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Electrocoagulation at low conductivity for drinking water treatment

Pilot Research, Cost estimation and LCA

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Electrocoagulation for drinking water treatment
Pilot Research

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Project manager
Erwin Beerendonk

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TKI, Brabant Water, Dunea, De Watergroep, QStone Capital

Author(s)
Julian Muñoz Sierra (KWR), Chuan Jiet Teo (KWR), Ruud Steenbeek (Dunea), Cesar Ortega (QSTone Capital)

Quality Assurance
Emile Cornelissen

Sent to
Project Partners

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More information
Dr.EngD.ir. Julian Muñoz Sierra
T +31 6 52891321
E julian.munoz@kwrwater.nl

PO Box 1072
3430 BB Nieuwegein
The Netherlands

T +31 (0)30 60 69 511
E info@kwrwater.nl
I www.kwrwater.nl

Summary

Electrocoagulation (EC) is a process in which iron sacrificial electrodes are submerged (and slowly dissolved) in water driven by an electric current, in which dissolved iron acts as a coagulant. It has been seen as a promising alternative to conventional chemical coagulation in drinking water treatment. The major reasons to consider electrocoagulation over conventional coagulation are (i) no chemical dosing, (ii) the creation of more settleable and heavier flocs, (iii) cheaper operational costs, (iv) a smaller environmental effect, and (v) no pH decrease and chloride increase in the treated water. Although electrocoagulation has been successfully applied to wastewater, it is still unknown whether successful application in sources for drinking water (in the Netherlands) is achieved with specific targets such as suspended solids, turbidity and TOC removal. Therefore, a pilot study on sources for drinking water was carried out to corroborate promising results already found at lab-scale experiments.
In this pilot study, the potential of electrocoagulation (EC) was investigated for treating three different types of water with a relatively low conductivity: surface (river) water from Dunea, backwash water from rapid sand filters of Dunea, and groundwater from Brabant Water. This project aims to evaluate electrocoagulation for treating water compared to conventional $FeCl_3$ coagulation, by means of a pilot removal efficiencies, cost calculations and Life Cycle Assessment. It includes conceptual design, cost estimates, and environmental impact analysis, focusing on Ecopoints and CO_2 emissions to assess electrocoagulation's potential.

The results showed that electrocoagulation was capable of achieving similar removal efficiencies compared to conventional coagulation but higher iron dosages were needed to get floc formation. For sand filter backwash water, with a Fe dosage of approximately 42 mg/L for EC, removal efficiencies were 80% for arsenic, 75% for total suspended solids, 86% for total phosphorus, 43% for total organic carbon, 56% for turbidity, and 48% for total color. After 8 hours of settling, turbidity removal improved to 64-93% and color removal to 56-88%. Conventional FeCl3 coagulation achieved turbidity and color removals of 78-85% and 63-86%, respectively.

The investment and operational costs for EC and conventional iron chloride dosing were determined by using available cost calculation tools and two workshops with the experts from the project partners. The flocculation and settling unit (lamella settling) accounts for the largest share of investment costs in both cases. The investment cost difference for Dunea sand filter backwash water is 556 k€ higher for electrocoagulation when considering about 604 k€ for the EC unit. For Dunea's benchmark on surface water, a difference of higher investment cost of 4452 K€ was determined for electrocoagulation. The total operational costs of the EC are 269 k€/year compared to 125 k€/year for iron chloride dosage treating 79 m3/h sand filter backwash water. For surface water, the operational costs are 2632 k€/year for electrocoagulation due to mainly the energy consumption cost of the EC being about 991 k€/year at an actual cost of 0.18 €/kWh, compared to about 1255 K€/year of conventional coagulation in which the main contribution is due to chemicals consumption costs. The sensitivity analysis indicates that energy price and consumption are the primary drivers of EC operational costs. Reverting energy prices to historical levels e.g. 0.09 €/KWh could reduce total OPEX by 19.6% for surface water and 15.6% for sand filter backwash water. Further reductions of up to 35% for EC are possible by cutting energy consumption. Iron electrode costs have minimal impact, while FeCl3 costs significantly influence OPEX, with potential reductions of 11% for surface water and 2% for backwash water if costs decrease. Optimizing energy efficiency to 5 KWh/kg Fe could make EC more financially and technically viable.

Modelling techniques can be employed to identify the process's key parameters, which can serve as the foundation for a theory-based design of the installation. Furthermore, sharing further knowledge between technologists and engineers from project partners, and technology suppliers can facilitate a more accurate estimation with a smaller band of the investment costs associated with a drinking water EC installation, especially when looking at a possible

modular/flexible configuration. Energy efficiency (KWh/kg Fe), the use of larger modules/electrodes for EC at high capacity, lifetime trade-off of the electrodes, and the feasibility of the technology given prevailing energy prices are critical factors that need to be carefully considered.
The Life Cycle Assessment (LCA) compared traditional coagulation and electrocoagulation methods for treating surface water and sand filter backwash water, using the ISO 14040/44:2006 framework, SimaPro 9.2 software, and the Ecoinvent database. For both water sources, the assessment within the boundaries selected indicated that EC benefits are limited, where the ecopoints are 6.14 µPt, 43.49 µPt, 17.21 µPt and 39.79 µPt for Coagulation sandfilter backwash water, Electrocoagulation sandfilter backwash water, Coagulation surface water and Electrocoagulation surface water respectively. Traditional coagulation had higher impacts on mineral and metal resource use, acidification, and eutrophication due to chemical inputs whereas electrocoagulation showed relatively higher climate change impacts due to electricity consumption. The environmental performance of electrocoagulation was heavily influenced by electricity consumption, highlighting the importance of energy sources. If iron dosage for electrocoagulation could be reduced, then the electricity consumption would be lower. Sensitivity analysis revealed that full renewable energy source would significantly (64-80%) reduced the environmental impacts measured in ecopoint of electrocoagulation.

Overall, electrocoagulation offers comparable removal efficiencies to conventional coagulation but required higher iron dosages based on the EC pilot configuration used which will make it incurs in higher costs, mainly due to energy consumption and efficiency, which significantly impacts its environmental performance. For drinking water applications, further optimization of the EC configuration is needed to obtain floc formation at similar iron dosages than CC and overcome any challenges imposed by the low conductivity of the water.

Contents

1 Introduction

1.1 Background

Electrocoagulation (EC) is a water treatment process mainly used to treat highly polluted water streams. It applies direct current to sacrificial electrodes that are submerged in an aqueous solution. Electrocoagulation involves the in situ formation of precipitating Fe-hydroxide salts by dissolving sacrificial Fe (0) anodes. The Fe (0) oxidizes to dissolved Fe (II), which further oxidizes to Fe(III) in the presence of oxygen, resulting in the precipitation of Fe flocs in water. EC is a straightforward and efficient technique to remove the flocculating agent produced by electro-oxidation of a sacrificial anode and generally made of iron or aluminum. Electrocoagulation is currently mainly used in wastewater treatment in which water has a medium or high conductivity, for the removal of for example, dyes, dissolved toxic organic compounds and cyanides (Larue et al 2003). Heavy metals and phosphate can also be easily removed with this technique. Excipients such as polyelectrolyte or other flocculation aids (e.g. FeCl3) are usually not needed.
The metal of the anode (usually Fe(0)) dissolved by forming ions, which act as coagulant. Electrolysis at the cathode creates hydrogen gas and nitrogen gas is created by reducing nitrate present in the water. There can be safety issues for closed installations or potential use of these gases. Oxidation/reduction of dyes and micropollutants present in the water can also take place on the electrode surface. The resulting flocs can be relatively small, part of which can be separated by means of sedimentation or flotation.

During drinking water production, iron (Fe) coagulants are dosed ($FeCl_3$, $FeSO_4$ etc.) for (rapid) sand filters, e.g. to remove turbidity, NOM and arsenic and to treat filter backwash water. The water and wastewater sectors in the Netherlands and Europe are committed to reducing their carbon footprint and therefore reducing the use of chemicals as much as possible is desirable. Electrocoagulation may be a promising alternative to chemical coagulation because of its CO_2 footprint and efficiency. Slow sedimentation and low dry matter content of iron sludge are often reported in applications of metered coagulants in drinking water production. The use of electrocoagulation can potentially result in a more compact settled sludge compared to the sludge from conventional coagulation [Larue et al, 2003]. The reasons for investigating electrocoagulation compared to conventional coagulation are:

- Electrocoagulation does not require dosing of chemicals; therefore, costs and risks associated with transportation and storage of coagulants can be reduced;

- Flocs generated by electrocoagulation can be more compact and settle faster compared to conventional coagulation;

- Based on the previous laboratory preliminary study of this project (KWR 2021.069), electrocoagulation appeared to have lower operating costs than conventional coagulation for sand filter backwash water from another location from Dunea (180 m3/h). This preliminary study showed about 50% lower operating costs for electrocoagulation (10.4 k €/year) compared to conventional coagulation (20.6 k €/year), based on an iron dosage of 13 mg Fe/l, similar for both EC and $FeCl_3$, EC energy consumption of 0.05 kWh/m3, iron electrode costs of 0.007 €/m3, energy efficiency of 3.8 KWh/kg Fe and energy cost of 0.004 €/m3 (based on lab data).

- Electrocoagulation does not lower the pH (no pH correction is needed) during water production and does not add chloride to water.

1.2 Electrocoagulation in drinking water production

EC has been shown to effectively remove a number of common drinking water contaminants such as arsenic (Roy et al. 2021), fluoride (Mameri et al. 2001, Zhu et al. 2007), clay (Holt et al. 2002), cobalt (Mansour et al. 2012) and viruses (Zhu et al. 2005). In this section, several insights from previous studies on the removal efficiency of EC are presented structured by the removal of different contaminants:

1.2.1 NOM removal

McBeath et al. (2020) evaluated recently the efficiency of electrocoagulation at a pilot-scale as an alternative drinking water treatment technology to conventional coagulation. Using iron anodes (A1008 cold-rolled steel iron), the influence of different parameters such as dissolved Fe (concentration), current density and inter-electrode gap on the reduction of natural organic matter (NOM) was studied using lake raw water. The four dissolved iron concentrations investigated were 27.8, 38.2, 51.1 and 60.8 mg/L, achieved by applying constant current at 16.0, 22.0, 29.4 and 35.0 A, respectively. Two inter-electrode gaps were investigated (1 and 2 mm). The results showed a clear effect of Fe dosage on the removal of DOC and UV254. DOC decreased by 29.2±5.6%, 35.1±4.2%, 37.1±3.4% and 37.2±4.2%, while UV254 was decreased by 38.7±3.0%, 48.8±2.5%, 52.5±1.2% and 54.7±0.9% for Fe dosage of 27.8, 38.2, 51.1 and 60.8 mg/L, respectively, for all current densities and inter-electrode gaps tested. As Fe dosages increased from 27.8 to 60.8 mg/L, specific ultraviolet absorbance decreased from 1.92±0.14 to 1.60±0.10 L/m.mg respectively, from an initial raw water value of 2.21 L/m.mg. In general, conditions that yielded the highest removal of NOM, a 1 mm gap and 4-cell configuration, had energy requirements between 0.480 and 0.602 kWh/m^3 of water treated. These results indicate that similar NOM removal performances can be achieved at both energy intensive and energy non-intensive conditions.

Lower energy requirements (0.11 kWh/m3) for an iron EC process for NOM removal has been reported at a small-scale batch but no fair comparison to full scale system can be made (Alimohammadi et al. 2017).

The presence of high concentrations of natural organic matter (NOM) and total organic carbon (TOC) in e.g., groundwater can impact the speciation of aqueous Fe(II) by complexation (Sundman 2014). Successful removal of NOM using EC has also been demonstrated at lab scale (Dubrawski et al. 2013, Dubrawski and Mohseni 2013, Jiang et al. 2002).

Research involving the effect of three NOM sources (Suwannee River (USA) DOC = 13.79 mg/L, Nordic Reservoir (Norway) DOC = 9.03 mg/L, and a natural source Lost Lagoon (Canada), DOC = 13.31 mg/L)) was carried out by Dubrawski et al. (2013). EC was found to predominantly remove NOM with higher concentrations of large hydrophobic and high molecular weight fractions. Moreover, relative reduction in DOC with two different initial NOM concentrations (13.79 and 21.59 mg/L) was not significantly different, performing equally for either treatment at the same EC operating conditions.

NOM of different origins (commercial, terrestrial and natural water) were treated by electrocoagulation (EC) process using aluminum, iron and hybrid electrodes (Al+Fe) (Ulu et al. 2015). The lowest effluent concentration of DOC was obtained as 5.05 mg L^{-1} with hybrid electrode for natural NOM source at its original pH 7.3. In addition, among the metal types, the best UV-abs-254 removal efficiency was obtained as 92.4% with 0.0312 cm^{-1} by hybrid electrode. The color removal efficiency of water occurred successfully by Al and hybrid electrodes. Aquatic NOM source was the most resistant to EC treatment with DOC reduction of 71.1%, 59.8%, and 68.6% for Al, Fe and hybrid electrodes, respectively. SUVA value was reduced to below 2 for three NOM sources studied. In another similar study, Ulu et al. (2014) found that the highest treatment efficiency for humic acids achieved 87% ($DOC_{treated}$ 2.1 mg/L) at an initial pH of 4 with iron electrodes.

Although EC has been previously researched for drinking water applications, much of the research has been bench-scale, batch (Ben-Sasson et al. 2013, Mahvi et al. 2011, Vepsäläinen et al. 2009) operations using synthetic water with humic acid and/or reverse-osmosis isolated NOM. A very limited number of studies incorporating continuous flow reactors exist and although successful, these remain at small bench-scale operations with flow rates below a practical capacity for piloting (McBeath et al. 2020).

Summarizing, NOM can be effectively removed by EC, depending on factors such as Fe dosage, current density, and electrode gap. Higher Fe dosages improve DOC and UV254 removal, with energy requirements varying based on operating conditions. EC's effectiveness varies with NOM source, and continuous flow reactor studies remain limited.

1.2.2 Nitrate removal

EC has been applied widely to remove nitrate from different types of waters including drinking water (Kumar and Goel 2010, Lacasa et al. 2011b). There is no need for post-neutralization in most cases with EC. Lacasa et al. (2011b) showed that nitrate adsorption onto growing metal hydroxide precipitates appears to be the primary mechanism behind the observed nitrate removal efficiency. The same amount of nitrate removal was measured using either iron or aluminum electrodes. Kumar and Goel (2010) achieved a removal efficiency of 84% for nitrate at 25V using tap water or distilled water after 3 hours of electrolysis, with an initial nitrate concentration of 300 mg/L. When the applied potential was 10 V, a 50.3% nitrate removal was achieved in 6 h. To avoid a decrease in electrocoagulation efficiency, the polarity of electrodes needs to change regularly.

Apshankar and Goel (2020) determined the operating conditions for maximizing nitrate removal using electrocoagulation, settling and filtration with four different types of waters: double distilled (DD) water, ground water (GW), tap water, and untreated water from the local water works plant. The best nitrate removal efficiencies were obtained at high initial concentrations of nitrate, high EC and settling times and high pH. The highest NO_3 removal efficiency of 38.2% was observed with double distilled (DD) water after EC and settling. EC and settling time were 4 h each. Initial pH was adjusted to ≈10. The voltage applied was low (10 V), with an energy consumption of 8.3 Wh/L. The average conductivity was around 1000 µS/cm because of the high initial nitrate concentration (104 mg/L).

Moradi and Ashrafizadeh (2020) evaluated the removal of nitrate from tap water by EC followed by flotation (ECF). An ECF reactor with a working volume of 8 L was used. The effect of different arrangements of anode/cathode electrodes (Al/Fe, Fe/Al, and SS/Fe) were assessed together with the initial pH of the aqueous solution, process duration, electrodes gap, electric potential, and the initial concentration of nitrate. The results showed that the percentage removal of nitrate increases with the initial concentration of nitrate, electrolysis time, the initial pH of the solution, and voltage. To achieve the maximum removal of nitrate, decreasing the gap between electrodes can be more economic than increasing other variables such as voltage. Finally, the results of this research showed that at optimum conditions (initial concentration of nitrate = 120 mg/L; solution pH = 8; voltage = 17 V; gap = 3 cm; electrolysis time = 240 minutes), the removal of nitrate by the Al/Fe electrodes arrangement achieved 93%.

NO3 removal can be achieved using EC depending on the applied voltage, electrode material, pH, and initial nitrate concentration. Higher voltage, lower electrode gaps, and longer electrolysis times improve efficiency, with removal rates reaching up to 93% under optimal conditions.

1.2.3 Phosphate removal

Lacasa et al. (2011a) showed that pH increases with current density, indicating phosphate adsorption onto metal hydroxide. In the case of aluminum electrodes, the coexistence of both direct precipitation and adsorption onto metal is observed, whereas in the case of iron electrodes, the adsorption mechanism is less effective. Due to the higher solubility of the iron phosphate, iron hydroxide was the main product of coagulation when iron electrodes were used. The optimum conditions to remove phosphates by electrocoagulation are at the lower current densities (<3.0 mA cm^{-2}) for iron and aluminum electrodes because it is favored the direct metal phosphate precipitation.

EC was investigated by Bektaş et al. (2004) under low phosphate concentrations (10–50 mg l^{-1}) and EC electrolysis duration equals to 5 min, achieving removal efficiencies in the range of 81–90%. The increase of the EC time provided better results on the phosphate. A 10 min EC process duration exhibited a 70 and 94% phosphate removal for 200 and 100 mg l^{-1} phosphate solutions, respectively.

Attour et al. (2016) indicated that the kinetics of the phosphate removal during EC was found to be accelerated by continuous adjustment of the pH. The total removal of phosphate was obtained after 140 minutes without pH adjustment and after only 40 minutes with adjustment of pH every 10 min. About 50% was reached after 20 min of electrolysis with an initial phosphate concentration of 100 mg/L and a current of 10 mA/cm^2. The choice of the aluminum electrode connection modes was found to strongly affect the efficiency of the treatment, and the consumption of energy. The mono-polar in parallels connection mode was found to be the most economic whereas bi-polar in parallels was found to favor a higher kinetic rate of treatment.

Franco et al. (2017) examined the effects of EC treatment parameters (initial pH, initial conductivity, power input, and initial P concentration) on the ability of the EC process to remove P in solutions with initial P concentrations less than 2 mg/L. Higher conductivity was demonstrated to have a higher P removal efficiency. 81% removal of P was achieved after 10 min of reaction time.

Vasudevan et al. (2008) used EC for the removal of phosphate from drinking water using mild steel as the anode and stainless steel as the cathode. The results showed that the maximum removal efficiency of 98% was achieved at a current density of 0.50 mA·cm^{-2} at a pH of 6.5.

In summary, various studies have shown that EC is an effective method for removing phosphate from surface water. The optimal conditions for phosphate removal are generally lower current densities and longer durations of EC. The type of electrode used, as well as the connection modes, can affect the efficiency of the treatment. Additionally, adjusting the pH during the EC process can accelerate the removal kinetics.

1.2.4 Removal of bacteria/viruses

Previous studies have demonstrated the effect of Fe-EC in the removal of a wide range of microorganisms from bacteria to viruses in different water matrices, mainly for drinking water applications (Delaire 2016, Ghernaout et al. 2019, Heffron et al. 2019a, Heffron et al. 2019b, Ryan et al. 2021).

Bacteria removal by Fe-EC was investigated by (Delaire 2016). The results show that removal is primarily due to bacteria encapsulation in Fe(III) flocs and removal by gravitational settling, while inactivation by germicidal reactive oxidants remains limited in the presence of HCO3 and at pH>7. Fe(III) precipitates are found to adhere to the surface of bacterial cells, primarily through interactions with bacterial phosphate groups, resulting in bacteria enmeshment in precipitate flocs.

The effect of major groundwater ions on bacteria/virus removal during EC is interpreted in light of this mechanism: the presence of Ca and Mg reduce bacterial removal by competing with complexing bacterial phosphate groups; Si and NOM, which do not strongly compete with phosphate groups for sorption to Fe(III) precipitates, do not affect attenuation; by contrast, P decreases attenuation significantly, except in the presence of bivalent cations, which can bridge between P sorbed to precipitates and bacterial phosphate groups. Finally, Fe-EC is shown to be equally effective towards Gram-positive and Gram-negative bacteria. Results show that Fe-EC can effectively remove all types of bacterial contamination from a range of groundwaters.

Ghernaout et al. (2019) reviewed the mechanisms for disinfection by applying EC. EC alone is effective for bacteria and algae removal to a certain extent (Barışçı and Turkay 2016, Ghernaout et al. 2008). Tanneru and Chellam (2012) established that for surface water carrying 5 mg-C/L NOM, EC/MF only eliminated 1.5 log of MS2 viruses at 13 mg Fe/L, whereas 6.5 log elimination was evaluated from synthetic water at an identical Fe dosage. The existence of 5 mg-C/L of Suwannee River-humic acid reduced virus elimination by roughly 4-log at pH 6.4 and approximately 2-log at pH 7.5 by EC/MF compared to FeSO4 coagulation/MF (Chellam and Sari 2016).

EC using Al electrodes performs better than EC using Fe electrodes in eliminating viruses from NOM-containing surface waters because of effective Al(OH)3(s) precipitation and enmeshment while Fe did not significantly precipitate following electrolysis upon these situations (Tanneru et al. 2013). Prolonged flocculation/contact times would be required to attain important deactivation through Fe/Al EC of surface water (Chellam and Sari 2016). Boudjema et al. (2014) studied the likely treatment of Oued El Harrach river water by Al EC. A 99% reduction in fecal coliforms and E. coli was obtained and a near-total inactivation of fungi was reached.

Fe EC has been shown to effectively remove a wide range of microorganisms from bacteria to viruses in different water matrices. The removal of bacteria by Fe-EC is primarily due to bacteria encapsulation in Fe(III) flocs and removal by gravitational settling.

1.3 Economic and Environmental Opportunities

The EC approach does not require dosing of chemicals (e.g., dosing of $FeCl_3$), therefore costs related to transportation and storage of chemicals in water treatment plants can be significantly reduced. Reduced demand for chemicals in the drinking water sector is expected to lead to reduced production of chemicals by industry, ultimately leading to reduced CO_2 emissions. Literature has described that flocs generated by electrocoagulation have a higher density and settle faster than flocs from conventional coagulation (Semerjian et al, 2015). Therefore, the costs associated with dewatering the sludge (e.g., extensive use of flocculants) might be reduced. This project promotes the use of iron (electro) coagulation instead of aluminium. The use of iron instead of aluminium is safer from the health point of view of drinking water users.
Furthermore, KWR conducted a preliminary LCA scan using SimaPro 8 based on lab data, and the results indicated that EC could have a significantly lower environmental impact compared to CC when dosing the same iron concentrations, which motivated to gather more detailed information from the operation and carry out a more refined evaluation with pilot results.

1.4 Aim

The primary aim of this project is to evaluate the applicability of electrocoagulation (EC) technology for treating surface water and sand filter backwash water in comparison to conventional coagulation ($FeCl_3$ dosage) by means of an integrated approach consisting of EC pilot operation, Life Cycle Assessment and Cost calculations (CAPEX and OPEX) for implementation.
By determining the necessary conceptual design and settings of the process based on the pilot experiments, overall cost estimates are made and compared with the current conventional coagulation-flocculation with the current process capacity. Additionally, a life cycle assessment is performed to compare the technologies in terms of Ecopoints and CO_2 emissions, so that the environmental impact/benefits of electrocoagulation as an alternative is elucidated.

1.5 Research Questions

Accordingly, with the aim, the following research questions were proposed:

1. What is the best current vs Fe dosage to achieve high and comparable removal efficiency to CC of TSS/Turbidity/ TP/Arsenic for each water type (surface water and sand filter backwash water) at Dunea's location?
2. What is the range of low Fe dosing that can be achieved with EC on top of the sand filtration columns to promote turbidity/suspended solids removal of groundwater at Brabant Water location?
3. What are the main differences in cost when comparing EC with conventional coagulation-flocculation and the applicability/limitations with the surface water and sand filter backwash water?
4. What are the benefits/gains from applying EC in terms of environmental impact (Ecopoints, CO_2 emissions)?.

2 Pilot Research

This chapter describes the materials, methods, and results obtained during the experimental work carried out with the EC pilot plant at Dunea drinking water production location with two different water types (surface water and sand filter backwash water); and the pilot trial at Brabant Water for iron dosing on groundwater using EC.

2.1 Iron Electrocoagulation Pilot at Dunea

2.1.1 Setup

EC experiments were carried out with a pilot-scale electrocoagulation setup (design by QStone Capital). Figure 1 shows a photo of the EC pilot. The EC unit comprises the following major components (see Figure 2) (drawing made by QStone Capital): Feed tank (1000 L), Clean water tank (1000 L), EC system, Settling Tank, Laboratory.

Feed Tank

A submersible pump continuously bring the water from the water basin/uptake point to the feed tank.

Clean water tank

To make the EC system efficient keeping the EC reactor clean is a priority. The bottom part of the system and the spaces between the plates must always be clean and sludge-free. Moreover, from an energy consumption perspective, sludge in between the plates will increase the voltage needed to deliver a certain amount of current, increasing the power consumption and therefore the OPEX. For these reasons, an automated cleaning process was added. From this tank, clean water is used to clean the bottom of the EC reactor through a high-pressure injection of water from both the left and right sides of the reactor.

Figure 1 EC (Qstone) Pilot set-up. Left: EC reactor and flocculation units. Right: Containerized pilot and feed tank.

EC system

The EC reactor that electrochemically treats the water is presented in Figure 1. Before going to the EC reactor, raw influent needs to be corrected if necessary to have a pH that fluctuates between 6.4 and 8.8. This is the optimal range to obtain the flocculation, a reaction that allows the aggregation of several pollutants with ions that come

from a transactional metal, in this case, iron. Apart from pH, the temperature plays an important role and should not exceed 40 °C. Conductivity is also important as it makes the process more energy efficient (less voltage is needed for the same amount of current when higher conductivity occurs).
At the same time, the system can run smoothly just with the influent's water characteristics with no need for pH, conductivity, and temperature correction.

The influent enters the EC reactor where the contaminants aggregate with iron hydroxide complexes. It represents the main process around the EC system. It consists of the following parts:

- The Intake Basin (distribution system) is separated from the electrodes. This basin is used to reduce turbulence at the intake side and feed the intake water below the electrodes until the aeration. (see Figure 2).
- EC reactor. Within this chamber, the wastewater is forced to flow from the bottom of the reactor to the top of it and then overflow to the next chamber. The upcoming flow needs to be mixed well in every section of the lower part, as each plate must reach the entire amount of water entered. All the water that enters needs to be equally EC treated. This is achieved by inserting an Inlet plate to ensure that the water always splashes downwards first. For this reason, the reactor needs to be placed on a leveled floor. The electrodes are made of steel. The aim of EC is to coagulate colloids so that these substances can easily be separated. Coagulant is realized by dissolving the electrodes. When they are dissolved, hydrogen gas (H_2) is released which results in a floatation effect.

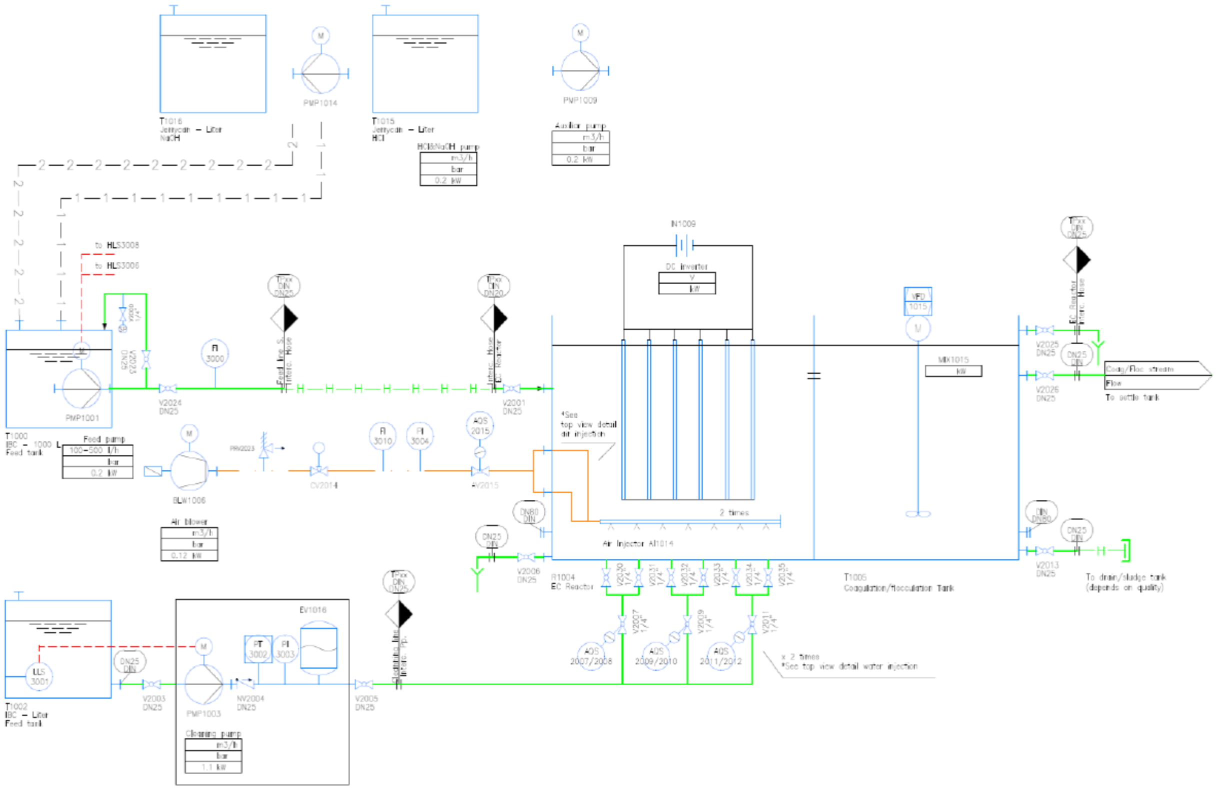

Figure 2 EC (Qstone) scheme of the Pilot setup

Flocculation tank

Once the charge of the pollutants is neutralized, a high-energy, rapid mix is needed to properly disperse the coagulant and to encourage particle collisions. This is done by air bubbling in the EC reactor. Overmixing does not affect the initial coagulation, but insufficient mixing will result in this step being incomplete. Therefore, an extra step of flocculation is needed to strengthen the flocs. It can be seen as a gentle mixing stage, which increases the particle size from micro flocs to visible suspended particles. During the experiments a rotation speed of 5.8 Hz was implemented .

