Proof of concept design of a black water disinfection system for on board use on recreational vessels and yachts

by

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Abstract

There are more than 160,000 recreational crafts in the Netherlands with a toilet on board. It is forbidden to discharge blackwater into the environment from these crafts. Stricter enforcement of the law by the sealing of the valves of the external drain, leaves the owners of these crafts with a major problem. It is allowed, however, to treat the blackwater on board and then discharge it if the following two boundary conditions for fecal indicators are met: a maximum concentration of 330 cfu per 100 ml of *Intestinal enterococci* and a maximum concentration of 900 cfu per 100 ml of *E. coli*. The overall objective of this research was to find and test a proof-of-concept (PoC) method to reach these boundary conditions. The method consisted of a series of conventional wastewater techniques: first a coagulation and flocculation step, then the filtration and subsequent dewatering of the TSS and lastly the filtrate was disinfected using UV-C irradiation. Bench scale experimental setups were used to get the data for this research. Results demonstrated that TSS was most efficiently removed by a combination of polyaluminnium chloride and a cationic polyacrylamide copolymer with a low charge density. Furthermore, polyaluminium chloride demonstrated to be highly effective at removing the fecal indicators. Dewatering experiments proved that increasing the polymer dosage did not affect the total dry solid content of the final sludge cake. Also, under the most ideal circumstances a dry solid content of the filter cake of 30% was achieved. Lastly, UV-C experiments were conducted on the filtrates of two different samples, with values for UV_{254} absorbance of 1.34 cm^{-1} and 11.85 cm^{-1} respectively. The first of which was sufficiently disinfected after a total irradiated UV-C dose of 180 mJ/cm^2, of which the effective dose was 54.9 mJ/cm^2. The fecal indicators in the second sample were not sufficiently removed after a total UV-C dose of 1100 mJ/cm^2. The absorbance of UV_{254} of the filtrate was so large that the effective dose was only 40.0 mJ/cm^2. The PoC method showed to have potential for implementation, but adjustments need to be made to make it more sufficient and viable for use of on board disinfection of blackwater. Preliminary screening can extract most of the fecal material, which is already high in dry solid content, before homogenising with the liquid fraction. The quantity of conditioning agents need to be adapted to the specific blackwater, and if necessary the pH will need to be adjusted to ensure the right coagulation mechanism for efficient TSS removal. Finally, humic matter in the filtrate will have to be reduced for UV-disinfection to be an effective treatment method. This could possibly be achieved by mechanisms such as adsorptive coagulation or by implementing an intermediate ozonation step after filtration of the TSS, which would also help in the removal of fecal indicators.

Contents

Acronyms

APS Advanced Protective Systems

BAL Besluit Activiteiten Leefomgeving

BLBI Besluit lozen buiten inrichtingen

BOD Biological oxygen demand

BW Blackwater

COD Chemical oxygen demand

DBP Disinfection by-products

EPA Environmental Protection Agency

EPS Extracellular polymeric substances

IMO International Maritime Organization

MSD Marine sanitation devices

NOM Natural organic matter

PoC Proof of concept

RLBI Regeling lozen buiten inrichtingen

SN Supernatant

TOC Total organic carbon

TS Total solids

TSS Total suspended solids

UV Ultraviolet

VSS Volatile suspended solids

WWTP Waste water treatment plant

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1. Introduction

1.1 Problem statement

There are around 200,000 recreational crafts in the Netherlands, from which approximately 162,000 have a toilet on board (Waterrecreatie Advies, 2005). Since January 2009 it is prohibited to discharge the toilet water, or black water, into the environment. Subsequently, on the 16th of March in 2011 the decree "Besluit lozen buiten inrichtingen" (BLBI) entered into force. This decree layed down rules for categories of discharges into the environment resulting from activities outside establishments. In the earlier years of the legislation, article 3.9 stated that the discharge of any toilet water from recreational crafts into surface waters was not permitted any more. This part of the BLBI was established due to health risks associated with pathogens from the toiletwater that is discharged in the recreational waters. Recreational crafts were required to have a storage tank for the toiletwater on board, and the government designated specific facilities for the discharge of this toiletwater. According to the ANWB Water Almanac part two (ANWB, 2020) there are 493 of these facilities in the Netherlands, most of which are situated near populated marinas. When the storage tank is full and needs to be emptied, the vessel will have to travel to one of the designated facilities and the black water subsequently can be pumped out. People who still discharge their wastewater do not only harm the environment and the water quality in the area of the berths, but they also risk a fine of 160 Euros. However, it is very hard to enforce this law on violators and till this day hardly any fines have been issued. This is due to the fact that in most ships, the discharge valves are located under the water level, which makes it very hard to catch offenders in the act of violating the law (Lamme, 2012).

In the beginning of October 2019 the Netherlands Government Gazette published an adaption of the BLBI. Article 3.9 now states that it is permitted to discharge toilet water from a recreational craft, if the toilet water, prior to discharge, is guided through a treatment system which complies with the requirements set by ministerial regulations. These requirements will be discussed in section 1.2. The addition to the BLBI gives an alternative to the storage tank that needs to be emptied at the approved facilities, which is a welcome change according to boat owners. A research conducted by the HISWA (HISWA, 2019) revealed that for many boat owners a treatment system on board is preferred over a storage tank.

On top of the above-mentioned cases, the environment law, which in Dutch is called "Besluit Activiteiten Leefomgeving" (BAL), is being adapted. Currently, the ministry of infrastructure and water management is working on a new legislation in the BAL that obliges owners of a boat to seal the shut-off valves of the external drain, making it virtually impossible to discharge toileltwater from a boat. However, at the time of writing, there are no systems that have been certified to treat toilet water and discharge it on Dutch waters. This is why the adaption in the BAL caused great opposition from the pleasure craft sector, and a transition period of five years was intercalated before the shut-off valvles will be sealed of.

This makes for a good market opportunity for entrepreneurs and water technology companies. Especially because the Netherlands is the first country of the European Union to introduce legislation's that allow for the treatment of blackwater on board, which means that other countries may follow in the future.

1.2 Boundary conditions

The discharge of untreated toilet water into recreational waters is particularly dangerous for swimmers, as swimming activities mostly occur close to berth places of these crafts. For this reason, the discharge requirements in the Netherlands are based on the microbiological standards of the bathing water directive and not on requirements set by the European law for inland vessels. The European law focuses on the preservation of the environment, which is why in this law the requirements of the water concern BOD, COD and TOC. The concentration of COD is a common measure for the concentration of organic pollutants present in the water (Hu & Grasso, 2019), which would give an indication for the potential impact it can have to the environment (oxygen depletion of surface waters). By solely adopting the requirements from the European law, there would be no guarantee for the necessary protection to public health, because there would be still too many fecal indicators in the discharge of the crafts. It would also create major barriers for the development of small and compact treatment systems, because effective COD treatment generally makes use of large bioreactors, which is not a feasible option for a small craft (Ministerie van Infrastructuur en Waterstaat, 2019).

The "Regeling lozen buiten inrichtingen" (RLBI) expands on some of the articles from the BLBI. Article 2.28 states the discharge requirements to be met by the system, which are only based on the concentration of two fecal indicators: *Escherichia coli*, from here on called *E. coli*, and *Intestinal enterococci*. Fecal indicators are used as intermediates to detect waterborne pathogens at limited cost (Rochelle et al., 2015). Identification and quantification of all pathogenic bacteria in a water body is impractical and time consuming. *E. coli* and *Intestinal enterococci* are commonly used as indicators of the water quality, when high values are present it is assumed that pathogenic bacteria are also present. To be allowed to discharge toilet water from a recreational craft, the permitted concentrations of fecal indicators in the toiletwater are:

I. A maximum concentration of 330 cfu per 100 mL of *Intestinal enterococci*

II. A maximum concentration of 900 cfu per 100 mL of *E. coli*

It is described in the RLBI that, to be accepted and certified, the treatment system needs to pass a test that is given by an organisation proposed by the government. The maximum values above are set as a 90-percentile, which means that at least 90% of the collected data has values equal or below these maximums. In the test the influent of the system will consist of parts toilet paper, faeces, urine and drinking water in a weight ratio of respectively 1, 3, 25 and 100 (Ministerie van Infrastructuur en Waterstaat, 2019). These ratios of course can fluctuate greatly per person, country and season. However, average values and ratios that were found in literature for the amount of urine and faecal matter in black water (Strande et al., 2014; Mawioo et al., 2016) mostly correspond to the ratio that is proposed by the Dutch government.

In literature it was found that the amount of *E. coli* in raw black water can have values of 1.70E08 cfu/100mL (Knerr et al., 2011), 5.01E08 cfu/100 mL (Eregno & et al, 2018) and 2.30E08 cfu/100mL (Mawioo et al., 2016). Also, common values for *Intestinal enterococci* in blackwater were found to be 1.26E08 cfu/100mL (Eregno & et al, 2018). All of the values found in literature were in the order of 1.0E08 cfu/100 mL. This means that to pass the treatment test the system needs to achieve at least a 6-log inactivation of the pathogens inside the black water, which is equal to inactivation of 99.9999% of the pathogens.

1.3 Proof-of-concept method

This master's thesis was conducted in collaboration with a start-up company called Advanced Protective Systems (APS). This is a company that is trying to develop a wastewater treatment system for raw blackwater on board of recreational crafts, and is aiming to be the first to realise and market such a system in the Netherlands.

The goal of this research was to present a PoC treatment method for on board sanitation of blackwater on recreational crafts. A PoC generally means: "Evidence, typically deriving from an experiment or pilot project, which demonstrates that a design concept is feasible" (Oxford University Press, 2020). To establish a method for the PoC, conventional techniques used in drinking water and wastewater treatment plants are weighed against each other. The considerations involved in choosing an appropriate treatment method, which is suitable for usage on board of recreational crafts, are elaborated in this section.

1.3.1 Biological treatment

A wastewater treatment system for on board a recreational craft is going to differ from a conventional bioreactor that is commonly used to treat large quantities of a continuous stream of sewage water. The system needs to be economically feasible for private individuals, should not take up much space and should focus exclusively on the removal of pathogens, whereas a bioreactor generally aims to also reduce substances like BOD, nitrogen and phosphorous from the effluent. As the hydraulic retention time of bioreactors is high, they often need big reservoir tanks which take up a lot of space. Also, these reactors use living microorganisms for the treatment of wastewater, which is not a feasible option on a smaller boat. This is because these microorganisms need a continuous feed of organics as nutrients to survive. When recreational crafts, and thus the toilets, are only sporadically used, which is the case for most of the boats owned by hobbyists, the organisms would likely die-off and the bioreactor would need to be started up again. Due to all of the above, a biological treatment step is not taken into consideration as a method for the PoC.

1.3.2 Physical treatment

In wastewater engineering, physical treatment is the step where the coarser material is separated by using, e.g., screens, clarifiers or filtration. It generally makes use of either a physical barrier like a filter mesh through which the suspended particles can not pass due to their size, or gravity settling of the solid material. A physical treatment step is essential to remove the majority of the suspended solid material of the size that is visible to the naked eye. As the suspended fecal material harbors a large part of the pathogenic bacteria, removing it will be an important step for the treatment system.

A clarifier or settling tank does not seem to be a convenient method for solid separation on board of a boat. Settling tanks generally occupy the largest part of a wastewater treatment plant, and as mentioned above, space is a decisive factor for the design of the system. Also, when a boat is not docked, waves can cause settled material to resuspend in the supernatant liquid. Furthermore, precipitating most of the solids takes a considerable amount of time and there are always solids, also called colloids, that wont settle due to them being too small to be affected by gravity, which makes them float freely through the supernatant liquid.

The benefit of filtration through a physical barrier is that, depending on the mesh size of the filter medium, virtually all the solids will get caught. This is due to the formation of a filter cake on the filter medium that retains the solids, a process called fouling. However, when the thickness of the cake layer increases, so does the flow resistance through the cake. This means that for filtration to be efficient a certain mechanical force will need to be applied on the wastewater to press it through the cake. This will increase the complexity and costs of a treatment system, but an advantage is that an excessive force can be used to dewater the solids to a certain degree. The solid material will retain a lot of water which also means that it takes up a larger volume when stored on board. Dewatering of the sludge can increase its storage capacity and decrease human interaction with the system, i.e., emptying or changing a solids container.

1.3.3 Physico-chemical treatment

Colloids are very fine particles (1 - 10,000 nm) that generally have a high stability due to their electrostatic surface charges of the same sign. This means that they are not able to aggregate and settle. Physico-chemical wastewater treatment involves the use of chemicals that can modify the physical state of these colloidal particles, which helps in making them more stable and coagulable for further treatment or filtration purposes (Chokhavatia, 2021). The processes involve coagulation and flocculation, and this will significantly increase the amount of solids that can be extracted by the system with a filtration step. The addition of chemicals will increase the complexity and costs of the system, but it could also create more revenue if it is sold as an addition to it.

1.3.4 Disinfection

Disinfection techniques are used to remove pathogenic bacteria from the wastewater and to make it suitable for reuse, or in this case discharge. Disinfection can be achieved by physical agents such as heat and light, or chemical agents like chlorine and its compounds, ozone, hydrogen peroxide and many more substances (Bhargava, 2016). The physical and physico-chemical treatment can also be seen as a disinfection step, as these steps also remove pathogen harboring solids.

The disinfection of the filtrate will be the final step before the wastewater is discharged. It is the most important step for reducing the pathogens so that the effluent meets the boundary conditions. Unfortunately, chemical agents do not seem to be a good option for the disinfection step, even though they can be very potent. This is because the addition of oxidants like chlorine, hypochlorite or ozone as disinfectants can often lead to simultaneous production of disinfection by-products (DBP). Normally these DBP are formed by reactions with organic matter, particularly at high concentrations, and these DBP can pose a serious risk to human health and to the ecology of the environment (Benjamin & Lawler, 2013). Furthermore, the required chlorine will increase almost exponentially with increasing turbidity levels in the water. The chlorine demand of a water with a turbidity of 100 NTU will be 70 times larger than that of water with 1 NTU turbidity. Particles in the water are also able to shield microorganisms from inactivation by chlorination (Smiech et al., 2020).

An alternative to the addition of chemicals is ultraviolet (UV) disinfection, as it is much less likely that this method forms potentially harmful substances in the effluent. Just like in chlorination, microorganisms can be shielded from UV light by particulate matter. This means that for UV-disinfection to be effective, most of the particles will need to be filtered out of the wastewater. Furthermore, UV radiation could be absorbed by substances other than the target organisms, in which reactive species such as superoxide radicals could be formed, particularly when natural organic matter (NOM) absorbs the radiation. Nonetheless, due to the relative short contact time of UV light with the particulate matter, very little by-products will be formed (Benjamin & Lawler, 2013).

The use of heat could also be an effective method for the disinfection of the filtrate. An advantage of heat over UV light is that heat will penetrate the wastewater much more effectively, even with higher concentrations of solids, whereas UV light is absorbed and scattered by contaminants in the wastewater, decreasing its efficiency. However, the energy requirement to heat up the wastewater is a major disadvantage as the specific heat capacity of water is very high. The energy requirement for thermal disinfection can be 20 to 30 times higher than that of UV-disinfection by low-pressure UV lamps(Smiech et al., 2020).

1.3.5 Preexisting treatment systems

As discussed in section 1.1, the regulations for recreational crafts have become stricter over the past years. Outside of the regulations that the International Maritime Organization (IMO) have established for the discharge of blackwater by bigger ships on international waters, more countries such as the Netherlands and America, are adopting legislations that also apply to smaller recreational crafts and in national waters. This has catalyzed the development of small-scale treatment systems, that are suitable to be used by private individuals on their recreational crafts. To develop a PoC method, inspiration could be drawn from these preexisting treatment systems.

America is currently leading the market when it comes to small-scale treatment systems. They distinguish between three types of systems that can be certified by the U.S. Coast Guard to meet the requirements in 33 CFR Part 159, which are then permitted to treat or hold blackwater on board. A type I system produces an effluent that has no more than 1000 cfu/100 ml and no "visible floating solids". Type I systems are allowed on vessels that have a maximum length of 20 meters, which falls in the category of this study (U.S. Coast Guard, 2021). A type II system is intended for bigger boats, and often uses biological treatment, and a type III system is typically a holding tank, which means that these type of systems are not of importance for this study.

As of November 2021 there are two type I marine sanitation devices (MSD) in America that have been certified by the U.S Coast Guard. An extensive evaluation on these two MSD has been carried out by the United States Environmental Protection Agency (EPA), in which their effectiveness was evaluated in reducing both fecal indicator bacteria and visible floating solids (EPA, 2010). Below both of these systems are briefly highlighted:

I. Raritan ElectroScan Waste Treatment Systems, which uses salt water and electricity for the disinfection process. Electrodes temporarily convert chloride in the salty water to hypochlorous acid, an effective disinfectant. However, even though it removed almost all of the fecal indicators in the evaluation by EPA (EPA, 2010) and fulfils the requirements for the certificate of the U.S. Coast Guard, there are certain downsides to this system making it not a viable option for in the Netherlands. It needs salt water to function, as the salt content in urine is most likely not enough for disinfection. Discharging this salt water in sweet water lakes could form an environmental problem. It can also only be connected to only one or two toilets, which poses a problem for on board use of middle to large yachts with multiple toilets. The biggest downside is that, just like described in subsection 1.3.4, disinfection by reactive chlorine species results in the formation of DBP, which pose serious risks to human health and the ecology. As the Electroscan system does not filter out the solid material and the NOM, the formation of DBP is ensured.

II. Thermopure-2 Non-chemical Waste Treatment System, which uses low level heat as the main source of disinfection. The waste stream is first macerated while its in the holding tank, after which it is pumped through a heating device. This means that no chemicals or additives are needed, and that the system should work in all types of water (Groco, 2021). However, the evaluation by EPA (EPA, 2010) demonstrated that the designated threshold temperature sufficient enough to kill bacteria was not reached. This resulted in almost no disinfection by the system.

1.3.6 Experimental proof-of-concept method

Taking all of the above into account an experimental PoC is devised, and this method is evaluated in this study through extensive experimentation by using a bench scale experimental setup. The PoC method is schematically shown in Figure 1.1. First, a physico-chemical treatment step is implemented in which chemicals are used for the destabilization of the solid fraction in the blackwater due the processes of coagulation and flocculation.

The formation of larger aggregates of solid material significantly benefits the next step, which is the filtration of the solid fraction from the blackwater. The filtration method used during the experimental phase of this study was dead-end filtration, where a mechanical force pushes the blackwater through a membrane (or dead-end) perpendicular to the force. The rejected solids form a cake near the membrane's surface, which has an influence on the flux through the membrane. Subsequently, the mechanical force is used to also dewater the sludge cake by compressing it and extracting most of the remaining available water. The dead-end method was chosen as a filtration and dewatering step because the equipment that was made available, namely a chamber filter press, did not allow for experimentation in any other way. There was also no other equipment available that could filtrate the solid from the liquid fraction, in the samples that were used during this research. Also, by using a chamber filter press, both the filtration of solids from the blackwater and the dewatering of the sludge cake could be performed in the same experiment.

The last step, is the disinfection of the filtrate, where UV light will be used to inactivate the fecal indicators in the water until the boundary conditions are met and the water is allowed to be discharged from a recreational craft.

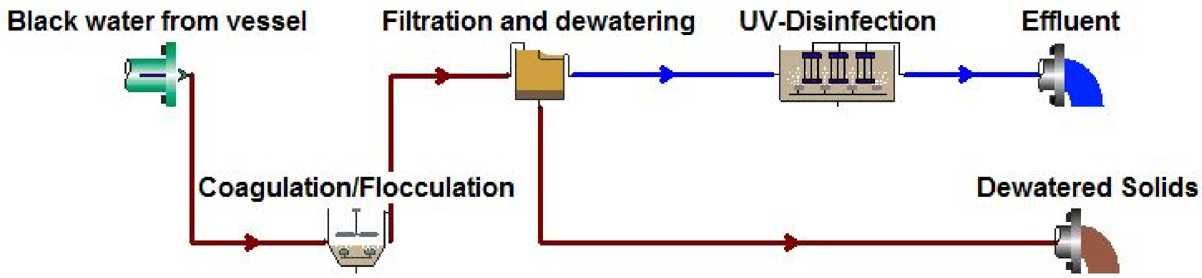

Figure 1.1: Schematic overview of the PoC method that was evaluated during this study. The diagram was made using BioWin

1.4 Research objectives

In section 1.1 a brief description is given of the problem that has occurred in the Netherlands where it will not be allowed to discharge untreated blackwater directly into the environment anymore. This presents a major problem for recreational boat owners, because no solution has yet been devised for it. The overall objective of this research is to find an applicable solution, using wastewater treatment practices, to reach the boundary conditions of fecal indicators in the effluent wastestream of a recreational vessel. The main goal of this research can be summarized by the following statement:

To present and investigate a proof-of-concept method for the disinfection of raw blackwater on board of recreational crafts. The goal is to investigate the coagulation/flocculation, filtration/dewatering and UV-disinfection steps required for a 6-log disinfection, and to find inventive and effective ways to achieve this.

This statement can be divided into the following research questions, that will contribute to the achievement of the main research goal:

I. *What is the most ideal combination and quantity of coagulants and polymers to remove as much TSS as possible from the raw blackwater derived from recreational crafts.*

II. *What is the effect of a polymer on the dewaterability of the sludge from raw blackwater, and what is the optimal dose of polymer with regard to dewaterability.*

III. *Which percentage of solids can be achieved by mechanically dewatering the TSS in raw blackwater. Also, how are the dry solid content and the compression pressure related and what is the most efficient pressure to dewater the sludge from raw blackwater.*

IV. *What is the effect of the particle size and organic matter on the inactivation of the indicator pathogenic bacteria by an increasing UV-C dose and is UV-disinfection a viable method for the disinfection of the filtrate.*

1.5 Thesis outline

This document is structured as follows: This chapter was an introduction to the problem behind this research. Theoretical background to this research will be given in chapter 2. Then, the methodology and experimental design will be discussed in chapter 3. This is where the wastewater samples and sampling methods, and experimental setups are explained that have been used during the experimental phase of this research. chapter 4 will focus on the results which have been categorized into four topics: wastewater quality, coagulation and flocculation, dewatering and UV-disinfection. Interpretations of the results are discussed in chapter 5, together with the implications that the results will have for further development, after which limitations of this research are discussed. This master's thesis will be finalized in chapter 6 with a summary of the most important conclusions and also recommendations for further research and development.

