Biofiltration in Single and Combined Pretreatments for Reclamation of Secondary Effluents to Drinking Water Quality

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Abstract

Growth of population and the contamination of water resources are depleting potable water sources. Wastewater reclamation is gaining interest as an alternative water source as it is a continuous source with low salinity; however, the current degree of wastewater treatment is such that the released wastewater streams called secondary effluents are suitable for restricted irrigation or sea disposal. The environmental effects of such a practice are negative, contamination of aquifers increases and with that the health risks associated with pathogens, suspended solids, nutrients and other pollutants commonly present in the secondary effluents. Thus, further treatment is necessary. Increasing the quality of secondary effluents to drinking water level is possible with Reverse Osmosis (RO) membranes. The back side of RO treatment of wastewater effluents is the high concentration of organics and anticipation of high degrees of organic and biofouling. The effective yet economically affordable and robust pretreatment is therefore a crucial prerequisite that will determine a success or a failure of the advanced wastewater treatment.

One of the options for pretreatment before wastewater desalination with RO is biofiltration. Biofiltration is a fixed film process responsible for the removal of biodegradable dissolved organic matter by attached biomass originating from the influents. Therefore, both organic and biofouling effects can be reduced in the RO membrane. Biofiltration is a simple technology, low cost, and chemical free which could be easily built and operated in every WWTP.
This work was performed with pilot scale biofiltration columns located in Sede Teiman WWTP. Continuous source of secondary effluents was supplied to the system and removal abilities of biofiltration were examined. Two approach velocities: 0.75 and 2 m/h were tested. The effect of different EBCT (examined by sampling of different filter depths: 60, 90, 130 and 160 cm) on removal efficiencies were tested as well. The medium used in the biofilters was exhausted granular activated carbon. Biofiltration was not only studied as a stand-alone pretreatment but also as part of a combined pretreatment with coagulation and ultrafiltration (pilot plant system created and designed by inge GmbH). Biofiltration from single or combined pretreatment was connected to RO pilot plant and fouling was measured.

For water quality examination the following analyses were performed: UV$_{254}$, DOC, turbidity, phosphate, Emission Excitation Matrix and LC-OCD. Fouling of RO membranes was assessed by permeability calculations.

It was found that biofiltration operated better in all parameters at the lower approach velocity. It was found that for all fractions measured (except biopolymers) main removal occurred in the beginning of the filter (first 60 cm). Biofiltration demonstrated the following removal abilities: average of 30-45% DOC removal was measured, turbidity removal efficiency was as high as 70% by average. Phosphate removal reached average of 70%. EEM was demonstrated to be an effective and accurate tool able of tracking humics. It was not applicable in measurement of proteins. Humics removal was shown to be as high as 70% by average. Using LC-OCD it was found that biofilter removes biopolymers at a higher EBCT since these are less biodegradable molecules. All DOC fractions were lowered in the biofiltration effluents.
From the combined pretreatment of biofiltration - coagulation – ultrafiltration it was found that biofiltration can achieve the same level of treatment as the UF without coagulant. The difference is in the retention of biopolymers when the UF pretreatment removes biopolymers completely and biofiltration can remove between 20 and 30% only. The retention of biopolymers is important to minimize the RO fouling. Indeed the flux drop in RO system in experiments performed with UF pretreatment was much less than in experiments performed with biofiltration. This means that there is no gain in a combined biofiltration - UF pretreatment. Both treatments retain the same compounds and their combination just increases capital and operational costs, however, biofiltration can serve as a sole pretreatment before the RO especially when the price for the entire setup is important.

**Keywords:** Biofiltration, secondary effluents, ultrafiltration, reverse osmosis pretreatment, wastewater reclamation, approach velocity, emission excitation matrix.
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1. Introduction

1.1 Wastewater Treatment

1.1.1 General

Growth of population and the contamination of water resources are depleting potable water sources. The global search for alternative water sources began a few decades ago and has found two promising technologies: seawater desalination and wastewater reclamation.

Seawater desalination has been considered to exchange the main water source from sweet to salty water. Rapid development occurred in this field during the last years and desalination plants were built in many arid countries. Desalination of seawater is a process composed from few pretreatment steps followed by high pressure reverse osmosis (RO) membranes, as salinity is the main factor to be removed. Israel is located in the Middle East where water resources are scarce and the future water supply depends on seawater desalination. In 2010, three large-scale seawater desalination plants provided 320 million cubic meters of Israel’s potable water requirements (to all sectors). This volume is equivalent to approximately 42% of the domestic water requirements. The water authority prognoses that desalination production capacity will increase to 577 and 750 million cubic meters/year by the years of 2014 and 2020 respectively (OECD Environmental Performance Reviews: Israel 2011 (2011); Tenne, 2010).

An alternative process, with many advantages over seawater desalination, is the utilization of secondary effluents. Wastewater is a continuous water source with low
salinity. However, the current degree of wastewater treatment is such that the released wastewater streams called secondary effluents are suitable for restricted irrigation or sea disposal. The environmental effects of such a practice are negative (Qin et al. 2004; Zhou and Smith, 2002). The contamination of aquifers increases and with that the health risks associated with pathogens, suspended solids, nutrients and other pollutants commonly present in the secondary effluents. In a projectory view of the situation the advanced tertiary treatment would produce high quality effluents with potential use for unrestricted irrigation or for aquifer recharge (Glueckstern et al. 2008).

In even more advanced quaternary treatment, the quality of the effluents can be increased to the drinking water level. That operation will require the introduction of high pressure membranes that will however face feed with much lower concentration of dissolved solids. Thus the cost of quaternary treatment is estimated as only one third of the cost of seawater desalination making the advanced treatment a feasible and a wise choice (Glueckstern et al. 2008). The back side of RO treatment of wastewater effluents is the high concentration of organics and anticipation of high degrees of organic and biofouling. The effective yet economically affordable and robust pretreatment is therefore a crucial prerequisite that will determine a success or a failure of the advanced wastewater treatment.

Possible pretreatment options include biofiltration for the removal of high molecular weight and biodegradable organics, coagulation/flocculation for removal of particles and organics, and ultrafiltration (UF) for removal of particles. Combinations of these treatment processes might be ideal, as each process has its strengths and weaknesses with
regard to the removal of different impurities from secondary effluents. My research centers on the biofiltration.

1.1.2 Biological Degradation as the Core Wastewater Treatment

Biological treatment is an important and integral part of any wastewater treatment plant (WWTP) that treats wastewater containing soluble organic impurities. The obvious economic advantage, both in terms of capital investment and operating costs, of biological treatment over other treatment processes (like chemical oxidation; thermal oxidation etc.) has cemented its place in any integrated wastewater treatment plant.

All biological treatment processes take advantage of the ability of bacteria and other microorganisms to remove dissolved organic matter by assimilation (converting the organic carbon into carbon dioxide and biomass). Biological processes can take place in aerobic or anaerobic conditions, directly influencing the type of biomass involved in the biodegradation process.

Biological treatment processes can be sub-divided into suspended growth and fixed film (attached growth) systems. In suspended growth systems the biomass is mixed with the influent and cannot form a biofilm. The suspended growth system requires an additional secondary clarification stage where bacteria are separated from the liquid phase. In fixed film processes bacteria is attached to a media inside the reactor. A formed biofilm is better controlled in terms of its location. Separation phase is not always required and depends on the type of the process.

Main treatment methods are described below.
1.1.2.1 Suspended Growth Systems

**Oxidation ponds** (stabilization ponds) are simple bodies of water where treatment of wastewater is dominated mostly by algae and bacteria. These ponds do not require any process interference (besides arrangement of flow) and are mostly used in small and isolated communities for wastewater treatment. Oxidation ponds are simple and low cost and do not require skilled manpower. However, large area is required, and the quality of treatment depends on weather conditions. Water losses occur due to seepage and vaporization and effluent quality is low because it contains algae. **Aerated lagoons** are improved wastewater treatment systems where artificial aeration is implemented to promote improved biodegradation by bacteria. These lagoons require higher capital and operation costs than the oxidation ponds but produce better quality effluents due to better degradation of organic matter. Quality of the effluents is aerated lagoons is still not very high because there is no precipitation stage, and no sludge recycling like in the activated sludge process which is described below.

**The activated sludge process** is the most commonly used technology in wastewater treatment. The process comprises a few stages: pretreatment includes the removal of rough materials by screening to prevent the possible damage to hydraulic equipment such as pumps and valves. Primary treatment includes the primary sedimentation tank responsible for the removal of suspended solids (basically colloidal material). The sludge comprised of particles is produced and removed by vacuum pumping from the bottom of the sedimentation tank. The secondary treatment is responsible for the biodegradation of organic compounds and sometimes also for removal of nutrients. This is accomplished by proper aeration and high biomass concentration in the reactor which allow the
decomposition of biodegradable organics by bacteria. This stage is followed by the secondary clarifier. Effluents from this stage are called secondary effluents and are disinfected or further treated by the tertiary treatment. The settled sludge in the secondary sedimentation tanks is partially recycled to the reactor- maintaining the concentration of the biomass, while the rest is collected and sent to sludge treatment. The advantages of activated sludge process make it the most common treatment option for municipal (and sometimes industrial) wastewater. The technology is reliable, operation costs are low comparing to more advanced treatment options (listed below), it can be designed to treat high flows of wastewater, and the quality of the secondary effluents meets the demands for restricted irrigation, therefore reclamation of the wastewater is possible. However, activated sludge has many disadvantages: it is not suitable for small communities since capital costs are rather high (comparing to more simple treatment options). In addition, space requirements are high and effluents must be farther treated to meet unrestricted irrigation standards.

