in the presence of chloride, nitrate and sulphate anions. They obtained the best results in the absence of these anions. Sulphate ion concentration was found to have a significant effect on flouride removal efficiency, most likely because of its ability to compete with fluoride ions from Aluminium III Floride. Vasudevan, Lakshmi, Jayaraj and Sozhan (2009) found that fluoride and arsenate are able to compete with phosphate ion and therefore enable its removal by EC.

2.3.5 Effect of Temperature

A number of studies have explored how temperature affects the removal of pollutants through EC. Yilmaz, Boncukcuoğlu, Kocakerim, Yilmaz and Paluluoğlu (2008) studied the effect of solution temperature on removal of boron using EC in the range of 2 93–333 K. Boron removal efficiency was found to increase from 84-96% when temperature was increased from 293 K to 333 K. However, the opposite effect has also been reported in a different study by Katal and Pahlavanzadeh (2011) when wastewater from a paper mill was treated at temperatures between 293K and 333K. In this study, removal of phenol, colour, and COD was found to decrease by 10–20% when the temperature increased from 293K to 333K.

Another study by Vasudevan *et al*. (2009) investigated the effect of temperature on removal of phosphate from wastewater by EC in the range of 293–333K. Removal efficiency was 29% lower at 293K than in higher temperature. Authors concluded that at low temperatures the dissolution of anode occurs at a lower rate. However, they did not present any results, such as concentrations of aluminium dissolved from electrodes, which would support this conclusion.

It has been suggested that when temperature is too high, there is shrinkage of large pores of the $Al(OH)_3$ gel, which causes the formation of dense flocs that are more likely to deposit on the electrode surface (Chen, 2004). Increasing temperature also enhances the solubility of aluminium. However, experiments suggest that increasing temperature can affect removal efficiency either positively or negatively (Chen, 2004). Therefore, there is possibility that the effect of temperature on removal efficiency is subject to the removal mechanism of pollutants.

2.4 Applications of Electrocoagulation

Most of studies that have investigated the EC process have concentrated on the removal of certain specific pollutant or pollutants from synthetic or real solutions. The majority of these application studies can be divided into the following categories (Vepsäläinen *et al*., 2009; Katal & Pahlavanzadeh, 2011).

1. Removal of metal ions and/or hydroxides from synthetic solutions, groundwaters or wastewaters. Typically, iron or combination electrodes (iron and aluminium) are used in EC system (Nouri *et al*., 2010; Rezaee *et al*., 2011; Shafaei *et al*., 2010; Meunier, Drogui, Mercier, & Blais, 2009).
2. Removal of organic material from wastewaters or synthetic solutions. High removal (> 70%) is typically obtained with optimum parameters. Aluminium, iron and combination electrodes can be used. In general, iron electrodes give higher organic matter removal, whereas higher colour removal is obtained with aluminium electrodes (Heidmann & Calmano, 2008; Kongjao *et al*., 2008; Kurt *et al*., 2008; Budiyono & Johari, 2010).
3. Purification of surface waters from natural organic matter, inorganic pollutants or microbes. Typically high removal of pollutants (> 90%) (Bayramoglu *et al*., 2007; Butler et al., 2011). Aluminium electrodes are more commonly used than iron electrodes in these applications (Saleem *et al*., 2011; Belkacem *et al*., 2008; Shafaei *et al*., 2010).

2.5 Economical and Ecological Considerations

A number of studies have determined operating cost of treating waste waters using different methods including EC. These calculations often consider cost of electrodes, energy and chemicals used. It is important to keep in mind that the cost of materials as well as energy often change over time hence operating costs are only estimates and not fixed prices. Cost calculations usually exclude investment costs such as power supplies, sludge separating systems and electrochemical cell containers, which may be considerable.

The treatment cost of dye polluted wastewaters has been estimated by Eyvaz, Kirlaroglu, Aktas and Yuksel (2009). The estimated operating cost was USD 1.3–3.4 per kg TOC removed depending on parameters, such as treatment time. Sridhar, Sivakumar, Immanuel and Maran (2011) carried out an economic analysis of the operating cost of EC treatment of bleaching effluents from pulp and paper industry. Operating costs varied from USD 1.52 per m^3 to 1.72 per m^3. Ölmez (2009) estimated the cost of removing Cr (VI) from wastewaters. The total cost of the EC process was about two-fold compared to the conventional process because of higher electricity consumption. Meunier *et al*. (2009) compared the cost of electrochemical and chemical precipitation with calcium hydroxide or sodium hydroxide. EC was up to five times cheaper than chemical precipitation.

Emamjomeh and Sivakumar (2009) estimated operating costs of fluoride removal from water. Operating cost was USD 0.26-0.44 per m^3 when initial fluoride concentration was 5 mg/l. Kobya *et al.* (2007) have published their operating cost calculations with regards to treatment of fluids containing waste metal cutting using EC. The operating cost was USD 0.025–0.90 per m^3 and USD 0.01–0.79 per m^3 with iron and aluminium electrodes, respectively. Espinoza-Quiñones *et al*. (2009) made operational cost calculations for tannery wastewater treatment with EC. They estimated that a 60 min treatment (high removal of pollutants) costs USD 1.64 per m^3, which was less than with conventional coagulants. According to Bayramoglu *et al*. (2007), the operating cost of chemical treatment of textile wastewater by chemical coagulation was 3.2 times as high as that of EC.

It is worth noting that there are few articles where the costs of chemical coagulation and EC have been compared. It might be difficult to obtain reasonable information on the price of common coagulants. Some chemical coagulants, such as aluminium sulphate and ferric chloride, are bulk chemicals and have a low price, whereas prehydrolysed metal coagulants are more expensive. The price of chemicals naturally depends on the amount that is required in the process and therefore, major consumers, such as pulp and paper mills or municipal water treatment plants, pay less for their chemicals than small users. However, iron and aluminium are typically more expensive in their metallic state than as metal salts because production of metals is energy consuming (Vepsalainen, Selin, Pulliainen, & Sillanpaa, 2007). Coagulation chemicals are produced from the minerals and are not transformed into the metallic state during this process. For example, aluminium sulphate is manufactured through the reaction of aluminium trihydrate and sulphuric acid. Aluminium trihydrate is made from bauxite mineral by purification. Also the ecological effects of EC are somewhat unknown.

As mentioned earlier, production of metallic aluminium and iron consume high amounts of energy (Emsle, 2003). It has been estimated that aluminium production consumes 5% of the electricity generated in the USA (Emsle, 2003). Typically, aluminium is recycled and this process requires significantly less energy than the production of pristine metal. The ecological effect of water purification with EC and chemical coagulants should be compared, taking energy and material consumption into account during the manufacture of metals or chemicals.

2.6 Performance of EC on the Treatment of Wastewaters from different Sources

2.6.1 Wastewater from Slaughterhouses

Treatment of wastewater from slaughterhouses by aerobic and anaerobic biological systems (Masse & Masse, 2005; Palatsi, *et al*., 2011; Cuetos *et al*., 2008), and hybrid systems (Tezcan, Koparal, & Bakir Öğütveren, 2009) have been extensively studied. Aerobic treatment processes have various disadvantages such as production of large amounts of sludge, and the high energy consumption required to enable aeration (Tezcan *et al.*, 2009). On the other hand, anaerobic treatment is often slowed by the accumulation of large amounts of suspended solids as well as floating fats in the equipment which reduce the methanogenic activity as well as biomass wash-out (Cuetos *et al*., 2008). Cuetos *et al*. (2008) added that anaerobic treatment has also been reported as being sensitive to high rates of organic loading. While biological processes have shown to be effective and economical, both of the two processes require high concentration of biomass and control of sludge loss, large reactor volumes, and long hydraulic retention times so as to avoid sludge wash-out (Asselin, Drogui, Benmoussa, & Blais, 2008).

Dissolved air flotation (DAF) and coagulation–flocculation units are amongst the most widely used physico-chemical processes total suspended solids, fats and colloids removal from slaughterhouse wastewaters (Bayar *et al*., 2011; Asselin *et al*., 2008). The study conducted by Asselin et al., concluded that total suspended solids was removed at 89%, turbidity 90%, BOD 86%, and oil and grease 99%, when completing electrocoagulation by combining mild steel or aluminium electrodes for treating slaughterhouse wastewater. In addition, the total cost of treating the wastewater was found to be 0.71 USD/m^3 for treated poultry slaughterhouse effluent, particularly including energy and electrode consumption and chemical and sludge disposal.

Bayar, Yıldız, Yılmaz and İrdemez (2011) also established the effectiveness of EC in successfully treating slaughterhouse wastewaters. These authors studied current density and stirring speed effect in the treatment of wastewater from poultry slaughterhouse using EC with aluminium electrodes. In the experiments, the best removal efficiency was obtained, when initial pH and current density were adjusted to 3.0 and 1.0 mA/cm^2, respectively. Increasing current density values decreased COD removal efficiency. The highest removal efficiencies of 85, 85, 81 and 71% were obtained with the current density of 0.5, 1.0, 1.5 and 2.0 mA/cm^2, respectively. Initial pH values of these removal efficiencies were 4.0, 3.0, 3.0 and 5.0. When experiments were performed to investigate the turbidity removal efficiency, it was found that current density of 1.0 mA/cm^2 provided 98% removal efficiency. Results from this study showed that electrocoagulation is a suitable method for treatment of effluents from poultry slaughterhouses

EC of cattle-slaughterhouse wastewater by aluminium and iron cylindrical anodes was also studied by Tezcan *et al*. (2009). EC method was found to remove COD, BOD, suspended solids and turbidity from the wastewater. In a different study, Budinyono *et al*. (2010) investigated performance of the electro-coagulation method in treatment of high strength slaughterhouse wastewater in batchwise mode. They studied the effect of various parameters and variables such as initial pH, operating time, electrode material and surface area of electrode, and content of suspended solids in the water. Content and operating time were investigated. All the tests were run at a constant current of 125 A/m^2. The study found the effective surface area for electrode pair to be 6.28 cm^2. The study also established that higher numbers of electrodes reduced the amount of time required to attain high removal efficiency of SS from the water. Based on the findings, EC was found to be an effective method in removing suspended solids from slaughterhouse wastewater.

Eryuruk *et al*. (2011) examined the effects of the operating parameters, such as supporting electrolyte concentration and polyelectrolyte addition on COD removal in the electrocoagulation process. The energy consumptions were also analyzed. The COD removal efficiency of 72% was obtained with the addition of polyelectrolyte after 90 min electrocoagulation. The study concluded that electrocoagulation is a relatively suitable process for removal of COD and effective treatment of slaughterhouse wastewater using iron electrodes.

2.6.2 Domestic Wastewater Treatment

Several researchers have studied application of EC in treating domestic wastewater. Sarala (2012), reports that wastewater usually consists of a number of contaminants, such as total dissolved solids (TDS), COD, colours and TSS. In that study, a wastewater sample was tested in experimental work, with EC after passing each current (0.12, 0.25 and 0.36) amp for each time period (5, 10, 15 and 20) minutes. The results of the study showed maximum reduction of COD and TDS at 20 minutes for 0.25 A. While in a different study, Saleem *et al.* (2011), found that the application of 24.7 mA⁄cm^2 current density with an inter electrode spacing of 5 cm may provide 91.8%, 77.2% and 68.5% removal in turbidity, COD, TSS within 30 min of electro-coagulation treatment. In their research, Rodrigo, Cañizares, Buitrón and Sáez (2010) demonstrated the capability of removing ionic phosphorus and COD, when using conductive-diamond electrochemical oxidation and electrocoagulation for persistent organic consumption, specifically regeneration of urban wastewater. The study stated that energy consumption is capable of removing at values lower than 4.5 kWh/m^3.

2.6.3 Industrial Wastewaters

Mansouri, Hannachi, Abdel-Wahab and Bensalah (2012) highlighted that EC using aluminium electrodes achieved a high removal efficiency of chemical oxygen demand (≥80%) from aqueous solutions containing 0.51 g·L^{-1} tannic acid. The primary mechanism implicated in eliminating tannic acid from water by EC using aluminium electrodes involves the adsorption of tannic acid molecules on the aluminium hydroxide surface. The results of the treatment of raw wastewater obtained from the pulp and paper industry with an initial chemical oxygen demand (COD) concentration of 1450 mg·L^{-1} have shown that more than 60% of COD can be removed by electrocoagulation using Al electrodes under optimized experimental conditions. The specific energy required for the electrochemical process with Al electrodes was estimated to range from 1 to 2 kWh·m^{-3}.

Merzouk *et al*. (2011) investigated the efficiency of electrocoagulation (EC) for the abatement of COD, total organic compounds (TOC) absorbance (i.e. color) and turbidity from a real textile wastewater, a pure red dye solution (disperse dyes 2-naphthoic acid and 2-naphthol) and a solution combining the two above fluids. Treatment of the solution combining the two above fluids allows for investigation into whether the removal of several polluting matters by electrocoagulation could be considered as the superimposition of the various treatments of single-species effluents, in a sort of additivity principle. According to the findings, TOC and turbidity and TOC are additive variables in the treatment of the industrial textile waste and dye solution as electrocoagulation seems to proceed with no interaction between the two types of matter to be removed, namely the dyestuff, and the lot of pollutants contained in the industrial waste.

In their study, Meas, Ramirez, Villalon and Chapman (2010) found that by using an electrocoagulator with sacrificial electrodes, COD, colour and turbidity removal was by (95%), (99%), and (99%) respectively when testing fluorescent penetrated liquid for non-destructive testing of parts, where the water was reused 4 times. Pajootan Pajootan, Arami and Mahmoodi (2012) found that decolorization can be achieved at 98% under an optimum condition of 1.875 A/cm^2 current density, a pH of 6, distance of 1.5 cm between electrodes and use of sodium chloride electrolyte when seeking to remove Direct Red 8 from synthetic wastewaters. The study by Merzouk *et al*. (2010) also established that; 76.2% turbidity, 85.5% suspended solids, 88.9% BOD, 93% colour and 79.7% COD can removed by the combination of electrocoagulation-electroflotation after ensuring optimum conditions for 300 mg/L silica, current density of 11.55 mA/cm^2, pH of 7.6, conductivity of 2.1 mS/cm, treatment time of 10 minutes, and electrode gap of 1 cm.

2.6.4 Heavy Metals

In another major study Shafaei *et al*. (2010) found that electrocoagulation was capable of removing Mn^{2+} ions with aluminium electrodes under an optimum pH of 7.0. The authors concluded that the density and electrolysis time, along with initial concentration were capable of determining successful removal rates.

Removal of Ni, Cu, and Cr from very heavily polluted industrial galvanic wastewater was carried out by batch- EC in a study by Akbal and Camci (2011). The raw pH of the wastewater was 1.5, which is strongly acidic, and adjustment to 5 was found necessary. The wastewater had very high conductivity (41 mS/cm) and its metal content was extremely high; around 2 g/l Ni, 2.5 g/l Cu, and 0.7 g/l Cr (70% present as Cr (VI)). The optimum electrode configuration of the EC system consisted of two separate anode-cathode-pairs used simultaneously instead of a single one made of Al or Fe. This novel EC process was found very efficient in removing metals from galvanic wastewater. It was concluded that EC could be a good alternative or an after-treatment (the varying composition of such wastewaters may limit the feasibility of EC as a primary method) to conventional methods in this application.

A study on decolorization of wastewater from restaurants containing inorganic pollutants and organic contaminants by a continuous EC process using Fe/Ti electrodes was conducted. The results showed great potential in EC-based decolorization (91% color removal efficiency) of this wastewater with only a slight initial pH alteration from the natural 7 to 8 needed. Adding NaCl to the water showed a substantial increase in removal efficiency, possibly due to electrogeneration of Cl_2, a strong oxidant (Zhang *et al.*, 2011).

2.6.5 Dye Removal

Pajootan *et al*. (2012) published a paper in which they described the binary system dye removal by electrocoagulation (EC) process using aluminium electrode in a batch electrochemical reactor. Acid Black 52 and Acid Yellow 220 were used as model dyes. However, the wool dyeing process had been performed and dye removal from the colored wastewater by EC investigated. It was found that an increase in the current density up to 40 A/m^2 increased dye removal efficiency and the optimum pH for EC process was 5.

Electrocoagulation has been applied successfully to treat potable water (Vik *et al*., 1984), wastewater containing proteins (Beck, Giannini, & Ramirez, 1974; Chen *et al*., 2000) yeast wastewater (Khristoskova, 1984), urban wastewater (Pouet & Grasmick, 1995), tar sand and oil shale wastewater (Renk, 1988), nitrate containing wastewater solutions (Souza, Silva, Mata, Martínez-Huitle, & Mata, 2012), metals and arsenic wastewater (Rocha & Martinez-Huitle, 2011; Gao, Chen, Shen, & Chen, 2005) clay minerals (Holt, Barton, Wark, & Mitchell, 2002), oil (Xu & Zhu, 2004), COD and dissolved solids (Chen *et al*., 2000; Xu & Zhu, 2004), color and turbidity (Jiang, Graham, Andre, Kelsall, & Brandon,., 2002) and organic substances (Jiang *et al*., 2002). This technique does not require supplementary addition of chemicals, reduces the volume of produced sludge (Mollah *et al*., 2004; Meunier *et al*., 2006) and economic studies also demonstrate a financial advantage compared to the conventional methods (Meunier *et al*., 2006).

