NUMBER B 2219 | DECEMBER 2015 | **REPORT**

Reuse of treated wastewater for non-potable use (ReUse)

Final report

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Funded by: Foundation for IVL Swedish Environmental Research Institute (SIVL) and Xylem Inc.
Photographer: Hammarby Sjöstadsverk
Report number: B 2219
Edition: Only available as PDF for individual printing

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This report has been reviewed and approved in accordance with IVL's audited and approved management system.

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Summary

Population growth, increasing living standards, but also environmental hazards with global climate change as the most significant are all contributing to an increasing water stress in many parts of the world. While access to fresh water for drinking water is getting more costly due to environmental pollution, uses of drinking water conflicts with water needs for agricultural and industrial use, which are in need of substantial water quantities. The use of reclaimed wastewater for non-potable purposes provides a solution for this. This is not new and has in fact been applied in many regions as the main water management approach. As water scarcity becomes more severe, also the need for more sustainable and holistic approaches to deal with our limited fresh water resources becomes more and more obvious. The traditional one-way water handling approach, with end-of-pipe treatment releasing “clean” effluent water to nature, has to be converted into a society-internal water reuse scheme where different water qualities and water uses are considered as an integral part of the water cycle.

The present report presents activities and results from an international project that aimed at developing and optimizing water treatment processes and systems for sustainable reuse of treated wastewater. The starting point is to combine the sequential batch treatment (SBR, sequencing batch reactors) with different conventional and emerging secondary and tertiary treatment techniques in various combinations, optimized from an overall sustainability perspective. Evaluation and optimization is achieved using life cycle assessment and life cycle cost assessment and their combination.

The ReUse-project worked with eight different lines comprising various state-of-the-art technologies combined differently and targeting various effluent qualities for agriculture reuse, industrial reuse or groundwater recharge. Contaminants investigated include a wide range of standard and emerging micropollutants as well as ecotoxicity. Further, greenhouse gas emissions were measured. Besides pilot-scale data, data from a number of full-scale treatment plants were used for the environmental and economic impact assessments.

Results show that the ***different treatment system setups can meet designated reuse effluent quality requirements.*** Moreover, an optimization of the treatment systems could be achieved for an improved resource efficiency of the treatment. New knowledge about operating, designing, controlling and combining various treatment processes was gained and implemented in reality by the project partners. Depending on treatment requirements, different treatment systems have been made available for implementation. Preferred options for various targeted substances and operational conditions/prerequisites are described in the report.

The report provides information for each reuse application about which treatment system that has the lowest environmental impact and best effluent quality. Impacts of various aspects such as additional nutrient removal and chlorination to achieve groundwater recharge qualities are reported and discussed. For industrial reuse, the lowest environmental impact was achieved with the line including submerged ultrafiltration and UV. The industrial reuse line that produced the best effluent quality with lowest con-

centrations of micropollutants has the highest environmental impact. For groundwater recharge, the treatment system including sand filter has the lowest environmental impact. The other groundwater recharge treatment systems are actually reaching industrial effluent quality but at a higher total environmental impact.

Different decisive system parameters are investigated and their impact on the overall environmental impact illustrated. Using for example anaerobic instead of aerobic stabilization of produced sewage sludge decreased the total global warming impact with 60%. In addition, the origin of the used electricity and nitrous oxide emissions from the secondary treatment (especially in agriculture reuse mode) have a significant impact on the global warming impact and other impact factors. The largest negative impact of reusing sewage sludge as a fertilizer is, however, for the terrestrial ecotoxicity.

The aggregation of LCA results including three different plant sizes (20 000, 100 000 and 500 000 person equivalents) shows that increased environmental impacts, caused by higher quality targets with more advanced treatment processes, become less significant with increasing plant size. Higher quality targets do not automatically imply an increase of environmental impacts. Instead, poorer water treatment can increase the environmental impact when both the treatment process and the downstream effects of, e.g. substances in the effluent are considered. ***The project indicates that the total environmental impact of the optimized ReUse-systems can be lower than for baseline scenario representing traditional treatment.***

Economic evaluation of eight studied reuse solutions showed that investment costs (CAPEX) of different treatment systems are not directly related to an increased effluent quality. Operating costs (OPEX), however, are generally increasing with increasing effluent quality. The sum of investment and operating costs over a whole plant lifetime, i.e. the Life Cycle Costs per m^3 of treated wastewater, decreases as the size of the plant increases. The project further showed that individual processes can have a significant impact on the overall treatment train costs and the LCC assessment provides a helpful tool to identify specific components or processes with high costs.

LCC evaluation of different ReUse trains also revealed that tertiary treatment steps, necessary for achieving a water quality corresponding to reuse quality standards, only contribute by few percent to the overall LCC of a treatment train. ***The evaluation showed that costs for producing water for different reuse applications are lower than reported costs for existing conventional sewage treatment plants in Sweden.*** This is true despite the fact that the economic benefits of reusing wastewater are not taken into account.

The project concludes that sustainable treatment systems for wastewater reuse require aggregation of environmental impact, cost and achieved water quality evaluations.

Results provide a clear indication that ***wastewater reuse for various reuse purposes is feasible without increasing the total environmental impact and without increase in costs and at the same time fulfilling regulation targets***. Various technical aspects that have to be considered are presented and implications discussed. Wastewater reuse is, as shown in this project, a both technically, environmentally and economically sustainable solution. This may also be true for standard effluent discharge to nature.

Sammanfattning

Befolkningstillväxt, ökad levnadsstandard, men också miljörisker, där den globala klimatförändringen är mest påtaglig, bidrar alla till en ökad vattenstress i många delar av världen. Samtidigt som tillgång till färskt vatten blir dyrare på grund, på grund av ökad vattenstress, uppstår konflikter mellan olika sektorer som är i behov av stora vattenmängder, såsom jordbruk och industri. Återanvändning av avloppsvatten spelar en nyckelroll för att kunna lösa denna problematik. Detta är inget nytt och har redan använts i flera regioner med vattenbrist som huvudalternativ för en uthållig vattenhantering. Med vattenbristen som blir allt mer påtaglig så ökar också behovet av mer hållbara och holistiska tillvägagångssätt för att ta itu med våra begränsade sötvattenresurser. Den traditionella enkelriktade vattenhanteringsstrategin med end-of-pipe rening som sedan släpper ut "rent" avloppsvatten till naturen, är i behov av uppdatering och anpassning till ett samhälle med systemintern återanvändning där olika vattenkvaliteter och vattenanvändning betraktas som en integrerad del av vattnets kretslopp.

Denna rapport presenterar aktiviteter och resultat från det internationella projektet ReUse som syftade till att utveckla och optimera vattenreningsprocesser och -system för en hållbar återanvändning av renat avloppsvatten. Utgångspunkten var att kombinera sekundär rening i en SBR (sekvenssatsreaktorer) med olika konventionella och nya kompletterande behandlingstekniker i olika kombinationer, och att optimera driften ur ett helhetsperspektiv. Utvärdering och optimering gjordes med hjälp av livscykelanalyser (LCA) och livscykelkostnadsbedömningar (LCC) och deras kombination.

ReUse-projektet arbetade med åtta olika vattenreningslinjer som består av olika state-of-the-art-reningstekniker som kombineras på olika sätt inriktade på olika kvaliteter på utgående vatten beroende på dess återanvändningsändamål; jordbruk, industri eller återföring till grundvatten. Föroreningar som undersöktes inkluderar ett brett utbud av standardföroreningar, mikroföroreningar samt ekotoxicitet. Vidare har utsläppen av växthusgaser mätts. Förutom data framtagen genom tester i pilotskala har även data från ett antal befintliga reningsverk i fullskala använts för både LCA och LCC.

Resultaten visar att de olika ***ReUse-reningssystemen kan möta specificerade kvalitetskrav för de olika återanvändningsområdena.*** En optimering av reningssystemen kan uppnås för en förbättrad resurseffektivitet. Ny kunskap om drift, design, styrning och hur olika reningsprocesser kan kombineras togs fram och implementeras redan i praktiken av en projektpartner. Beroende på behandlingskraven har olika reningssystem ställts till förfogande för implementering. Lämpliga alternativ för rening mot olika krav, driftförhållanden och förutsättningar beskrivs i rapporten.

Rapporten ger information om varje ReUse-reningssystem, vilket system som har lägst miljöpåverkan och bästa reningseffekt. Inverkan av olika aspekter som exempelvis en mer effektiv rening av näringsämnen och klorering för att kunna återföra renat avloppsvatten till grundvatten beskrivs och diskuteras. Vid industriell återanvändning uppnås t.ex. den lägsta miljöpåverkan med ett reningssystem med ultrafiltrering och UV. Det system för industriell återanvändning som producerade den högsta vattenkvalitén med lägsta koncentrationerna av mikroföroreningar har även den högsta miljöpåverkan. För reningssystem för återföring av renat avloppsvatten till grundvatten har

system med sandfilter lägst miljöpåverkan. Övriga reningssystem för återföring till grundvatten uppnår industrikrav men med en högre total miljöpåverkan.

Olika systemparametrar utreds och deras inverkan på den totala miljöpåverkan illustreras. Användning av exempelvis anaerob istället för aerob slamstabilisering minskar den totala klimatpåverkan med 60%. Dessutom har energiursprunget och lustgasutsläpp från sekundär rening (särskilt vid minskad kväverening för återanvändning vid bevattning) en betydande miljöpåverkan. Den största negativa effekten av att återanvända avloppsslam som gödsel på åkermark är för markbunden ekotoxicitet.

Aggregering av miljöpåverkan för tre olika anläggningsstorlekar; 20 000, 100 000 och 500 000 personekvivalenter, visar att även om högre kvalitetskrav som kräver mer avancerade reningstekniker ger en ökad miljöpåverkan, så blir detta mindre betydelsefullt med en ökande anläggningsstorlek. Högre kvalitetskrav innebär alltså inte automatiskt en ökning av miljöpåverkan. Istället kan även en sämre vattenrening leda till en ökad miljöpåverkan när både reningsprocessen och relaterade effekterna beaktas. Projektet visar att ***den totala miljöpåverkan av de optimerade ReUse-reningssystem kan vara lägre än för dagens traditionella avloppsvattenrening***.