**Membrane Bio- Reactor (MBR)** is the latest technology for biological degradation of soluble organic impurities. In the MBR, as in the activated sludge process, the influent is mixed with the microorganisms in an aeration tank. The difference lies in the method of separation of bio-solids. In MBR the bacteria are separated by microfiltration or ultrafiltration polymeric membrane. In the activated sludge the separation is based on the gravity settling process. MBR has many advantages over the conventional activated sludge process. The system has more effective sludge separation, can be operated in a variety of sizes and can reach higher effluent quality (due to the membrane filtration).
However, capital and operation costs are much higher comparing to the conventional activated sludge and the reactor requires skilled operators (Krzeminski et al. 2012).

The **sequenced batch reactor** (SBR) is another variation of the classic activated sludge system. SBR process covers all the functions of a conventional activated sludge plant (biological removal of pollutants, solids/liquid separation and treated effluent removal) in a single multifunctional volume basin in an alternating mode of operation. SBR systems are suitable for intermitted flows and require less space since all processes are taking place in the same reactor. The disadvantage of SBR is that the system is much more complex and fragile and requires precise control of timing, mixing and aeration.

### 1.1.2.2 Fixed Film Systems

Fixed film processes are based on the capacity of different microorganisms to grow on surfaces and develop a biofilm. Biofilm forms when bacteria adhere to surfaces in moist environments. Bacteria are held together in biofilms by Extracellular polymeric substances (EPS) allowing the development of a complex three dimensional, resilient attached communities. A biofilm community can be formed by a single bacterial species, but usually biofilms consist of rich mixtures of many species of bacteria as well as algae, protozoa and other microorganisms.

Biofilm formation is preferred in fixed film processes for various reasons such as greater substrate availability and protection from harmful environment (e.g. shear stress of flowing water) (Mara and Horan, 2003).

Biofilm systems have many advantages over suspended growth systems: The intrinsic resistance of biofilm communities to changing environmental conditions make...
based treatment systems more resilient to influent variation in toxicity and nutrient concentrations (Lear and Lewis, 2012). Another major advantage is the high microbial density that can be achieved, leading to smaller treatment system footprints, and the inherent development of aerobic, anoxic and anaerobic zones which enable simultaneous biological nutrient removal. However, formation of anaerobic zones could also be a disadvantage since it can cause the prosperity of unwanted bacterial communities. This formation of anaerobic zones is caused by a less controlled oxygen supply due to the complex structure of the biomass inside the biofilm.

The major types of fixed film technologies are described below.

**Trickling filters** are the oldest fixed film process. Wastewater is applied to the surface of the filter, which is simply a bed of rocks, gravel or plastic, and percolates down through the bed. Biofilm forms on the media and disintegrates the organic matter. As the biofilms grow thick, parts of them disconnect from the media, drop off, and a new layer of biofilms begin to grow. For this reason a sedimentation tank is needed after the trickling filter. Conventional trickling filters are round in shape with centrally mounted rotating arms for distribution of the wastewater. Nozzles on the arms spray the wastewater evenly across the media. Main advantage of this technology is its’ simplicity. This process has been used for nearly a century to provide low cost, low maintenance biological wastewater treatment. It is suitable for low flows since wastewater must be sprayed over the media. No aeration is needed because porosity of the media is high and ensures sufficient supply of oxygen. Disadvantages of trickling filters are: secondary sedimentation tank is needed because of dropping-off of biomass, sometimes (depending
on quality of the raw wastewater) primary sedimentation tank is also needed to avoid blocking of the media, high space requirements and disability to treat high flows.

**Sand filtration** is a technology that enables the removal of particles by adsorption and the removal of biodegradable organics by biomass accumulation. Sand filters are subdivided into slow and rapid types. Rapid sand types are operated at rates 20-50 times faster than the slow sand filters (SSF) (normally operated at flow rates between 0.1 and 0.4 m³/hr per square meter of surface), and hence require (in theory) only 2-5% of the area needed for slow sand filters. In practice, the reduction in space requirements is partially offset by the additional pretreatment stages needed for rapid filtration. Besides differences in approach velocities, other intrinsic differences exist.

In slow sand filtration fine sand is used, and the filter could run for weeks before cleaning. The suspended solids and colloidal matter are deposited at the very top of the filter (in association with a clogging layer known as Schmutzdecke), from which they can be removed by scraping off the surface layer. Attributes of the SSF are its low cost, operational simplicity, applicability to decentralized systems, and the possibility of it functioning as a stand-alone treatment system. Limitations of the SSF process are high land requirements, need for the occasional removal/ restoration of the Schmutzdecke and the perception that SSF is not a very robust process because of its shallow media depth (0.75-1.25 m) and over reliance on the Schmutzdecke for surface straining.

Rapid sand filtration (RSF) is a process that involves chemical pretreatment (coagulation/flocculation) and relies on depth filtration for removal of turbidity and microorganisms. The medium used for rapid filtration is considerably coarser (effective
grain size 0.6-2 mm). The impurities are deposited more rapidly and are carried more deeply into the filter and the necessity of cleaning (backwashing) occurs at frequent intervals (often only 1-4 days) (Huisman and Wood, 1974; Amy et al. 2006). An obvious advantage of RSF is its’ ability to handle higher flow rates. Disadvantages of the process are the need of chemicals addition prior to the filter and the need of backwash performance.

**Soil aquifer treatment** (SAT) is another fixed film process also known as infiltration. Unlike the processes described above, SAT is not performed in a controlled reactor, but in the natural environment. SAT is applied as tertiary treatment of wastewater. Water is recharged to the soil (usually sand) through infiltration basins. After the ground passage, water is collected underground in wells, and is later pumped out for further treatment or use. Depending on the water’s residence time underground and the hydrological situation, contaminates may not be removed by ground passage but only blended and diluted in some cases. Passage of water underground removes particles, bacteria, viruses, parasites, micro-pollutants, and other organic and inorganic compounds. (Kuehn and Mueller 2000). SAT has very low capital and operational costs and water quality after SAT is high and suitable for unrestricted irrigation. However, as noted, high water quality could be attributed to dilution and therefore contamination of the soil and the aquifer could occur.

**Biofiltration** is a process taking advantage of the ability of bacteria originating from the influents to biodegrade organic matter. This is accomplished by promoting conditions for the prosperity of biofilms. Biofilters are an advanced type of trickling filters, where water flows through the media and not sprayed- allowing higher flow rates. A sedimentation
tank is not needed after the biofilter since controlled backwashes are performed in order to avoid the blocking of the filter with excess biomass. Biofiltration is further discussed below.

1.2 Biofiltration

1.2.1 General Description

A biofilter is a fixed film reactor filled with granular material. Bacteria originating from the influents enter the biofilter and settle on the granular media. The media provide a location for colonization and growth of the biomass and protection from shear stress, allowing the formation of a biofilm. The dissolved organic matter is continuously supplied within the influents and is accumulated by the attached biomass, lowering the concentration of the organic matter in the effluents. Oxygen, which is crucial for biomass prosperity, is dissolved in the influents. In some biofilters, the oxygen concentration is increased by aeration.

Most commonly used types of media in biofiltration include sand, anthracite and spent granular activated carbon (GAC). In many cases, it has been shown that the use of GAC medium results in more bioactivity than that by the use of other filtration media, including anthracite and sand (Urfer et al. 1997).

In order to allow a biodegradation inside the biofilter the following criteria must be met: biomass concentration inside the reactor is increased (allowing the consumption of dissolved organic matter by more bacteria); oxygen is provided; optimal contact between
biomass, nutrients in the influents and dissolved oxygen is ensured, and the concentration of toxic substances within the reactor is below a critical level.

Studies investigating biofiltration report significant removal values, mostly of dissolved organic carbon (DOC) fraction (which consist of biodegradable dissolved organic matter). Removals of turbidity, total nitrogen and biopolymers were reported as well (Kalkan et al. 2011; Basu and Huck, 2004; Chinu et al. 2009; Hu et al. 2005).

1.2.2 Biofilter Operational Parameters

A biofilter is a simple reactor. Inlet stream is usually fed to the reactor by gravity. Inside the filter is the granular media, which typically has a large surface area to volume and at the bottom of the filter is the outlet stream. Backwashing of the filter is often needed for the removal of particles and excessive biomass- so usually backwashing water pipes exist. Some biofilters contain an inlet for aeration at the bottom of the filter- for preferred conditions for the proliferation of the biomass.

Various factors affect the removal of biodegradable organic matter during biofiltration. Two key parameters are contact time, expressed as empty bed contact time (EBCT), and the approach velocity (Zhang and Huck, 1996). Those parameters are calculated as follows:

\[ EBCT = \frac{V}{Q} \]  

where \( V \) is the volume of the biofilter and \( Q \) is the volumetric flow rate of influent stream.
\[ v = \frac{Q}{A} \] (2)

The approach velocity is represented by \( v \) and \( A \) is the cross-section bed area of the filter.

Those two parameters affect the growth of biomass. Velocity must be low enough and EBCT must be high enough to minimize shear stress and promote the growth of biomass and the accumulation of dissolved organic matter. But EBCT too high and velocity too low will limit the supply of organic matter to the biomass. Therefore an optimum must be reached.

Other factors influencing the effectiveness of the biofilter include the type of media, the concentration of dissolved oxygen, the concentration of biodegradable organic matter and backwashing parameters such as the flow rate, the frequency and the duration of a backwash.

1.2.3 Advantages and disadvantages of biofiltration

Biofiltration is a technology with many advantages. A biofilter is a simple low cost reactor, it could be easily built in every WWTP and it is a small reactor and therefore does not require large territory. A biofilter would be easy to handle and operate by inexperienced manpower. Biofiltration is a “green technology”- it is chemical free, leaves a small footprint in the effluents, does not require sludge treatment (no settling phase) and does not cause any environmental damage.

Biofiltration has some disadvantages. Periodic backwashing is required because of head loss due to solids accumulation and attached biomass growth. Another drawback is the
relatively low approach velocity needed to allow biodegradation by biomass. Low approach velocity means large reactor area for a given flow rate.

