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Abstract

The goal of this research was to investigate the feasibility of using electrocoagulation (EC) as a primary surface water treatment process that could be used with solar panels in decentralized locations. In this study, the removal of turbidity, UV 254 and dissolved organic carbon (DOC) was studied. Operating parameters such as current density (0.91 - 10 mA/cm^2) and charge loading (0.083 – 1.250 A.h/L) in bipolar and monopolar configuration were studied using a DC power supply. Experimental results showed that a current density of 4.55 mA/cm^2 and a treatment time of 60 min, in the bipolar configuration, allowed the best removal of turbidity, UV 254 and DOC (97.0%, 93.0 %, and 95.2%, respectively). For the second part of the study, the DC power supply was replaced by two photovoltaic panels without the use of a set of batteries and a charge controller. According to the results, the turbidity removal is directly proportional to the solar radiation intensity. This suggests that it is possible to control the charge loading by adjusting the treatment time as a function of solar intensity. Such outcomes suggest that solar-powered electrocoagulation (SPEC) would be a suitable technology for areas not connected to centralized water treatment networks.

Introduction

Still to this day, more than 700 million people worldwide do not have access to improved drinking water sources (WHO. and UNICEF., 2015). Unsafe drinking water can cause waterborne diseases such as typhoid fever, cholera, *Hepatite A* and *E*. It is also directly related to severe diseases such as diarrhea and meningitis following the contraction of bacteria such as *E.coli*. Waterborne diseases are a serious issue that must be addressed as they account for 90 % of deaths in developing countries, affecting mainly young children. Indeed, 88 % of all cases of diarrhea in the word are a direct result of unsafe water use (Ashbolt, 2004). This issue is critical in regions like sub-Saharan Africa, which account for 43 % of the world's population without access to improved drinking water sources (325 million) (WHO. & UNICEF., 2014). Unlike improved water sources, unimproved waters (surface water, tanker trucks, unprotected springs or dug wells) are susceptible to microbiological (bacteria, viruses, parasites) and chemical contaminations (nitrate and phosphate). It should be noted that the classification of water between improved and unimproved refers only to the protection of the source and does not consider water quality nor water safety. The latest update published by WHO and UNICEF attests that still in 2015, almost half of the population in sub-Saharan Africa uses unsafe water for consumption and sanitation purposes (WHO. and UNICEF., 2015). This is often the result of a laborious geographical environment and a context of economic development often portrayed in rural areas of Africa. As a result, 83% of the population without access to safe drinking water lives in rural communities (WHO. & UNICEF., 2013).
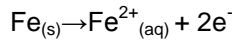
In Africa, the lack of drinking water is not necessarily caused by the scarcity of water resources but may be due to the lack of appropriate treatment (Thompson, 2015). It has been shown in the past that between 30 % and 50 % of water treatment systems fail after 3 to 5 years in rural areas due to a lack of maintenance and management between the organizations in charge of these facilities (Walters and Javernick-Will, 2015). Implementation of appropriate drinking water treatment in remote areas not connected to centralized treatment networks should be selected based on: (i) local water quality, (ii) regional specificity, (iii) ease of operation and maintenance, (iv) use by non-expert users (Glade and Pagilla, 2015).