The electrogenerated flocs separate rapidly and remove colour plus turbidity from dyeing wastewaters (Chen *et al*., 2000). Electrocoagulation treatments of textile dye-containing solutions or wastewater samples have been conducted on a laboratory scale and good removal of COD, color, turbidity and dissolved solids at varying operating conditions have been obtained (Gurses, Yalcin, &nDogan, 2002; Jiang *et al*., 2002). Electrocoagulation process has been found to be very efficient in COD removal and decoloration with low-energy consumption (Naje & Abbas, 2013).

While previous research has been able to demonstrate that electrocoagulation is able to treat wastewater, research specific to the Kenyan context is not available. Previous research comparing effectiveness of iron to aluminium as electrode materials is also scarce yet electrode material is a key factor when calculating costing of treatment. This current study seeks to address this research gap by addressing these two issues.

CHAPTER THREE

RESEARCH METHODOLOGY

3.1 Collection of Slaughter House Wastewater Samples

The wastewater used in this study was collected from a slaughterhouse plant located at Gachororo, Juja in Kiambu County, Kenya.

Figure 3.1: Map showing Location of Gachororo Slaughterhousen

The wastewater was sampled twice daily, once in the early morning and later at noon to coincide with the time the slaughter took place and the cleaning to be done. The samples were taken from the pipes that drain wastewater from the slaughterhouse to the wastewater collection tank. The two water samples were then put in one large jerrycan and mixed by shaking to form one sample. The sampling procedure was done on two different days of each week for a period of four months. The wastewater was then immediately stored in a cold box after its temperature was recorded. This was to avoid any changes or biodegradation prior to analysis. It was then transported to the laboratory and stored in the refrigerator at 4°C to prevent degradation as they awaited analysis for pH, total suspended solids (TSS), conductivity, turbidity, biochemical oxygen demand (BOD), total alkalinity, and chemical oxygen demand (COD) in the laboratory.

3.2 Experimental Set Up

The experiment set up involved a bipolar batch reactor consisting of several parallel aluminium and iron electrodes which were connected to a source of power. Three litres of wastewater was used for each run. The setup is represented in Figures 3.1 and plate in Appendix 1. Each horizontal plate measured 1 cm by 10 cm with a space of 1cm between the different plates. The study used up to 10 electrodes with iron plates and four electrodes for aluminium. The electrical voltage ranged from 5V to 25 V (at a current of 0.4 A). Sampling for the determination of treated wastewater characteristics was done after 2 hours of electrocoagulation. The set up was connected to Hewlett Packard HP dual DC power supply, model 6205B which converted the alternating current to direct current. The power supply was used to regulate the voltage and from it measurements could be taken. This is illustrated in figures 3.2 and 3.3.

The appropriate voltage and current as well as electrocoagulation period were determined after a series of trials with variations in voltage as well as time.

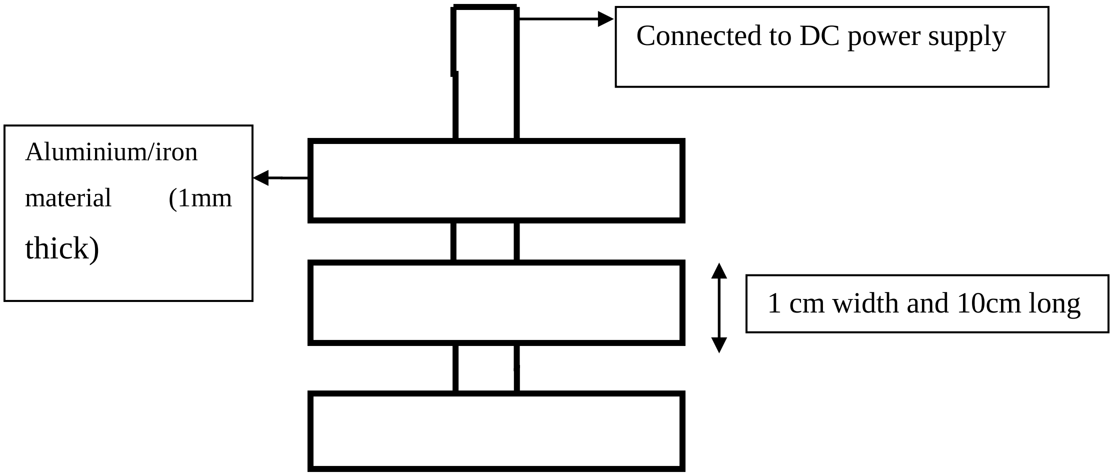

Figure 3.2: Diagram of Horizontal Electrode Configuration

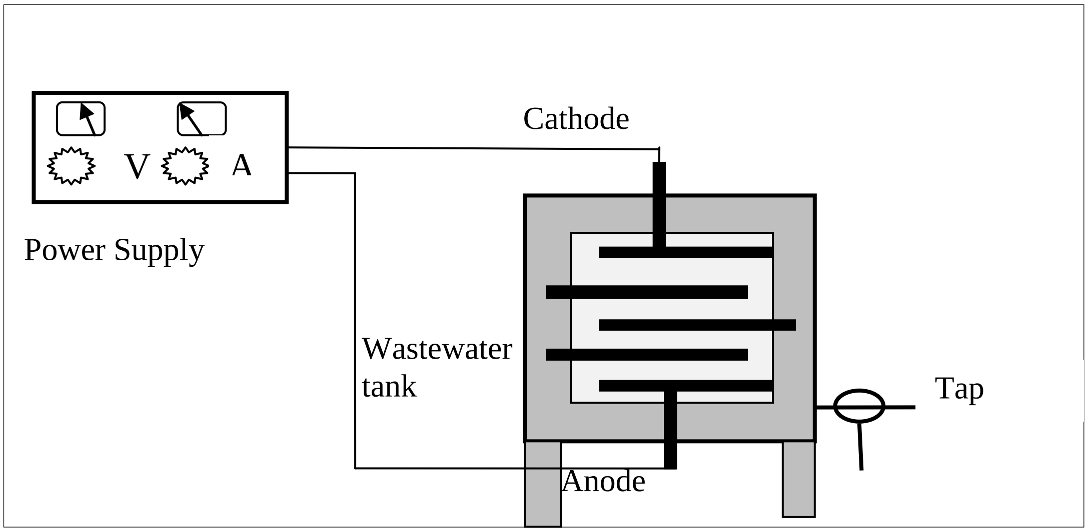

Figure 3.3: Schematic diagram of the whole set-up of the horizontally arranged plates

3.3 Determination of the Physiochemical Properties of the Wastewater in order to characterise it

3.3.1 Dissolved Oxygen

Dissolved oxygen (DO) was measured by applying standard titration method and used in determination of BOD_5.

Apparatus included: Pipette, BOD bottles, Erlenmeyer flask and a siphon.

The reagents included;

A. Manganese sulphate solution (prepared to 364g/L)
B. Alkali-azide-iodide reagent which was pepared by dissolving 500g of sodium hydroxide and 135g of sodium iodide in water and diluting to 1 litre. To this was added 10g of sodium azide dissolved in 40ml of water.
C. Concentrated Sulphuric acid (undiluted)
D. Starch indicator solution prepared by dissolving 2g of starch in 1000 ml of hot distilled water. This was preserved using 0.2g of salicyclic acid.
E. 0.025M of Standard sodium thiosuphate solution prepared by dissolving 6.205g hydrous sodium thiosulphate in distilled water and adding 1.5 ml 6N of sodium hydroxide. The mixture was then diluted to 1 litre using distilled water.

The procedure involved;

1. The sample was collected in BOD bottles by displacing the volume of each bottles for at least one minute. A siphon was used to sip samples from the container. The bottle stoppers were then carefully replaced to avoid trapping of any air bubble in the bottles.
2. The stoppers were removed and in each bottle, 1 ml of each of reagents A and B above were added in succession with the tip of the pipette well below the water

level in the bottle. The stopper was then replaced again carefully to ensure that no air bubbles were trapped

3. The contents of each bottle were mixed several times and the precipitates allowed to settle halfway down the bottle. The contents were mixed again and the precipitates allowed settling as before (halfway).
4. 2ml of concentrated acid was added using a bulb to the contents of each bottle, with the tip of the pipette well below the water level. The stopper of each bottle was replaced and the contents mixed again until all the precipitates dissolved.
5. A suitable amount of the eliquot from the bottle was transferred to an Erlenmeyer flask (depending on volume of flask). This was then titrated against standard sodium thiosulphate solution until the colour changed to pale yellow. 1ml of starch indicator solution was added and titration continued until the blue colour disappeared. Reappearance of the blue colour after first appearance was disregarded.

Dissolved oxygen was calculated using equation 17:

$$DO = \frac{a \times V_1}{V_2} \times \frac{1000}{(V_1 - 2)} \times 0.2 \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 17$$

Where;

DO = dissolved oxygen in mg/l

A = volume of titrant used

V_1 = Volume of BOD bottles

V_2 = Volume of sample partly taken from BOD bottle for titration

0.2 = oxygen equivalent of 0.025M sodium thiosulfate solution

3.3.2 Biological Oxygen Demand

This was determined using standard titration method that uses dissolved oxygen (DO). The following reagents were used;

A. Phosphate buffer (8.5g of KH2PO4, 21.75g of K2HPO4, 33.4g of Na2HPO4.7H2O and 1.7g of NH4Cl dissolved in 5000ml of distilled water and diluted to 1 litre.
B. Magnesium Sulphate solution (prepared by dissolving 22.5g MgSO4.7H2O in distilled water and diluted to 1 litre.
C. Calcium Chloride solution (prepared by dissolving 27.5 g of CaCl2 in distilled water and diluted to 1 litre.
D. Ferric chloride solution (prepared by dissolving 0.25g of FeCl3.6H2O in distilled water and diluted to 1 litre.

The Procedure involved;

1. 1 ml of each of the above reagents was added into 1000 ml distilled water
2. The solution was aerated until the DO was at least 8 mg/l.
3. 5 ml and 10 ml amounts of raw waste water and each of the treated samples were placed into 250 ml beakers. One of the beakers was left as blank containing the aerated water.
4. Volumes in each of the beakers were made to 200 ml using aerated water.
5. Dissolved oxygen of each of the samples was measured using the method described in section 3.4.1 above and recorded as D_1.
6. After determining D_1, each of the solutions in the beakers was transferred into BOD bottles ensuring that there was no air trapped
7. The bottles were then covered with cellophane and incubated for 5 days at 20^0C

8. After 5 days the dissolved oxygen of the incubated samples was measured using the same method described in section 3.4.1 as D_2

BOD_5 (mg/l) was then calculated using equation 18;

$$BOD = \frac{D_1 - D_2}{p} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots..18$$

Where,

D1 = DO of the diluted sample immediately after preparation (mg/l)

D2 = DO of the diluted sample after 5 days of incubation at 20^{o}C for 5 days

p = Decimal volumetric fractions of sample used

$$\mathrm{p} = \frac{\text{volume of sample used}}{\text{volume of dilution} + \text{wastewater}} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots.\,.\ldots\ldots\ldots\ldots 19$$

3.3.3 Chemical Oxygen Demand

The standard laboratory method for determining COD was used.

50 ml of sample was placed into a refluxing flask and several boiling stones and 0.1g of mercuric sulphate added. To this solution, 5 ml of concentrated sulphuric acid (undiluted) was added. To ensure that the mercuric sulphate dissolved completely, the solution was swirled slowly while adding the sulphuric acid. To this, 0.1 g of Silver sulphate and Potassium dichromate was added and the solution swirled to ensure thorough mixing of the content. The flask was then placed in a water bath to recover any volatile substances that may have escaped from the liquid state. The flask was connected to a condenser to cool the contents. 20 ml of Sulphuric acid was slowly added to the flask as its contents continued to cool while swirling it to mix the solution. The solution was then refluxed for an hour.

A blank run (using 50 ml distilled water instead of sample) was conducted simultaneously using the same procedure.

After cooling, the solution was transferred to an Erlenmeyer flask. The reflux flask was rinsed thrice and the rinsing water poured to the Erlenmeyer flask. The solution was diluted to about 300 ml after which 8 drops of Phenanthroline ferrous sulphate was added to the solution as an indicator.

The solution was then titrated against 0.1N ferrous ammonium sulphate and the titre volume required for the colour change from blue-green to reddish blue recorded.

The procedure was repeated for the blank run and COD determined using the following formulae;

$$COD = \frac{(a-b)N \times 8000}{V} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 20$$

Where, a = volume of ferrous ammonium sulphate used for the blank (ml)

b = volume of ferrous ammonium sulphate used or the sample (ml)

N = volume of sample (ml)

8000 is the multiplier to express COD (mg/l)

3.3.4 Total Suspended Solids (TSS)

To determine TSS of wastewater, a 100 ml of a well-mixed sample was placed in a gooch crucible of known weight. The weight of the crucible together with the water sample was established and then placed in an oven set at 110ºC to dry. After drying, the crucible was placed in a dessicator to allow the contents to cool after which the weight of the crucible together with the dry contents was determined. The crucible was then returned to the oven and cooled repeatedly until a constant weight was achieved.

TSS in mg/l was calculated using the following formulae;

$$TSS, mg/l = \frac{Wd \times 10^6}{V} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots..21$$

Where, Wd = weight of dry content in grams

V = the volume of the sample in ml

3.3.5 Turbidity

Turbidity of the water was determined by use of the Lovibond photometer (in NTU).

3.3.6 Total Alkalinity

This was measured using the standard titration method.

The following reagents were used;

Sulphuric Acid Solution (0.02N) which was prepared using the following procedure;

a) 500 ml of distilled water was added into a 1000 ml standard flask
b) 20ml of concentrated 0.1N sulphuric acid was pipetted and slowly added to the standard flask containing the 500 ml water. This was diluted to the 1000 ml mark to make 0.02N H_2SO_4
c) Phenolpthalein Indicator which was prepared using by weighing 1g of phenolphthalein and diluting it in 100ml of distilled water.
d) Mixed Indicator which was prepared by dissolving 100mg of Bromocresol green and 20mg of methyl red in 100 ml of distilled water.

COD was tested using the following procedure;

1. The burette was filled with the 0.02N sulphuric acid, adjusted to zero and placed on a stand.
2. 50 ml of the sample to be tested was poured in a 250 ml conical flask.
3. Two drops of phenolphthalein indicator was added into the flask containing the waste or treated water until the colour turned pink. This was then titrated against 0.02N sulphuric acid until the pink colour disappeared. The titter value (V_1 was noted). This was used to calculate the phenolphthalein alkalinity.
4. A few drops of the mixed indicator were then added to the same solution which turned blue. Titration was continued till the solution turned red. The entire volume of second titration was recorded as V_2 and used to calculate the total alkalinity.

Total alkalinity was obtained using the formulae;

$$Alkalinity\left(mg\ \frac{C_aCo_3}{L}\right)=\frac{A\times N\times 50000}{Ml\ Sample}\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots 22$$

Where;

A = ml standard acid used (V_2)

N = normality of standard acid (0.02N)

Ml sample = volume of sample used (50ml)

3.3.7 pH and Temperature

These were measured using pH meter and a thermometer respectively.

3.3.8 Conductivity

This was measured by use of a conductivity meter (in µS/cm). Lovibond SD 70 Hand-held meter was used.

3.4 Data Analysis

The data was analyzed using statistical methods. Descriptive methods including means were used to provide a summary of the findings using bar graphs. A set of three samples were used for each characteristic. Quality control was ensured by using a laboratory that is certified by the Government of Kenya through NEMA. Precautions were taken to ensure that the various procedures for determining each characteristic are followed to the later and that the various samples were analysed immediately after treatment to avoid further degradation. ANOVA analysis was used to statistically examine the differences between all of the variables used in the experiment (electrode material, surface area and voltage) in order to compare performance of iron to that of aluminium using SPSS computer software version 19.

CHAPTER FOUR

RESULTS AND DISCUSSION

4.1 Introduction

This chapter covers data analysis, findings and discussions of the research. The study used 30 samples for iron electrodes and 12 samples for aluminium electrodes. Three trials for each run were used. The mean value was then considered as the reading for each test.