1.3 Wastewater Treatment in Israel

Israel has achieved the highest rate of water reclamation in the world- 75% of total treated raw sewage in 2005 (350 million cubic meters) (Mekorot). In most wastewater treatment plants, wastewater is brought to secondary level according to regulations published in the early 90’s. Most commonly used technology in WWTPs in Israel is the activated sludge process. Nowadays, with the understanding of the great potential in the utilization of effluents for irrigation thereby saving limited fresh water sources for domestic use, along with the damages stemming from utilization of low quality effluents, Israel’s water authority is implementing tertiary treatment in most WWTPs according to “Inbar committee regulations”- as the main goal is the production of tertiary effluents for unrestricted irrigation which will not damage soil or water sources (Inbar, 2007).

Most common advanced tertiary treatment in Israel is soil aquifer treatment (SAT) implemented to all “Shafdan” secondary effluents. The “Shafdan” is the largest WWTP in Israel treating 137 million cubic meters/ year of Israel’s domestic wastewater. The effluents are filtered through sandy soils, and are later pumped out of the aquifer and used for irrigation in Northern Negev (Shafdan WWTP).

Tertiary treatment produces high quality effluents which could be utilized for unrestricted agricultural irrigation (as well as landscape irrigation and process water in some industrial applications). Possible tertiary treatments are granular and membrane filtration, and disinfection. The filtration schemes differ by the hydraulic load that directly affects
the filtration velocity. The slow sand filtration and biofiltration are performed at relatively low velocities thus promoting the development of the biological layer during the filtration cycle. The membrane and rapid sand filtration are often performed by the conventional filtration scheme that includes the stages of coagulation, flocculation and filtration itself. Typically the microfiltration (for removal of suspended solids and large microorganisms like bacteria and protozoa) and ultrafiltration (suitable for the removal of viruses and organic macromolecules in addition to what is being removed by the microfiltration) membranes are used. Sometimes powdered activated carbon (PAC) is also added prior to the membrane and is used to treat water contaminated with dissolved organic matter and micro-pollutants (Wintgens et al. 2005).

Quaternary treatment can produce effluents of drinking water quality. High pressure membrane systems remove small organics and multivalent ions (nanofiltration NF) or even all dissolved species (reverse osmosis RO). Effective operation of NF and RO systems is depended upon avoiding conditions leading to fouling. Hence, tertiary treatment stage must be carefully chosen in regards to the composition of contaminates in the water.
1.4 RO fouling in wastewater reclamation

1.4.1 General

RO technology is a pressure driven process performed by a semi-permeable membrane. The pressure must be high enough to overcome the osmotic pressure which is generated by the different concentrations of solute on two sides of the membrane. RO membrane pores allow the passage of water, retaining the solutes. RO membranes are widely used in seawater desalination applications, because the major fraction which needs to be removed is inorganic salts. Use of RO technology in wastewater reclamation is not as common mainly due to the lack of sufficient research and practical experience in effective pretreatment. Pretreatment is a crucial step in wastewater desalination with RO membranes because of high concentrations of organics and particles in the effluents that cause high degrees of fouling on RO membranes.
In spite of all the research and advances in technology, membrane fouling is still one of the major shortcomings that membrane processes face. The accumulation or deposition of materials on the membrane or within its pores causes a reduction of productivity— the fouling results in decrease in the permeate flux or in significant increase of the feed pressure required to maintain the permeate flow.

The material that fouls reverse osmosis membranes is diverse, and is composed of inorganic particles (precipitated metal oxides and hydroxides, colloids, etc.), natural organic matter (NOM), and bacterial, fungal, algal, and protozoan cells. Dissolved organic matter has a significant contribution to fouling effects in RO membranes. Organics form cake/gel layer on membrane surface and reduce membrane permeability (Bourgeois et al. 2001). Studies investigating the characteristics of foulants show that dissolved extracellular polymer substances (EPS) or soluble microbiological products, normally identified as proteins and polysaccharides, are major organic foulants (Rosenberger et al. 2006; Liang et al. 2007). The rate and extent of fouling is a strong function of the quality of water applied to the membranes. It has traditionally been held that fouling material is the result of concentration and retention of constituents from the bulk. Another mechanism of fouling is the proliferation of organisms in biofilms on the membranes (Flemming et al. 1997).

1.4.2 Mitigating Membrane Fouling by pretreatment

There are several strategies used to control and minimize fouling including a feed pretreatment, a membrane modification, and a chemical cleaning. However, in terms of sustainability and membrane integrity, the best way to reduce fouling is by a
pretreatment, via the reduction of the amount of foulants that may come into contact with
the membrane and the minimization of their potential to cause problems. The common
types of pretreatment used when employing membrane processes in water and
wastewater treatment include the low pressure membrane filtration (microfiltration or
ultrafiltration), coagulation with or without sedimentation, and adsorption with powdered
activated carbon (PAC) (Malleviale et al. 1996).

Choice of pretreatment before RO membranes is a crucial step. Pretreatment steps will
directly affect the efficiency of the whole process in terms of cost, footprint and
environmental impacts.

In light of what was said above- tertiary treatment must be as efficient as possible,
meaning- pretreatment before RO membranes should remove as many fractions to the
greatest extent possible (within cost limitations). Theoretically- it would be best if RO
membrane would be left to “deal” only with salinity, which will minimize fouling to
scaling only. This approach claims that under some circumstances low pressure
membranes (MF or UF) could be “sacrificed” instead of RO membranes, for example.

Most effective and economically affordable pretreatment chain will include a
combination of pretreatment options- as each pretreatment is capable of the removal of
different impurities from the water.

Most common pretreatment to RO membrane is filtration through low pressure
membranes like MF and UF. As a simple physical process it is suitable for the production
of effluents with low concentration of suspended solids and organic matter. Secondary
effluents are fed to these membranes in the tertiary treatment stage. An alternative to this
“end-of-pipe” treatment is the application of membrane bioreactor (MBR) as a straight combination of biological treatment processes (e.g. activated sludge) and biomass retention by MF or UF membranes.

Low pressure membranes are attractive in wastewater treatment because this technology is able to produce water of uniform quality regardless of the normally wide variation in the concentrations or physicochemical properties of the wastewater influent. Low pressure filtration has proved to be efficient in lowering fouling effects in RO membranes (Wintgens et al. 2005). However, a number of major drawbacks of membrane utilization as tertiary treatment could be noted. Secondary effluents are water with high fouling potential. Fouling can cause severe flux decline, reduce membrane productivity and affect the filtrate quality (therefore increasing operational costs).

Reversible fouling can be removed by backwashing of the membrane, but physical cleaning might not be enough to maintain process stability. In the case of severe irreversible fouling - chemical cleaning is required (affecting operational costs and environmental impact).

Chemical cleaning is done with the use of acids, bases and oxidants. Cleaning actions are performed at low frequency (once in hours or days) due to high costs of chemical consumption, and irreversible damages to the membrane. Furthermore, chemical cleaning is a major disturbance to the sequence of the membrane operation due to long required soaking times of the chemicals on the membrane surface.

Besides the drawbacks of chemical utilization, low pressure membrane operation is not effective without the added treatment step of coagulation (direct filtrating of secondary
effluent was identified as an uneconomical strategy (Xie et al. 2006)). This is another disadvantage that should be taken into account when implementing MF or UF pretreatments.

Coagulation is the process by which colloidal particles and organic macromolecules initially present in a wastewater are combined into larger agglomerates. This is achieved by adding different kinds of chemicals (coagulants) which promote destabilization of the colloid dispersion and agglomeration of the individual colloidal particles. These “enlarged” foulants are then more easily retained by the membrane. Furthermore, addition of coagulants has been identified as a successful approach to minimize fouling in membranes treating secondary effluents (Felder et al. 2011; Kim et al. 2005). The main reason for this is that the aggregates created are too large to cause fouling effects like pore narrowing and pore plugging (Fan et al. 2007).

Pretreatment by the sequential process of coagulation and membrane filtration has proved to be an effective yet a high cost option with environmental shortcomings following the extensive use of chemicals.

Biofiltration is a possible pretreatment aimed at reducing the biologically degradable organic compounds that either contribute directly to organic fouling or provide nutrients for the development of biofilms on the membrane surfaces. Use of this technology encourages biofilm growth inside the biofilter as opposed to having it occur on the membrane (Flemming et al. 1997). The approach is ecologically sound because it uses biological processes to control subsequent biological activity and reduces the dependence on disinfectants. Biofiltration could be implemented as a stand-alone tertiary treatment,
before RO membranes or in combined treatment with coagulation and low pressure membrane filtration, lowering operational costs and chemical demands.

1.4.3 Previous Studies

In previous studies, biofiltration has been demonstrated to be effective in the reduction of fouling.

Zheng and co-workers (Zheng et al. 2009b) studied the effect of biological filtration of secondary effluents at filtration rates of 0.25-0.5 m/h prior to UF membranes. Results showed a decrease in concentration of biopolymers. The reported average removal rates for proteins, polysaccharides and biopolymers were 10%, 27% and 34%, and 6%, 19% and 26% for 0.25 m/h and 0.50 m/h approach velocities respectively. Results also showed a reduction in TMP development in the UF process: direct filtering of secondary effluents led to TMP increase from 220 mbars to 700 mbars within 12 h. Under the same operational condition, the operation time filtering biofilter filtrate was extended to around 30 days.

Mosqueda-Jimenez and Huck (2009) measured the effect of the biofiltration on the fouling behavior of nanofiltration (NF) membranes in drinking water treatment. Biofiltration was able to considerably reduce the rate of flux decline to one third or less of a decline observed when the NF membranes directly treated the water.