Den ekonomiska utvärderingen av de åtta studerade ReUse-reningssystemen visade att investeringskostnaderna (CAPEX) av olika reningssystem inte är direkt relaterade till en ökad vattenkvalitet. Driftskostnaderna (OPEX) ökar emellertid i allmänhet med ökande vattenkvalitet. Summan av investerings- och driftskostnader över en hel anläggningslivstid, dvs livscykelkostnaderna (LCC) per m^3 renat avloppsvatten, minskar med ökande anläggningsstorlek. Projektet visade dessutom att individuella reningsprocesser kan ha en betydande inverkan på de totala kostnaderna av ett reningssystem och att LCC-bedömningar ger ett användbart verktyg för att identifiera specifika komponenter eller processer med höga kostnader.

LCC-bedömningen av olika reningssystem visade också att tertiära reningssteg, som möjliggör en ökad vattenkvalitet och därmed återanvändning, endast bidrar med några procents ökning av den totala livscykelkostnaden.. Utvärderingen visade att ***kostnaderna för att rena avloppsvatten så att det kan användas för olika återanvändningsändamål kan vara lägre än redovisade kostnader för dagens konventionella reningsverk i Sverige***. Detta trots att dagens reningsverk har lägre kvalitetskrav och att de ekonomiska fördelarna med att återanvända avloppsvatten och därmed minskad vattenstress inte togs med i utvärderingen. Projektet drar slutsatsen att en hållbar rening av avloppsvatten för återanvändning i samhället kräver hänsynstagande och aggregering av både miljöpåverkan, kostnader och reningseffektivitet.

Resultaten ger en tydlig indikation på att ***återanvändning av avloppsvatten för olika återanvändningsändamål är möjligt utan att öka den totala miljöpåverkan och utan ökade kostnader samtidigt som kvalitetsmålen kan uppfyllas***. Olika tekniska aspekter som måste beaktas presenteras och konsekvenser diskuteras. Återanvändning av avloppsvatten är, som visas i det här projektet, en både tekniskt, miljömässigt och ekonomiskt hållbar lösning. Detta gäller också om återanvändningen endast gäller återföring till naturen som sådan.

List of Abbrevations

AOC	Assimilable Organic Carbon
AOB	Ammonium Oxidizing Bacteria
ATP	Adenosine Triphosphate, ATP test measures microorganisms' activity
BAF	Biologically Active Filter
BOD	Biochemical oxygen demand
cBOD	Carbonaceous Biochemical Oxygen Demand
CEB	Chemically Enhanced Backwashes
CML	database for LCA, also referring to Institute of Environmental Sciences (CML), University Leiden, Nederlands
DCP	2,4 dichlorophenol
DF	Disk Filter
DFZ	spectral absorption coefficient (Deutsche Farbzahl)
DO	Dissolved oxygen
EBCT	Empty Bed Contact Time
EPS	Extracellular polymeric substances
ESEM	Environmental Scanning Electron Microscopy
ETP	Ecotoxicity potential
FAETP	Freshwater Aquatic Ecotoxicity Potential
GaBi	Life Cycle Assessment Software
ICEAS	Sanitaire Intermittent Cycle Extended Aeration process
LCA	Life Cycle Assessment
LCC	Life Cycle Costs
MAETP	Marine Aquatic Ecotoxicity Potential
MLSS	Mixed Liquor Suspended Solids
MP	Micropollutants
MRZ	Main Reaction Zone of the ICEAS
MTBE	Methyl tertiary butyl ether
NDMA	Nitrosodimethylamine
NDN	Operational mode; reaction phases are under different combination of aerobic/anoxic conditions to enhance nitrification and denitrification process
NTU	Nephelometric Turbidity Units
ORP	Redox potential
PAO	Phosphate Accumulating Organisms
PFOS	Perfluorooctanesulfonic Acid
PPCPs	Pharmaceuticals and Personal Care Products
PRZ	Pre-Reaction Zone of the ICEAS
RGSF	Rapid Gravity Sand Filter
SDI	The Silt Density Index
SS	Suspended material
SRT	Sludge retention time
STP	Sewage treatment plants
SV	Sludge Volume
TET	Terrestric Ecotoxicity potential

TOC	Total organic carbon
TSS	Total Suspended Solids
VFA	Volatile Fatty Acids
VSS	Volatile Suspended Solids
WAS	Mixed Liquor Suspended Solids
YAS	Yeast Androgen Screen
YES	Yeast Estrogen Screen

1 Introduction

Discharge of large quantities of pollutants to surface waters is a contributing factor to lack of water suitable for drinking water. In addition, supply and demand of fresh water is skewed in the world, which increases water stress in many regions, with some serious conflicts as a result. While there is a severe shortage of fresh water, many applications for used water do not require water of such high quality. Wastewater reclamation, i.e. the reuse of treated wastewater has been identified as one of the most significant approaches to meet current and future water demands (ACWUA 2010; National Research Council, 2012; U.S. Environmental Protection Agency, 2012; WHO 2006). While access to fresh water is getting more costly due to environmental pollution, climate change and increased demand on water resources, the use of water for non-potable purposes can be based on reclaimed wastewater. In order to reuse water safely, solids and pathogens need to be removed for most reuse applications. Furthermore, micropollutants and emerging contaminants may need to be removed in other reuse applications. Since this cannot be achieved with traditional secondary treatment alone, additional tertiary and disinfection steps are required. Besides the efficiency of a process to reduce targeted substances, the environmental impact of the wastewater treatment process itself has been discussed by several authors (Falk *et al.,* 2011; Friedrich 2007; Kennedy 2005; Lundie 2004; Memon *et al.* 2007; Muňoz *et al.* 2009; Ortiz *et al.* 2007; Pasqualino 2010; Pillay 2002; Tangsubkul 2005; Zhang 2009).

1.1 Background

A pilot study called ReUse was initiated by IVL Swedish Environmental Research Institute in collaboration with Xylem Inc. at the R&D-facility Hammarby Sjöstadsverk in Stockholm to understand the sustainability aspects of wastewater reuse treatment systems. The project consisted of several components, which are all linked to the central optimization of wastewater treatment by using Life Cycle Assessment LCA. The starting point of this LCA was to first assess the existing water treatment system. Based on this inventory, optimization proposals were developed that aimed at a substantial improvement of the various treatment processes and systems, and to create a comprehensive knowledge base for application of apply these different treatment systems in different parts of the world with different abilities and needs.

In order to implement this project, a global screening of different standards for the reuse of water for different purposes was conducted (ADSSC Design Guidelines, 2008, 2009; Norma Chilena Oficial, 1984; Republica de Colombia, 2007). Furthermore, before the project start, a review of available treatment processes was performed and most relevant technologies identified for inclusion in this project. Selected treatment processes consist of best available and emerging technologies that are commercially available in order to allow direct implementation. Treatment system optimization was performed within the project in order to achieve highest possible resource efficiency.

1.2 General objectives of the project

The general objectives of the ReUse project were to

(i) optimize state-of-the-art treatment processes and systems for non-potable water reuse applications worldwide;
(ii) assess treatment processes in terms of sustainability to achieve the lowest life cycle costs now and in the future;
(iii) achieve the best possible micropollutants reduction; and
(iv) create basic information about treatment systems that can be adapted to local and regional requirements and conditions. As such, the project aimed at developing sustainable solutions to reclaim treated wastewater for urban, agriculture, recreation, industry, and groundwater recharge uses.

From the start of the ReUse project, these objectives were divided into different goals that defined the work for the different project actions:

- Mapping existing standards and guidelines to identify compounds of interest and synthesize global reuse quality targets for Urban, Agricultural, Industrial, Environmental & Recreational, Groundwater Recharge, and Augmentation of Potable Water (Indirect Potable) reuse applications.
- Reviewing and identification of applicable secondary, tertiary and disinfection processes/technologies which when combined allow non-potable reuse quality goals to be achieved.
- Evaluation of the different treatment trains concerning their performance if operated in designed mode. For this the design, installation, start-up and commissioning of a non-potable pilot reuse equipment and treatment trains which will be used in research efforts. This work package will describe the “reference situation”, defining the most relevant and available treatment trains implemented at present. This system will evaluate wastewater and combined wastewater and storm water. The “reference situation” will provide the data of the processes and systems that is necessary for the following work packages.
- To assess the “reference treatment trains” which considers social, environmental, and economic factors which optimizes resource utilization, such as LCA.
- Optimizations of investigated non-potable wastewater reuse solutions.
- Efficient distributions of beneficial information on non-potable reuse solutions.
- Identification and description of possible follow-up projects on non-potable reuse solutions.

1.3 Project organization and management

The project was carried out in collaboration between Xylem Inc. and IVL Swedish Environmental Research Institute. The project was managed by a main project leader from IVL, manager for the R&D facility Hammarby Sjöstadsverk, Christian Baresel, the Head of R&D for Treatment Business Unit, Xylem (USA) Glen Trickle, overall project manager Alexis de Kerchove, Xylem (Sweden), and sustainability project manager Aleksandra Lazic, Xylem (Sweden). A steering group, consisting of representatives from both organizations, met every quarter to follow up progress and define overall adjustments to the project goals and actions. The working group was the operative part of the project making evaluations and decisions on a daily basis. This group consisted of a high number of experts from both organizations. Apart from the involvement of more than 20 experts from each partner, the project involved a number of international external experts to support the project with analytics, technologies, and other competence.

For the environmental impact evaluation, three of the internationally most acknowledged and skilled experts within LifeCycleAssessment of water reuse have helped throughout the project to ensure that used assessment approaches, evaluations and result interpretation was done in accordance to high quality research.

The pilot-wastewater reuse trains at the R&D-facility Hammarby Sjöstadsverket were operated by operators from IVL (Mila Harding, Jesper Karlsson and Elin Ottosson) in close collaboration with treatment experts from IVL and Xylem responsible experts for the various treatment units. The operators were further responsible for sample collection and shipment to external laboratories and the comprehensive onsite analyses program.

The treatment unit modeling, related data analyses and the environmental impact assessment was performed by IVL experts in cooperation with Xylem Inc., which was responsible to provide full-scale data for various treatment units and sizes required for a realistic evaluation in accordance to the objectives of the project.

The cost analyses were performed by Xylem Inc. with support from IVL based on real data from Xylems full-scale plants and sales construction and organization.

2 Project Scope

The overall scope of the Reuse project was defined to bring up sustainable treatment processes and systems for non-potable water reuse and augmentation of potable water. Because the immense options in the water reuse field, however, the ReUse project had to be limited to the most significant and relevant issues and techniques by limiting the scope of covered subject with the following restrictions.