4.2 Wastewater Characterization

In order to characterize the wastewater, means of all the samples used in the analysis (30 samples of water for the Iron plates and 12 samples for aluminium plates) was used. Table 4.1 shows the characteristics of the untreated slaughterhouse wastewater in terms of BOD_5, COD, total alkalinity, conductivity, turbidity, TSS and pH before treatment. Also presented are the recommended standards for treated water by the World Health Organization (WHO) and the Government of Kenya (GoK) through NEMA.

Table 4.1: Characterization of Untreated Slaughterhouse Wastewater and Comparison with Government of Kenya (GoK) and WHO Standards for Treated Water.

PARAMETER	VALUES FOR RAW WASTEWATER SAMPLES	WHO STANDARDS	GoK (NEMA) STANDARDS
BOD_5 (mg/l)	1278.00 ±7.44 mg/l	5 mg/l	30 mg/l
COD (mg/l)	1425.00 ±14.59 mg/l	5 mg/l	50 mg/l
Total Alkalinity (mg/l)	258.00 ±76.43 mg/l	200mg/l	500 mg/l
Conductivity (µS/cm)	369.00 ±45.38 µS/cm	EC<500µS/cm	EC<500µS/cm
Turbidity NTU	978.00 ±10.30NTU	<1NTU	5NTU
TSS (mg/l)	405.00 ±2.61 mg/l	30 mg/l	30 mg/l
pH	6.30 ±0.10	6.5-8.5	6.5-8.5

Table 4.1 presents comparison of untreated slaughterhouse wastewater characteristics with the Kenya and WHO standards. The results indicate that COD, BOD_5, Turbidity, TSS, Total and Alkalinity of the wastewater were way above the recommended values for release into the environment while conductivity was found to lie within the Kenyan recommended range of not more than 500µS/cm. The pH of the water was found to be lower than is required by the standards. Lower pH indicates acidity of the wastewater due to presence of blood (Budiyono *et al.*, 2011). Therefore, the slaughterhouse effluent needed to be treated prior to discharge into the environment or before discharge to the receiving water.

The high BOD indicates that slaughterhouse wastewaters contain high organic matter. Thus the slaughterhouse wastewater is highly biodegradable. Such water if discharged directly into the environment without being treated would be harmful as it would cause deoxygenation of rivers and also contaminate ground water (Budiyono *et al.*, 2011). The wastewater was also found to contain high concentrations of total suspended solids which include grit, manure, hair, undigested feed, pieces of fat, skin and grease. These insoluble materials contribute to biodegradable organic matter in the wastewater.

These values however differ from those found by previous studies that have characterised wastewater from slaughterhouses. Bazrafshan, Mostafapour, Farzadkia, Ownagh and Mahvi, (2012) for example found higher values for five of the parameters; BOD_5 (2543mg/l), COD (5817mg/l), TSS (3247mg/l), pH (7.3) and Conductivity (9140µS/cm). Sarairah and Jamrah (2008) characterised wastewater from a slaughterhouse in Amman, Jordan and found the following values; BOD_5 (1235mg/l), COD (2144mg/l), TSS (863.57mg/l), pH (6.69) and conductivity of 1.47dS/m. Masse and Masse (2001) found the following values for wastewater from a slaughterhouse in Canada; COD (2941mg/l), TSS (2244mg/l), Alkalinity (333mg/l) and pH of 6.5. Seif and Moursy (2011) found slaughterhouse wastewater in Egypt to have the following characteristics: pH (6.8), TSS (400mg/l), COD (4400 (mg/l). Padilla-Gasca, López-López & Gallardo-Valdez (2011) analysed wastewater from a characterised municipal slaughterhouse in Ethopia and found the following values; pH (7.7), COD (4306mg/l), BOD (2733mg/l), TSS (1900mg/l) and alkalinity of 1320mg/l.

4.3 Effectiveness of Electrocoagulation in Treating Slaughterhouse Wastewaters

4.3.1 Effectiveness of Electrocoagulation in Reducing BOD_5 in Slaughterhouse Wastewater

Figure 4.1 is a graph of BOD concentration of the treated wastewater plotted against voltage for the different surface area of iron electrodes after two hours of treatment. The graph demonstrates decrease in BOD concentration with increasing electrode surface area and with increase in voltage. The recommended BOD_5 level was achieved after two hours at surface area of 100 cm^2 and 25V with iron electrodes. At this point, BOD_5 was reduced to 23.20 mg/l which is below the recommended NEMA standards of 30 mg/l.

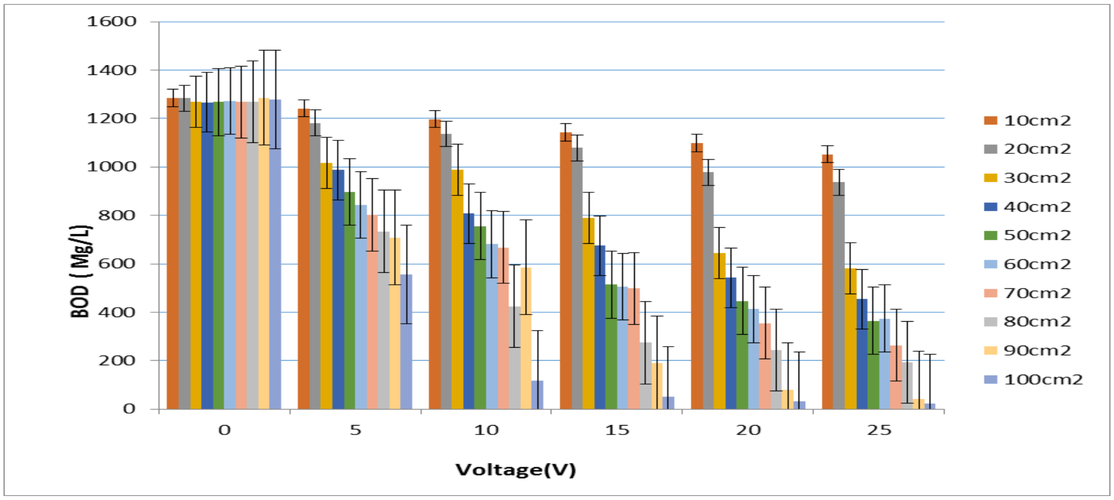

Figure 4.1: Graph of BOD against Voltage for the Different Surface area of Iron Plates after Two Hours

Figure 4.2 is a graph of BOD concentration plotted against voltage for the different surface area of aluminium electrodes after two hours. The graph indicates a sharp decrease in BOD concentration in the water being treated with increasing surface area of electrodes as well as with increase in voltage. The results demonstrate that the decrease was sharper when aluminium electrodes were used than when iron electrodes were used for the same surface area as well as voltage.

When Aluminium was used for electrodes, the recommended NEMA standard for BOD_5 of 30 mg/l was reached at surface area of 30 cm^2 and 20V. The optimum surface area and voltage for aluminium electrodes were found to be 4cm^2 and 25V respectively after two hours of treatment. At this point, BOD_5 of 20.78 mg/l was achieved.

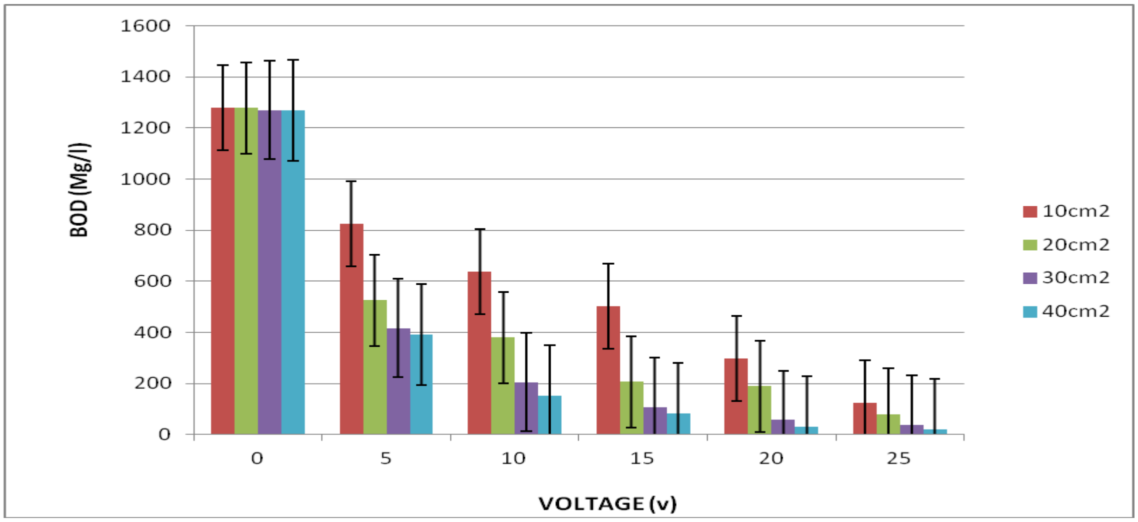

Figure 4.2: Graph of BOD against Voltage for the Different Surface Area of Aluminium Plates after Two Hours

The results from figures 4.1 and 4.2 indicate that electrocoagulation is effective in reducing BOD_5 in slaughterhouse wastewaters to levels that lie within the recommended standards. These findings are similar to those by previous studies including Bayar *et al*. (2011), Tezcan *et al*. (2009) and Asselin *et al*. (2008).

4.3.2 Effectiveness of Electrocoagulation in Reducing COD in Slaughterhouse Wastewater

Figure 4.3 is a graph of COD concentration of the water plotted against voltage for different surface area of iron electrodes after two hours.

Higher levels of chemical oxygen demand (COD) were recorded for the raw wastewater from the slaughterhouse. This is undesirable since continuous discharge of untreated effluent has negative impact on the receiving water body lowering the quality of the freshwater as well as causing harm to the aquatic life especially fish, downstream (Kanu & Achi, 2011).

Figure 4.3 shows that there was gradual decrease in COD concentration as the treatment process proceeded. COD concentration decreased with increase in electrode surface area and with increase in the applied voltage. According to the findings, when iron electrodes were used, the recommended NEMA standards of maximum 50 ml/g was obtained at surface area of 90 cm^2 and 25V (30 mg/l). Further decrease of COD was observed at 100 cm^2 and 25V giving a COD value of 18.67 mg/l. 90 cm^2 and 25V were found to be the optimum surface area and voltage for COD removal using iron electrodes.

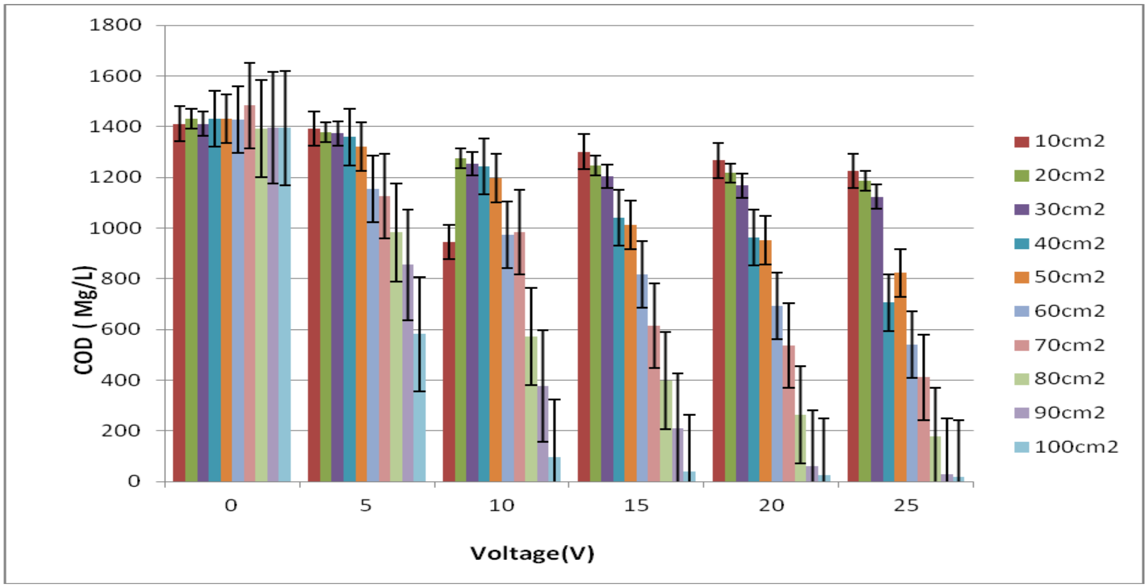

Figure 4.3: Graph of COD against Voltage for the Different Surface Area of Iron Plates after Two Hours

Figure 4.4 is a graph of COD concentration of the water plotted against voltage for different surface area of aluminium electrodes after two hours. The graph shows that there was decrease in COD concentration in the slaughterhouse wastewater with increasing surface area as well as voltage. The results further demonstrate that the decrease was sharper when aluminium electrodes were used compared to iron electrodes for the same surface area and same voltage.

When Aluminium was used for electrodes, the recommended NEMA standards for COD (50 mg/l) were attained at a much smaller surface area of 30 cm^2 and 25V. COD value of 16.33 mg/l was achieved at surface area of 40 cm^2 and 25V.

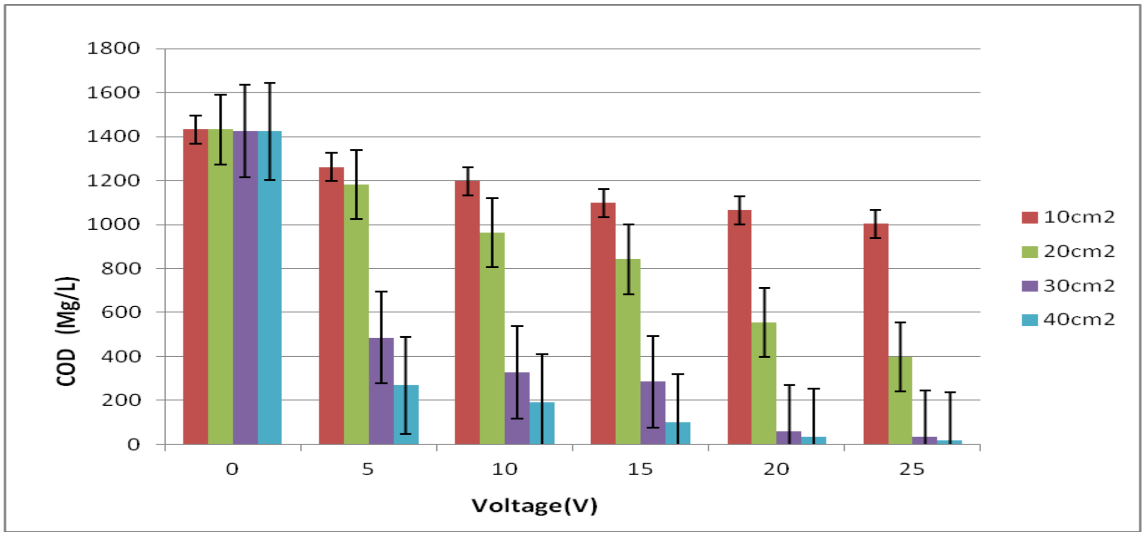

Figure 4.4: Graph of COD against Voltage for the Different Surface Area of Aluminium Plates after Two Hours

The results from figures 4.3 and 4.4 demonstrate that electrocoagulation is very effective in reducing COD in slaughterhouse wastewaters to levels that lie within the recommended standards. These findings are similar to those found by previous studies including Eryuruk *et al*. (2011), Tezcan et al. (2009), and Asselin *et al*. (2008) who found that electrocoagulation was 98% effective in treating slaughterhouse wastewaters of COD.

4.3.3 Effectiveness of Electrocoagulation in Treating Slaughterhouse Wastewater of Total Alkalinity

Results of effectiveness of EC in treating wastewater from slaughterhouses of total alkalinity are presented in figures 4.5 and 4.6.

Figure 4.5 is a graph of total alkalinity plotted against voltage for the different surface area of iron electrodes after two hours of treatment for iron electrodes while figure 4.6 is a graph of total alkalinity plotted against voltage for the different surface area of iron electrodes after two hours of treatment for aluminium electrodes.

Graphs in figures 4.5 and 4.6 demonstrate unusual pattern in the values of total alkalinity as the experiments proceeded. The Total alkalinity values fluctuated throughout the experimental period, though relatively satisfactory values were observed throughout the follow-up process. In Figure 4.5 which presents results obtained with iron electrodes, there was an initial increase in Total Alkalinity of the wastewater at surface area of 10 cm^2 as the experiment began at 5V. These values however decrease gradually with increase in surface area and voltage until at surface area of 100 cm^2 and 25V where a value of 136 mg/L $CaC0_3$ which is within the standards was attained. Figure 4.6 also demonstrates a similar pattern whereby there was an initial sharp increase in total alkalinity at surface areas of 10 cm^2 and 20 cm^2 at low voltages. These values however decreased sharply as surface and voltage is increased until a value of 120.67 mg/L $CaC0_3$ which was within the accepted NEMA standard values was achieved at 40 cm^2 and 25V.