2. Theoretical background

This chapter provides the theoretical background of this research and will introduce all the fundamental concepts that are needed in order to reach the main research goal. The main topics that will be covered in this chapter are derived from the proof-of-concept method and consist of: the fecal material itself, coagulation and flocculation, filtration and dewatering, and UV-disinfection. The chapter will end with a small summary to sum up the main findings.

2.1 Coagulation and flocculation

Wastewater streams generally contain high quantities of dissolved and suspended particles. Along with disinfection, the removal of the suspended solids from the water are the primary goals of water treatment. Also, efficient removal of solids is generally necessary for disinfection processes to be effective, as solids can shield microorganisms off from the disinfectants. The effective removal of the suspended and especially the colloidal particles (1 to 10,000 nm) can pose a real problem for conventional treatment methods like settling and filtration. This is why, most of the wastewater is "pre-treated" by methods of coagulation and flocculation, which is from now on referred to as the conditioning of the wastewater. In other words, wastewater conditioning is a process where solids in the water are treated with chemicals or other means to form particles large enough for the improvement of the filtration and dewatering characteristics of the solids.

The main cause that the colloidal material in wastewaters, without conditioning, do not form bigger aggregates or flocs, is the presence of an electrical charge on the surface of the particles. Colloids are very small in size, which means that in a colloidal suspension gravitational effects are neglected. However, all the colloids together have an extremely large surface area, which makes surface phenomena predominate in colloidal suspensions (Ghernaout & Ghernaout, 2012).

Most of the suspended particles that originate from human faeces have the same negative charge. This means that these particles repel each other and collisions, causing attachment of one particle to another, do not occur. When two particles have enough momentum and come sufficiently close to each other, London-van der Waals forces overcome the repulsive forces and brings the particles together. However, in most suspensions this is very uncommon, which means that a so called stable suspension is retained. When particles are stable in a suspension it means that the particles can stay unchanged in the suspension for a long period of time, without changing the particle size distribution. The key to removing these particles is to destabilize them (Benjamin & Lawler, 2013).

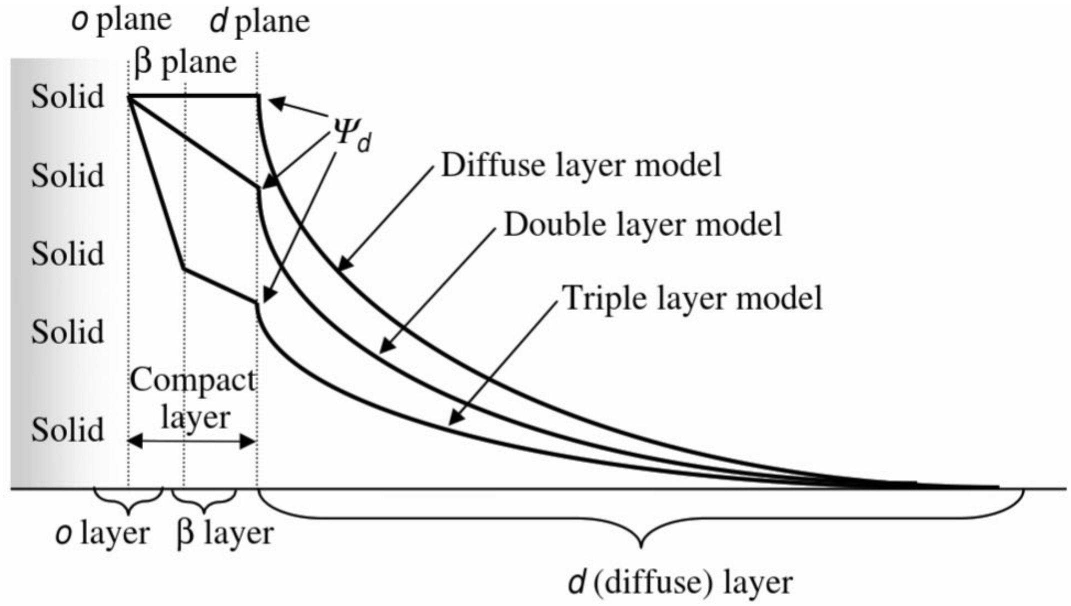

Figure 2.1: *Conceptual model of the ionic distribution around a charged particle's surface. The compact layer is very thin (1 nm) compared to the diffuse layer (10-100 nm). The different models deviate from one another in the description of the compact layer (Benjamin & Lawler, 2013)*

Figure 2.1 shows a conceptual model of electrical potential in the area next to a charged particle. In the triple layer model the *o* layer, also called the "stern layer"", consists of one or a few layers of ions of opposite sign that are adsorbed directly on the particles surface. Located next to the stern layer, is the β layer, where all ions and chemically bound water moves in bulk, bounded to the particle, through the suspension. The *d* plane, also reffered to as the "slipping plane", separates the bound charge from the diffuse charge around the particle. In the double layer model, the adsorbed and bound substances around the particle are combined into the "compact layer". In both models, the *d* plane is the is interface between the particle together with its adsorbed and bound constituents, and the diffuse layer (Andreoli et al., 2007). The ions in the diffuse layer are in the aqueous solutions and ions of the same sign as the particle are less concentrated there than in the bulk solution. The dominant charge in the diffuse layer is critical for the interactions between different particles when they approach each other. The electrical potential at the *d* plane, or the start of the diffuse layer, is specified as the so-called zeta potential, and can also be seen as the “modified” or “effective” surface charge. The zeta potential can be measured and is used as an effective tool for coagulation control as changes in zeta potential indicate changes in the repulsive force between colloids (Ravina, 1993).

The surface charge on particles is the results of the following three processes (Buffle & van Leeuwen, 1992; Ghernaout & Ghernaout, 2012; Benjamin & Lawler, 2013):

I. Due to isomorphic substitution species that have the same size but a different valence replace species in the solid lattice.

II. Chemical reactions at the surface, mostly induced by a change in pH. Most of the solids in feces are biogenic organic solids, that contain carboxyl (−COOH) and amino ($-NH_2$) groups. The surfaces of most bacteria also contain these these groups, which could be of interest for their removal from the wastewater before UV-disinfection even initiates. By decreasing pH of a suspension, these groups that are adhered to these surfaces of organic solids and bacteria interact with protons and hydroxide ions to form positively and negatively charged surface species. At lower pH values, the dominant species exist in their most protonated forms, namely −COOH and $-NH_3^+$. At higher pH the equilibrium shifts and these species exist in their deprotonated form $-COO^-$ and $-NH_2$. At average pH values the carboxyl groups are generally deprotonated and the amino groups are protonated. In wastewaters, both raw and treated, the surface concentration of the carboxyl groups greatly surpasses that of the amino groups, which explains why in a pH range of 6.5 to 8.5 particles are generally negatively charged. The pH at which the surface charge is neutral is called the isoelectric point (IEP)

III. Lastly by adsorption on the particle surface. Free ions in the solution can adhere to the surface of a particle, whereby changing its surface charge. These can be the same ions present in the particle, or ions that are different from those that form the solid.

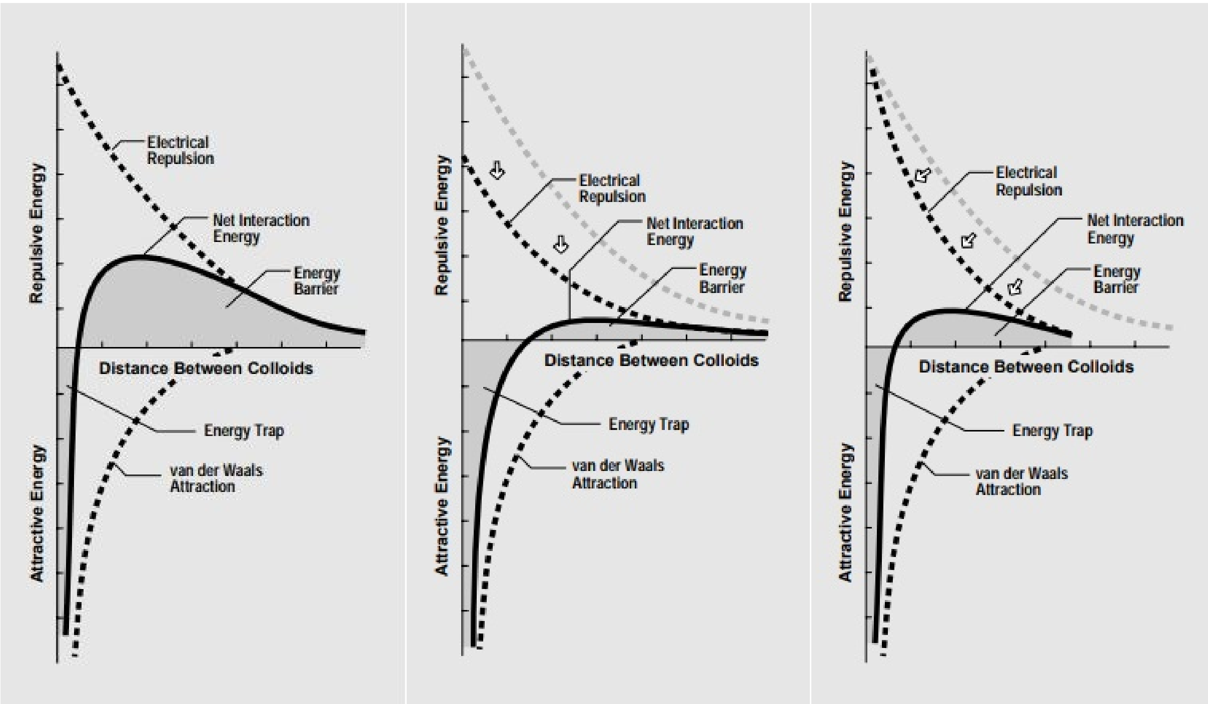

__Figure 2.2:__ The net interaction curve is formed by subtracting the attraction curve from the repulsion curve (left). Coagulant addition lowers the surface charge and drops the repulsive energy curve. More coagulant can be added to completely eliminate the energy barrier (middle). Double layer compression squeezes the repulsive energy curve reducing its influence. Further compression would completely eliminate the energy barrier (right) (Ravina, 1993)

Figure 2.2 gives a schematic overview of the so called DLVO theory, which describes the forces between charged particles and how they interact with each other. Two forces are taken into account: the electrostatic repulsion (positive curve) describes the repulsion that emerges when the diffuse layers of two particles start to overlap, and the Van der Waals attraction (negative curve) between colloids. These two forces combined create the "net interaction curve" on the left in Figure 2.2. If the distance between two particles is large enough they repel each other, and the height of this "energy barrier" indicates their resistance to coagulation. However, if particles posses enough kinetic energy, they can overcome this barrier, after which the net interaction energy is all attractive. Van der Waal forces then bring the particles irreversibly together, which can be seen as an "energy trap" (Ravina, 1993).

As the stability created by the energy barrier prevents particles from aggregating together, the successful removal of colloidal particles depends on destabilisation and overcoming this barrier. This is generally triggered by the addition of chemicals to the suspension. The destabilisation process is achieved by the following four mechanisms (Ghernaout & Ghernaout, 2012; Benjamin & Lawler, 2013):

I. Compression of the diffuse layer, which happens through an increase of the ionic strength of a solution. When for instance a salt is added, more counterions will surround a particle, and this compresses the diffuse layer. This process is shown on the right curve in Figure 2.2. It can be seen that the height of the energy barrier is decreased by the compression, so the particles have to overcome less repulsive energy to fall into the energy trap. It is good to note that the compression of the diffuse layer has an effect on the sphere of influence, but not on the charge of the particles (Ravina, 1993). As this process requires the addition of massive amount of salts to be effective, it is not a feasible method for this study (N. Wei et al., 2015).

II. Adsorption and charge neutralization. This is the process of adsorption of positive charged ions on the surface of a particle, generally through the addition of a so called "coagulant" (such as alum, iron chloride or aluminium chloride). The surface charge of the particle is neutralized , which can result in a net charge of around zero, eliminating the problems that create particle stability (Ghernaout & Ghernaout, 2012). The process of charge neutralization is depicted in the middle curve in Figure 2.2. The value for electrical repulsion of the particle drops due to the adsorption of positive ions, which in turn decreases the height of the energy barrier. The surface charge does not have to be reduced to zero for this process to be effective. The energy barrier should just be lowered enough so that the velocity of particles due to mixing can overcome it (Ravina, 1993). The necessary dose of chemical for effective charge neutralization is correlated with the amount of solids in the suspension, particularly the total surface area, as this determines the cumulative surface charge. This could lead to operational difficulties when the TSS concentration of a wastewater is highly variable, as the coagulant dose will need to be adjusted to the TSS (Benjamin & Lawler, 2013). This should be considered during this study, as the composition of the supplied blackwater will be highly variable.

III. Sweep flocculation, which is also referred to as enmeshment in a precipitate or colloid entrapment. It happens when a high enough quantity of metal salts like $AlCl_3$ or $FeCl_3$ are added, after which a reactions in the water takes place that form the metal hydroxide precipitates $Al(OH)_3(s)$ or $Fe(OH)_3(s)$. To achieve this, an overdose of coagulant is required, compared to the charge neutralization process. The positively charged metal ions are first adsorbed on the surface of the colloids, the same as in charge neutralization, which destabilizes the particles. However, then metal hydroxides will form onto the adsorbed metal ions, which enmesh the particle, and will form larger flocs with the other precipitates in the suspension. These precipitates are also positively charged, which means that more negatively charged colloidal material will adhere to it. All of the above results in the rapid formation of relatively large, white fluffy flocs that will sweep the solids they it comes across out of the suspension by enmeshing them in their floc structure, hence the name "sweep flocculation" (Ghernaout & Ghernaout, 2012; Benjamin & Lawler, 2013). Sweep flocculation does not work when alkalinity is too low as the metal ions combine with the minerals that make up the alkalinity to form $Al(OH)_3$, using up this alkalinity (Ye et al., 2007). The effectiveness of metal salts has been studied by Amirtharajah, who made a stability diagram for the effective zones of pH and coagulant dose for the formation of the hydroxides, which is shown in Figure 2.3. It can be seen that the sweep flocculation mechanism occurs in the range of pH 6 to 8.5 where the conditions are suitable for the rapid formation of $Al(OH)_3$.

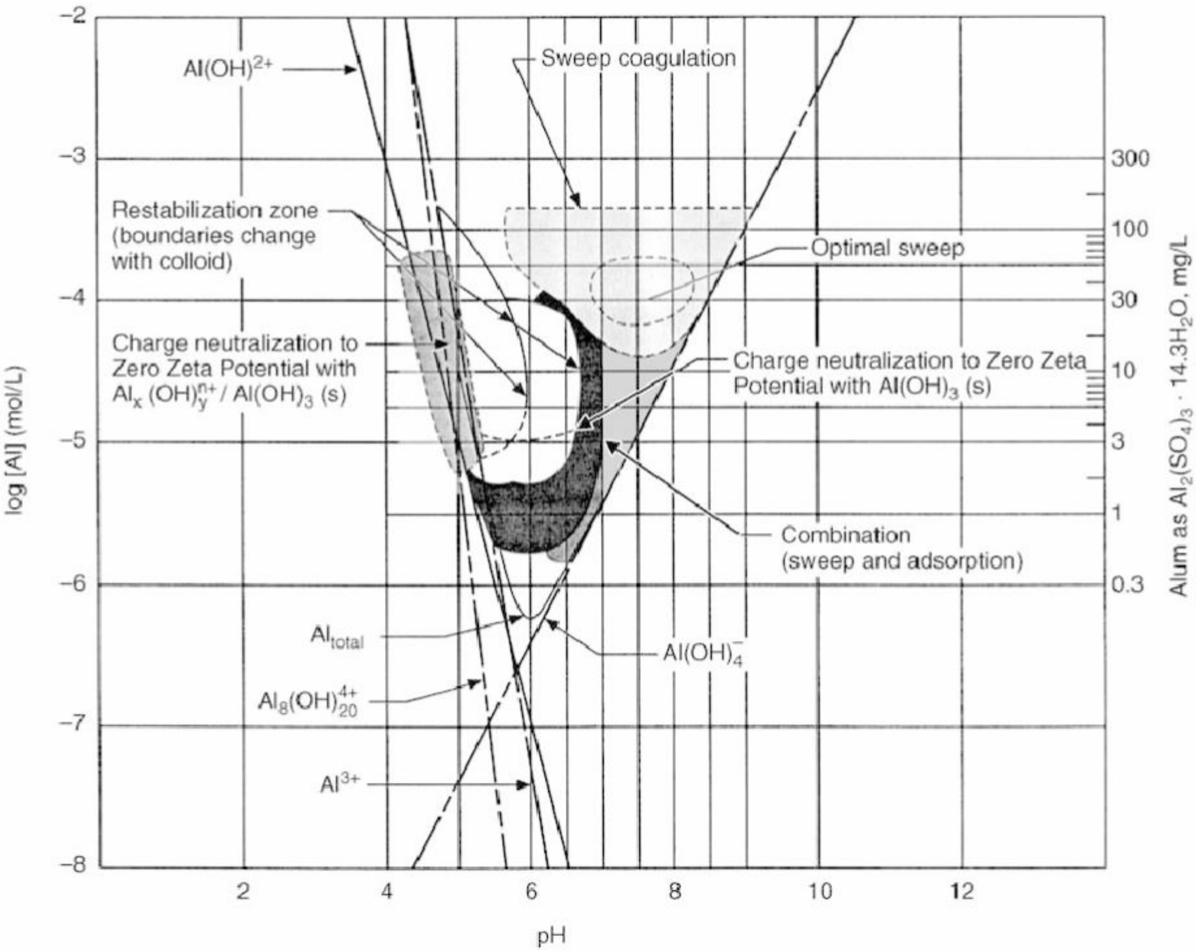

Figure 2.3: *Aluminium destabilization diagram (Amirtharajah & Mills, 1982)*

IV. Inter particle bridging, which is generally achieved by using high-molecular weight polymers. These polymers destabilise the colloids by forming bridges that extend between them. The polymers have reactive groups that bind to specific sites on the surface of the colloidal particles. When a group on a polymer molecule attaches to a colloid, the remainder of the long-chain molecule extends away into the water. Once the extended portion of the polymer gets attached to another colloidal particle, the two particles become tied together or bridged by the polymer(Wang et al., 2005). When a polymer is overdosed, which can also mean that there are not enough particles to bind with, the polymer could wrap around the particle and restabilize it again, redispersing the particle back into the suspension

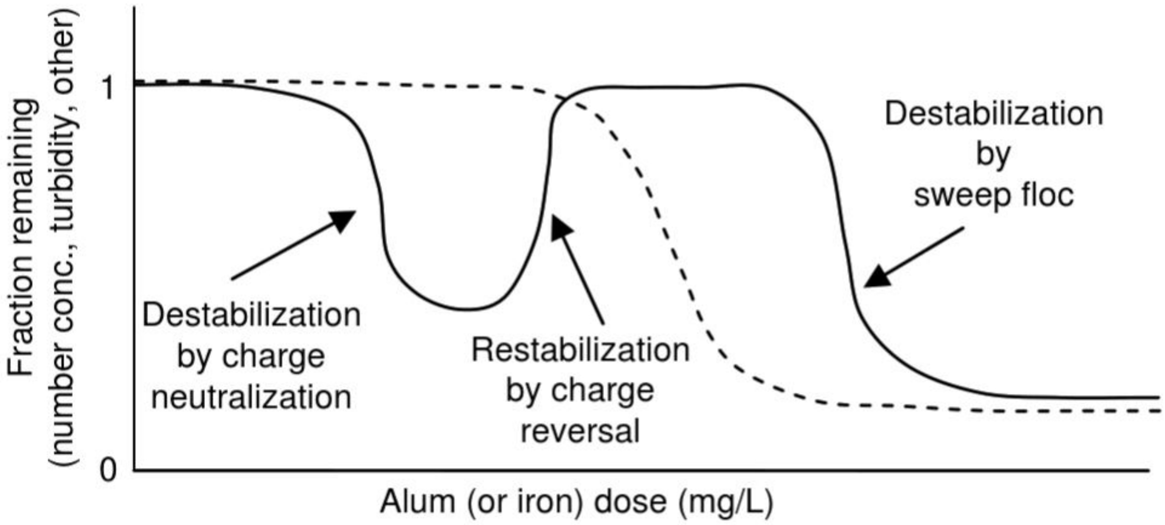

Figure 2.4: *Curve of two different methods of destabilization due to the addition of metal salts. The solid line represents a less concentrated suspension in which the pH is favourable for the two mechanisms to be separated. The dotted line is a more concentrated suspension at the same pH level (Benjamin & Lawler, 2013)*

As explained above the two ways that a coagulant can destabilize particles in a suspension are charge neutralization and sweep flocculation, the first requiring significantly lower quantities. Figure 2.4 gives a schematic view of the dominant process at increasing coagulant dose. It shows that overdosing can reverse the charge of the colloid. The driving force of the adsorption of the positive ions onto the surface of solids is not primarily electrostatic, so dosing a coagulant beyond the quantity needed for charge neutralization will result in charge reversal (Benjamin & Lawler, 2013). This is of course undesirable because solids will restabilize and redisperse in the suspension as positive colloids (Ravina, 1993). However, after restabilization due to reversal of the original particle charge, destabilization can again be achieved by overdosing the coagulant, inducing sweep flocculation. With high particle concentrations, like in the blackwater that will be used for this study, both the neutralization and sweep floc regions blend into each other, making it impossible to differentiate where one region ends and the other begins.