Polarity Inverter

A polarity inverter in electrocoagulation periodically reverses the electrical polarity of electrodes. This prevents the buildup of scale and contaminants on the electrode surfaces, ensuring consistent treatment efficiency. By alternating the current, it extends electrode lifespan, and reduces the need for frequent maintenance. The main characteristics of inverter are depicted in Table 1.

Table 1 Characteristics of the Inverter

Inverter	
Voltage (+ 20% contingency)	12 V
Current (+ 20% contingency)	259 A
Power consumption	3.1 kW
Max. current	400 A
Charge setting	Dipolar
Reactor current flow	Series
Length inverter to plates cable	1.5 m
Thickness cable	285 mm^2

Reverse polarity every 300 seconds, applied voltages between 3.4 and 12 V and hydraulic retention time (HRT) of 20 minutes were implemented during the pilot experiments.

Aeration system

The aeration part consists by default of two pipes distributing air under the electrodes into the water for two reasons. First, the applied air bubbles mix the water through the electrodes which provides a better mixing for higher efficiency of iron release. Second, it oxygenizes the water to eliminate the possible excess of iron still contained in the solution.

Settling tank

In the settling tank, the effluent is stored, and the settlement time of the flocs is noted. Sludge is quantified periodically.

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2.1.2 Monitoring and measurements Dunea EC Pilot

The experiments were carried out at Dunea's Bergambacht drinking water plant. For each experiment run, approximately 1000 L of water was taken from the different water types (pretreated with CC surface river water and sand filter backwash water) and stored temporarily in a tank that was connected to the pilot plant inlet pump. A monitoring plan was used to obtain the needed data for the evaluation of the performance of EC with two water types. A completely raw surface water was not available at the location since at the intake of the river $FeCl_3$ is dosed.

Daily measurements during the experimental runs were taken following the prepared data Table 2, in which the following parameters were measured once or twice for day in both influent and effluent: Flow (flowmeter), pH (pH meter), T, color (254nm, 435 nm, 525nm, 620 nm via spectrophotometer), total color, turbidity, As, TSS, Total P (analysis carried out by the laboratory), with a frequency as indicated in Table 3.

Table 2. Measurements during EC pilot experimental work.

SAMPLING TIME	DATE	Current	Fe dosage	EC PARAMETERS									MIXING TANK PARAMETERS		RAW INFLUENT													EC EFFLUENT 1											
				Sludge production	Reverse Polarity	Voltage $_{Min}$	Voltage $_{Max}$	Avg. Voltage	Fe dosage/As	Fe dosage/color	HRT	Power consumption	Mixer		Flow	pH	T	EC	col. 254 nm	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity	As	TSS	Total P	pH	T	EC	col. 254 nm	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity	As	TSS	Total P
		A	mg/L	kg/m3	sec	V	V	V	mg/mg	mg/a.u	min	kW/m^3	Hz	Rpm	L/hr		°C	mS/cm	a.u	a.u	a.u	a.u	a.u	NTU	mg/L	mg/L	mg/L		°C	mS/cm	a.u	a.u	a-u	a.u		NTU	mg/L	mg/L	mg/L

Table 3 Frequency of sampling per test/day of the Dunea Water. Measurements from Monday to Friday when experiment were carried out.

Parameter	Raw influent	Effluent 1
Flow rate	2x	2x
Current	1x	1x
Voltage	1x	1x
pH	2x	2x
Temperature	2x	2x
Conductivity (uS/cm)	2x	2x
Color 254 nm	1x	1x
Color 436 nm	1x	1x
Color 525 nm	1x	1x
Color 620 nm	1x	1x
Total Color	2x	2x
Turbidity (NTU)	2x	2x
Arsenic (As) and Other metals	1x	1x
Total Suspended Solids (TSS)	1x	1x
Total P (PO4 mg/L)	1x	1x
TOC (mg/l)	1x	1x
Fe (mg/L)	1x	1x

When possible, the analyses were carried out in situ using kits and a spectrophotometer, and the rest were carried out at Dunea's laboratory following the standard methods.

Furthermore, for indication of the settleability of flocs, the sludge volume was observed using an Imhoff settling cone (see Figure 3) for a period of 48 hours.

Figure 3 Photo of an Imhoff flask, empty and filled within the experiments.

2.2 Dosing of Fe with EC (electrode cassette) tests at Brabant Water

2.2.1 EC dosing Setup

An EC electrode cassette composed of electrode plates of two different dimensions: 80 mm x 60 mm x 10 mm (first test) and 160 mm x 60 mm x 10 mm (second test), was placed on the top of the sand filter column at Brabant Water Schijf drinking water location, so that iron is dosed electrochemically at low concentrations (Figure 4). The target dosing for Brabant Water in different locations is between 1.5 – 4 mg Fe /L. The objective was to test EC as an alternative for iron dosing before sand filtration.

Figure 4 EC dosing cassette on pilot sand filter column.

2.2.2 Measurements at Brabant Water for dosing Fe via EC cassette

Two different tests were carried out at Brabant Water to explore EC as an alternative for iron dosing before sand filtration instead of dosing FeCl3, after one short trial. The trial results showed that low Fe-dosages were not achieved during EC all the time and were not easy to control because the initial setup used was not completely appropriate (the current supply was not controllable). Iron concentrations of 2.2, 10.2, 9.8 and 20.2 mg Fe/l were measured when targeting about 4 mg Fe/L. Therefore, two more experiments were carried out to improve the iron dosage control with two strategies:

- Using CC (continuous current) mode instead of CV (continuous voltage).
- Doubling the area of electrodes

Samples were taken in the influent, in the backwash water of the filter, and effluent of the sand filter column as blank samples, and from the water on the top of the sand filter and effluent after applying the corresponding current (see Table 6).

2.3 Results and Discussion

2.3.1 Dunea EC pilot: Sand filter backwash water

Removal Efficiencies

The removal efficiencies obtained at different applied currents within the EC unit are presented in Table 4.
It can be observed that at about 180 A (124 mg/L Fe dosed), very high removals efficiencies were observed for most of the parameters: total removal values are reported for color 76-82%, turbidity 89-92%, arsenic 89-90%, total suspended solids 78-85%, total phosphate 92-100%, and total organic carbon 48-49%. However, at this applied current or dosage, the power consumption is already at 1.7 KW/m^3 and the iron dosage equivalent is high compared to state of the art dosing in conventional coagulation, and in this case, there will be iron overdosing ending up with a residual iron concentration in the effluent and the formation of $Fe(OH)_3$ (green color). Increasing further the applied current to 240 A only slightly improved the removal of the different parameters. On the contrary, when looking at the lowest applied current with good floc formation (about 60 A), the removal efficiencies are considered similar to what has been observed in Dunea by applying conventional coagulation with FeCl3, with removal values of 48-56% for total color, 70-80% for arsenic, 69-75% for total suspended solids, 80-86% for total phosphate, and 33-43% for total organic carbon. Color at 525 nm and 620nm was received by 57-61% and 51-67%, correspondingly.
Removal efficiencies of total color of 55-86% and turbidity of 64-93% at 60 A were measured after a settling time of about 8 hr. When comparing these results with the conventional coagulation-flocculation, total color removal of 63-86% and turbidity of 78-86% of the backwash sand filtration water are achieved currently (see Table 13 Appendix I). Consequently, after discussion with Dunea's process engineers it was decided to choose this applied current of 60A, equivalent to an iron dosage of about 42 mg Fe/L, for further assessment of the sand filter backwash water. It must be pointed out that with CC an average dosage of 43.6 mg $FeCl_3$ (40%) is used, i.e., corresponding to an iron dose of 6 mg Fe/L. This shows that the observed minimum dose for obtaining good floc formation with EC and similar removal efficiencies than CC is about 7 times higher. This is contrary to what was expected, likely due to the low conductivity of the water.
During the experiments minimum variation of influent/effluent parameters such as pH (see Figure 35, Appendix I) and temperature (see Figure 37, Appendix I) were observed. In the case of conductivity, it decreased from 525 to 498 µS/cm on average after applying EC (Figure 36, Appendix I).
Substantial metals removal was observed for EC at different applied currents, but they were in most of the cases comparable despite the current applied. For arsenic, which is the target for Dunea, removal efficiencies between 72-88% were observed, but also for other metals present in the influent water (see Figure 5).

It must be pointed out that for that niquel, molybdenum and iron, negative removal or increased concentrations were observed in the water effluent of the EC compared to the influent. This is aligned and expected due to the electrode composition. In the case of the applied current of 60 A, for these metals, the effluent concentrations were: 9.5, 3.0, and 3701 µg/L, respectively.

Table 4 Removal efficiencies of monitored parameters for sand filter backwash water. Color was measured at different wave lengths (436, 525 and 620nm). TC: Total Color. As: Arsenic. TSS: Total suspended solids. TP: Total phosphorus. TOC: Total organic carbon.

EC PARAMETERS				RAW INFLUENT										REMOVAL EFFICIENCES								
Current	Fe dosage	Voltage	Power consumption	EC	436 nm	525 nm	620 nm	TC	Turbidity	As	TSS	TP	TOC	436 nm	525 nm	620 nm	TC	Turbidity	As	TSS	TP	TOC
A	mg/L	V	kW/m³	µS/cm	a.u	a.u	a.u	a.u	NTU	mg/L	mg/L	mg/L	mg/L	%	%	%	%	%	%	%	%	%
59	41	3,4	0,2	526	0,6	0,4	0,4	1,4	50	4,0	69,0	0,1	5,8	39%	57%	51%	48%	56%	80%	75%	86%	43%
62	43	3,4	0,2	519	0,6	0,6	0,5	1,7	58	3,5	88,6	0,2	6,0	-33%	22%	29%	3%	26%	59%	66%	67%	34%
59	41	3,3	0,2	519	0,9	0,7	0,6	2,3	59	3,0	76,7	0,2	4,9	45%	61%	67%	56%	53%	70%	69%	80%	33%
119	83	6,4	0,8	530	0,5	0,4	0,4	1,3	42	3,2	42,0	0,1	6,7	29%	64%	80%	55%	79%	85%	83%	92%	56%
120	83	6,5	0,8	528	0,3	0,2	0,2	0,8	21	2,7	24,0	0,1	5,3	-128%	-43%	-2%	-68%	14%	80%	42%	91%	41%
120	83	6,5	0,8	547	0,3	0,2	0,2	0,7	23	5,2	18,0	0,2	7,2	1%	26%	54%	22%	39%	87%	55%	94%	48%
179	124	9,3	1,7	555	0,3	0,2	0,2	0,7	20	5,5	18,7	0,1	7,1	-11%	34%	52%	18%	70%	86%	53%	85%	54%
179	124	9,6	1,7	519	0,8	0,6	0,5	1,9	53	3,0	54,0	0,1	5,1	71%	86%	92%	82%	91%	89%	85%	92%	48%
179	124	9,3	1,7	526	0,7	0,5	0,5	1,7	53	3,2	42,0	0,2	4,8	62%	81%	90%	76%	89%	90%	78%	100%	49%
236	164	12	2,8	519	0,7	0,6	0,4	1,7	54	3,5	56,0	0,2	4,8	50%	75%	83%	67%	91%	89%	75%	94%	49%
232	161	12	2,8	503	0,5	0,4	0,3	1,3	35	4,2	34,0	0,1	4,1	44%	71%	79%	62%	74%	93%	71%	92%	46%
236	164	12	2,8	513	1,2	1,0	0,8	2,9	88	3,7	91,0	0,2	5,1	69%	81%	89%	78%	88%	92%	90%	95%	54%

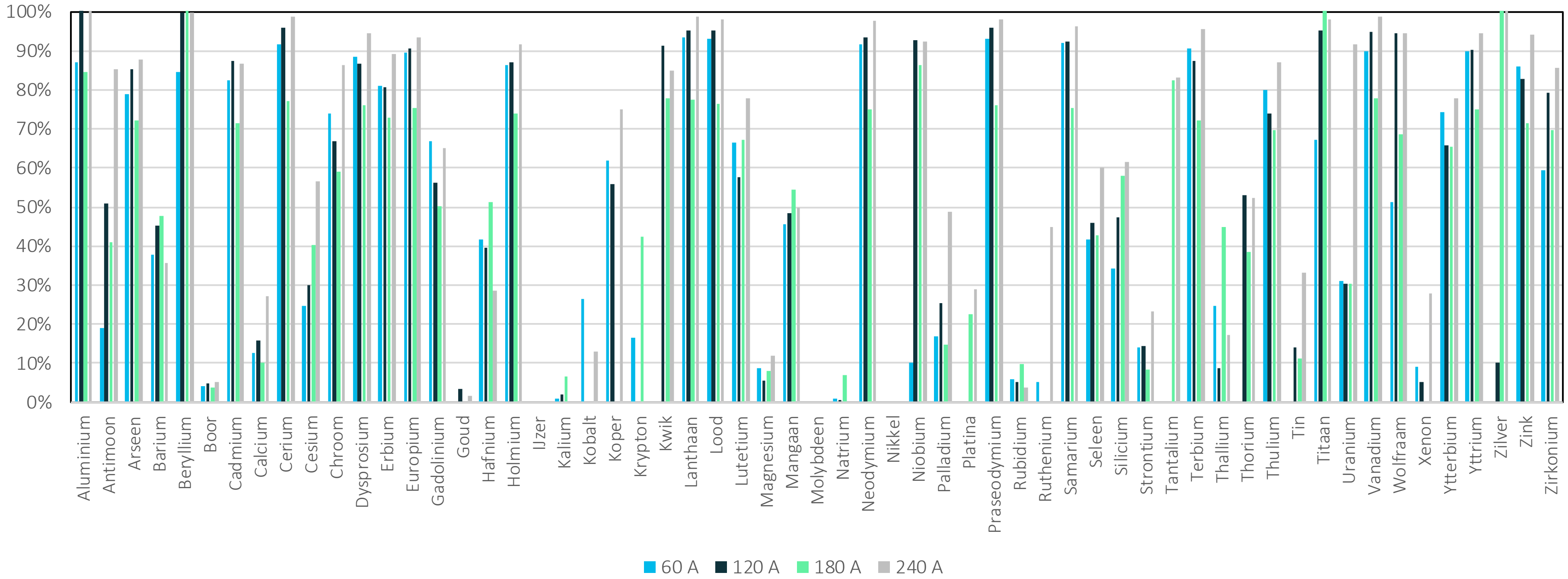


Figure 5 Metals removal efficiency (y-axis) at different applied currents.

Concentrations in the Effluent

When looking at concentrations in the effluent of the EC, in most of the cases for arsenic, concentrations lower than 1 µg/L were achieved in the effluent. The higher the current applied the lower the effluent concentration was. At about 60A, concentrations of 0.78, 0.89 and 1.44 µg As/L were observed in the effluent, compared to 0.32 µg As/L or 0.28 µg As/L achieved by applying 180 A or 240 A respectively.

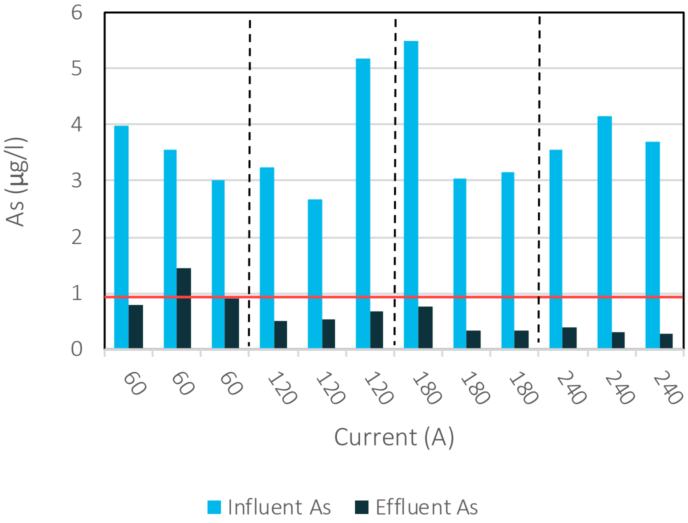


Figure 6 Achieved arsenic concentrations in effluent by applying different currents in the EC unit.

Total phosphate lower than 0.05 mg/L was achieved for all the applied currents (Figure 7), showing the EC can remove efficiently phosphate by likely forming FePO4 precipitate during the coagulation/flocculation process.

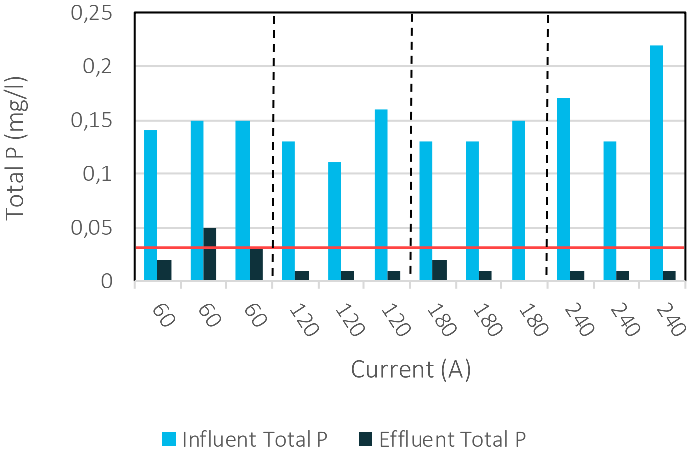


Figure 7 Achieved total phosphorus concentrations in effluent by applying different currents in the EC unit.

Regarding suspended solids, it was observed that the influent presented a high variability of TSS concentration in the range of 18 to 91 mg TSS/L. Concentrations lower than 15 mg TSS/L in the effluent were achieved only at 120, 180 and 240 A applied currents. At about 60 A, the concentrations achieved were between 17 and 30 mg TSS/L (Figure 8). This is in alignment with the turbidity and color removals achieved, which are slightly better or comparable with CC (see Table 14).

Total organic concentration in the effluent was in most of the cases lower than 3.5 mg TOC/L (as indicated by the red line) for the applied currents and for 240A, concentrations lower than 2.5 mg TOC/L were achieved (Figure 9).

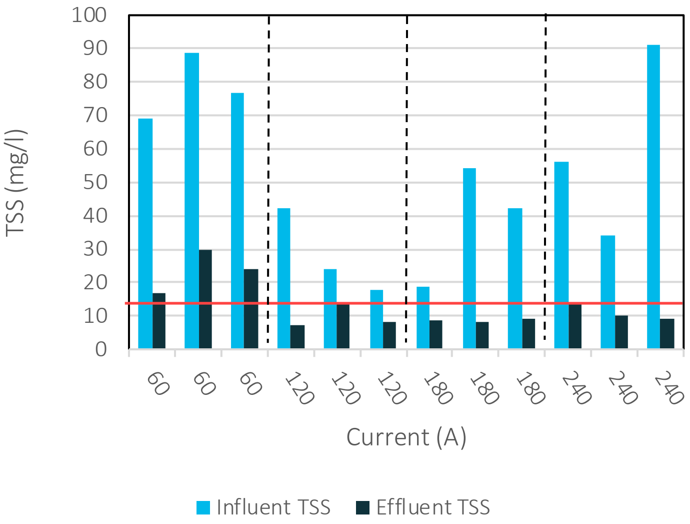


Figure 8 Achieved total suspended solids concentrations in effluent by applying different currents in the EC.

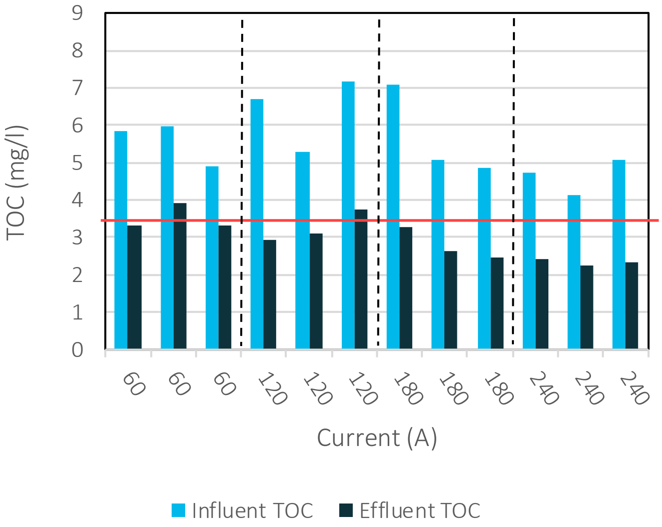


Figure 9 Achieved total organic carbon concentrations in effluent by applying different currents in the EC.

2.3.2 Dunea EC pilot: Surface water

Removal Efficiencies

The removal efficiencies in surface water obtained after different applied currents within the EC unit are presented in Table 5. It must firstly point out that the surface water used for the experiments has passed already through coagulation-flocculation with $FeCl_3$ at the intake of the river water, and therefore the influent water presents already lower concentrations than expected if it would have been raw influent. From the EC effluent concentrations and the calculated removals, it was observed that about 120 A (84 mg/L Fe dosed), competitive removals efficiencies were observed for Arsenic, phosphate and TOC: arsenic 69-72%, total phosphate 50%, and total organic carbon 25-26%. However, for total color, turbidity and total suspended solids, all the obtained values were negative, indicating that due to the relatively clean influent water iron overdose was needed to get the formation of flocs, being detrimental for these parameters. Even at 60A, there is a negative impact on these parameters. At this applied current or dosage, the power consumption at pilot scale is about 1.2 KW/m3 and the iron dosage equivalent is significantly higher than what would be normally used with conventional coagulation, confirming there is substantial iron overdosing.

Further increased of the applied current to 240 A increased only the arsenic and phosphate removal to about 80%, but remained similar for the other parameters.

Removal efficiencies of total color of 44-48% and turbidity of 71-80% at 120 A were measured after a settling time of about 8 hr (see Table 14 Appendix I). After discussion with the project partners, it was decided that knowing that the surface water was already pretreated with conventional coagulation on the inlet 30 km away, the minimum applied current of 60A might be sufficient for further assessment of the surface water, which is equivalent to an iron dosage of about 42 mg Fe/L. The water at Dunea in Bergambacht is a combination of pretreated surface water, rainwater and run-off water.

For the conventional coagulation (CC) which is applied to this surface water, an average dosage of 145.2 $FeCl_3$ (40%) is used, i.e., corresponding to an iron dose of about 20 mg Fe/L.

During the experiments pH in the influent varied from 7.8 to 8.2. An average pH increase from 8.1. to 8.7 was observed after EC (see Figure 38, Appendix I). Temperature increased from 23.6 to 24.9. (Figure 40, Appendix I) were observed. In the case of conductivity, the influent decreased from 525 to 498 µS/cm during the time of the experiments and effluent increases were observed mainly at higher applied up to 580 µS/cm at 236 A. (Figure 39, Appendix I).

Similar to in sand filter backwash water, substantial metals removal was observed for EC at different applied currents. For arsenic, removal efficiencies between 69-78% were observed. Higher removal values were observed for metals such aluminium , beryllium, lead, titanium, vanadium with removals higher than 80% (see Figure 14). Niquel, cobalt, manganese, molybdenum and iron showed negative removal or increase in the water effluent of the EC compared to the influent surface water. For example, for 120 A their EC effluent concentrations were: 7.4, 0.9, 451, 3.5 and 4781 µg/L, respectively.

Concentrations in the Effluent

For arsenic, concentrations lower than 1 µg/L were achieved in the effluent. The higher the current applied the lower the effluent concentration obtained. At about 60 A, concentrations in the range of 0.65-1.78 µg As/L were observed in the effluent, compared to <0.6 µg As/L achieved by applying 180 A or 240 A (Figure 10). The effluent concentrations were not always consistent at the same current, likely due to variations in the influent matrix.

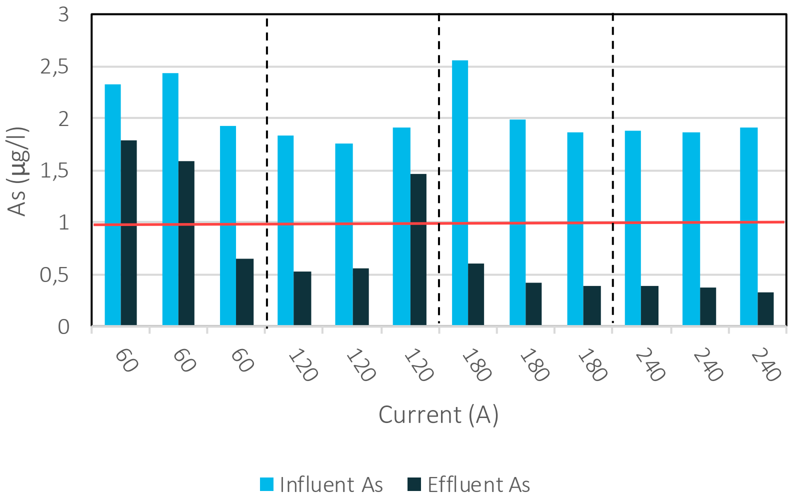


Figure 10 Achieved arsenic concentrations in effluent by applying different currents in the EC.

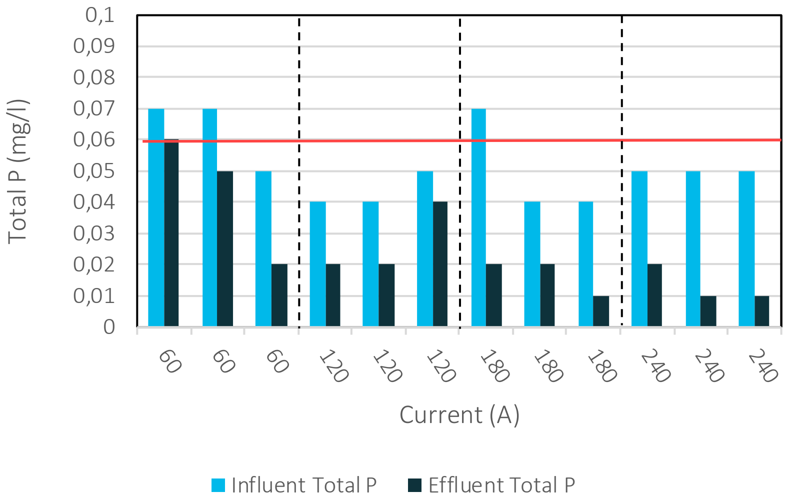


Figure 11 Achieved total phosphorus concentrations in effluent by applying different currents in the EC.

Total phosphate lower than 0.06 mg/L was achieved for all the applied currents (Figure 11). In terms of suspended solids, the influent water showed a relatively low TSS concentration in the range of 5 to 17 mg TSS/L. After EC, concentrations were higher than 10 mg TSS/L at 60 and 120 A. At 180 and 240A, the concentrations achieved were lower but still higher than in the influent (Figure 12). The latter corresponds with the negative turbidity and color removals achieved, which are due to the iron overdosing.

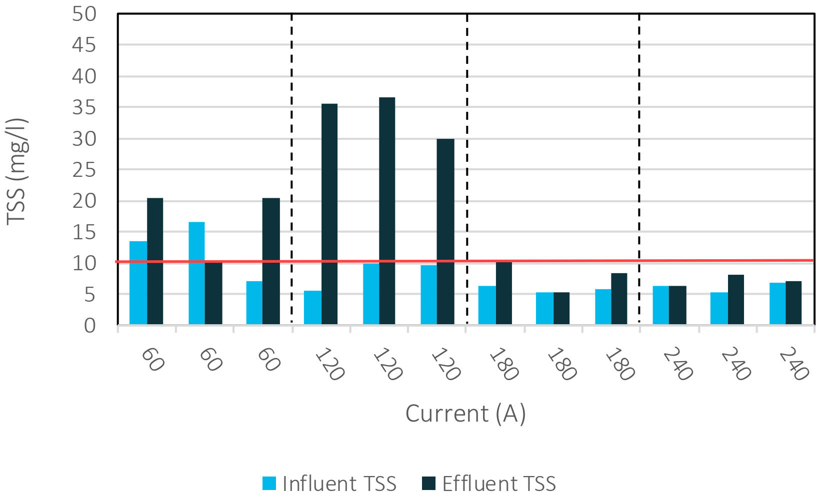


Figure 12 Achieved total suspended solids concentrations in effluent by applying different currents in the EC.

Even though TOC removal was considerably less with surface water when compared to sand filter backwash water, the TOC effluent concentrations were lower than 4.5 mg TOC/L for 60 A and 120 A. However, it is clear that the attainable effluent concentration depends on the influent quality and the removal was comparable and about 325% for all experiments.