Electrocoagulation (EC) is a promising alternative to conventional chemical coagulation since it does not require the use of additional chemicals (Bukhari, 2008) (Ricordel et al., 2014) (Delaire et al., 2015) (Moreno-Casillas et al., 2007). The development of this technology is of great interest in the context of developing countries, as it is rather simple and easy to use and requires little maintenance and cost. Therefore, electrocoagulation has been considered as a rising decentralized water treatment (Dubrawski and Mohseni, 2013) (Holt et al., 2005), for the treatment of organic matter (Ulu et al., 2014) (Ulu et al., 2015) (McBeath et al., 2018) and microbial contamination (Ndjomgoue-Yossa et al., 2015). Electrocoagulation has been shown to be effective on suspended colloidal particles, organic matter, bacteria and viruses [9, 15]. One of the main potential benefits of this technology is that it can be powered by solar panels, making it easily usable in rural areas. Indeed, electrocoagulation powered by solar energy has been tested in previous studies and has given promising results (Sharma et al., 2011) (Nawarkar and Salkar, 2019). These authors have shown that solar-powered electrocoagulation (SPEC) is effective for water treatment and that more data would be valuable.
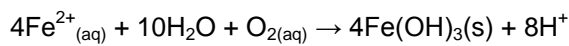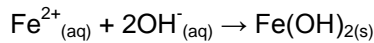
EC involves *in situ* formation of coagulants by the dissolution of metal from the anode with an electric current between two electrodes in solution. The role of coagulants is to destabilize colloids by decreasing repulsive forces and to improve the agglomeration of flocs for a better solid/liquid separation (Moussa et al., 2017). In the EC process, several parameters can greatly influence the performance of the treatment: (i) charge loading (applied intensity x treatment time), (ii) current density (applied intensity/anode surface area), (iii) pH (Dubrawski and Mohseni, 2013) (Den and Huang 2006), (iv) space between electrodes, (v) type of anodes, (vi) pollutant concentration and (vii) conductivity of the solution (Hakizimana et al., 2017). The first parameter controls the coagulant dosing rate or the metal loading rate and is dictated by Faraday's Law (Den and Huang, 2006).

Therefore, for a defined water quality, a too high charge loading results in excessive metal dissolution and affects the residual Fe concentration and the turbidity removal. On the other hand, a too weak charge loading does not guarantee a generation of coagulant sufficient to ensure good coagulation. Charge loading is one of the most important parameters and is always related to a specific active electrode area. This active electrode area introduces the notion of current density. The current density and the charge loading are interconnected since they both depend on the current intensity. The removal of inorganic, organic and microbiological contaminants by EC is a complex mechanism

involving metal dissolution and the formation of metal hydroxides occurring at the anode and the bulk solution, respectively. On the other hand, the hydrogen evolution reaction occurs at the cathodic side (Benaissa et al., 2016) (Moussa et al., 2017).
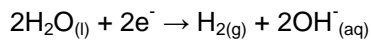
Anode:

$Fe_{(s)} \rightarrow Fe^{2+}_{(aq)} + 2e^-$

Bulk solution:

$Fe^{2+}_{(aq)} + 2OH^-_{(aq)} \rightarrow Fe(OH)_{2(s)}$

$4Fe^{2+}_{(aq)} + 10H_2O + O_{2(aq)} \rightarrow 4Fe(OH)_3(s) + 8H^+$

Cathode:

$2H_2O_{(l)} + 2e^- \rightarrow H_{2(g)} + 2OH^-_{(aq)}$

The objective of this project is to evaluate the performance of SPEC as a promising water treatment system that can be used continuously for the treatment of water from areas not connected to a drinking water network that meets the guidelines of the World Health Organization (WHO). These regulations imply a residual turbidity of less than 1 NTU, the absence of fecal coliforms and a residual chlorine concentration of 0.5 mg/L. This study focused on certain operating parameters such as current density and charge loading in order to evaluate the effectiveness of EC as part of a surface water treatment system. In addition, the feasibility of SPEC via the direct connection mode (without batteries and charge controller) has been tested.

1   **Methodology**

1.1   **Simulated rural surface water**

The initial effluent was obtained from the *Quebec Est* municipal wastewater treatment plant after secondary treatment (after biofiltration and prior to UV disinfection) in Quebec City. In order to simulate the water quality of African rural surface waters, the initial effluent was modified based on the characteristics of a previous study in terms of turbidity and fecal coliforms (Zamxaka et al., 2004). Thus, a concentration of 0.14 g/L of kaolin was added to increase the concentration of suspended colloidal particles to achieve a turbidity of about 100 NTU. A concentration of 5 mg/L of humic acid was added to simulate the presence of natural organic matter (NOM) found in African surface waters (Volk et al. 2002). It should be noted that the composition of organic matter in drinking water sources