The terms coagulation and flocculation are sometimes used interchangeably and ambiguously, which may cause confusion. That is why their significant meaning in this report should be explained. Coagulation is seen as the process of lowering the energy barrier and destabilizing the particles in a solution, or the processes of charge neutralization. The process of bringing particles together with the intentioon of creating larger particles is referred to as flocculation.

Coagulants are the chemicals that induce the charge neutralization, and flocculants are the high-molecular weight molecules that cause interparticle bridging, but they can also be referred to just as polymers. It should, however, be noted that cationic polymers also could cause slight charge neutralization.

2.2 Filtration and dewatering mechanisms

The residue that accumulates in the treatment of wastewater, which is generally comprised of by-products of the wastewater treatment process, is classified as primary or secondary sludge. In this study, the sludge will be the raw, unprocessed, and unchanged fecal material that has been filtered from the blackwater. Without a thickening or dewatering step, the sludge is on average 93-99% made up of water (Andreoli et al., 2007; Strande et al., 2014; Gold et al., 2017). The main objective of the dewatering of sludges is to reduce the water content in the sludge, which reduces its total volume and weight, increasing the storage capacity and also preparing the sludge for a more cost efficient disposal. In waste water treatment the dewatering step forms a bottleneck in sludge handling, as it is a very energy intensive step.

Both the filtration of the solids and the dewatering of the sludge will be an important step in the PoC method. One of the objectives of the treatment systems will be to filter out as much fecal material as possible as they contain fecal indicator organisms and will hinder the UV-disinfection. Another perquisite is that the storage capacity of the dewatered sludge is large enough so that human interactions with the system, like removing and emptying a storage reservoir, are reduced to a minimum. The following paragraphs elaborate on the water distribution in sludges, filtration of solids from a waste stream, and the dewatering characteristics of sludges.

2.2.1 Water distribution

To better understand the reasons for difficulties in dewaterability of sludges, appropriate indices are needed. An approach for this is the knowledge of how water is distributed within a sludge. The representation of the moisture distribution within the sludge has always been considered to be essential for the examination of dewatering problems. However, scientific literature in this field is often hard to apply because of controversial data and definitions (Vaxelaire & Cezac, 2004). The water in a sludge can be subdivided into different types. These types of water are mainly distinguished from each other on the basis of the mechanism and the intensity of their physical bonding to the solids (Kopp & Dichtl, 2001a). The type of water content in a sludge is said to determine the extent of solids dewaterability, because some types of water are easier to remove than others. Depending on the overall chemistry, the physical distribution of water within the slurry particles can vary (Avila & Novak, 2019).

Water in sludge is available in free or in bound forms. This is an important differentiation in understanding treatment mechanisms because the free water is said to be fairly easily removed, while removal of the bound water poses much more problems (Strande et al., 2014). Free water, or better called bulk water, usually represents the majority of the water in untreated wet sludge. The proportion of free water accounts for approximately 70 to 90 % of the total water content in wet sludges (H. Wei et al., 2018; Kopp & Dichtl, 2001b). It moves freely between the individual sludge particles, without experiencing forces from the solid particles.

It is not adsorbed or bound to solid particles. Free water is not influenced by capillary forces and behaves physically and chemically as pure water (Strande et al., 2014).

Bound water is water whose properties are modified due to the presence and influence of solid particles. As discussed previously, most of these solid particles have a negative surface charge, which interacts with the polar water molecules. The portion of bound water can be further differentiated, depending on the strength of its interaction with the solid phase, into three physical forms. Although this classification is a simplification of the true behaviour of water in different kind of sludges, it is generally sufficient for dewatering considerations. The different types of bound water are:

I. Interstitial water, or also referred to as capillary water. It represents the water that is trapped in interstices and pore spaces of solid materials like the flocs and organisms. It is bound to the solid fraction by capillary forces. Some of this water is held within the floc structure, and can become free water if the floc structure is destroyed. The effects of the solid surfaces on this water are unclear, but evidence suggests that some of this interstitial water may not behave physically as bulk water (Verslind, 1994).

II. Vicinal water, which is also called surface water. This is the water that is physically bound to the surfaces of particles due to adsorption and adhesion. In section 2.1 the compact layer was discussed, wherein everything moves in bulk, bounded to the particle. The water inside of the compact layer is referred to as the vicinal water. As discussed in the previous chapter, this includes the water molecules that are adsorbed onto the surface of the negatively charged particles, and the water molecules that are not directly adsorbed, but chemically bound onto the solids. The properties of this vicinal water, near solid particles, differ from water in the bulk liquid. The density of the vicinal water is 3% lower, but heat capacity is 25% higher. Just as the diffuse layer theory states is that the structuring of vicinal water decays in an exponential manner with distance from the interface and hydrogen bonding near water molecules is enhanced at shorter distance near the solid surface (Etzler et al., 1990).

III. Intracellular water, represents the water that is present on the inside of microbial cells and is also called water of hydration.

J.Kopp et al. (Kopp & Dichtl, 2001a) states that the water binding energy is less than 0.28 kJ/kg water for the free water content, and more than 5 kJ/kg water for the bound water. Another study by Chu et al. (Chu et al., 2005), reported similar results of a bond energy close to 1 kJ/kg water for an excess water content of 30kg/kg DS. In their study the bonding energy exceeded a tremendous 1 MJ/kg when the water content was less than 0.5 kg/kg DS. What these studies imply is that when the residual water content is predominantly bound water, the dewaterability of a sludge decreases rapidly, as the force to overcome the bond energy becomes exceedingly high. This implies that there is a trade-off between applied force and dried solid content to a point where the extra dried solid content does not outweigh the extra force that is needed to achieve this.

2.2.2 Filtration and dewatering mechanisms of the solid fraction

After the conditioning of the solids in the blackwater, the next step is the filtration and dewatering of this material from the solution. The removal of the solid fraction reduces the concentration of fecal indicators in the effluent, and it also is necessary for UV-disinfection to be effective. In section 1.3 it was established that a settling tank is not a feasible option for on board of a boat. For the PoC method, dead-end filtration was adapted, where a mechanical force pushes the blackwater through a physical barrier like a membrane, which filters out most of the solids. The force is then used to also compress and dewater the sludge cake that is formed

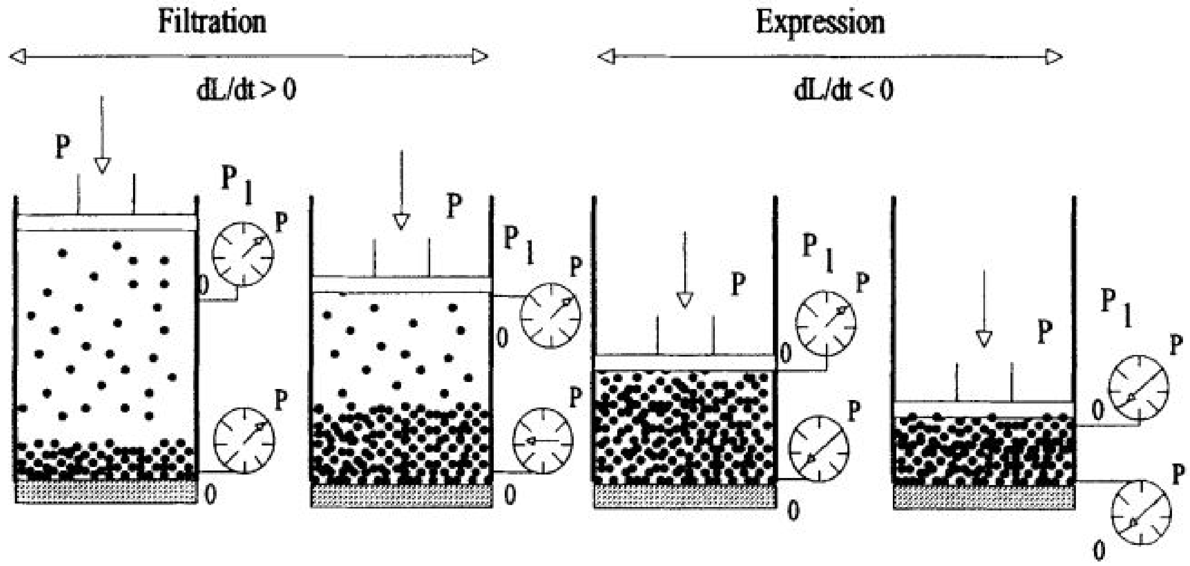

Figure 2.5: Schematic representation of the filtration and expression/dewatering mechanism. Here L is the cake thickness and p is the hydraulic pressure (Heij & Kerkhof, 1996)

Dead-end filtration with the use of a chamber filter press can be distinguished into two phases: the filtration stage and the expression or compression stage. In Figure 2.5 these mechanisms are schematically depicted for an experiment in a filter cell with a piston type dewatering device, which corresponds to the equipment that was used in the experiments. Once the press chamber is full, the filtration stage starts. A compressive piston pushes the water with the solids in the direction of the physical barrier. A filter cake will build up ($dL/dt > 0$, L is cake thickness), and as the filtrate flows through this cake more and more solids are deposited. As the cake builds up, the hydraulic pressure at the top of the cake and in the suspension remains constant and equal to the applied pressure of the piston (Heij & Kerkhof, 1996). The ratio of the pressure drop across the physical barrier to the pressure drop across the cake decreases with time (Qi et al., 2011). The cake compression or expression stage begins when the compressive piston comes in contact with the solids of the filtration cake ($dL/dt < 0$). During this stage, the cake is squeezed into a more compact form, decreasing the cake thickness L, which leads to a lower final moisture content (Heij & Kerkhof, 1996). A successful sludge dewatering process produces a permeable and rigid lattice structured cake that remains porous and incompressible under high pressure during the compression step, maintaining its porosity through which the water is expressed (Mowla et al., 2013).

2.2.3 Dewaterability of sludges

As already has been discussed, the content of bound water is one of the major influencing factors that limit sludge dewaterability (H. Wei et al., 2018; Kopp & Dichtl, 2001b, 2001a; Verslind, 1994). The part of the bound water that has a chance to be removed from the sludge cake during the expression phase is the interstitial water content, when it is released by breaking up the floc structure, which amounts to approximately 4-13% of the total water content (Kopp & Dichtl, 2001b). Vicinal water has a very limited availability, is impossible to dissolve and, due to the high heat capacity, has a large evaporation enthalpy. As the water molecules are physically bound to solid particles surface, it cannot be separated by any mechanical means. As previously discussed, one of the few ways to decrease the vicinal water around a particle is by compression of the diffuse layer, which decreases the length to the slipping plane, and by adsorption of positively charged ion on the particle surface, which replaces adsorbed water molecules and decreases the negative charge of a particle. The intracellular water can be released only by thermo chemical destruction of the particles at temperatures above 105 C.

Besides the bound water, the particle size and floc structure also have an effect on the degree of dewaterability that can be reached. The coagulation and especially the flocculation process is an essential pretreatment for sludge dewatering, as it reduces the specific resistance to filtration (H. Wei et al., 2018), which is a measure of the resistance of the filter cake to the transport of filtrate through the cake and is used extensively as a measure of dewaterability (Ng & Hermanowicz, 2005). The size of the particles in the suspension is crucial for the specific resistance. When a sludge is not conditioned sufficiently, the smaller particles will fill up the cakes porosity, which significantly decreases its dewaterability. The inner particle bridging mechanism initiated by a polymer has can have a positive effect on the dewaterability, as it will absorb lots of the smaller particles on the long polymer chains (H. Wei et al., 2018).

Achieving high final cake solids is also hindered by compressibility of the flocculated sludge during the expression stage of mechanical dewatering. Reducing the sludge's solids compressibility, which means increasing the sludge's resilience against high pressures, will improve the sludge cake filterability. The compressibility of a sludge cake could be reduced by the addition of inert materials with a high porosity such as fly ash and chemical gypsum, or materials containing carbon like coal, wood chips and lignite (Qi et al., 2011). These materials will act as skeleton builders, which are materials that increase the dewatering performance greatly due to their different pore structure. The materials will construct porous channels in a sludge cake, which are responsible for the effective drainage of water. Skeleton builders will decrease the compressability of a sludge and improve the mechanical strength, while maintaining its porosity under high pressures. The porous channels created by these skeleton builders can increase the dewatering efficiency by approximately 30% (Zhang et al., 2018). Skeleton builders could have a positive effect on the dewaterability, and decrease the amount of water in the sludge cake, but the volume of the sludge cake will also increase due to the addition of incompressible solids. Experiments can determine if the higher solid content in the sludge cake by the addition of skeleton builders is really worth the extra step and addition of extra materials.

2.3 UV-disinfection

The last step in the PoC method is the disinfection of the filtrate, which will now mainly consist of water, urine, salts, humic matter and very small pieces of colloidal and suspended faecal material. Disinfection is considered to be the primary mechanism for the inactivation and destruction of fecal indicators, which makes it an essential step considering that the boundary conditions for the treatment system only imply a reduction in fecal indicators. The ability of UV-disinfection to be effective is dependent on the successful removal of most of the particulate matter, which will be discussed in this section. The removal of the fecal material also has a disinfecting effect, as a lot of fecal indicators are associated with the particles. This reduces the pressure on the UV-disinfection step.

The mechanism of disinfection by UV light differs from chemical disinfectants such as ozone or chlorine species. UV-disinfection systems transfer electromagnetic energy from a mercury arc lamp to an organism's genetic material, its DNA and RNA. When UV radiation penetrates the cell wall of an organism, it destroys the cell's ability to reproduce, thus resulting in the inactivation of the microorganism (Bolton & Linden, 2003). The disinfection is only effective when the UV light has a wavelength that the DNA will absorb inside its structure. This absorption does not occur above wavelengths of approximately 300 nm. Furthermore, the fraction of UV with wavelengths below 200 nm cannot penetrate water. This portion of the UV spectrum, which is effective in disinfection, is called the "germicidal range" and peak absorbance for germicidal activity has been established at 265 nm (Mamane, 2008). Low-pressure mercury lamps emit UV light that is almost monochromatic at a wavelength of 253.7 nm, which is close to the peak absorbance value for microorganisms. This makes the low-pressure lamps effective at disabling fecal indicators.

The efficiency of UV disinfection of a water column strongly depends on certain parameters that decrease the UV intensity throughout this column. The ability of UV light to penetrate wastewater is typically measured in a spectrophotometer at the same wavelength as is produced by low-pressure UV lamps. This measure is called the absorbance and is a function of the transmission, or transmittance. The absorption of light by a liquid comprises of all the factors that absorb, reflect or disturb the UV light in its path from source to destination. As the UV absorption through as medium increases, the effectiveness of UV disinfection decreases and the residual fecal indicators after treatment increase (Das, 2001). The contact time required for UV disinfection is typically less than 1 min. More important is the proximity of the organisms to the light source, which is ensured by maintaining turbulent conditions and a small distance between the lamp and the sample (Benjamin & Lawler, 2013).

2.3.1 Particle effects on UV-disinfection

The main impact on the transmission of UV light through wastewater is from suspended particles in the water column (Kollu & Ormeci, 2012) (Emerick et al., 2000) (Winward et al., 2008) (Madge & Jensen, 2006) (Mamane, 2008). Fecal indicators in wastewater may be either particle or non-particle associated. This means that the fecal indicators are either partly or completely enmeshed within a floc structure, or they are dispersed in the solution. If the fecal indicator is non-particle associated, then the UV light can potentially inactivate the microorganism by direct path to the target (Mamane, 2008). Disperse fecal indicators in a solution are readily inactivated because they are almost fully exposed to UV light. However, when a floc particle is harboring a fecal indicator, the disinfectant must first pass through the associated particle before reaching the target DNA and RNA of the microoganism. Therefore, disinfection problems most often result from the influence of particle-associated organisms (Emerick et al., 2000).

The general representation for disinfection kinetics is by means of a dose response curve, which is usually presented in a semi logarithmic form. The surviving CFU's are put on a logarithmic y-axis and the UV dose on the x-axis. The total suspended solids (TSS) in the water column will have two major effects on UV-disinfection that can be distinguished in the dose response curve. Firstly, an increase in TSS will cause a decrease in the rate of disinfection, characterised by a less steep log-linear graph at lower UV doses. Second, the tailing of the dose-response curve at higher doses, which is attributed to suspended solids as indicated above. This is noticeable in the dose-response curve as a so called "tail", which is a transition from the initial log-linear decrease, through an inflection point to a region of lesser decrease or even to a near plateau in the tailing region (Azimi et al., 2011). Tailing phenomena usually occur at high UV dosages, because at lower doses the initial stage of the disinfection of disperse microorganisms is still in progress and the disinfection follows the single exponential model. Even though this effect does not appear until after 3 or 4 logs of inactivation have already occurred, it is considered the primary limiting factor to disinfection capacity of UV light (Madge & Jensen, 2006). Because dose response curves of filtered and unfiltered wastewater are similar down to approximately 1% survival or two logs of inactivation, it has been estimated that approximately 99% of measurable microorganisms exist in a free-floating state, while 1% of measurable microorganisms are associated with floc or inert solids (Madge & Jensen, 2006).

The total suspended solids cause the aforementioned effects by interfering with UV disinfection in the following ways:

I. By scattering, absorbing, reflecting or refracting UV light. When UV light is scattered, reflected or refracted by particles, it is still capable of inactivating microbes. UV light that is absorbed by particles, on the other hand, is no longer available for inactivation (Kollu & Ormeci, 2012).

II. By shading fecal indicators located behind the suspended solids from UV light.

III. By shielding embedded organisms from UV light. In this phenomenon microorganisms can associate with particles to such a degree that they are partly or sometimes even completely shielded from incoming UV light. This will result in a "residual" concentration of fecal indicators, that even after high UV light doses is still not inactivated (Emerick et al., 2000).

The critical size at which suspended particles are capable of shielding bacteria from UV disinfection has been reported as 10 μm (Winward et al., 2008; Kollu & Ormeci, 2012; Benjamin & Lawler, 2013). Particles that are smaller then 10 μm in diameter will mostly have an insignificant effect on the tailing region of the dose response curve as UV light can fully penetrate particles up to this thickness (Loge et al., 1999; Jolis et al., 2001). Particles from 10 μm up to approximately 40 μm in diameter can in general be penetrated by UV light, but the UV demand increases (Das, 2001). Therefore, these particles will have a measurable effect on the tailing region of the dose response curve. Particles bigger than 40 μm will often not be fully penetrated by UV light, which means that there is a chance that fecal indicators enmeshed within these particles have a chance to survive even the higher doses of UV.

It is unlikely that the treatment processes can be tailored to allow light transmission into solid material that exceeds a certain thickness, which is why the relation between filtration and UV disinfection is important for this research. Also, at UV doses lower than 10 mJ/cm^2, neither particle size nor degree of flocculation has a significant effect on the inactivation of fecal indicators. The majority of the bacteria inactivated at these low UV doses are likely non-particle associated that have full exposure to UV, dispersed bacteria are likely much higher in numbers compared to particle-associated bacteria (Kollu & Ormeci, 2012).

The irradiance is a measure of how much energy falls on a particular surface. Average sample irradiance for UV-disinfection of wastewater can be calculated using measured absorbance coefficients and the integrated form of the Beer–Lambert law described by Morowitz (1950) (Bolton & Linden, 2003). If stirring is applied, in a completely mixed liquid sample with depth *D*, each parcel would receive an average UV intensity I_{avg}. This average intensity can be determined by the following equation, which is an integration of the Beer-Lambert's law over the entire liquid depth (Kuo et al., 2003):

$$I_{avg} = I_0 \left(\frac{1 - \epsilon^{\alpha D}}{\alpha D} \right) \tag{2.1}$$

In this equation the α parameter is the measured absorbance coefficient. In UV experiments the irradiance (I_0) is generally measured with a radiometer. To establish the dose-response relationship, the total applied dose is then calculated as the product of average UV intensity and exposure time *t* as follows:

$$Dose = t \times I_{avg} \tag{2.2}$$

The UV exposure dose is usually given in mJ/cm^2. In the above mentioned approach, the light reflected by the wall of the petri dish is not taken into consideration. As a result, it may underestimate the amount of UV intensity that the liquid sample receives (Kuo et al., 2003).

3. Methodology

3.1 Methodological approach

The main goal of this research was to present a PoC method for the disinfection of wastewater on board recreational crafts. In order to reach the main research goal and to answer the different research questions, quantitative data was needed. There was limited data available regarding this specific type of treatment of raw blackwater, which left some research gaps between the literature and the research goal. Based on these research gaps, the research questions were formed. The data that was required to answer the research questions was obtained by conducting several bench scale experiments that tried to simulate the PoC method as closely as possible.

Most of the experimental data was gathered by controlling and changing certain variables, such as coagulant and flocculant dose, pressure and time, and UV dosage. To quantify certain experiments, the properties of the samples were measured by the instruments provided for that purpose. Furthermore, when optimal values for the variables were known, some comprehensive experiments were performed in order to observe how an actual treatment line would work on different sample types.

The experimental approach of this research is one that is seen more often in the field of wastewater treatment, often performed using conventional experiments and machinery. However, what set this study apart, was that the sample worked with, namely the rawest fecal material possible, is rarely used in this type of research approach. Most of the research conducted in this field is done on secondary wastewaters, and some on primary fecal waste that is for instance collected from septic tanks. These wastewaters have a composition that is different from raw fecal wastewater, as a more mature waste stream has undergone numerous processes affecting it.

3.2 Materials and methods of data collection

3.2.1 Fecal material

The WaterLab is a water treatment research laboratory, situated in the faculty of civil engineering and geosciences at the TU Delft. Biosafety limitations at the WaterLab strictly forbid openly testing with raw fecal material, as there is a high risk of pathogenic transfer. Because nearly all of the experiments needed to be carried out with raw fecal material, it was not possible to use some of the machinery in the lab. Also, due to the high pathogenic load of the material, a laminar flow cabinet was used to conduct most the experiments in throughout this research. Having a restricted area to conduct experiments in with a laminar air flow, minimized the chance of contamination and the UV-C germicidal lamp made the disinfection of the workstation convenient.