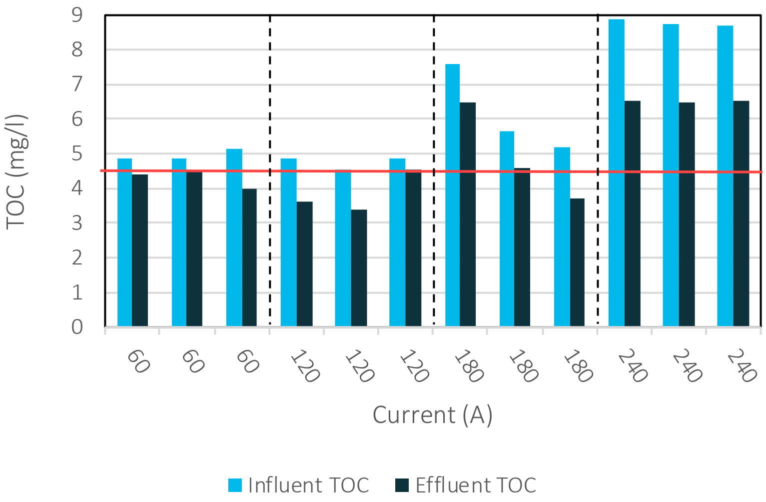


Figure 13 Achieved total organic carbon concentrations in effluent by applying different currents in the EC.

Table 5 Removal efficiencies of monitored parameters for Surface Water. Color was measured at different wavelengths (436, 525 and 620nm). TC: Total Color. As: Arsenic. TSS: Total suspended solids. TP: Total phosphorus. TOC: Total organic carbon.

EC PARAMETERS				**RAW INFLUENT**										**REMOVAL EFFICIENCIES**								
Current	Fe dosage	Voltage	Power consumption	EC	col.436 nm	col.525 nm	col.620 nm	Total Color	Turbidity	As	TSS	Total P	TOC	col.436 nm	col.525 nm	col.620 nm	Total Color	Turbidity	As	TSS	Total P	TOC
A	mg/L	V	kW/m³	µS/cm	a.u	a.u	a.u	a.u	NTU	mg/L	mg/L	mg/L	mg/L	%	%	%	%	%	%	%	%	%
60	42	3,7	0,3	528	0,2	0,1	0,1	0,4	15	2,3	13,4	0,1	4,9	-131%	-49%	-14%	-73%	-67%	23%	-53%	14%	9%
59	41	3,4	0,3	526	0,2	0,1	0,1	0,4	14	2,4	16,6	0,1	4,9	-134%	-74%	-26%	-88%	-36%	35%	37%	29%	8%
61	42	3,4	0,3	519	0,1	0,1	0,1	0,3	8	1,9	7	0,1	5,2	-265%	-151%	-86%	-188%	-138%	66%	-193%	60%	23%
120	83	6,8	1,3	517	0,1	0,1	0,1	0,2	10	1,8	5,6	0,0	4,9	-241%	-140%	-59%	-171%	-70%	72%	-534%	50%	25%
121	84	6,2	1,2	518	0,1	0,1	0,1	0,2	8	1,8	9,8	0,0	4,5	-369%	-269%	-124%	-280%	-213%	69%	-272%	50%	26%
121	84	6,2	1,2	516	0,1	0,1	0,1	0,2	7	1,9	9,6	0,1	4,9	-328%	-197%	-81%	-228%	-200%	24%	-212%	20%	6%
180	125	9,1	2,5	517	0,1	0,1	0,1	0,2	7	2,6	6,2	0,1	7,6	-263%	-131%	-96%	-182%	-200%	76%	-65%	71%	15%
180	125	9	2,5	510	0,1	0,1	0,0	0,2	7	2,0	5,2	0,0	5,7	-208%	-86%	-9%	-123%	-29%	79%	0%	50%	19%
179	124	9,1	2,5	509	0,1	0,1	0,0	0,2	5	1,9	5,8	0,0	5,2	-317%	-192%	-79%	-219%	-320%	79%	-45%	75%	29%
232	161	12	4,3	496	0,1	0,1	0,1	0,2	6	1,9	6,2	0,1	8,9	-273%	-178%	-94%	-202%	-133%	80%	-3%	60%	26%
236	164	12	4,4	502	0,1	0,1	0,1	0,2	6	1,9	5,2	0,1	8,8	-156%	-71%	-3%	-92%	-167%	80%	-58%	80%	26%
236	164	12	4,4	492	0,1	0,1	0,1	0,2	8	1,9	6,8	0,1	8,7	-212%	-103%	-40%	-139%	-75%	83%	-6%	80%	25%

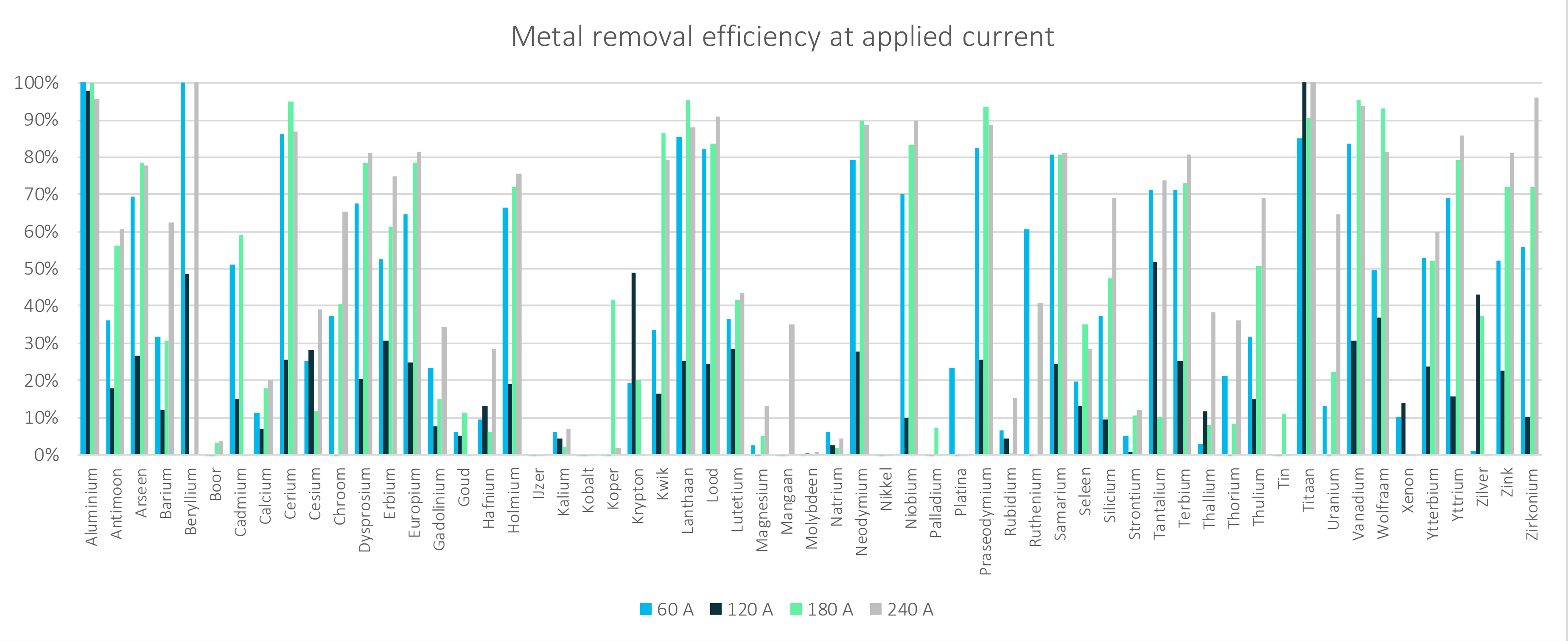


Figure 14 Metals removal efficiency (y-axis) at different applied currents.

2.3.3 Brabant Water Dosing of Iron with EC (electrode cassette) in groundwater sand filtration

The results of the Fe concentrations achieved are depicted in Table 6. For the experiment 1 repeated at CV, the current could not be controlled and a dosage of about 3.4 mg Fe/L was achieved when striving for less than 2 mg Fe/L. Once the current was controlled in CC mode, a lower dose of 0.75 mg Fe/L was achieved, however it was expected to be higher, since the applied current was higher. In experiment 3, with the same current as 1, a dose of about 1 mg Fe/L was achieved. However, the last experiment with a lower applied current and expected lower concentration resulted in a high iron dose of 8.6 mg/L. This misleading result is likely due to sampling-associated errors (there are iron particles in the walls of the sampling point Above water EC+RSF that could enter into the water sample). After the EC cassette was modified with a double area of plates (experiments 5-7), only at 0.55A an iron dose of 1.28 mgFe/L was determined. Once the current was diminished, the balance of iron between the influent and the water above the rapid sand filtration after the EC dosing exhibited negative values, which could not be explained. It is suggested for potential follow up tests to have longer runs of electrolysis (at least 10 min) and better mixing under the electrode cassette to guarantee homogenous iron concentration in all the sampling points.

Table 6 Results dosing of iron with EC cassette (1-4: plates of 80mmx60mmx10mm; 5-7: plates of 160mmx60mmx10mm)

							EC parameters		Result
	Sample	Monster	Time	Flow	Height volume to treat	Height sampling	Current	Voltage	Fe
1			min	L/h	cm	cm	A	V	µg/l
CV	Backwash of filter		3						
	Influent	1	0	3.9	55	25	--	--	4383.8
	Effluent (RSF)	1.1	0		--	--	--	--	330.6
	Above water (EC+RSF)	1.2	1	3.9	55	25	0.35	30	7759.6
	Effluent (EC+RSF)	1.3	2	3.9	55	25	0.34	30	250.5
	Iron Dosed								**3376**
2 CC	Backwash of filter		3						
	Influent	2	0	3.9	55	25	--	--	3978.8
	Effluent (RSF)	2.1	0		--	--	--	--	--
	Above water (EC+RSF)	2.2	1	3.9	55	25	0.55	57	4723.4
	Effluent (EC+RSF)	2.3	2	3.9	55	25	0.55	57	303.7
	Iron Dosed								**745**
3 CC	Backwash of filter		3						
	Influent	2	0	3.9	55	25	--	--	3850.1
	Effluent (RSF)	2.1	0		--	--	--	--	--
	Above water (EC+RSF)	2.2	1	3.9	55	25	0.35	36	4839.4
	Effluent (EC+RSF)	2.3	2	3.9	55	25	0.35	37	138.8
	Iron Dosed								**989**
4 CC	`cc		3						
	Influent	2	0	3.9	55	25	--	--	3830.1
	Effluent (RSF)	2.1	0		--	--	--	--	--
	Above water (EC+RSF)	2.2	1	3.9	55	25	0.15	17.5	12410

	Effluent (EC+RSF)	2.3	2	3.9	55	25	0.15	17.5	116
	Iron Dosed								**8580**
5 CC	Backwash of filter		3						
	Influent	2	0	3.9	55	25	--	--	3886.4
	Effluent (RSF)	2.1	0		--	--	--	--	125.65
	Above water (EC+RSF)	2.2	1	3.9	55	25	0.55	57	5168.5
	Effluent (EC+RSF)	2.3	2	3.9	55	25	0.55	57	148.3
	Iron Dosed								**1282**
6 CC	Backwash of filter		3						
	Influent	2	0	3.9	55	25	--	--	3860.4
	Effluent (RSF)	2.1	0		--	--	--	--	--
	Above water (EC+RSF)	2.2	1	3.9	55	25	0.15	17.5	2640.8
	Effluent (EC+RSF)	2.3	2	3.9	55	25	0.15	17.5	118.3
	Iron Dosed								**-1219**
7 CC	Backwash of filter		3						
	Influent	2	0	3.9	55	25	--	--	3855.1
	Effluent (RSF)	2.1	0		--	--	--	--	--
	Above water (EC+RSF)	2.2	1	3.9	55	25	0.15	17.5	2315.4
	Effluent (EC+RSF)	2.3	2	3.9	55	25	0.15	17.5	170.4
	Iron Dosed								**-1540**

2.4 Concluding remarks

The following concluding remarks were drawn for sand filter backwash water pilot operation at Dunea's location:

- A current of 60 A was determined as the best for further evaluation (lowest power consumption, lower Fe dosage, high enough removal of targeted parameters).
- Removal efficiencies achieved right after EC for the target parameters at 60 A (Fe dosage of about 42 mg/l) were: 80% As, 75% TSS, 86% Total P, 43% TOC, 56% Turbidity, 48% Total color. After 8 hours of settling Turbidity and total color removal achieved were between 64-93% and 56-88% respectively. Compared with conventioncal coagulation with FeCl3 dosage, turbidity and color removals of 78-85% and 63-86% were achieved correspondingly.
- Metals removal was favorable with EC. Niquel, molybdenum and iron concentrations were shown to increase (coming from the electrodes dissolution).

The following concluding remarks were drawn for (pretreated) surface water at Dunea's location:

- Based on the pilot experimental results, a current of 120 A would need to be applied for the EC unit with the surface water quality that was used (medium power consumption, medium Fe dosage, removal of targeted parameters). However, this surface water was already pretreated with conventional coagulation using a dosage of about 20 mg Fe/L, and therefore for further economical and life cycle assessment to use a current of 60 A to minimize energy consumption and to avoid overdosing of iron.

- Removal efficiencies achieved right after EC for the target parameters at 120 A (Fe dosage of about 84 mg/l) were: 72% As, 50% Total P, and 26% TOC. For TSS, Turbidity, and total color, EC effluent quality was less than influent due to overdosing of iron because of the relatively clean influent (CC in the intake of the river). However, after 8 h of effluent settling Turbidity and total color removal achieved were between 71-80% and 44-48% respectively.

- Niquel, cobalt, manganese, molybdenum and iron concentrations were shown to increase likely due to electrodes dissolution and less floc formation.

The following concluding remarks were drawn for EC dosing for groundwater at Brabant Water’s location:

- Dosages of 0.7 mg/L to 8.6 mg/L could be achieved. The current was better controlled with an inverter of higher capacity and running the experiment in CC mode instead of CV mode.

- Fe dosage obtained does not follow a clear pattern. The influence of low electrical conductivity might be a reason for it, as well as the uncertainty of sampling points, some of them with a visual accumulation of iron. Increasing the area of plates did not show any improvements in better dosing control results.

- EC proved to be able to dose low concentrations of iron, however, reproducibility is still a question mark and more experiments would be needed to demonstrate it. It is suggested for future trials to run the experiments with longer electrolysis time and improving mixing to guarantee homogeneous distribution of dosed iron for the sampling points.

3 Cost Calculations

This chapter provides an overview of a calculated cost comparison between iron EC and CC dosing iron chloride, by translating the pilot experimental results into an operational scale of the system for surface water and sand filter backwash water treatment.

3.1 Materials and Methods

3.1.1 Basis and boundaries of the cost calculations

For the calculations described in the following section, we have considered two practical scenarios (1) sand rapid filter backwash water treatment at Dunea, with a capacity of 79 m^3/h and (2) treating surface water (new sources) for Dunea's new drinking water treatment development with a capacity of 1000 m^3/h.
For all scenarios, the results (i.e, applied current and corresponding Fe dosage) of the electrocoagulation pilot experiments and current practices at Dunea were taken into consideration for the calculations considering the following Fe dosages:

1. Surface water conventional coagulation, in which 145 mg $FeCl_3$ (40%)/l is dosed resulting in 20 mg Fe/l.
2. Surface water electrocoagulation, in which 42 mg Fe/l is dosed (60 A).
3. Sand filter backwash water conventional coagulation, in which 43.6 mg FeCl3 (40%)/l is dosed resulting in 6 mg Fe/l.
4. Sand filter backwash water conventional coagulation, in which 42 mg Fe/l is dosed (60 A).

For each of these scenarios, the drinking water treatment process boundaries used for the cost estimation (and further LCA) are depicted in Figure 15. Only coagulation/electrocoagulation, followed by flocculation and lamella sedimentation are evaluated since here is where the differences are expected to occurred. As a result of the discussions with the process engineers and water technologist during the two workshops carried out on the conceptual design, current practices, and take away points from the pilot trials, it was decided that despite the slight differences in sludge/flocs formation, the flocculation and sedimentation is assumed as equal for both coagulation with $FeCl_3$ and electrocoagulation for the cost calculation and further LCA.

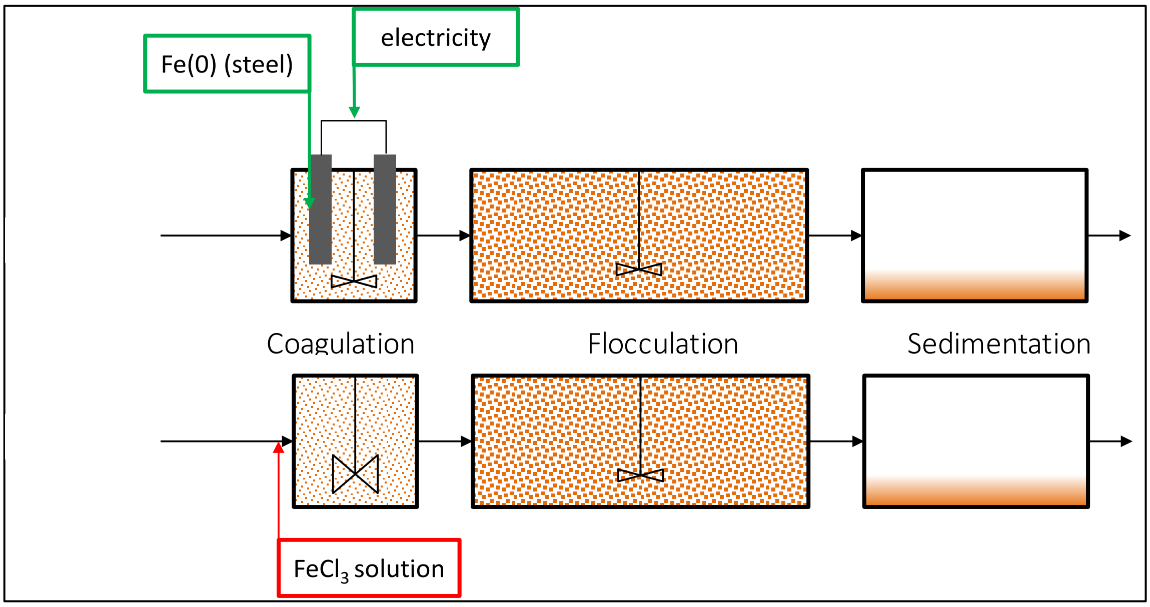


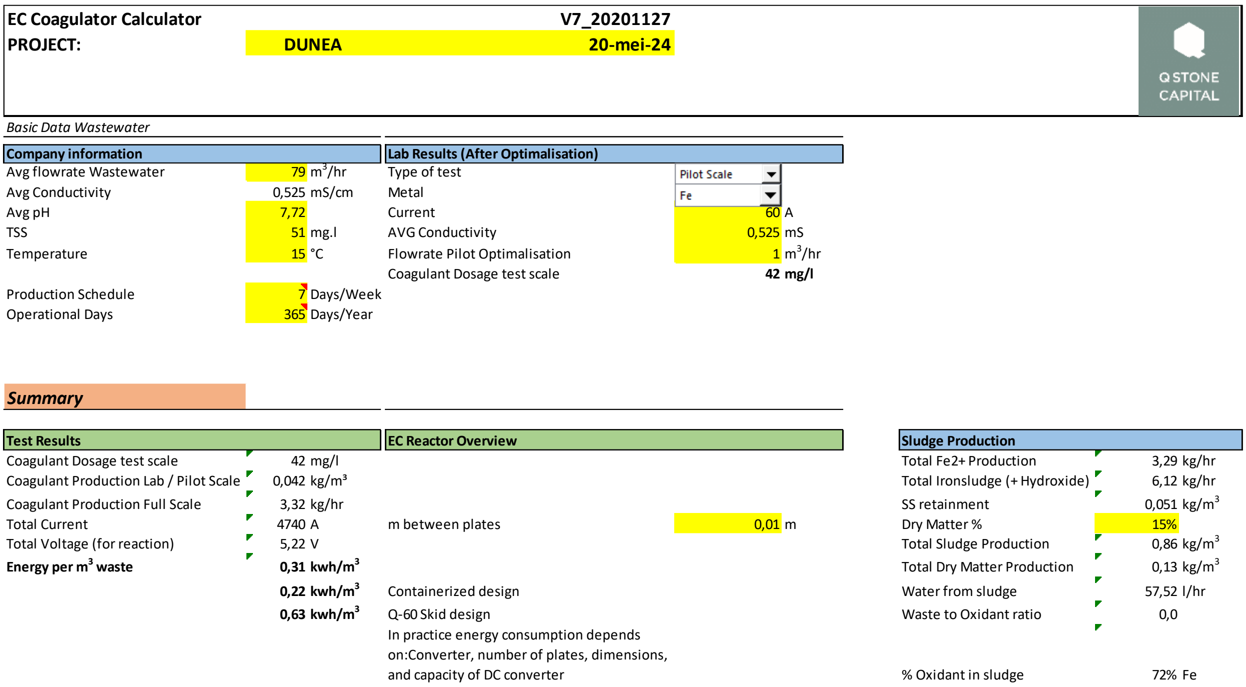
Figure 15 Boundaries of the cost evaluation (and LCA).

3.1.2 Cost and scale-up calculations

Electrocoagulation unit

With an Excel calculation sheet from QStone Capital aimed at dimensioning of the system, data required for the evaluation have been calculated. Figure 16 shows an example of a printout of one of the QStone Capital's calculation sheets.

Input from the pilot result experiments are: the current [A], conductivity [mS/cm], flow rate of the pilot [l/h] and measured iron dose [mg/l Fe]. Input for practical applications are: flow rate [m^3/h], pH [-], TSS [mg/l], temperature [°C], electrode gap distance [m], dry matter content [%] and iron cost [€/kg]. Key technology parameters such as the iron dose [kg/h], the current [A], the voltage [V], the energy consumption [kWh/m^3] and the sludge production [kg/m^3] were calculated.



EC Coagulator Calculator **V7_20201127**
PROJECT: **DUNEA** **20-mei-24**

Basic Data Wastewater

Company information		
Avg flowrate Wastewater	79	m^3/hr
Avg Conductivity	0,525	mS/cm
Avg pH	7,72	
TSS	51	mg.l
Temperature	15	°C
Production Schedule	7	Days/Week
Operational Days	365	Days/Year

Lab Results (After Optimalisation)		
Type of test	Pilot Scale	
Metal	Fe	
Current	60	A
AVG Conductivity	0,525	mS
Flowrate Pilot Optimalisation	1	m^3/hr
Coagulant Dosage test scale	**42**	**mg/l**

Summary

Test Results		
Coagulant Dosage test scale	42	mg/l
Coagulant Production Lab / Pilot Scale	0,042	kg/m^3
Coagulant Production Full Scale	3,32	kg/hr
Total Current	4740	A
Total Voltage (for reaction)	5,22	V
Energy per m^3 waste	**0,31**	**kwh/m^3**
	0,22	**kwh/m^3**
	0,63	**kwh/m^3**

EC Reactor Overview		
m between plates	0,01	m
Containerized design		
Q-60 Skid design		

In practice energy consumption depends on:Converter, number of plates, dimensions, and capacity of DC converter

Sludge Production		
Total Fe2+ Production	3,29	kg/hr
Total Ironsludge (+ Hydroxide)	6,12	kg/hr
SS retainment	0,051	kg/m^3
Dry Matter %	15%	
Total Sludge Production	0,86	kg/m^3
Total Dry Matter Production	0,13	kg/m^3
Water from sludge	57,52	l/hr
Waste to Oxidant ratio	0,0	
% Oxidant in sludge	72%	Fe

Figure 16 Example of the QStone Capital calculation sheet (this excel tools is not shared, calculation are based on QStone Capital data and technical team input).

The investment costs of the EC installation were estimated by QStone Capital with an accuracy of ± 15%. The CAPEX was calculated excluding commissioning and excluding VAT. Furthermore, based on the CAPEX provided, and the standard distribution of the cost used by Dunea, KWR has proposed a distribution of the costs accordingly (following Bakker, 2015). The substantiation of EC's operating costs is based on information using QStone Capital basics of design input and calculations from KWR. KWR has calculated the energy costs [€/m^3 and €/year], the iron costs [€/m^3 and €/year], the sludge waste management [€/year], etc. The electrodes for electrocoagulation cost about 0,55 €/kg Fe.

Conventional coagulation-flocculation ($FeCl_3$) and sedimentation

Iron chloride dosing is currently the preferred technique used in water treatment to add an iron flocculant to water. The investment and operating costs of conventional coagulation-flocculation with $FeCl_3$ dosage followed by lamella sedimentation were determined with the Cost Standard drinking water tool of Royal HaskoningDHV (Bakker, 2015) used by all Dutch drinking water companies. The key figures for the investment cost calculations are presented in Table 15, Appendix II.

It is assumed that flocculation takes place with a residence time of 20 minutes, followed by a lamella settling with a design surface load 0.9 $m^3/h/m^2$. The storage capacity of 40% ferric chloride is 12 days (single-walled), with a strategic reserve of 15%, 100% reserve rack pumps, and 1 emergency/eye wash provided (see Appendix II). These are common practice values in the Dutch water sector. Transport costs of ferric chloride are not included in the calculations.

Iron chloride (FeCl3) cost was assumed to be 0,54 €/kg $FeCl_3$ (100%) for Dunea (based on actual costs). The energy cost used was 0,18 €/kWh for Dunea (current energy costs, assuming 100% green energy). The summary of all the input values used for the operational cost calculations is presented in Table 16, Appendix II.

A sensitivity analysis of total OPEX taking into account the energy price, energy consumption, iron electrodes cost (steel plates) and FeCl3 cost was made for both surface water and sand filter backwash water.

3.2 Results and Discussion

Table 7 provides an overview of the determined costs. They are classified as follows:

- Calculation QStone Capital EC, based on their EC calculation spreadsheet in which the results of experiments are taken into account;
- Investment costs RHDHV tool, floc formation, lamella settling and chemicals (ferric chloride) storage and dosing;
- Operating costs RHDHV tool, floc formation, lamella settling and chemicals (ferric chloride) storage and dosing;
- Calculation by KWR(*), determined with the results from the spreadsheet of QStone Capital and experiments; and standard distribution of costs;

Table 7 Overview of costs of iron electrocoagulation (EC) and iron chloride dosage (CC)

	Unit	SF Backwash water		Surface Water	
Design basis					
Design average flow rate	m^3/h	79		1000	
Design peak flow rate	m^3/h	103		1300	
Design iron chloride (40%) dose CC	mg/l FeCl3	43,6		145,2	
Design iron dose CC	mg/l Fe	6		20	
Design iron dose EC	mg/l Fe	42		42	
Water quality					
Conductivity	mS/cm	0,525		0,513	
pH		7,7		8,0	
Temperature	°C	15		15	
TSS	mg/l	51		8	
		EC	CC	EC	CC
Calculation QStone Capital EC					
Energy	kWh/m^3	0,63	--	0,63	--
Total sludge production	kg/m^3	0,62	0,42	0,62	0,31
Sludge production (dry matter)	kg/m^3	0,09	0,06	0,09	0,05
Lifetime of plates	Days	53	--	53	--
Cost of energy*	€/m^3	0,113	--	0,113	--
Cost iron electrodes*	€/m^3	0,023	--	0,023	--
Cost of FeCl3*	€/m^3	--	0,009	--	0,031
Energy efficiency*	kWh/kg Fe	15	--	15	--
Increase in water chloride content*	mg/l Cl	0	11,44	0	38,1
Investment costs RHDHV+QStone					
Flocculation	K€	392	392	3,851	3,851
Lamella settling	K€	835	835	6,031	6,031
EC Unit	K€	604	--	4,707	--
Dosing unit $FeCl_3$	K€	--	48	--	155
Subtotal	**K€**	***1,831***	***1,275***	***14,589***	***10,037***
Operational costs RHDHV+Qstone					
Flocculation	K€/year	36	36	358	358
Lamella settling	K€/year	64	64	477	477
EC Unit*	K€/year	169	--	1,797	--
Dosing unit $FeCl_3$	K€/year	--	25	--	420
Subtotal	**K€/year**	***269***	***125***	***2,632***	***1,255***

* Values calculated by KWR based on the input provided by the partners.

Total CAPEX (14,589 K€ Surface water and 1,831 K€ sand filter backwash water) and OPEX (2,632 K€/year Surface water and 269 K€/year sand filter backwash water) are substantially higher for EC than for CC at the selected settings. As it can be seen, the selection of a 42 mg/L Fe dosage for electrocoagulation (EC) in treating both surface water and sand filter backwash water is significantly higher than the 20 mg/L and 6 mg/L applied in conventional $FeCl_3$ coagulation for surface water and backwash water, respectively. Here, the question that remains is why EC required more iron dosage to form flocs compared to CC. At Dunea, the calculated energy required for EC is approximately 0.63 kWh per cubic meter (m^3) of treated water, as shown in Table 7. Consequently, the cost of energy per m^3 of water treated by EC is around 0.113 euros, leading to an energy efficiency of about 15 kWh per kilogram (kg) of iron. This indicates potential for optimization through different EC design choices since energy efficiency of about three times less is attainable. The current EC design, which employs a skid-based system with a Bipolar Serial connection of Fe plates, consumes roughly three times more energy than a container-based system with a Monopolar Parallel connection (refer to Appendix III for details).
The cost of iron electrodes stands at approximately 0.023 euros per m^3 for both surface water and backwash water due to the same iron dosage, whereas the cost for $FeCl_3$ is about 0.009 euros per m^3 for backwash water and 0.031 euros per m^3 for surface water, as indicated in Table 7.