2.1 Selected regions of interest and reference regions

The regions of interest for wastewater reuse were identified as the Middle East, India, Latin America, and Australia. However, for the actual impact and cost assessment and comparison of different reuse systems, the ReUse project selected Spain as the location for the hypothetical STP. This was partly because data necessary for baseline scenarios and environmental impact assessment was easier to gather for Spain than for the general areas of interest. This implies that assumptions about upstream and downstream process such as electricity, sludge quality regulations etc. are based on the Spanish situation. However, the project includes evaluations of region-specific parameters that may affect the outcome of the overall assessment.

2.2 Selected reuse applications

As a starting point for the ReUse project, the most common non-potable wastewater reclamation applications were identified and out of the commonly defined alternatives (see e.g. (Chen *et al.,* 2012; Dalahmeh and Baresel, 2014; National Research Council,

2012) three main reuse alternatives were selected for further evaluation and analysis. The reuse applications selected were:

- **Agriculture:** Irrigation with reclaimed wastewater is one of the most important applications as roughly 2/3 of all water use goes to agriculture irrigation. The alternative combines recycling of both water and nutrients. Note that the application may be divided into restricted and non-restricted irrigation including food crops, non-food crops, fodder, fibre and seed crops. The hygienic quality of the wastewater is the major aspect to consider when wastewater is used in agriculture.
  This reuse alternative may also include reuses commonly named as recreational or urban such as use in parks, irrigation of landscaped areas surrounding homes, commercial buildings, industrial developments, and golf courses etc.

- **Industry:** Wastewater reuse for industrial applications includes water used for cooling, boiler make-up water; industrial process water in pulp & paper, chemical, petrochemical, coal & cement industries, etc. Here, high water purity is needed to avoid rusting, biological fouling and scale formation, which involves advanced treatment of wastewater for the removal of ammonia and phosphates, reduction in alkalinity, hardness, and reduction in suspended and dissolved solids.

- **Groundwater augmentation:** Augmentation of aquifers provides storage of reclaimed water for subsequent retrieval and reuse, helps to minimize or prevent ground subsidence caused by decreased groundwater levels. This reuse alternative may further include infiltration basins, percolation ponds, and augmentation of other natural water bodies for wetland enhancement, wildlife habitat, stream augmentation etc., with or without further use of that water body as fresh water resource for potable use after additional treatment.

The ReUse project did not include urban applications such as for vehicle washing, laundry, window washing, fire protection water, toilet flushing in commercial and industrial buildings etc. mostly because these applications require a separate infrastructure in the form of a pipe systems to avoid a contact between drinking water and reclaimed wastewater.

2.3 Water quality and effluent water qualities targets

The pilot system was operated with the real wastewater inflow to Stockholm's largest wastewater treatment plant in Stockholm, Henriksdal. As the flow to the pilot system was controlled by the main WWTP inflow, an equivalent load to the pilots system was achieved as to the main WWTP including at storm-water events.

For the general modeling and evaluation of the treatment systems in full-scale, the German standard ATV –DVWK-A 131E (2000) was used as a base for the dimensioning of the biological treatment for three selected full-scale plant sizes (20 000 pe, 100 000 pe and 500 000 pe) as presented in Table 2.1. In addition, peaking factors of 3 for the smallest size (20K pe), 2.5 for the middle size (100K pe) and 2 for the largest plant size (500K pe) were used. All internal backwash waters from tertiary treatment as well as supernatant water from sludge dewatering were included as internal loading to the plant. Note, that different Population Equivalent, pe, are used around the world. While the BOD5 value of 60 g/d, pe as used here may be assumed to be internationally accepted even if e.g. Sweden uses BOD7 but with a comparable value. In addition, the water flow used here is widely used. However, in Sweden more diluted wastewater to the WWTPs implies a higher water flow of about 300-400 L/pe and thus lower concentrations.

Table 2.1. Standard load values from ATV –DVWK-A 131E (German standard, 2000).

Parameter	Value	Comment
Water flow	230 l/d, pe	(as this is a worldwide average and used as a design parameter by Xylem)
BOD5	60 g/d, pe	
COD	120 g/d, pe	
SS	70 g/d, pe	
Ntot	11 g/d, pe	
Ptot	1.8 g/d, pe	

Furthermore, the minimum temperature of wastewater of 10 °C was used for sizing the biological secondary treatment step as the full-scale plants are to be positioned in Spain (see Section 2.1).

A working group further mapped the global non-potable reuse quality standards and guidelines to identify compounds of interest and synthesize global reuse quality targets for the above reuse applications. The guidelines and regulations for the reuse applications were clearly defined for Australia, China, India, Spain, USA, Kuwait, Latin America, Saudi Arabia, Western Europe, and UAE (United Arab Emirate). In addition, the country specific guidelines and rules were evaluated where available. As this work has been performed as preparatory task within Xylem Inc. the reader is referred to similar compilations of data such as provided by Dalahmeh and Baresel (2014), National Research Council (2012) and U.S. EPA (2012).

The screening revealed that the guidelines and regulations predominantly focused on the following parameters for almost all reuse applications, which were then used during the selection of relevant treatment technologies (see next section):

- Micro-biological parameters
  - Total and fecal Coliforms
  - Helminth ova eggs (in some territories)
  - Viruses (in some territories)
- Solids, Organic, and Inorganic Parameters:
  - Total suspended solids
  - Total dissolved solids
  - Turbidity
  - BOD5
  - Nutrients (Nitrogen and Phosphorus)
  - Color

As described, each water reuse purpose requires a certain effluent quality. From the review of various regulations and standards in regions of interest, effluent quality targets for the considered reuse alternatives were defined as shown in Table 2.2. The treatment configurations that were supposed to meet the required qualities for the different reuse applications are described in Section 3.1.

Table 2.2. Required main effluent qualities as monthly average for the different reuse applications.

Parameter	Unit	Agriculture AG	Industrial IN	Groundwater Recharge GW
Chlorine Residual		0	0.15	1
Microbiology				
Total Coliforms	/100 ml	2.2	2.2	2.2
Max Total Coliforms	/100 ml	23	23	23
Helminth Eggs Ova	Count/L	<1	<1	0.1
Solids & Turbidity				
Total Suspended Solids	mg/L	5	2	5
Average Turbidity	NTU	2	1	2
Maximum Turbidity	NTU	10		2
Organic & Inorganic				
BOD_5	mg/L	<8	<5	<5
COD	mg/L	<40	<30	<30
Total Nitrogen	mg/L	20	10	<10
Ammonia Nitrogen	mg/L	5	1	1
Nitrate Nitrogen	mg/L	10	5	10
Organic Nitrogen	mg/L	5		5
Total Phosphorus	mg/L	2	1	1
Dissolved Oxygen	mg/L	3		3
pH	-	Neutral	Neutral	Neutral

Due to the variety of reuse applications, it was found that some regions are requiring more stringent effluent quality of certain parameters like organics, while other regions focus more on solids removal. Therefore, the selection of the targeted effluent quality for the ReUse project was done according to Table 2.2, to allow both stringent quality and wide applications of reuse water.

Target values for the pharmaceutical residues and other persistant substances were defined based on mainly two sources; Ökotoxzentrum (Switzerland) and MKULNV (Ministerium für Klimaschutz, Umwelt, Landwirtschaft, Natur- und Verbraucherschutz des Landes Nordrhein-Westfalen, 2010). These recommendations are under discussion and international or national regulations do still not exist. Generally, the criteria differ between acute and chronic effects:

- **acute:** Injury to an organism within 24-96 h cannot be excluded
- **chronic:** continuous exposure of an organism over a long period (> 96 h)

The chronic quality criteria will be relevant for effluents of STP due to the continuous exposure of the specific substances in water bodies. The values from MKULNV are based on the recommendation of Ökotoxzentrum (Switzerland). Additional guide values defined for drinking water are based on an assessment approach MKULNV (2010).

Table 2.3 summarizes the target values for selected substances. Not for all analyzed substances, target values are defined. The selection of the listed substances is based on the Swiss approach for a definition of substances to be reduced before discharged to sensitive water and removal behavior during treatment processes.

Table 2.3. Summary of specific criteria from Ökotoxzentrum and MKULNV.

Substance [µg/L]	Acute quality criteria (Switzerland)	Chronical quality criteria (Switzerland)	Chronical quality criteria (Germany)	Drinking water quality criteria (Germany)
Carbamazepine	2550	0.5	0.5	0.1
Diclofenac	-	0.05	0.1	0.1
Sulfamethoxazole	2.7	0.6	0.15	0.1
Mecoprop-P	187	3.6	-	-
Metoprolol	76	64	7.3	0.1
Benzotriazole	120	30	30	4.5
Ibuprofen	23	0.3	-	-
MTBE	-	-	-	-
Bisphenol A	-	1.5	-	-
17-ß-Estradiol	-	0.4 ng/L	-	-
17α-Ethynylestradiol	-	0.037 ng/L	-	-

For the removal of these substances in this project, targets values indicated with the brownish frame in Table 2.3 were used.

2.4 Selected treatment technologies

The review of different wastewater reclamation technologies during the first project period showed that the most used treatment methods in wastewater reclamation are based on different combination of conventional primary and secondary wastewater treatment with nitrogen and phosphorous removal. The conventional treatment is commonly followed by membrane separation and ends with a disinfection/oxidation step. The working group, which consisted of experts with extensive water treatment experience, reviewed and identified applicable secondary, tertiary and disinfection processes/technologies, which when combined allow non-potable reuse quality goals to be achieved. The technologies applied in the ReUse project have thus been selected based on an initial screening of standard available technologies that can archive a reduction or removal of different substances related to the parameters in water reclamation standards as presented in Section 2.2.

The pilot-system used within the project was defined to include one secondary and several tertiary and disinfection treatment steps. The ICEAS active sludge process, a sequential batch reactor with continuous inflow, was selected as secondary treatment (Section 3.2.1). The tertiary treatment included microfiltration (Section 3.2.2), various ultrafiltration techniques (Section 3.2.6), sand filters (Section 3.2.3), ozonation (Section 3.2.4), biological active filters (Section 3.2.6), granulated active carbon (GAC) filter (Section 3.2.4), UV (Section 3.2.8) and chlorination (section 3.2.9). After reviewing and benchmarking existing treatment processes, the selected treatment technologies represent the most widely used techniques for advanced treatment of wastewater. The different reclamation technology trains that were investigated in the project were thus considered the most relevant combinations of state-of-the-art techniques even so various other technologies for e.g. the secondary wastewater treatment exist.