Throughout this research two distinctly different samples of raw fecal wastewater were used. Both represent blackwater streams that the system could be treating in the future. The first sample was collected from a super yacht docked in the Rotterdam Marina harbor situated in Kop van Zuid, Rotterdam. The yacht is 28 meters long and on average two to three people are living on it continuously. The yacht has a total of four toilets driven by a vacuum generator connected to a 440 liter wastewater storage tank. Figure 3.1 shows a schematic overview of the toilet and piping system on board of the yacht. The storage tank is connected to a pipe that leads to the upper deck, which is used to discharge the wastewater in a designated facility, and also a pipe leading to an exit point beneath the surface level of the water. The latter is the commonly used option by these kinds of crafts to discharge the wastewater on the water surface. A vacuum toilet like this uses on average 0.5 to 1.5 liters of water per flush.

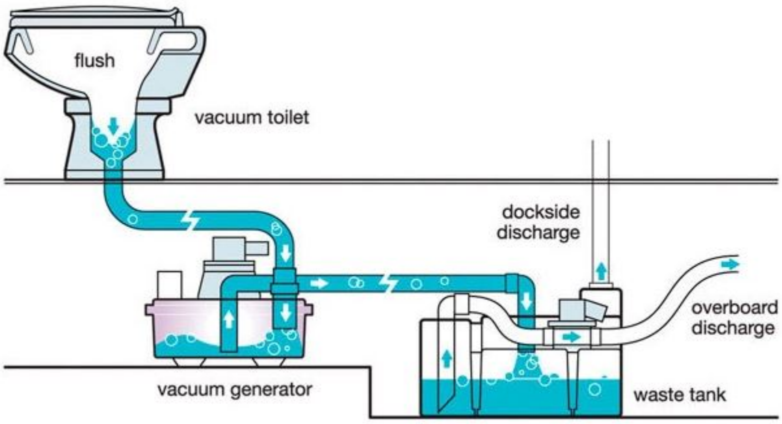

Figure 3.1: *Schematic overview of a vacuum toilet system. This is a toilet system that is most commonly used in yacht that have more than one toilet connected to the system (Marine sanitation and supply, 2021)*

To extract the wastewater from the system in a controlled manner, an extra tube was inserted into the waste tank, reaching all the way to the bottom. When the tube does not reach all the way to the bottom of the tank there is a chance that the faster settling solids precipitate and do not end up in the sample. A macerator with a pump was added on this tube to homogenise the wastewater, and before extracting a sample the wastewater was left to homogenise for 10 to 15 minutes. The average temperature of the wastewater tank was 15 to 20 degrees Celsius.

The other sample used during this research, which was distinctly different from the previously described sample, was collected from a campervan. This campervan was generally used for only a few days per week by 3 people on average and it contains a single chemical cassette toilet. This is a type of toilet where the feces and urine are collected in a removable 15 to 20 liter waste-holding tank beneath the toilet head. Only a very small amount of water is manually used per session, so it's therefore hard to establish the average addition of flush water. However, the water use is distinctly less compared to a vacuum toilet like the one on the yacht. This characteristic is the main reason why both of the samples are evidently different from each other. The waste-holding tanks are, in general, supplemented with product that contains formaldehyde, mainly against smells and to break down the fecal material into a liquid. However, it was very important for this research that this product was left out and that the the holding tank was completely free of it, as it has a major effect on the characteristics, mainly pH and fecal indicators, of the wastewater. This will be explained in more detail in subsection 5.2.1. A macerator was used to homogenise this blackwater before sampling.

To make this study repeatable, a standard sampling method has been applied throughout the span of this research. After homogenization by the macerator the sample was collected in a 10 liter jerrycan and brought to the WaterLab, where it was stored in a cooling cell at six degrees Celsius. To prevent excess die-off of pathogens and to suppress reactions in the wastewater due to an increase in temperature, the time between collecting the sample and storing it in the cooling cell was kept at a maximum of one hour. Another thing to take into account was the time that the wastewater spend in the storage tanks before it was collected. As higher temperatures lead to faster die-off of pathogens and catalyses transformation reactions in the wastewater, it was important that the wastewater was collected as soon as possible. Even though this was not always possible, during this study efforts have been made to keep the wastewater for a maximum of three days in the storage tank.

3.2.2 Coagulants and flocculants

During this study, several different conditioning agents have been used. Almost all of these additives were supplied by Kemira, a Finnish global chemicals company situated in i.a. Rotterdam. Kemira was chosen as a supplier because it is one of the largest companies in the water chemistry industry and a contact person from Kemira already contributed to preliminary research before this study was conducted. Following is an explanation of the different conditioning agents and how they were used.

KEMIRA PAX-18

KEMIRA PAX-18, from now on called PAX-18, a mid-basicity polyaluminum chloride and a effective coagulant for treatment of wastewater. PAX-18 is based on highly charged aluminum, which significantly decreases the required amount. This results in lower dosages and therefore reduces sludge volumes and pH adjustment demands. The total aluminum (Al^{3+}) of PAX-18 is $9.0 \pm 0.3\%$ and its basicity is $42 \pm 3\%$ (Kemira, 2017).

KEMIRA PAX-HP800

KEMIRA PAX-HP800, from now on called PAX-HP800, is highly similar to PAX-18. It is a high basicity polyaluminum chloride. PAX-HP800 is designed for high performance in cold water applications and also for a more efficient application using less total aluminum. The total aluminum (Al^{3+}) of PAX-HP800 is $5.5 \pm 0.3\%$ and its basicity is $67 \pm 2\%$ (Kemira, 2014).

Superfloc®C-2230

Superfloc®C-2230, from now on called C-2230, is an ultra high molecular weight flocculant. It is a cationic polyacrylamide copolymer (PAM), which is a highly water absorbant emulsion, and it is added to water to form a viscous gel. The relative charge density of C-2230 is low. The recommended range of working solutions is 0.15% to 0.5% concentration; however, stock solutions can be prepared up to 1%, but not lower than 0.15% as activators in the product function poorer when the concentration is too low. It was recommended by an employee of Kemira that for the samples used in this research a concentration of 0.15% to 0.25% is maintained (earlier testing on the samples in 2020 showed a concentration of 0.2% to be ideal). When this report states a dosage of C-2230 in ml/l, the amount of stock solution (0.2%) per liter of blackwater is meant. Every day a new stock solution was freshly prepared, as the efficiency of polyacrylamide copolymers in a solution decreases over time. It was very important to wear adequate protection in the form of rubber gloves, goggles and protective clothing when handling C-2230 as direct contact with skin or eyes can have severe consequences (Kemira, 2015). To create the solution a 1mL syringe is used to dose 0.2mL per 100 mL of demineralized water in a glas beaker. The most efficient way to incorporate the C-2230 into the water is by slowly dripping it into the beaker while stirring vigorously. When all of the C-2230 is dissolved the solution should be aged for 15 minutes to achieve maximum effectiveness.

3.2.3 Samples analyses

The WaterLab in TU Delft has numerous practical set-ups and research experiments, which were made available to conduct experiments on. The following paragraphs will expand on the practical set-ups that were utilized for analysis and data collection during this research.

Jar test

For over 50 years, the jar test is the standard technique that is applied when an optimization is needed for the addition of coagulants and polymers in drinking water and the wastewater treatment industry (ASTM, 1995). Polymer and coagulant interactions are too complex to model, so laboratory studies are needed to determine the optimal dosage, duration and intensity of mixing, flocculation and settling time, and pH. The coagulation–flocculation tests of the conditioning agents were carried out following standard practice, used to evaluate the chemicals, dosages, and conditions required to achieve optimum results (ASTM, 1995; Ebeling et al., 2005). Nonetheless, in some cases the results that are obtained from a laboratory setup may deviate from the realistic scenario. This is something that should to be taken into consideration when designing the system.

For the tests a standard jar testing set-up was used, with six 1 liter glass beakers. A single variable speed motor drives the six flat paddles all at the same rate from from 0 to 300 rpm, the paddles could not be driven at different speeds. A built-in light behind the beakers helps monitoring of the coagulation and flocculation processes.

For each jar test, the following procedure was followed (Ebeling et al., 2005). Each jar was filled with one liter of blackwater sample measured inside of the glass beakers. During a jar test experiment it is essential that the conditioning agents are dosed simultaneously in all of the six beakers, which can be a tedious task. Small beakers, just big enough to contain the dosage of the conditioning agent, were used for convenient dosing during the experiment. A jar test experiment may include the addition of solely a coagulant, solely a flocculant, or the addition of both after each other. As a coagulant is generally added before a flocculant, the first set of jar test experiments was done with solely a coagulant to determine what the ideal dose was. In the following round of experiments, the same ideal coagulant dose was added to all of the beakers after which an increasing dose of flocculant was added.

For the addition of the coagulant the stirrer was set to the 'flash mix' value of 140 rpm for 60 seconds. After the predetermined 'flash mix' duration, the mixing speed was reduced to the flocculation or 'slow mix' value: 80 rpm for 10 minutes. This was also when the polymer was added during the second round of experiments. Lastly, two minutes of really slow mixing at 30 rpm followed after which the flocs were allowed to settle for 10 minutes. Samples of the supernatant and of the flocs were extracted for analysis. The lowest dose of conditioning agents that achieved satisfactory results is the dose used to treat the wastewater from this point on in the research.

The addition of polyaluminiumchloride will lower the pH of the wastewater. During conventional jar test experiments this drop in pH is compensated for by adding enough NaOH to the sample to keep the pH at the same level as it was before the addition of the coagulant. This is done by bringing the pH up to the level of the original sample in all of the beakers, after the different dosages of coagulant are added. The amount of added NaOH is then noted and the samples are discarded. Then in the next run, the NaOH is added to the beakers before the coagulant is added, so that the pH will be the same in all of the beakers after the addition of coagulant. However, as the pH was still in the range of sweep coagulation after the decrease due to coagulant dosing, it was thought that the effect of decreasing pH was insignificant for the coagulation step. Also, the addition of NaOH would mean an extra step for the treatment system, so to simulate the real life scenario the addition of NaOH was neglected.

Dewatering

The dewaterability of the flocculated sludge was measured using a bench scale chamber filter press: the Mareco laboratorium minipers MMP-3. Figure 3.2 gives an overview of the filter press. It has a pneumatic time delayer, which starts counting when the selected pressure is reached. For all of the experiments the specified time under pressure the maximum value of 999 seconds.

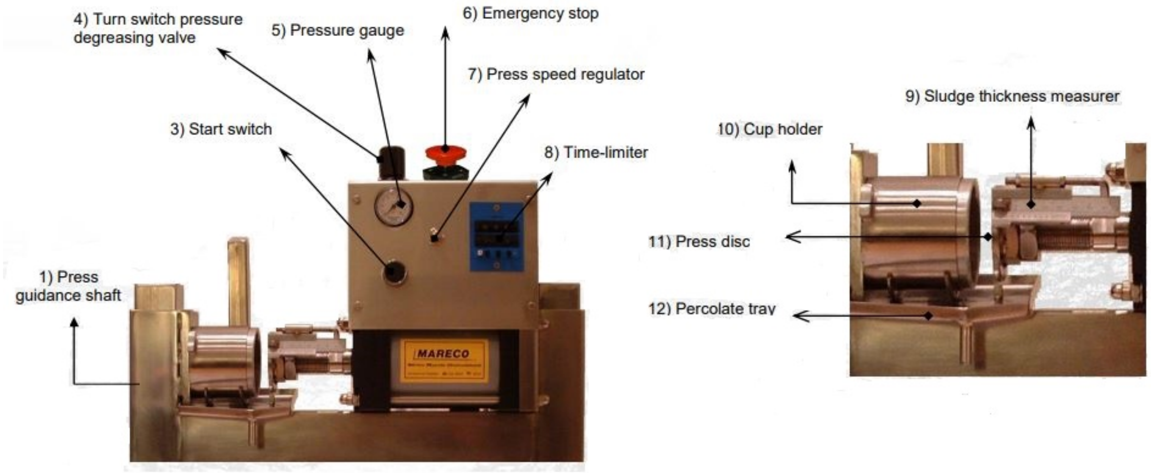

Figure 3.2: *Overview of the Mareco chamber filter press and its components*

The filter press can not function properly without good flocculation of the solids that are fed into the machine. An unconditioned wastewater sample will not dewater properly and has a chance to be pressed through the sides of the cup and the elastic band of the filter. That's why, depending on the experiment, a certain amount of coagulant and polymer was added to the wastewater sample. When a certain amount of blackwater is specified in a experiment in this report it means that the total amount of solids present in this sample added to the filter press. It was not possible to add the whole sample, including the water, as the filter cell only had a volume of 250 ml and most experiments were conducted with samples containing 1000 ml or more.

In a regular filter press experiment, when big and settling flocs were achieved, the solids were first allowed to settle in the beaker. After this, the supernatant was poured of, by hand, until most of the water was gone. Then the watery solids were added to the filter cell and the filter cell was assembled as depicted in Figure 3.3. When flocculation was insufficient or when the flocs did not settle enough the sample was poured on a very fine sieve to extract the solids and release most of supernatant water. Next, the solids were added to the filter cell as described below. The cake that formed between the filters was put in a 105 degrees Celsius oven for 24 hours and the weight of the dried solids was compared to the wet weight (%) to determine the dewaterability of the sample.

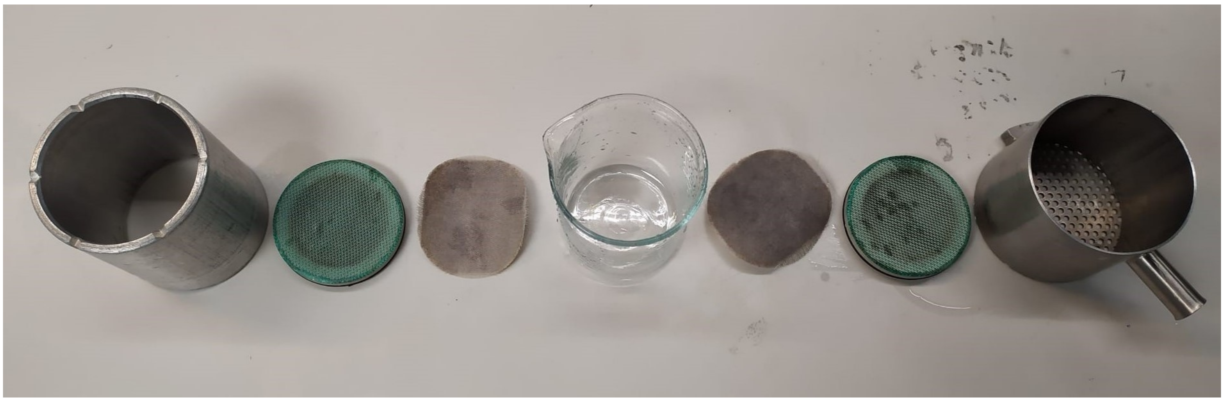

Figure 3.3: Overview of the Mareco chamber filter press and its components. From left to right: beaker (with grooves upward to prevent spillage), filter disc, support filter, sample, support filter, filter disc, beaker holder

Collimated beam

The collimated beam apparatus was used as a method to establish a dose-response relationship for UV-disinfection of the two fecal indicators on a laboratory scale. Figure 3.4 gives a schematic overview of the experimental setup that was used. It consists of an enclosed chamber where part of the output of a set of UV lamps is directed through two successive concentric apertures and down a long collimator onto a horizontal surface. Six parallel low pressure TUV TL-D 15W UV-C lamps by Philips were used in an enclosed and ventilated segment on top of the machine. The collimator room, perpendicular to the UV lamp source, was made of non-reflective material. However, due to some dispersion the beam is never truly collimated, which has to be considered if long water path lengths are used (Bolton & Linden, 2003). A manually controlled shutter was used for shutting off the UV radiation from collimated tube to regulate the time of exposure on the sample. In order to keep the sample homogeneous, it was placed on top off a magnetic stirrer. However, the stirring must be done without creating a "vortex" to ensure a horizontal plane of incidence, so the speed was carefully controlled and the stir bar was small. The UV-C irradiance was measured with a Radiometer system RM-22 with USB cable, UV-C sensor, power supply and software, connected to a laptop. The calibrated plane of the radiometer detector must be placed at exactly the same height as that of the top of the water during UV exposure for proper irradiance measurement (Bolton & Linden, 2003). Safety is very important when working with UV equipment, so at all times during the experiments UV protective glasses, gloves and a lab coat were worn.

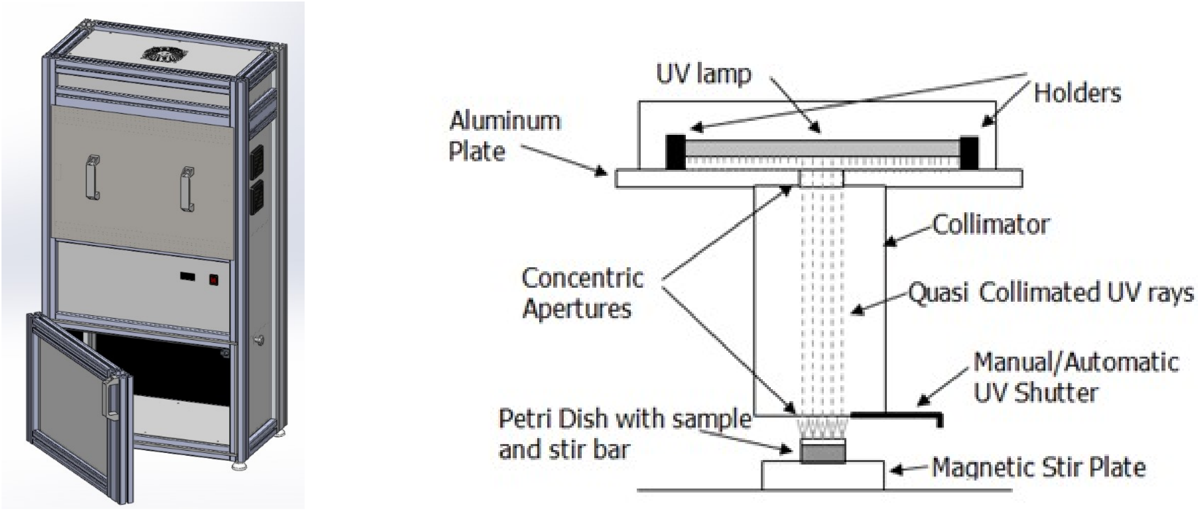

Figure 3.4: *Overview of the collimated beam setup designed, constructed and tested by Dr. Gröbel UV-Elektronik GmbH*

Before conducting the dose-response experiments the lamps inside the machine were turned on to preheat for a period 15 minutes, while the shutter closed off the bottom chamber of the machine from harmfull UV-C light. A sample was prepared according to previously discussed methods, and 80 mL of the filtrate was put into a petri dish with a diameter of 6.6 cm. This means that the irradiated surface of the sample was 34.21 cm^2 and the depth of the water column was 2.34 cm. Next, the petri dish was placed on top of the magnetic stirrer right next to the UV-C meter, at the center of the opening where the collimated UV rays come through. Stirring was initiated at least ten seconds before the shutter was opened to ensure a homogeneous sample. To start the disinfection experiment, the shutter was opened. In standard UV-C dose response procedures, samples are tested for a dose of 0, 5, 10, 20, 30, 40, 60, 80 and 100 mJ/cm^2 . When the specific dose is reached, the sample is collected from the bottom chamber and a new sample is put in. This procedure was repeated in triplets until a sufficient dose response curve could be made.

While conducting the experiments, it became clear that the irradiance at the bottom of the machine was not strong enough for proper disinfection of some of the wastewater samples. For these samples, the ingenious solution was found to put them a level higher in the machine, inside of the collimator room. This solution is far from ideal, as the UV rays in that chamber are still only "quasi collimated" (Figure 3.4), but it was the only way in which a higher irradiance on the surface of the sample could be achieved.

Particle size distribution

The particle size distribution (PSD) of the samples was measured by a particle size analyzer (Bluewave, Microtrac, Germany) with Microtrac FLEX 11.1.0.2 software and a set flow rate of 20%. It measures particle size distribution through light scattering technology with a measuring range from 10,7 nm up to 2000 µm. The device is generally used for wastewaters, so the samples do not need to be diluted before analysis.

Microbial analysis

For this research only two species of fecal indicators were important, namely *E. coli* and *Intestinal enterococci*. The assessment of these species was done using culture based analysis. Hereby, bacteria are cultivated on a selective medium containing nutrients that are specific to a bacteria to enhance microbial growth. The culture plates used for the determination of *E. coli* contained Chromocult coliform agar and cultivates *E. coli* as violet and dark blue colonies. The pink and red colonies are from coliforms and the white are other groups, but they are not important for this research. Cultivation was done by incubating the Chromocult plates at 37 degrees Celsius for 24 hours. For the determination of *Intestinal enterococci*, Slanetz and Bartley medium is used, which cultivates *Intestinal enterococci* as pink and red colonies and develops no other bacteria. Cultivation of Slanetz and Bartley was done by incubating the plates at 37 degrees Celsius for 40 to 48 hours. Figure 3.5 shows how both plates will look when they contain a countable amount of colonies.