Table 8 OPEX Iron electrocoagulation (EC) for surface water

OPEX Costs (€/year)	Flocculation	Lamela Settling	Electrocoagulation	TOTAL COSTs
Interest/Depreciation	€ 237.654	€ 360.772	€ 309.496*	**€ 907.900**
Energy	€ 31.536	€ 7.884	€ 990.662*	**€ 1.030.082**
Chemicals	€ 0	€ 0	€ 149.010*	**€ 149.010**
Waste	€ 0	€ 0	€ 243.734*	**€ 243.734**
Maintenance	€ 55.159	€ 72.391	€61.899*	**€ 189.500**
Operator	€ 23.797	€ 23.797	€ 30.949*	**€ 78.500**
Security	€ 3.831	€ 5.999	€ 5.261*	**€ 15.100**
Administration	€ 5.526	€ 5.959	€ 6.190*	**€ 17.700**
TOTAL	€ 358.000	€ 477.000	€ 1.797.000*	**€ 2.632.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 40%	+/- 40%

Table 9 OPEX Iron chloride dosing (CC) for surface water

OPEX Costs (€/year)	Flocculation	Lamela Settling	FeCl3 dosing	TOTAL COSTs
Interest/Depreciation	€ 237.654	€ 360.772	€ 10.444	**€ 608.900**
Energy	€ 31.536	€ 7.884	€ 4.588	**€ 44.008**
Chemicals	€ 0	€ 0	€ 266.499	**€ 266.499**
Waste	€ 0	€ 0	€127.743	**€ 127.743**
Maintenance	€ 55.159	€ 72.391	€5.364	**€ 132.900**

Operator	€ 23.797	€ 23.797	€ 4.631	**€ 52.200**
Security	€ 3.831	€ 5.999	€ 154	**€ 10.000**
Administration	€ 5.526	€ 5.959	€ 957	**€ 12.400**
TOTAL	€ 358.000	€ 477.000	€ 420.000	**€ 1.255.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 40%	+/- 40%

When examining both scenarios independently, the major investment costs for EC and conventional coagulation are similar, primarily due to the comparable costs of the flocculation and settling units, which represent the largest portion of the investment. For backwash water treatment, the investment difference is 556 k€, with EC requiring 604 k€ (for a maximum footprint of 140 m²) compared to 48 k€ for CC. In surface water treatment, the difference is even greater, with EC costing 4707 k€ (for a maximum footprint of 1517 m²) versus 155 k€ for CC. This significant difference in investment costs is attributed to the complexity of EC units compared to $FeCl_3$ dosing units (see CAPEX cost distribution in Table 17, Table 18, Table 19 , Table 20, Appendix IV).

The operating costs for EC, based on the skid-based, bipolar serial connections design, are 2632 k€ per year, which is 110% higher than the costs for iron chloride dosing for surface water treatment (1000 m³/h). For backwash water treatment, the operating costs are 144 k€ per year (115% higher) than those for conventional coagulation, treating 79 m³/h. The primary factor driving these higher costs is the substantial difference in iron dosage between the two technologies, leading to an EC energy consumption cost of approximately 991 k€ per year for surface water (Table 8) and 78 k€ per year for backwash water (Table 10) at a current energy cost of 0.18 €/kWh. If energy costs return to historical values of around 0.07 €/kWh, the energy cost for EC in surface water treatment would drop to about 396 k€ per year, making it more competitive with the total operational expenditure (OPEX) of conventional coagulation, which stands at 420 k€ per year (
Table 9).

Table 10 OPEX Iron electrocoagulation (EC) for sand filtration backwash water

OPEX Costs (€/year)	Flocculation	Lamela Settling	Electrocoagulation	TOTAL COSTs
Interest/Depreciation	€ 24.200	€ 49.925	€ 39.711	**€ 113.800**
Energy	€ 2.491	€ 623	€ 78.262	**€ 81.377**
Chemicals	€ 0	€ 0	€ 18.636	**€ 18.636**
Waste	€ 0	€ 0	€ 19.255	**€ 19.255**
Maintenance	€ 5.617	€ 10.018	€ 7.942	**€ 23.600**
Operator	€ 2.423	€ 2.423	€ 3.177	**€ 8.000**
Security	€ 390	€ 830	€ 794	**€ 2.000**
Administration	€ 563	€ 651	€ 794	**€ 2.000**
TOTAL	€ 36.000	€ 64.000	€ 169.000	**€ 269.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 40%	+/- 40%

For backwash water, the total OPEX for EC is 169 k€ per year compared to 25 k€ per year for conventional coagulation (see Table 10 and Table 11).

For FeCl3 dosing, the primary OPEX contributions are chemical consumption costs, amounting to 266 k€ per year, and waste management (sludge) costs of 128 k€ per year for surface water (

Table 9). For backwash water, these costs are 6 k€ and 13 k€ per year, respectively (Table 11).

Table 11 OPEX Iron chloride dosing (CC) for sand filtration backwash water

OPEX Costs (€/year)	Flocculation	Lamela Settling	FeCl3 dosing	TOTAL COSTs
Interest/Depreciation	€ 24.200	€ 49.925	€ 3.265	**€ 77.400**
Energy	€ 2.491	€ 623	€ 362	**€ 3.477**
Chemicals	€ 0	€ 0	€ 6.322	**€ 6.322**
Waste	€ 0	€ 0	€ 13.094	**€ 13.094**
Maintenance	€ 5.617	€ 10.018	€ 1.677	**€ 17.300**
Operator	€ 2.423	€ 2.423	€ 472	**€ 5.300**
Security	€ 390	€ 830	€ 48	**€ 1.300**
Administration	€ 563	€ 651	€ 104	**€ 1.300**
TOTAL	€ 36.000	€ 64.000	€ 25.000	**€ 125.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 40%	+/- 40%

Clearly, energy costs play a crucial role in determining the feasibility of EC in terms of OPEX. Therefore, EC must be optimized in terms of Fe dosage, electrode lifetime, and energy consumption to become competitive with FeCl3. The EC electrodes are estimated to need replacement every 53 days, compared to the processing of an iron chloride supply every 12 days. Additionally, it is important to note that dosing iron chloride increases the chloride concentration in treated water by 38.1 mg Cl/L for surface water and 11.44 mg Cl/L for backwash water, respectively (see Table 7).

The sensitivity analysis results for the total OPEX taken into account the energy price, energy consumption, iron electrodes cost (steel plates) and FeCl3 are shown in Figure 17 and Figure 18 for surface water and sand filter backwash water, correspondingly.
Result showed that mainly energy price and energy consumption (directly related to the iron dosage) are the main factors driving the EC operational costs. If Energy prices revert to historical values of 0.09 €/kWh a reduction of about 19.6% in total OPEX for surface water and 15.6% for backwash water will be achieved (Figure 17 A and Figure 18 A). Even higher reduction will be obtained if the EC unit energy consumption (actual 0.63 KWh/m3) is reduced by half or by four achieving 23% and 35% reduction of the total OPEX (see Figure 17 B and Figure 18 B). It must be highlighted, the latter might be an even higher reduction since this calculation does not taken into account the CAPEX reduction associated with it and therefore the interest/depreciation which is significant in the case of EC. Furthermore, this energy consumption reflects the energy efficiency, which will improve from 15 KWh/kg Fe to 3.8 KWh/kg Fe. This efficiency can be also improved if the EC system is optimized and managed to achieve similar removal efficiencies with similar or lower dosages than the ones observed during the pilot (42 mg Fe/l). If an energy efficiency of about 5 KWh/kg Fe is achieved, EC will become more financially and technically attractive. Additionally it was found that the iron electrodes (plates) cost will not impact significantly the operational costs with only a reduction of 3% if price is reduced to half and 6% if is double for surface water (Figure 17 C), and similarly for backwash water (Figure 18 C), under the set EC design conditions. On the other hand, the cost of FeCl3 for conventional coagulation-flocculation will have an impact higher or lower depending on the iron dosage. In the

case of surface water in which the dosage is higher, then 11% OPEX reduction is attainable if cost is reduced, and 21% increase if the price is double (Figure 17 D). For backwash water, the OPEX impact will be 2% reduction and 6% increase for half and double cost, respectively (Figure 18 D).

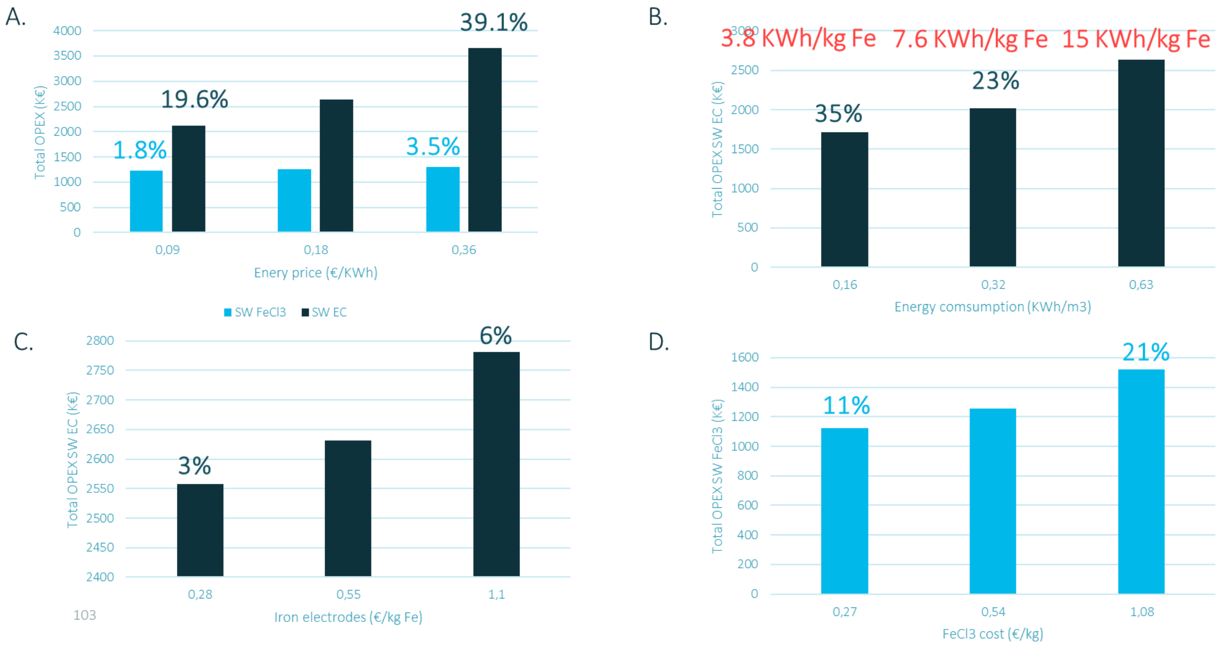


Figure 17 Total OPEX Sensitivity analysis for surface water. A. Energy price. B. Energy consumption . C. Iron electrodes cost. D. FeCl3 cost.

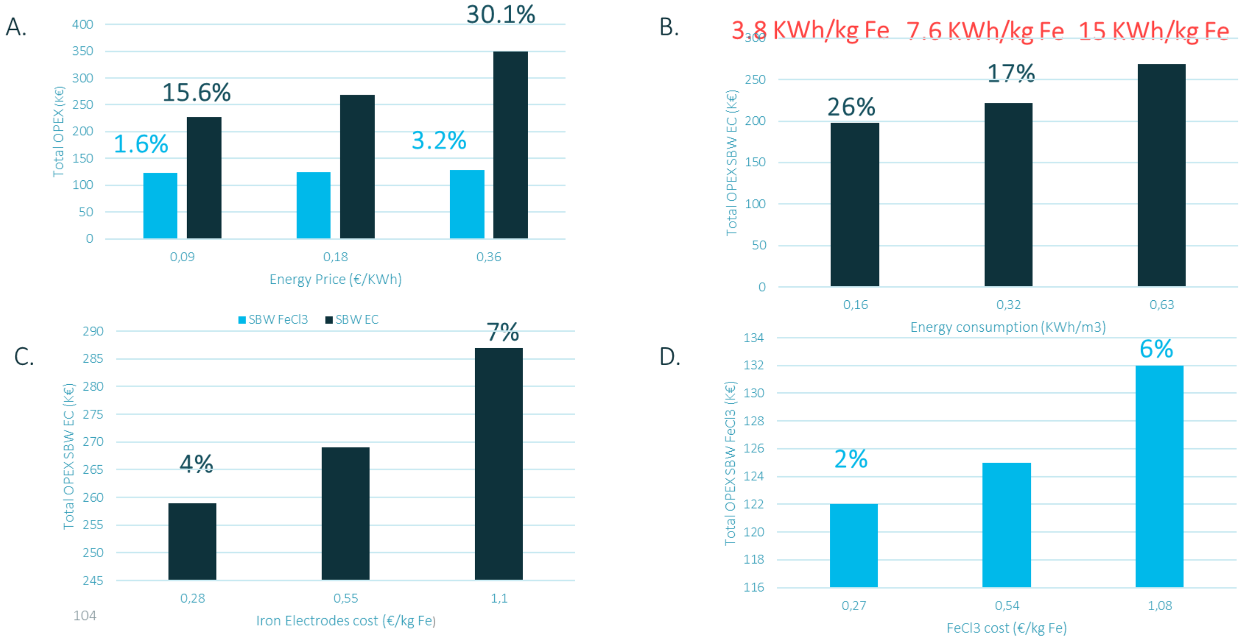


Figure 18 Total OPEX Sensitivity analysis for sand filter backwash water. A. Energy price. B. Energy consumption . C. Iron electrodes cost. D. FeCl3 cost.

3.3 Concluding remarks

From the cost evaluation, the following main remarks were formulated:

- The investment costs for EC are considerably higher than for conventional $FeCl_3$ coagulation, primarily due to the complexity of the EC units. For surface water treatment, EC requires an investment of 4707 k€, whereas CC requires only 155 k€. For backwash water, the investment is 604 k€ for EC compared to 48 k€ for CC. Additionally, the operating costs for EC are significantly higher, with EC costing 2632 k€/year for surface water treatment, 110% more than CC, and 169 k€/year for backwash water, 115% more than CC.

- Optimization of the Fe dosage and floc formation, and therefore energy consumption and plate lifetime, are crucial for EC to be competitive with $FeCl_3$ CC. The feasibility of EC largely hinges on energy costs, given its high energy consumption when Fe dosages are needed to be higher than in conventional coagulation. If energy costs revert to historical values of 0.09 €/kWh, the energy cost for EC in surface water treatment would make this EC design closer to the total operating costs of CC (420 k€/year).

- EC requires frequent electrode replacement every 53 days, compared to $FeCl_3$ supply processing every 12 days. These factors, along with waste management and chemical consumption costs, influence the overall sustainability and operational efficiency of the treatment processes.

4 Life Cycle Analysis

4.1 Materials and Methods

This section outlines the methodology used to conduct the Life Cycle Assessment (LCA) for the previously described scenarios. The methodological foundation was based on the ISO 14040/44:2006 framework (International Standards Organisation, 2006).The system boundary is described in 3.1.1 with the functional unit of 1 m^3 of influent water.

SimaPro 9.2 software (Sustainability, 2024) and the Ecoinvent database (EcoInvent, 2024) were utilised to facilitate the LCA. Detailed information regarding the life cycle inventory is provided in Appendix III.

Additionally, an Environmental Footprint (EF) assessment was conducted as part of the LCA. This sub-method of LCA, commonly used in the European Union (Commission, Environmental Footprint Methods, 2024), evaluates the environmental impact of various scenarios. The EF assessment provides a standardised approach for industries to assess their production processes using reliable data. The method defines specific impact categories, which quantify the environmental burden of a process. These categories are detailed in Table 6.

Table 12 Different impact categories used in this study.

Impact Category
Climate change
Ozone depletion
Ionising radiation
Photochemical ozone formation
Particulate matter
Human toxicity, non-cancer and cancer
Acidification
Eutrophication, freshwater
Eutrophication, marine
Eutrophication, terrestrial
Ecotoxicity, freshwater
Land use
Water use
Resource use, fossils
Resource use, minerals and metals

Each impact category generates a score reflecting its severity, but direct comparison between categories can be challenging. To address this issue, the analysis normalised the scores into a single unit, facilitating a direct comparison of impact magnitudes. Subsequently, subjective weights were assigned to different categories based on their perceived importance, expressed in EcoPoints per cubic meter of influent water. This approach simplifies the results and provides a clear understanding of the overall environmental impact (Commission, Normalisation method and data for Environmental Footprints, 2014).

This study acknowledges limitations due to data availability. Comprehensive LCAs require in-depth data for each stage of the product's life cycle. When specific details are unavailable, generic industry processes are used as substitutes. This approach can introduce uncertainties into the calculations.

In this study, we assumed an average Dutch energy mix for electricity consumption. The data on electricity technology shares relies on 2018 figures derived from calculations by data providers and statistics from the International Energy Agency. While the company may be purchasing green electricity, this electricity gets mixed within the national grid. Consequently, using an average Dutch energy mix provides a more accurate representation for this study. Green electricity procurement doesn't guarantee green consumption due to the nature of the grid, where all sources of electricity are blended together, making it difficult to trace specific green energy use directly to the end consumer (Zhu, Y. (2020) & *When 100% renewable energy doesn’t mean zero carbon*. (2019, May 23)). This serves as the baseline for the study.

4.2 Results and Discussion

The results section of this study is structured to provide a detailed analysis of four different scenarios for water treatment. The scenarios include:

1. CC, Flocculation, and Sedimentation of Surface Water: This scenario examines the environmental impacts of using coagulation methods combined with flocculation and sedimentation for treating surface water.
2. EC, Flocculation, and Sedimentation of Surface Water: This scenario explores the use of electrocoagulation in place of coagulation for surface water treatment, followed by flocculation and sedimentation.
3. CC, Flocculation, and Sedimentation of Sand Filter Backwash Water: This scenario focuses on treating the more contaminated sand filter backwash water using coagulation, followed by flocculation and sedimentation.
4. EC, Flocculation, and Sedimentation of Sand Filter Backwash Water: This scenario evaluates the environmental impacts of using electrocoagulation for treating sand filter backwash water, combined with flocculation and sedimentation.

The results section is organized into several key comparative analyses to provide a comprehensive understanding of the environmental impacts associated with each treatment method and influent type. The structure includes:

- Analysis of Individual Scenarios: Each scenario is individually analysed to understand its specific environmental impacts across various categories, such as climate change, acidification, and resource use.
- Comparative Analysis of Scenario 1 and Scenario 2: This section compares the CC method with EC for surface water treatment, highlighting the differences in chemical and energy usage.
- Comparative Analysis of Scenario 1 and Scenario 3: This comparison focuses on the differences between treating surface water and sand filter backwash water using the coagulation method, emphasizing the impact of influent type on environmental outcomes.
- Comparative Analysis of Scenario 2 and Scenario 4: This section compares electrocoagulation for surface water versus sand filter backwash water, illustrating how the level of contamination in the influent affects the overall environmental impacts.
- Comparative Analysis of Scenario 3 and Scenario 4: This comparison focuses on coagulation with electrocoagulation for treating sand filter backwash water, providing insights into the trade-offs between chemical usage and energy consumption.

- Subsequently, a sensitive analysis is carried out to compare different energy type to understand the changes in impact when using a more renewable energy sources as the country is transitioning towards a greener energy agenda

- Lastly, an overview of carbon footprint in kg CO_2 equivalent will be presented

This structured approach allows for a clear and systematic evaluation of the environmental impacts of each scenario, facilitating a better understanding of the trade-offs and potential benefits associated with different water treatment methods and influent types.

4.2.1 Analysis of Individual Scenarios

CC, Flocculation, and Sedimentation of Surface Water

Figure 19 shows that the biggest environmental impact is driven by the significant chemical usage and energy consumption. The production and transportation of iron (III) chloride, along with associated infrastructure materials such as reinforcing steel and concrete blocks, contribute to the overall environmental burden. These factors result in higher impacts on climate change, resource use, acidification, eutrophication, human toxicity, and particulate matter formation.

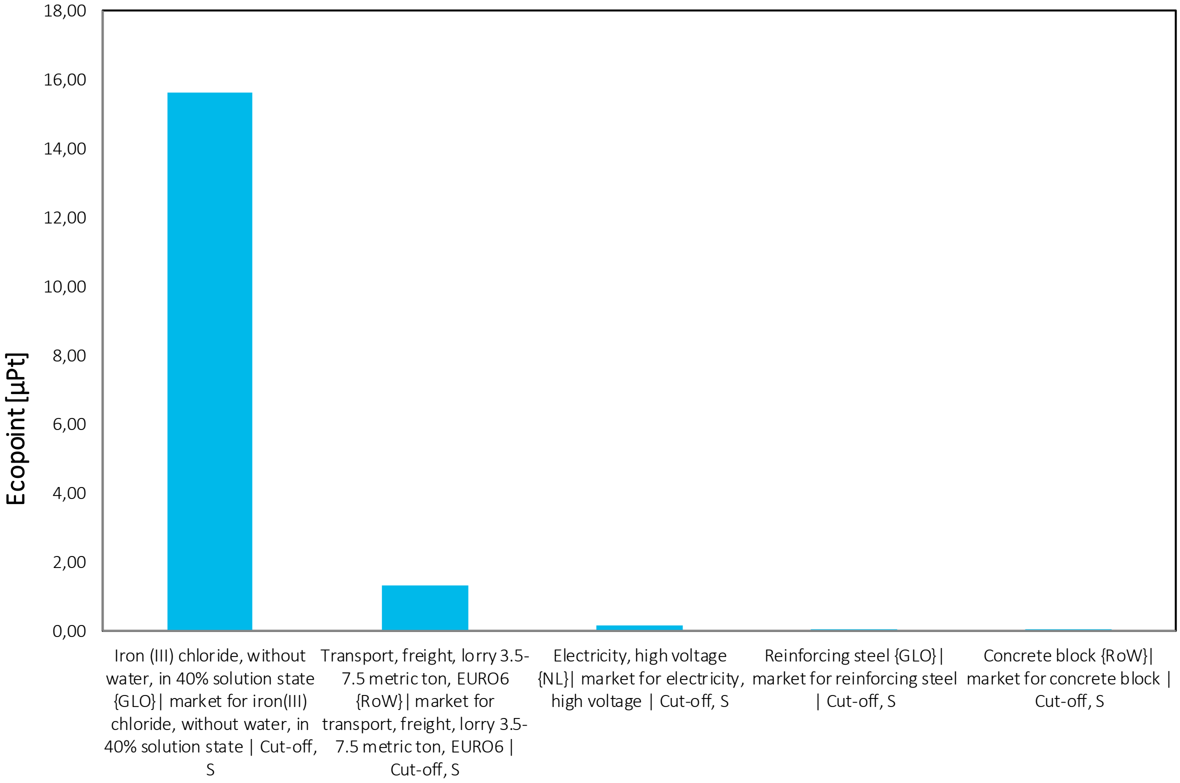


Figure 19 Process contribution towards environmental impact from coagulation, flocculation and sedimentation for 1m³ of surface water

CC demonstrates higher environmental impacts in most categories, compared to Flocculation and Sedimentation. Relatively, flocculation and sedimentation processes exhibit much lower environmental impacts as shown in Figure 20. These methods require fewer chemical inputs and less energy, translating to reduced environmental footprints. For instance, the climate change impact for flocculation (0.00059 µPt) and sedimentation (0.29 µPt) are significantly lower than for coagulation. Similarly, the resource use and toxicity impacts are also minimal compared to coagulation.

Some key findings from the LCA study of 1 m^3 of coagulation, flocculation, and sedimentation of surface water are:

- Climate Change: The conventional coagulation process has the highest impact on climate change, with a value of 3.56 µPt, largely due to the energy-intensive nature of producing and transporting coagulants such as iron (III) chloride.
- Resource Use (Minerals and Metals): Conventional coagulation shows a significantly higher impact (4.47µPt) compared to flocculation (0.000078 µPt) and sedimentation (0.046 µPt). This is due to the extensive use of chemicals and infrastructure materials in the coagulation process.
- Acidification and Eutrophication: These categories also exhibit higher impacts for conventional coagulation (0.93 µPt) for acidification and 1.10 µPt for freshwater eutrophication), driven by the chemical inputs and their environmental consequences.

This comparison underscores the necessity of considering environmental impacts comprehensively when selecting water treatment methods. While coagulation is effective, its environmental footprint suggests the need for optimization or alternative methods. Flocculation and sedimentation present promising alternatives with lower environmental impacts but their operational costs must be carefully evaluated.

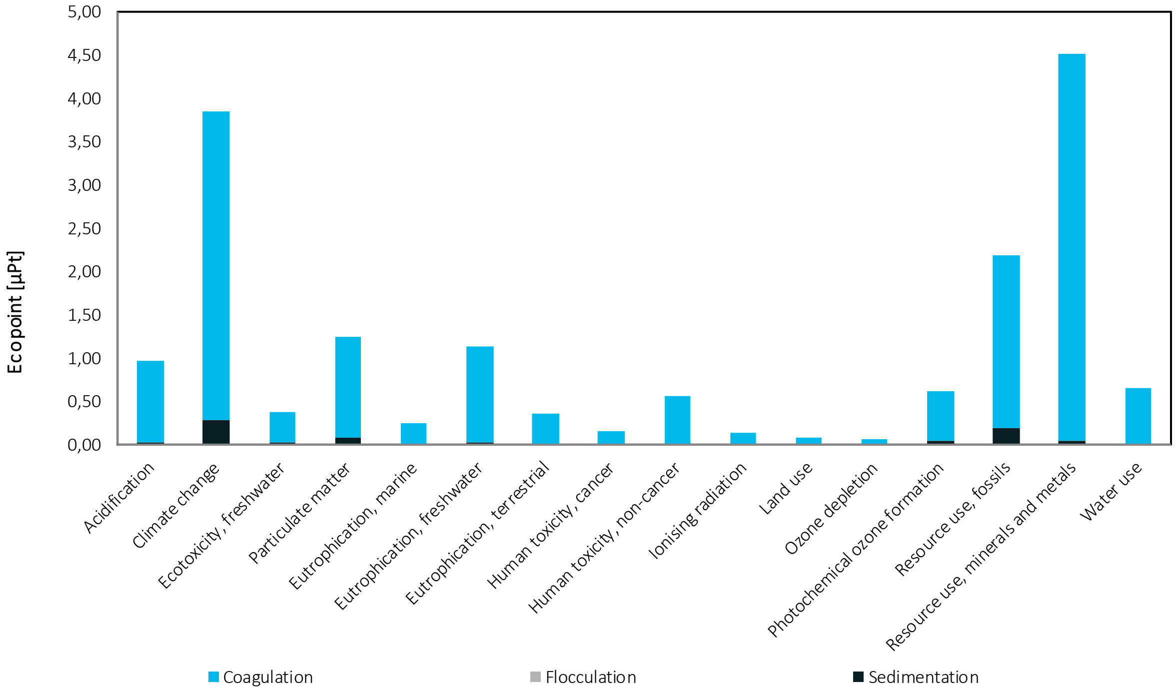


Figure 20 Life cycle impact assessment of coagulation, flocculation and sedimentation for 1m3 of surface water

Coagulation, Flocculation, and Sedimentation of Sand Filter Backwash Water

The LCA study of 1 m^3 of coagulation, flocculation and sedimentation of sand filter backwash water exhibits similar trends of impact assessment compared to that of surface water (Figure 21) with key findings summarised below, also in Figure 22:

- Climate Change: The coagulation process shows a significant impact on climate change with a value of 1.10 µPt, which is higher than flocculation (0.00098 µPt) and sedimentation (0.40 µPt). This high impact is mainly due to the use of chemicals and energy, particularly the production and transportation of iron (III) chloride.
- Resource Use (Minerals and Metals): Coagulation has the highest impact on the use of minerals and metals (1.34 µPt), reflecting the heavy reliance on chemicals and construction materials. Flocculation and sedimentation have significantly lower impacts in this category, with values of 0.00013 and 0.063 µPt, respectively.
- Acidification and Eutrophication: Coagulation leads to higher acidification (0.28 µPt) and eutrophication impacts (0.34 µPt) for freshwater eutrophication) due to the intensive use of chemicals in the process.
- Human Toxicity: The impacts on human toxicity, both cancer (0.046 µPt) and non-cancer (0.16 µPt), are higher for coagulation compared to flocculation and sedimentation. The use of chemicals and their emissions contribute significantly to these impacts.
- Particulate Matter: Coagulation also shows a higher impact on particulate matter formation (0.35 µPt), indicating a significant contribution to air pollution, compared to much lower values for flocculation (0.00043 µPt) and sedimentation (0.11 µPt).