Results presented in the two figures above demonstrate that there was an initial increase in Total Alkalinity of the waste water with increase in voltage and surface which later decreased. Previous studies on EC as a method for treating wastewater have not considered total alkalinity as a parameter. However, Wang, Li, Su, and Sun (2010) also discovered that there was a relationship between an increase of pH and alkalinity with increase in temperature, and electrolysis time.

In the current study, the initial increase is attributed to increased OH^- ions in the wastewater due to action of current on the electrodes before formation of coagulants as per equation (9). Formation of CO_3^- ions due to presence of organic matter in the wastewater is also another possible explanation. Higher dissociation constants force a higher concentration of bicarbonate and carbonate to be present for a given concentration of carbonic acid. Hence, they result in a higher alkalinity as explained by Rajakumar, Meenambal, Saravanan, and Ananthanarayanan (2012). As pH increases, the rate of hardness and total alkalinity removal also increase as the effect of pH on coagulants depends on the produced reactions on different conditions (Malakootian & Yousefi, 2009). Evidenced literature demonstrates that there is strong correlation between alkalinity and partial acid build-up with the composition of the wastewater. Protein-rich effluents, such as those from slaughterhouses, the acids accumulate in the same proportion, but pH and total alkalinity tend to increase due, probably, to the formation of ammonia during anaerobic degradation of proteins (Rajakumar *et al.*, 2012).

The findings indicate that electrocoagulation process is effective in treating total alkalinity in slaughterhouse wastewater to the recommended levels of 130-200 mg/l. These findings are similar to those found by Malakootian and Yousefi (2009) who found electrocoagulation to be effective in removal of alkalinity and harness from water.

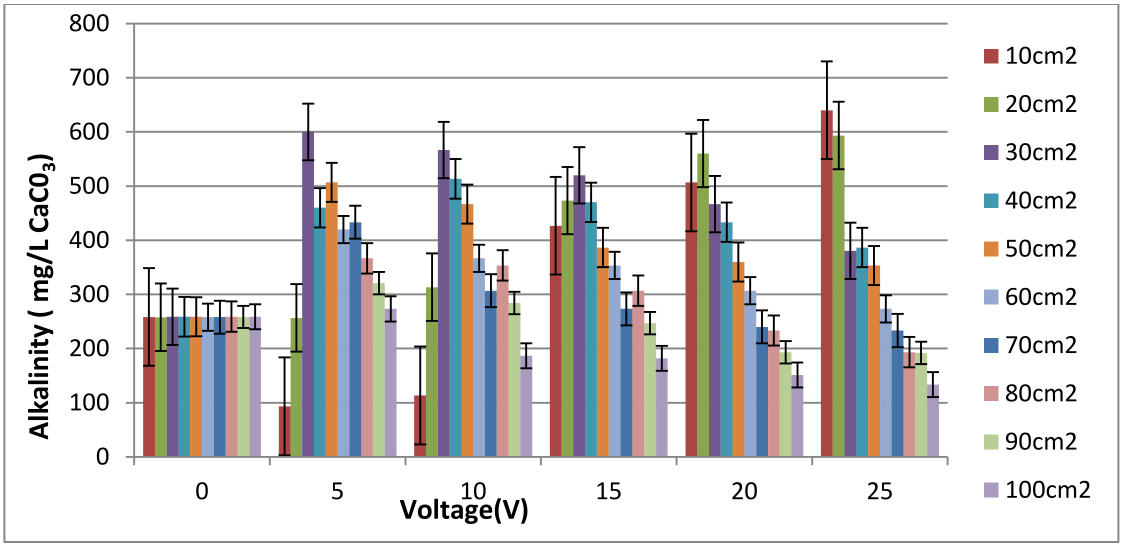

Figure 4.5: Graph of Total Alkalinity against Voltage for the Different Surface Area of Iron Electrodes after Two Hours

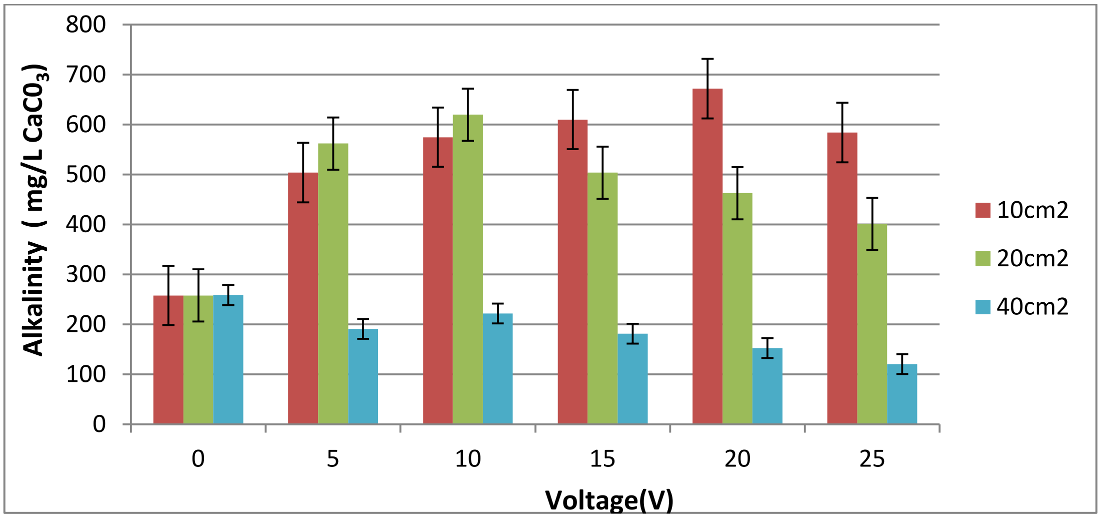

Figure 4.6: Graph of Total Alkalinity against Voltage for the Different Surface Area of Aluminium Plates after Two Hours

4.3.4 Effectiveness of Electrocoagulation in Treating Slaughterhouse Wastewater of Total Suspended Solids (TSS)

The results obtained when effectiveness of electrocoagulation in treating slaughterhouse wastewater of total suspended solids (TSS) was examined are presented in figures 4.7 and 4.8.

Figure 4.7 is a graph of TSS concentration in the water plotted against voltage for the different surface area of iron electrodes after two hours of treatment. The graphs demonstrate that when Iron was used as electrode material, there was a very sharp decrease in TSS in the wastewater with increase in surface area of the electrode as well as with voltage from the initial value of 405 mg/l. Increase in current and surface area increased the process. The recommended NEMA standards of 30 mg/l was achieved at surface area of 80 cm^2 and 15V when Iron was used as electrode material. TSS value of 6.0 mg/l was achieved at surface area of 100 cm^2 and 25V.

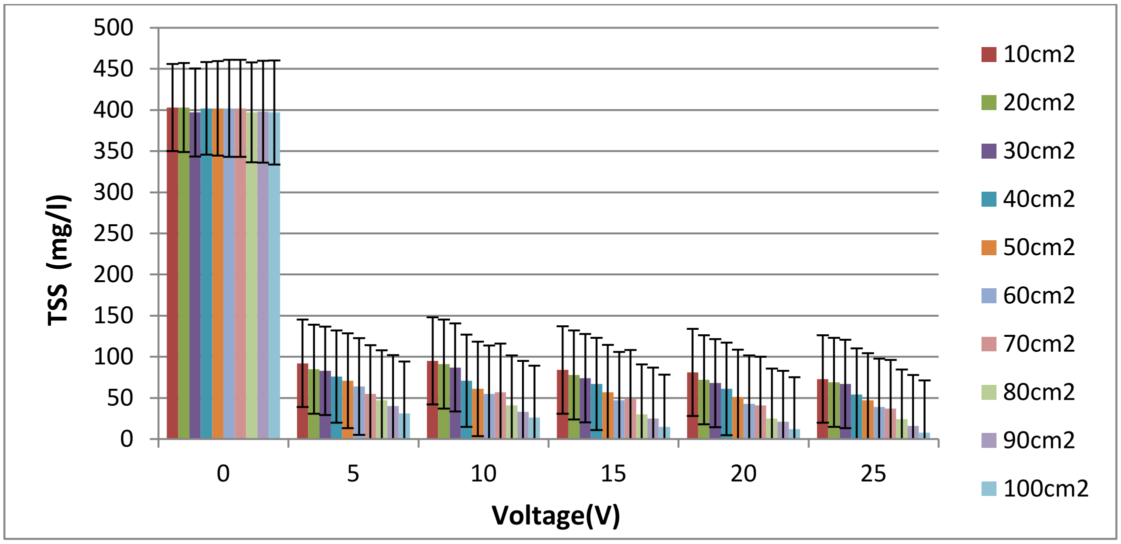

Figure 4.7: Graph of Total Suspended Solids against Voltage for the Different Surface Area of Iron Plates after Two Hours

Figure 4.8 is a graph of TSS concentration in the water plotted against voltage for the different surface area of iron electrodes after two hours of treatment. Figure 4.8 indicates that when Aluminum was used as electrode material, there was also a very sharp decrease in amount of TSS in the wastewater. The concentration of TSS decreased as the aluminium electrode surface area was increased as well as with increase in voltage. The observed decrease with aluminium electrodes was much sharper than that observed when Iron electrodes were used. Untreated wastewater was found to have high concentration of TSS (406 mg/l). Treatment of the wastewater using aluminium and iron reduced the TSS concentration so that it was within the recommended NEMA standards. TSS was removed from the water by direct adsorption of the pollutants on the electrodes, precipitation, flocculation and aggregation of the particulates.

The recommended NEMA standard of 30 mg/l was reached at surface area of 30 cm^2 (3 plates) and 25V when aluminium was used as electrode material. TSS of 2 mg/l was achieved at 40 cm^2 and 25V, implying an almost 100 percent removal efficiency.

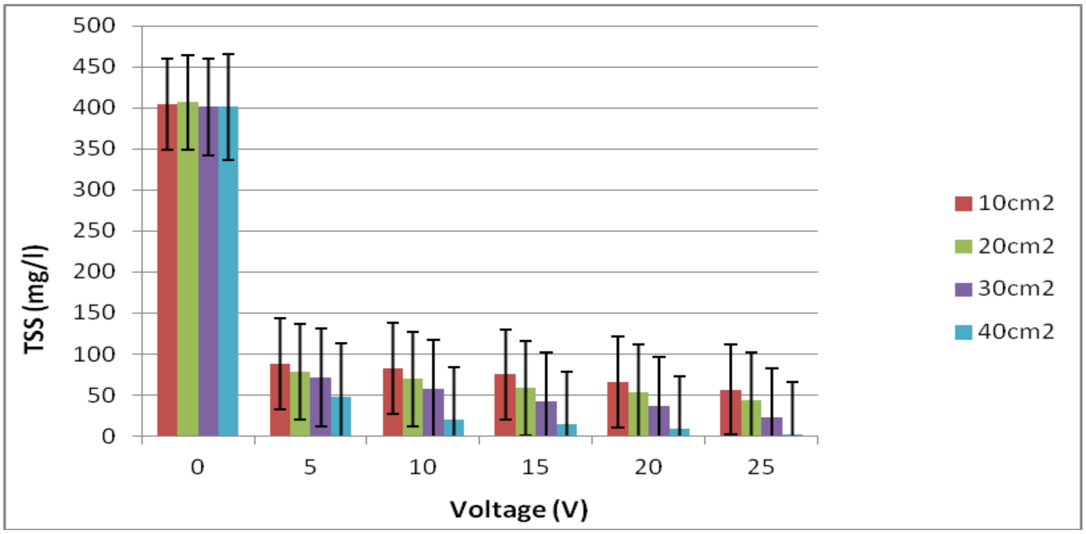

Figure 4.8: Graph of Total Suspended Solids against Voltage for the Different Surface Area of Aluminium Plates after Two Hours

These findings illustrated in figures 4.7 and 4.8 are consistent with those established by Bazrafshan *et al.*, (2012) who conducted an investigation of treatment of slaughterhouse wastewater using combined chemical coagulation and electrocoagulation process. Removal efficiency of 65% was registered in this study. Asselin *et al*., (2008) also found the TSS was removed at 89% using the EC method.

The results indicate that aluminium performed significantly better than iron for the same surface area as well as voltage in removing TSS hence is a better electrode than Iron.

4.3.5 Effectiveness of Electrocoagulation in Reducing Turbidity in Slaughterhouse Wastewater

When examination of effectiveness of electrocoagulation in reducing turbidity in slaughterhouse wastewater was examined using Iron and aluminium electrodes, the results obtained are as are presented in figures 4.9 and 4.10.

Figure 4.9 is a graph of turbidity plotted against voltage for the different surface area of iron electrodes after two hours of treatment. This graph demonstrates that when Iron was used as electrode material, there was a gradual decrease in the wastewater's turbidity from the initial value of 978 NTU. Turbidity was observed to decrease with increase in surface area of the iron electrode as well as with increase in voltage.

The water became very clear at surface area of 90 cm^2 and 25V giving a turbidity 30 NTU. At surface area of 100 cm^2 and 25V Turbidity of 13 NTU was achieved. This value is within the acceptable standards of NTU as required by NEMA for effluent discharged into the environment.

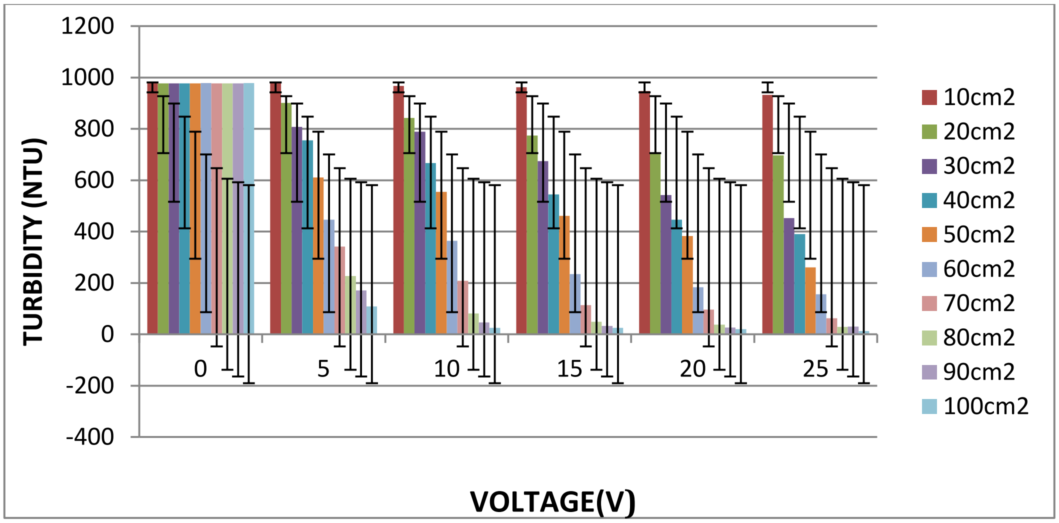

Figure 4.9: Graph of Turbidity against Voltage for the Different Surface Area of Iron Plates after Two Hours

Figure 4.10 is a graph of turbidity plotted against voltage for the different surface area of aluminium electrodes after two hours of treatment. It indicates that when Aluminum was used as electrode material, there was a gradual decrease in Turbidity in the wastewater with increase in surface area as well as current. The decrease observed was much sharper than that observed with Iron electrodes of same surface area and at same voltage. The water became very clear at surface area of 30 cm^2 and 25V giving a turbidity 20.67 NTU. Turbidity of 2 NTU was achieved at surface area of 40 cm^2 and 25V. This value meets the recommended NEMA standards of a maximum value of 15 NTU.