Basu and Huck (2004) conducted an integrated biofilter-immersed membrane study to determine the effect of placing a biofilter before or after a membrane for the treatment of humic type water. They found that when a bench-scale UF module was located after biofiltration, the membrane fouled at a lower rate. The placement of the equipment in the
study allowed for a direct comparison of the system as a membrane alone versus a biofilter alone or in a combined configuration. They found that the biofiltration added to TOC removal in comparison to the membrane alone or the membrane followed by a biofilter.

Chinu and co-workers (Chinu et al. 2009) compared biofiltration with different media (anthracite and GAC) as pretreatments for RO membranes and assessed the efficiency of the pretreatment by two fouling indices: modified fouling index (MFI) and slit density index (SDI). They concluded that biofiltration reduces a significant amount of organic matter and leads to a lower RO fouling. Both biofilters demonstrated similar fouling reduction behavior in terms of SDI and MFI.

Hu and co-workers (Hu et al. 2005) studied the feasibility of using biofiltration as a pretreatment process to control the biofouling of RO membranes. The 35–45% reduction in concentrations of dissolved organic carbon (DOC) was reported. The reduced biofouling increased the operational length five-fold to over 300 hrs.

The studies presented above give important insight on biofiltration removal abilities, and the potential of this technology as pretreatment of RO membranes in wastewater reclamation applications. However, operational information is still lacking. Pilot scale experiments are rare, and most researches are conducted in the controlled environment of a laboratory.

As of media types- the traditional use of sand in biofilter applications is the much more common, although there is growing interest in GAC media and reports on better removal
abilities are found. Still, little information can be found on operational parameters with
this media type.

Few of the many questions that remain open after deep searches in literature findings are:
under what operational parameters should biofilters operate? What are the dominant
fractions removed by this technology and to what extent? How does the approach
velocity affect removals? Should biofiltration act as a stand-alone pretreatment to RO or
should it be combined with other treatment options? What is the added value of combined
pretreatment and how should it be carried out?

This work will try and answer these and other questions.

1.5 The Current Research

1.5.1 Motivation

Although biological filtration is often implemented in gas purification, its use in
wastewater treatment is somewhat at its infancy. Several reports on bench scale lab
investigation of biofiltration are far from a full knowledge and precise examination of
operating parameters and removal abilities is lacking. As of now no reports on full-scale
biofiltration experiments with a continuous supply of secondary effluents of a municipal
WWTP exist.

The lab-scale investigation and the general knowledge gathered in the field suggest that
the biofiltration might have some advantages over the other pretreatment options in
advanced treatment of secondary effluents. These advantages are explored in the
pioneering study performed on a pilot plant located in Sede Teiman. The biofiltration as a
single and as a part of pretreatment was compared to and combined with coagulation – UF. The ultimate goal of the study was to reduce the fouling of RO quaternary treatment performed to obtain the water of a tap quality.

1.5.2 Research Goals and Objectives

The objective of this study was to evaluate the biofiltration of secondary effluents as a pretreatment to reduce fouling in RO membranes as a stand-alone technology and in combination with coagulation/ flocculation and UF membranes. This was accomplished by executing the following stages:

1. Examination of the fractions of secondary effluents removed by biofiltration and revision of the removal efficiencies of each fraction.
2. Evaluation of the effect of approach velocity on removal abilities.
3. Testing biofiltration combined with UF membrane, and analysis of the character of the effluents from the combined treatment.
4. Understanding feasibility of utilization of biofiltration as a pretreatment to RO membranes in wastewater desalination by assessing the degree of fouling and the retention of organic matter.
2. Material and Methods

2.1. Biofiltration columns- experimental setup

Secondary effluents were supplied by Sede Teiman WWTP to the pilot plant by an irrigation tap which was connected to the head tank of the biofilter (the secondary effluents are used by the WWTP for irrigation). The secondary effluents flowed from the head tank by gravity into downward-sloping pipes that were loosely connected to filtration columns located 2.5 m below the head tank. Such a loose connection promoted a raising pressure as the filtration column clogged during the run and enabled each experiment to be performed at a constant rate. Filtration rate was monitored using a flow meter and a control valve on the effluent line. The filtration column was constructed from a transparent, 10-cm diameter acrylic pipe and had a total height of 2.2 m. Seventeen sampling ports and 17 pressure ports were arranged in pairs on opposite sides of the column. Sampling ports had internal tubes that were extended 5 cm inside the column to avoid the collection of samples from filtration column walls. The filter was filled with 1.6 m of exhausted granular activated carbon (GAC) with effective size of 0.8 mm and a uniformity coefficient of 1.7 (AquaSorb 2000). Prior to the experiments the GAC was washed for 5 months with secondary effluents and therefore was considered exhausted. Figure 2 shows removal efficiency of the biofilter regarding \( \text{UV}_{254} \) analyses.
Figure 2- UV$_{254}$ removal efficiency of the biofilter with time. At $t=0$ the column was filled with new granular activated carbon.

It can be shown from figure 2 that after around 20 days stabilization is reached. Although flow rate was not maintained constant throughout these months, removal efficiency of UV$_{254}$ analyses stays in the range of 25-50%.

Filtration was performed at constant approach velocities of 0.75 and 2 m/h. Once a week the filter was backwashed for 20 minutes with tap water at a 150 L/hr flow rate. Backwash water was drained to waste. Schematic of the filter column is shown in figure 3.
2.2 Ultrafiltration pilot plant

The UF pilot plant, created and designed by *inge GmbH* (Greifenberg, Germany), consists of a 50 kDa molecular weight cut-off (MWCO) polyethersulfone (PES) UF membrane, packed in an "inside-out" hollow fiber module with total surface area of 1 m², operated under a semi dead-end mode (cross flow is applied between filtration cycles by forward flush cleaning). "Outside-in" backwash cleaning with filtrate water is performed between cycles. Pilot specification is shown in table 1.
<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pilot size</td>
<td>120<em>80</em>81 cm³</td>
</tr>
<tr>
<td>Module length</td>
<td>1.5 m</td>
</tr>
<tr>
<td>Total membrane surface</td>
<td>1 m²</td>
</tr>
<tr>
<td>Filtration pressure</td>
<td>200-800 mbar</td>
</tr>
<tr>
<td>Backwash pressure</td>
<td>1500-2800 mbar</td>
</tr>
<tr>
<td>Filtration flow rate</td>
<td>50-130 LMH</td>
</tr>
<tr>
<td>Backwash flow rate</td>
<td>120-300 LMH</td>
</tr>
<tr>
<td>Operational temperature</td>
<td>0-40 °C</td>
</tr>
</tbody>
</table>

Table 1 – Specification of the pilot plant (inge GmbH, 2011)

The pilot’s data logger stores the following over time: trans-membrane pressure (TMP), permeability, flux, filtrate pH, feed turbidity and temperature.

The pilot was fed by secondary effluents from Sede Teiman waste water plant, when connected to the biofiltration system, the reservoir tank of the UF system was filled with biofilter effluents. The standard mode of operation included 30 min filtration cycles with 60 LMH (L/m²/h) and 56 sec backwashes with 230 LMH. Process stability was investigated in a term of changes in TMP over time.

Basic diagram of the plant is shown in figure 4.
In-line coagulation was performed as follows: A 10 wt% Fe$^{3+}$ solution was prepared by mixing 1 kg granular FeCl$_3$·6H$_2$O (CARLO ERBA, Italy) with 1 L deionized water (DW), using standard magnetic stirrer. The solution was injected during the filtration to a 350 ml dynamic mixer (inge GmbH, Germany) in the feed line of the UF pilot plant by simple dosing pump (Watson-Marlow, UK). The contact time of the coagulant in the dynamic mixer was between 12 to 21 sec (depends on the flux) with additional residence time of 25–50 sec in the feed line before the membrane (Epsztein, 2012).

### 2.3 RO pilot plant

4 stainless steel test cells were designed and constructed at BGU workshop. The purpose of the test cells was to simulate a segment of a spiral wound modul. Characteristics of the RO pilot plant are presented in table 2.
Parameter | Characteristics
--- | ---
Channel dimensions | 896mm x 40mm x 0.7mm
Membrane area | 358 cm$^2$
Segment area | 114.4 cm$^2$
Material | Stainless steel membrane
Overflow velocity | 0.05-0.2 m/sec
Pressure | 12 bars

Table 2- Parameters of the RO test cells

The semi-automated RO pilot plant was placed in Sede Teiman WWTP. Feed stream was collected in a 40L tank and pumped into the system using a metering pump (M03 Hydracell, WANNER ENG. Minneapolis, USA) via a pulsation dumpener (WANNER ENG.).

Motor inverter was connected to the pump and trigged it to 19 H at all times, electrodes in the feed tank enabled automatic shutdown of the pilot when water level was low and re-activation when water was supplied again, thus maintaining continuous run. Volume flow was measured with flow meter (T&go, Israel). By - pass of the inlet was used for maintaining the desired cross-flow velocity. Flow velocity through the cells and system pressure were both maintained constant using 2 back-pressure regulators (BP-3, GO regulator, USA). The by-pass also maintained sufficient mixing in the feed tank so no stirrer was needed. Basic Flow chart of the system can be seen in Figure 5.
Trial procedure was as follows: new ESPA-2 membranes were taken from the module, cut into 50*910 mm pieces and placed inside the test cells. Twenty four - forty eight hours compaction step was performed, using tap water, until the flux remained stable, and this was used as the reference to normalize permeability later. After the compaction phase, the inlet to the pilot was replaced from tap water to treated/untreated secondary effluents, according to the experiments matrix.

Permeate was collected from each of the four test cells. Flux was calculated and water quality analyses were performed.