The general evaluated system further included standard sludge treatment with thickening, aerobic stabilization, and dewatering (including additives).

2.5 Data from pilot system and full-scale plants

The pilot treatment plant at the R&D-facility Hammarby Sjöstadsverk provided treatment-related data for the reuse system evaluation. The information the pilot system provided was, however, limited to treatment efficiencies for various substances under various predefined conditions and for different combinations of single treatment steps to complete treatment trains.

In addition, data from full-scale plants all over the world was used in the project. This data, comprising energy use for pumping, aeration, chemical use, transport etc. for a number of different plant sizes, was completed with treatment unit and plant design and construction data necessary in order to gather as relevant data as possible for the impact evaluation. Data received from the pilot system and full-scale data could further be compared and verified, and in the case of inconsistencies be checked.

2.6 Environmental impact assessment and LCC

The environmental impact assessment and cost analyses incorporated, except for the processes as included in the pilot system, environmental impacts of generating and supplying energy and chemicals to the modelled treatment processes. Figure 2.1 shows a schematic view of the evaluated system with its system boundaries and the included pilot treatment system. The functional unit was set to one (1) m^3 of reclaimed water fulfilling specified effluent requirements for water reuse in regions with the highest water reclamation potential. The downstream boundary considers all the effluents including reclaimed water and sludge treatment (aerobic/anaerobic sludge stabilization step (AD), thickening (TH) and dewatering (DW)). The wastewater treatment part of the studied system was physically in operation at the R&D-facility Hammarby Sjöstadsverk in Stockholm.

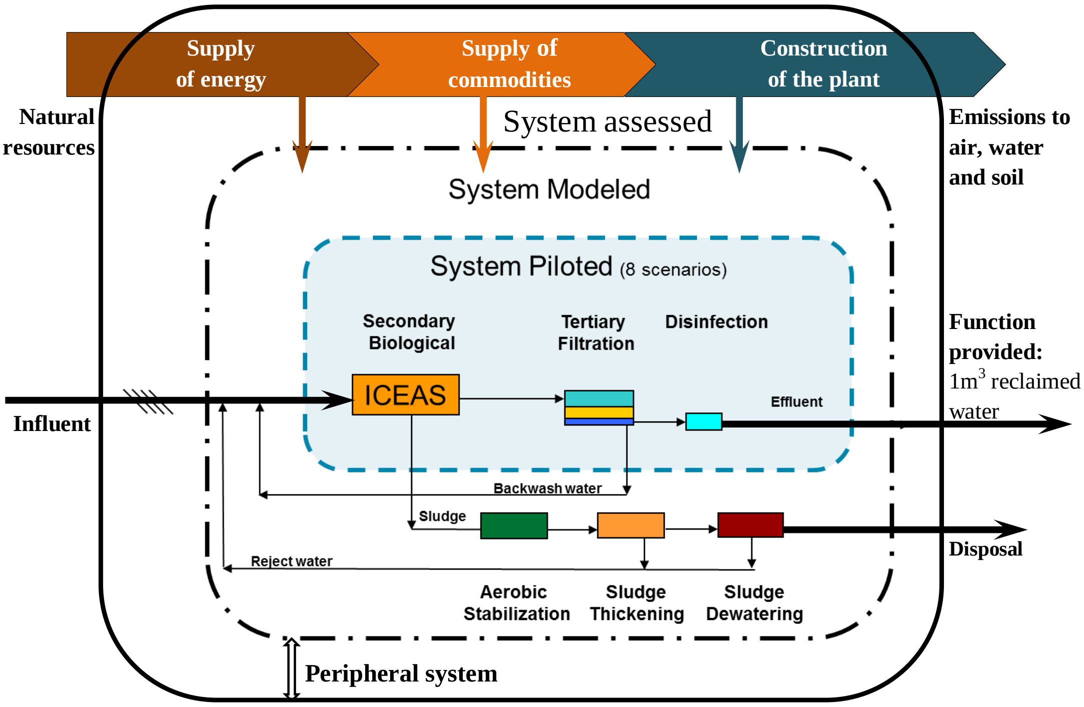

Figure 2.1. Schematic view of the system setup considered in the project including processes of the pilot plant and modeled processes.

2.7 Optimizing of treatment processes and systems

The optimization of treatment processes and systems for non-potable water reuse applications was defined to include several parts. The main optimization work of the ReUse project aimed at benchmarking tests of each single treatment process and complete treatment trains, respectively. Operational parameters such as oxygen control, sludge age, ozone dose, contact times etc., should be varied in most relevant ranges in order to obtain the most optimal process configuration to treat water of a certain characteristic to a predefined effluent quality. This should also be repeated for different treatment trains consisting of various treatment units to find most favorable operation configuration to meet certain effluent quality requirements.

Optimizations in peripheral processes that affect the wastewater reuse sustainability should be determined by utilizing created assessment models without the need for actual testing in pilot-scale but instead direct implementation in reality.

From the sustainability evaluation of the various treatment systems for wastewater reuse applications, modified or complete new treatment train configuration were aimed be suggested in order to be tested using the pilot treatment plant. Those optimized systems should provide treatment processes that achieve the lowest environmental impact, life cycle costs, or the best possible micropollutant reduction to reclaim treated wastewater for the defined reuse applications within agriculture, industry, and groundwater recharge.

2.8 Recommendations, roadmap

The project scope also included the recommendation of further work after finalizing of the ReUse-project related activities. This should include further optimization work that is relevant for the investigated reuse system and reuse applications but that are outside the scope of the project or that require additional completion of the pilot system. The project further should suggest a realistic outline on how to perform such complementary studies.

3 Project methodology

The following sections describe the technologies, methodologies and approaches as used in the ReUse-project including targeted water qualities, treatment technologies and systems, pollutants and parameters investigated including their analyses methods, pilot characteristics, LCA and LCC methodologies.

3.1 Pilot facility

The overall setup of the pilot facility at the R&D-facility Hammarby Sjöstadsverk is shown in Figure 3.1. It consisted of pilot-units for secondary, tertiary and disinfection treatment of municipal wastewater, i.e. the same wastewater as to Stockholm's main WWTP Henriksdal. The effluent from the pilot was returned to the Henriksdal WWTP located next to the R&D-facility. Hence, no effluent treatment requirements applied during the project, which provided the prerequisite for the various tests.

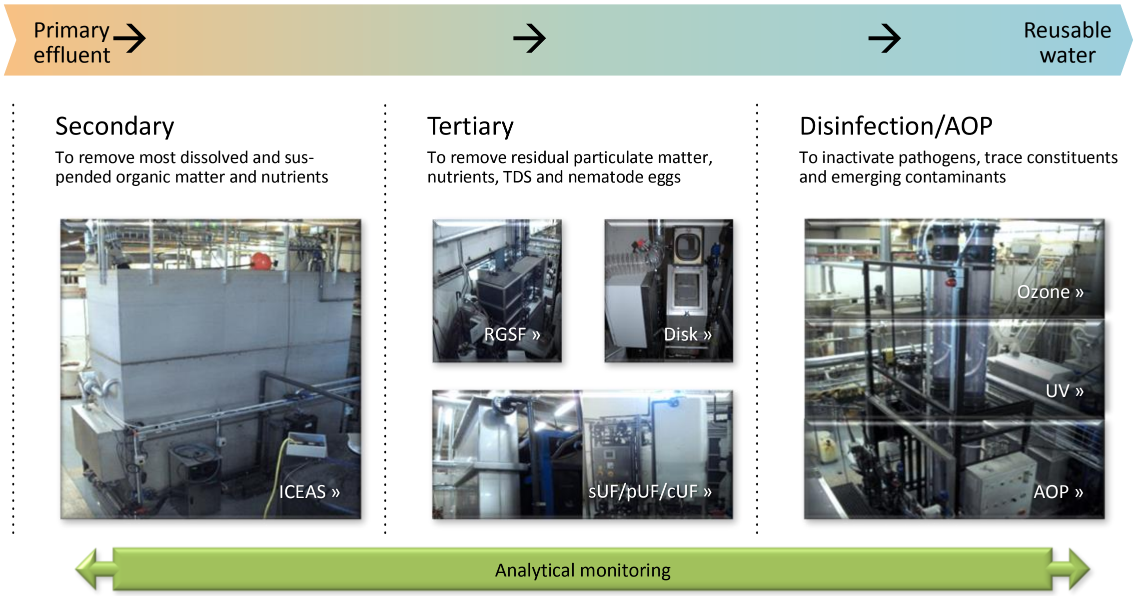

Figure 3.1. Schematic view of the pilot system setup.

3.1.1 **Limitations and simplification**

The ReUse-pilot was designed for the complete range of possible performance and wastewater characteristics. However, during the first period of the project with several extreme flow conditions including storm events and low flow conditions, maximum and minimum flows hade to be defined in the control system to avoid affecting the biology by flushing out or starving periods. Between those limiting set points, the inflow to the pilot was directly controlled by real flow variations into the main STP Henriksdal.

Due to limitations in the size of tertiary pilot processes, the flow to tertiary processes was significantly lower than the effluent quantity produced by the secondary treatment with the ICEAS. This facilitated a better operation and evaluation of the performance of the technologies, but it implied that return flows such as filter backwash waters could not be realized in the pilot facility. Such flows were still analyzed and considered in the modeling of the treatment trains and during the evaluation.

Another limitation the ReUse-project had to face was the relatively low or high concentrations of some substances in the sewage water treated. Spiking equipment was on site at the pilot to overcome this problem in case necessary. As an example of lower concentration in the influent than for the regions of interest, the phosphorous concentration can be named. Various regulations to reduce the use of phosphorous in e.g. detergents have resulted in a steady decrease of the concentrations in sewage during the last decades. On the other hand, higher iron concentrations than expected were observed in the incoming wastewater.

3.1.2 **Sampling and onsite analyses**

For daily follow up of the treatment processes, conductivity, pH, temperature, redox (ORP), dissolved oxygen (DO), Sludge blanket, and turbidity (NTU, Nephelometric Turbidity Units) were measured at several places with portable hand meters.