is very complex and varies considerably from one source to another. Although humic acids represent an important fraction of the NOM, other compounds such as fulvic acids and hydrophilic compounds may be present. Since the effluent was not sufficiently conductive (0.6 mS/cm), sodium sulfate was added to the water to increase the conductivity of the solution. A concentration of 1 g/L of Na_2SO_4 was used for a current density of 4.55 mA/cm^2 and 3 g/L for higher current densities. The conductivity of surface waters in Africa can vary considerably from 0.12 to 1.77 mS/cm (Nkambule et al., 2012). The initial pH ranged from 6.5 to 7.

1.2 Electrocoagulation reactor using DC electrical power supply and PV panels

The reactor consisted of the EC reactor and a recirculation tank (Figure 1). The total volume distributed between the two parts was 2.3 L. In total, eight electrodes composed of four iron anodes and four graphite cathodes were used. Aluminum and iron anodes are the most widely used (Moussa et al., 2017). In this case, iron was used as a safer option preventing the release of aluminum ions in the water, which could be responsible for diseases such as Alzheimer's disease. Graphite cathodes were used for its non-damaging material and for its disinfecting ability to produce hydrogen peroxide. The dimensions of the electrode were (10 × 11 × 0.1) cm (L × W × T) with a total active surface area of 110 cm^2 and a spacing of 1 cm between each electrode. The treated water was recirculated by a peristaltic pump at a flow rate of 31.28 mL/min. The electrodes were connected to a DC power supply used to impose a specific current intensity (maximum rated current of 70 A at a 40 V open circuit potential) (XFR 40–70, Xantrex Technology, B.C, Canada). The iron electrodes were cleaned with a 3.7% HCl solution for one hour before each test to remove the oxide layer on the surface. All tests were performed in batch mode at room temperature (20°C). A volume of 100 mL of the initial sample was collected before each test and stored at 4°C for later analysis. The sludge was separated from the supernatant water using a decantation column after addition of a polymer (Zetag 7563, BASF) at a concentration of 10 mg/L and stirred for 1 minute. For each test, 100 mL of supernatant was collected after a settling time of 20 minutes and stored at 4°C for later analysis.

1.3 EC solar panels

After optimizing the operating parameters of the EC, the SPEC system was evaluated by replacing the power supply with two 25 W solar panels made of monocrystalline silicone connected in series (maximum power current of 1.55 A and a maximum power voltage of 21.6 V) (Fig. 1). The EC reactor was directly connected to the solar panels during testing without using batteries. These tests were carried out on the rooftop of the INRS – ETE building at the end of July 2016.

1.4 Analytical measurements

Turbidity was measured with a turbidity meter (HACH 2100N). An estimate of the removal of dissolved organic carbon (DOC) was obtained using UV absorbance at 254 nm with a spectrophotometer (VARIAN Cary 50 UV-VIS). To find the relation between UV 254 nm and the DOC value, a DOC standard and a UV 254 nm phthalate standard were used. Given the proportion of carbon in phthalate, it was possible to establish a relationship between DOC (mg/L) and UV absorbance at 254 nm. The main objective of this analytical estimation method was to easily evaluate the organic carbon concentration with simple technical resources. A control solution was tested to verify the correlation between the two methods. Controls were analyzed with a Shimadzu VCPH with a calibration curve between 0 and 5 mg/L. The error percentage was 12.3%, which means that the estimate of dissolved organic carbon is greater than the theoretical result. This error percentage did not affect the removal percentages, as it was considered both for initial and final samples. The pH was measured using a pH meter (Fisher Scientific Accumet Excel XL25). For tests using solar panels, a multimeter (Fluke 87V Industrial Multimeter) was used to measure voltage and intensity. The solar UVA and visible light intensities were measured using a radiometer (DO9721 - Delta OHM, Padova, Italia). The fecal coliforms concentration was evaluated according to the 9222 D method for the examination of water and wastewater (APHA 2002). Residual metal concentrations were monitored by ICP-AES where samples were acidified to a final HNO_3 concentration of 0.2 %.