Change in feed water temperature significantly affected the transmembrane flux. Hence deviations in flux due to changes in the temperature were normalized to the reference temperature ($25^0C$) with the following empiric equation:

$$TCF = \exp\left(K \cdot \left(\frac{1}{273 + t} - \frac{1}{298}\right)\right)$$ (3)

where TCF is temperature correction factor, K is a constant characteristic for a given membrane material ($K=2700$ for ESPA-2 membranes), and $t$ is feed water temperature in
degrees Celsius. In this equation, a temperature of 25°C is used as a reference point, with TCF = 1. Due to frequent temperature changes, and the large amount of water, it was decided to calculate all performance results when they are all normalized to 25°C. \( J_{25} \) is calculated by multiplying the observed flux, \( J_{\text{obs}} \), with TCF.

\[
J_{25} = J_{\text{obs}} \cdot TCF \quad (4)
\]

RO flux was measured for each segment by dividing the flow through every exit point by the segment area (114.4 cm²). For precise unit conversion flux was calculated in m³/h and segment area in m². Eventually, permeability (specific flux - SF) was calculated by normalizing the measured flux (corrected to 25°C) by the calculated TMP (\( P_n \)) across the membrane,

\[
SF_{n,25} = \frac{J_{n,25}}{P_n} \quad (5)
\]

where \( J_{n,25} \) is normalized flux through the segment n, and \( SF_{n,25} \) is the specific normalized flux. \( P_n \) is the pressure of cell n calculated as follows:

\[
P_n = P_{in} - n \cdot 0.5 \cdot (P_{in} - P_{out}) \quad (6)
\]

where n is the number of test cell (1-4). Pressures were measured before the 1st cell (\( P_{in} \)), and after cell no. 4 (\( P_{out} \)), pressure loss within the cells was calculated for each cell (Felder, 2013).
2.4 Plan of Experiments:

The experiments performed as part of this work included the following:

1. **Biofiltration as a stand-alone tertiary treatment of secondary effluents:** two approach velocities were tested: 0.75 and 2 m/h. The media type used was Granular Activated Carbon (GAC). Water quality analyses were performed for raw influents (secondary effluents) and for samples at various filter depths: 60, 90, 130 and 160 cm. For each analysis removal efficiency was calculated.

2. **Biofiltration combined with coagulation and UF as tertiary treatment of secondary effluents:** Biofiltration column was connected to UF pilot plant. Media type for biofiltration column was GAC, and approach velocity was 0.75 m/h. The membrane used for the UF system was 50 kDa PES membrane. UF system operated at constant flux of 60 LMH, cycle time was 30 min and between cycles backwashing with filtrate (56 sec, 230 LMH) was performed. Water quality analyses were performed for the following samples: raw influents, biofiltration samples at depths 60, 90, 130 and 160 cm and UF effluents. Stability of UF system was examined as well and compared to UF pilot plant experiments without the biofiltration connection. The experiments were performed with and without 10 ppm ferric chloride coagulation.

3. **Biofiltration as a stand-alone pretreatment of RO membrane:** biofiltration column was connected to the RO pilot plant. Media type for biofiltration column was GAC, and approach velocity was 0.75 m/h. The membrane used for the RO system was ESPA-2. RO system operated at constant pressure of 12 bars. Water quality analyses were performed for the following samples: raw influents,
biofiltration samples at depths 60, 90, 130 and 160 cm and RO effluents from each test cell (1-4). The influence of the degree of fouling was assessed by permeability measurements.

4. **Biofiltration- coagulation- UF as combined pretreatment before RO membrane:** biofiltration column, coagulation-UFS system and RO pilot plant were connected in a raw. Media type for biofiltration column was GAC, and approach velocity was 0.75 m/h. Operational parameters for the UF system were as follows: UF membrane was 50 kDa PES, the system operated at a flux of 60 LMH, cycle time was 30 min and between cycles backwashing with filtrate (56 sec, 230 LMH) was performed. Coagulation was applied prior to UF membrane with concentration of 1 ppm ferric chloride. RO membrane was ESPA-2 and the system was operated at constant pressure of 12 bars. Water quality analyses were performed for the following samples: raw influents, biofiltration samples at depths 60, 90, 130 and 160 cm, UF effluents and RO effluents from each test cell (1-4). The influence of the degree of fouling was assessed by permeability measurements.

**2.5 Raw water (secondary effluents):**

Raw water was supplied from the secondary effluents of Sede Teiman WWTP. The WWTP treats 45,000 m$^3$/day of municipal wastewater by the activated sludge process that includes primary settling, biological degradation, and secondary clarification (Israel ministry of environmental protection).

Table 3 shows the average characteristics of the secondary effluents along with standard deviation values.
<table>
<thead>
<tr>
<th>Data measured in this research</th>
<th>Reported by Sede Teiman WWTP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Parameter</td>
<td>Average Value</td>
</tr>
<tr>
<td>DOC (mg/L)</td>
<td>13.45 ± 1.46</td>
</tr>
<tr>
<td>UV$_{254}$ (RAU)</td>
<td>0.276 ± 0.027</td>
</tr>
<tr>
<td>Phosphate (mg/L)</td>
<td>7 ± 3.2</td>
</tr>
<tr>
<td>Turbidity (NTU)</td>
<td>2.3 ± 0.5</td>
</tr>
<tr>
<td>pH</td>
<td>7.6 ± 0.2</td>
</tr>
</tbody>
</table>

Table 3- Sede Teiman wastewater treatment plant average effluent values

On the left side of table 3, data measured in this research (from January 2011 to January 2012) is presented. On the right side of table 3, data reported by Sede Teiman WWTP from March 2011 is presented (Israel ministry of environmental protection).

### 2.6 Effluent quality analyses

Samples were collected on daily basis from biofiltration columns. Samples were collected from raw water (secondary effluents) which were influent water and from several sampling ports in different depths within the biofilters: 60 cm, 90 cm, 130 cm and 160 cm. The 160 cm depth is in fact the biofiltration effluent sample, and was collected from the filtrate collectors. In case of an experiment which included UF membrane system or RO membrane system, membrane filtrate was also collected. Some of the analyses were performed on site (details are given below). The samples were transported to the laboratory at Ben-Gurion University once a day not later than one hour after their collection. In the laboratory, the samples were kept at 4 °C until the analysis in case it was not performed immediately.
The following analyses were performed:

The dissolved organic carbon was detected in the lab with Apollo 9000 (Teledyne Tekmar, USA) DOC analyzer using the standard method 5310B (standard methods, 1995). The samples were vacuum filtered through 0.45 µm (Tamar laboratories, Israel) prior to the analysis.

UV\textsubscript{254} absorbance was detected in the lab with a spectrophotometer (Agilent Technologies, 8453 UV-visible spectroscopy system, USA) using the standard method 5910B (standard methods 1995). The samples were vacuum filtered through 0.45 µm (Tamar laboratories, Israel) prior to the analysis.

Turbidity was detected on site with a turbidimeter (Hach 2100N, Hach Company, Loveland, Colorado).

Phosphate was detected on site with a colorimeter (DR/890, Hach Company, USA), using Molybdovanadate method: Hach method 8114. Phosphate determination reagent (Molybdovanadate reagent, Hach Company, USA) was used for this method.

LC-OCD (Liquid chromatography - organic carbon detection) analysis was performed at TU Dresden using LC-OCD (LC-OCD-OND New Model 8, DOC – Labor Dr. Huber, Karlsruhe, Germany). Samples were collected and pasteurized as follows: the samples were heated to 65-70°C and the temperature was kept constant for 15-30 minutes, then the samples were cooled down again and sent to Germany with a special tray of ice. The LC-OCD system is equipped with size exclusion column and online dissolved organic carbon, UV\textsubscript{254} and dissolved organic nitrogen detectors. The LC unit separates organic compounds according to their molecular size and the separated compounds are detected
by online detectors. The corresponding peak area could be converted into concentrations in mg C/L, mg N/L and UV 254 in 1/m. As state-of-the-art equipment, it has been used to analyze biofiltration removal abilities (Zheng et al. 2010a; Peldszus et al. 2011) and the fouling-causing substances in membrane processes (Zheng et al. 2009a; Zheng et al. 2010b).

Emission Excitation Matrixes (EEMs) were obtained with a fluorometer (SCINCO model FluoroMate FS-2, S. Korea) in the lab. EEMs were analyzed by principal component analysis (PCA). The principal component groups which were referred were based on a literature report (Chen et al. 2003). The principal component groups differ by different emission and excitation wavelength boundaries. The location of EEM peaks and the defined wavelength boundaries are shown in figure 6.

Figure 6- Location of EEM peaks (symbols) based on literature reports and operationally defined excitation and emission wavelength boundaries (dashed lines) for five EEM regions (Chen et al. 2003)
Figure 6 shows the location of the EEM peaks and the division for five different regions as follows: region 1 – “aromatic protein” (Emission wavelengths range: 280-330 nm, Excitation wavelengths range: 200-250 nm), region 2- “aromatic protein 2” (Emission wavelengths range: 330-380 nm, Excitation wavelengths range: 200-250 nm), region 3- “fulvic acid like” (Emission wavelengths range: 380-550 nm, Excitation wavelengths range: 200-250 nm), region 4- “soluble microbial by product like” (Emission wavelengths range: 280-380 nm, Excitation wavelengths range: 250-340 nm) and region 5- “humic acid like” (Emission wavelengths range: 380-550 nm, Excitation wavelengths range: 250-400 nm). Two main component groups were expected to be identified in both raw (secondary effluent) and biofilter effluent samples and therefore referred: “soluble microbial by-product like” (region 4) and “humic acid like” substances (region 5).

Fluorescence spectroscopy has received attention in the water industry as a potential monitoring technique. Fluorescence monitoring is attractive as it is a rapid, reagentless technique that requires no sample preparation prior to analysis. Fluorescence spectroscopy has been investigated as a monitoring tool for a range of applications including water and wastewater quality analyses. (Henderson et al. 2009).