Collection of grab and composite samples was performed by onsite samplers with options for various interval sampling and local cooling. The placement of the different samplers is indicated in Figure 3.2. Grab samples could also be collected by manual samplers and at different valves. Media samples were taken from special openings in the pilot columns after draining the columns.

Following parameters were determined on filtered samples (0.45 µm):
- dissolved Aluminum (Al) and iron (Fe)
- Ammonium (NH4-N), nitrate (NO3-N) nitrite (NO2-N)
- Chloride
- dissolved chemical oxygen demand (COD)
- Phosphate (PO4-P)
- Potassium (K)
- Color

Following parameters were determined on unfiltered samples:
- Alkalinity
- Total aluminum and iron
- Chlorine
- Total nitrogen (TN)
- Total phosphorous (TP)
- chemical oxygen demand (COD)
- UV Transmittance (UVT)
- Biological oxygen demand (BOD)

Several onsite parameters were determined using spectrophotometric methods and the WTW photoLab 6600:
- COD WTW 2503-01,-03,-06, 252071 (total and dissolved)
- NH4-N WTW 250495, 250329, 252027
- TN WTW 250494, 252018
- PO4-P WTW 252075, 252076
- Ptot WTW 252075, 252076
- NO3-N WTW 252085
- Alkalinity WTW 1.01758.0001
- Fe Total WTW 205361, 250349
- Fe Soluble WTW 205361, 250349
- Al Total WTW 250425
- Al soluble WTW 250425
- Free chlorine WTW 252013

For onsite analyses, duplicates or triplicates have been performed from time to time and when new parameters have been added to the analyses program.

3.1.3 Online analytical monitoring

The online analytical monitoring of water quality and process performance was established by a digital modular multi-parameter system network based on the WTW IQ Sensor Net (http://www.wtw.de/en/products/online/iq-sensor-net.html). The system allowed for monitoring of single or several parameters through single probes and data gathering in a central online database for easy follow up. Parameters were measured before, in and after the tertiary treatment process, in sludge, and around all tertiary treatment steps. Sensors included were for example pH/Temp, DO, Turbidity, TSS, NH4, NO3, COD, UVT, Conductivity, and Redox.

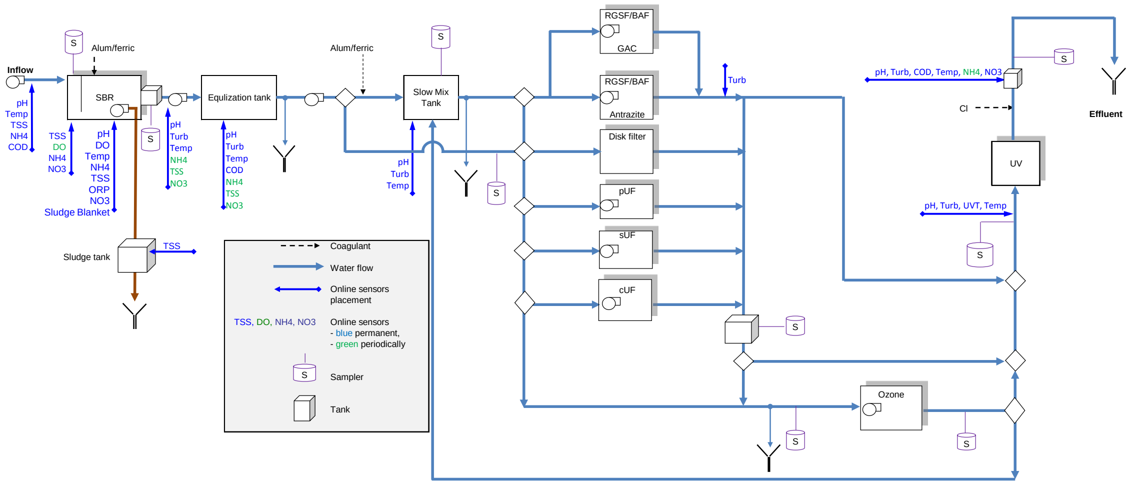

Figure 3.2. Schematic picture of the ReUse-pilot, placement, and type of online sensors.

Online probes haven been frequently serviced according to agreed intervals with the manufacturer and in continuous evaluation of the measurement values and onsite and external analyses. The maintenance to ensure best reliability and performance of the sensors included weekly cleaning of the probes, references sampling and adjustments if necessary as well as matrix adjustments if the acceptable variation from the control sample measurement was more than 10% three times. Sensor trends were continuously evaluated by the project team and remotely by an expert team from WTW to detect any problems in time.

3.2 Treatment technologies

Wastewater treatment as considered in this project includes processes from the raw wastewater influent over the secondary treatment (continuous inflow Advanced SBR called ICEAS) and tertiary filtration and disinfection. Figure 2.1 illustrates which parts of the system that are actually setup and tested in the pilot scale and which are considered in the assessment analyses. Sludge treatment consisting of sludge stabilization and dewatering is not part of the pilot setup but is modeled. However, parameters defining sludge quality and quantity were measured in the ReUse pilot. The preliminary treatment with a grid was excluded from the LCA and LCC analysis. It is, however, included as the common pre-treatment for the pilot-system at the R&D-facility Hammarby Sjöstadsverk.

The process flow combinations, here called treatment trains, studied in the ReUse project pilot are presented after a brief description of the treatment processes selected. The process goals for the scenarios studied are to achieve the agricultural, groundwater recharge or industrial reuse quality levels as presented in Section 2.3. Some of the processes were operated under different conditions, e.g. the ICEAS-SBR in NIT mode (only nitrification), NDN mode (nitrification/denitrification), and NDNP mode (biological P removal) with and without addition of chemicals for chemical P removal. RGSF and other units were operated with and without addition of flocculent.

Not all configurations were supposed to deliver the same effluent quality and grouping of the treatment lines in the three reuse groups (Agriculture, Industrial, Urban) was made for comparison of lines that belong to the same group. Comparison of different groups will be possible only on a high level, for example, analysis on how more stringent effluent qualities influence the overall cost of the treatment as well as environmental impacts.

The considered treatment processes for the pilot tests and the assessment studies included a number of readily available state-of-the-art technologies that are operated in any sewage treatment plants of varying size around the world. For a better understanding of their use, limitations, and how they complete each other for various wastewater reuse applications; the following sections provide a brief description of these techniques.

A common sewage treatment plant consists of pretreatment, primary treatment and secondary treatment processes. In the pretreatment, all materials that can be easily collected from the raw sewage before they damage or clog the pumps and sewage lines are removed. In the ReUse pilot, this is done by a rotary sieve ConSieve 20 from ConPura (www.conpura.com). The normal primary treatment consists of clarifiers. However, because of several advantages primary treatment was not used in the system setup of this project. Secondary treatment is used to degrade the biological content of the sewage. For more stringent water effluent qualities, as required for water reuse, tertiary treatment is applied to improve the effluent quality further. More than one tertiary treatment process may be used. As final treatment step, disinfection reduces the number of microorganisms in the effluent water.

3.2.1 Secondary treatment: Sanitaire ICEAS™, advanced SBR

As the common secondary treatment in the ReUse pilot, a modification of a conventional activated sludge plant is applied. A sequencing batch reactor (SBR) process allows the unit processes of react, settle, and discharge to occur sequentially in one basin. As a result, the “footprint” of a SBR is typically much smaller than that of a conventional activated sludge plant. The Intermittent Cycle Extended Aeration System (ICEAS) process is a modification of a conventional SBR.

The ICEAS process allows continuous inflow of wastewater into the treatment basins during all phases of the cycle without any pretreatment except for a grid. The continuous inflow is an advantage over conventional SBRs in that it optimizes biological treatment by supplying a constant food source for the process and equalizes the flow loadings in multiple-basin systems. A cycle consists of different phases (react, settle, and decant) during which treatment takes place. The cycles operate continuously in each basin to meet the treatment goals of the plant.

An ICEAS basin has two compartments: a pre-react zone and a main-react zone. The pre-react zone acts as a biological selector and receives the continuous influent flow. The two compartments are separated by a baffle wall that spans the tank width and has openings at the basin floor. The baffle wall prevents short-circuiting and allows the two zones to be hydraulically connected as it directs the flow to enter the main-react zone at the bottom of the basin.

The following is a brief process overview of the three phases of a typical treatment cycle:

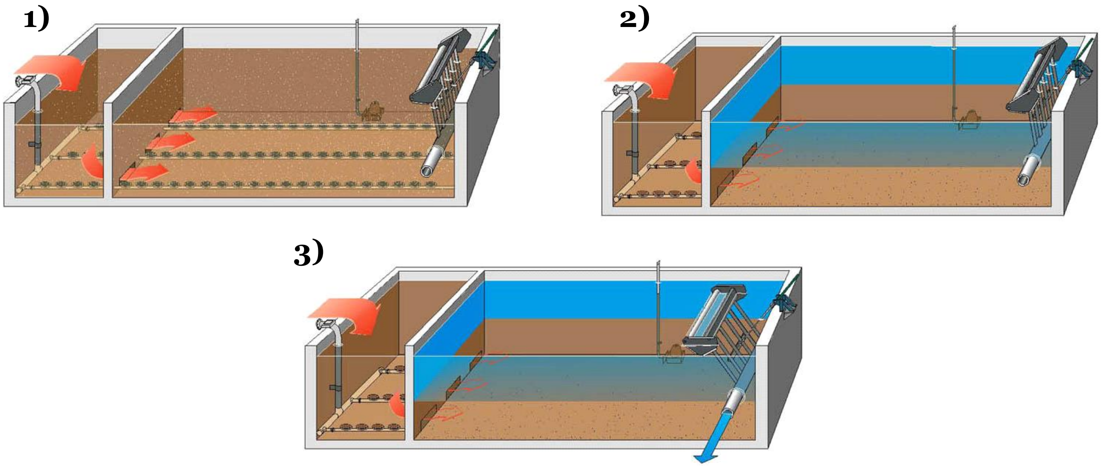

Figure 3.3. ICEAS™, advanced SBR operations phases 1) React, 2) Settle, and 3) Decant (Xylem Inc.).

1) React Phase
During the react phase, raw wastewater flows into the pre-react zone continuously to react with the mixed liquor suspended solids. Depending on the process scheme, the basin contents are aerated, anoxically mixed, allowed to react anaerobically, or a combination thereof. As the basin continues to fill, biological oxidation/reduction reactions take place simultaneously to treat the wastewater.