2 Results and discussion

2.1 Effect of current density on EC process

The current density is an important parameter to consider in EC because it is related to the concentration of metal ions and hydroxides in solution. Although the current density was varied, the metal dissolution was not held constant. This could affect the results since an excessively high current

density can lead to an accumulation of residual metal ions in the solution, which would reduce the turbidity removal efficiency. In this study, different current densities were first tested both in bipolar (BP) and monopolar (MP) configurations using a power supply. In the MP configuration, all the anodes and cathodes are connected to the power supply and each side carries the same charge. In this configuration, the applied current is divided between the electrodes, which gives a voltage lower than that of the BP configuration. The BP configuration is defined when the external anode and cathode are connected to the power supply while the inner electrodes are not connected. Once the current flows through the outer anode and cathode, each adjacent side of each next inner electrode is polarized. In this case, the larger interelectrode distance gives a higher potential difference. When standardized in terms of current densities, the MP and BP configurations may have relatively similar removal efficiencies but may differ in terms of ease of use and operating costs. The imposed intensities were calculated according to the MP or BP configurations. The current densities examined ranged from 0.91 to 10 mA/cm^2 and were selected based on previous research and publications (Dubrawski et al., 2013, Dubrawski and Mohseni, 2013). As shown in Fig. 2, in BP configuration, except for 0.91 mA/cm^2, turbidity, DOC and UV 254 removal increased with current density up to 4.55 mA/cm^2, to remain relatively constant at higher current densities. The optimal current density was 4.55 mA/cm^2 with optimal removal percentages of 97.8 %, 95.3 % and 97.7 % for turbidity, DOC and UV 254 nm, respectively. For monopolar configuration (Fig. 3), a significant decrease in removal efficiency was observed at current densities greater than 1.82 mA/cm^2. The optimal current density was 1.82 mA/cm^2 with removal percentages of 97.8%, 90.5% and 92.4% for turbidity, DOC and UV 254 nm, respectively. It is important to note that, even though the optimal current density was lower in the MP configuration, the removal percentages obtained with the BP configuration were slightly higher than those achieved with the MP configuration. In addition, at the optimal current densities, the charge loading (Q) was lower in MP configuration (0.5 A.h/L) compared to the BP configuration (0.625 A.h/L). The different behavior between BP and MP electrocoagulation systems could be explained by the difference in charge loading values.

2.2 Effect of charge loading (Q)

Charge loading is an essential factor because it controls the amount of dissolved metallic ions. The latter is defined by Faraday's law (Moussa et al., 2017) given in the following equation:

$$m_{Fe} = \frac{M.I.t}{z.F}$$

Therefore, the coagulant dose and reaction rate are proportional to the charge loading. Since charge loading ($Q = I.t$) influences the generation of metal ions, evaluating both the effects of current density and charge loading on the removal of turbidity and organic matter could facilitate the use of EC as a water treatment. For this study, treatment times of 15, 30, 45, 60 and 90 minutes were tested for both configurations at their respective optimal current density. The treatment time was tested for both configurations in order to determine whether the system performs the same way for similar charge loading ranges. Turbidity and DOC removals were plotted as a function of the charge loading for both configurations (Fig. 4). According to the results, it may be observed that regardless of the configuration and current densities (MP – 1.82 mA/cm^2 and 4.55 mA/cm^2), the turbidity removal tends to follow slightly the same pattern. Indeed, at a charge loading less than 0.25 A.h/L, the amount of coagulant generated was not sufficient to completely destabilize the suspended solids, which resulted in a partial turbidity removal. On the other hand, an increase in charge loading (> 0.25 A.h/L) resulted in an increase in the production of metal ions and a better turbidity removal. However, a charge loading greater than 0.83 A.h/L did not favor better turbidity removal since it remained relatively the same. Therefore, under these specific currents, the type of configuration had a minor influence on turbidity removal. In agreement with these results, a study on the effect of electrode connections showed that the removal efficiency was similar in terms of turbidity (Demirci et al., 2015). However, the removal of organic matter (DOC) as a function of the charge loading did not show the same trend. Under these conditions, the BP configuration allowed better organic matter removal than the MP configuration. For a charge loading of 0.83 A.h/L, a 95.22 % DOC removal was achieved in the BP configuration compared to 88.65 % in the MP configuration. In the BP configuration, some sides of the graphite electrodes are positively polarized in comparison to the MP configuration, where each graphite electrode is only negatively charged. As a result, the graphite electrodes are oxidized, releasing carbon matter into the solution, which could interfere with the removal of DOC.