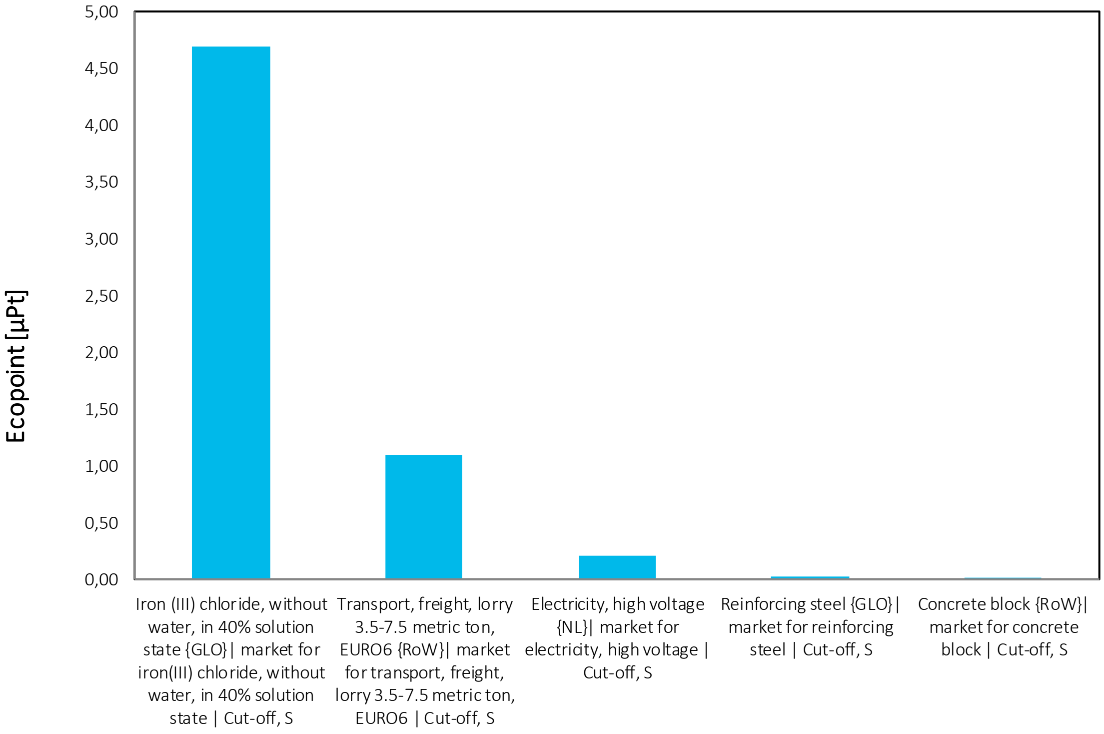


Figure 21 Process contribution towards environmental impact from coagulation, flocculation and sedimentation for 1m^3 of sand filter backwash water

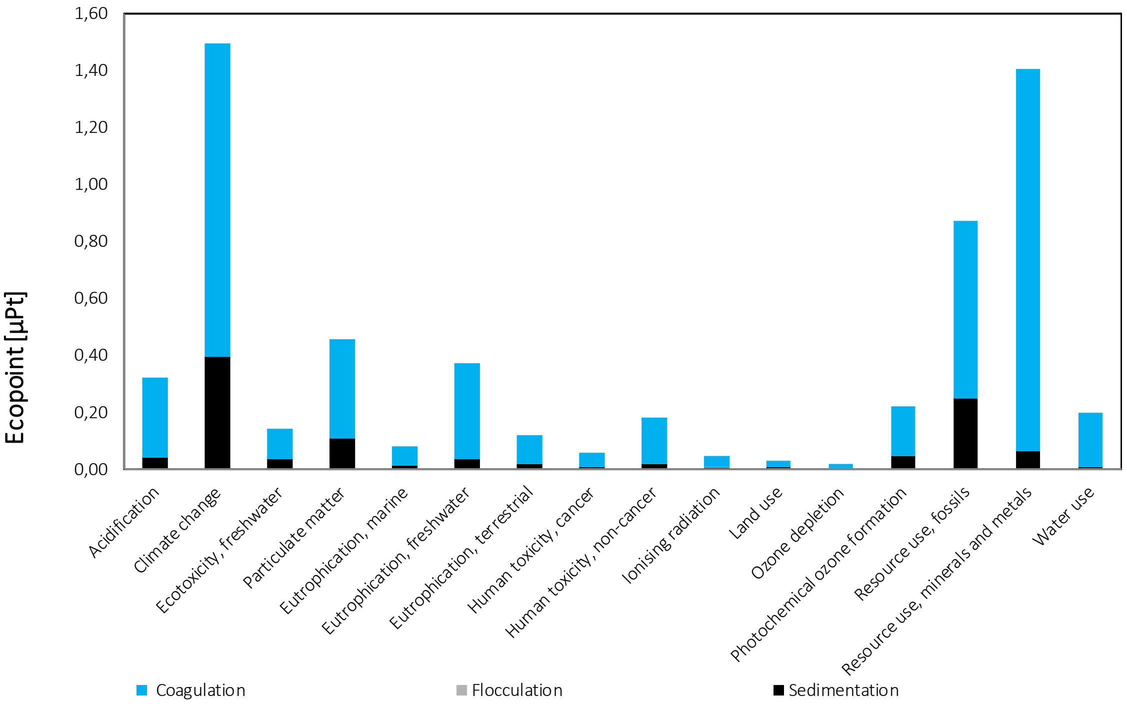


Figure 22 Life cycle impact assessment of coagulation, flocculation and sedimentation for 1m3 of sand filter backwash water

Electrocoagulation, Flocculation, and Sedimentation of Surface Water

The Life Cycle Assessment (LCA) results for electrocoagulation on surface water reveal significant environmental impacts across various categories (Figure 23 and Figure 24) . The analysis includes categories such as acidification, climate change, ecotoxicity, particulate matter, eutrophication (marine, freshwater, and terrestrial), human toxicity (cancer and non-cancer), ionising radiation, land use, ozone depletion, photochemical ozone formation, resource use (fossils and minerals/metals), and water use.

The electrocoagulation process shows environmental impacts primarily driven by high electricity consumption. The key contributing processes include:

- Electricity Consumption: With an impact score of 20.24 µPt, it is the most significant contributor, affecting climate change, resource use (fossils), acidification, particulate matter, and photochemical ozone formation.
- Transport: Freight transport, particularly lorry transport, contributes significantly (1.49 µPt), impacting fossil resource use and climate change.
- Material Production: The production of coke (0.44 µPt), pig iron (0.41 µPt), and hard coal (0.34 µPt) for equipment and electrode manufacturing also contribute notably to resource use and human toxicity.

In comparison, flocculation and sedimentation processes have lower environmental impacts due to reduced energy and material demands. The primary contributions from these processes are related to the minimal use of chemicals and energy, resulting in lower impacts across all categories.

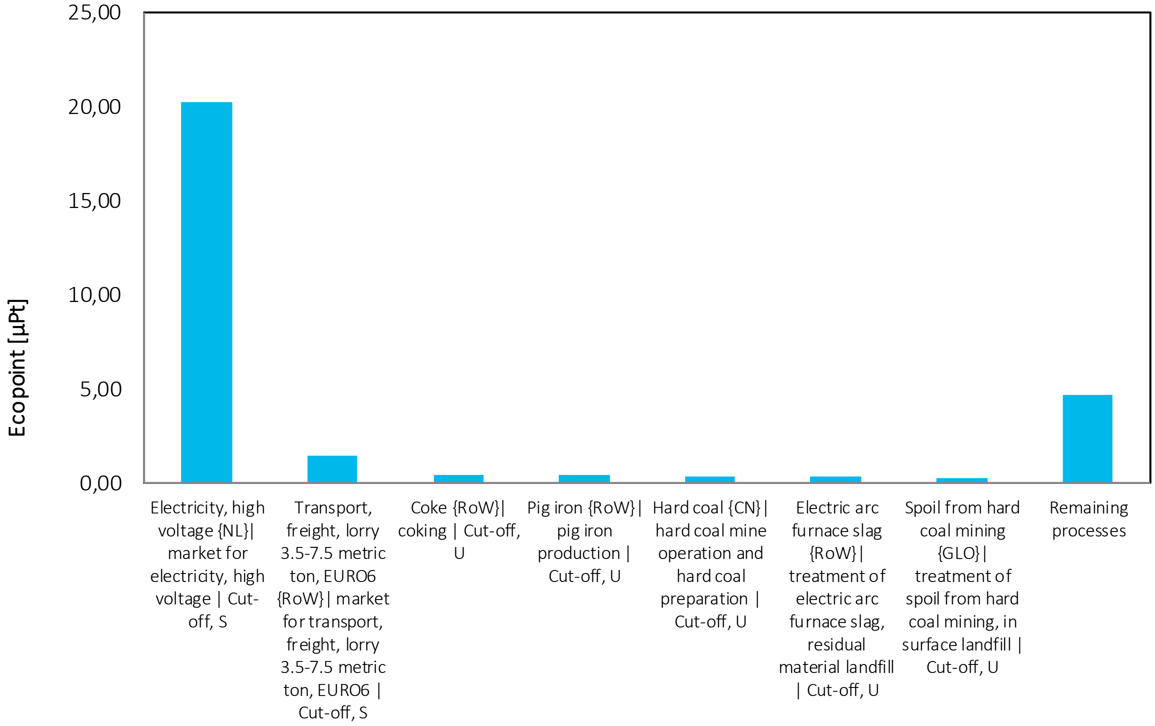


Figure 23 Process contribution towards environmental impact from electrocoagulation, flocculation and sedimentation for 1m3 of surface water with 1% cut off point

Key Findings:

- Climate Change: The electrocoagulation process shows a significant impact on climate change with a value of 10.50 µPt. This high impact is primarily due to the electricity consumption required for the process. The production and consumption of electricity, especially if sourced from non-renewable sources, contribute heavily to greenhouse gas emissions, thus affecting climate change.
- Resource Use (Fossils): Electrocoagulation has a high impact on fossil resource use, with a value of 6.71 µPt. This is likely driven by the energy demands of the process (including production), highlighting the dependency on fossil fuels for electricity production.

- Acidification: The impact score for acidification is 0.90 µPt, indicating a considerable contribution to environmental acidification. This could be due to the emissions of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) during electricity generation.
- Eutrophication: The impact scores for marine (0.35 µPt), freshwater (2.04 µPt), and terrestrial eutrophication (0.49 µPt) show significant nutrient loading to various ecosystems due to the process. This is likely linked to the release of nutrients and chemicals that promote excessive plant and algae growth in water bodies, leading to oxygen depletion and harm to aquatic life.
- Resource Use (Minerals and Metals): The impact score is 0.73 µPt, reflecting the use of mineral and metal resources. This includes materials used in the construction and maintenance of electrocoagulation equipment such as the steel plates as cathode and anode.

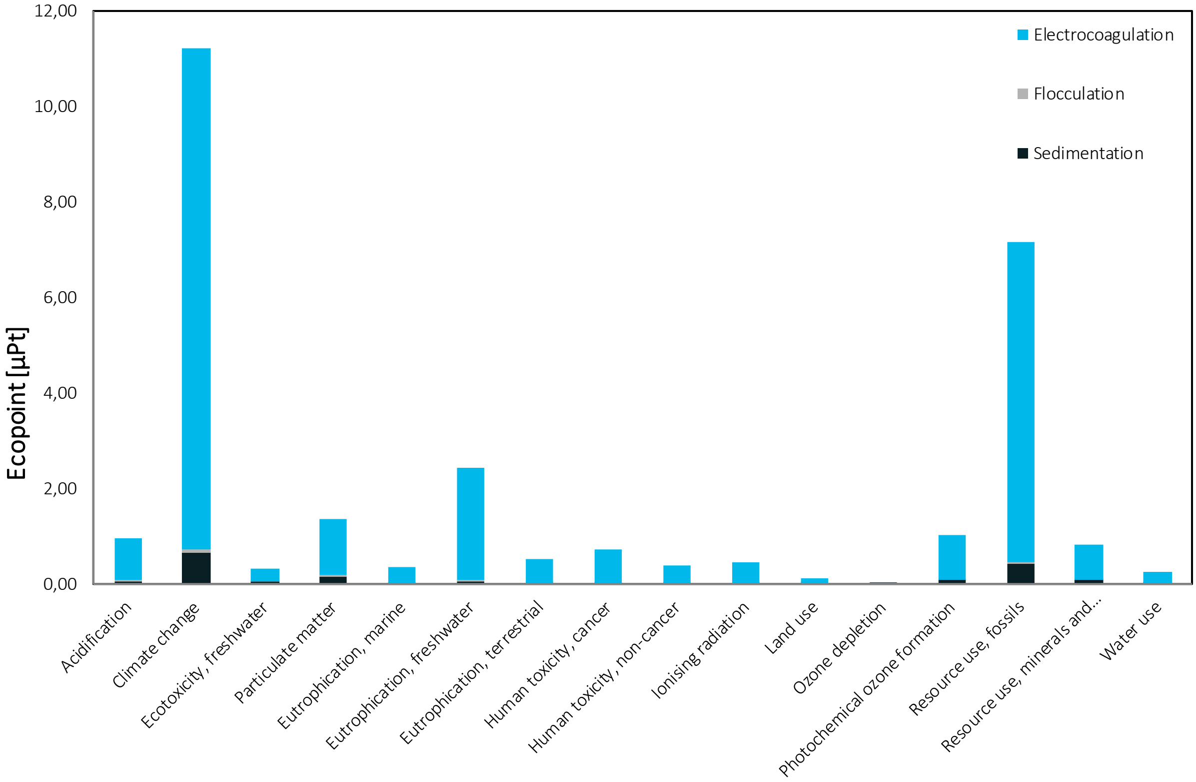


Figure 24 Life cycle impact assessment of from electrocoagulation, flocculation and sedimentation for 1m3 of surface water

These impact scores highlight the environmental burdens associated with the energy and material inputs required for the electrocoagulation process, particularly the reliance on high-voltage electricity, which significantly influences the overall environmental performance. The cut-off point for process contribution is capped at 1%, ensuring that only the most impactful processes are included in the analysis. This approach helps to focus on the most significant contributors to environmental impacts, making the assessment more manageable and targeted.

Electrocoagulation, Flocculation, and Sedimentation of Sand filter Backwash Water

Similar to above, the results (Figure 25) for electrocoagulation on sand filter backwash water again reveal significant environmental impacts across various categories. The electrocoagulation process (compared to flocculation and sedimentation) shows its environmental impacts primarily driven by high electricity consumption. The key contributing processes include:

- Electricity Consumption: With an impact score of 20.24 µPt, it is the most significant contributor, affecting climate change, resource use (fossils), acidification, particulate matter, and photochemical ozone formation.
- Transport: Freight transport, particularly lorry transport, contributes significantly (1.56 µPt), impacting fossil resource use and climate change.
- Material Production: The production of coke (0.69 µPt), pig iron (0.64 µPt), and hard coal (0.55 µPt) for equipment and electrode manufacturing also contribute notably to resource use and human toxicity.

In comparison, flocculation and sedimentation processes have lower environmental impacts due to reduced energy and material demands. The primary contributions from these processes are related to the minimal use of chemicals and energy, resulting in lower impacts across all categories.

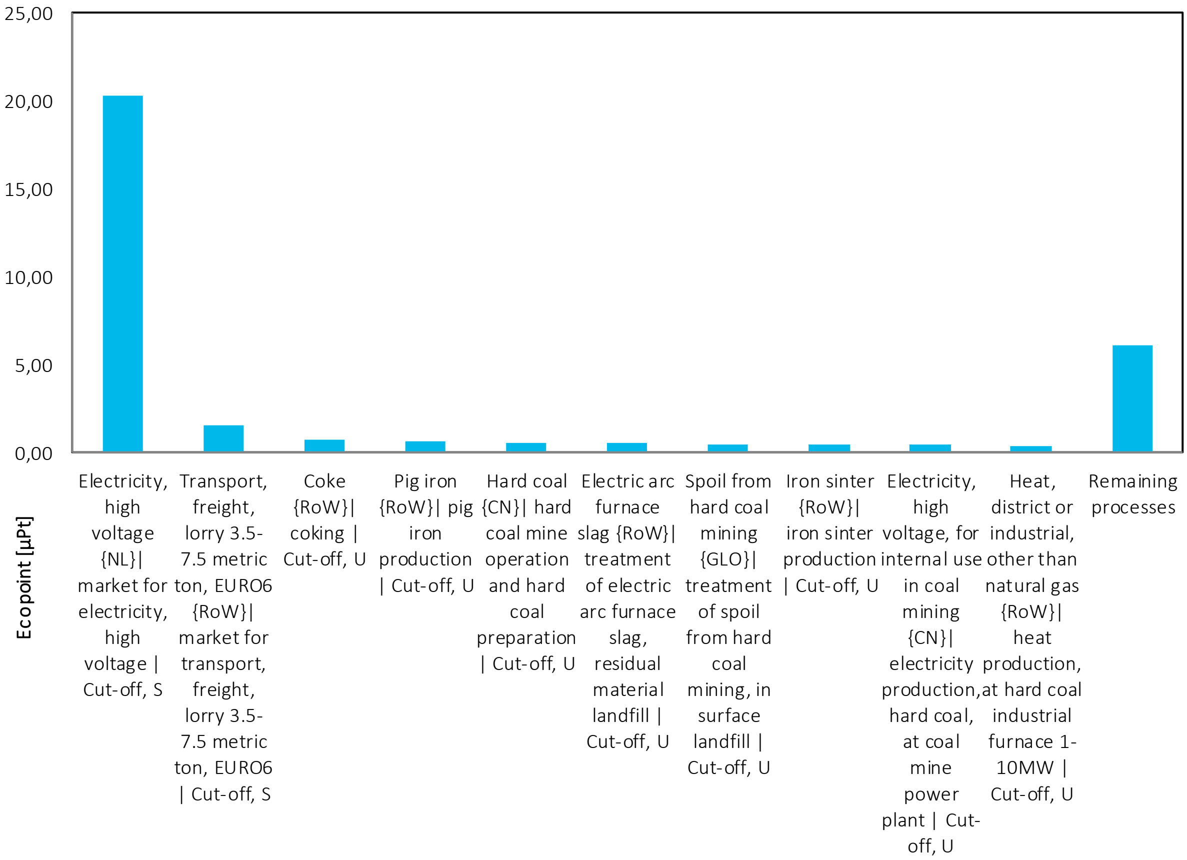


Figure 25 Process contribution towards environmental impact from electrocoagulation, flocculation and sedimentation for 1m3 of sand filter backwash water with 1% cut off point

The life cycle impact assessment results are shown in Figure 26 and some key findings are shown below:

- Climate Change: The electrocoagulation process shows a significant impact on climate change with a value of 11.53 µPt. This high impact is primarily due to the substantial electricity consumption required for the process. The production and consumption of electricity, especially if sourced from non-renewable sources, contribute heavily to greenhouse gas emissions, thus affecting climate change.
- Resource Use (Fossils): Electrocoagulation has a high impact on fossil resource use, with a value of 7.22 µPt. This is driven by the energy demands of the process, highlighting the dependency on fossil fuels for electricity production.

- Acidification: The impact score for acidification is 1.08 µPt, indicating a considerable contribution to environmental acidification. This could be due to the emissions of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) during electricity generation.
- Eutrophication: The impact scores for marine (0.40 µPt), freshwater (2.67 µPt), and terrestrial eutrophication (0.57 µPt) show significant nutrient loading to various ecosystems due to the process. This is likely linked to the release of nutrients and chemicals that promote excessive plant and algae growth in water bodies, leading to oxygen depletion and harm to aquatic life.
- Particulate Matter: The impact on particulate matter formation is 1.65 µPt, suggesting substantial contributions to air pollution and associated health risks. Particulate matter can cause serious health problems, including respiratory and cardiovascular diseases.
- Ecotoxicity, Freshwater: The impact score for ecotoxicity in freshwater ecosystems is 0.33 µPt, highlighting potential harm to aquatic life. This could be due to the release of toxic substances that affect the health of aquatic organisms.
- Ionising Radiation: The impact score is 0.47 µPt, reflecting potential radiation exposure risks. This can be associated with the electricity generation process, especially if nuclear energy is part of the energy mix.
- Land Use: The impact score for land use is 0.12 µPt, indicating the area of land affected by the process. This includes land used for mining raw materials and infrastructure.
- Ozone Depletion: The impact on ozone depletion is relatively low, with a score of 0.01 µPt. This suggests minimal contributions to the release of substances that deplete the ozone layer.
- Photochemical Ozone Formation: The impact score is 1.14 µPt, indicating contributions to ground-level ozone formation, which affects air quality and human health. This is primarily due to emissions of volatile organic compounds (VOCs) and NO_x.
- Resource Use (Minerals and Metals): The impact score is 1.04 µPt, reflecting the use of mineral and metal resources. This includes materials used in the construction and maintenance of electrocoagulation equipment.
- Water Use: The water use impact score is 0.22 µPt, indicating the volume of water consumed during the process. This is significant as it reflects the demand on water resources.

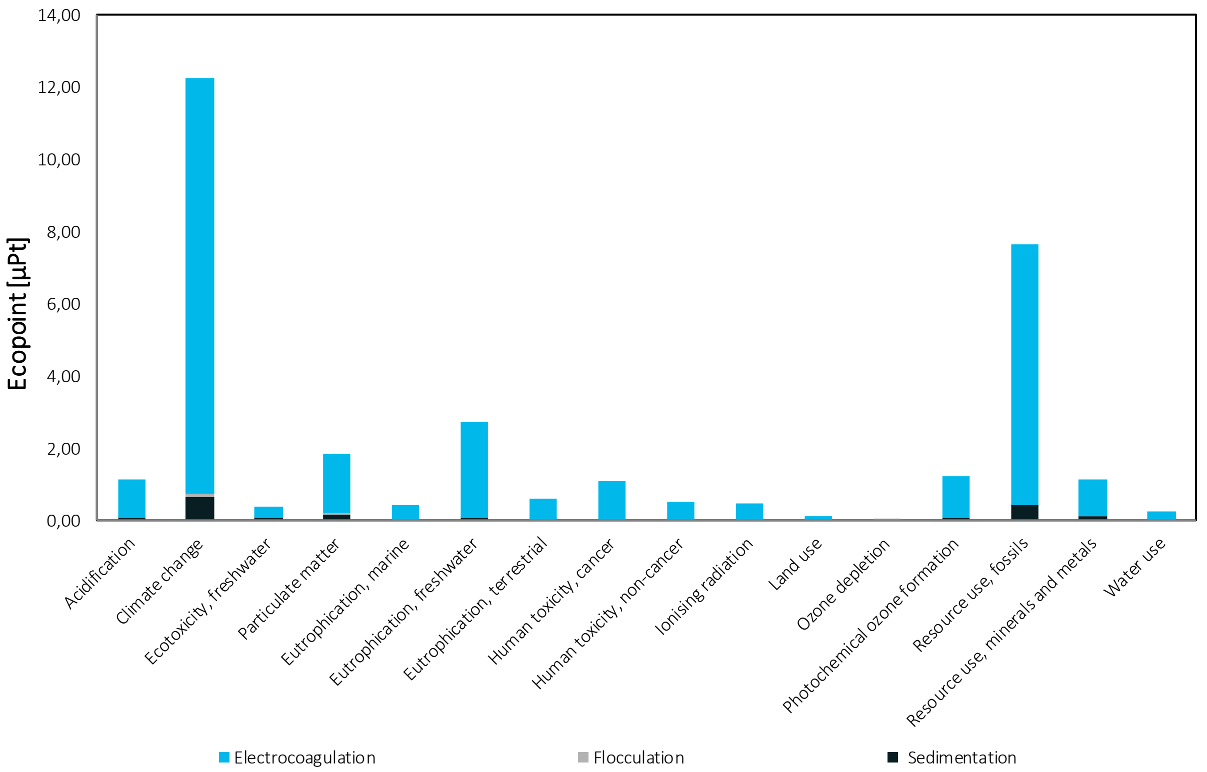


Figure 26 Life cycle impact assessment of from electrocoagulation, flocculation and sedimentation for 1m3 of sand filter backwash water

These impact scores highlight the substantial environmental burdens associated with the energy and material inputs required for the electrocoagulation process, particularly the reliance on high-voltage electricity, which significantly influences the overall environmental performance. The cut-off point for process contribution is capped at 1%, ensuring that only the most impactful processes are included in the analysis. This approach helps to focus on the most significant contributors to environmental impacts, making the assessment more manageable and targeted.

4.2.2 Comparative Analysis

Comparative Analysis of Surface Water and Backwash Water for CC (Scenario 1 and Scenario 3):

Figure 27 focuses on the differences between treating surface water and sand filter backwash water using the coagulation method, emphasizing the impact of influent type on environmental outcomes. The primary difference between treating sand filter backwash water and surface water using CC, flocculation and sedimentation process lies in the influent characteristics and the subsequent environmental impacts. Sand filter backwash water contains higher concentrations of suspended solids and potentially different contaminant profiles compared to surface water, leading to variations in the required treatment processes and the associated environmental impacts:

- Transport and Handling: The need for more frequent or higher doses of coagulants in surface water treatment leads to increased transport impacts. This is reflected in the overall higher resource use for fossil fuels in surface water treatment (2.19 µPt) compared to sand filter backwash water (0.87 µPt).
- Energy Consumption: The treatment of surface water demonstrates higher energy-related impacts, as seen in the climate change category (3.85 µPt for surface water vs. 1.05 µPt for sand filter backwash water). This is due to the additional energy required for handling and processing the higher contaminant load in surface water.

The overall environmental impacts of treating sand filter backwash water are generally lower across most categories compared to surface water treatment. This is reflected in the lower values for acidification, climate change, and resource use (minerals and metals) in the LCA results.

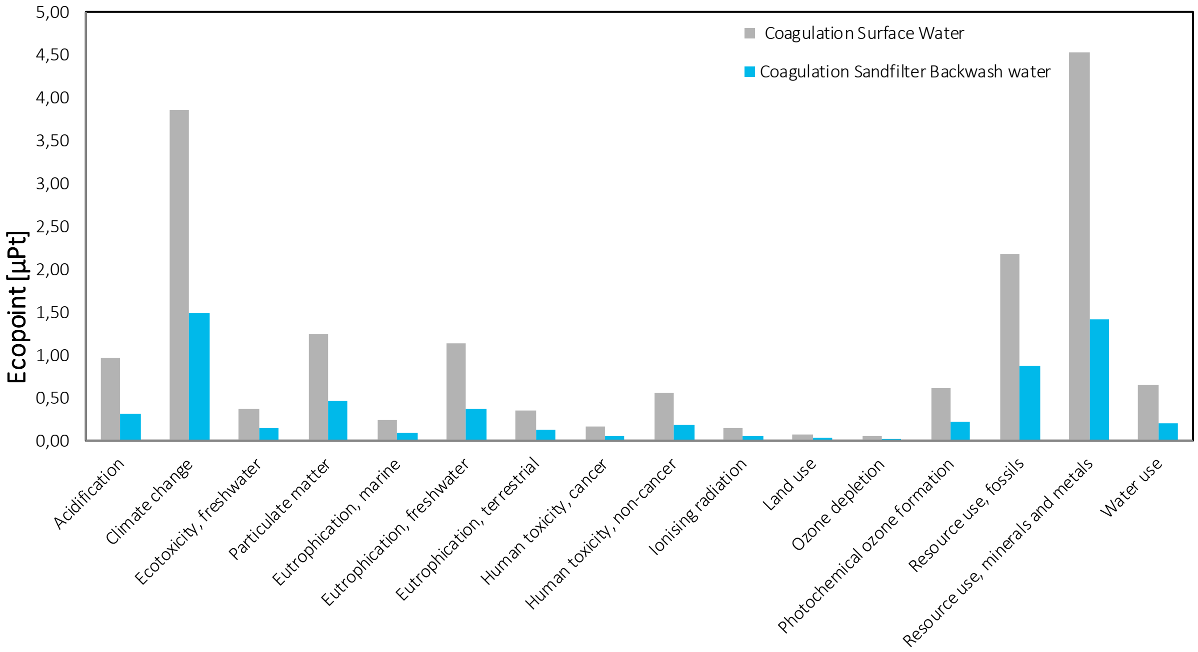


Figure 27 Life cycle impact assessment comparing treating surface water and sand filter backwash water using the coagulation method

Comparative Analysis of Surface Water and Backwash Water for EC (Scenario 2 and Scenario 4):

Figure 28 compares electrocoagulation for surface water versus sand filter backwash water, illustrating how the level of contamination in the influent affects the overall environmental impacts. The comparative results for electrocoagulation of surface water versus sand filter backwash water show some differences across various environmental impact categories. For climate change, electrocoagulation of sand filter backwash water has a higher impact score of 12.25 µPt compared to 11.22 µPt for surface water. This could be due to the higher energy consumption required in the production and transportation of steel plate for treating the more contaminated backwash water, as a higher amount of steel plate is required to treat 1m3 of sand filter backwash water compared to surface water. Resource use impacts are also higher for sand filter backwash water, with fossil resource use at 7.66 µPt compared to 7.16 µPt, and minerals and metals use at 1.15 µPt compared to 0.84 µPt. The higher environmental impacts associated with treating sand filter backwash water are primarily due to its higher contaminant load, which necessitates more intensive treatment processes, increased energy consumption, and greater material usage, particularly steel plates for electrodes in electrocoagulation.