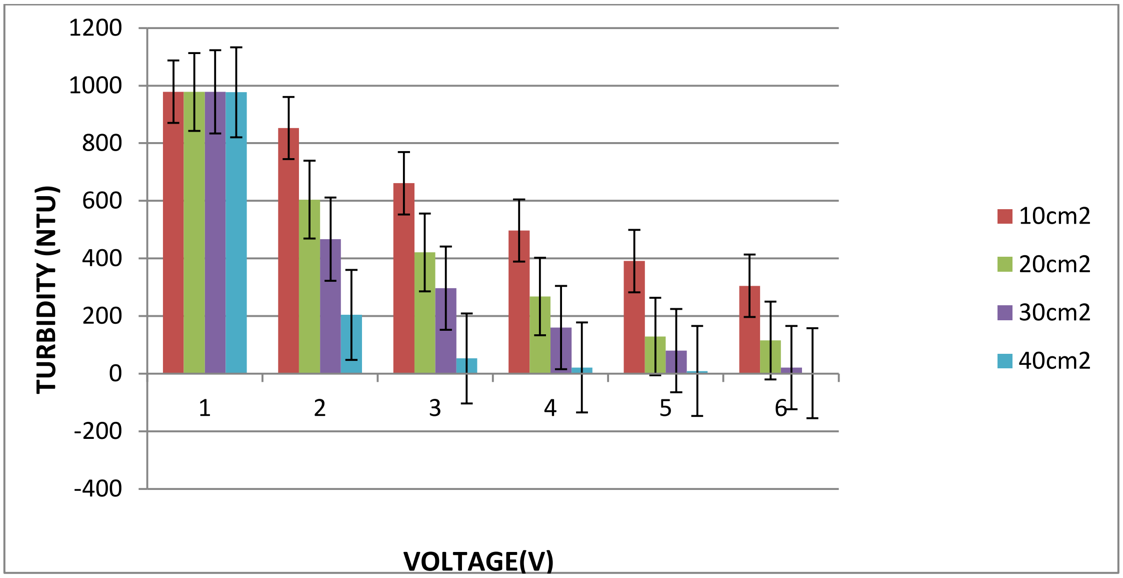

Figure 4.10: Graph of Turbidity against Voltage for the Different Surface Area of Aluminium Plates after Two Hours

The turbidity profile varied significantly amongst different surface areas and voltage/current for both aluminium and iron electrodes throughout the study. The turbidity values obtained from the sampling points was higher than WHO standard of 15 NTU. The values obtained using Aluminium of surface area of 40 cm^2 and Iron of surface area of 100 cm^2 at 25V qualify the treated water for direct domestic use as they lie within the recommended WHO standards. The results also indicate that aluminium performed significantly better than Fe for the same surface area as well as current in removing TSS hence is a better electrode than Iron.

Kuokkanen *et al*. (2013) explain that Turbidity in water is as a result of presence of suspended matter, in this case, clay, finely divided organic and inorganic matter, silt and other microscopic organisms. Very high removal efficiencies were achieved rapidly for both iron and aluminium electrodes and at low current just as was observed in the study conducted by Kuokkanen *et al*. (2013). Electrocoagulation has been found to remove turbidity up to 99% with low energy consumption as demonstrated by Terrazas, Vázquez, Briones, Lázaro, and Rodríguez (2010) and, Kuokkanen *et al*. (2013). The findings of the current study are therefore consistent with those from other studies on EC and removal of turbidity from wastewater. The decrease in turbidity values demonstrates that EC is effective in treating slaughterhouse wastewaters of suspended solids.

4.3.6 Effectiveness of Electrocoagulation in Reducing Conductivity in Slaughterhouse Wastewater

In order to determine the effectiveness of EC in reducing the conductivity of the slaughterhouse wastewater, the results presented in figures 4.11 and 4.12 were obtained.

Figure 4.11 is a graph of conductivity plotted against voltage for the different surface area of iron electrodes after two hours of treatment for iron electrodes. The figure demonstrates that when iron was used as electrode material, there was rapid decrease in conductivity of the wastewater being treated with increase in surface area as well as with increase in current. The acceptable limit for conductivity in domestic water supply is 70 µs/cm (DWAF, 1996). This limit was well achieved and surpassed using the EC method at surface area of 100 cm^2 and 25V as conductivity of 65.67 µs/cm was achieved.

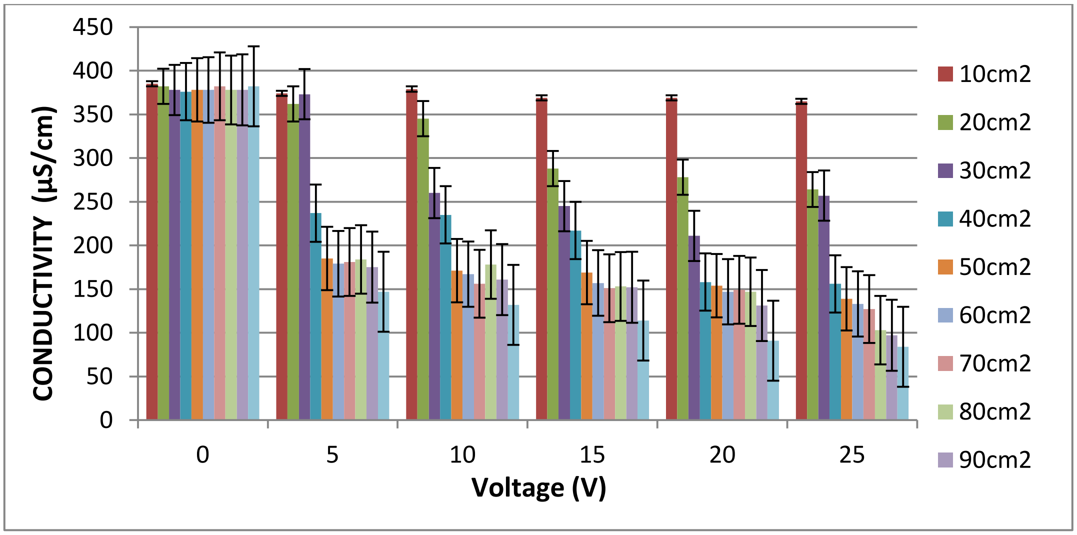

Figure 4.11: Graph of Electrical Conductivity against Voltage for the Different Surface Area of Iron Plates after Two Hours

Figure 4.12 is a graph of conductivity plotted against voltage for the different surface area of aluminium electrodes after two hours of treatment. It demonstrates that when Aluminum was used as electrode material, there was a rapid decrease in conductivity the wastewater being treated with increase in surface area and current. The decrease was much sharper than observed with Iron electrodes. The FEPA acceptable limit for conductivity in domestic water supply of 70 µs/cm was exceeded at surface area of 40 cm^2 and 25V, 62.47 µs/cm.

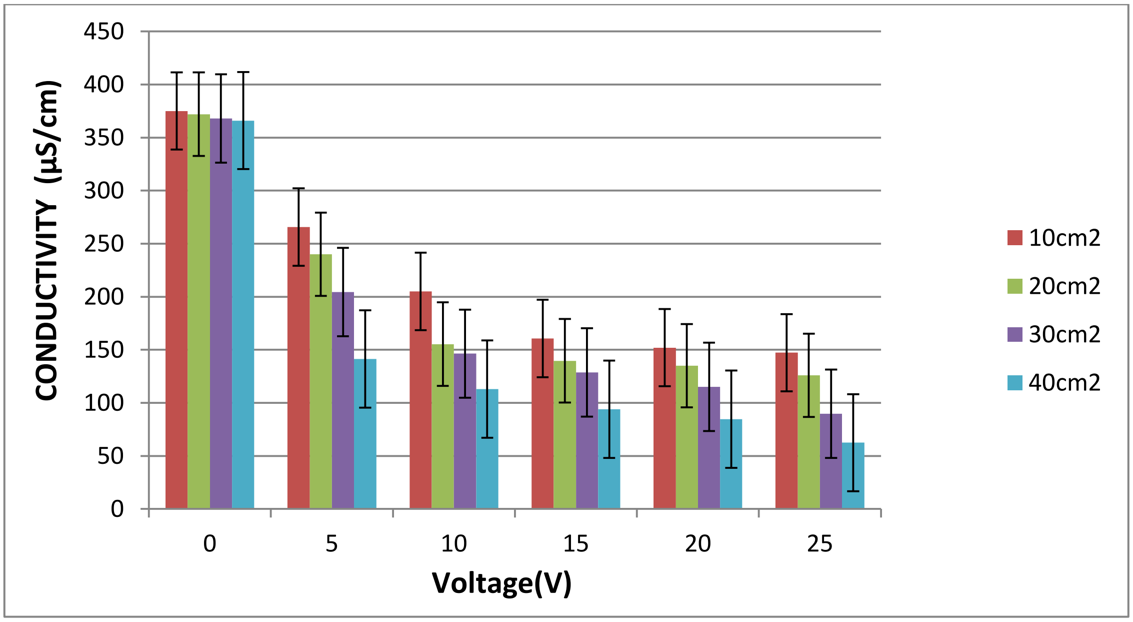

Figure 4.12: Graph of Conductivity against Voltage for the Different Surface Area of Aluminium Plates after Two Hours

Electrical conductivity is a function of ions concentration, which determines the quality of water (Tariq, Ali, & Shah, 2006). According to these authors, electric conductivity is a measure of how much total salt (inorganic ions such as sodium, chloride, magnesium, and calcium) is present in the water. The higher the concentration of these ions, the higher the higher the conductivity. Conductivity can serve as an indicator of other water quality problems. If the conductivity of a stream suddenly increases, it indicates that there is a source of dissolved ions in the vicinity.

The results demonstrate that Aluminium performed significantly better than iron for the same surface area as well as current in reducing conductivity hence is a better electrode.

The electrical conductivity of the treated water samples generally varied with surface area and current density. The raw wastewater was observed to have high conductivities The FEPA acceptable limit for conductivity in domestic water supply is 70 µs/cm (DWAF, 1996). This limit was exceeded in the final treated water by both Aluminium and Iron electrodes. The conductivity values obtained in the current study are similar to the findings of previous studies by Budiyono & Johari (2010) and Eryuruk *et al*. (2011).

4.3.7 pH

The pH of raw wastewater was found to be slightly acidic at about 6.3. It then increased steadily with increase in surface area and voltage/current for both Al and Fe electrodes until the standard value of 7.0-7.5 was achieved.

Previous studies demonstrate that the wastewater's initial pH is a key operating factor that influences EC performance (Chen *et al*., 2000; Adhoum & Monser, 2004). According to these studies, the removal yields of COD and turbidity are dependent on the initial pH. These studies further explain that maximum removal of COD and turbidity is observed at neutral pH 6-7. Studies also show that the more acidic the initial liquid (raw wastewater) is, the higher is the increase in pH during the run. This phenomenon was ascribed to hydrogen evolution at cathode and according equation (5) (Vik *et al*., 1984).

Generally, the obtained pH value for the treated wastewater using both Al and Fe fall within the World Health Organization standard of 7.0 to 8.5 and the water quality ranges 6.5 to 8.5 for drinking water and water meant for full contact recreation, respectively (DWAF, 1996).

4.3.8 Efficiency Levels Achieved

For this study, 25V is chosen as optimum operating voltage for the electrocoagulation process, while 100 cm^2 and 40 cm^2 are chosen as the optimum electrode surface area for iron and aluminium plates respectively. Although these did not give removal efficiency of 100%, they gave values that were within the permissible limits by NEMA standards for domestic water. A summary of the findings comparing the values of the various characteristics before and after treatment for the EC process for the two electrodes are presented in Table 4.2.

Table 4.2 provides a summary of the results indicating the values obtained after treatment process for aluminium electrodes surface area of 40 cm^2 at 25V and iron electrodes of surface area of 100 cm^2 at 25V. The values demonstrate that elecrocoagulation process was able to treat the wastewater of the various characteristics to levels that meet NEMA and WHO standards.

Table 4.2: Summary of the Results before and after Treatment with Aluminium and Iron Electrodes at 40 cm^2 and 100 cm^2 at 25V

PARAMETER	BEFORE TREATMENT	NEMA GUIDELINES	AFTER TREATMENT	
			AL ELECTRODES	Fe ELECRODES
BOD$_5$	1278.00±7.44 mg/l	30 mg/l	20.78 mg/l	23.2 mg/l
COD	1425.00±14.59 mg/l	50 mg/l	16.33 mg/l	18.67 mg/l
TOTAL ALKALINITY	258.00±76.43 mg/l	500 mg/l	120.67 mg/l	136 mg/l
CONDUCTIVITY	369.00±45.38 µS/cm	EC<500 µS/cm	62.67 µS/cm	65.67 µS/cm
TURBIDITY	978.00±10.30 NTU	5 NTU	2 NTU	13 NTU
TSS	405.00±2.61 mg/l	30 mg/l	2.33 mg/l	6 mg/l
pH	6.30±0.10	6.5-8.5	7.2	7.3

Based on these findings, efficiency levels were calculated and the values presented in Table 4.3.

Table 4.3 presents results of efficiency levels achieved at the end of treatment process for each of the parameters that were being measured for iron and aluminium electrodes. The table demonstrates that high efficiency levels were obtained with both aluminium and iron electrodes (more than 98% for most of the parameters part from conductivity, where efficiency levels of 82.2% and 85% were obtained with Iron and aluminium electrodes respectively and total alkalinity which gave the lowest efficiency level.

The results demonstrate that aluminium is a superior material for electrodes compared to Iron as it yielded higher efficiency levels for removal of all the parameters that were being examined. The efficiency levels were calculated using the following formulae;

$$Efficiency\ (\%) = \frac{C_O - C}{C_O} \times 100 \qquad\text{.......... 23}$$

Where; C_o is original value before treatment and C is value after treatment.

Total alkalinity was not easily treated using the electroacoagulation process because of the high levels of $[OH^-]$ ions produced during the process which increase alkalinity of the solution. These findings are consistent with those obtained by Kuokkanen *et al*. (2013); Terrazas *et al.* (2010); Bazrafshan *et al.* (2012); and Asselin *et al*. (2008) who found that electroagulation registered high removal efficiency of Turbidity, BOD, COD, conductivity, alkalinity , colour and TSS.

Table 4.3: Efficiency Levels Obtained after Treatment with Aluminium (Al) and Iron (Fe) Electrodes

PARAMETERS	**EFFICIENCY (%)**	
	Al ELECTRODES	**Fe ELECTRODES**
BOD	98.37	98.18
COD	98.9	98.73
TOTAL ALKALINITY	53.22	47.28
CONDUCTIVITY	83	82.2
TURBIDITY	99.98	98.67
TSS	99.42	98.5

4.4 Comparison of Performance of Aluminium and Iron as Electrodes during Electro-Coagulation Treatment of Wastewater from Slaughterhouses

In order to compare performance of aluminium to that of iron, ANOVA analysis was conducted to statistically examine the differences between all of the variables used in the experiment (electrode material, surface area and voltage). The analyses investigated whether the quality of the wastewater from Gachororo slaughterhouse treated using aluminium electrodes is different from wastewater treated using iron electrodes.
The following results were obtained for each parameter at different surface area and voltage.

4.4.1 Total Alkalinity

The study formulated the following hypothesis;

$H1_0$: The alkalinity of wastewater treated using aluminium electrodes is not significantly different from the alkalinity of wastewater treated using iron electrodes at p =0.05

This hypothesis was tested at different surface areas to determine the performance of each electrode with regards to the quality of the wastewater as presented below.

4.4.1.1 Total Alkalinity at Surface Area of 10 cm^2

Table 4.4 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing alkalinity at surface area of $10cm^2$. Since the obtained p-value=0.324 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of Iron and Aluminum at $10cm^2$ for the observed conditions (type of electrode and surface area). The observed difference at this surface area is therefore due to effect of current on the different types of electrodes.

Table 4.4 Difference in Alkalinity at 10 cm^2 for Aluminium and Iron

ANOVA

Mean value

	Sum of Squares	**Df**	**Mean Square**	**F**	**Sig.**
Between Groups	34847.430	1	34847.430	1.077	0.324
Within Groups	323629.587	10	32362.959		
Total	358477.017	11			

4.4.1.2 Total Alkalinity at Surface Area of 20 cm^2

Table 4.5 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing alkalinity at surface area of 20cm^2. Since the obtained p-value=0.782 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of Iron and Aluminum at surface area of 20cm^2 for the observed conditions (element and surface area). The observed difference is therefore due to effect of current on the different electrode materials.

Table 4.5: Difference in Alkalinity at 20 cm^2 for Aluminium and Iron

ANOVA

Mean value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	2133.067	1	2133.067	0.081	0.782
Within Groups	264172.082	10	26417.208		
Total	266305.148	11			

4.4.1.3 Total Alkalinity at Surface Area of 30 cm^2

Table 4.6 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing alkalinity at surface area of 30 cm^2. Since the obtained p-value=0.154 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum at three plates for the observed conditions (element and surface area). The observed difference is therefore due to effect of current on the different electrode materials.

Table 4.6: Difference in Alkalinity at Three Plates for Aluminium and Iron

ANOVA

Mean value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	28356.481	1	28356.481	2.374	0.154
Within Groups	119457.841	10	11945.784		
Total	147814.322	11			

4.4.1.4 Alkalinity at Surface Area of 40 cm^2

Table 4.7 presents ANOVA analysis that measured difference in performance of iron and Aluminum in removing alkalinity at surface area of 40 cm^2. Since the obtained p-value=0.000 is less than 0.05, the study's null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminum at surface area of 40cm^2. This implies that as surface area is increased, efficiency of Aluminium becomes better than that of Iron. This is attributed to the fact that Al is a better adsorbent than Iron hence performs better at large surface area. It can therefore be argued that surface area contributes significantly to the differences observed in Alkalinity removal at 4 plates (surface area of 40 cm^2) and that it is not only due to action of current on the electrodes.