Recent technological advances have allowed the rapid detection of 3- dimensional excitation–emission matrixes. Each EEM is a composite of emission scans from a single sample recorded at incrementing excitation wavelengths and arranged in a grid (excitation x emission x intensity).

The data provided from fluorescence analysis of each sample in this research includes a 2201 rows and 33 columns table of intensity units for each emission excitation.
wavelength. Using these data becomes a challenge. Extremely high peaks which represent self-reading peaks screen the relevant intensity peaks for each sample. SigmaPlot software was used to neutralize those self-reading peaks. Intensity readings over 2000 (self-reading) and below 200 (background noise) were ignored. In some cases readings over 1000 intensity units were ignored for a higher resolution of low organic content samples. Quantification of FIU (fluorescence intensity units) was achieved by summing FIU for each emission excitation couple in each region.

Examples of typical EEM graphs of secondary effluents along with boundaries for region 4 and 5 can be seen in figure 7.

Figure 7 – The EEM fluorescence analysis of raw water samples (secondary effluents) from Sede Teiman WWTP on different dates. Sum of FIU in each sample was as follows: region 4- 7.91, 10.01, 9.07, 10.55; region 5- 26.00, 32.23, 28.71, 30.47 in samples collected on dates: December 12, 19, 20, 21 2012 respectively (left to right). Samples were analyzed on the sampling date with a fluorometer and the graphs were achieved using SigmaPlot software.

Figure 7 depicts secondary effluent samples in different days of the experiment displayed with the aid of SigmaPlot. Average of sum of FIU for secondary effluents in region 5 was 29.28±1.67*10^6 FIU. Average of sum of FIU for secondary effluents in region 4 was 9.14±0.47*10^6 FIU.
Assessment of precision margins was achieved with triplicate measurements of the same sample depicted in figure 8.

Figure 8 – Triple repetition of the same sample taken on December 21, 2012. The intensity values were as follows: region 4: 10.55, 10.77, 10.41 *10^6 FIU; region 5: 30.47, 31.16, 30.19 *10^6 FIU for repetition number 1,2,3 respectively (left to right).

Fig 8 shows three EEM graphs of the same sample. Average readings for these samples were 30.60±0.49*10^6 FIU and 10.57±0.18*10^6 FIU for regions 5 and 4 respectively.

The linearity of fluorescence response of concentrated and diluted samples was examined by a gradual dilution of secondary effluents followed by EEM analysis. Results are shown on figure 9.
Figure 9 shows EEM results of gradual dilution of secondary effluents. Left side of figure 9 shows change in FIU with the dilution rate and the right side shows the graphs achieved by SigmaPlot. The results presented on Figure 9 manifest that indeed a linear correlation between the presence of organic matter within the sample and the fluorescence intensity can be claimed. Since the purpose of this curve was to assess linearity and not to convert fluorescence intensity to concentration, a minor number of points was sufficient.

The different analyses described above were performed for all samples and removal efficiencies (R.E.) were calculated as following:

$$R.E.(\%) = \left(1 - \frac{X_S}{X_R}\right) \times 100\% \quad (7)$$

Were $X_R$ and $X_S$ are the values of parameter $X$ in raw water (secondary effluents) and the specific sample respectively.
3. Results and discussion

3.1 Impact of influent velocity and filter depth on DOC and turbidity removal

DOC plays an important role in membrane fouling (Zheng et al. 2009a). Figure 10 shows the light absorbance at 254 nm (UVC range). It is generally known that above one mg/l of organic matter the UV$_{254}$ absorbance linearly correlates with DOC (Goren et al, 2008). The UV absorbance as a function of the filtration time showed a higher absorbance level in the feed than inside the filter. Inside the filter the lower absorbance was measured as the filter becomes deeper. In general it corresponded to an intuitively logical trend of a gradual removal of the organic matter inside the filter. The same trend was observed for the higher approach velocity of 2 m/h. The net reduction of organic matter measured as DOC in the biofilter effluents was 37.4% for 0.75 m/h and 30% at 2 m/h approach velocity.
Removal efficiency was calculated as the $\text{UV}_{254}$ ratio in the entrance and in the exit from the filter. The calculated ratios were placed as a function of the filtration time at the right part of Figure 10. Little differences in removal efficiencies were measured at different depths of the filter, as high removal efficiency was reached at the top of the filter. Small improvements in the removal efficiency at higher filter depths were explained by a gradual depletion of oxygen in deeper layers of the filter. A similar trend was observed in lab experiments performed earlier. Kalkan and co-workers (2011) reported that highest reduction in DOC level was observed at first 25 cm of the filter depth. The authors explained the observed by a presence of a gradient of biomass. If the gradient exists the
maximum biomass is at the top part of the biofilter and thus the highest removal is achieved at the filter layer that accumulates more biomass. Yapsaklı and Çeçen (2010) reported that 42% of the DOC was biodegraded in the first centimeters of the column and only additional 5% in the lower part of the column.

Results of turbidity measurements are shown on figure 11.

Figure 11- On the left side: turbidity analyses for 0.75 and 2 m/h approach velocities and 60, 90, 130, 160 cm filter depths. On the right side: removal efficiencies calculated through a UV$_{254}$ ratio in the effluent and the influent.

Figure 11 depicts changes in turbidity and its removal efficiency as a function of the filter depth. Although the turbidity level in the raw samples highly varied, similar absolute values were measured inside the filter. The difference between filter depths was not
significant. It seems logical and it is also supported by the previous findings (Zheng et al. 2010a, Halle et al. 2009) that the turbidity was removed by the attachment of turbidity particles to the media grains and not accumulated by biomass.

Higher removal efficiencies, in both UV$_{254}$ and turbidity analysis, were reached at the lower 0.75 m/h approach velocity. Low velocity reduces the shear stress inside the filter and increases the protection of biomass for a prosperous growth. Lower velocities are also preferred since more time is given for the accumulation of the dissolved organic matter by biomass. Zheng et al. (2010a) investigated the removal of organic foulants from secondary effluents and reported better performance of the biofilter at lower 0.25 m/h approach velocity, in comparison with 0.5 m/h. Hu et al. (2005) reported that the increase in EBCT from 7.5 to 60 min, meaning the reduction in the approach velocity, improves DOC removal. Similar trends were reported also in other studies (Reungoat et al. 2011, Peldszus et al. 2011).

### 3.2 EEM analysis of biofiltration samples

EEM graphs of samples collected from different filter depths during a typical biofiltration run are presented in figure 12.
As can be seen in figure 12, already at 60 cm depth the yellow-red region observed in the raw sample disappeared and much more of a blue color appeared. According to the scheme provided on the left side of Figure 12, the fluorescence intensity peaks reduced from the area of 1000-2000 FIU to the area of 500-600 FIU. Much less organic material can currently be detected in the sample in both 5 and 4 regions. Deeper inside the filter, minor changes in organic content can be seen (the differences stand out more when looking at the bottom row that presents a lower range of FIU). The presence of peaks of humic substances and biopolymer substances (tyrosine like and tryptophan like) in treated wastewater effluents was reported in other studies as well (Baker et al. 2001, Hudson et al. 2007, Henderson et al. 2009).

Quantification of EEM graphs is presented in figure 13.
Figure 13- Relative FIU with filter depth in approach velocities 0.75 and 2 m/h, and in two principal component regions: region 5 (R5 on the figure)- “humic acid-like” and region 4 (R4 on the figure)- “soluble microbial by product-like”. Filter media was GAC, and sampling was performed during 31 days.

Figure 13 shows relative FIU in EEMs of the two tested approach velocities: 0.75 m/h and 2 m/h for both 4 and 5 regions. Relative FIU was calculated as the ratio of FIU in the sample to the initial FIU at the depth of 0 cm (which is in fact the secondary effluents). Absolute FIU were 28.57±3.23*10^6, 10.17±1.50*10^6 FIU in regions 5 and 4 respectively when 0.75 m/h approach velocity was applied, and 29.28±1.67*10^6, 9.14±0.47*10^6 FIU in regions 5 and 4 respectively when 2 m/h approach velocity was applied. It can be seen that higher removal of both soluble microbial by-product like (region 4) and humic acid like substances (region 5) are obtained at lower 0.75 m/h approach velocity. Figure 13 also shows that the main removal for both principal component groups occurs at the first 60 cm of the biofilter.

Changes in fluorescence intensity with time and filter depth are shown on figure 14.
Figure 14- FIU for regions 4 and 5 in the different depths of the biofilter. Top two figures are regarding the lower 0.75 m/h approach velocity and bottom two figures are for the higher 2 m/h approach velocity.

Top two figures of figure 14 show changes in FIU for regions 5 (left) and 4 (right) when the lower 0.75 m/h approach velocity was applied. Bottom two figures show changes in FIU for regions 5 (left) and 4 (right) when the higher 2 m/h approach velocity was applied. When observing each one of the above graphs, it is clear that the main removal of both humic acid substances (region 5) and microbial by products (region 4) occur in the first 60 cm of the filter. Still additional removal of organic matter deeper inside the filter is observed. Comparing the two approach velocities – higher removal is reached at lower velocity. The new and interesting finding in these graphs is the similarity between the graphs of regions 4 and 5 for every approach velocity. For better analysis of these similarities, removal efficiencies for each velocity were compared in figure 15.
Figure 15 shows the removal efficiency of soluble microbial by-product like (region 4) and humic acid like substances (region 5), as a function of filtration time. A higher 80% retention was observed in experiment performed at lower 0.75 m/h approach velocity. An average 60% retention was observed in experiment performed at 2 m/h. An unexpectedly high similarity in retention percentage at both regions 4 and 5 at each approach velocity was detected. These results indicate that biofiltration removes humic acid like and microbial by product like substances to the exact same extent, which is highly unlikely.