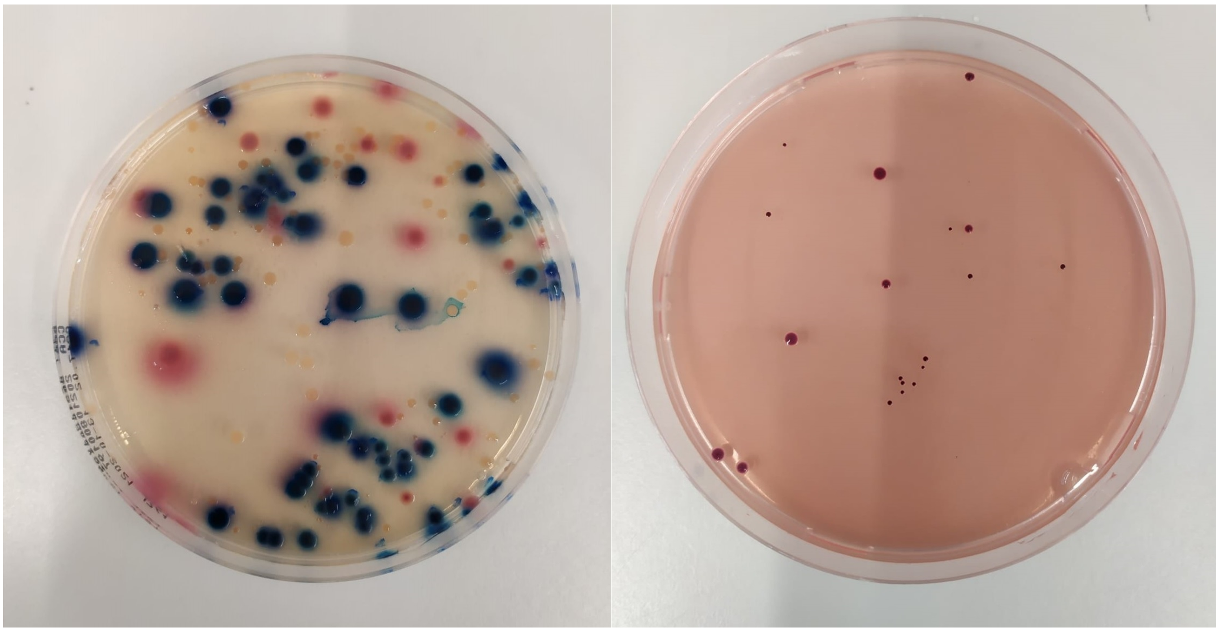

Figure 3.5: *The left plate shows how a typical Chromocult coliform agar plate is going to look and the right how a Slanetz and Bartley plate is going to look, after application of a small amount of wastewater*

The cultivation plates indicate one colony forming unit (cfu) with a coloured dot, which is counted to determine the total cfu/100 mL. As the plate is relatively small, too many colonies will make it uncountable. To determine the colonies in pathogenic dense samples a method called "serial dilution" is applied. As diluting agent phosphate-buffered saline (PBS) is used, which maintains a constant pH. Half a mL of the sample is added to 4.5 mL of PBS aiming to dilute the sample by a factor 10^{-1} for every performed dilution step. The sample needs to be diluted until the amount of countable cfu is between 30 and 300 . This is because below 30 cfu there is an increasing effect of randomness and above 300 the plates become harder to count and thus more prone to errors (Breed & Dotterrer, 1916).

To prepare the plates, 0.1 mL of pure sample or serial dilution was extracted with a pipette and placed on the agar. Exceptionally, a larger amount was extracted, but the amount that is placed on a plate should never be larger then 0.2 mL as it will not be absorbed. After the sample is pippeted on the plate a plate spreader is used to ensure an even distribution of sample. Sometimes during disinfection experiments, a larger amount than 0.2 mL is needed to ensure that the cfu on the plate are above 30. In these cases the samples were filtered through a 0.45 μm membrane filter, after which the filter was put on the agar plate facing upwards. The amount of the filtered sample varied depending on the stage of the experiment. All of the experiments were conducted in triplicates for certainty reasons, thus for every serial dilution three plates were prepared.

Wastewater quality parameters

The following water quality parameters were tested for during the course of the conducted experiments:

- The absorbance of the samples was measured with a spectrophotometer (GENESYS 10S UV-Vis) set at wavelengths of 254 nm and 475 nm. Absorbance at a 254 nm wavelength gives an indication of the NOM and also the penetration depth of UV-C light in the sample. 254 nm light will not pas through a normal plastic cuvette so a quartz one is used. The spectrophotometer could not measure samples with a higher absorbance than 3.0 so some samples needed to be diluted. The 475 nm wavelength was measured to quantify the colour of the samples.

- The pH was measured by using the inoLab_IDS multimeter with the WTW pH-Electrode Sentix 940 probe. It was important to know the pH when conducting the coagulation and flocculation steps as it has a major influence on these.

- The turbidity is a measure for the cloudiness or haziness of a liquid caused by suspended particles. The particles scatter the incident light, and the liquid loses its transparency. The changes in turbidity due to the different treatment steps have been closely observed as they give an indication of the amount of clarification achieved by the experiments. Furthermore, it gives an indication on the effectiveness of UV-C as more cloudiness will lead to more absorbance and scattering of the incident light rays. The turbidity was measured using a TB1 portable turbidimeter, which gives it in Nephelometric Turbidity Units (NTU).

- TS, and TSS of samples were measured based on standard methods (APHA, 2017).

4. Results and interpretations

4.1 Wastewater quality

Table 4.1: *Raw wastewater parameters of the two different samples. The parameters on both samples have been gathered throughout the whole research spanning a total of six months, wherein many different batches of both samples were collected, to give an approximation of general values for both wastewaters*

	Yacht sample		**Camper sample**	
	Mean values	*Std. dev.*	*Mean values*	*Std. dev.*
pH (-)	7.3	0.12	9.24	0.14
Turbidity (NTU)	1322	632	916	389
TS (g/l)	1.46	0.69	12.28	0.76
TSS (g/l)	0.87	0.84	7.63	0.79

The unfiltered wastewater quality parameters of the two different samples are shown in Table 4.1. Quality parameters like total phosphorous and total nitrogen are not of importance for this research so they have been left out of the experimental procedure. These results indicate that both of the wastewaters are distinctly different from each other. The main difference between the unfiltered samples seems to be the TS and TSS fractions, and the pH. The difference in TSS also became evident when experimenting on both of the samples, as the camper wastewater needed a notably higher dose of both coagulant and polymer for the same degree of flocculation as the yacht sample. More negatively charged particles in the suspension meant that more coagulant was needed to destabilize them, and subsequently more polymer was needed for the inner particle bridging mechanism to form larger flocs.

Besides the fact that the unfiltered samples differ distinctly from each other, the TS, TSS and turbidity also differed in the same samples between each sampling moment. The standard deviation of some of the mean values reflects this well, especially when looking at the values for turbidity, where the standard deviation is equal to almost half of the mean values in both samples. This deviation had an impact on the amount of conditioning agents that were required for sufficient flocculation (In section 5.1 the implications that this will have on a treatment system are interpreted). There are several possible explanations for the deviations in turbidity between sampling days, the first of which linking it to the non-homogeniety and high variability in substances of the provided wastewater. Such problems are expected with raw and complex wastewater with high amounts of solids in it. Another possible explanation for the big deviation is the amount of water used by the specific toilet system and the amount of urine in the wastewater, as more water and urine will lower the turbidity, TS and TSS. This of course is arbitrary as the amount of times that one goes to the toilet only to urinate is not equal to the amount of times one goes to defecate, and every day this amount will also be different from the previous day.

4.1.1 Difference in pH between the two samples

Another characteristic on which the two wastewaters differed greatly was the pH, where the pH of the camper sample was almost 2 degrees higher than the pH of the yacht sample. The mean value of 7.3 for the pH of the yacht sample is, according to literature, in the general range of raw sewage (Metcalf & Eddy inc., 2002). However, the pH values of the samples from the camper were notably higher then even the high general pH range of 8 for raw wastewater (Metcalf & Eddy inc., 2002). These high pH levels have a major effect on the coagulation and flocculation processes. A possible explanation for this high pH value in the camper samples is the formation of ammonia (NH_3) from urea ($CO(NH_2)_2$). Ammonia in the wastewater is formed by the hydrolysis of urinary urea and is catalyzed by microbial urease that is present in human feces. The enzymatic decomposition of urea into carbonic acid (H_2CO_3) and ammonia is initiated when urine and feces contact one another after being excreted (Dai & Karring, 2014):

$$CO(NH_2)_2 \xrightarrow{\text{Urease}} H_2CO_3 + NH_3$$

In aqueous solutions, the carbonic acid and ammonia generated from urea hydrolysis are in equilibrium with bicarbonate (HCO_3^-) and ammonium (NH_4^+) ions, respectively. Consequently, urea hydrolysis is associated with a subsequent increase in pH (Mobley et al., 2021), which explains the high pH of the sample. This reaction is also a major problem in for instance the livestock sector.

4.1.2 Turbidity

It is noteworthy that the mean turbidity values of the samples from the yacht are higher than the sample from the camper, while the TS and TSS are lower. However, this data must be interpreted with caution because turbidity units have no inherent value. Turbidity is mainly caused by the TSS in the suspension, but as it depends on many more parameters, it is not an accurate measurement for TSS. It is a qualitative, rather than a quantitative, measurement (YSI Incorporated, 2001). For instance, the value for turbidity may be greatly influenced by a residual haze, which may represent an extremely small proportion of the original solids. There is not a standard conversion between various turbidity units (NTU or FNU) and quantative mass measurements (mg/l) (Campbell Scientific Inc, 2014), which is why turbidity data must always be taken in context (YSI Incorporated, 2001). Also, the turbidity of water is based on the amount of light scattered by particles in the water column. The more particles present in the water, the more light that will be scattered (Perlman, 2014). Wastewater from the yacht was homogenised by a high velocity macerator for 10 minutes before it was sampled, while the samples from the camper only passed through the macerator once. This would mean that the amount of smaller particles in the samples from the yacht could have been a lot higher, which explains the higher value for turbidity even though the amount of TSS in the camper samples was greater.

4.2 Coagulation and flocculation

Before the actual jar tests were conducted, first a series indicative tests have been carried out. This was mainly to test the effectiveness of the selected coagulants and polymers and to get a sense of the range in which they work most efficiently on the samples. These tests had shown that the coagulation results of PAX-HP800 were not satisfactory, and it was therefor not included in the rest of the study. Both PAX-18 and C-2230 showed good coagulation and flocculation results in the indicative testing phase, and a rough estimate could be made on the range in which they worked most effectively. These are also the conditioning agents that were used in the rest of the research.

4.2.1 Coagulant based jar test

The first jar test was performed to determine the optimal dosage of coagulant. Figure 4.1 shows the results after ten minutes of settling time and Table 4.2 gives the relevant parameters. Sample IV with 2.0 ml/l PAX-18 had the best settling abilities after ten minutes. The turbidity of the supernatant decreased substantially from 322 NTU with 0.5 ml/l PAX-18, to 43 NTU with 2.0 ml/l PAX-18. However, the addition of 2.5 and 3.0 PAX-18 reduced the turbidity only slightly more till 41 and 38 NTU respectively. This would indicate that 2.0 ml/l is roughly the ideal dose of PAX-18 for this kind of blackwater. The pH of the supernatant increased gradually from 7.29 to 6.54 due to the increasing PAX-18 dose.

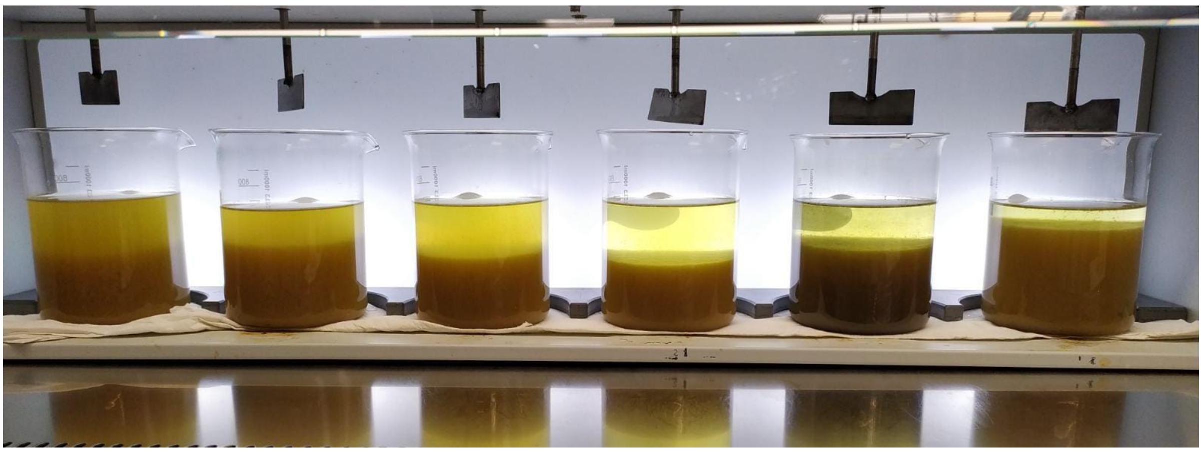

Figure 4.1: *Jar test to determine the ideal PAX-18 dose on samples from the vacuum toilet of the yacht, after 10 minutes of settling. The respective PAX-18 dosages from left to right were: 0.5, 1.0, 1.5, 2.0, 2.5, and 3.0 ml/l*

Table 4.2: *Parameters regarding the jar test from Figure 4.1, the initial pH of the sample was 7.32 and the initial turbidity was 1699 NTU*

Sample	I	II	III	IV	V	VI
PAX-18 dose (ml/l)	0.5	1.0	1.5	2.0	2.5	3.0
Turbidity SN (NTU)	322	167	97	43	41	38
pH SN (-)	7.29	7.10	6.98	6.77	6.62	6.54

The values for pH were all in the range of sweep flocculation, which can be seen in the Amirtharajah diagram (Figure 2.3). As described in subsection 3.2.3 the pH of the wastewater was not adjusted to the coagulant dose, and this was also not necessary due to the pH being in the sweep flocculation region in all of the samples. As shown on the diagram, the boundaries of the different regions of destabilization vary with different suspensions, both because of the particle concentration and because of the surface charge density. This means that the aluminium dose on the y-axis is only an indication, and can vary depending on the total amount of solids in the wastewater, which explains the large quantities of coagulant used on the wastewaters of this research compared to the Amirtharajah diagram.

Both the turning point and the beginning of the mechanism of sweep flocculation were clearly visible in this first jar test. It can be seen that most of the solid material has precipitated in the first three jars, but the supernatant is still murky with colloidal material in it. On the other hand, the supernatant in the fourth jar was notably clearer. It still had a relatively high turbidity of 43 NTU, but it was easy to see through. The turbidity of the supernatant also did not change substantially when more PAX-18 was added. The small flocs that were formed in the fourth to the sixth had a whitish colour and were "fluffy", which is a feature of the formation of hydrous aluminium oxides (Benjamin & Lawler, 2013).

4.2.2 Polymer based jar test

The next jar test was carried out with addition of only the polymer C-2230, but in different doses. Figure 4.1 shows a picture of the two jars that yielded the best results. The flocs that formed were relatively strong and compact, but the solids would stick to the sides of the glass. The supernatant was cloudy, and it was not possible to see through it in the beaker, no turbidity was measured.

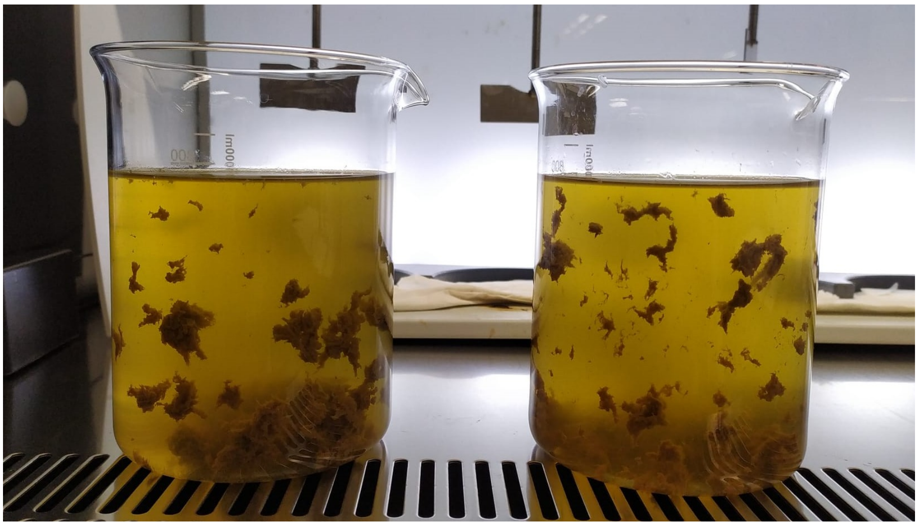

Figure 4.2: *Jar test with the addition of solely flocculant on samples from the vacuum toilet of the yacht. Respectively, dosages of 0.25 ml/l and 0.3 ml/l were used*

A reason that the sole addition of C-2230 did not provide satisfying results could be because it is a cationic polyacrylamide with a low relative charge (Kemira, 2015). Generally, cationic polyelectrolytes should act in two ways, namely charge neutralization, together with inter particle bridging (Ravina, 1993). However, due to the low relative charge of the specific polyelectrolyte C-2230, the charge neutralisation characteristics are also low. This could mean that the negatively charged colloidal material in the wastewater, which adhered to the polyacrylamide chains, create a mean negative surface charge of the larger sludge flocs, as the reduction in the magnitude of the zeta potential was not large enough. These sludge flocs, after conditioning with only C-2230, would then be able to adhere to the glass and are hard to extract (see Figure 4.2). Another possible explanation for the ineffectiveness of the dosing of solely C-2230 is that due to a lack of solids in the wastewater, the polymer does not have enough solids to attach to. As described in section 2.1, when the long chains do not bind enough particles on them they can wrap around the particles or flocs, which restabilizes them and gives them a large total negative charge. A solution to this problem could be to dose a polymer with a higher charge density. When a polymer with a higher charge density would be used, the reduction in the magnitude of the zeta potential of the formed flocs is also higher. These flocs would than have a total surface charge that is less negative than the flocs in the experiment of Figure 4.2, which would make them less "sticky". It should be noted however, that it is hard to say something about the charge neutralization characteristics of C-2230, as Kemira keeps the real formulas for their polyelectrolytes a secret and the charge neutralization is only categorized as "low" on their technical data sheets.

4.2.3 Jar tests with both a coagulant and a polymer

The consistency of the flocs from both the coagulant and the polymer based jar tests did not seem to be sufficient for the purpose of this study. The flocs that were formed in the coagulation jar tests were to weak and small to be filtered out, without passing through the membrane of the filter cell. When attempts were made to filter these flocs in the chamber filter press, the filtrate still contained many solids. The coagulated flocs would pass through the filter cloth and also through the sides of the filter cell. The flocs from the polymer based jar tests were to sticky due to their negative charge. It was difficult to extract them from the glass beaker and they would adhere to everything that they touched. The supernatant liquid was also still murky which would have a negative effect on the UV-disinfection step.

For these reasons, in the next two jar tests, a combination of the two was chosen. The first jar test, shown in Figure 4.3, was conducted by first adding the ideal PAX-18 dosage of 2 ml/l, which was determined in the first jar test (see Table 4.2). After which, an increasing dose of C-2230 was added to the beakers. Table 4.3 shows the important data regarding this experiment. It's evident that the pH is not affected by the addition of C-2230, as it only decreased 0.04 over the whole range of C-2230 addition. The turbidity seems to remain approximately the same value at dosages 10, 15, 20 and 25 ml/l, but increased from 24.8 NTU to 38.3 NTU after the dosage of polymer is increased from 25 to 30 ml/l. However, even with the small increase, the turbidity is still lower then when only PAX-18 was dosed. In Figure 4.3 the structure of the flocs is somewhat visible. The left beaker, with the lowest dose of C-2230, showed small, non ideal flocs, which were not interconnected. The flocs in the beaker with the highest dose of C-2230 were large and firm, and they were almost all part of the larger floc structure.

Figure 4.3: *First jar test to determine the ideal C-2230 dose when the samples are pre-treated with an ideal PAX-18 dose of 2.0 ml/l with samples from the vacuum toilet. The uppermost picture was during the mixing phase and the bottom picture was after two minutes of settling. The respective dosages of C-2230 from left to right were: 10, 15, 20, 25, 30, and 35 ml/l*

Table 4.3: Parameters regarding the jar test from Figure 4.3. The initial pH of the sample was 7.36 and the initial turbidity was 2827 NTU

Sample	I	II	III	IV	V	VI
PAX-18 dose (ml/l)	2.0	2.0	2.0	2.0	2.0	2.0
C-2230 dose (ml/l)	10	15	20	25	30	35
Turbidity SN (NTU)	21.8	25.5	24.4	24.8	38.3	35.5
pH SN (-)	6.69	6.67	6.67	6.69	6.68	6.65

The next jar test was performed to determine the required dose of C-2230 for satisfying floc formation, when a less ideal quantity of PAX-18 is used. The jar test is shown in Figure 4.4, and its corresponding data in Table 4.4. The floc formation was distinctly less dense than Figure 4.3, as all of the beakers showed smaller flocs that were not interconnected. At the lower quantities of C-2230, namely 10 and 15 ml/l, the turbidity of the supernatant increased to 66.4 and 53.5 NTU respectively, and the supernatant became noticeably murkier. The turbidity of the supernatant after 2.0 ml/l PAX-18 and 35.5 ml/l C-2230 addition is virtually the same as after 1.0 ml/l PAX-18 and 35.5 ml/l C-2230 addition.