2.3 Optimization and WHO recommendations

The chosen configuration of the EC process (60 minutes of treatment in BP configuration, 4.55 mA/cm^2) gave a final turbidity of 4.60 NTU. This configuration was chosen since both turbidity and UV 254 nm were strongly eliminated. Under these experimental conditions, the average voltage was 31.60 V, leading to an energy consumption of 6.87 kWh/m^3. This energy consumption is higher than that of similar studies on the use of EC to remove natural organic matter from drinking water sources (i.e. ((Alimohammadi et al., 2017)). The difference could be explained by a larger gap between electrodes, a longer treatment time and a more complex water to treat. The final pH was 7.60 ± 0.20. The initial iron concentration was 0.049 mg/L and the final concentration was 2.47 ± 0.50 mg/L. Although WHO does not have guidelines for residual iron, a limit of 0.3 mg/L Fe_{total} would be ideal, according to a study on aesthetic recommendations for drinking water (Sain and Dietrich, 2015). However, according to the same study, 0.05 mg/L to 10 mg/L of iron is a common concentration range in source waters. In order to evaluate the disinfection potential of EC, the initial and final *E.coli* concentrations were measured. The initial *E.coli* concentration was 2.37 ± 0.19 x 10^4 CFU/100 mL, while the final concentration was 1.83 ± 0.23 x 10^3, resulting in an elimination of 92.3 ± 3.4 % and a 1.11 log reduction. To comply with WHO recommendations for residual turbidity and *E.coli* concentration, an additional water treatment system, such as a filtration column (sand, anthracite, activated charcoal) would be required. In addition, a solar-powered UV disinfection system could be used as a final disinfection step, while a chlorine dosing pump would be necessary to ensure a chlorine residual of 0.5 mg/L.

2.4 Electrocoagulation powered by solar panels

For the second part of this study, the power supply was replaced by two solar panels connected in series directly to the EC reactor. For SPEC tests, solar radiation intensity varied with the time of day during which the experiments were conducted. This series was conducted on July 26th, 2016 at 10:00 A.M, 1:00 P.M and 4:00 P.M. The turbidity as a function of the treatment time (0 - 45 minutes) and the intensity generated by the solar panels is given in Fig.5. The three tests carried on the same day gave very different outcomes from each other. This variation results from the addition of a second parameter: the solar radiation intensity. For the test at 10:00 A.M, turbidity decreased rapidly after only 15 minutes and increased slowly over the rest of the test. The solar radiation intensity generated