2) Settle Phase
During the settle phase, basin agitation from the react phase (i.e. aeration or mixing) is stopped to allow the solids to settle to the bottom of the basin. Raw wastewater continues to flow into the pre-react zone as the main-react zone settles. The sludge blanket forms on the bottom of the basin as the mixed liquor suspended solids (MLSS) settle and a clear layer of water will remain on top of the basin.

3) Decant phase
During the decant phase, the decanter rotates downward to draw off the clarified supernatant and discharge it to the effluent line. Raw wastewater continues to flow into the pre-react zone as the main-react zone is decanted. Sludge is typically wasted from the basin during this phase in the cycle.

The ICEAS-SBR has different operation modes depending on the targeted effluent water quality and characteristics of the incoming wastewater flow.

- **The nitrification (NIT) mode** operates to remove BOD, TSS, and ammonia-nitrogen (NH_3-N) through nitrification. In the NIT process, aeration is supplied during the complete react phase to supply oxygen to the biomass for BOD oxidation and nitrification.
- **The nitrification-denitrification (NDN) mode** operates to remove the same as the mode before and nitrite-nitrogen (NO_2-N)/nitrate-nitrogen (NO_3-N) through denitrification. In the NDN process, the react phase consists of alternating periods of aeration and anoxic mixing. The aeration periods supply oxygen to the biomass for BOD oxidation and nitrification. The anoxic mixing periods provide minimal oxygen and mixing of the biomass for denitrification.
- **The nitrification-denitrification-phosphorus (NDNP) mode** operates to remove the same as the mode before and phosphorus through biological luxury uptake. In the NDNP process, the react phase consists of alternating periods of aeration and air-off. The aeration periods supply oxygen to the biomass for BOD oxidation and nitrification. The air-off periods provide anoxic/anaerobic conditions for denitrification and phosphorus release. When the aeration is started after the air-off period, the phosphorus that was released plus extra phosphorus is taken up in the biomass with oxygen present. Note that the process is also equipped with chemical dosing units for chemical phosphorous removal when needed.

A normal ICEAS treatment cycle is completed after 4 hours. Each cycle is divided into the main phases described above with configurable sub-periods of various operations that can be adapted to treatment requirements as was done during the project. Figure 3.4 provides an example of such a cycle for normal inflow conditions and at storm water events, were the cycle has to be reduced to 3 hours to increase the capacity of the treatment process.

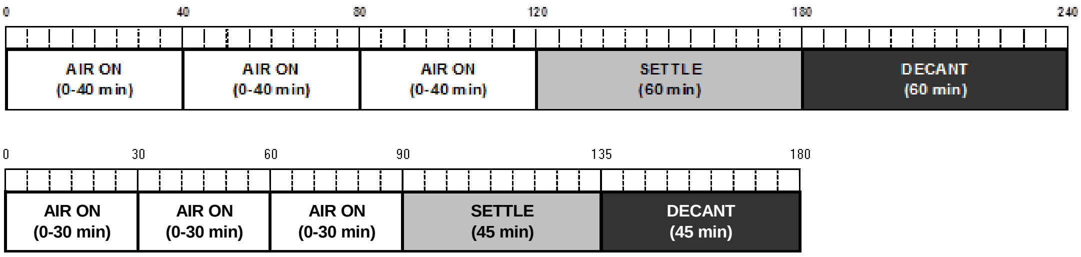

Figure 3.4. Example of ICEAS-SBR cycles in normal and storm mode for the nitrification operation mode.

In NDN and NDNP mode, the total cycle time can be increased to 4.8 hours. Similar as for the NIT-mode, the system switches from normal cycle operation to storm cycle operation (and back), when the monitoring system detects such conditions. Storm water events could also be simulated in the pilots for controlled performance studies.

The control system for the ICEAS process was separated from the other control systems at Hammarby Sjöstadsverk and was connected to a number of process instrumentations for the inflow, effluent, PRZ and MRZ control. The inflow is proportional to the inflow to Stockholm's main STP Henriksdal. Two different sources of wastewater were used during the project; Henriksdal wastewater, which corresponds to raw wastewater from the inner town of Stockholm, and Sickla wastewater, which consists of wastewater from the Nacka suburbs of Stockholm and supernatant from sludge dewatering.

The main ICEAS-SBR control systems applied during the project were Dissolved Oxygen (DO) Control System and Solids Retention Time (SRT) Control System (SIMS).

3.2.2 **Tertiary treatment: Disk Filter**

The disk filter unit from Nordic Water (www.nordicwater.com) is a mechanical disk filter with filter openings ranging from 10 to 100 microns to remove suspended solids for subsequent tertiary treatment processes. These provide high capacity on a very small area with a backwash process parallel to the filtration process, which reduces amount of wash water.

The operation of the disk filter can be described as follows. The discs are submerged to approximately 60% and when the water level inside the filter rotor increases to a preset point due to clogging of the filter by solids, the filter starts rotating and the backwash of the filter media starts. The high-pressure backwash removes the accumulated suspended solids into the reject flume inside the filter. The suspended solids are then discharged via the reject pipe and normally returned to the inflow of the treatment plants. In the ReUse pilot system, however, the backwash water was not returned because of scale issues.

The average feed flow was 600 L/h. The filter was operated continuously for two specific periods using meshes with different pore size:

- Mesh with pores of 10 µm (July 2013)
- Mesh with pores of 18 µm (from Oct 20 to May 09, 2012)

The goal of the DF-operation was to establish the general performance of the DF under conventional operation as a tertiary treatment downstream of the ICEAS. Operation was adjusted to meet the design influent and targeted effluent quality for the ground water recharge (GWR) and agriculture application given in Table 2.2.

3.2.3 Tertiary treatment and disinfection: Ozone treatment

Ozone treatment is commonly used for disinfection through oxidation of contaminants and it provides removal of colored substances, odors, bacteria and most viruses. It further provides removal of endocrine and pharmaceutical substances called micropollutants (MP). In difference to chlorination, there are no harmful chlorinated by-products. Three main issues were tested regarding the ozone process.

i. Improve the overall water quality by using ozone to treat secondary effluent
    a. Significantly remove Odor and Color
    b. Significantly improve UV Transmission
    c. Slightly reduce COD
    d. Slightly increase BOD
    e. Achieve certain disinfection effect
    f. Ensure the bromate level is under 10 µg/L in the final treated water
ii. Study the micropollutants removal efficiency by ozone
iii. Study the synergetic benefits of ozone process on downstream filter process

The secondary treated water (by ICEAS or MBR) passed or passed not through a filter (a Disc Filter 10µm or a pressurized Ultrafiltration pUF) and flowed into the WEDECO Ozone Pilot. The heart of the ozone pilot was a WEDECO MODULAR HC8 ozone generator (nominal ozone production 8 g/h). The set-up of ozone pilot consisted of two columns operated in series. The first column is operated in downstream mode; the second column is operated in upstream mode. The ozone gas is continuously bubbled into the water through a ceramic diffusor built in the bottom of each column. Then the ozonated effluent is fed to the downstream media filter pilot (see following sections). The media filter pilot consisted of two filter columns. One column is filled with anthracite as media; the other one is filled with Granular Activated Carbon (GAC) as media.

Table 3.1. The location of ozone process in different treatment scenarios.

Treatment configuration				
DF-O3-BAF	SBR	Disc Filter	Ozone	BAFs
O3-BAF	SBR	-	Ozone	BAFs
pUF-O3-BAF	SBR	pUF	Ozone	BAFs
MBR-O3-BAF	MBR		Ozone	BAFs

3.2.4 Tertiary treatment: Granulated Active Carbon Filter

One of the investigated filters was operated as Granulated Active Carbon (GAC) Filter. A dual granular media of fine sand and GAC effectively removes particles and various substances through sorption processes. The filtration process was driven by a hydraulic head on the top of the filter. If the predefined maximum head-loss was exceeded, back-washing of the filter with air and effluent water was initiated. The backwash sequence could also be time-controlled for maintenance of a high capacity.

Empty bed contact times (EBCT) were varied.

3.2.5 **Tertiary treatment: Sand filter (RGSF)**

The Leopold FilterWorx Performance Filter represents Rapid Gravity Sand Filter (RGSF) treatment technology. The dual granular media of fine sand (bottom layer of 0.305 m (1 foot) ES 0.5 mm UC 1.4) and anthracite (0.61 m (2 feet) ES 1.0 mm UC 1.4) effectively removes fine suspended solids through chemical coagulation and interception during filtration. The filtration process is driven by a hydraulic head on the top of the filter. If the predefined maximum head-loss is exceeded, backwashing of the filter with air and effluent water is initiated. The backwash sequence can also be time-controlled for maintenance of a high capacity. The pilot-unit was equipped with two filters (8 inches (20.3 cm) diameter and 12 feet (3.66 m) tall with a cross section area of 0.35 ft^2 (0.0325 m^2)), with independent control systems.

The average hydraulic loading of the filter was set at 3 gpm/ft^2 (7.3 m/h) and during a twelve hours storm event, the peak flow was 9 gpm/ft^2 (22 m/h). Empty bed contact times (EBCT) are variable and a turbidity meter is located immediately downstream of the filtration units for process control.

The objective of the study relating to media filtration was to demonstrate that the tertiary filtration system could achieve the California Title 22 reuse standards after the upstream sequential batch reactor (SBR). The California Title 22 requests for water passed through natural undisturbed soils or a bed of filter media to pursue the following:

1) At a rate that does not exceed 5 gpm/ft^2 (12.2 m/h) in mono, dual or mixed media gravity, upflow or pressure filtration systems, or does not exceed 2 gpm/ft^2 (4.9 m/h) in traveling bridge automatic backwash filters;
2) So that the turbidity of the filtered wastewater does not exceed any of the following:
    a. An average of 2 NTU within a 24-hour period;
    b. 5 NTU more than 5 percent of the time within a 24-hour period; and
    c. 10 NTU at any time.