Table 4.7: Difference in Alkalinity at 40 cm^2 Aluminium and Iron

ANOVA

Mean value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	275930.210	1	275930.210	41.200	0.000
Within Groups	66973.934	10	6697.393		
Total	342904.143	11			

ANOVA analysis shows that there is no significant difference in performance of Iron and Aluminum plates in reducing alkalinity of wastewater al lower surface area (up to 30 cm^2). The difference in the performance at this stage is therefore due to action of current except at surface area of 40 cm^2 which show significant difference in performance of the two metals. With regards to reduction of Alkalinity of wastewater, it can be concluded that Aluminum is more efficient than Iron as an electrode. Therefore, the alkalinity of wastewater treated using an aluminium plate is significantly different from the alkalinity of wastewater treated using iron electrodes.

4.4.2. Biological Oxygen Demand (BOD)

The study formulated the following hypothesis

$H2_0$: The biological oxygen demand of wastewater treated using aluminium electrodes is not significantly different from the biological oxygen demand of wastewater treated using an iron electrodes at p =0.05

This hypothesis was tested at different surface areas to determine the performance of each electrode with regards to quality of the treated water as presented below.

4.4.2.1 BOD_5 at Surface Area of 10 cm^2

Table 4.8 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing BOD_5 at surface area of 10 cm^2. Since the obtained p-value = 0.008 is less than 0.05, the formulated null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminum at surface area of 10 cm^2 in removing BOD.

Table 4.8: Difference in BOD Removal at 10cm^2 for Aluminium and Iron

ANOVA
Mean value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	937370.331	1	937370.331	10.699	0.008
Within Groups	876123.390	10	87612.339		
Total	1813493.721	11			

4.4.2.2 BOD_5 at Surface Area of 20 cm^2

Table 4.9 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing BOD_5 at surface area of 20 cm^2. Since the obtained p-value=0.001 is less than 0.05, the hypothesized null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminum at surface area of 20 cm^2 in removing BOD.

Table 4.9: Difference in BOD_5 Removal at 20 cm^2 for Aluminium and Iron

ANOVA
Mean value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	2123777.741	1	2123777.741	22.090	0.001
Within Groups	961424.331	10	96142.433		
Total	3085202.072	11			

4.4.2.3 BOD_5 at Surface Area of 30 cm^2

Table 4.10 presents ANOVA analysis that measured difference in performance of iron and aluminium in removing BOD_5 at surface area of 30 cm^2. Since the obtained p-value=0.001 is less than 0.05, the formulated null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminium at surface area of 30 cm^2 in removing BOD.

Table 4.10: Difference in BOD_5 Removal at 30 cm^2 for Iron and Aluminium

ANOVA **Mean value**					
	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	851680.801	1	851680.801	5.884	0.036
Within Groups	1447410.137	10	144741.014		
Total	2299090.937	11			

4.4.2.4 BOD_5 at Surface Area of 40 cm^2

Table 4.11 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing BOD_5 at surface area of 40 cm^2. Since the obtained p-value=0.073 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant statistical difference in performance of Iron and Aluminum at surface area of 40 cm^2 in removing BOD for the observed conditions (element and surface area). The observed difference is therefore due to effect of current on the different types of electrodes.

Table 4.11: Difference in BOD_5 removal at 40 cm^2 for Iron and Aluminium

ANOVA **Mean value**					
	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	648572.704	1	648572.704	4.008	0.073
Within Groups	1618007.783	10	161800.778		
Total	2266580.487	11			

4.4.3 Chemical Oxygen Demand (COD)

The study sought to test the hypothesis;

$H3_0$: COD of wastewater treated using aluminium electrodes is not significantly different from the COD of wastewater treated using iron electrodes at p =0.05

This hypothesis was tested at different surface areas to determine the performance of each electrode with regards to quality of the treated water as presented below.

4.4.3.1 COD at Surface Area of 10 cm^2

Table 4.12 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing COD at surface area of 10 cm^2. Since the obtained p-value=0.061 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant statistical difference in performance of Iron and Aluminum at surface area of 10 cm^2 in removing COD for the observed conditions (element /type of electrode and number of plates (surface area). The observed difference is therefore due to effect of current on the different types of electrodes.

Table 4.12: Difference in COD Removal at 10 cm^2 for Iron and Aluminium

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	66156.750	1	66156.750	4.469	0.061
Within Groups	148031.285	10	14803.128		
Total	214188.035	11			

4.4.3.2 COD at Surface Area of 20 cm^2

Table 4.13 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing COD at surface area of 20 cm^2. Since the obtained p-value=0.036 is less than 0.05, the formulated null hypothesis is rejected and a conclusion made that there is significant difference in performance of Iron and Aluminum at surface area of 20 cm^2 in removing COD.

Table 4.13: Difference in COD Removal at 20 cm^2 for Iron and Aluminium

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	469262.068	1	469262.068	5.883	0.036
Within Groups	797692.767	10	79769.277		
Total	1266954.835	11			

4.4.3.3 COD at Surface Area of 30 cm^2

Table 4.14 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing COD at surface area of 30 cm^2. Since the obtained p-value=0.003 is less than 0.05, the formulated null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminum at surface of 30 cm^2 in removing COD.

Table 4.14: Difference in COD Removal at 30 cm^2 for Iron and Aluminium

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	2011726.119	1	2011726.119	14.522	0.003
Within Groups	1385332.265	10	138533.227		
Total	3397058.384	11			

4.4.3.4 COD at Surface Area of 40 cm^2

Table 4.15 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing COD at surface area of 40 cm^2. Since the obtained p-value=0.005 is less than 0.05, the formulated null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminium at surface of 40 cm^2 in removing COD.

Table 4.15: Difference in COD Removal at 40 cm^2 for Iron and Aluminium

ANOVA
Mean Value

	Sum of Squares	**Df**	**Mean Square**	**F**	**Sig.**
Between Groups	2091683.350	1	2091683.350	12.622	0.005
Within Groups	1657119.530	10	165711.953		
Total	3748802.880	11			

4.4.4 Conductivity

The study sought to address the following hypothesis

> $H4_0$: The conductivity of wastewater treated using aluminium electrodes is not significantly different from the conductivity of wastewater treated using iron electrodes at p =0.05

This hypothesis was tested at different surface areas to determine the performance of each electrode with regards to quality of the treated water as presented below.

4.4.4.1 Conductivity at Surface Area of 10 cm^2

Table 4.16 presents ANOVA analysis that measured difference in performance of iron and aluminium in removing COD at surface area of 40 cm^2. Since the obtained p-value=0.7981 is greater than 0.05, the study's null hypothesis is accepted and a conclusion made that there is no significant difference in performance of Iron and Aluminum at surface area of 10 cm^2 in reducing conductivity of waste water for the observed conditions (type of electrode and surface area). The observed difference is therefore due to effect of current on the different types of electrodes.

Table 4.16: Difference in Conductivity values at 10 cm^2 for Iron and Aluminium when Voltage/current is considered

ANOVA
Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	9734.948	5	1946.990	0.126	0.981
Within Groups	93045.016	6	15507.503		
Total	102779.964	11			

Table 4.17 presents ANOVA analysis that measured difference in performance of iron and aluminum in treating conductivity at surface area of 10 cm^2. Since the obtained p-value=0.000 is less than 0.05, the formulated null hypothesis is rejected and a conclusion is made that there is significant difference in performance of iron and aluminium at 10cm^2 in reducing conductivity of wastewater. The difference in performance of Iron and aluminium at 10 cm^2 is due to material used rather than due to action of varying current.

Table 4.17: Difference in Conductivity values at 10 cm^2 for Iron and Aluminium when Type of Electrode is considered

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	87835.896	1	87835.896	58.776	0.000
Within Groups	14944.067	10	1494.407		
Total	102779.964	11			

4.4.4.2 Conductivity at Surface Area of 20 cm^2

Table 4.18 presents ANOVA analysis that measured difference in performance of iron and aluminum in treating conductivity at surface area of 10 cm^2 when electrode material is considered. Since the obtained p-value=.0.001 is less than 0.05, the formulated null hypothesis is rejected and a conclusion is made that there is significant difference in performance of Iron and Aluminum at surface area of 20 cm^2 in decreasing conductivity of waste water.

Table 4.18: Difference in Conductivity values at $10cm^2$ for Iron and Aluminium when Type of Electrode is considered

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	60445.149	1	60445.149	20.256	0.001
Within Groups	29840.913	10	2984.091		
Total	90286.062	11			

Table 4.19 presents ANOVA analysis that measured difference in performance of iron and aluminum in treating conductivity at surface area of 20 cm^2 when voltage is considered. Since the obtained p-value=0.737 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum at surface area of 20 cm^2 in reducing conductivity of waste water for the observed conditions (electrode and surface area). The observed difference is therefore due to effect of current on the different types of electrodes.

Table 4.19: Difference in conductivity values at 20 cm^2 for Iron and Aluminium when voltage/current is considered

ANOVA
Mean Value

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	28309.860	5	5661.972	0.548	0.737
Within Groups	61976.202	6	10329.367		
Total	90286.062	11			

4.4.4.3 Conductivity at Surface Area of 30 cm^2

Table 4.20 presents ANOVA analysis that measured difference in performance of iron and aluminum in treating conductivity at surface area of 30 cm^2. Since the obtained p-value=0.104 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum in reducing Conductivity of wastewater at surface area of 30 cm^2 for the observed conditions (electrode and surface area). The observed difference is therefore due to effect of current on the different types of electrodes.

Table 4.20: Difference in Conductivity for Iron and Aluminium at 30 cm^2

ANOVA
Mean Value

	Sum of Squares	Df	Mean Square	F	Sig.
Between Groups	21958.974	1	21958.974	3.195	0.104
Within Groups	68727.631	10	6872.763		
Total	90686.605	11			

4.4.4.4 Conductivity at Surface Area of 40 cm^2

Table 4.21 presents ANOVA analysis that measured difference in performance of iron and aluminum in treating conductivity at surface area of 40 cm^2. Since the obtained p-value=0.182 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of Iron and aluminum in reducing Conductivity of wastewater at surface area of 40cm^2 for the observed conditions (electrode type and surface area). The observed difference is therefore due to effect of current on the electrodes. Although performance increases with increase in surface area for both aluminium and iron electrodes, aluminium demonstrated to perform better and this was due to the reaction that takes place when current acts on it.

Table 4.21: Difference in Conductivity for Iron and Aluminium at 40 cm^2

ANOVA

Mean Value

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	18695.360	1	18695.360	2.059	0.182
Within Groups	90784.087	10	9078.409		
Total	109479.447	11			

4.4.5. Total Suspended Solids (TSS)

The research sought to test the hypothesis;

$H5_0$: The total suspended solids of wastewater treated using aluminium electrodes are not significantly different from the total suspended solids of wastewater treated using iron electrodes at p =0.05

This hypothesis was tested at different surface areas to determine the performance of each electrode with regards to quality of the treated water as presented below.

4.4.5.1 TSS at Surface Area of 10 cm^2

Table 4.22 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing TSS at surface area of 10 cm^2. Since the obtained p-value=0.837 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of Iron and aluminum in reducing TSS of wastewater at surface area of 10 cm^2 for the observed conditions (electrode and surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.22: Difference in TSS at 10 cm^2 for Iron and Aluminium

ANOVA
Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	864.239	1	864.239	0.045	0.837
Within Groups	173691.912	9	19299.101		
Total	174556.150	10			

4.4.5.2 TSS at Surface Area of 20 cm^2

Table 4.23 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing TSS at surface area of 20 cm^2. Since the obtained p-value=0.950 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum in reducing TSS of wastewater at 10 cm^2 for the observed conditions (element (type of electrode and number of plates (surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.23: Difference in TSS at 20 cm^2 for Iron and Aluminium

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	72.763	1	72.763	0.004	0.950
Within Groups	192730.564	11	17520.960		
Total	192803.327	12			

4.4.5.3 TSS at Surface Area of 30 cm^2

Table 4.24 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing TSS at surface area of $30cm^2$. Since the obtained p-value=0.793 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum in reducing TSS of wastewater at surface area of $30cm^2$ for the observed conditions (element and surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.24: Difference in TSS at 30 cm^2 for Iron and Aluminium

ANOVA

Mean Value

	Sum of Squares	Df	Mean Square	F	Sig.
Between Groups	1415.492	1	1415.492	0.073	0.793
Within Groups	194954.497	10	19495.450		
Total	196369.989	11			

4.4.5.4 TSS at Surface Area of 40 cm^2

Table 4.25 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing TSS at surface area of 40 cm^2. Since the obtained p-value=0.660 is greater than 0.05, the formulated null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum in reducing TSS of wastewater at one plate for the observed conditions (electrode and surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.25: Difference in Turbidity at 40 cm^2 for Iron and Aluminium Plates

ANOVA

Mean Value

	Sum of Squares	**df**	**Mean Square**	**F**	**Sig.**
Between Groups	4485.720	1	4485.720	0.206	0.660
Within Groups	217894.552	10	21789.455		
Total	222380.272	11			

4.4.6. Turbidity

The study formulated the hypothesis:
$H6_0$: The turbidity of wastewater treated using aluminium electrodes is not significantly different from the turbidity of wastewater treated using iron electrodes at p =0.05

This hypothesis was tested at different surface areas to determine the performance of each electrode regards to quality of the treated water as presented below.

4.4.6.1 Turbidity at Surface Area of 10 cm^2

Table 4.26 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing turbidity at surface area of 10 cm^2. Since the p-value=0.660 is greater than 0.05, the study's null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum in reducing turbidity of wastewater at 10 cm^2 for the observed conditions (electrode material) and surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.26: Difference in Turbidity at 10 cm^2 for Iron and Aluminium Plates

ANOVA

Mean Value

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	4485.720	1	4485.720	0.206	0.660
Within Groups	217894.552	10	21789.455		
Total	222380.272	11			

4.4.6.2 Turbidity at Surface Area of 20 cm^2

Table 4.27 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing turbidity at surface area of 20 cm$^{2.}$ Since the obtained p-value=0.422 is greater than 0.05, the study's null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminum in reducing turbidity of wastewater at surface area of 20 cm^2 for the observed conditions (element (electrode material) and surface area)). The observed difference is therefore due to effect of current on the electrodes.

Table 4.27: Difference in Turbidity at 20 cm^2 for Iron and Aluminium Plates

ANOVA

Mean Values

	Sum of Squares	**Df**	**Mean Square**	**F**	**Sig.**
Between Groups	83335.000	1	83335.000	0.700	0.422
Within Groups	1190334.090	10	119033.409		
Total	1273669.090	11			

4.4.6.3 Turbidity at Surface Area of 30cm^2

Table 4.28 presents ANOVA analysis that measured difference in performance of iron and aluminum in removing turbidity at surface area of 30 cm^2. Since the obtained p-value=0.435 is greater than 0.05, the study's null hypothesis is accepted and a conclusion made that there is no significant difference in performance of Iron and Aluminum in reducing turbidity of wastewater at surface area of 30 cm^2 for the observed conditions (element/ electrode material and surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.28: Difference in Turbidity at 30cm^2 for Iron and Aluminium Plates

ANOVA

Mean Values

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	83665.330	1	83665.330	0.662	0.435
Within Groups	1264152.614	10	126415.261		
Total	1347817.944	11			

4.4.6.4 Turbidity at Surface Area of 40 cm^2

Table 4.29 presents ANOVA analysis that measured difference in performance of iron and aluminium in removing turbidity at surface area of 40 cm^2. Since the obtained p-value=0.625 is greater than 0.05, the study' null hypothesis is accepted and a conclusion made that there is no significant difference in performance of iron and aluminium in reducing turbidity of wastewater at surface area of 40 cm^2 for the observed conditions (element/ electrode material and surface area). The observed difference is therefore due to effect of current on the electrodes.

Table 4.29: Difference in Turbidity at 40 cm^2 for Iron and Aluminium Plates

ANOVA

Mean Values

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	34776.333	1	34776.333	0.254	0.625
Within Groups	1368012.227	10	136801.223		
Total	1402788.560	11			

4.5. The Effect of Electrode Material

Performance of iron and aluminium electrodes was compared under similar operational conditions (Voltage, surface area, and treatment time) for all the parameters. The results obtained from the experiments for both electrode types are presented in figures 4.1 to 4.12. As seen from these figures, performances of electrodes were not the same under similar conditions. Aluminium electrodes showed higher treatment efficiency than iron ones with regards to removal of COD, BOD_5, Turbidity, TSS, conductivity and Total Alkalinity from slaughterhouse waste water. The rates of removals for Aluminium electrodes were faster at lower surface area and voltage. The optimum surface area was found to be 40 cm^2 at 25V compared to Iron electrodes where optimum surface area was found to be 90 cm^2. Superior performance of Aluminium is attributed to the fact that Al $(OH)_3$ is known to be a better adsorbent than Fe $(OH)_3$ (Bagga *et al*. 2008).