Another experiment was conducted in order to understand these results: known concentration of humic acid alone, and bovine serum albumin (BSA) alone, were analyzed by EEM (data not shown). Results show the expected peaks of humic acid and protein in the typical emission-excitation location (region 5 for humic acid and region 4 for BSA). Then, the mixture of the two solutions was analyzed as well. EEM of the mixed solution shows humic acid peaks only and not protein peaks. These interesting results could explain the similar removal efficiencies for regions 5 and 4: humic acids screen the protein in the sample, so in fact what is seen in region 4 is not soluble
microbial by-product like proteins but some humic acid “tails”. No previous reports on the observed phenomenon were found in the scientific literature.

A number of analyses made are summed in figure 16 and differences in removal efficiencies in the two approach velocities- 0.75 m/h and 2 m/h are shown.

![Figure 16 – Removal efficiencies of biofiltration with approach velocities of 0.75 and 2 m/h in different analyses](image)

As can be seen in figure 16, all analyses result in higher removal efficiencies for the lower velocity. Turbidity was removed significantly in the lower velocity. Good correlation is found between analysis of UV$_{254}$ and TOC- as they both measure dissolved organic carbon (indirect and direct measurements respectively).

Phosphate removal was also higher at the lower approach velocity. Phosphate could be accumulated by biomass inside the reactor, but further analysis should be done in order to characterize the specific microorganisms.
The similar removal efficiencies at both regions 4 and 5 observed at EEM analysis were explained previously.

3.3 Liquid Chromatography - Organic Carbon Detection (LC-OCD) results

LC-OCD results of three raw samples (secondary effluents) in different days of the experiment are shown on figure 17.

Figure 17- LC-OCD analysis of three different raw samples collected on January 1st, 2nd and 16th 2012

Figure 17 shows black curves are for the organic carbon, green lines are for the organic nitrogen and blue lines are for the UV. The relative amounts of inorganic colloids, biopolymers, humics, building blocks, low molecular weight (LMW) acids and humic
substances (HS) and ammonium (urea) can be detected. A slight difference among three samples, mainly on the ammonium, was found.

LC-OCD analysis of biofiltration effluents in different filter depths and at different approach velocities: (a) 0.75 m/h and (b) 2 m/h are shown on figure 18.

It can be seen from figure 18 that in both approach velocities- clear decreases in peaks of humics and LMW acids are seen. Interestingly – the deepest sample (biofilter effluent) shows a new peak of nitrate in both approach velocities.
This interesting peak could be evidence of nitrification of bacteria inside the biofilter. This deepest sample is unique because it is after the biofiltration column but is also well aerated (the sample is collected from the filtrate collectors). Oxygen is obligatory for nitrification.

This nitrification peak is not present in any biofiltration samples besides the last sample, and is not present in the raw samples. This peak was also seen in UF effluents after connection to biofiltration columns since UF does not remove nitrate. Results are shown on page 61.

Figure 19 shows quantification of removal values shown in figure 18.
In figure 19- HOC and cDOC are hydrophobic and hydrophilic fractions of DOC respectively. Biopolymers, Humic Substances (HS), building blocks (BB), low molecular weight neutrals (LMWn) and low molecular weight acids (LMWa) are all fractions of cDOC. In the lower velocity (0.75 m/h) clear decrease in the amount of biopolymers is seen. This is an important result because the biomass inside a biofilter produces biopolymers (EPS), but a net reduction means a high biodegradation of these organic molecules. Biopolymers were highly removed only at the lower velocity and at the deepest sampling port. The probable reason for the observed trend is the higher contact time needed for the biodegradation of biopolymers.

Biopolymer removal is significant to pretreatment stage before RO because it has been reported that the fraction of the large dissolved organic molecules, identified as
biopolymers with LC-OCD, causes significantly more membrane fouling than other DOC fractions (Zheng et al. 2009a; Zheng et al. 2010b).

Figure 20 shows removal values (for biofiltration effluents) for all LC-OCD fractions in the two approach velocities.

Figure 20 - Removal values in the different LC-OCD fractions for both approach velocities 0.75 and 2 m/h, sampling was on December 29 and January 16, 2012 respectively.

Figure 20 shows removal in all DOC fractions. Lower velocity resulted in higher removals.

3.4 A combination of biofiltration and Ultrafiltration membrane

Biofiltration effluents were used as influents for ultrafiltration. The goal was to understand removal abilities of the combined pretreatment in comparison to a single treatment by each system. Added advantage of coagulation was also examined as the experiment was conducted with and without 10 ppm ferric chloride coagulation. That
concentration of the coagulant was defined as the “optimal dose” for the UF system with 50 kDa PES membrane in a previous study (Epsztein, 2012). Biofilter was operated at approach velocity of 0.75 m/h which was found (in this work) to be more effective in removal of impurities from the secondary effluents of two tested approach velocities.

Removal efficiencies of different pretreatments as calculated from results of three analyses: UV$_{254}$ and TOC, which are two independent methods for the detection of the organic matter in a sample, and PO$_4$ analysis are shown on figure 21.

Figure 21 depicts removal efficiencies of biofiltration as a single treatment, ultrafiltration as a single treatment and combined treatment of biofiltration-ultrafiltration. The dark grey columns represent biofiltration samples and coagulation is irrelevant for these results (coagulant is added prior to UF system). Light grey columns represent UF without (left figure) or with (right figure) coagulation, and stripped columns represent the combination.
of the pretreatments (without or with coagulation). UF pretreatment without coagulation results in a very low removal values. The removal of organic matter (analyzed by UV$_{254}$ and TOC) is relatively similar in biofiltration and UF with 10 ppm coagulation pretreatments, but removal values of the combination of the pretreatments is significantly higher. This result suggests that different fractions are removed in the two pretreatments and there is an added value in a combination of these two pretreatments.

Previous study also showed improvement in effluent quality when UF membrane was placed after biofiltration (Basu and Huck 2004).

Figures 22 and 23 show EEM graphs of raw water samples, biofiltration effluents and ultrafiltration effluents from the combined treatment of biofiltration followed by ultrafiltration without and with coagulation (figures 22 and 23 respectively).

Figure 22- EEM’s of raw samples, biofiltration effluents and UF effluents without coagulation prior to UF treatment. Samples were collected and analyzed on January 1st 2012. Top figures show values in the range of 200-2000 FIU, bottom figures show values in the range of 200-1000 FIU.
Figure 23- EEM’s of raw samples, biofiltration effluents and UF effluents with 10 ppm ferric chloride coagulation prior to UF treatment. Samples were collected and analyzed on January 2nd 2012. Top figures show values in the range of 200-2000 FIU, bottom figures show values in the range of 200-1000 FIU.

Figure 22 visibly shows the removal of humic acid like substances and microbial by product like substances in different samples of a combined pretreatments experiment. Significant removal is achieved after biofiltration but there is no further removal after UF membrane without coagulation. The last EEM even shows an increase in the fluorescence intensity after UF, probably due to a minor contamination of the permeate in pipes, a reservoir or membrane system. The results of EEM with an addition of the coagulation step show an additional removal after UF treatment, as can be seen on figure 23.

Figure 24 presents the quantification of the matrixes presented in figures 22 and 23.
Figure 24- FIU values for regions 4 and 5 in different treatments: raw influents (secondary effluents), biofiltration effluents, combined pretreatment of biofiltration- UF without coagulation and combined pretreatment of biofiltration- UF with 10 ppm ferric chloride coagulation. Samples were taken on January 1st and 2nd 2012 for the experiments without and with coagulation respectively. Raw and biofiltration FIU values are an average of the samples taken in those two dates.

In average the fluorescence intensity decreased from $27 \times 10^6$ and $9 \times 10^6$ FIU in raw samples to $7-9 \times 10^6$ and $2-3 \times 10^6$ FIU after pretreatment in regions 5 and 4 respectively.

According to the observed fluorescence levels more than 70% of the organic matter is retained by a pretreatment. Along with that the biofiltration followed by a UF without coagulation displayed fluorescence levels higher than the levels observed after the biofiltration alone. The addition of coagulation at the UF stage resulted in a continuous improvement of the filtrate quality in both regions 4 and 5.

Figure 25 shows LC-OCD analysis of raw samples and UF effluents.
Figure 25- LC-OCD curves for raw sample (bottom curve) taken on dates January 1st and 2nd 2012, filtrate of combined treatment biofiltration-Uf, without coagulation taken on January 1st (middle curve) and filtrate of combined treatment biofiltration-Uf, with 10 ppm ferric chloride coagulation taken on January 2nd (top curve).

In figure 25, top curve belongs to UF filtrate after biofiltration and 10 ppm ferric chloride coagulation. Middle curve belongs to the UF filtrate sample after biofiltration, without coagulation, and bottom curve belongs to the raw sample. As can be seen in figure 25, the peak of biopolymers which was present in the raw sample, was completely removed by UF in both samples. The peaks of the other DOC fractions (humics, building blocks and LMW acids and HS) were diminished in UF samples in comparison to raw sample. Nitrate peak is evident in both UF samples and is not seen in the raw sample, a result which was discussed earlier. Quantification and further analysis of these LC-OCD results are shown below in figure 26.
Figure 26 – Concentrations of organic matter in ppb of organic carbon in raw, biofiltered, biofiltered and UF treated and biofiltered, coagulated with 10 ppm FeCl₃ and UF treated samples. Here DOC is the dissolved organic carbon, HOC is the hydrophobic organic carbon and CDOC is hydrophilic organic carbon. Details of CDOC concentrations as a function of the treatment are detailed on the right graph. The samples were collected at January 1st 2012 for the experiment without coagulation and on January 2nd 2012 for the experiment with coagulation.