The validation of the reuse quality of the treated effluent is done under constant average dry weather flow (ADWF) and under the diurnal flow conditions with a peaking factor during storm event of 3; and peak dry weather flow (PDWF) normal operation with a peaking factor of 2.2. Specifically, the following items were examined:

- Filter runtimes at different loading rates
- Effluent qualities (TSS, turbidity, phosphorus, SDI15 etc.) during normal dry weather flows and wet weather flow
- The need for coagulation for complying California Title 22 – 2 NTU

3.2.6 Tertiary treatment: Ultrafiltration

The ReUse pilot included two different ultrafiltration options:

- pressurized ultrafiltration (pUF)
- submerged ultrafiltration (sUF)

This report covers the pressurized UF with polymeric membranes and the sUF. The performance of ultrafiltration was focusing on two aspects: Capacity, and permeate (effluent) quality. I addition, some initial tests have been carried out using ceramic UF. Results from these tests are however not presented here.

There were three campaigns running the pUF and one with the sUF. Ultrafiltration produces high quality effluent suitable for direct reuse or discharge because of its high removal rates of particulates and pathogens. In addition, very low turbidity in the ultrafiltration effluent improves the performance of downstream process such as disinfection.

All used techniques have in common that a pressure difference across the membrane drives the water through membrane. Colloids and particulates are stopped by the membrane and if the pressure difference exceeds a certain value, the flow through the membrane is reversed for a backwash (even scouring with air) to remove particles from the membrane. The backwash water was not returned to the process inflow as normally done because the chosen system setup.

The polymer membranes used in pUF and sUF had nominal pore size between 0.02 and 0.35 µm. The pUF was fitted with two membrane elements from X-Flow, type AQF each with 6.2 m^2 of membrane area. The feed was treated with a 100µm screen that, however, was bypassed during most of the first trials. Subsequent trials used the disk filter fitted with the same screen size. Following the screen, coagulant was dosed during some tests using an inline mixer before a slow mixing tank (SMT) in the first test campaign. However, as this proved unreliable, the coagulant was dosed directly into SMT in remaining campaigns. The level in the slow mix tank was set at the lowest option to provide minimum retention time, i.e. 10 - 60 minutes depending on feed flow. From the slow mix tank there was gravity feed to the suction of the pUF feed pump.

Both membrane elements were in use during the first campaign, and one was used in the later campaigns. Permeate was driven into the permeate tank from where it flowed via the downstream instruments to the hypochlorite dosing and then to drain. The permeate tank also provided permeate to the backwash pump for backwash and Chemically Enhanced Backwashes (CEB). The waste from the backwash was discharged to the drain.

The general operating sequence was as advised by the membrane supplier: Filtration was carried out in a dead end mode. At the end of a defined time, the membrane was backwashed with permeate, while a feed flow was maintained. Simultaneously, the waste valve was opened allowing the membrane to be flushed by retentate. This cycle was repeated for a defined number of times, after which two chemically enhanced backwashes were applied. The first was with alkaline sodium hypochlorite (Sodium

Hydroxide: 525 mg/L; Sodium hypochlorite: 200 mg/L) to remove organics, and the second was with hydrochloric acid (450 mg/L) to remove scale.

The sUF was fitted with a single GE Zenon ZW500 membrane module in a custom designed frame. The GE advice is to use a minimum of three modules, but this was not a viable option to match the capacity of the upstream process. The general flow scheme was similar to the pUF, but no coagulant or screening was used.

The membrane was operated in dead end mode, with the membrane tank level being maintained. When the desired volume had been treated, the membranes were backwashed, and the tank was drained. The sequence was then repeated. A backwash involved both air scour and back-pulsing the membranes. There were also intermediate back-pulses carried out at fixed intervals during the filtration cycle.

3.2.7 **Tertiary treatment: Biological Active Filter (BAF)**

The Leopold Filters (see 3.2.4 and 3.2.5) can also be operated as Biological Active Filtration System (BAF) to remove organic substances next to suspended solids. One filter was loaded with anthracite and sand while the other one was loaded with granular activated carbon (GAC) and sand. These two filters were operated in a parallel manner to compare process performance. Empty bed contact times (EBCT) are variable and extended compared to operation as RGSF. The process is also here driven by a hydraulic head on the top of the filter. If the predefined maximum head-loss is exceeded, backwashing of the filter with air and effluent water is initiated. The backwash procedure was identical with the one for the RGSF and backwash water was not returned to the pilot inflow. A turbidity meter was located immediately downstream of the filtration units for process control.

3.2.8 **Disinfection: UV**

Ultraviolet light eliminates most of waterborne bacterial pathogens in seconds without the need for chemicals additives or harmful side effects. UV light is energy rich light with a wavelength of 200 – 400 nm that destroys harmful microorganisms. At the wavelength range of 254 nm UV light directly impacts the DNA of microorganisms (see Figure 3.5). By changing the DNA, the cell division of the microorganism is interrupted – it can no longer reproduce itself and loses its pathogenic effect.

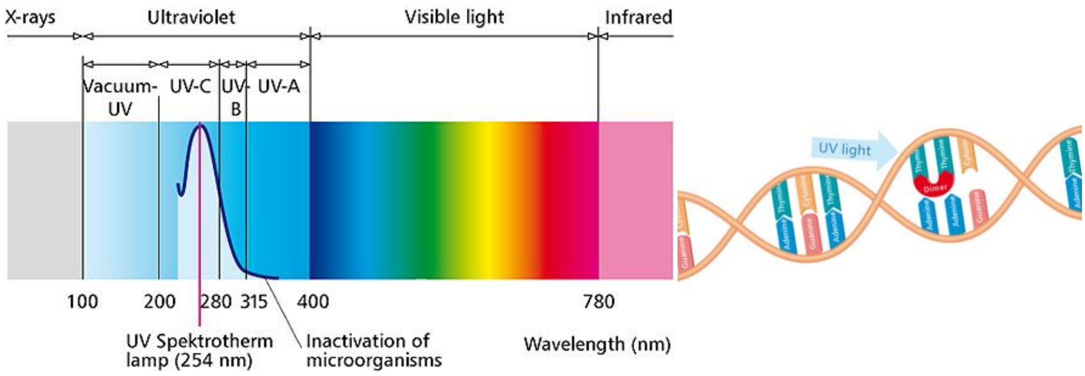

Figure 3.5. Schematic illustration of Ultraviolet light spectrum and effect on microorganism.

In the ReUse-project, a Collimated Beam Device (CBD, WEDECO) was used for UV-disinfection. The purpose of the tests was to get the proper UV-dose for the inactivation of bacteria in the wastewater effluent. For this, total coliform and faecal coliform have been selected as indicator bacteria. For each CBD-trial, 18 water samples were radiated. Then each sample was examined for faecal coliforms and total coliform in triplicate to get reliable results!

3.2.9 **Disinfection: Chlorination**

Hypochlorite was used during the pilot research to study the chlorine demand to reach a stabilization of the final effluent for non-potable reuse application. For all applied regions, the stabilization is defined by reaching a free chlorine residual of 1 mg/L after a contact time of minimum 30 min. The chlorine demand is defined as the difference between the amount of chlorine added to a water system and the amount of free available chlorine at the end of a specified time. The demand is the amount of chlorine consumed by oxidation or substitution reactions with inorganic and organic materials, such as H_2S, Fe^{2+}, Mn^{2+}, NH_3, phenols, amino acids, proteins, and carbohydrates.

The Break point chlorination is defined as the point where sufficient chlorine was added to a system to maintain a free-available chlorine residual. Factors that affect breakpoint chlorination are initial ammonia nitrogen concentration, pH, temperature, and demand exerted by other inorganic and organic species. The weight ratio of chlorine applied to initial ammonia nitrogen must be 7.5:1 or greater for the breakpoint to be reached. If the weight ratio is less than 7.5:1, there is insufficient chlorine present to oxidize the chlorinated nitrogen compounds initially formed.

The evaluation of water stabilization by chlorination was initially intended by online injection of hypochlorite in the pilot plant final effluent. Results have shown to be very inconsistent, independently of the upstream treatment. Despite the control of the chlorine dose as a function of the water flow rate, the frequent variation in the flow rate made the contact time of chlorination very variable. The residual free chlorine could not be directly correlated to the injected dose. Chlorination was then tested on batch system in the laboratory. Water sample of the final effluent of the targeted lines were collected. UV disinfection was applied by column bean device (CBD) for very exact exposure of the sample. Contact time was maintained to 30 min before testing for residual free chlorine.

The goal of the batch testing was to define the exact chlorine demand required for effluent stabilization after disinfection. The effect of the solid and carbon content in the final effluent on the chlorine demand was considered small and the batch test were all performed on either the disk filter or media filter (RGSF) effluent. The disinfection technologies testes were chlorination, ozonation and UV exposure. The secondary disinfection of the final effluent after stabilization was not investigated.

3.3 Treatment trains – Combination of treatment processes

The different treatment technologies were combined to create different treatment trains that were identified as the most promising combinations to archive various effluent qualities. Below, the trains for the three reuse-applications, Agriculture, Industry, and groundwater recharge are described. The general setup including all flow configurations is shown in Figure 3.6.

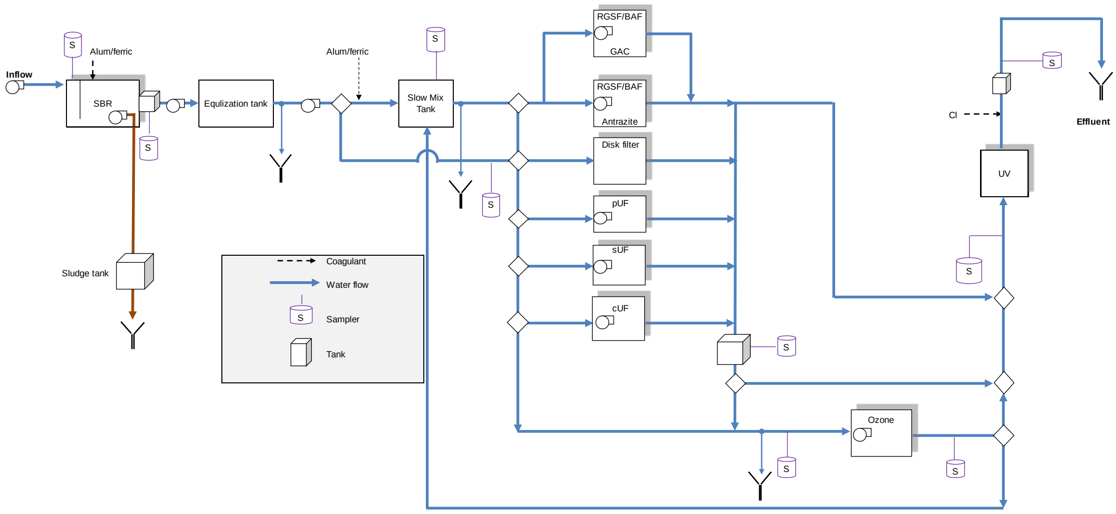

Figure 3.6. ReUse-pilot flow configurations including all treatment options.