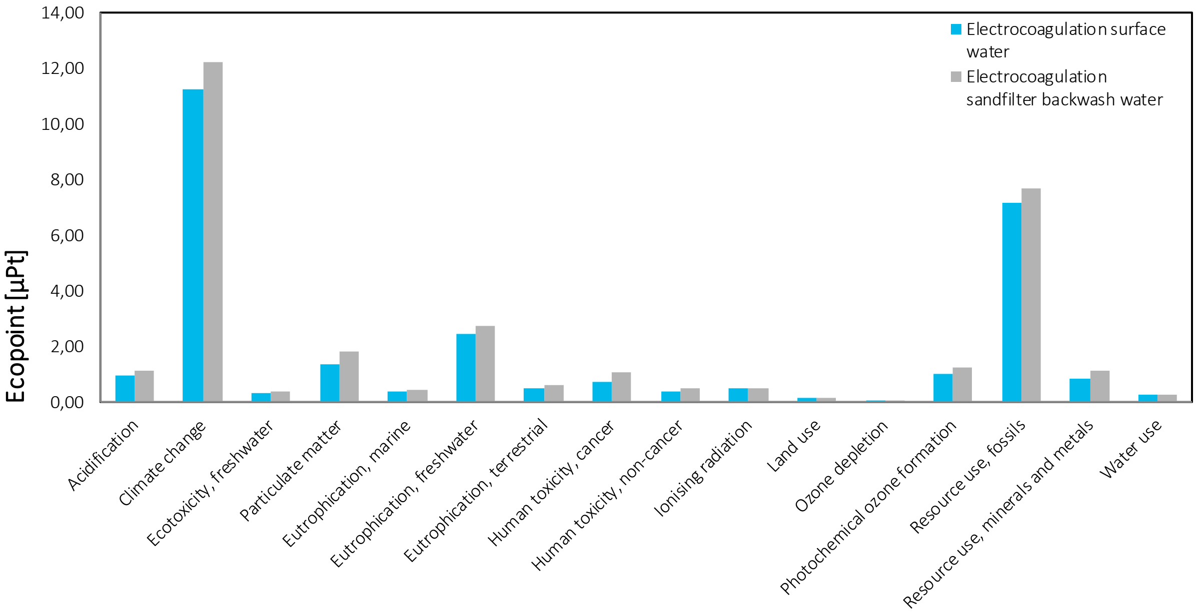


Figure 28 Life cycle impact assessment comparing treating surface water and sand filter backwash water using the electrocoagulation method

Comparative Analysis of Surface Water for CC and EC (Scenario 1 and Scenario 2)

Figure 29 compares the coagulation method with electrocoagulation for surface water treatment, highlighting the differences in chemical and energy usage. The most striking difference is observed in the climate change category, where electrocoagulation has a much higher impact (11.22 µPt) compared to coagulation (3.85 µPt). This disparity is primarily due to the substantial electricity consumption (which is related to the iron dosage difference) required by electrocoagulation. The process relies heavily on high-voltage electricity to drive the coagulation reaction, resulting in higher greenhouse gas emissions, especially if the electricity is sourced from non-renewable energy.

Another notable difference is in the resource use of fossils, where electrocoagulation again shows a significantly higher impact (7.16 µPt) compared to coagulation (2.19 µPt). This reflects the energy-intensive nature of electrocoagulation, which requires more fossil fuels to generate the necessary electricity. On the other hand, coagulation, which involves the chemical dosing of coagulants like iron salts, tends to have a lower energy footprint. Consequently, the reliance on fossil fuels for energy in electrocoagulation not only increases climate change impacts but also exacerbates fossil resource depletion.

Furthermore, for human toxicity, particularly cancer-related impacts, electrocoagulation shows higher values (0.72 µPt) compared to coagulation (0.16 µPt). This is likely due to the materials used in the electrodes, such as steel plates, which can release hazardous substances during their production and disposal. Additionally, the increased use of chemicals and energy in electrocoagulation can lead to higher emissions of toxic substances that pose health risks. Despite its higher impacts in several categories, electrocoagulation has a lower impact on mineral and metal resource use (0.84 µPt) compared to coagulation (4.52 µPt), suggesting that while it is more energy-intensive, it may require fewer raw materials for chemical production. These differences highlight the trade-offs between the two processes, emphasizing the need for careful consideration of environmental impacts when selecting a treatment method for surface water.

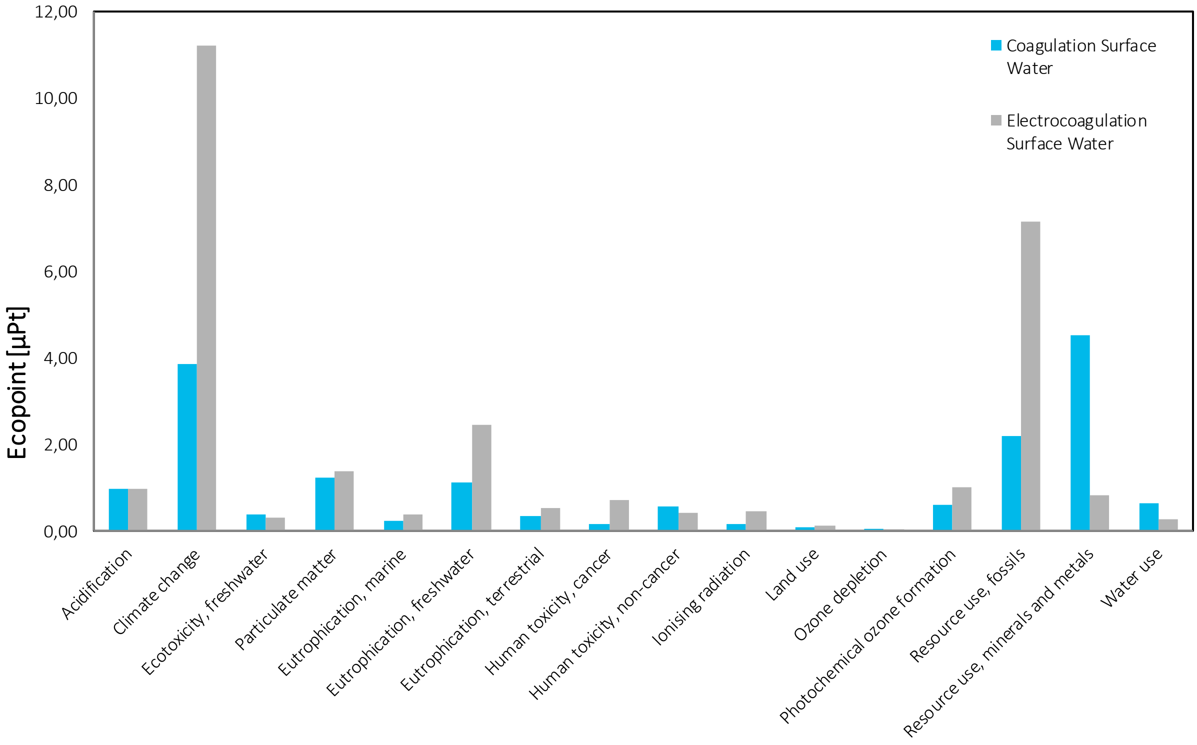


Figure 29 Life cycle impact assessment comparing the coagulation method with electrocoagulation for surface water treatment

Comparative Analysis of EC vs CC for Backwash Water (Scenario 3 and Scenario 4):

Figure 29 focuses on coagulation with electrocoagulation for treating sand filter backwash water, providing insights into the trade-offs between chemical usage and energy consumption. Similar to above, several differences emerge across various environmental impact categories. Overall similar trend as Figure 30. Again, the most notable distinction is observed in the climate change category, where electrocoagulation shows a much higher impact (12.25 µPt) compared to coagulation (1.49 µPt). This substantial difference is primarily due to the high energy consumption associated with electrocoagulation. Interestingly, the relative difference in certain impacts is higher because less coagulant is used for sand filter backwash water compared to surface water (0.044 kg/m^3 feed vs. 0.14 kg/m^3 feed), which significantly impacts the results by reducing the chemical-related burdens in the sand filter backwash water treatment vs surface water treatment.

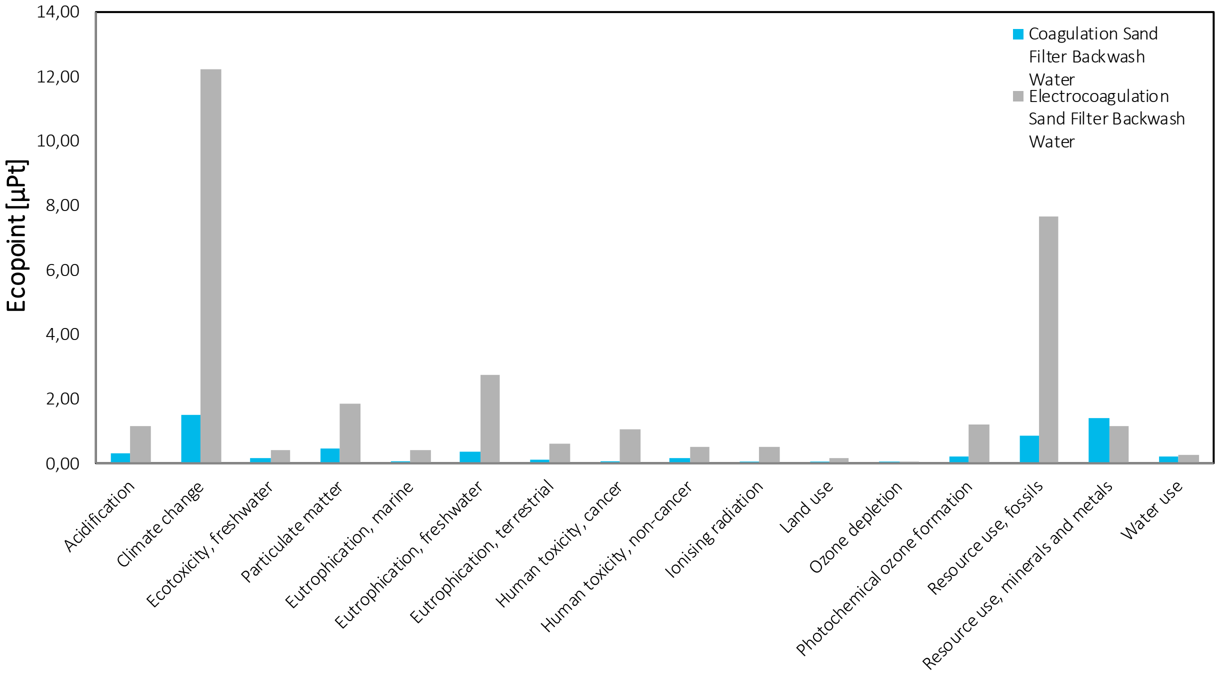


Figure 30 Life cycle impact assessment comparing coagulation with electrocoagulation for treating sand filter backwash water

Overview of all scenarios:

Figure 31 shows the comparative analysis highlights significant lifecycle trade-offs between coagulation and electrocoagulation processes in water treatment. Electrocoagulation offers potential advantages in terms of reduced chemical usage and lower impacts on mineral and metal resource use. However, it comes with significantly higher energy demands and associated environmental impacts, particularly in terms of climate change, fossil resource use, and human toxicity. Coagulation, while less energy-intensive, involves higher chemical use, leading to greater impacts on mineral and metal resource depletion.

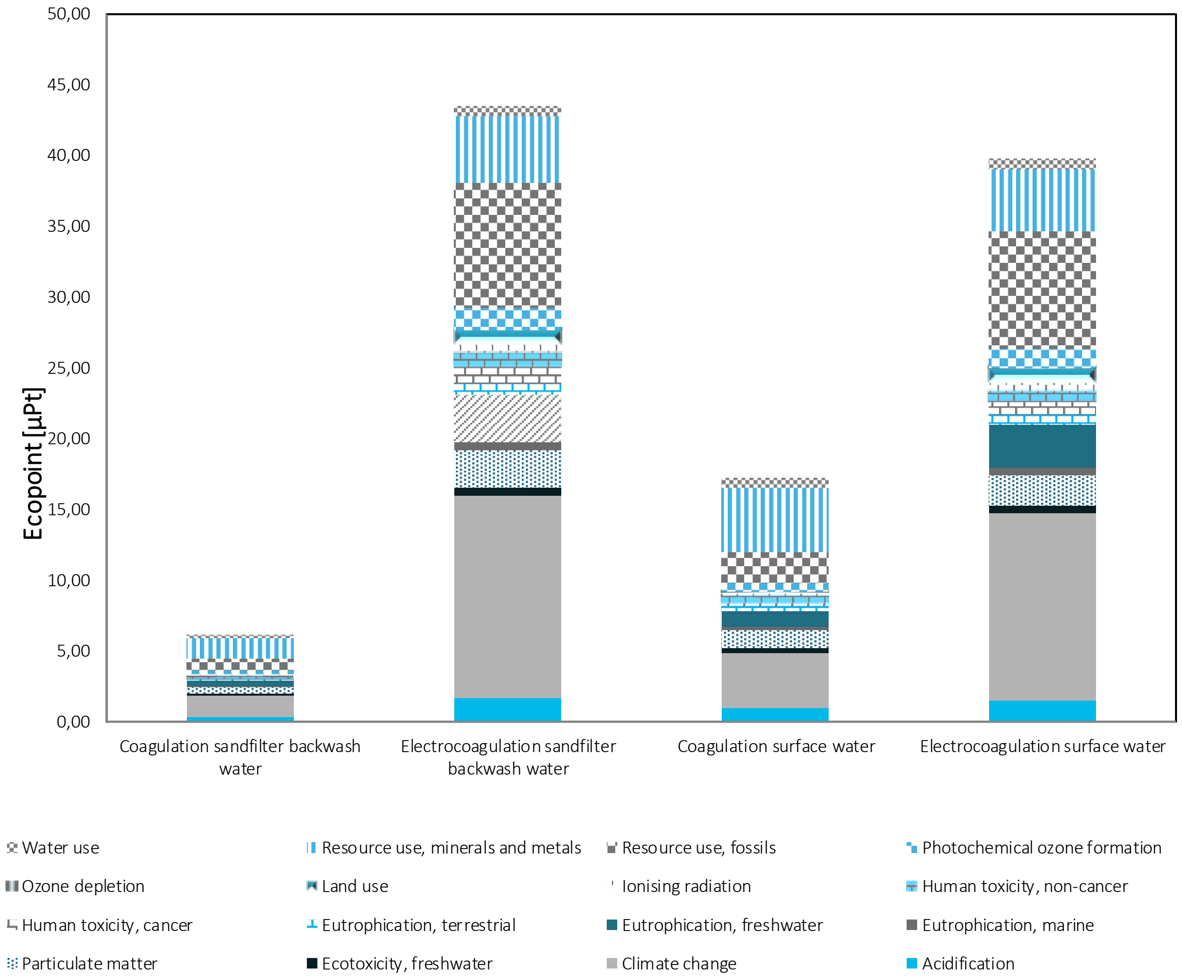


Figure 31 Overview of all scenarios

It is important to acknowledge that the environmental impact results presented so far in this analysis are based on the average Dutch electricity mix for grid power. As advancements in renewable energy technologies continue and the Netherlands transitions towards a greener grid, the relative environmental impacts of the treatment methods will likely change. To account for this uncertainty and provide more informed decision-making, a sensitivity analysis has be conducted. This analysis would explore how variations in the electricity source might influence the environmental footprint of each water treatment scenario. By incorporating such an analysis, we can gain a more nuanced understanding of how future energy landscapes may impact the environmental trade-offs between coagulation and electrocoagulation.

As expected and shown in Figure 32&Figure 33, a shift towards renewable energy sources significantly reduces the environmental impact associated with electrocoagulation, particularly for climate change. For example, using wind power for electrocoagulation of sand filter backwash water reduces the climate change impact from 14.29 to 3.65 µPt. This 74.43% percentage decrease highlights the potential environmental benefits of integrating renewable energy with electrocoagulation.

However, the impact on other environmental indicators varies. For instance, human toxicity associated with electrocoagulation remains a concern regardless of the energy source. This emphasizes the need for further research into electrode materials and potential life cycle optimizations within the electrocoagulation process itself.

In conclusion, this sensitivity analysis underscores the importance of considering future energy landscapes when evaluating water treatment methods. As the Netherlands progresses towards a greener grid, electrocoagulation powered by renewables may become a more environmentally friendly option. However, a holistic approach that considers all environmental aspects and ongoing research into improving both coagulation and electrocoagulation methods is crucial for sustainable water treatment.

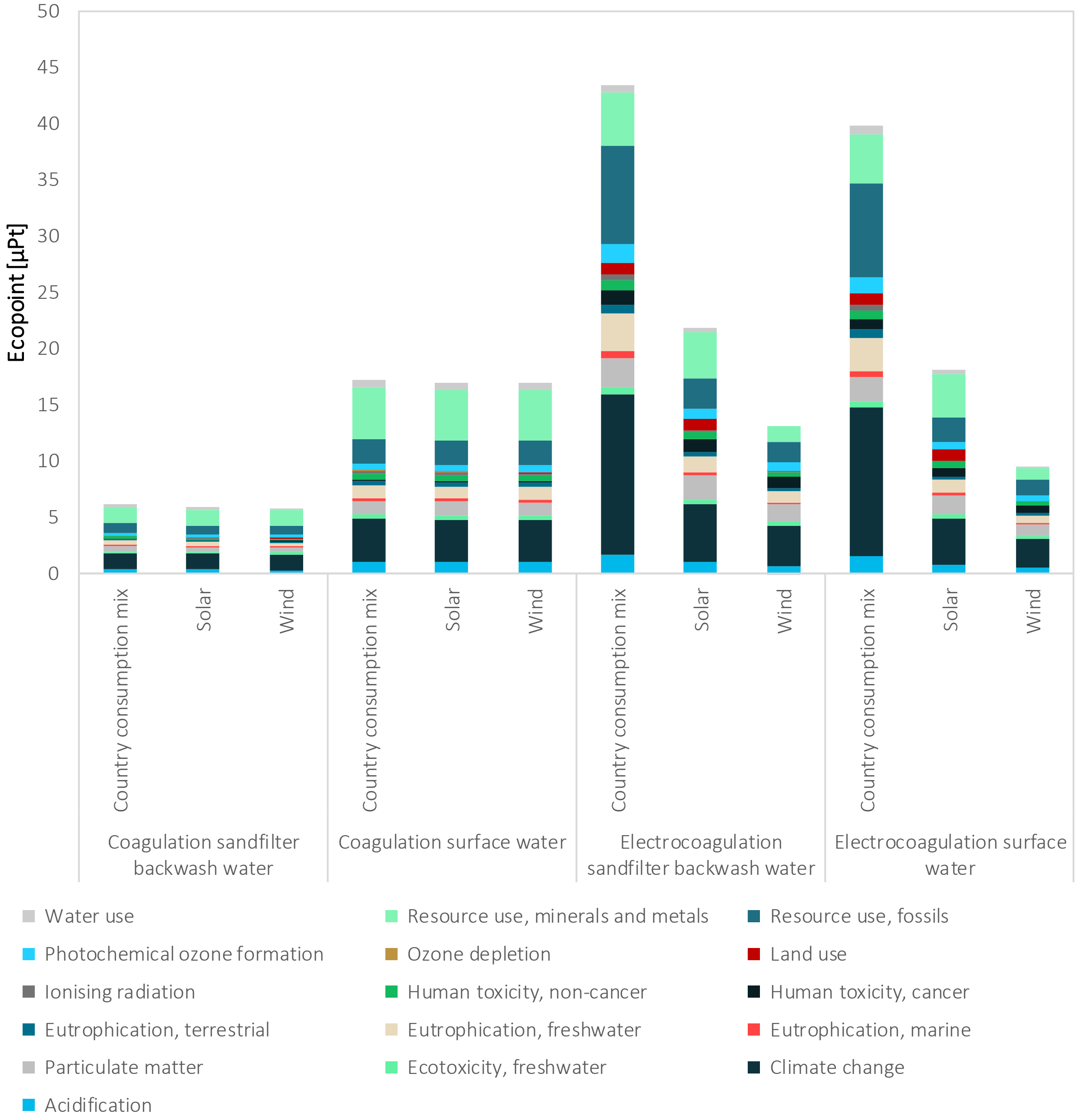


Figure 32 Sensitivity analysis using different energy type

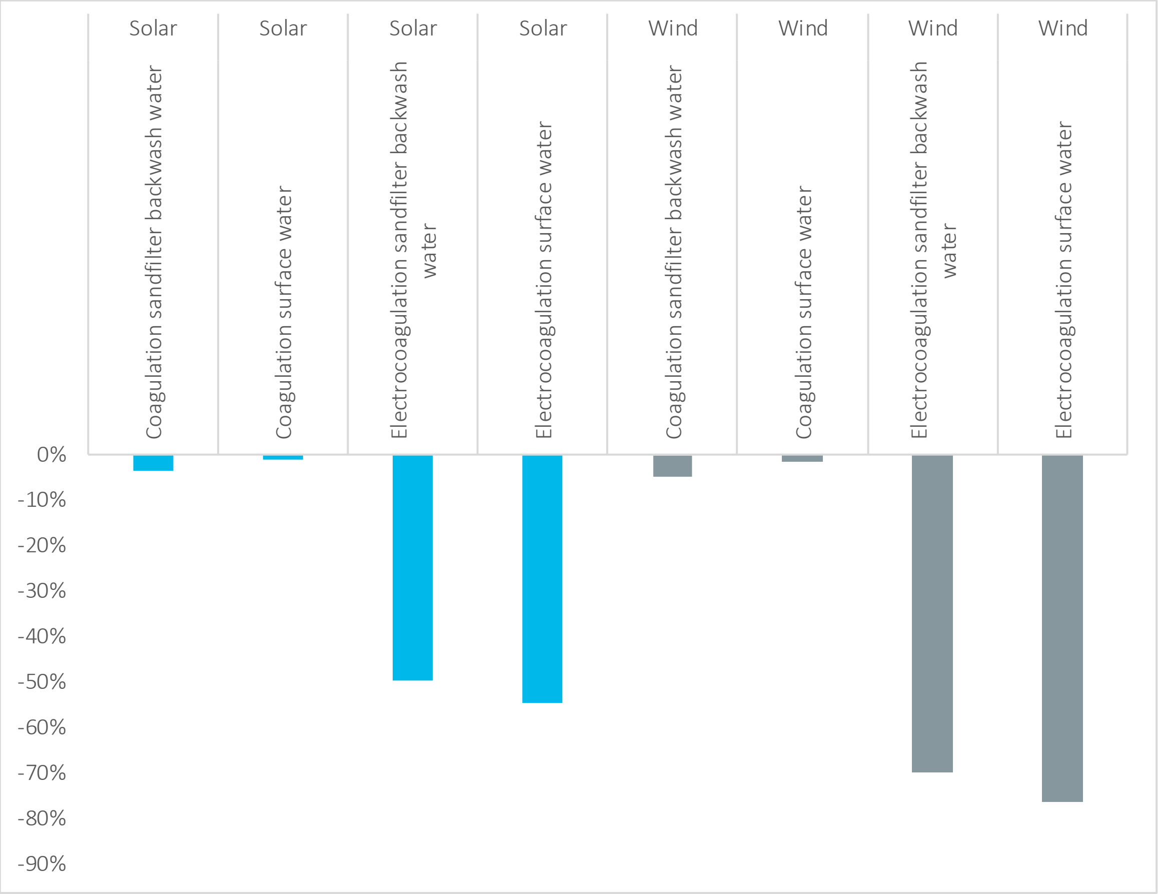


Figure 33 Percentage difference for each energy type relative to Netherlands' average consumption mix

CO2 footprint analysis:

Figure 34 CO_2 equivalent footprint for all 4 scenarios

As demonstrated in Figure 34, assessing the CO_2 footprint of coagulation and electrocoagulation for surface water treatment, the primary contributors to CO_2 emissions are the use of iron (III) chloride and electricity. For coagulation of surface water, the highest CO_2 emission is associated with iron (III) chloride (0.12 kg CO_2 eq), followed by transportation (0.018 kg CO_2 eq) and electricity consumption (0.0028 kg CO_2 eq). These emissions are relatively low due to the lesser energy requirements of traditional coagulation. In contrast, electrocoagulation for surface water shows a significantly higher CO_2 emission, predominantly driven by electricity consumption (0.32 kg

CO_2 eq), which is considerably higher than the traditional method. Additional significant contributors in electrocoagulation include the use of pig iron (0.013 kg CO_2 eq) and the transport emissions (0.02 kg CO_2 eq).

When treating sand filter backwash water, similar trends are observed. Coagulation for sand filter backwash water is primarily impacted by iron (III) chloride (0.036 kg CO_2 eq) and electricity consumption (0.0033 kg CO_2 eq). The total CO_2 emissions for coagulation remain lower due to the reduced chemical and energy requirements. Electrocoagulation, on the other hand, demonstrates higher emissions mainly due to electricity use (0.32 kg CO_2 eq) and other material inputs such as pig iron (0.021 kg CO_2 eq) and iron sinter (0.0065 kg CO_2 eq). These materials contribute significantly to the overall carbon footprint, highlighting the energy-intensive nature of electrocoagulation.

4.3 Concluding remarks

This chapter presents a comprehensive Life Cycle Assessment (LCA) comparing traditional coagulation and electrocoagulation methods for treating surface water and sand filter backwash water based on pilot trial results. The analysis was based on the ISO 14040/44:2006 framework and utilized SimaPro 9.2 software and the Ecoinvent database. It aimed to evaluate the environmental impacts of these treatment methods across various categories such as climate change, acidification, resource use, and human toxicity, while considering the influence of different energy sources on these impacts.
Key findings can be summarised below:

- Climate Change Impact: Electrocoagulation shows significantly higher impacts on climate change due to its substantial electricity consumption, especially when using the average Dutch electricity mix. This impact is notably reduced when renewable energy sources are considered.
- Resource Use: Traditional coagulation demonstrates higher impacts on mineral and metal resource use due to extensive chemical inputs, whereas electrocoagulation, though more energy-intensive, has lower impacts in this category.
- Acidification and Eutrophication: Conventional coagulation shows higher impacts in these categories, driven by the chemical inputs required for the process.
- Electricity Consumption: The reliance on high-voltage electricity in electrocoagulation significantly influences its overall environmental performance, emphasizing the importance of energy source in lifecycle assessments. The fact that the determined iron dosage for EC was substantially higher than for CC is reflected in the energy consumption. If EC would manage to get flocculation with the same iron dosage, then the impact would be substantially less.
- Sensitivity Analysis: Transitioning to renewable energy sources significantly reduces environmental impacts, particularly for electrocoagulation.

5 Conclusions and Recommendations

An integrated evaluation of electrocoagulation (EC) compared to conventional coagulation (CC) for treating surface water and sand filter backwash water was conducted through experimental pilot research. This research provided results that were translated into a concept design to estimate both CAPEX (capital expenditure) and OPEX (operational expenditure), as well as to perform a comprehensive Life Cycle Assessment (LCA). The conclusions drawn from this study are critical in understanding the viability and efficiency of EC compared to conventional $FeCl_3$ coagulation methods in the drinking water sector.

For Sand Filter Backwash Water, the pilot study determined that a current of 60 A was the best for treating sand filter backwash water. This current level was chosen due to its balance of low power consumption, low Fe dosage, and effective removal of targeted contaminants. At 60 A, with an Fe dosage of approximately 41 mg/L, removal efficiencies for key parameters were achieved: 80% for arsenic (As), 75% for total suspended solids (TSS), 86% for total phosphorus (Total P), 43% for total organic carbon (TOC), 56% for turbidity, and 48% for total color. Post an 8-hour settling period, the removal efficiencies improved, with turbidity removal between 64-93% and total color removal between 56-88%. Comparatively, conventional coagulation with $FeCl_3$ achieved turbidity and color removals of 78-85% and 63-86%, respectively. The removal of metals was generally favourable using EC. However, it was noted that concentrations of nickel (Ni), molybdenum (Mo), and iron (Fe) increased, likely due to the dissolution of the electrodes.
For surface water, the pilot results suggested a current of 120 A (84 mg Fe/L) was necessary based to meet the target removals. However, the surface water had already been pretreated with conventional coagulation (on average 20 mg Fe/L) which was a limitation of the pilot evaluation, and therefore not optimal results were obtained. For economic and life cycle assessments, a reduced current of 60 A was. We still need to understand why EC flocs were not successfully formed at similar iron dosages than in CC. Less energy (less Fe dose) in EC is crucial for a comparable effectiveness as in CC, and therefore, more research is needed to fine-tune the EC configuration (electrode gap, current density, contact time) and flocculation (mixing speed). The hypothesis is that EC flocs were not successfully formed at similar iron dosages than in CC, with this specific pilot design, because of a poor destabilisation mechanism of the contaminants.

The iron dosing via EC for groundwater demonstrated that dosages ranging from 0.7 mg/L to 8.6 mg/L could be achieved, but it was not successfully controlled. Better control of the current was achieved using a higher capacity inverter and running the experiment in constant Current mode instead of constant Voltage mode. However, uncertainty in iron dosage patterns was found, likely influenced by low electrical conductivity and sampling. Therefore, while EC cassette was able to dose low concentrations of iron, reproducibility remains uncertain, demanding further experiments to confirm consistent results. Reproducibility can be certain whenever the flow in between the plates is controllable into an EC reactor.

Within the chosen design and boundaries of the cost evaluation and life cycle assessment, EC incurs significantly higher investment costs than conventional $FeCl_3$ coagulation, and the main reason might be the significant differences in Fe dosage compared. For surface water treatment, EC requires an investment of 4707 k€, compared to 155 k€ for CC. For backwash water, the investment costs are 604 k€ for EC versus 48 k€ for CC. The operational costs for EC are also substantially higher. For surface water treatment, EC costs 2632 k€/year, which is 110% more than CC dosing. For backwash water, EC operating costs are 169 k€/year, 115% more than CC dosing. The findings indicated that the feasibility of EC is highly dependent on the Fe dosage and therefore, energy consumption and energy costs. Energy consumption and corresponding energy efficiency need to be reduced to values around 0.2 KWh/m3 and 5 KWh/kg Fe to become competitive with CC. If energy prices revert to historical values e.g. 0.09 €/kWh, the energy cost for only the EC unit in surface water treatment would be about 248 k€/year compared to the total OPEX of conventional coagulation (420 k€/year). Optimizing mainly Fe dosage and floc formation, plate lifetime, and energy consumption is essential for EC to be competitive with CC; since they also impact the waste management and chemical consumption costs, affecting the sustainability and operational efficiency of the treatment processes. This work demonstrated that comparing the technologies with different iron dosages (substantially higher for EC) will result in EC not being competitive for drinking water treatment. Future research should focus on achieving effective EC at similar iron concentrations than by adding $FeCl_3$.