Figure 26 shows LC-OCD results of raw water samples, biofiltration effluents and UF effluents (after biofiltration) with and without coagulation. A total DOC level in analyzed raw samples is 11 ppm, among them there are approximately 1.5 ppm hydrophobic organic carbon and 9.5 ppm hydrophilic organic carbon. The hydrophilic organic carbon comprises of 1 ppm biopolymers, 4 ppm humic substances, 2 ppm building blocks and 2.5 ppm of low molecular weight neutrals. A negligible quantity of low molecular weight acids did not contributed much to the overall balance. The retention of all fractions of organic carbon after a treatment was detected. The most significant retention was observed in biopolymers. When the biofiltration did not retain almost any biopolymer, the UF treatment with and without coagulation removed the biopolymers completely. Between 20 and 30% retention was observed in all other fractions of hydrophilic organic carbon. More than 70% of the hydrophobic carbon was retained. Somewhat better
removal of humic substances with the addition of coagulant was attributed to a presence of a charged fraction of the organic carbon that was neutralized by the coagulant. No difference in the retention of other fractions with the addition of the coagulant was observed.

Almost same organic carbon levels were observed after each pretreatment (results were also compared with LC-OCD results of UF treatment with and without coagulation without prior treatment by biofiltration (Epsztein, 2012)). Not only there is no difference in the retention after biofiltration and UF, the combination of the two has no gain over a single pretreatment. In a choice between the biofiltration and UF the latter eliminates completely the biopolymers, a most persistent foulant in RO membranes, and in that sense the UF compares preferably to the biofiltration.

Figure 27 shows the stability of the membrane system after different pretreatments.
In figure 27, as expected and indeed observed, the direct UF treatment of secondary effluents resulted in unstable operation. The TMP raise per each 30 min cycle increased and already after 3 h of operation the TMP at the end of a cycle increased up to 500 mbars level. With a maximal 800 mbars allowable, it is likely that a chemical cleaning will be required before 24 h as during a normal operation. The biofiltration as the pretreatment improves the stability of the operation and for the displayed 3 h operational time shows that a stable UF operation can be achieved even without a coagulant.
3.5 Biofiltration followed by RO membrane

The ultimate test for the efficiency of the pretreatment was the permeate flux of RO membranes. For that the biofilter permeate was used as a feed of 4 RO modules connected in line.

Figure 28- Normalized permeability of the four cells with time. Biofiltration effluents were used as influents for RO. Biofilter was operated at approach velocity of 0.75 m/h. Operational parameters for the RO system were as follows: RO membrane was ESPA-2, and was operated at constant pressure of 12 bars.

Figure 28 depicts the normalized permeability drop with time for each of the four test cells in the RO system. No meaningful difference can be seen between the test cells. If recovery rate was high enough a different trend would have been expected since the first test cell would suffer more from scaling, which would affect permeability. However, the RO system in the experiment was maintained with relatively low recovery rate (~30%). The correlation between scaling and recovery rate was presented in previous studies (Glueckstern et al. 2008; Vrouwenvelder et al. 2010; Greenberg et al. 2005). Lack of a
trend in a comparison between the test cells was observed in a previous study with the same system and similar operation parameters (Felder, 2013). The small differences in relative permeability were attributed to differences in membrane surface properties originated in the production process.

Figure 29 shows changes in relative permeability (normalized to 25°C) with time. Low pressure membranes used as pretreatments were UF (50 kDa) and MF modules (0.1 micron). Two coagulant doses were used: 10 ppm and 1 ppm named optimal and minimal dose respectively. The optimal dose enhances the filtrate quality and recovery rate, and the minimal dose is a dose that still allows a stable long term operation (Epsztein 2012). Results of UF and MF pretreatments were taken from a previous study with the same RO system and operation parameters (Felder 2013).

Figure 29- Normalized permeability (to 25°C) drop with time for different RO pretreatments: no pretreatment (direct connection of secondary effluents to RO system), UF and MF “optimal” and “minimal” with coagulant dosing of 10 and 1 ppm ferric chloride respectively (data taken from previous work (Felder, 2013)) and biofiltration pretreatment.
As can be seen in figure 29, introduction of secondary effluents directly to the RO system resulted in a massive fouling that caused a sharp permeability drop (80% flux decrease). Pretreatment of low pressure membranes was demonstrated to be effective in terms of stability of the RO system. The expected trend can be seen—smaller membrane pores and higher coagulant dose result in a more moderate permeability drop. Pretreatment with biofiltration mitigated the sharp drop in comparison to no pretreatment, but still stability was not preserved.

3.6 Combined pretreatment: biofiltration, coagulation and ultrafiltration prior to RO membrane

Figure 30 expands the scope of results presented in figure 27 with an addition of the combined pretreatment: biofiltration following coagulation and ultrafiltration.
Figure 30- Normalized permeability (to 25°C) drop with time for different RO pretreatments: no pretreatment (direct connection of secondary effluents to RO system), UF and MF “optimal” and “minimal” with coagulant dosing of 10 and 1 ppm ferric chloride respectively (data taken from previous work (Felder, 2013)), biofiltration pretreatment and combined biofiltration-coagulation-UF pretreatment. Biofilter was operated at approach velocity of 0.75 m/h. Operational parameters for the UF system were as follows: UF membrane was 50 kDa PES, the system operated at a flux of 60 LMH, cycle time was 30 min and between cycles backwashing with filtrate (56 sec, 230 LMH) was performed. Coagulation was applied prior to UF membrane with concentration of 1 mg/L ferric chloride. RO membrane was ESPA-2 and the system was operated at constant pressure of 12 bars.

As can be seen in figure 30, at the first 12 days (288 hours) of the experiment, pressure drop was sharp and the combined pretreatment presented similar relative permeability like were presented in the experiment with no pretreatment at all. After the 12th day (~300 hrs) a sharp elevation in permeability was seen. At the same time- a sharp rise in temperature also occurred. Comparison between relative permeability and temperature rise can be seen in figure 31.
Figure 31- Normalized permeability drop in RO membrane of combined pretreatment: biofiltration-coagulation-UF and temperature changes with time.

Figure 31 depicts changes in temperature and normalized permeability of RO permeate after the combined pretreatment of biofiltration-coagulation-UF. It can be seen that the sharp elevation in permeability occurred at the same time with the rise of temperature. Temperature can affect RO membrane performance as well as pretreatment performance.

Water quality was measured via the analysis of absorbance in UV_{254}. Results are presented in figure 32.
Figure 32- Absorbance in UV$_{254}$ analysis of raw samples, biofiltration effluents, UF effluents and RO effluents with time in the combined pretreatment. Biofiltration operated at 0.75 m/h approach velocity with GAC medium, UF operated with 50 kDa PES membrane, at a flux of 60 LMH, cycle time was 30 min and between cycles backwashing with filtrate (56 sec, 230 LMH) was performed. Coagulation was applied prior to UF membrane with concentration of 1 mg/L ferric chloride. RO membrane was ESPA-2 and the system was operated at constant pressure of 12 bars.

Figure 32 shows light absorbance at 254 nm. Oppositely to the previous trends displayed and discussed above, the same UV$_{254}$ levels in raw, biofiltered and biofiltered-UF treated samples were observed. The only observable difference was between the UV$_{254}$ levels in the 3 above samples and the RO permeate. The average 0.05 RAU after RO membrane was much lower than the average 0.25 RAU in raw and pretreated samples. The average 75% retention of organic matter can be viewed as a good sign for the retention of the organic matter, although it looks like a direct treatment of organics by the RO membrane. The general results of this approach is a severe flux drop in RO membrane, which was in fact observed on Figure 30.
4. Conclusions

The current study compares different pretreatment options for the production of high-quality tertiary effluents. The feed stream of secondary effluents from a wastewater treatment plant Sede Teiman was forwarded through biofiltration, UF and coagulation-UF operations. Effluents were treated with ESPA-2 RO membranes. Each experiment was performed for a period not less than three weeks. A suitability of the treatment was assessed by the achieved retention rates and by the flux through UF and RO membranes.

The idea of using biofiltration as a pretreatment to RO membranes is relatively new and was not discussed before. Single reports coming from short lab experiments do not really serve as a base for conclusions drawn in this study. In that sense the performed study is first of a kind.

Biofiltration can serve as a sole pretreatment before the RO especially when the price for the entire setup is important. The advanced wastewater treatment is definitely one of the niche applications when the option of biofiltration can be considered. Biofiltration can achieve the same level of treatment as the UF without coagulant. The difference is in the retention of biopolymers when the UF pretreatment removes biopolymers completely and biofiltration can remove between 20 and 30% only. The retention of biopolymers is important to minimize the RO fouling. Indeed the flux drop in RO system in experiments performed with UF pretreatment was much less than in experiments performed with biofiltration. Interestingly the biofiltration followed by UF although completely removes the biopolymers still releases some nitrogen-containing compounds that significantly reduce RO flux.
Biofiltration can serve as a stand-alone advanced wastewater treatment. The biofiltration displays 30-40% reduction in concentrations of humic substances, building blocks, low molecular weight neutrals and hydrophobic compounds. The same degree of retention is achieved by UF without coagulation. The addition of a coagulant improves the retention of hydrophobic compounds and of humic substances, two fractions of organic carbon that contain charged compounds. Still the difference in the retention of biopolymers is critical for the RO operation. The biofiltration was effective in the retention of turbidity and phosphorus. The turbidity level was kept below an average of 1 NTU regardless of variations in turbidity values in the influents. Phosphate removal reached average of 70%. Higher efficiency of biofiltration was constantly observed at lower 0.75 m/h approach velocity. Most of removals were observed at the first 60 cm of the filter depth. There is no gain in a combined biofiltration- UF pretreatment. Both treatments retain the same compounds and their combination just increases capital and operational costs. Still even after the retention of 40% of organic matter the operation of RO produces a permeate stream that contains some organic compounds. In general more than 90% retention of organic matter is observed. The retention level is below the expected.
5. References


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