was higher at the beginning of the test (1.3 A) and then decreased considerably after 15 minutes (0.5 A). For the test at 1:00 P.M, the solar radiation intensity was low at the beginning (0.5 A) and very high after 15 minutes (1.2 A). As a result, turbidity increased due to an accumulation of total solids produced by the dissolution of the iron anodes. For the test at 4:00 P.M, the solar radiation intensity was also low during the first minutes (~0.3 A), then higher at 15 minutes (1.05 A), but remained low for the rest of the test (0.3 A). This lower intensity may explain why turbidity did not increase as in the 1:00 P.M test. Overall, the lowest turbidity (6.7 NTU) was achieved after 15 minutes of treatment with the test at 10:00 A.M. Two exhaustive conclusions can be drawn from this series of tests. Firstly, it is obvious that there is a strong relationship between solar radiation intensity and turbidity removal. The solar radiation intensity determines the intensity of the generated current, the current density and the charge loading. As a result, the solar radiation intensity also determines the metal loading rate, which will affect the potential for turbidity removal. As mentioned previously, if the metal charge is excessive in relation to the coagulation demand, turbidity will increase. Secondly, it is possible to conclude that in order to obtain the best results for SPEC, treatment time and current intensity must be controlled simultaneously. As can be seen from Fig. 5, turbidity removal is strongly correlated to the charge loading. Therefore, it is possible to establish the charge loading necessary to obtain an appropriate turbidity removal by adjusting the treatment time as a function of the light intensities received or the current intensities generated by the solar radiation. Although this would be possible, the performance of the system would depend on the solar radiation received and the local weather conditions. Thus, the use of batteries could compensate for variations in solar radiation when the weather conditions are not optimal.

Conclusion

The main objective of this study was to investigate the feasibility of using electrocoagulation powered by solar energy as a water treatment system that could be used in rural areas such as Sub-saharan Africa. For the first part of this study, the operating parameters of the EC using iron electrodes were optimized with a DC power supply. Optimal removal of turbidity, UV 254 nm and organic matter was achieved at 4.55 mA/cm^2 with 60 minutes of treatment in bipolar configuration (97.0%; 93.0%; 95.2%). These parameters suggest that 0.83 A.h/L is the ideal charge loading for best results. The EC system was then tested using solar panels as a power supply without batteries and a charge controller. It was

concluded that the removal efficiency is directly determined by the solar radiation intensity related to natural weather conditions. The highest removal of turbidity (92.1%) was achieved at 10:00 AM after 15 minutes of treatment. Adaptation on a larger scale can be facilitated using a charge controller to regulate the intensity of the generated solar current. Thus, by controlling the current intensity supplied to the system, the treatment time or the hydraulic retention time can be modified while keeping the charge loading constant. At the same time, the current density can be kept constant by varying the number of electrodes. Given these results, SPEC has a lot of potential as part of a water treatment system that could be tested in rural or non-connected areas.

List of notation

m_{Fe} is the mass of dissolved iron (g)

I is the current intensity (A)

T is the time (s),

z is the number of electrons transferred per atom and

F is the Faraday's constant (96,485 C/mol).

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Figure captions

Figure 1. Experimental solar electrochemical reactor (reactor volume: 2.3 L, 31.28 mL/min, 4 Iron electrodes and 4 graphite electrodes). **A**: solar panels, **B**: Radiometer with UVA and visible probes, **C**: pump, **D**: electrocoagulation reactor, **E**: multimeter

Figure 2. Efficiency of the bipolar configuration in function of the current density (4 iron electrodes, 4 graphite electrodes, Treatment time: 60 min, flow rate: 31.28 mL/min, working volume: 2.3 L, surface simulated water, 10 mg/L cationic polymer). White bar = turbidity; Black bar = UV 254 nm; Grey bar = DOC

Figure 3: Efficiency of the monopolar configuration in function of the current density (4 iron anodes, 4 graphite cathodes, treatment time: 60 min, flow rate: 31.28 mL/min, working volume: 2.3 L, simulated surface water, 10 mg/L cationic polymer). White bar = turbidity; Black bar = UV 254 nm; Grey bar = DOC

Figure 4: Efficiency of the bipolar and monopolar configurations in function of the charge loading (0.5 A for BP, 0.8 A for MP, Treatment times: 15, 30, 45, 60 and 90 mins, 4 iron electrodes, 4 graphite electrodes, 100 RPM, Flow rate: 31.28 mL/min, working volume: 2.3 L, surface simulated water, 10 mg/L cationic polymer). (■) MP-turbidity, (▲) BP-turbidity, (●) MP-DOC (♦) BP-DOC