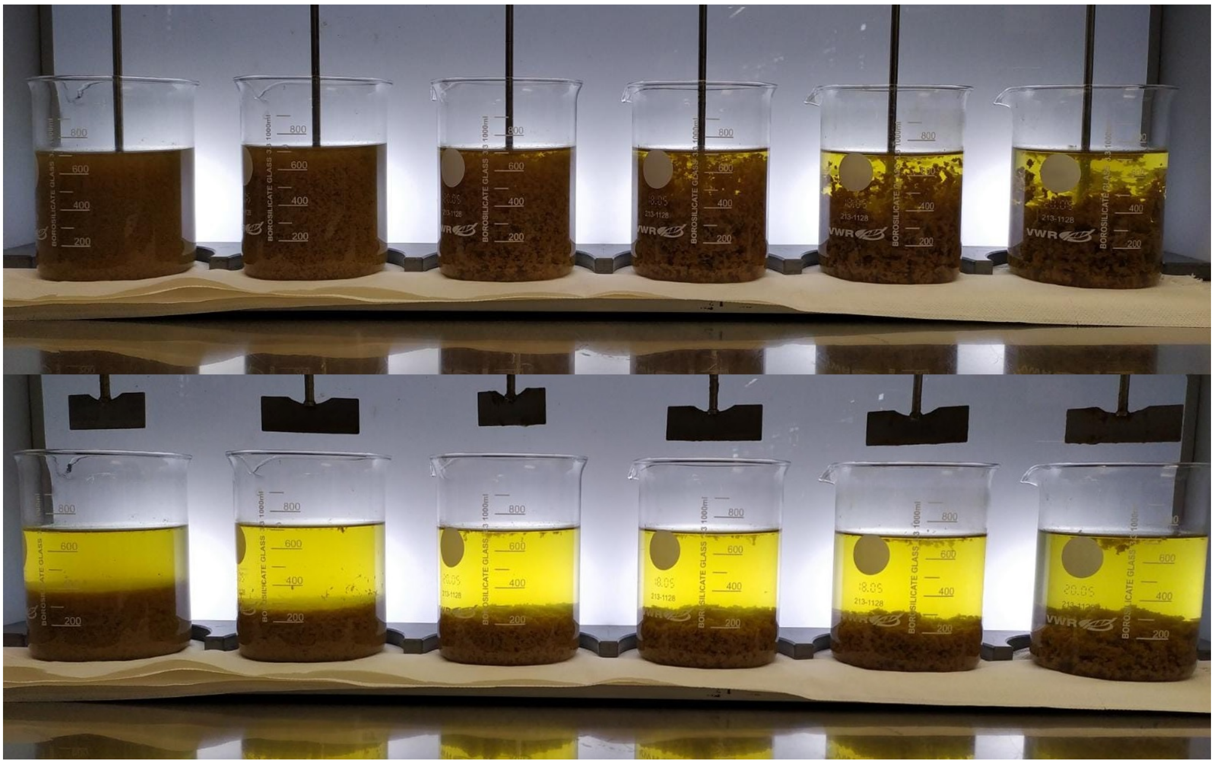

Figure 4.4: Second jar test to determine the ideal C-2230 dose when the samples are pre-treated with an less ideal PAX-18 dose of 1.0 ml/l on samples from the vacuum toilet. The uppermost picture was during the mixing phase and the bottom picture was after two minutes of settling. The respective dosages of C-2230 from left to right were: 10, 15, 20, 25, 30, and 35 ml/l

Table 4.4: *Parameters regarding the jar test from Figure 4.4. The initial pH of the sample was 7.42 and the initial turbidity was 2860 NTU*

Sample	I	II	III	IV	V	VI
PAX-18 dose (ml/l)	1.0	1.0	1.0	1.0	1.0	1.0
C-2230 dose (ml/l)	10	15	20	25	30	35
Turbidity SN (NTU)	66.4	53.5	41.6	43.2	37.4	36.2
pH SN (-)	6.99	6.99	6.98	6.98	6.97	6.97

Mixing conditions

During the execution of the jar tests, different mixing conditions were experimented with on the coagulation and flocculation processes. However, the variations in mixing speed and time did not seem to have a substantial effect on the coagulation and flocculation processes. The only thing that seemed to be significant was that a short phase of relatively rapid mixing was initiated after the addition of the coagulant so that the aluminium can react and form small flocs, after which a slower but lengthier period of mixing needed to be initiated for the addition of C-2230. The exact speed and length for these phases proved to be of no importance for the floc formation. These results tie well with previous studies wherein rapid mixing conditions made a difference only in the adsorption-destabilization zones. In the zone where sweep flocculation dominates, there was no measurable distinction between different intensities of mixing (Amirtharajah & Mills, 1982).

Alkalinity

The amount of alkalinity in the wastewater is another important characteristic for the initiation of sweep flocculation, as it reacts with aluminium to form hydrous aluminium oxides. Without alkalinity, the pH will be lowered to the point where soluble aluminum ions (Al_3^+) are formed instead of aluminum hydroxide, which are ineffective coagulants (Ravina, 1993). This would mean that no sweeping of the sollids would take place and that the removal of the solids would be not sufficient.

As the molar mass of PAX-18 is approximately 300 g/mol, and the aluminum content is 9%, the total amount of alkalinity that is used for sweep flocculation is about 0.08 g $CaCO_3$ per g of PAX-18 (Kemira, 2017). However, this calculation is a rough estimate as the real formula for the polyaluminium chloride solution is kept a secret by Kemira. Unfortunately, the alkalinity was not measured during this research, however the sweep flocculation was achieved in both samples which means sufficient amounts were present in the solutions.

4.2.4 Removal of fecal indicators

Lastly, the experiment depicted in Figure 4.5 was conducted to determine the effect of both the coagulant and the polymer addition on both fecal indicators. As almost all the solid material will be filtered out, the unbound free suspended bacteria in the solution are of interest for this study, which is why the supernatant of the samples was used for this analysis. Figure 4.5 shows that a higher quantity of PAX-18 will decrease the amount of indicator pathogens in the supernatant. An addition of 2 ml/l PAX-18 achieves a 1.5-log removal of *E. coli* and a 2-log removal of *Intestinal enterococci*, and an addition of 4 ml/l PAX-18 achieves at least a 2-log removal of *E. coli* and a 3-log removal of *Intestinal enterococci*. The increasing quantity of C-2230 does not seem to have an effect on the removal of fecal indicators. When the total pathogens at 2.0 ml/l of the left graph are compared to those at 10, 20, 50, and 80 of the right graph, no substantial changes in removal can be distinguished.

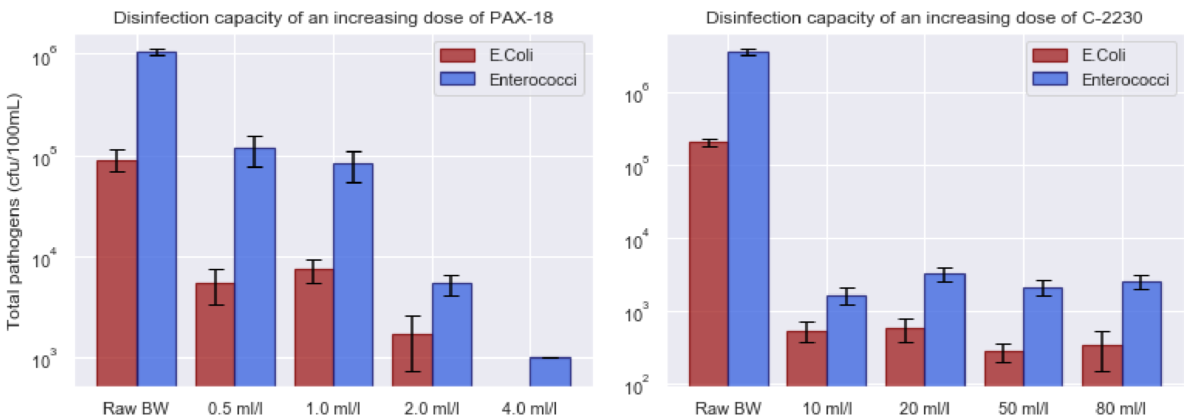

Figure 4.5: *The left chart shows the effect that an increasing quantity of PAX-18 has on both fecal indicators in the supernatant. The right chart shows the effect on both fecal indicators when 2 ml/l PAX-18 is added to all samples, and an increasing dose of C-2230. The same sample from the vacuum toilet was used for both experiments*

As most bacteria, including *E.Coli* and *Enterococci*, carry a net negative surface charge, adhesion of these bacteria takes palace on positively charged surfaces (Gottenbos et al., 2001). This means that these pathogens can be partly removed by destabilization, and even more effectively by sweep flocculation, using Al^{3+}. The results shown in Figure 4.5 provide evidence of this removal, as it can be clearly distinguished that an increasing PAX-18 has a substantial effect on the total amount of fecal indicators in the supernatant. A small amount of removal took place with minimal dosages of PAX-18, but remarkably better results were obtained with the higher dosages. Thus, to effectively entrap the pathogens in the flocs, higher coagulant concentrations should be dosed.

Additionally, it is noteworthy that the removal of fecal indicators seems to happen mainly due to the addition of a coagulant, rather than the addition of a flocculant. The data for the 2.0 ml/l addition of PAX-18 in the left graph can be compared with the right graph in Figure 4.5, where each entry was coagulated with 2ml/l, and no difference can be distinguished. From these results, it appears to be clear that the increased dosage of C-2230 does not effect the total amount of pathogens in the supernatant. If no PAX-18 and solely the C-2230 is dosed, it would of course have a measurable effect on the pathogens, as it would remove most of the solid material, which has the highest bacterial density. However, this effect is not greater than the effect that a coagulant has, as the sweep flocculation will not only remove part of the fecal indicators by precipitating the solid material but will also entrap free-floating bacteria from the supernatant in the floc structures and adhere them onto the sweep flocs.

Further research is strongly encouraged, in order to determine an ideal dosage of PAX-18 for the removal of fecal indicators from the wastewater. The values for *E.Coli* at 4.0 ml/l in the left graph seem to be zero, but this was due to a too small an amount of supernatant that was put on the culture plates. At these high quantities of coagulant it would be wise to use a 0.45 micron filter and to filter 1 to 10 ml of supernatant to get a good estimation of the amount of bacteria per 100 ml.

4.3 Dewatering

In this subchapter the results of the different dewatering experiments are reported. Almost all of the data that is shown in this chapter is obtained from the Mareco chamber filter press, and by using the standard method for TS determination afterwards (APHA, 2017). Table 4.5 shows the results that were obtained after using the press for the first series of tests. Two different quantities of PAX-18 were used, together with an increasing quantity of C-2230. The dry solid content after applying the pressure to the cake was approximately 30% for all of the samples.

Table 4.5: *Results from pressing the sludge cakes at 6 bar. For these tests samples from the vacuum toilet of the yacht were used*

	PAX-18 dose (ml/l)	**C-2230 dose** (ml/l)	**DS content** (%)
Sludge from 1L of BW	1.0	20	28.2
	1.0	25	27.0
	1.0	30	29.0
	1.0	35	31.3
Sludge from 1L of BW	2.0	20	30.3
	2.0	25	29.7
	2.0	30	30.2
	2.0	35	29.8

In hindsight, these first dewatering experiments were not executed properly. The wastewater from the vacuum toilet were used, which had a low TSS content (Table 4.1). Also, the solid material of only one single liter of this blackwater was inserted in the filter cell, and together with a pressure of 6 bar all of the sludge cakes were flat as a dime. This made it impossible to distinguish any effect of an increasing dose of coagulant or polymer on the dewaterability. However, what this experiment did show is that the maximum amount of dry solids that can be reached by the filterpress is around 30%, which was in line with previous studies on dewaterability (Mowla et al., 2013). After this process nearly all of the free water and part of the interstitial water is extracted, and the water that is left in the sludge cake is the bound water, consisting of trapped interstitial, vicinal and intracellular water. This water can only be removed by thermal treatment of the sludge (STOWA, 1998).

The next series of dewatering test were carried out using a larger amount of blackwater, which should also mean more total solids and a thicker sludge cake. Table 4.6 shows the results of the tests. The floc structure of the second sample was substantially better than the first sample, which had many more smaller flocs. Both the third and fourth sample had good floc structures and a clear separation of the free water.

Table 4.6: *Results from pressing the sludge cakes at 6 bar. For these tests samples from the vacuum toilet of the yacht were used*

	PAX-18 dose (ml/l)	**C-2230 dose** (ml/l)	**DS content** (%)
Sludge from 2l of BW	1.0	20	17.3
	1.0	30	27.2
Sludge from 4l of BW	1.0	25	8.1
	0.5	50	9.8

The opposite of what happened in Table 4.5 occurred in the dewatering experiments of Table 4.6. In the last two samples, the solid material from four liters of blackwater was added to the filter cell and this gave dry solids contents of 8.1 and 9.8 %. The sludge cakes in both experiments looked similar to the sludge cakes shown in the right picture in Figure 4.6. The filter cake seemed to have been compressed on both sides of the filter cell near both the filter membranes, but the middle part of the cake still contained a lot of trapped water. The same thing happened when pressing an excess of blackwater from the camper samples , which was less macerated but contained a lot more TSS. The last entry of Table 4.7 shows that the solids of one and a half liters seemed to be too much to dewater for the filter press, even at high pressures. This is common in a suspension with a large PSD that contains small and compressible materials. During the filtration stage of dewatering, an extremely compressible layer is formed at the membrane surface that is highly impermeable and changes the diffusion of solid material into and through the filter cake. Most of the hydraulic pressure drop is located within that narrow region. Work to characterise this process has shown the expected filter cake structure for a compressible material, whereby there is a solid gradient across the filter cake that becomes homogeneous at equilibrium, which forms a relatively dry and impermeable cake near the filter medium (Tiller & Green, 1973; Skinner et al., 2015; Heij & Kerkhof, 1996).

Table 4.7 presents the results of the experiments concerning the relation between applied pressure and dry solid content of the sludge cake. A gradual increase in dry solid content can be observed when the pressure increases. Summing up, a five bar increase from two to seven bar of applied pressure results in an 8% increase in dry solid content of the sludge cake.

Table 4.7: *Results from pressing the sludge cakes with an increasing pressure. For these experiments samples from the camper were used, which had more TS and TSS*

	Pressure (bar)	**Cake thickness** (mm)	**DS content** (%)
Sludge from 1.0 l BW	2.0	17.25	11.1
	3.0	13.0	13.8
	4.0	11.5	13.9
	5.0	8.75	16.3
	6.0	8.75	17.2
	6.0	8.25	19.6
	7.0	8.5	10.8
	7.0	8.0	19.0
Sludge from 1.5 l BW	6.0	28.5	9.6

The dewaterability experiments did not always go as expected. Figure 4.6 shows what happened when an excessive amount of solids are put in the filter cell. The right picture of Figure 4.6 corresponds to the last entry of Table 4.7, where 1.5 l BW is used instead of 1.0 l. A thick cake layer with a lot of water inside was the results of this experiment.

Figure 4.6: *This is a picture taken of a successful dewaterability experiment with a well-formed sludge cake (left) and of a poorly formed sludge cake (right)*

The last dewaterability experiments were carried out to determine whether the amount of polymer that was used to flocculate the sample had a substantial effect on the dry solid percentage in the sludge cake. Table 4.8 shows four results of a sludge with a high dosage of polymer (150 ml/l) and five results of a sludge with a minimal dose for flocculation (70 ml/l). Figure 4.7 shows how the flocculation of both scenarios looked. For all of the experiments the same ideal amount of 4 ml/l of PAX-18 was used to ensure the best floc formation.

Table 4.8: *Results of the experiments to determine the significance of the effect of a high or a low dosage of C-2230 on the dewaterability of a sludge. For these experiments samples from the camper were used, high in TSS*

C-2230 dose (ml/l)	**Cake thickness** (mm)	**DS content** (%)
150	14.75	14.6
150	16.75	12.3
150	15.75	12.7
150	17.25	12.2
70	14	15.3
70	15.25	14.7
70	15	14.4
70	12.25	15.7
70	16	16.0

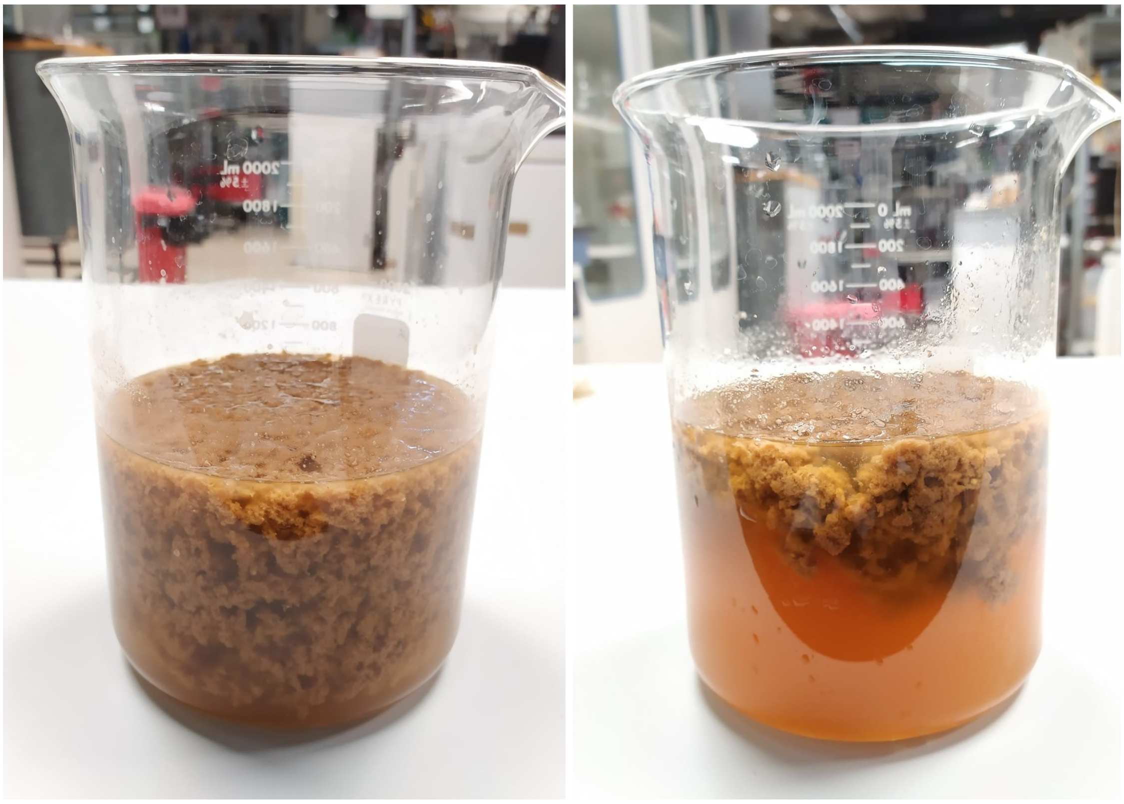

Figure 4.7: *Flocculation with a low amount (70 ml/l) of C-2230 (left) and with a high amount (150 ml/l) of C-2230 (right). Photo was taken of both sludges from the experiment in Table 4.8*

Effect polymer dose on dewaterability

The experiments to determine the significance of the effect of a high and low dose of polymer were carried out at three bar. The reason for this was that three bar was the minimum amount of pressure required to form a somewhat compact filter cake, so it was thought that the difference in dewaterability could best be distinguished at lower pressures. When the difference would only be small, the higher pressure could completely destroy the structure of the flocs and diminish the effect of the polymer dose.

Looking at the data of the experiment in Table 4.8, it seems really hard draw a conclusion on the effect of the polymer dose. The dry solid contents in the cake with the lower dose of polymer is approximately 2 to 3% higher, but whether this is due to the use of polymer is difficult to say. The high variability in the cake thickness, and that this seems to have almost no correlation with the dry solid content, shows that the difference in dewaterability could also be attributed to the inhomogeneity of the blackwater and inconsistency of the sampling method, rather than the use of a polymer.

It must be pointed out that an excessive amount of polymer could have been used in this study, even in the experiments where the dose was categorized as "low". Previous studies describe ideal polymer quantities whose maximum values do not exceed 16 g/kgDS (H. Wei et al., 2018; Almonani & Ormeci, 2014; Yeneneh et al., 2016), while the "low" dose of polymer in this research was equal to 21 g/kgDS. This means that the "high" dose of polymer, which was assumed to be the more ideal dose as it generated the strongest and most compact flocs, exceeded the ideal dose for dewaterability substantially. However, an excessive amount of polymer should not have too big of an impact on dewaterability, as research by STOWA revealed that the free water content was not influenced by an over dosage of polymer (Korving, 2012). Nevertheless, the fact that every sample was overdosed with polymer means that results are unusable for the determination of the effect that a polymer has on the dewaterability.

In future experiments a smaller quantity of polymer should be used to determine an "optimum dose range" in relation to dewaterability. However, there is a problem, as the flocs generated by such a low quantity of polymer are impossible to separate from the free water without high pressure filtration. As the TSS content of the blackwater is too low for the Mareco filter press, other test apparatus would have to be utilized.

4.4 UV-disinfection

In this subchapter the results of the UV-disinfection experiments are depicted and interpreted. These results were all obtained by using the collimated beam apparatus as described in subsection 3.2.3. The radiometer was used to measure the irradiance in both the bottom chamber and inside of the collimator room, the latter of which was located closer to the lamps. Table 4.9 gives the mean irradiance values of two sets of lamps with different power at both levels in the machine, and it can be seen that tthe difference was negligible. This is why the 15 Watt lamps were chosen for the rest of this research, as it is the most conventional practice in this kind of study.

Table 4.9: *The UV-C irradiance in the collimated beam setup that was measured by the radiometer in mW/cm² at the two different levels using 15 Watt and 25 Watt lamps*

	Bottom shelf		Top shelf	
	Mean values	*Std. dev.*	*Mean values*	*Std. dev.*
Six 15 Watt lamps (mW/cm^2)	0.15	2.29e-3	2.84	6.90e-2
Six 25 Watt lamps (mW/cm^2)	0.16	1.06e-3	2.88	6.45e-2

Before the UV-experiments were conducted on the two wastewater samples, a calibration measurement was done on demiwater that contained bacteria. This was done to determine whether the collimated beam performed in accordance to studies found in literature. A stock solution was made using 200 ml of demi water and 400 ul of WR1 *E. coli* culture from the WaterLab. As can be seen in Figure 4.8, an increasing dose decreased the number of colonies on the plates and after 10 mJ/cm^2 the amount of *E. coli* in the water is below the detection limit. More plates of higher UV quantities were made, but none of those plates showed any colonies. It could therefore be concluded that in demiwater, without solids or other organic material and with a turbidity of 0 NTU, a UV-C dose of 10 mJ/cm^2 was enough for satisfying disinfection results.