Iron electrodes transfer higher numbers of Iron ions into solution and they produce a higher amount of sludge compared with Aluminium electrodes. Considering the fact that the costs of both types of electrodes are almost same, Aluminium would be a good choice for electrode material as it provides higher treatment efficiencies.

Findings from ANOVA Analysis also indicate that aluminium performed significantly better than iron electrodes and attribute the superior performance of aluminium electrodes to the action of the current/voltage on the aluminium plates. This study therefore concludes that aluminium is the better electrode.

4.6 The Effect of Applied Voltage and Current Density

The voltage applied at the electrodes is one of the most crucial parameters that influence the performance as well as economy of the EC process. To understand how applied voltage affects the efficiency of EC process in treating slaughterhouse wastewater, several voltages in the range of 5V to 25V were applied between the electrodes in the electrocoagulation cell, and pollutants removal was determined at the conditions given as indicated by figures 4.1 to 4.12

The applied voltage was observed to exhibit a strong effect on electrocoagulation, especially on the abatement of BOD, COD, Turbidity, TSS and conductivity: higher the current (voltage) resulted in shorter the treatment time. The quantity of the resulting coagulant as well as amount of Al^{3+} and Fe^{3+} ions produced by the respective electrodes were determined by supply of current to the EC system. Thus, increase in voltage/current resulted in higher concentration of Al^{3+} and Fe^{3+} ions that got dissolved into the solution and hence increasing the formation rate of $Al(OH)_3$ and $2Fe(OH)_3$. It is also well-known that in addition to determining the coagulant dosage rate, electrical potential also determines the bubble production rate and size as well as the flocs growth (Holt, Barton, Wark, & Mitchell, 2002), which could have influenced efficiency of the EC treatment process.

Graphical presentations have demonstrated that the removal efficiency of pollutants is very high and as expected, it appears that for a given time, the efficiency increased significantly with increase of electrical potential. As the results indicate, the removal efficiencies increased as the electrical potentials increased. As an example, COD concentration decreased from 268 mg/l at 5V to 16.33 mg/l at 25V for surface area of 40cm^2 which was way below the permissible level and approximately 89% COD removal efficiency after electrocoagulation process with electrical potential of 25V compared to 4.5% with 5V.

The effect of current density can also be explained the same way. During the reactive phase (electrocoagulation process), the reduction rate of COD, BOD$_5$, Total Alkalinity, TSS, Conductivity and turbidity were observed to increase with the current density. This can be explained by the fact that the amount of Al^{3+} and Fe^{3+} species formed by dissolution of the anode, increases with the current density according to Faraday’s law (see Eq. (4)). Higher amounts of dissolved aluminium and Iron allowed higher coagulation efficiency and more significant destabilization of the emulsion. Moreover as shown by Khemis, Leclerc, Tanguy, Valentin, and Lapicque (2006), higher production rates of hydrogen allowed by higher currents, favours the flotation of the flocculated matter.

CHAPTER FIVE

CONCLUSION AND RECOMMENDATIONS

5.1 Introduction

This chapter provides conclusions made based on the findings and goes further to highlight areas that require further research in order to enhance knowledge and understanding of the electrocoagulation process in treating wastewater from slaughterhouses.

5.2 Conclusions

Based on the findings of the experiments, this study concludes the following;

1. Characterization of the wastewater from Gachororo slaughterhouse in terms of COD, BOD_5, total alkalinity, turbidity, total suspended solids (TSS), electrical conductivity and pH indicates that it is highly contaminated with both organic and inorganic contaminants as values of these parameters were found to be way above the recommended GoK (through NEMA) standards for release of water into the environment.
2. Electrocoagulation is very effective in treating slaughterhouse wastewaters at optimum voltage/current and surface area as it was able to treat the water of high concentrations of suspended solids, COD, BOD_5, total alkalinity, , turbidity, conductivity and low pH to values that meet the recommended GoK (through NEMA) standards for water discharged into the environment. The treated water can therefore be recovered and re-used.
3. Aluminum was found to perform better as an electrode at the same voltage and surface area compared with Iron as the recommended values for all the parameters being tested were achieved at a lower surface area ($40cm^2$)

compared to $100cm^2$ required for iron electrodes. This is because Aluminum hydroxide is a better adsorbent than the hydroxides of iron.

5.2 Recommendations

5.3.1 Recommendations from the Study

Based on the findings of this study, the following is recommended;

1. Further inquiry into the process through which electro-coagulation reduces total alkalinity of wastewater.

5.3.2 Recommendations for Further Studies

This study recommends the following;

1. Further studies on the use of solar or wind as sources of energy in the electrocoagulation process.
2. Investigation of other electrode materials that may be more effective and environmentally friendly than aluminium or Iron.

REFERENCES

Adhoum, N., & Monser, L. (2004). Decolourization and removal of phenolic compounds from olive mill wastewater by electrocoagulation. *Chemical Engineering and Processing: Process Intensification*, *10*(43), 1281-1287.

Al Aji, B., Yavuz, Y., & Koparal, A. S. (2012). Electrocoagulation of heavy metals containing model wastewater using monopolar iron electrodes. *Separation and Purification Technology*, *86*, 248-254.

Akbal, F., & Camci, S. (2010). Comparison of electrocoagulation and chemical coagulation for heavy metal removal. *Chemical Engineering & Technology*, *33*(10), 1655-1664.

Akbal, F., & Camcı, S. (2011). Copper, chromium and nickel removal from metal plating wastewater by electrocoagulation. *Desalination*, *269*(1-3), 214-222.

American Public Health Association. (1998). American Water Works Association and Water Environment Federation. *Standard Methods for the Examination of Water and Wastewater*, *1*(2), 39-42.

Asselin, M., Drogui, P., Benmoussa, H., & Blais, J. F. (2008). Effectiveness of electrocoagulation process in removing organic compounds from slaughterhouse wastewater using monopolar and bipolar electrolytic cells. *Chemosphere*, *72*(11), 1727-1733.

Bagga, A., Chellam, S., & Clifford, D. A. (2008). Evaluation of iron chemical coagulation and electrocoagulation pretreatment for surface water microfiltration. *Journal of Membrane Science*, *309*(1-2), 82-93.

Bayar, S., Yıldız, Y. Ş., Yılmaz, A. E., & İrdemez, Ş. (2011). The effect of stirring speed and current density on removal efficiency of poultry slaughterhouse wastewater by electrocoagulation method. *Desalination*, *280*(1-3), 103-107.

Bayramoglu, M., Kobya, M., Can, O. T., & Sozbir, M. (2004). Operating cost analysis of electrocoagulation of textile dye wastewater. *Separation and Purification Technology*, *37*(2), 117-125.

Bayramoglu, M., Eyvaz, M., & Kobya, M. (2007). Treatment of the textile wastewater by electrocoagulation: Economical evaluation. *Chemical Engineering Journal*, *128*(2-3), 155-161.

Bazrafshan, E., Mostafapour, F. K., Farzadkia, M., Ownagh, K. A., & Mahvi, A. H. (2012). Slaughterhouse wastewater treatment by combined chemical coagulation and electrocoagulation process. *PloS one*, *7*(6), e40108.

Beck, E. C., Giannini, A. P., & Ramirez, E. R. (1974). Electrocoagulation clarifies food wastewater. *Food Technology*. 28, 18-19

Belkacem, M., Khodir, M., & Abdelkrim, S. (2008). Treatment characteristics of textile wastewater and removal of heavy metals using the electroflotation technique. *Desalination*, *228*(1-3), 245-254.

Budiyono, I., & Johari,S. (2010). Study on Treatment of Slaughterhouse Wastewater by Electro-coagulation Technique. *International Journal of Science and Engineering*, *1*(1), 25-28.

Budiyono, B., Seno, J., & Sunarso, S. (2011). Study on slaughterhouse wastes potency and characteristic for biogas production. *International Journal of Waste Resources (IJWR)*, *1*(2), 4-7.

Butler, E., Hung, Y. T., Yeh, R. Y. L., & Suleiman Al Ahmad, M. (2011). Electrocoagulation in wastewater treatment. *Water*, *3*(2), 495-525

Cañizares, P., Jiménez, C., Martínez, F., Rodrigo, M. A., & Sáez, C. (2009). The pH as a key parameter in the choice between coagulation and electrocoagulation for the treatment of wastewaters. *Journal of Hazardous Materials*, *163*(1), 158-164.

Chen, G. (2004). Electrochemical technologies in wastewater treatment. *Separation and Purification Technology*, *38*(1), 11-41.

Chen, X., Chen, G., & Yue, P. L. (2000). Separation of pollutants from restaurant wastewater by electrocoagulation. *Separation and Purification Technology*, *19*(1-2), 65-76.

Chou, W. L., Wang, C. T., & Huang, K. Y. (2009). Effect of operating parameters on indium (III) ion removal by iron electrocoagulation and evaluation of specific energy consumption. *Journal of Hazardous Materials*, *167*(1-3), 467-474.

Cuetos, M. J., Gómez, X., Otero, M., & Morán, A. (2008). Anaerobic digestion of solid slaughterhouse waste (SHW) at laboratory scale: influence of co-digestion with the organic fraction of municipal solid waste (OFMSW). *Biochemical Engineering Journal*, *40*(1), 99-106.

Deniel, R., Bindu, V. H., Rao, A.V., & Anjaneyulu, Y. (2008). Removal of arsenic from wastewaters using electrocoagulation. *Journal of Environmental Science Engineering*, 50(2), 283-288.

Department of Water Affairs and Forestry, DWAF, (1996). South African Water Quality Guidelines. Domestic Uses. 2nd. Ed. Department of Water Affairs and Forestry, Pretoria, Vol. 1.

Emamjomeh, M. M., & Sivakumar, M. (2009). Fluoride removal by a continuous flow electrocoagulation reactor. *Journal of Environmental Management, 90*(2), 1204-1212.

Emsley, J. (2003). Book Review: Nature's building blocks: an AZ guide to the elements/Oxford University Press, New York, 538 pp., 2002, ISBN 0-198-50341-5. *Astronomy, 31*(2), 87-88.

Eryuruk, K., Tezcan, U., & Ogutveren, U. B. (2011). Treatment of cattle-slaughterhouse wastewater using tubular electrocoagulator. *International Conference on Chemical Engineering and Applications IPCBEE*, *23*, 134-137.

Espinoza-Quiñones, F. R., Fornari, M. M., Módenes, A. N., Palácio, S. M., da Silva Jr, F. G., Szymanski, N., ... & Trigueros, D. E. (2009). Pollutant removal from tannery effluent by electrocoagulation. *Chemical Engineering Journal*, *151*(1-3), 59-65.

Eyvaz, M., Kirlaroglu, M., Aktas, T. S., & Yuksel, E. (2009). The effects of alternating current electrocoagulation on dye removal from aqueous solutions. *Chemical Engineering Journal*, *153*(1-3), 16-22.

Fia, R., Pereira, E. L., Fia, F. R., Emboaba, D. G., & Gomes, E. M. (2015). Start-up of anaerobic reactors for slaughterhouse wastewater treatment. *Engenharia Agrícola*, *35*(2), 331-339.

Gao, P., Chen, X., Shen, F., & Chen, G. (2005). Removal of chromium (VI) from wastewater by combined electrocoagulation–electroflotation without a filter. *Separation and Purification Technology, 43*(2), 117-123.

Gomes, J. A., Daida, P., Kesmez, M., Weir, M., Moreno, H., Parga, J. R., ... & Cocke, D. L. (2007). Arsenic removal by electrocoagulation using combined Al–Fe electrode system and characterization of products. *Journal of hazardous materials*, *139*(2), 220-231

Gürses, A., Yalçin, M., & Doğar, C. (2002). Electrocoagulation of some reactive dyes: a statistical investigation of some electrochemical variables. *Waste Management, 22*(5), 491-499.

Hanay, Ö., & Hasar, H. (2011). Effect of anions on removing Cu2+, Mn2+ and Zn2+ in electrocoagulation process using aluminium electrodes. *Journal of Hazardous Materials, 189*(1-2), 572-576.

Hasson, D., Lumelsky, V., Greenberg, G., Pinhas, Y., & Semiat, R. (2008). Development of the electrochemical scale removal technique for desalination applications. *Desalination*, *230*(1-3), 329-342.

Heidmann, I., & Calmano, W. (2008). Removal of Cr (VI) from model wastewaters by electrocoagulation with Fe electrodes. *Separation and Purification Technology, 61*(1), 15-21.

Holt, P. K., Barton, G. W., Wark, M., & Mitchell, C. A. (2002). A quantitative comparison between chemical dosing and electrocoagulation. *Colloids and Surfaces A: Physicochemical and Engineering Aspects*, *211*(2-3), 233-248.

Hu, C. Y., Lo, S. L., & Kuan, W. H. (2003). Effects of co-existing anions on fluoride removal in electrocoagulation (EC) process using aluminium electrodes. *Water Research*, *37*(18), 4513-4523.

Janpoor, F., Torabian, A., & Khatibikamal, V. (2011). Treatment of laundry wastewater by electrocoagulation. *Journal of Chemical Technology and Biotechnology, 86*(8), 1113-1120.

Jiang, J. Q., Graham, N., André, C., Kelsall, G. H., & Brandon, N. (2002). Laboratory study of electro-coagulation–flotation for water treatment. *Water Research*, *36*(16), 4064-4078.

Kaluli, J. W., Githuku, C., Home, P., & Mwangi, B. M. (2011). Towards a national policy on wastewater reuse in Kenya. *Journal of Agriculture, Science and Technology*, *13*(1), 116-125.

Kanu, I., Achi, O. K., Ezeronye, O. U., & Anyanwu, E. C. (2006). Seasonal variation in bacterial heavy metal biosorption in water samples from Eziama River near soap and brewery industries and the environmental health implications. *International Journal of Environmental Science & Technology*, *3*(1), 95-102.

Kanu, I., & Achi, O. K. (2011). Industrial effluents and their impact on water quality of receiving rivers in Nigeria. *Journal of Applied Technology in Environmental Sanitation*, *1*(1), 75-86.

Katal, R., & Pahlavanzadeh, H. (2011). Influence of different combinations of aluminium and iron electrode on electrocoagulation efficiency: Application to the treatment of paper mill wastewater. *Desalination*, *265*(1-3), 199-205.

Kenya National Bureau of Statistics. (20140). Kenya 2009 Population and housing census highlights. www.Planning.go.ke. Accessed in July , 2015.

Khemis, M., Leclerc, J. P., Tanguy, G., Valentin, G., & Lapicque, F. (2006). Treatment of industrial liquid wastes by electrocoagulation: experimental investigations and an overall interpretation model. *Chemical Engineering Science*, *61*(11), 3602-3609.

Khristoskova, S. (1984). Possibility of purification and decoloring wastewaters from the yeast industry by electrocoagulation. *Nauchni Tr-Plovdski Uni.(Bul.)*, *22*, 177-185.

Kobya, M., Hiz, H., Senturk, E., Aydiner, C., & Demirbas, E. (2006). Treatment of potato chips manufacturing wastewater by electrocoagulation. *Desalination*, *190*(1-3), 201-211.

Kobya, M., Ciftci, C., Bayramoglu, M., & Sensoy, M. T. (2008). Study on the treatment of waste metal cutting fluids using electrocoagulation. *Separation and Purification Technology*, *60*(3), 285-291.

Kolics, A., Polkinghorne, J. C., & Wieckowski, A. (1998). Adsorption of sulfate and chloride ions on aluminium. *Electrochimica Acta*, *43*(18), 2605-2618.

Kongjao, S., Damronglerd, S., & Hunsom, M. (2008). Simultaneous removal of organic and inorganic pollutants in tannery wastewater using electrocoagulation technique. *Korean Journal of Chemical Engineering*, *25*(4), 703.

Kuokkanen, V., Kuokkanen, T., Rämö, J., & Lassi, U. (2013). Recent applications of electrocoagulation in treatment of water and wastewater—a review. *Green and Sustainable Chemistry*, *3*(02), 89.

Kurt, U., Gonullu, M. T., Ilhan, F., & Varinca, K. (2008). Treatment of domestic wastewater by electrocoagulation in a cell with Fe–Fe electrodes. *Environmental Engineering Science*, *25*(2), 153-162.

Linares-Hernández, I., Barrera-Díaz, C., Roa-Morales, G., Bilyeu, B., & Ureña-Núñez, F. (2009). Influence of the anodic material on electrocoagulation performance. *Chemical Engineering Journal*, *148*(1), 97-105.

Liu, Y. (2003). Chemically reduced excess sludge production in the activated sludge process. *Chemosphere, 50*(1), 1-7.