Figure 5: Turbidity in function of the received light intensity at 10 am, 1 pm and 4 pm. White bar = 10:00 AM; Grey bar = 1:00 PM; Black bar = 4:00 PM, (♦) 10:00 PM Intensity (▲) 1:00 PM Intensity, (■) 4:00 PM Intensity
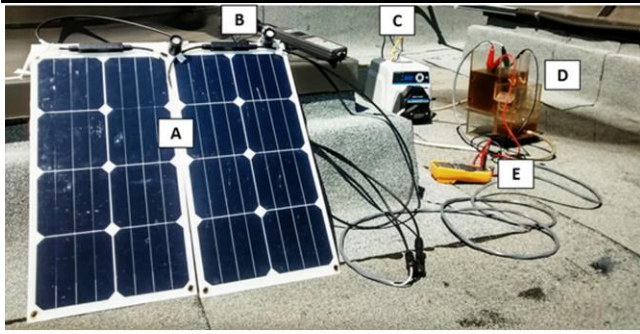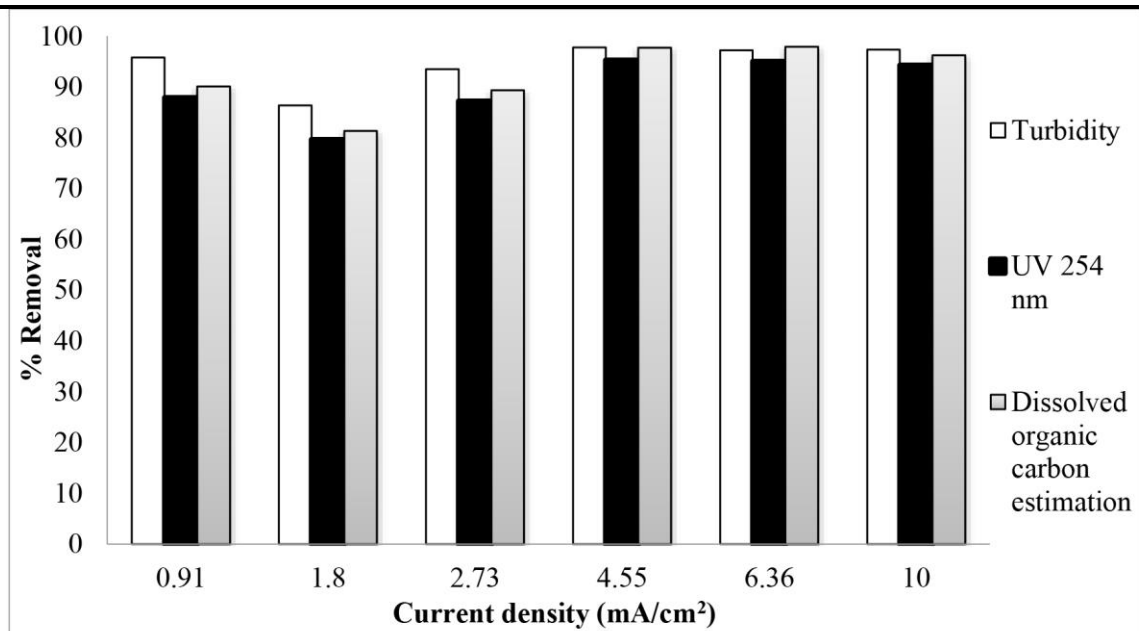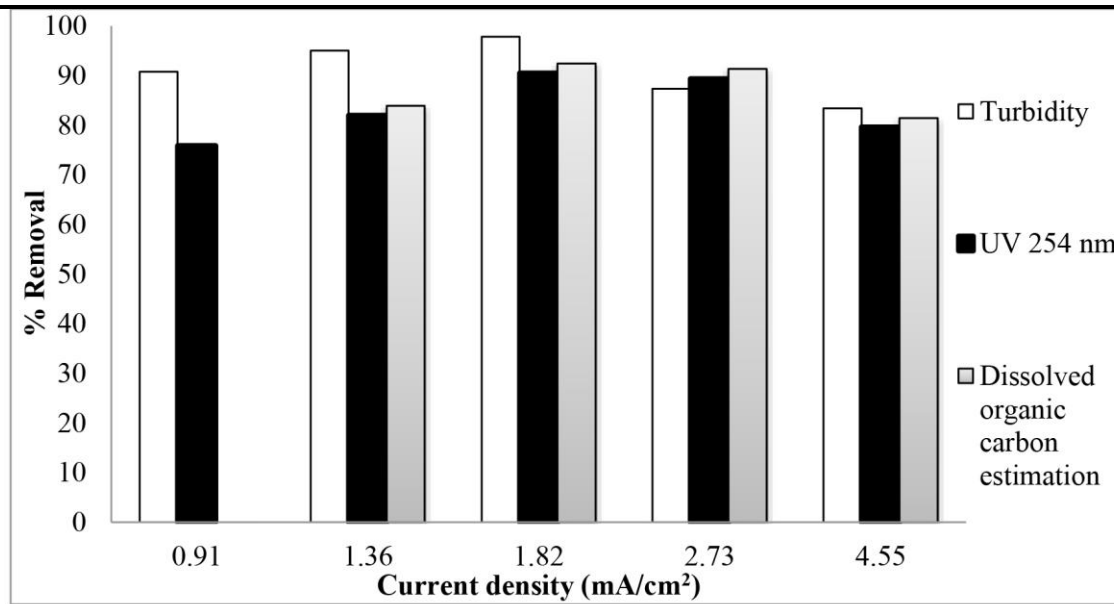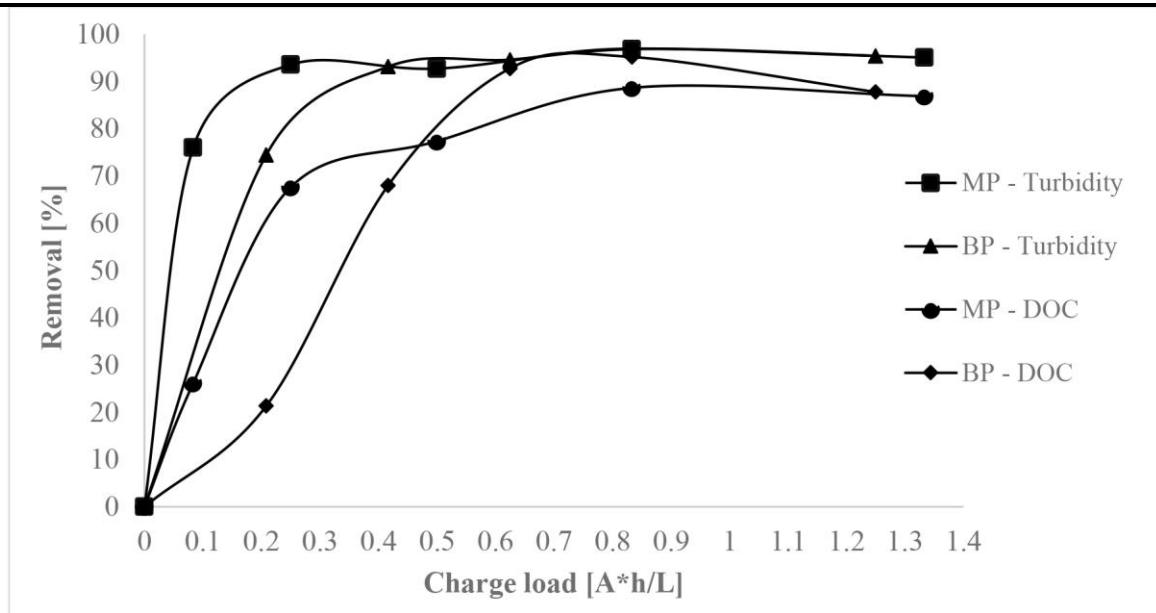
Figure 1_ 600dpi.jpg

Fig. 2

Fig. 3

Fig. 4

Fig. 5