The LCA indicated that EC has significantly higher impacts on climate change due to its substantial electricity consumption, especially when using the average Dutch electricity mix. However, this impact is notably reduced when renewable energy sources are considered. CC has higher impacts on mineral and metal resource use due to extensive chemical inputs. In contrast, EC, while more energy-intensive, has lower impacts in this category. CC shows higher impacts in Acidification and Eutrophication, driven by the chemical inputs required for the process. The reliance on electricity in EC significantly influences its overall environmental performance, highlighting the importance of the energy source and the need for an effective iron dosage similar to CC to become attractive. Transitioning to renewable energy sources significantly reduces environmental impacts, particularly for EC.

Based on the above conclusions drawn from the integrated evaluation of EC for treating surface water and sand filter backwash water, several recommendations can be made to enhance the viability and efficiency of EC:

- Optimization of applied current and Fe dosage: Further research should focus on optimizing the current and Fe dosage to balance energy consumption, removal efficiencies, and operational costs. For raw surface water, reducing the applied may help minimize energy use while maintaining acceptable removal efficiencies.
- Improvement in Design: Adopting a Monopolar Parallel connection could significantly reduce energy consumption compared to the current design with Bipolar Serial connections. This design change could enhance the overall cost-effectiveness of the EC process, extend the lifetime of the electrodes and reduce the frequency of replacements. Also, lower energy costs would make EC more competitive with conventional coagulation methods.
- Future studies should continue to perform comprehensive LCAs to evaluate the long-term environmental impacts of EC, particularly in comparison to conventional coagulation methods. This includes assessing the difference in water quality and sludge differences delivered by the two technologies.
- Conducting studies to understand how variations in water quality, such as changes in contaminant levels or seasonal fluctuations, affect the performance of EC and to develop strategies for dealing with these variations.
- Additionally, more knowledge exchange between technologists and engineers from the drinking water companies and EC technology suppliers can provide a better estimate of the investment costs, risks and environmental benefits of a drinking water EC installation.
- Points of attention include the energy efficiency (kWh/kg Fe) which was high about 15 , in which the feasibility of the technology relies on energy prices between 0.07 – 0.18 €/kWh.
- On-site pilot continuous experiments with longer durations that allow for EC configuration optimization (Fe dosage, energy consumption, electrode configuration) are expected to provide better insights into the application of EC in the drinking water sector and are recommended for further investigation, especially with the backwash water from rapid sand filters, in which EC seems more directly suitable for application.

Overall, the evaluation of electrocoagulation for surface water and sand filter backwash water treatment has provided valuable insights into the technology's capabilities and challenges to become competitive in drinking water treatment. While EC offers promising removal efficiencies for various contaminants and has potential environmental benefits, it also presents higher investment and operational costs and LCA impact compared to conventional CC at the chosen settings and pilot results. Only if optimization of current/ iron dosage and design configuration improvements are applied to reach comparable removals at a similar iron dosage concentration than CC the feasibility and competitiveness of EC will be significantly enhanced in drinking water treatment despite the dependency on energy costs.

References

Alimohammadi, M., Askari, M., Dehghani, M.H., Dalvand, A., Saeedi, R., Yetilmezsoy, K., Heibati, B. and McKay, G. (2017) Elimination of natural organic matter by electrocoagulation using bipolar and monopolar arrangements of iron and aluminum electrodes. International Journal of Environmental Science and Technology 14(10), 2125-2134. 10.1007/s13762-017-1402-3

Apshankar, K.R. and Goel, S. (2020) Nitrate removal from drinking water using direct current or solar powered electrocoagulation. SN Applied Sciences 2(2), 304. 10.1007/s42452-020-2069-9

Attour, A., Ben Grich, N., Mouldi Tlili, M., Ben Amor, M., Lapicque, F. and Leclerc, J.-P. (2016) Intensification of phosphate removal using electrocoagulation treatment by continuous pH adjustment and optimal electrode connection mode. Desalination and Water Treatment 57(28), 13255-13262. 10.1080/19443994.2015.1057537

Bakker, M. (2015). *Kostencalculator drinkwater: Handleiding* p. 29. Royal HaskoningDHV.

Barışçı, S. and Turkay, O. (2016) Domestic greywater treatment by electrocoagulation using hybrid electrode combinations. Journal of Water Process Engineering 10, 56-66. https://doi.org/10.1016/j.jwpe.2016.01.015

Bektaş, N., Akbulut, H., Inan, H. and Dimoglo, A. (2004) Removal of phosphate from aqueous solutions by electro-coagulation. Journal of Hazardous Materials 106(2), 101-105. https://doi.org/10.1016/j.jhazmat.2003.10.002

Ben-Sasson, M., Zidon, Y., Calvo, R. and Adin, A. (2013) Enhanced removal of natural organic matter by hybrid process of electrocoagulation and dead-end microfiltration. Chemical Engineering Journal 232, 338-345. https://doi.org/10.1016/j.cej.2013.07.101

Bicudo, B., van Halem, D., Trikannad, S.A., Ferrero, G. and Medema, G. (2021) Low voltage iron electrocoagulation as a tertiary treatment of municipal wastewater: removal of enteric pathogen indicators and antibiotic-resistant bacteria. Water Research 188, 116500. https://doi.org/10.1016/j.watres.2020.116500

Boudjema, N., Drouiche, N., Abdi, N., Grib, H., Lounici, H., Pauss, A. and Mameri, N. (2014) Treatment of Oued El Harrach river water by electrocoagulation noting the effect of the electric field on microorganisms. Journal of the Taiwan Institute of Chemical Engineers 45(4), 1564-1570. https://doi.org/10.1016/j.jtice.2013.10.006

Chellam, S. and Sari, M.A. (2016) Aluminum electrocoagulation as pretreatment during microfiltration of surface water containing NOM: A review of fouling, NOM, DBP, and virus control. Journal of Hazardous Materials 304, 490-501. https://doi.org/10.1016/j.jhazmat.2015.10.054

Zhu, Y. (2020). RE100 Green Electricity Certificate (GECS) of China. In *RE100 Green Electricity Certificate (GECs) of China*. https://www.there100.org/sites/re100/files/2020-10/Chinese%20GEC%20Paper_RE100_2020%20FINAL.pdf#:~:text=URL%3A%20https%3A%2F%2Fwww.there100.org%2Fsites%2Fre100%2Ffiles%2F2020

Commission, E. (2014). *Normalisation method and data for Environmental Footprints.* Retrieved from https://publications.jrc.ec.europa.eu/repository/handle/JRC91531

Commission, E. (2024). *Environmental footprint methods.* Retrieved from https://environment.ec.europa.eu/news/environmental-footprint-methods-2021-12-16_en

Delaire, C. (2016) Improving access to safe water in West Bengal, India: From Arsenic and bacteria removal to household behavior change, University of California, Berkeley.

Dubrawski, K.L., Fauvel, M. and Mohseni, M. (2013) Metal type and natural organic matter source for direct filtration electrocoagulation of drinking water. Journal of Hazardous Materials 244-245, 135-141. https://doi.org/10.1016/j.jhazmat.2012.11.027

Dubrawski, K.L. and Mohseni, M. (2013) Standardizing electrocoagulation reactor design: Iron electrodes for NOM removal. Chemosphere 91(1), 55-60. https://doi.org/10.1016/j.chemosphere.2012.11.075

EcoInvent. (2024). Ecoinvent database. Zürich, Switzerland.

EU Commission, 2023, EU Commission, Environmental Footprints Methods, Available at: 〈https://ec.europa.eu/environment/eussd/smgp/pdf/EF%20simple%20guide_v7_clen.pdf〉 (Accessed 01 March 2023).

Franco, D., Lee, J., Arbelaez, S., Cohen, N. and Kim, J.-Y. (2017) Removal of phosphate from surface and wastewater via electrocoagulation. Ecological Engineering 108, 589-596. https://doi.org/10.1016/j.ecoleng.2017.07.031

Ghernaout, D., Aichouni, M. and Touahmia, M. (2019) Mechanistic insight into disinfection by electrocoagulation — A review. Desalination and Water Treatment 141, 68-81

Ghernaout, D., Badis, A., Kellil, A. and Ghernaout, B. (2008) Application of electrocoagulation in Escherichia coli culture and two surface waters. Desalination 219(1), 118-125. https://doi.org/10.1016/j.desal.2007.05.010

Gu, Z., Liao, Z., Schulz, M., Davis, J.R., Baygents, J.C. and Farrell, J. (2009) Estimating Dosing Rates and Energy Consumption for Electrocoagulation Using Iron and Aluminum Electrodes. Industrial & Engineering Chemistry Research 48(6), 3112-3117. 10.1021/ie801086c

Heffron, J., McDermid, B., Maher, E., McNamara, P.J. and Mayer, B.K. (2019a) Mechanisms of virus mitigation and suitability of bacteriophages as surrogates in drinking water treatment by iron electrocoagulation. Water Research 163, 114877. https://doi.org/10.1016/j.watres.2019.114877

Heffron, J., Ryan, D.R. and Mayer, B.K. (2019b) Sequential electrocoagulation-electrooxidation for virus mitigation in drinking water. Water Research 160, 435-444. https://doi.org/10.1016/j.watres.2019.05.078

Holt, P.K., Barton, G.W., Wark, M. and Mitchell, C.A. (2002) A quantitative comparison between chemical dosing and electrocoagulation. Colloids and Surfaces A: Physicochemical and Engineering Aspects 211(2), 233-248. https://doi.org/10.1016/S0927-7757(02)00285-6

ISO. (2006a). ISO 14040:2006. Environmental management — life cycle assessment — principles and framework. Geneva, CH: International Organization for Standardization. Retrieved from https://www.iso.org/obp/ui/#iso:sTd:iSo:14040:eD-2:v1:en (Accessed on 14/06/2018). [Google Scholar]

ISO. (2006b). ISO 14044:2006. Environmental management – life cycle assessment – requirements and guidelines. Geneva, CH: International Organization for Standardization. Retrieved from https://www.iso.org/standard/38498.html

Jiang, J.-Q., Graham, N., André, C., Kelsall, G.H. and Brandon, N. (2002) Laboratory study of electro-coagulation–flotation for water treatment. Water Research 36(16), 4064-4078. https://doi.org/10.1016/S0043-1354(02)00118-5

Karamati Niaragh, E., Alavi Moghaddam, M.R. and Emamjomeh, M.M. (2017) Techno-economical evaluation of nitrate removal using continuous flow electro-coagulation process: optimization by Taguchi model. Water Supply 17(6), 1703-1711. 10.2166/ws.2017.073

Kumar, N.S. and Goel, S. (2010) Factors influencing arsenic and nitrate removal from drinking water in a continuous flow electrocoagulation (EC) process. Journal of Hazardous Materials 173(1), 528-533. https://doi.org/10.1016/j.jhazmat.2009.08.117

Lacasa, E., Cañizares, P., Sáez, C., Fernández, F.J. and Rodrigo, M.A. (2011a) Electrochemical phosphates removal using iron and aluminium electrodes. Chemical Engineering Journal 172(1), 137-143. https://doi.org/10.1016/j.cej.2011.05.080

Lacasa, E., Cañizares, P., Sáez, C., Fernández, F.J. and Rodrigo, M.A. (2011b) Removal of nitrates from groundwater by electrocoagulation. Chemical Engineering Journal 171(3), 1012-1017. https://doi.org/10.1016/j.cej.2011.04.053

Larue, O., & Vorobiev, E. (2003). Floc size estimation in iron induced electrocoagulation and coagulation using sedimentation data. International Journal of Mineral Processing, 71(1–4), 1–15. https://doi.org/10.1016/S0301-7516(03)00026-7

Llanos, J., Cotillas, S., Cañizares, P. and Rodrigo, M.A. (2017) Electrocoagulation as a key technique in the integrated urban water cycle – A case study in the centre of Spain. Urban Water Journal 14(6), 650-654. 10.1080/1573062X.2016.1223322

Mahvi, A.H., Malakootian, M. and Heidari, M.R. (2011) Comparison of polyaluminum silicate chloride and electrocoagulation process, in natural organic matter removal from surface water in Ghochan, Iran. Journal of Water Chemistry and Technology 33(6), 377-385. 10.3103/S1063455X11060051

Mameri, N., Lounici, H., Belhocine, D., Grib, H., Piron, D.L. and Yahiat, Y. (2001) Defluoridation of Sahara water by small plant electrocoagulation using bipolar aluminium electrodes. Separation and Purification Technology 24(1), 113-119. https://doi.org/10.1016/S1383-5866(00)00218-5

Mansour, S., Hasieb, H. and Khalaf, H. (2012) Remaval of Cobalt from drinking water by alternating current, Electrocoagulation technique. Journal of Applied Science. 10.3923/jas.2012.787.792

McBeath, S.T., Mohseni, M. and Wilkinson, D.P. (2020) Pilot-scale iron electrocoagulation treatment for natural organic matter removal. Environmental Technology 41(5), 577-585. 10.1080/09593330.2018.1505965

Moradi, M. and Ashrafizadeh, S.N. (2020) Nitrate removal from tapwater by means of electrocoagulation-flotation process. Separation Science and Technology 55(8), 1577-1587. 10.1080/01496395.2019.1602143

Roy, M., van Genuchten, C.M., Rietveld, L. and van Halem, D. (2021) Integrating biological As(III) oxidation with Fe(0) electrocoagulation for arsenic removal from groundwater. Water Research 188, 116531. https://doi.org/10.1016/j.watres.2020.116531

Ryan, D.R., Maher, E.K., Heffron, J., Mayer, B.K. and McNamara, P.J. (2021) Electrocoagulation-electrooxidation for mitigating trace organic compounds in model drinking water sources. Chemosphere 273, 129377. https://doi.org/10.1016/j.chemosphere.2020.129377

Semerjian, L., Damaj, A. & Salam, D. Comparative study of humic acid removal and floc characteristics by electrocoagulation and chemical coagulation. Environ Monit Assess 187, 670 (2015). https://doi.org/10.1007/s10661-015-4886-7

Sundman, A. (2014) Interactions between Fe and organic matter and their impact on As (V) and P (V), Umeå Universitet

Tanneru, C.T. and Chellam, S. (2012) Mechanisms of virus control during iron electrocoagulation – Microfiltration of surface water. Water Research 46(7), 2111-2120. https://doi.org/10.1016/j.watres.2012.01.032

Tanneru, C.T., Rimer, J.D. and Chellam, S. (2013) Sweep Flocculation and Adsorption of Viruses on Aluminum Flocs during Electrochemical Treatment Prior to Surface Water Microfiltration. Environmental Science & Technology 47(9), 4612-4618. 10.1021/es400291e

Ulu, F., Barışçı, S., Kobya, M., Särkkä, H. and Sillanpää, M. (2014) Removal of humic substances by electrocoagulation (EC) process and characterization of floc size growth mechanism under optimum conditions. Separation and Purification Technology 133, 246-253. https://doi.org/10.1016/j.seppur.2014.07.003

Ulu, F., Barışçı, S., Kobya, M. and Sillanpää, M. (2015) An evaluation on different origins of natural organic matters using various anodes by electrocoagulation. Chemosphere 125, 108-114. https://doi.org/10.1016/j.chemosphere.2014.11.063

Vasudevan, S., Sozhan, G., Ravichandran, S., Jayaraj, J., Lakshmi, J. and Sheela, M. (2008) Studies on the Removal of Phosphate from Drinking Water by Electrocoagulation Process. Industrial & Engineering Chemistry Research 47(6), 2018-2023. 10.1021/ie0714652

Vepsäläinen, M., Ghiasvand, M., Selin, J., Pienimaa, J., Repo, E., Pulliainen, M. and 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. https://doi.org/10.1016/j.seppur.2009.08.001

When 100% renewable energy doesn't mean zero carbon. (2019, May 23). Stanford Doerr School of Sustainability. https://sustainability.stanford.edu/news/when-100-renewable-energy-doesnt-mean-zero-carbon

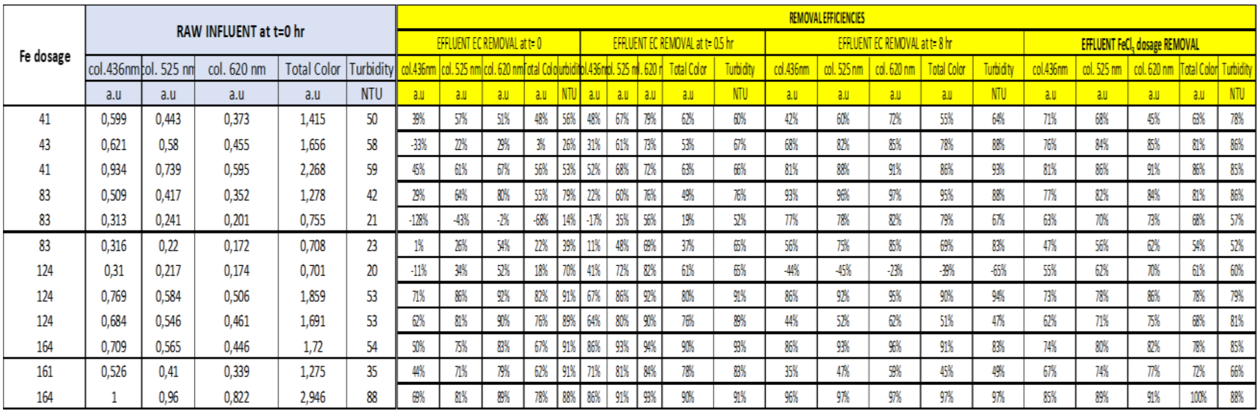
Zhu, B., Clifford, D.A. and Chellam, S. (2005) Comparison of electrocoagulation and chemical coagulation pretreatment for enhanced virus removal using microfiltration membranes. Water Research 39(13), 3098-3108. https://doi.org/10.1016/j.watres.2005.05.020

Zhu, J., Zhao, H. and Ni, J. (2007) Fluoride distribution in electrocoagulation defluoridation process. Separation and Purification Technology 56(2), 184-191. https://doi.org/10.1016/j.seppur.2007.01.030

Appendix I Pilot Research

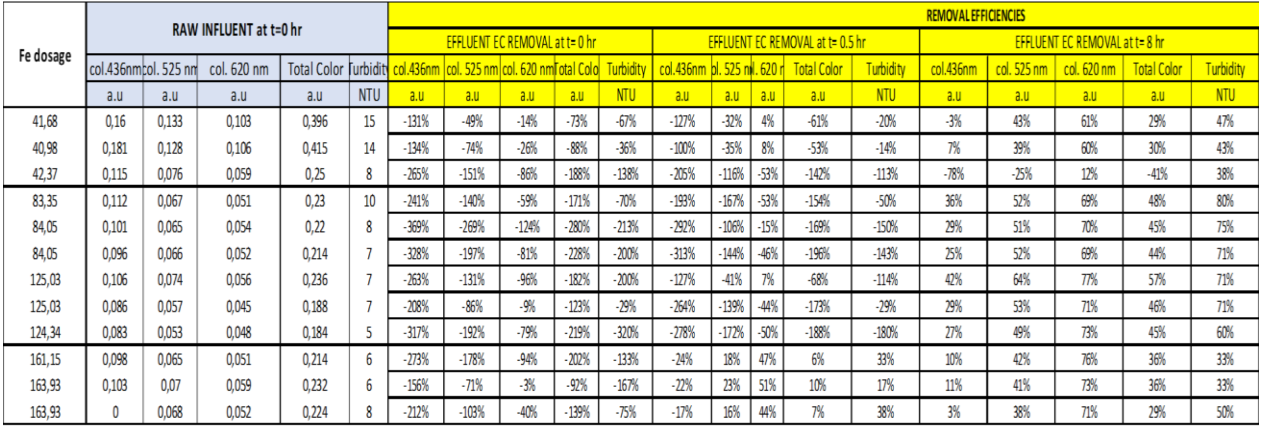
Removal Efficiencies

Table 13 Comparison of color and turbidity removal of EC after 0, 0.5 and 8 h of settling for sand filter backwash water, and the current conventional coagulation-flocculation with FeCl3.



Fe dosage	RAW INFLUENT at t=0 hr					REMOVAL EFFICIENCIES																			
						EFFLUENT EC REMOVAL at t= 0					EFFLUENT EC REMOVAL at t= 0.5 hr					EFFLUENT EC REMOVAL at t= 8 hr					EFFLUENT $FeCl_3$ dosage REMOVAL				
	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity	col.436nm	col. 525 nm	col. 620 nm	otal Colo	urbidit	ol.436n	ol. 525 n	. 620 r	Total Color	Turbidity	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity
	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU
41	0,599	0,443	0,373	1,415	50	39%	57%	51%	48%	56%	48%	67%	79%	62%	60%	42%	60%	72%	55%	64%	71%	68%	45%	63%	78%
43	0,621	0,58	0,455	1,656	58	-33%	22%	29%	3%	26%	31%	61%	73%	53%	67%	68%	82%	85%	78%	88%	76%	84%	85%	81%	86%
41	0,934	0,739	0,595	2,268	59	45%	61%	67%	56%	53%	52%	68%	72%	63%	66%	81%	88%	91%	86%	93%	81%	86%	91%	86%	85%
83	0,509	0,417	0,352	1,278	42	29%	64%	80%	55%	79%	22%	60%	76%	49%	76%	93%	96%	97%	95%	88%	77%	82%	84%	81%	86%
83	0,313	0,241	0,201	0,755	21	-128%	-43%	-2%	-68%	14%	-17%	35%	56%	19%	52%	77%	78%	82%	79%	67%	63%	70%	73%	68%	57%
83	0,316	0,22	0,172	0,708	23	1%	26%	54%	22%	39%	11%	48%	69%	37%	65%	56%	75%	85%	69%	83%	47%	56%	62%	54%	52%
124	0,31	0,217	0,174	0,701	20	-11%	34%	52%	18%	70%	41%	72%	82%	61%	65%	-44%	-45%	-23%	-39%	-65%	55%	62%	70%	61%	60%
124	0,769	0,584	0,506	1,859	53	71%	86%	92%	82%	91%	67%	86%	92%	80%	91%	86%	92%	95%	90%	94%	73%	78%	86%	78%	79%
124	0,684	0,546	0,461	1,691	53	62%	81%	90%	76%	89%	64%	80%	90%	76%	89%	44%	52%	62%	51%	47%	62%	71%	75%	68%	81%
164	0,709	0,565	0,446	1,72	54	50%	75%	83%	67%	91%	86%	93%	94%	90%	93%	86%	93%	96%	91%	83%	74%	80%	82%	78%	85%
161	0,526	0,41	0,339	1,275	35	44%	71%	79%	62%	91%	71%	81%	84%	78%	83%	35%	47%	59%	45%	49%	67%	74%	77%	72%	66%
164	1	0,96	0,822	2,946	88	69%	81%	89%	78%	88%	86%	91%	93%	90%	91%	96%	97%	97%	97%	97%	85%	89%	91%	100%	88%

Table 14 Color and turbidity removal of EC after 0, 0.5 and 8 h of settling for surface water.



Fe dosage	RAW INFLUENT at t=0 hr					REMOVAL EFFICIENCIES														
						EFFLUENT EC REMOVAL at t= 0 hr					EFFLUENT EC REMOVAL at t= 0.5 hr					EFFLUENT EC REMOVAL at t= 8 hr				
	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity	col.436nm	col. 525 nm	col. 620 nm	Total Colo	Turbidity	col.436nm	ol. 525 n	l. 620 r	Total Color	Turbidity	col.436nm	col. 525 nm	col. 620 nm	Total Color	Turbidity
	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU	a.u	a.u	a.u	a.u	NTU
41,68	0,16	0,133	0,103	0,396	15	-131%	-49%	-14%	-73%	-67%	-127%	-32%	4%	-61%	-20%	-3%	43%	61%	29%	47%
40,98	0,181	0,128	0,106	0,415	14	-134%	-74%	-26%	-88%	-36%	-100%	-35%	8%	-53%	-14%	7%	39%	60%	30%	43%
42,37	0,115	0,076	0,059	0,25	8	-265%	-151%	-86%	-188%	-138%	-205%	-116%	-53%	-142%	-113%	-78%	-25%	12%	-41%	38%
83,35	0,112	0,067	0,051	0,23	10	-241%	-140%	-59%	-171%	-70%	-193%	-167%	-53%	-154%	-50%	36%	52%	69%	48%	80%
84,05	0,101	0,065	0,054	0,22	8	-369%	-269%	-124%	-280%	-213%	-292%	-106%	-15%	-169%	-150%	29%	51%	70%	45%	75%
84,05	0,096	0,066	0,052	0,214	7	-328%	-197%	-81%	-228%	-200%	-313%	-144%	-46%	-196%	-143%	25%	52%	69%	44%	71%
125,03	0,106	0,074	0,056	0,236	7	-263%	-131%	-96%	-182%	-200%	-127%	-41%	7%	-68%	-114%	42%	64%	77%	57%	71%
125,03	0,086	0,057	0,045	0,188	7	-208%	-86%	-9%	-123%	-29%	-264%	-139%	-44%	-173%	-29%	29%	53%	71%	46%	71%
124,34	0,083	0,053	0,048	0,184	5	-317%	-192%	-79%	-219%	-320%	-278%	-172%	-50%	-188%	-180%	27%	49%	73%	45%	60%
161,15	0,098	0,065	0,051	0,214	6	-273%	-178%	-94%	-202%	-133%	-24%	18%	47%	6%	33%	10%	42%	76%	36%	33%
163,93	0,103	0,07	0,059	0,232	6	-156%	-71%	-3%	-92%	-167%	-22%	23%	51%	10%	17%	11%	41%	73%	36%	33%
163,93	0	0,068	0,052	0,224	8	-212%	-103%	-40%	-139%	-75%	-17%	16%	44%	7%	38%	3%	38%	71%	29%	50%

Variation at influent and effluent for pH, conductivity and temperature for Sand filtration backwash water Dunea

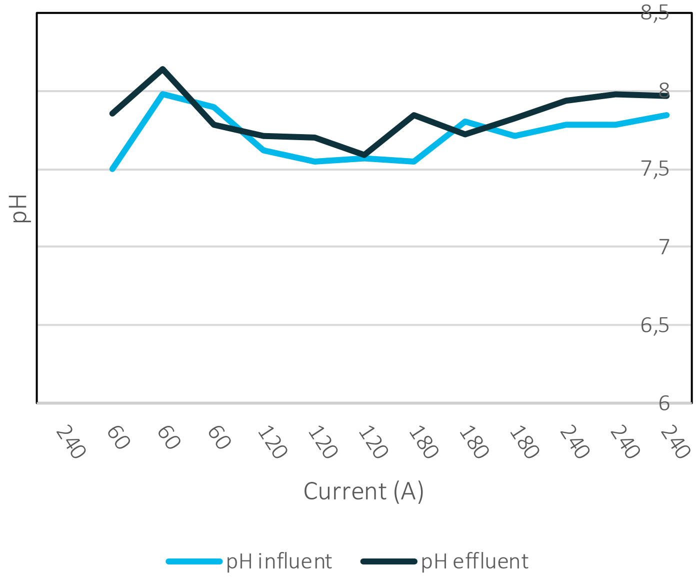


Figure 35 pH variation observed at the influent and effluent of the EC pilot.

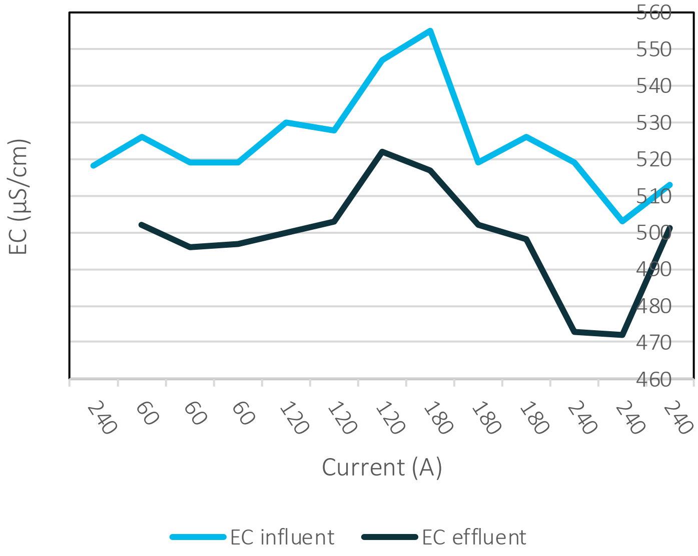


Figure 36 Conductivity variation observed at the influent and effluent of the EC pilot.

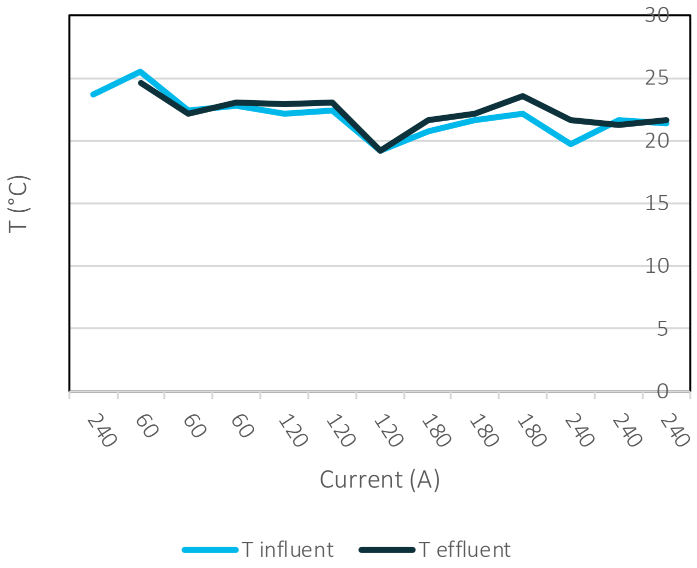


Figure 37 Temperature variation observed at the influent and effluent of the EC pilot.