Malakootian, M., & Yousefi, N. (2009). The efficiency of electrocoagulation process using aluminium electrodes in removal of hardness from water. *Iranian Journal of Environmental Health Science & Engineering (IJEHSE)*, *6*(2).

Mansouri, K., Hannachi, A., Abdel-Wahab, A., & Bensalah, N. (2012). Electrochemically dissolved aluminium coagulants for the removal of natural organic matter from synthetic and real industrial wastewaters. *Industrial & Engineering Chemistry Research*, *51*(5), 2428-2437.

Massé, D. I., & Masse, L. (2001). The effect of temperature on slaughterhouse wastewater treatment in anaerobic sequencing batch reactors. *Bioresource Technology*, *76*(2), 91-98.

Masse, L., & Masse, D. I. (2005). Effect of soluble organic, particulate organic, and hydraulic shock loads on anaerobic sequencing batch reactors treating slaughterhouse wastewater at 20 C. *Process Biochemistry*, *40*(3-4), 1225-1232.

Matis, K. A., & Peleka, E. N. (2010). Alternative flotation techniques for wastewater treatment: focus on electroflotation. *Separation Science and Technology, 45*(16), 2465-2474.

McCornick, P. G., Hijazi, A., & Sheikh, B. (2004). *From wastewater reuse to water reclamation: progression of water reuse standards in Jordan. Wastewater use in irrigated agriculture: Confronting the livelihood and environmental realities*. CABI Publishing, Cambridge, pp. 153-162

Meas, Y., Ramirez, J. A., Villalon, M. A., & Chapman, T. W. (2010). Industrial wastewaters treated by electrocoagulation. *Electrochimica Acta, 55*(27), 8165-8171.

Merzouk, B., Yakoubi, M., Zongo, I., Leclerc, J. P., Paternotte, G., Pontvianne, S., & Lapicque, F. (2011). Effect of modification of textile wastewater composition on electrocoagulation efficiency. *Desalination*, *275*(1-3), 181-186.

Meunier, N., Drogui, P., Mercier, G., & Blais, J. F. (2009). Treatment of metal-loaded soil leachates by electrocoagulation. *Separation and Purification Technology, 67*(1), 110-116.

Meunier, N., Drogui, P., Montané, C., Hausler, R., Mercier, G., & Blais, J. F. (2006). Comparison between electrocoagulation and chemical precipitation for metals removal from acidic soil leachate. *Journal of Hazardous Materials, 137*(1), 581-590.

Ministry of Water and Irrigation(2007). Draft National Policy on Irrigation and Drainage Development. Republic of Kenya.

Mollah, M. Y. A., Schennach, R., Parga, J. R., & Cocke, D. L. (2001). Electrocoagulation (EC)—science and applications. *Journal of Hazardous Materials, 84*(1), 29-41.

Mollah, M. Y., Morkovsky, P., Gomes, J. A., Kesmez, M., Parga, J., & Cocke, D. L. (2004). Fundamentals, present and future perspectives of electrocoagulation. *Journal of Hazardous Materials*, *114*(1-3), 199-210.

Moreno C, H. A., Cocke, D. L., Gomes, J. A., Morkovsky, P., Parga, J. R., Peterson, E., & Garcia, C. (2009). Electrochemical reactions for electrocoagulation using iron electrodes. *Industrial & Engineering Chemistry Research*, *48*(4), 2275-2282.

Morgan, B., & Lahav, O. (2007). The effect of pH on the kinetics of spontaneous Fe (II) oxidation by O2 in aqueous solution–basic principles and a simple heuristic description. *Chemosphere*, *68*(11), 2080-2084.

Mouedhen, G., Feki, M., Wery, M. D. P., & Ayedi, H. F. (2008). Behavior of aluminium electrodes in electrocoagulation process. *Journal of Hazardous Materials*, *150*(1), 124-135.

Naje, A. S., & Abbas, S. A. (2013). Electrocoagulation Technology in Wastewater Treatment: A Review of Methods and Applications. *Civil and Environmental Research*, *3*(11),29-42

Ngigi, A., & Macharia, D. (2006). Education Policy Overview Paper. http://www.enable.nu/publication/D_1_7_kenya_Education_Policy_Overview.pdf (Accessed 13th December, 2019).

Nouri, J., Mahvi, A. H., & Bazrafshan, E. (2010). Application of electrocoagulation process in removal of zinc and copper from aqueous solutions by aluminium electrodes. *International Journal of Environmental Research*, *4*(2), 201-208.

Ölmez, T. (2009). The optimization of Cr (VI) reduction and removal by electrocoagulation using response surface methodology. *Journal of Hazardous Materials*, *162*(2-3), 1371-1378.

Padilla-Gasca, E., López-López, A., & Gallardo-Valdez, J. (2011). Evaluation of stability factors in the anaerobic treatment of slaughterhouse wastewater. *Journal of Bioremediation and Biodegradation*, *2*, 114.

Pajootan, E., Arami, M., & Mahmoodi, N. M. (2012). Binary system dye removal by electrocoagulation from synthetic and real colored wastewaters. *Journal of the Taiwan Institute of Chemical Engineers*, *43*(2), 282-290.

Pal Palatsi, J., Viñas, M., Guivernau, M., Fernandez, B., & Flotats, X. (2011). Anaerobic digestion of slaughterhouse waste: main process limitations and microbial community interactions. *Bioresource Technology*, *102*(3), 2219-2227.

Parawira, W., Kudita, I., Nyandoroh, M. G., & Zvauya, R. (2005). A study of industrial anaerobic treatment of opaque beer brewery wastewater in a tropical climate using a full-scale UASB reactor seeded with activated sludge. *Process Biochemistry*, *40*(2), 593-599.

Picard, T., Cathalifaud-Feuillade, G., Mazet, M., & Vandensteendam, C. (2000). Cathodic dissolution in the electrocoagulation process using aluminium electrodes. *Journal of Environmental Monitoring*, *2*(1), 77-80.

Pouet, M. F., & Grasmick, A. (1995). Urban wastewater treatment by electrocoagulation and flotation. *Water Science and Technology*, *31*(3-4), 275-283.

Rajakumar, R., Meenambal, T., Saravanan, P. M., & Ananthanarayanan, P. (2012). Treatment of poultry slaughterhouse wastewater in hybrid upflow anaerobic sludge blanket reactor packed with pleated poly vinyl chloride rings. *Bioresource Technology*, *103*(1), 116-122.

Refait, P. (1998). Mechanisms of oxidation of Ni (II)-Fe (II) hydroxides in chloride-containing aqueous media: Role of the pyroaurite-type Ni-Fe hydroxychlorides. *Clay Minerals*, *32*(4), 597-613.

Renk, R. R. (1988). Electrocoagulation of tar sand and oil shale wastewaters. *Energy Progress*, *8*(4), 205-208.

Rezaee, A., Hossini, H., Masoumbeigi, H., & Soltani, R. D. C. (2011). Simultaneous Removal of Hexavalent Chromium andNitrate from Wastewater using

Electrocoagulation Method. *International Journal of Environmental Science and Development*, *2*(4), 294.

Rocha, J. H. B., & Martinez-Huitle, C. A. (2011). Electrochemical technology for removing petroleum hydrocarbons from produced water. *Exploration and Production: Oil and Gas Review*, *96*,(1), 80–87

Rodrigo, M. A., Cañizares, P., Buitrón, C., & Sáez, C. (2010). Electrochemical technologies for the regeneration of urban wastewaters. *Electrochimica Acta*, *55*(27), 8160-8164.

Rulkens, W. (2006). Increasing the environmental sustainability of sewage treatment by mitigating pollutant pathways. *Environmental Engineering Science*, *23*(4), 650-665.

Saleem, M., Bukhari, A. A., & Akram, M. N. (2011). Electrocoagulation for the treatment of wastewater for reuse in irrigation and plantation. *Journal of Basic and Applied Sciences*, *7*(1), 11-20.

Sarairah, A., & Jamrah, A. (2008). Characterization and assessment of treatability of wastewater generated in Amman slaughterhouse. *Dirasat Engineering Sciences*, *35*(2), 71-83.

Sarala, C. (2012). Domestic wastewater treatment by electrocoagulation with Fe-Fe electrodes. *International Journal of Engineering Trends and Technology*, *3*(4), 550-533

Sarkar, M. S. K. A., Evans, G. M., & Donne, S. W. (2010). Bubble size measurement in electroflotation. *Minerals Engineering*, *23*(11-13), 1058-1065.

Sasson, M. B., Calmano, W., & Adin, A. (2009). Iron-oxidation processes in an electroflocculation (electrocoagulation) cell. *Journal of Hazardous Materials, 171*(1-3), 704-709.

Seif, H, & Moursy, A. (2011). Treatment of Slaughterhouse Wastes. *Sixth International Water Technology Conference (IWTC), 2011* PP. 269-275

Shafaei, A., Rezayee, M., Arami, M., & Nikazar, M. (2010). Removal of Mn2+ ions from synthetic wastewater by electrocoagulation process. *Desalination*, *260*(1-3), 23-28.

Sridhar, R., Sivakumar, V., Immanuel, V. P., & Maran, J. P. (2011). Treatment of pulp and paper industry bleaching effluent by electrocoagulant process. *Journal of Hazardous Materials, 186*(2-3), 1495-1502.

Souza, K. R., Silva, D. R., Mata, W., Martínez-Huitle, C. A., & Mata, A. L. (2012). Electrochemical technology for removing heavy metals present in synthetic produced water. *Latin American Applied Research*, *42*(2), 141-147.

Tariq, M., Ali, M., & Shah, Z. (2006). Characteristics of industrial effluents and their possible impacts on quality of underground water. *Soil and Environnent*, *25*(1), 64-69.

Terrazas, E., Vázquez, A., Briones, R., Lázaro, I., & Rodríguez, I. (2010). EC treatment for reuse of tissue paper wastewater: aspects that affect energy consumption. *Journal of Hazardous Materials*, *181*(1-3), 809-816.

Tezcan Ün, Ü., Koparal, A.S., & Bakir Öğütveren, Ü. (2009). Hybrid processes for the treatment of cattle-slaughterhouse wastewater using aluminium and iron electrodes. *Journal of Hazardous Material*, *164(2-3)*, 580-586.

Trompette, J. L., & Vergnes, H. (2009). On the crucial influence of some supporting electrolytes during electrocoagulation in the presence of aluminium electrodes. *Journal of Hazardous Materials, 163*(2-3), 1282-1288.

Vasudevan, S., Lakshmi, J., Jayaraj, J., & Sozhan, G. (2009). Remediation of phosphate-contaminated water by electrocoagulation with aluminium, aluminium alloy and mild steel anodes. *Journal of Hazardous Materials, 164*(2-3), 1480-1486.

Vepsäläinen, M. (2012). *Electrocoagulation in the treatment of industrial waters and wastewaters*. VTT Technical Research Centre of Finland.

Vepsäläinen, M., Ghiasvand, M., Selin, J., Pienimaa, J., Repo, E., Pulliainen, M., & Sillanpää, M. (2009). Investigations of the effects of temperature and initial sample pH on natural organic matter (NOM) removal with electrocoagulation using response surface method (RSM). *Separation and Purification Technology*, *69*(3), 255-261.

Vepsäläinen, M., Pulliainen, M., & Sillanpää, M. (2012). Effect of electrochemical cell structure on natural organic matter (NOM) removal from surface water through electrocoagulation (EC). *Separation and Purification Technology*, *99*, 20-27.

Vepsalainen, M., Selin, J., Pulliainen, M., & Sillanpaa, M. (2007). Combined electrocoagulation and chemical coagulation of paper mill mechanically cleaned water. *Journal of Pulp and Paper Science*, *33*(4), 233-239.

Vik, E. A., Carlson, D. A., Eikum, A. S., & Gjessing, E. T. (1984). Electrocoagulation of potable water. *Water Research*, *18*(11), 1355-1360.

Wang, X., Li, H., Su, D., & Sun, T. (2010). Treatment of printing and dyeing wastewater by DC electrocoagulation method. *Environmental Science & Technology (China), 33*(2), 150-153.

WHO, (2004). Rolling revision of the WHO guidelines for drinking-water quality, Draft for review and comments. Nitrates and Nitrites in drinking-water, World Health Organization. (WHO/SDE/WSH/04.08/56).

Yilmaz, A. E., Boncukcuoğlu, R., Kocakerim, M. M., Yilmaz, M. T., & Paluluoğlu, C. (2008). Boron removal from geothermal waters by electrocoagulation. *Journal of Hazardous Materials, 153*(1-2), 146-151.

Zhang, L., Xue, G., Zhang, N., Liu, S., Duan, A., & Wang, L. (2011, May). Decolorization study of cooking wastewater by continuous electrocoagulation process. In *Materials for Renewable Energy and Environment (ICMREE), 2011 International Conference on* (Vol. 1, pp. 860-863). IEEE.

APPENDICES

Appendix 1: Photos of Raw Wastewater, Experimental Set up, ongoing Treatment and treated Water

Plate 1: Raw Wastewater in a Plastic Container

Plate 2: Experimental Set-up

Plate 3: Ongoing Treatment for Iron Plates (with formation of Coagulants)

Plate 4: Complete Process with Clean Water below the Coagulants

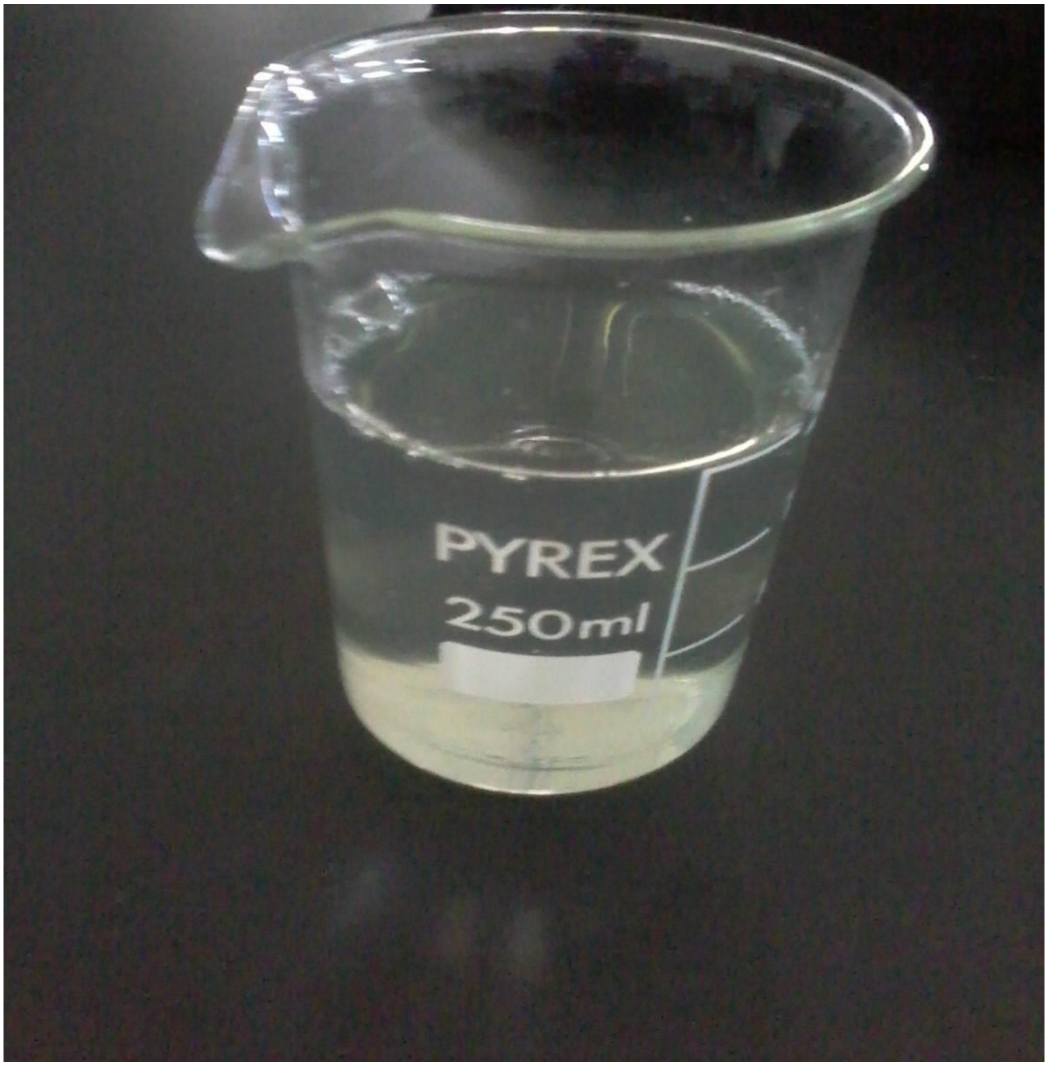

Plate 5: Sample of Treated Water