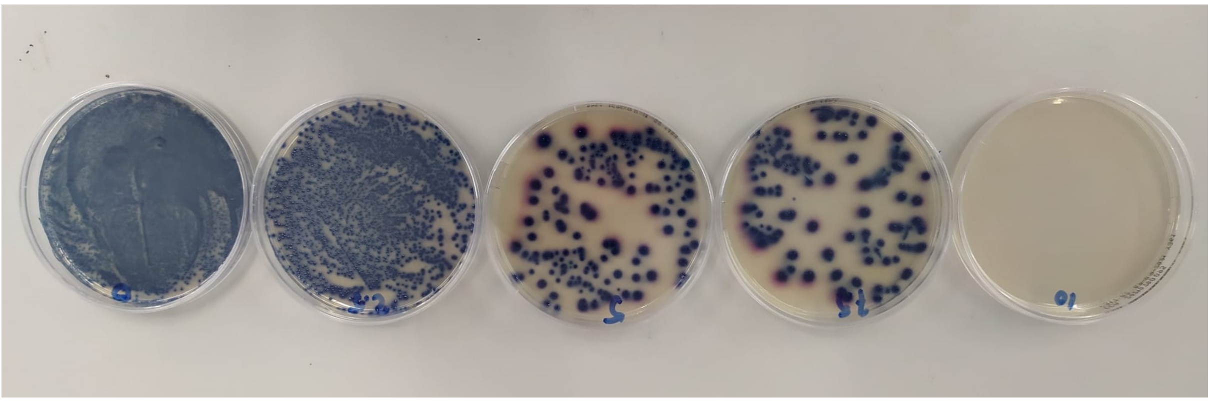

Figure 4.8: *E. coli cfu on plates from the calibration measurement. The UV radiation dose from left to right were: 0, 2.5, 5, 7.5, and 10 mJ/cm² and the sample was placed in the bottom chamber of the machine*

Unfortunately, the dilution rates of the calibration experiment were not properly estimated, which is why there were too many colonies to count on the plates that received a low UV-C dose of 0 and 2.5 mJ/cm^2. However, this does not mean that the experiment was without added value. A clear decrease in colonies can be seen at 5 and 7.5 mJ/cm^2, proving that the collimated beam apparatus is capable of UV-disinfection. Previous studies demonstrated that a dose of 10 mJ/cm^2 is needed in crystal clear demi water to inactivate almost all of the *E.Coli* bacteria (Chang et al., 1985), which would indicate that the collimated beam apparatus is working accordingly.

In contrast to the clear water used for the calibration measurement, the wastewaters used for the UV-disinfection experiments contained solids and organic matter, and had higher turbidity values. Except for the experiment depicted in Figure 4.9, all of the other experiments were conducted on supernatant or filtrates of the wastewaters. Table 4.10 shows some of the characteristic parameters of the filtrates of both of the different wastewaters, obtained by using the chamber filter press. The turbidity's are practically equal, however there is a considerable difference in the absorbance values of UV_{254}. The absorbance value converts logarithmically to the transmittance of UV_{254} into the specific water. The absorbtion of UV_{254} could also be an indication of the humic acid concentration, as there is a good correlation between the humic acid concentration and UV_{254} absorption (mohammadi et al., 2013).

Table 4.10: *Filtrate parameters of the two different samples, after filtration by the Mareco filter press*

	Yacht sample		**Camper sample**	
	Mean values	*Std. dev.*	*Mean values*	*Std. dev.*
Turbidity (NTU)	16.4	12.7	17.68	14.3
UV_{254} (cm^{-1})	1.34	0.06	11.85	0.56

Figure 4.9 depicts the first UV-disinfection experiment, which was conducted on a sample of pure, unfiltered and unconditioned blackwater collected from the vacuum toilet. This was to determine the effectiveness of UV-disinfection, before the coagulation/flocculation and filtration process had taken place.

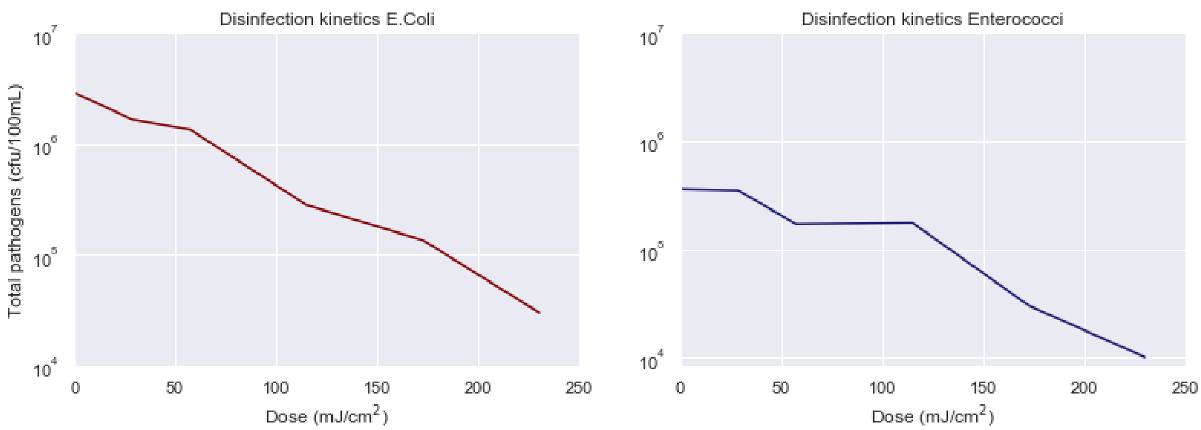

Figure 4.9: *UV-C experiment performed on raw unfiltered BW sample from the vacuum toilet. It had a turbidity of 455 NTU and the petri dish with the sample was placed on the top shelf of the collimated beam apparatus*

It was anticipated that hardly any disinfection would take place in the experiment shown in Figure 4.9, due to the high TSS concentration (870 mg/l) of the raw blackwater. Mainly because research has shown that UV disinfection efficiency decreases with increased suspended solids concentrations and that wastewater solids have high absorbance values (Hill et al., 2002; Emerick et al., 2000; Loge et al., 1999). However, despite the substantial amount of TSS in the raw sample, the inactivation rate of the pathogenic bacteria does not seem to have been reduced by a substantial amount compared to the experiment on the filtrate of the same sample (Figure 4.12). Unfortunately, the experiment was only done up to a total UV irradiance of 230 mJ/cm^2. If higher rates of UV would have been examined, there is a probability that a tailing effect could be seen in the graph, caused by the many particles in the suspension.

The next two UV-disinfection experiments were conducted on the unfiltered supernatant of the different type of samples. The disinfection kinetics of the sample from the vacuum toilet in Figure 4.10 show a decrease of *E. coli* and *Intestinal enterococci* that amounts to almost 1.5log and the 0.5log respectively after a dose of approximately 50 mJ/cm^2. Figure 4.11 shows the disinfection kinetics of the sample from the camper. Here *E. coli* decreases roughly 0.5log after a dose of 80 mJ/cm^2, but the amount of *Intestinal enterococci* seems to not have decreased at all after the same dose was applied.

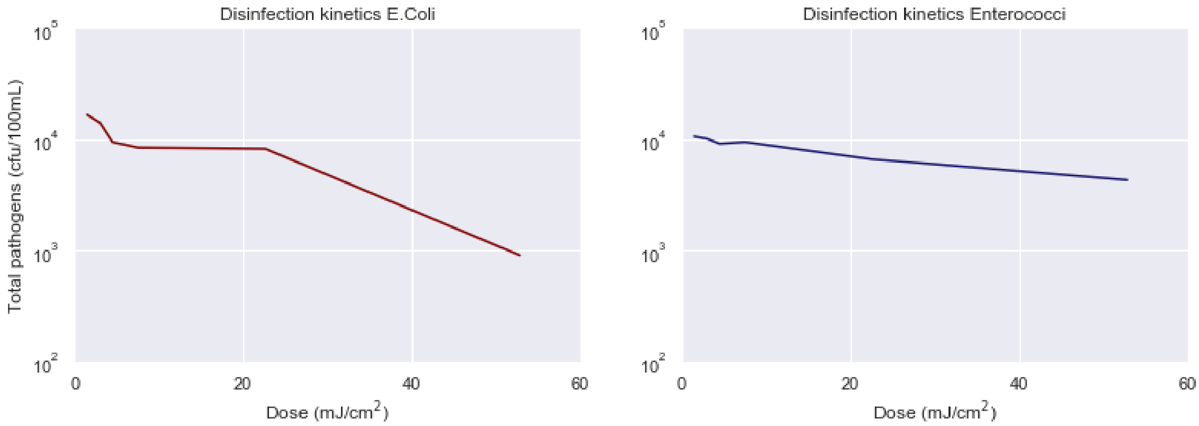

Figure 4.10: *UV-C experiment performed on supernatant from a sample from the vacuum toilet. For flocculation 1 ml/l PAX-18 and 50 ml/l C-2230 was used, and the turbidity of the supernatant was 30.4 NTU. The petri dish with the sample was placed on the bottom shelf of the collimated beam apparatus*

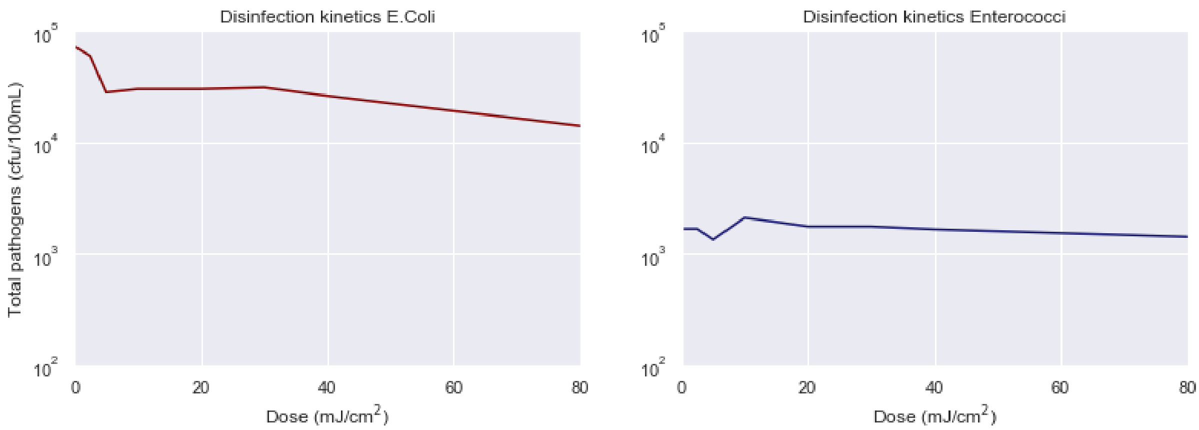

Figure 4.11: *UV-C experiment performed on supernatant from a sample from the camper. For flocculation 4 ml/l PAX-18 and 120 ml/l C-2230 was used, and the turbidity of the supernatant was 36.3 NTU. The petri dish with the sample was placed on the bottom shelf of the collimated beam apparatus*

The experiments showed unsatisfying results for the disinfection of the wastewaters for this total UV-C dose. It is evident that the effective dose of UV light decreases rapidly inside the samples, as the dosage needed for disinfection is considerably higher than in the calibration experiment (with the demi water samples). As almost all of the solid material was removed from the samples, the chance that this rapid decrease in UV effectiveness is attributed to the shading and shielding effects that solids have is unlikely. Previous studies have established that humic acids, which are the main components of humic matter, absorb strongly in the UV range and that this specific light absorption can substantially affect the survival of bacteria exposed to UV-C light (Bitton et al., 1972; Qualls et al., 2015; mohammadi et al., 2013). The higher this absorption, the less the effective dose of UV light is in the sample, and the better the survival of bacteria after UV-irradiation.

Absorbance values and TSS concentrations in the wastewaters that were investigated in this research (see Table 4.1 and Table 4.10) were considerably higher than what has generally been studied in previously conducted research. Most research on UV-disinfection of "low quality" wastewater has focused on secondary-treated wastewater having absorbance values of 0.1 to 0.5 cm^{-1} and TSS concentrations of less than 60 mg/L (Mahmoud et al., 2013; Azimi et al., 2011; Loge et al., 1999). On the other hand, the mean absorbance values of the filtrates during this research were 1.34 and 11.85 cm^{-1}, and the mean TSS concentrations were 870 and 7630 mg/L, for the samples from the yacht and from the camper respectively. These levels are far higher than have typically been investigated and indicative of high concentrations of humic acids or other dissolved compounds that absorb UV radiation at a wavelength of 254 nm (Hill et al., 2002; mohammadi et al., 2013).

The humic matter that is dissolved in these wastewaters could be one of the reasons why the experiments shown in Figure 4.10 and Figure 4.11 were not scaled properly and show unsubstantial disinfection. To determine a good range on the y-axis the effective UV-C dose needs to be calculated, which will give a better estimation of the disinfection performance. With Equation 2.1 a factor for the decrease in irradiance which arises from absorption is calculated. This equation gives an estimation for the average UV intensity that each liquid parcel receives in a completely mixed liquid, called the water factor (Kuo et al., 2003). For the vacuum toilet samples, this water factor was 0.305 and for the camper samples 0.0361. The total average UV dose is then calculated by multiplying with the total time (Equation 2.2). This would mean that the total effective UV irradiation at the end of the UV experiments from Figure 4.10 and Figure 4.11 equalled 15.86 mJ/cm^2 for the vacuum toilet sample and 2.89 mJ/cm^2 for the camper sample respectively. In this report the UV-disinfection graphs are all plotted with the total UV irradiation on the Y-axis instead of the effective irradiation. This choice was made because the effective irradiation would not give a good picture of the amount of energy it took to remove the fecal indicators from the sample. When the total UV irradiation is known, better estimates can be made about the effectiveness of the PoC method and of the characteristics of the treatment system. Nonetheless, the effective dosages of UV are given in the text.

This is, however, still a highly simplified way of calculating the effective dose, as it is dependent on many more parameters like a divergence, a reflection and a petri dish factor (Bolton & Linden, 2003). The reason that these factors are not included in the calculations in the following paragraph is that the effective UV dose arriving at the pathogenic bacteria in the wastewater is likely to be higher than the calculated values. Spectrophotometric measurements have been shown to overestimate UV absorbance by particles in wastewater because of the scattering effect. Of the spectrophotometric absorbance caused by particles, about 75% was true absorbance and 25% was scattering (Loge et al., 1999; Qualls et al., 2015). Even though both sampled were filtrated properly, they still had a turbidity of around 17 NTU, which would mean that some degree of scattering by particles still is taking place. Therefore, it was decided that the amount of scattering equals the divergence, reflection, and petri dish factor, which is why the simplified equation for the effective dose is used, which will then give a rough estimate of the effective UV-dose.

Finally, the last two UV-disinfection experiments were performed on the filtrates of both samples, after they were flocculated and pressed through the chamber filter press. Figure 4.12 depicts the UV-disinfection kinetics of the filtrate from the vacuum toilet samples. The black lines in the graphs indicate the boundary conditions that need to be met by the system. After an applied UV-C dose of approximately 180 mJ/cm^2, the filtrate seems sufficiently disinfected to be allowed to be discharged from a recreational craft. Multiplying this with the water factor of the toilet samples, the effective UV dose to meet the boundary conditions of both fecal indicators is 54.9 mJ/cm^2. The UV-disinfection kinetics of the filtrate from the camper are shown in Figure 4.13. Even after a high applied UV-C dose of 1100 mJ/cm^2, the disinfection does not seems sufficient for the water to be allowed to be discharged. The effective UV dose at the termination of the experiment was 40.0 mJ/cm^2.

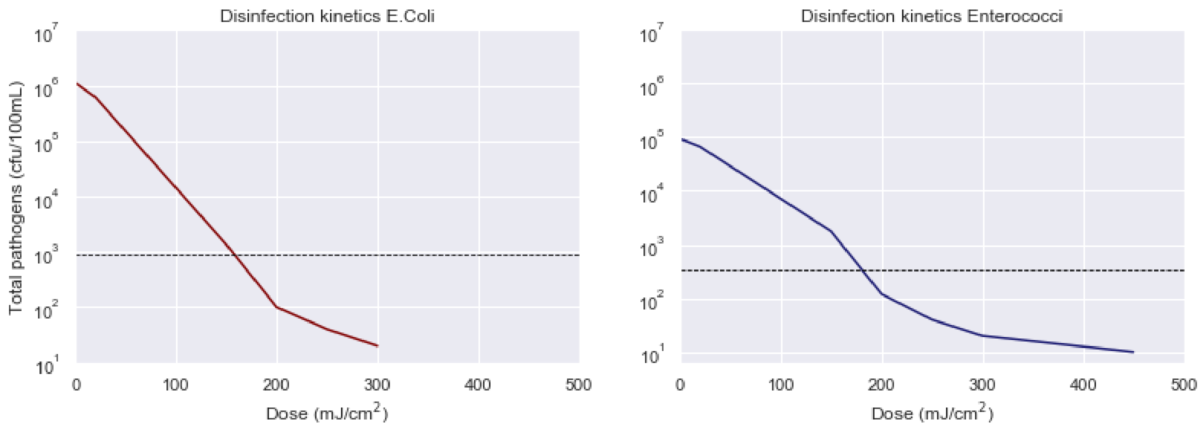

Figure 4.12: *UV-C experiment performed on the filtrate of sample from the vacuum toilet after it was pressed through the chamber filter press. For flocculation 1 ml/l PAX-18 and 30 ml/l C-2230 was used, and the turbidity of the filtrate was 99.1 NTU. The petri dish with the sample was placed on the top shelf of the collimated beam apparatus, with a much higher irradiance. The black dotted line represents the boundary condition for the specific fecal indicator*

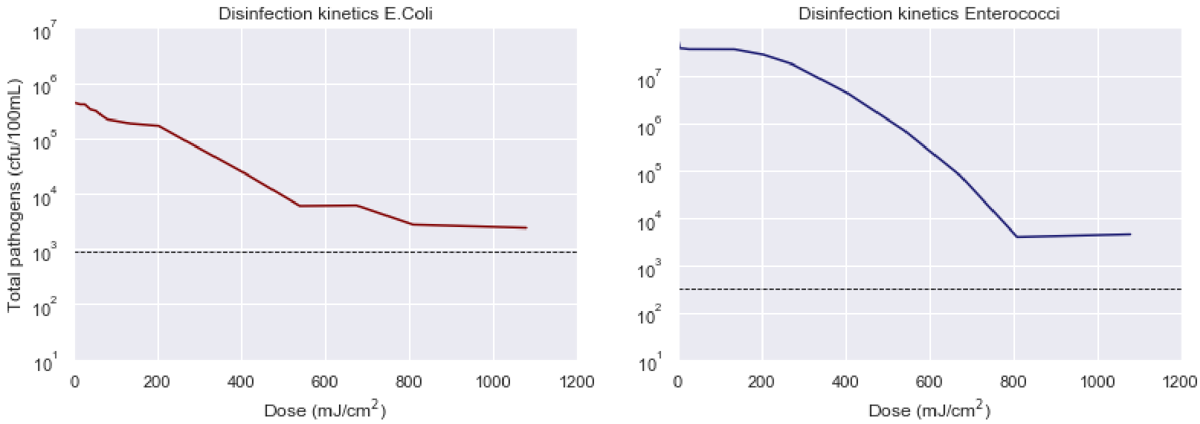

Figure 4.13: *UV-C experiment performed on the filtrate of sample from the camper toilet after it was pressed through the chamber filter press. For flocculation 4 ml/l PAX-18 and 160 ml/l C-2230 was used, and the turbidity of the filtrate was 27.8 NTU. The petri dish with the sample was placed on the top shelf of the collimated beam apparatus, with a much higher irradiance. The black dotted line represents the boundary condition for the specific fecal indicator*

5. Discussion

In this chapter, the results of the research will be discussed and placed into perspective. Additionally, the limitations that were experienced while conducting the research are discussed and the implications of the findings of this research for the intended treatment system are suggested.

5.1 Implications of the research findings

This section discusses the implications that the research findings can have on the future development of a treatment system that can be used on board of recreational crafts, which was the main goal of this research. Also, the importance of the findings of this research for wastewater treatment in general is discussed.

5.1.1 Removal of TSS

The coagulation and flocculation experiments showed promising results for the removal of the TSS fraction of both of the wastewaters. The results showed that, regardless of the high variability in the consistency of the blackwater, an efficient removal of solids can be achieved. However, the ideal dosages of coagulant and polymer that were used in this research may not apply to the finalized treatment system. They should be reconsidered and adapted to the consistency of the blackwater that the system is going to treat and to the system itself, and the best practices to do this are explained in the next paragraphs.

It has been shown that the TSS is the most decisive factor for the determination of the quantity of conditioning agents to be added. Ideally the dosing mechanism of the treatment system should be adjusted directly to the situation specific characteristics of the blackwater at that moment. At the beginning of this study, an attempt was made to link turbidity to the amount of conditioning agents. However, this idea was dismissed as the results in section 4.1 showed that turbidity does not represent the TSS in the wastewaters, but rather the amount of particles that scatter light. For instance, the value for turbidity may be greatly influenced by a residual haze, which may represent an extremely small proportion of the original solids. Additionally, there was not enough correlation found between the turbidity values that were measured during this research, and the quantity of conditioning agents needed. An idea that could be considered to tackle the high variability of the blackwater and to save on conditioning agents is to place a TSS measuring device on the influent pipe of the system. The "Valmet TS" system gives accurate measurements of the TSS in the range of 0 to 50%. It ensures long term reliability, which is something that is needed when it is being used by people who are not skilled in wastewater engineering (Valmet, 2021). However, the costs saved on the use of conditioning agents may not be able to match the capital costs that a device like this will add to the system.