Variation at influent and effluent for pH, conductivity, and temperature for surface water Dunea

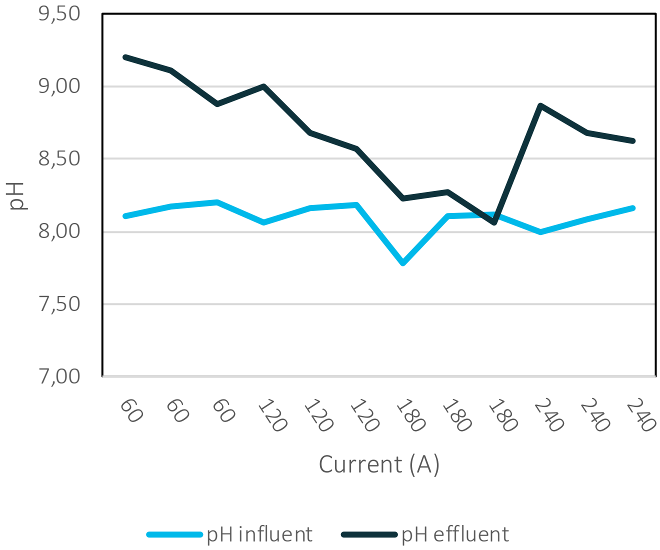


Figure 38 pH variation observed at the influent and effluent of the EC pilot.

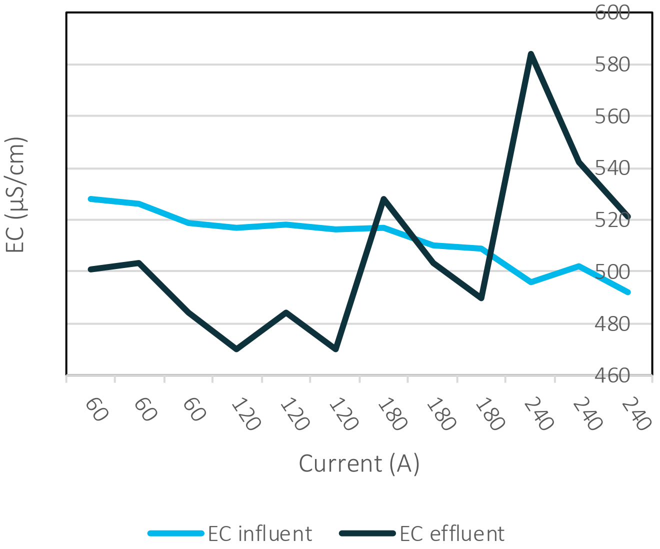


Figure 39 Conductivity variation observed at the influent and effluent of the EC pilot.

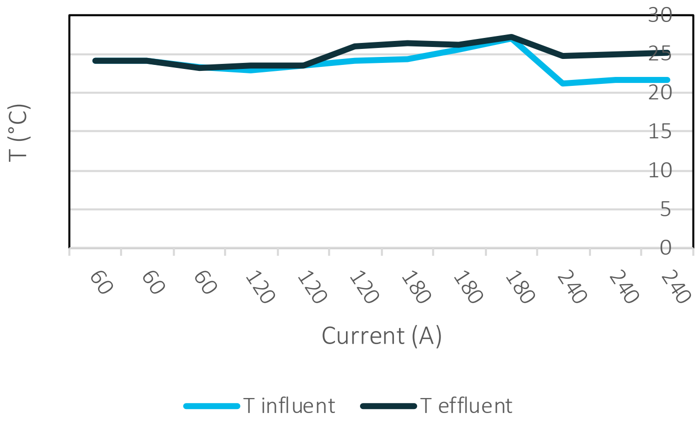


Figure 40 Temperature variation observed at the influent and effluent of the EC pilot.

Appendix II Cost estimation basis

1. Flocculation Unit

Flocculation is achieved by dosing a coagulant (iron salt), possibly with flocculants. After the (rapid) mixing of raw water and flocculants, the growth of the flocs takes place in the flocculation chamber. Energy is introduced into these flocculation chambers using agitators. Compartmentalized flocculation spaces are often used.
The construction costs for flocculation are mainly determined by the residence time in the flocculation space(s). In practice, flocs formation times of 10 - 30 minutes are used. The contents of the flocculation space then follow from the design flow and the flocculation time. The content is used as a cost-determining parameter in the derived construction cost function.

Investment parameters

Residence time: 20 min
Reserve position: 0 %
Process Auto: 20 I/O
Existing capacity: 0 %
Share CTB costs: 40 %
Share WTB costs: 40 %
Share of E&I costs: 20 %

Operation Parameters

Energy consumption: 20 Wh/m³

2. Lamella sedimentation Unit

Lamella sedimentation is applied in practice in surface water treatment and backwash water treatment. In the preparation of drinking water from surface water, sedimentation is applied after flocculation. The purpose of sedimentation is to remove floc, sand and clay. Sedimentation is based on the differences in density between these suspended solids and the water. In practice, the efficiency of settling is found to be independent of the height of the settling tank. It is therefore possible to install plates (lamella) in the settling tank, providing more settling surface. This can save a factor of 10 - 20 in building volume.

The lamella plates are placed at an angle (55- 60°), so that the settled material sinks to the bottom. The settled sludge is collected in sludge cones under the plates. A natural thickening takes place in the sludge cones, sometimes promoted by reamers. From the sludge cones, the sludge is removed to sludge processing. Often the sludge is disposed of periodically using sludge pumps.

The typical surface load is 0.8 - 1.0 m3/h per m2 of settling surface.
The settling surface area is used as a cost-dependent parameter.

Investment parameters

Surface load: 0.9 m3/h/m2
Reserve position: 0 %
Process Auto: 30 I/O

Loss: 3%
Existing capacity: 0 %
Share CTB costs: 50 %
Share WTB costs: 40 %
Share of E&I costs: 10 %

Operation Parameters

Energy consumption: 5 Wh/m³

3. FeCl3 dosing and storage

Investment parameters

Storage capacity: 12 days
Strategic reserve: 15%
Reserve pump: 100 %
Process Auto: 10 I/O
Existing capacity: 0 %
Share CTB costs: 10 %
Share WTB costs: 70 %
Share of E&I costs: 20 %

Operation Parameters

Energy consumption: 3 Wh/m³
FeCl3 (100%): 58.08 gr/m3 (Surface Water), 17.44 gr/m3 (sand filter Backwash water).

Table 15 Input data provided by Dunea for investment cost calculations and comparison with EC.

	Value	Unit
Algemeen		
Kengetal PA bouwkosten (alleen software/ uren, hardware in post E&I)	0	Eur per I/O
Percentages bijkomende kosten (toeslagen op bouwkosten)		
A- Directe BK- Alg. voorzieningen/ N.T.D.	5	% bouwk. C/ W / E / PA (bv kantoren/geb. install./ tijd.maatregel)
B- Directe BK- Inrichtingskosten	2	% bouwk. C/ W / E / PA
C- Directe BK-Beveiligingskosten	1	% bouwk. C/ W / E / PA
D- Eng. OG+IB zuiveringsprojecten	30	% bouwk. C / W / E / PA +A+B+C
E- Eng. OG+IB leidingprojecten	20	% bouwk. C / W / E / PA +A+B+C
F- Overige bijkomende kosten - leges etc	1	% bouwk. C / W / E / PA +A+B+C
G- Totale bouwrente	4,5	% bouwk. C / W / E / PA +A+B+C+(D of E)+F

H- Risico scope wijz. / marktwerking	0	% bouwk. C / W / E / PA +A+B+C+(D of E)+F+G
Indexering investeringskengetallen		
Indexeren tot jaar	2024	
Inflatie per jaar tot indexatiejaar	3	procent

Table 16 Input data provided by Dunea for operational cost calculations and comparison with EC.

	Value	Unit
Fixed costs depreciation periods		
Rente op basis van annuïteit	3,2	%
Levensduur Winning	33,3	jaar
Levensduur Civiele Techniek / Bouwkunde	33,3	jaar
Levensduur Werktuigbouwkunde	20	jaar
Levensduur Elektrotechniek	20	jaar
Levensduur PA	10	jaar
Levensduur Transportleidingen	40	jaar
Levensduur Membranen UF / HF	8	jaar
Levensduur Membranen NF / HF	5	jaar
Levensduur Membranen RO zeewater	4	jaar
Vervanging aktief kool (nieuw)	2	jaar
Vervanging aktief kool (regeneratie)	1,5	jaar
Levensduur overige (filterzand etc.)	10	Jaar
Consumption costs: energy		
Energiekosten (transport/ levering/ belasting)	0,18	€/kWh
Percentage groene stroom	100	%
Fictieve kosten per ton CO2	300	€/ton CO2
Consumption costs: chemicals		
NaOH (100%)	0,225	Eur /kg
FeCl3 6H2O (100%)	0,54	Eur /kg
CO2 (100%)	0,091	Eur /kg
PE (100%)	2,45	Eur /kg
Consumption costs: consumables		
Granaatzand ontharding	380	Eur/ton
Antraciet SF	282	Eur/m3
Zand SF	120	Eur/m3
Zand LZF	120	Eur/m3
Consumption cost: disposal cost		
Drinkwaterslib gewoon	150	Eur/ton ds
Drinkwaterslib chemisch	300	Eur/ton ds

Lozingskosten riool	0,03	Eur/m3
Onderhoudskosten		
Civiele Techniek / Bouwkunde	0,5	% van bouwkosten CT / B
Werktuigbouwkunde	2	% van bouwkosten WTB
Elektrotechniek	4	% van bouwkosten E
Proces Automatisering	6	% van bouwkosten PA
Onderhoud inrichting /algemene voorzieningen	6	% van bouwkosten inrichting /alg. voorzieningen
Specific operating expenses		
Personeelskosten (bediening/ onderhoud/optimalseren)	75000	Euro per manjaar
Analysekosten	2,8	Euro per KIWA-punt
Kosten beveiliging	15	% van bouwkosten beveiliging
Administrative management costs		
Administratieve beheerskosten	20	% van specifieke bedrijfskosten

Appendix III Conceptual design basis EC

Process parameters of Skid based design provided by Qstone.

Q-60 unit

General Specification

Design flow rate: **60 m3/h**

Maximum flow rate: **180 m3/h**

Total shipping weight: **3.5 ton**

Operational weight: **10 ton**

Footprint dimensions: **7000 X 7000 mm**

Chamber Specification

Number of chambers: **6**

Chamber dimensions (L x W x H):

1448 x 860 x 1217 mm

Chamber material: **Fibre Reinforced Plastic (FRP)**

Skid Specification

Number of skids: **3**

Skid dimensions (L x W x H): **2971 x 1898 x 1768 mm**

Skid material: **Mild Steel**

Electrodes Specification (per chamber)

Total number of plates: **145**

Number of power electrodes: **2**

Plates dimensions (L x W x H):

800 x 400 x 3 mm

Plates material: **Mild Steel (C37)**

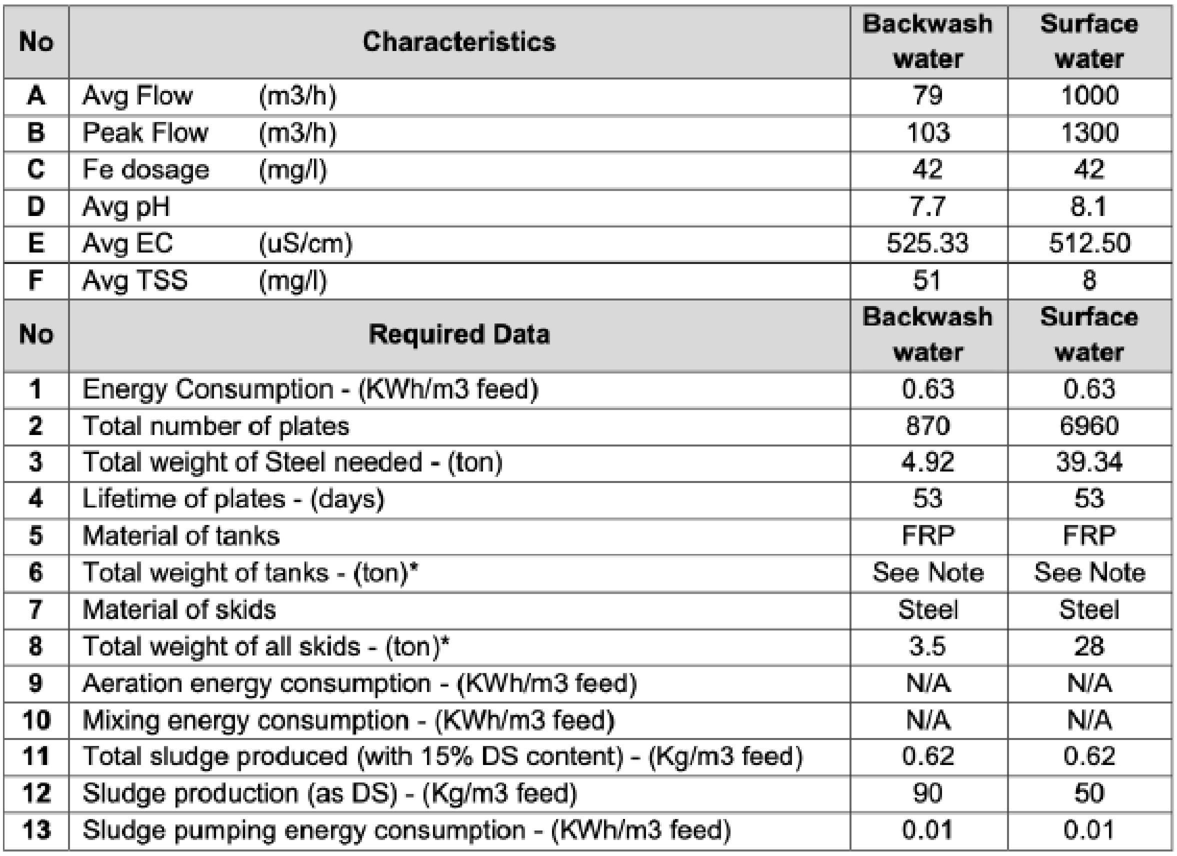
Electrical Panel Specification

Panel capacity: **500 A**

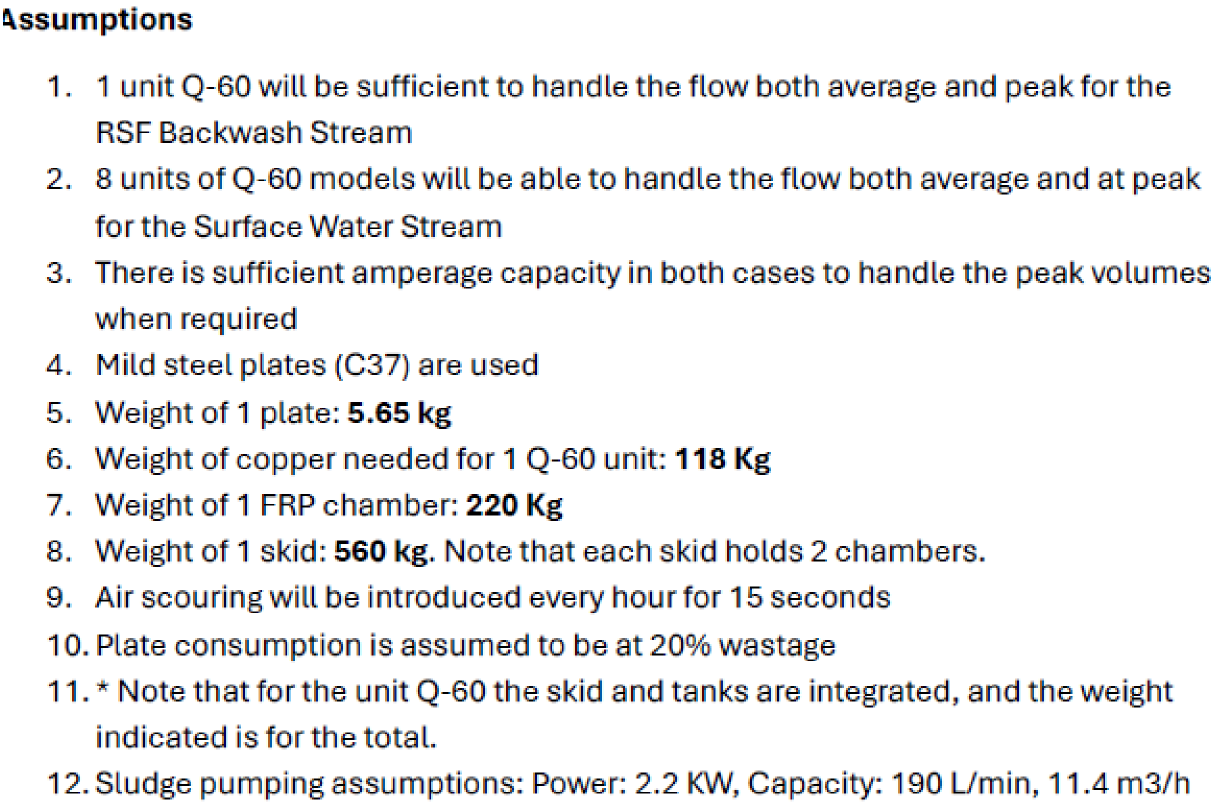
Power requirements: **380-400 V | 50 Hz | 3 Phase**

Panel dimensions (L x W x H): **1350 x 620 x 2080 mm**

Panel casing IP code: **IP55**



No	Characteristics	Backwash water	Surface water
A	Avg Flow (m3/h)	79	1000
B	Peak Flow (m3/h)	103	1300
C	Fe dosage (mg/l)	42	42
D	Avg pH	7.7	8.1
E	Avg EC (uS/cm)	525.33	512.50
F	Avg TSS (mg/l)	51	8
No	**Required Data**	**Backwash water**	**Surface water**
1	Energy Consumption - (KWh/m3 feed)	0.63	0.63
2	Total number of plates	870	6960
3	Total weight of Steel needed - (ton)	4.92	39.34
4	Lifetime of plates - (days)	53	53
5	Material of tanks	FRP	FRP
6	Total weight of tanks - (ton)*	See Note	See Note
7	Material of skids	Steel	Steel
8	Total weight of all skids - (ton)*	3.5	28
9	Aeration energy consumption - (KWh/m3 feed)	N/A	N/A
10	Mixing energy consumption - (KWh/m3 feed)	N/A	N/A
11	Total sludge produced (with 15% DS content) - (Kg/m3 feed)	0.62	0.62
12	Sludge production (as DS) - (Kg/m3 feed)	90	50
13	Sludge pumping energy consumption - (KWh/m3 feed)	0.01	0.01



Assumptions

1. 1 unit Q-60 will be sufficient to handle the flow both average and peak for the RSF Backwash Stream
2. 8 units of Q-60 models will be able to handle the flow both average and at peak for the Surface Water Stream
3. There is sufficient amperage capacity in both cases to handle the peak volumes when required
4. Mild steel plates (C37) are used
5. Weight of 1 plate: **5.65 kg**
6. Weight of copper needed for 1 Q-60 unit: **118 Kg**
7. Weight of 1 FRP chamber: **220 Kg**
8. Weight of 1 skid: **560 kg.** Note that each skid holds 2 chambers.
9. Air scouring will be introduced every hour for 15 seconds
10. Plate consumption is assumed to be at 20% wastage
11. * Note that for the unit Q-60 the skid and tanks are integrated, and the weight indicated is for the total.
12. Sludge pumping assumptions: Power: 2.2 KW, Capacity: 190 L/min, 11.4 m3/h

Connection of Iron plates

Below the different ways of connecting the Fe plates are depicted, the EC container-based design adopts the Monopolar Parallel while the skid-based design (chosen) use the Bipolar Serial:

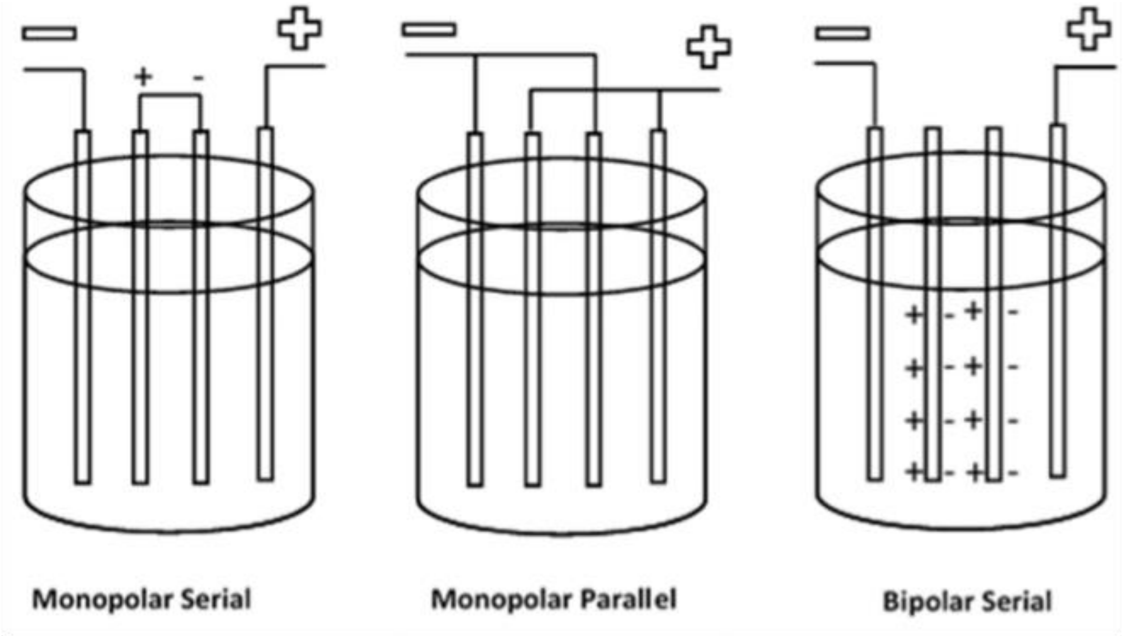


Figure 41 Electrode configurations

Electrode Configurations

1. Monopolar-Serial (MS)

- Each electrode pair (anode and cathode) is connected in series.
- The current flows through one electrode pair at a time.
- This setup typically has a lower overall resistance because the current path is longer.

2. Monopolar-Parallel (MP)

- Anodes and cathodes are connected in parallel.
- The current is divided among all electrode pairs.
- This setup allows for a uniform current distribution, reducing localized resistance and potential drops.

3. Bipolar-Serial (BS)

- No electrical connection between inner electrodes.
- The outermost electrodes are connected to the power supply, while the inner electrodes act as both anodes and cathodes.
- The current flows through multiple electrode pairs in series, but because there are no direct connections in the middle, each pair has its own potential drop.

Reasons for Increased Energy Consumption in Bipolar-Serial Configuration:

1. Multiple Voltage Drops

- In the Bipolar-Serial setup, each electrode pair introduces a potential drop.
- The total voltage required to drive the current through all pairs is the sum of the individual voltage drops across each pair.
- This results in a higher overall voltage requirement, increasing energy consumption.

2. Internal Resistance

- The internal resistance of each electrode pair adds up in series.
- Higher resistance in the circuit increases the power required to maintain the same current level.
- Power (P) is given by $P = V\times I$ where V is voltage and I is current. With increased resistance, more power is needed to overcome it.

3. Electrode Surface Area

- The effective electrode surface area in the Bipolar-Serial configuration might be less optimized compared to parallel configurations.
- Less surface area can increase resistance and decrease the efficiency of electrochemical reactions, thus requiring more energy.

4. Heat Generation

- Higher resistance also leads to more heat generation within the system.
- Heat loss means more energy input is needed to achieve the same process efficiency.

Appendix IV Capex Cost Distribution

Table 17 CAPEX iron electrocoagulation for surface water

CAPEX COST	Flocculation	Lamella Settling	Electrocoagulation	TOTAL COSTS
Construction costs	€ 2.553.680	€ 3.999.526	€ 3.106.649*	**€ 9.659.854**
Civil costs	€ 1.021.472	€ 1.999.763	€ 621.330*	**€ 3.642.565**
Mechanical costs	€ 1.021.472	€ 1.599.810	€ 1.553.325*	**€ 4.174.607**
Electrical costs	€ 510.736	€ 399.953	€ 931.995*	**€ 1.842.683**
Proces autom.	€ 75.511	€ 118.263	€ 106.944*	**€ 300.718**
General facilities	€ 127.684	€ 199.976	€ 156.086*	**€ 483.746**
General building equipment costs	€ 51.074	€ 79.991	€ 62.415*	**€ 193.480**
Protection costs	€ 25.537	€ 39.995	€ 31.208*	**€ 96.740**
Design / supervision treatment	€ 827.392	€ 1.295.846	€ 1.011.308*	**€ 3.134.547**
Miscellaneous additional costs	€ 27.580	€ 43.195	€ 33.702*	**€ 104.477**
Construction interest costs	€ 162.583	€ 254.634	€ 198.731*	**€ 615.948**
TOTAL CAPEX COSTS	€ 3.851.000	€ 6.031.000	€ 4.707.000	**€ 14.589.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 15%	+/- 30%

Table 18 CAPEX iron chloride dosage for surface water

CAPEX COST	Flocculation	Lamella Settling	FeCl3 dosing	TOTAL COSTS
Construction costs	€ 2.553.680	€ 3.999.526	€ 102.757	**€ 6.655.963**
Civil costs	€ 1.021.472	€ 1.999.763	€ 10.276	**€ 3.031.510**
Mechanical costs	€ 1.021.472	€ 1.599.810	€ 71.930	**€ 2.693.212**
Electrical costs	€ 510.736	€ 399.953	€ 20.551	**€ 931.240**
Proces autom.	€ 75.511	€ 118.263	€ 3.038	**€ 196.812**
General facilities	€ 127.684	€ 199.976	€ 5.138	**€ 332.798**
General building equipment costs	€ 51.074	€ 79.991	€ 2.055	**€ 133.119**
Protection costs	€ 25.537	€ 39.995	€ 1.028	**€ 66.560**
Design / supervision treatment	€ 827.392	€ 1.295.846	€ 33.293	**€ 2.156.532**
Miscellaneous additional costs	€ 27.580	€ 43.195	€ 1.110	**€ 71.884**
Construction interest costs	€ 162.583	€ 254.634	€ 6.679	**€ 423.895**
TOTAL CAPEX COSTS	€ 3.851.000	€ 6.031.000	€ 155.000	**€ 10.037.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 40%	+/- 40%

Table 19 CAPEX iron electrocoagulation for sand filtration backwash water

CAPEX COST	Flocculation	Lamella Settling	Electrocoagulation	TOTAL COSTS
Construction costs	€ 260.034	€ 553.465	€ 398.607*	**€ 1.212.106**
Civil costs	€ 104.014	€ 276.732	€ 79.721*	**€ 460.467**
Mechanical costs	€ 104.014	€ 221.386	€ 199.304*	**€ 524.703**
Electrical costs	€ 52.007	€ 55.346	€ 119.582*	**€ 226.935**
Proces autom.	€ 7.689	€ 16.366	€ 13.722*	**€ 37.776**
General facilities	€ 13.002	€ 27.673	€ 19.930*	**€ 60.605**
General building equipment costs	€ 5.201	€ 11.069	€ 7.972*	**€ 24.242**
Protection costs	€ 2.600	€ 5.535	€ 3.986*	**€ 12.121**
Design / supervision treatment	€ 84.251	€ 179.322	€ 129.149*	**€ 392.722**
Miscellaneous additional costs	€ 2.808	€ 5.977	€ 4.305*	**€ 13.091**
Construction interest costs	€ 16.555	€ 35.237	€ 25.995*	**€ 77.787**
TOTAL CAPEX COSTS	€ 392.000	€ 835.000	€ 604.000	**€ 1.831.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 15%	+/- 30%

Table 20 CAPEX iron chloride dosage for sand filtration backwash water

CAPEX COST	Flocculation	Lamella Settling	FeCl3 dosing	TOTAL COSTS
Construction costs	€ 260.034	€ 553.465	€ 32.118	**€ 845.617**
Civil costs	€ 104.014	€ 276.732	€ 3.212	**€ 383.958**
Mechanical costs	€ 104.014	€ 221.386	€ 22.483	**€ 347.882**
Electrical costs	€ 52.007	€ 55.346	€ 6.424	**€ 113.777**
Proces autom.	€ 7.689	€ 16.366	€ 950	**€ 25.004**
General facilities	€ 13.002	€ 27.673	€ 1.606	**€ 42.281**
General building equipment costs	€ 5.201	€ 11.069	€ 642	**€ 16.912**
Protection costs	€ 2.600	€ 5.535	€ 321	**€ 8.456**
Design / supervision treatment	€ 84.251	€ 179.322	€ 10.406	**€ 273.980**
Miscellaneous additional costs	€ 2.808	€ 5.977	€ 347	**€ 9.133**
Construction interest costs	€ 16.555	€ 35.237	€ 2.088	**€ 53.880**
TOTAL CAPEX COSTS	€ 392.000	€ 835.000	€ 48.000	**€ 1.275.000**
Bandwidth Calculation	+/- 40%	+/- 40%	+/- 40%	+/- 40%