When no meassuring device for TSS is implemented in the system, an approach to deal with the high variability in TSS in the blackwater is to assume a "worst case scenario" and to adapt the system according to those values. This way, a good flocculation is ensured, regardless of the consistency of the influent. The "worst case scenario" would have to be determined specifically on site, or a set of worst case scenarios would have to be made that are linked to a particular toilet or toilet system. This research has shown that an overdose of coagulant is not likely to cause any problems, except for higher cost, and that it would even increase the disinfection capacity of the system due to more bacteria being enmeshed within the flocs created by sweep coagulation. As most of the experiments were done with an overdose of the polymer, it is clear that this is also not likely to cause problems for the flocculation of the solid material. It will however, result in a residual amount of flocculant in the supernatant and filtrate liquid stream. The effect of this residual amount of polymer has not been investigated in this research, but it could cause problems during UV-disinfection, due to absorption of UV-C light by the polymer molecules. This would need to be researched further. Something else to keep in mind is that the amount of coagulant that is added, also effects the amount of polymer that will be needed.

Finally, it is the design of the system itself that will have the greatest influence on how the solids need to be flocculated and on the removal of TSS. The type of filtration, the filter mesh, the method of dewatering (a piston, centrifuge or electro-dewatering), the method of disinfection, etc., will determine what type of flocs will need to be formed. This research has mainly provided proof and guidance in the operation and application of certain conditioning agents for the subsequent treatment processes.

Sweep flocculation

The choice was made to pursue sweep flocculation at all times during this research, and it's also recommended to do so in the future when designing the treatment system, as this is the most effective way to precipitate the colloids and even a part of the fecal indicators. Nonetheless, when particle concentrations are as high as in both of the samples used in this research, the charge neutralization and sweep flocculation zones tend to blend into each other (see Figure 2.4), which makes it impossible to distinguish the end of the charge neutralization mechanism from the beginning of sweep flocculation (Benjamin & Lawler, 2013). However, a downside to the pursuance of sweep flocculation is that the humic substances, which are presumably the reason for the poor functioning of the UV-C irradiation, are not extracted by sweep flocculation.

Effect pH on coagulation

Due to the high pH values of the blackwater samples from the camper (see Table 4.1), a 10% acetic acid solution was gradually added to these samples until the pH value was in the range of 7.2 to 7.5, which is common for raw blackwater (Metcalf & Eddy inc., 2002). The pH had to be lowered because at these high pH levels, most of the solid material is deprotonated, giving the solid material an even higher negative charge. The Amirtharajah diagram (Figure 2.3) shows that the high pH values fall outside of the region of insolubility of $Al(OH)_3$, and that sweep flocculation with aluminium is only achieved in a pH range of 6 to 8.5. When the treatment system would have to treat blackwater with a high pH value, it could be considered to use an iron coagulant instead of an aluminium one. Iron coagulants have a much wider region of insolubility of $Fe(OH)_3$. For wastewater treatment in which the pH is higher than 8.0, iron based coagulants are always preferred because of this difference in the solubility of the metals (Benjamin & Lawler, 2013). Nontheless, it is likely that the values of pH of the blackwater that the system is going to treat are closer to the common values for blackwater. The formation of ammonium from urea happens when urine and feces are mixed together, and it happens fast but not spontaneous. If coagulation and flocculation in the treatment system happens before this conversion takes place, the problems involved with the high values of pH can be bypassed.

5.1.2 Maceration

Homogenisation of the blackwater during sampling was necessary for this research. It was needed to decrease the variability of the characteristics of the samples, which, in spite of that, was still high, so that better and more accurate predictions could be made before and during the performance of the experiments. However, for the design of the treatment system this step should be reconsidered. The intestinal tract is capable of high levels of dewatering, as the solid content of human feces can range from 10-50% (Nishimuta et al., 2006). It seems like a redundant step in the system to macerate all of this, relatively dry, fecal material into a homogeneous mass, to subsequently try to flocculate and, after that, dewater it again.

The possibility of removing most of the solid material directly after defecation should be considered. This is also referred to as preliminary screening, and is a standard treatment procedure in WWTPs. It has been defined as the removal of gross solids in wastewater and the removal of grit prior to subsequent treatment (Sidwick, 1991). Of course, the remaining waste stream will still contain solid material, but this can be flocculated, filtered and dewatered as proposed in this research. This would result in a substantial decrease in the required amount of coagulant and flocculant, which would also reduce the costs. It would also relieve some of the pressure on the dewatering mechanism, which should increase its performance. As results of this research have shown that the effectiveness of cake filtration is strongly dependent on the total amount of solids in the cake. In an ideal situation, where almost all of the solid material is directly separated after defecation, a dewatering step could even be omitted, as the amount of flocs that form after the flocculation step would be minimal. However, a disadvantage of the above mentioned method is that the consistency of fecal material is not guaranteed, and that if the consistency is too watery the material could pass through the coarse preliminary filter. When the system is tuned to a low amount of solids in the wastewater stream, a sudden spike in TSS could mean that the effluent of the system no longer meets the requirements for fecal indicators.

5.1.3 Dewatering

Section 4.3 has shown that under ideal conditions a dry solid content of 20-30% can be achieved. An 8% increase in dry solid content of the sludge cake is achieved with a five bar increase in pressure. It is questionable if this slight increase in dewaterability is worth the extra costs and effort of such a high pressure dewatering system. The dewatering step would be implemented purely to decrease the total volume of the solid material so that the storage capacity of the solid material increases, but such a small decrease in volume will hardly be noticeable in the end.

What could be interesting is the comparison between solid content of the material with and without compression. This way the usefulness of a dewatering step can be determined. If, for instance, costs need to be saved on the system design, a choice can be made to leave the dewatering step out and another way of separating the solids can be implemented. This would then again also change the quantity and ratio of coagulants and flocculants, because compact strong flocs are then preferred.

5.1.4 Costs of conditioners

When the conditioning agents are ordered in intermediate bulk containers, which have a volume of around 1,000 liter, the C-2230 costs 3.00 €/kg and the PAX-18 costs 0.45 €/kg. Of course, the amount of C-2230 that is actually used per liter is substantially lower than the PAX-18, as only 0.2 ml is used per 100 ml of stock solution. The price per liter of C-2230 stock solution is 0.006 €/l. Table 5.1 shows the price predictions of the conditioning agents per liter of blackwater if the optimal ratio for the specific sample is used, which is 1 ml/l PAX-18 and 40 ml/l C-2230 fot the yacht samples and 4 ml/l PAX-18 and 150 ml/l C-2230 for the camper samples.

Table 5.1: *This table shows the costs associated with the use of the conditioning agents*

	Camper sample	**Yacht sample**
PAX-18 (€/l BW)	1.8E-3	4.5E-4
C-2230 (€/l BW)	9.0E-4	2.4E-4

5.1.5 UV-Disinfection

After an applied total UV-C dose of approximately 180 mJ/cm^2, which was equal to an effective UV-C dose in the sample of 54.9 mJ/cm^2, the removal of the fecal indicators in the samples of the yacht seems sufficient enough for the wastewater to be discharged on open water. As the samples were placed on the top shelf of the collimated beam (with an irradiance of 2.84 mW/cm^2), the total irradiation time was 64 seconds. However, the removal of fecal indicators from the samples from the camper did not reach the boundary conditions, not even with a substantially large dose of UV-C light of 1100 mJ/cm^2. The effective dose inside the wastewater was only 40.0 mJ/cm^2, and the total irradiation time was 390 seconds.

Even though the boundary conditions were reached with the samples from the yacht, the effectiveness of UV radiation still needs to be increased for it to be an efficient disinfection technique in the PoC method and in the actual treatment system. 80 mL of sample was used in the perti dish, and the depth of the sample was only 2 cm, but it still took 64 seconds of irradiation time by six low pressure UV-C lamps to disinfect the relatively small amount of filtrate. When larger volumes need to be disinfected, the UV-disinfection will experience difficulties with reaching the boundary conditions. It should also be realized that the experiments in the collimated beam apparatus were conducted with a total of six UV-C lamps, which is not conventional as in a smaller UV-disinfection system generally a total of one or two lamps are used.

5.1.6 PoC method evaluation

To reach the limits of 330 cfu/100 ml of *Intestinal Enterococci* and 900 cfu/100 ml of *E.Coli* in an effective way, improvements need to be made within the PoC method. Steps in the PoC need to be re-evaluated or extra steps need to be added. This section will explain some of the adjustments that can be made to reach the boundary conditions for fecal indicators of the effluent. For convenience the PoC method that was reasearched is displayed again in Figure 5.1.

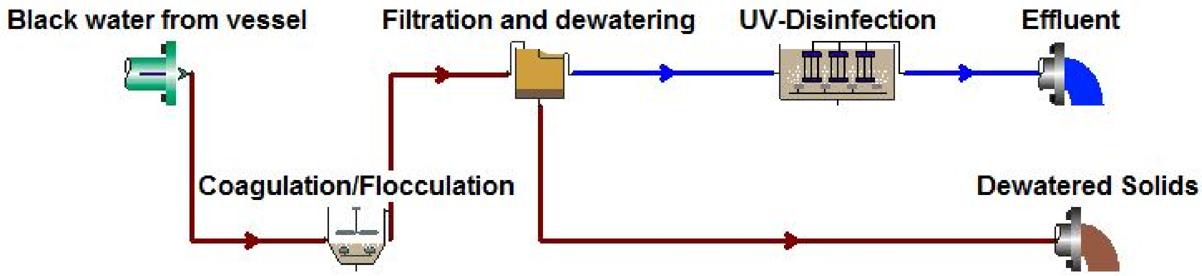

Figure 5.1: *Schematic overview of the PoC method that was evaluated during this study. The diagram was made using BioWin*

The biggest problem for the effectiveness of the UV-C light is the colour of the filtrate, which is caused by humic substances that are dissolved in the wastewater. The humic substances have a detrimental effect on the internal transmittance of UV_254 light inside of the wastewater. The extraction of the humic matter from the wastewater is therefore seen as a necessary step to increase the effectiveness of UV-disinfection.

As explained in subsubsection 5.1.1, sweep flocculation was pursued during this research because it was the most effective way to remove the TSS fraction from the wastewater. However, for the removal of dissolved humic substances, adsorption onto the surface of positive charged ions is required. Due to the neutrally charged sweep flocs ($Al(OH)_3$), adsorption onto the flocs is low. In adsorptive coagulation, humic matter is adsorbed onto the cationic hydrolyses products $Al(OH)^{2+}$ (Mathuram et al., 2018). Adsorptive coagulation mechanisms can be achieved by lowering pH and using higher dosages of coagulant, which can also be seen in the Amirtharajah diagram shown in Figure 2.3.

In the PoC method, the choice could be made to pursue a two-step coagulation method. The first step would be to ensure sweep flocculation to remove the TSS fraction of the wastewater.

The pH value of the vacuum toilet was already sufficient for this, and, as discussed in subsubsection 5.1.1, the pH values of the camper samples would also be sufficient if the coagulation step was employed instantly after the moment the urine and feces get in contact with each other. After the TSS fraction has been extracted, an extra dosage of coagulant could be added and the pH slightly lowered to ensure the formation of $Al(OH)^{2+}$ for adsorptive coagulation of the humic matter. However, this method has its shortcomings as inorganic coagulants only remove a small fraction of this humic matter. The hydrophilic, low molecular weight fraction are removed less efficiently by these coagulants (Bhatnagar & Sillanpaa, 2017).

Another method to remove the humic matter and thus the colour from the wastewater is to add intermediate ozonation to the PoC method, following the filtration step. Ozone preferentially reacts with humic substances (mostly by electrophilic substitution and oxidation reactions), which results in substantial color removal, decreasing the absorbance values for UV_{254} and also decreasing the potential formation of DBP. Intermediate ozonation performance is specific to the water characteristics and ozonation conditions, so it would have to be tailored to the specific wastewater type and treatment system (Plourde-Lescelleur et al., 2015). Ozonation also rapidly reacts with bacteria over a wide pH range and even has stronger germicidal properties than chlorination, making it a very effective disinfection step, which maybe even obviates the UV-disinfection step.

To increase the removal of fecal indicators, the coagulant dose for sweep coagulation could also be increased. The experiment in Figure 4.5 showed that increasing the PAX-18 dose above 2 ml/l, substantially increased the removal of bacteria from the wastewater. As the previous section described, the costs of the conditioning agents are relatively low. This means that increasing the dose of coagulant to increase the removal of fecal indicators by sweep coagulation could be considered if the final disinfection of the system is not sufficient. Also, less polymer is needed when a higher dose of coagulant is used. However, care must be taken not to have free aluminium in the effluent, which can cause aluminium phytotoxicity in humans when high levels of aluminium are ingested.

5.1.7 Alternative solutions

As an alternative solution that would require a completely different PoC method, there is the option of separating the urine and feces before they even come in contact with each other with the use of a so called composting toilet. Such a toilet immediately directs the urine into a holding tank, and the feces into a separate holding tank, which is often filled with saw dust, coconut coir or peat moss and is connected to a fan and venting tube to remove any odor and assist in the dehydration and composting process. The aerobic digestion of the feces results in the disinfection of the material, after which a nutrient dense compost is left. The urease in the feces does not come in contact with the ureum in urine as discussed in section 4.1, which ensures that virtually no odour is released by the system. The storage capacity of a composting toilet is also high, as the fecal material is loosing almost all of its moisture in the holding tank. The urine tank, however, needs to be emptied every several days, depending on its size. A system like this can be an alternative to flush based sanitation, and saves a lot on water usage. It is also convenient that the toilet does not have to be connected to a central water system. The greatest barriers to the use of composting toilets are public acceptance, regulations and lack of knowledge and experience in design and operation (Anand & Apul, 2014).

Another alternative solution is the use of heat for disinfection. Microorganism inactivation by heat is the oldest and simplest method of disinfection. It has been proven exceptionally effective against bacteria, protozoa and viruses, including those resistant to chemical treatment or irradiation. No disinfection byproducts are formed, and heat penetrates deep into particles. The efficiency of thermal disinfection is not affected by TSS, natural organic matter or other pollutants in the wastewater, which would make a coagulation/flocculation and filtration and dewatering step absolete. Studies have shown that a disinfection up to 8-log can be achieved in high turbidity samples (100 NTU) (Smiech et al., 2020). A crucial aspect of thermal disinfection systems is the energy per m^3 of treated water aspect ratio. To make a thermal disinfection system viable it is important to efficiently recover heat that is used in operating the system. Inventive ways can also be found to recycle heat used on the boat, from the motor and the machine chamber for instance, into the thermal disinfection system.

Table 5.2: *Comparison of energy demand for water disinfection by UV irradiation and thermal inactivation, retrieved from a study by Smiech et al. on the thermal disinfection capacity of a pilot-scale continuous-flow system. The disinfection values for the UV were taken from the "Emerging Technology database" from Washington State University and are not applicable for the values from this research (Smiech et al., 2020)*

	Energy demand (W h L^{-1})	**Disinfection level (log)**
Low-pressure UV lamps	0.4 - 0.9	2 - 4
Medium-pressure UV lamps	1.7 - 2.1	2 - 4
Thermal disinfection pilot	< 10	6 - 8

5.2 Limitations

In this section, a brief explanation of the limitations of this research follows. Because the research involved non-conventional samples and methods, a lot of creative solutions had to be devised. Also, a lot of the machinery and equipment had not been used in a long time and were damaged or sometimes even broken.

5.2.1 Samples

The samples that were used for this study are not conventional in the field of wastewater treatment. In general, secondary and sometimes primary sludge is used, and this already has different characteristics then the raw blackwater that was used here. Pathogenic density in the raw samples was also a lot higher then in primary or secondary sludge, which meant that certain restricting guidelines had to be followed for handling the blackwater in the WaterLab.

Samples from only two different locations have been used during this study. There is a high chance that these samples do not represent the black water of most of the boats that are going to be using the system, as a vacuum toilet system is only seen on large boats, such as yachts with several toilets on board. The camper sample came from a chemical toilet. Conducting experiments on more types of samples would drastically increase the length of this study and was outside the scope and period of this research.

A transition was required from the samples of the yacht to the samples of the camper, halfway through this research. This had to be done because the owner of the yacht went on a long holiday, which would leave the blackwater tank empty. However, the samples from the camper turned out unsuitable for the UV-disinfection experiments, and it may be questioned whether its characteristics are similar to the blackwater that the system is supposed to treat in the future. Also, in chemical toilets it is customary to add a certain chemical, often with formaldehyde added, that helps against the smell because it stops the conversion of urea into ammonia and carbonic acid. However, this is a toxic chemical that alters the characteristics of the blackwater and kills off bacteria, which was unfavourable during this research. Unfortunately, during some of the sampling days, this chemical did end up in the blackwater. This has caused several experiments to fail, especially the UV-disinfection experiments.

Efforts were made during this research to collect the samples as quickly as possible, so that they would spend the least amount of time in the blackwater tank where temperatures were relatively high (15-20 degrees Celsius). But, as the planning did not always allow immediate collection of the samples, most of the times the blackwater spend more time in the storage tank than was desirable. Also, occasionally the samples would spend a considerable amount of time in the fridge before before they were used for experiments. Higher temperatures and longer storage time may have altered the blackwater a certain way, which would be of course not favorable for this study. An example is the severe increase in pH of the camper samples.

5.2.2 Microbial analysis

Bacteria are living organisms, which is why there is always a high degree of randomness involved when working with them. Attempts have been made to minimise the randomness of the measurements, among other things by using triplicates for each measuring point and by increasing the plated volume of wastewater. However, there were still large deviations in the results.

Also, before each experiment an estimate had to be made of the amount of bacteria in the blackwater, and on the effectiveness of the disinfection over time. This proved to be difficult as the blackwater varied greatly from batch to batch. As a result, a large number of the cultivation plates had to be declared invalid because they contained too many or not enough bacteria, and sometimes even entire experiments had to be redone or invalidated.

5.2.3 Experimental setup

The proof of concept method that was researched in this study was designed to resemble a real life treatment system as good as possible. The Waterlab has many possibilities to investigate wastewater, but there were certainly limitations encountered with regard to the use of machinery during this study. In the first place, it must not be forgotten that most of the equipment used was bench scale. For instance, the filter press was only able to dewater a small quantity of solids effectively, as the filter cell only had a small volume. Due to the filtration and expression phases being highly complex, increasing the solid content or filter area, or using a different filter cloth will result in completely different results. The WaterLab also did not have any other apparatus for the dewatering of the blackwater then the filter press, so no other methods could be tested. The filter press became a staple during this research as it was also used often to generate a filtrate that could be used in the collimated beam. This means that a lot of the results were tied to the effectiveness of the press, which could make them research somewhat lopsided.

The collimated beam made use of six 15 Watt lamps and the distance to the sample was relatively high, even after the sample was placed higher up in the machine. Both of these things will not be the case when a UV-disinfection step is implemented in the treatment system, as those systems will probably only use one or two lamps and the wastewater will flow really close to those lamps. However, it was not possible to conduct scientific experiments on such a system as one has limited control over it. During the experiments it was necessary to exactly measure the UV-irradiation and regular samples needed to be taken. This does mean, however, that the UV-disinfection by the collimated beam displays different results than if an existing UV- disinfection system was used.

Unfortunately, the utilization of some apparatus was not possible during this research. To quantify the effect of coagulation the zeta potential is generally measured. However, the majority of the particles in both of the blackwaters was far too big for the zeta potential device. It measures particle velocity in an electric field by having the particles move through an interference pattern of two intersecting laser beams. If the particle size is much larger than the lines in the interference pattern, the measurement is not possible. The detector could also only handle a certain amount of photons per second per surface, and when the particles would be too big only the light of the single particles would be measured. Another measuring instrument that did not work properly was the radiometer, used for measuring the irradiation of the collimated beam. It had not been used in approximately five years and some sections were partially broken. When connected to a laptop it could just shutdown and stop working for a couple of days. Some UV-disinfection experiments were therefore conducted with a timer and a mean value for the irradiation of the lamps to calculate the total UV-dose on the sample, which makes te calcualted dose less accurate.

6. Conclusions

6.1 Conclusions

The main goal of this research was to present and investigate the validity of a proof-of-concept for the disinfection of raw blackwater on board of recreational crafts. Based on conventional wastewater treatment methods like coagulation and flocculation, filtration and dewaterability, and UV-disinfection, a proof-of-concept method was devised that aimed to achieve a high degree of disinfection of different types of raw blackwater. The main findings of this research were:

- TSS was most efficiently removed by using a combination of polyaluminnium chloride (PAX-18) and a high molecular weight cationic polyacrylamide copolymer with a low charge density (C-2230)
- Polyaluminnium chloride also was highly effective at removing the fecal indicators
- Increasing the dosage of polymer demonstrated to have no effect on the dry solid content of the sludge
- Under the most ideal circumstances a dry solid content of the filter cake of 30% was achieved
- When pressure was increased from two to five bar, an average increase in dry solid content of 8% was achieved
- The samples from the vacuum toilet met the boundary conditions after a 4-log inactivation following a total UV-C irradiance rate of 180 mJ/cm^2, of which the effective dose was 54.9 mJ/cm^2.
- The samples from the chemical toilet did not met the boundary conditions after a 4-log inactivation following a total UV-C irradiance rate of 1100 mJ/cm^2, of which the effective dose was 40.0 mJ/cm^2.

The PoC method showed to have potential for implementation, but adjustments will have to be made to make it more sufficient and viable for use of on board disinfection of blackwater. Preliminary screening can extract most of the fecal material, which is already high in dry solid content, before homogenising with the liquid fraction. The quantity of conditioning agents need to be adapted to the specific blackwater, and if necessary the pH will need to be adjusted to ensure the right coagulation mechanism for efficient TSS removal. Finally, humic matter in the filtrate will have to be reduced for UV-disinfection to be an effective treatment method. This could possibly be achieved by mechanisms such as adsorptive coagulation or by implementing an intermediate ozonation step after filtration of the TSS, which would also help in the removal of fecal indicators.

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