3.3.1 Treatment trains for agriculture reuse applications

Here treatment trains that can reach agricultural reuse standards, that do not require removal of nitrogen, are considered. The main target of these treatment trains is nutrient reclamation. The investigated systems are as follows:

- AG1: SBR (AG-NIT) > RGSF > UV
- AG2: SBR (AG-NIT) > DF > UV

The first flow configuration was evaluated for ICEAS in partial NIT mode. The special AG-NIT mode aimed to maintain a high ammonium in the effluent (around 5 mg/L) needed for the agriculture reuse of the water. The configuration of the second AG-train was the same as the first one except that the RGSF was replaced with a disc filter (DF).

3.3.2 Treatment trains for industrial reuse applications

The main target of treatment trains for industrial reuse applications was the optimal nutrient and solid reduction. The investigated systems were as follows:

- I1: SBR (NDN) > pUF > UV > Cl
- I2: SBR (NDN) > sUF > O3 > Cl
- I3: SBR (NDN) > sUF > UV > Cl

The main difference between the three trains was the use of either submerged or pressurized ultrafiltration and UV and ozone treatment, respectively.

3.3.3 Treatment trains for groundwater augmentation applications

The main target of treatment trains for groundwater augmentation applications is the optimal nutrient, solid, and merged contaminant reduction. Therefore, the ICEAS-system was operated in NDN mode in all configurations. The investigated systems are as follows:

- GW1: SBR (NDN) > RGSF > UV > Cl
- GW2: SBR (NDN) > DF > O3 > BAF > UV > Cl
- GW3: SBR (NDN) > O3 > BAF > UV > Cl

The main difference between the three trains is the alternate use of either RGSF, disk filter or none of those, in combination with ozone and biological active filters with final UV and chlorination.

3.4 Contaminants investigated and analysis methods

Parameters and substances that were of interest in the ReUse project include a number of common targeted parameters as well as emerging substances and substances of higher interest in other parts of the world. The following sections provide a description of the different parameters, why they are of interest for wastewater reuse and their analyses method.

3.4.1 Common targeted parameters

Wastewater is characterized in terms of its physical, chemical, and biological constituents. The most commonly used parameters were analyzed within the project.

3.4.1.1 Nutrients

Nutrients as nitrogen and phosphorus can damage environment as they lead to oxygen consumption in aquatic systems with “dead bottoms” in marine systems as one of the known consequences. Nutrients are however also beneficial as they are required for plant growth. Thus, irrigation with nutrient-rich water is one of the applications of wastewater reclamation schemes, while nutrient for other reuse applications such as industrial use would create problems.

Analyses on nutrients in water and sludge were performed according to common standards that are not presented here as these analyses are widely spread and can be performed with good confidence. Different concentrations ranges were applied depending on the effluent target. Onsite, fast analyses were possible but at lower detection limits. Various certified laboratories were used during the course of the project including Erken laboratory in Norrtälje and Alcontrol AB.

3.4.1.2 Organic matter

Biochemical oxygen demand (BOD) is the amount of oxygen used by organisms while consuming organic matter in a wastewater sample, usually measured as BOD_5, the oxygen consumption after 5 days. In Europe, the method of measurement for the BOD_5 requires that the sample is homogenized, unfiltered, and undecanted; and that a nitrification inhibitor is added. Overseas the standard BOD_5 analyses exclude the inhibitor and therefore both analyses were performed in the ReUse project.

Sample preparation and filling of the measuring bottles were done following DIN 38409 part 52. The analyses were then mostly preformed onsite with help of the WTW

Manometric BOD Measuring Devices OxiTop using the method WTW 208211. Samples were also frequently sent to Alcontrol laboratory for comparison and quality assurance.

Total organic carbon (TOC) was determined by first eliminating the inorganic carbon. This is achieved by bubbling (sparging) the preserved sample (pH of 1-2) with synthetic air which causes the inorganic carbon to gas-off as carbon dioxide. Then, the sample is placed in the combustion tube and heated. Any remaining carbon compounds form CO_2 and the carrier gas moves the sample (now as gas) to a dehumidifier where the gas is cooled and dehumidified. The sample is purified with respect to chlorine and other halogens and finally inserted into an analysis cell in which the CO_2 content is determined by NDIR (Non-Dispersive Infra-Red sensor). The TOC-analyses were performed at IVL laboratory in Gothenburg.

3.4.1.3 Solids, particles and sludge characteristics

To determine *Total Suspended Solids (TSS)*, a well-mixed sample was filtered through a weighted standard glass-fiber filter, 1.6 µm. The retained residue was dried at 105°C and reweighted after cooling. The increase in weight of the filter represents the TSS in the sample. The sample volume was adjusted according to the expected concentrations, i.e. 100 ml for influent wastewater, 100-250 ml for secondary effluent water, and 25 ml for activated sludge. Suspended material in the backwash water was determined in some instances by the standard method SS 02 81 12-3.

Volatile Suspended Solids (VSS), which are classed as organic material, are defined as the residue from TSS when ignited in 550^{0}C for a minimum of 1 hour and weighted after cooling in a desiccator.

Sludge Volume (SV) was frequently used to measure the settling quality of the sludge and for calculating the *Sludge Volume Index (SVI)*. For the determination of SVI, it is necessary to determine the Total Suspended Solids (TSS) and SV_{30} simultaneously. A 1000 ml cylinder Ø 80mm or 2000 ml cylinder Ø 127 mm was filled with well mixed, carefully homogenized (not to destroy the sludge flocks) and freshly collected sample of activated sludge during the aeration or mixing period. The samples settled for a predetermined time, usually 30, 60 or 120 min. If the volume was more than 250 ml/L after 30 min, a new sample was collected and diluted with effluent 1:1, 1:2 or more to get a value that is 250 ml or less (Diluted SVI).

The Silt Density Index (SDI) standard test method can be used to indicate the quantity of particulate matter in water and is applicable to relatively low turbidity waters such as tertiary treated wastewaters. Water is passed through a 0.45 µm membrane filter at a constant applied gauge pressure of 207 kPa (30 psig), and the rate of plugging of the filter is measured. The SDI is calculated from the rate of plugging.

3.4.2 Emerging substances and parameters of interest

3.4.2.1 Pathogens

Water contamination in terms of the number of the colonies of coliform-bacteria Escherichia coli (E. coli) per 100 milliliter of water indicates the extent of fecal matter present in the water after various treatment steps. The project used fecal and total coliforms as an indicator for pathogens. Colilert® (Colilert Most Probable Number (MPN)

Method (Colilert-18)) is a commercially available enzyme-substrate liquid-broth medium (IDEXX Laboratories, Inc., Westbrook, Maine) that allows the simultaneous detection of total coliforms and Escherichia coli (E. coli). It can also be used for detection of fecal coliforms. The MPN method is facilitated by use of a specially designed disposable incubation tray called the Quanti-Tray®. Analyses were performed at the National Veterinary Institute, SVA.

Total coliforms 35°
The analyses method enumerates between 1 and 2,400 MPN/100 mL, which implies dilution of the sample if counts higher than this are observed. The sample is combined with the Colilert reagent and mixed. The reagent/sample mixture is poured into the incubation tray, which is then run through the Quanti-Tray sealer. After incubation at 35±0.5°C, Colilert-18 results are definitive at 18–22 hours. In addition, positives observed before 18 hours, and negatives observed after 22 hours are also valid. The MPN table is then used to obtain results. For diluted samples, the result has to be multiplied with the corresponding factor.

Fecal coliforms
Here the same procedure as for total coliforms 35° is used with exception for the incubation procedure. The incubation tray is incubated in a water bath with temperature of 44.5 ± 0.2°C for 18-22 hours before count of positive wells.

3.4.2.2 Helminth ova
Helminth eggs are the infective agents for the types of worm diseases known globally as helminthiases. Eggs are microscopic and are contained in variable amounts in excreta and thus also in wastewater and sludge. Because of the issues associated with introducing viable helminth ova into the works, it has been decided to use surrogate polymer beads for the tests. Beads were 20µm diameter spheres, with a density close to that of helminth ova. The diameter has been chosen to be less than the smallest minor axis dimension for ova. The beads were yellow-green fluorescing to make analysis easier. As much as up to 1000 liters of sample water was collected at each test and filtered through a 10µm filter. The filters were examined using a fluorescence microscope with the aid of a grid, printed on overhead transparency. The filter and grid is placed between two glass slides by the commissioner before being sent to SP Sweden for counting.

3.4.2.3 Pharmaceutical residues and Endocrine substances
Pharmaceuticals and Personal Care Products (PPCPs) refer commonly to any product used for personal health or cosmetic reasons. This comprises a variety of chemical substances, which eventually end up in sewage treatment plants (STP) where most of them are not broken down completely (Loos *et al.,* 2013; SEPA 2008). As they pose a risk of irreversibly disturbing ecosystems in recipients (Gerrity and Snyder, 2011; Hollender *et al.,* 2009; Wahlberg *et al.*, 2010; Wert *et al.,* 2007), current STPs have to complement their treatment process with additional systems for reducing emissions. In July 2013, it was decided by the European Parliament to, for the first time, include three pharmaceuticals in a "watch list" of emerging pollutants that could one day be placed on the priority list (Directives 2000/60/EC and 2008/105/EC as regards priority substances in the field of water policy, European Parliament, 2013).

In the past, various methods to remove PPCP residues have been evaluated mostly within the framework of larger projects, such as the EU projects POSEIDON and REM-PHARMAWATER and the on-going Swedish MistraPharma project. The latter has demonstrated that some of the substances can be broken down more effectively in the existing activated sludge treatment by adding carriers (Falas *et al.* 2012). The study indicates a faster degradation (per amount of biomass) for diclofenac, ketoprofen, gemfibrozil, clofibric acid and mefenamic using carriers. For ibuprofen and naproxen, no significant effect was observed. Falas *et al.* (2013) further found similar relationships for a few more compounds. However, a breakdown of the six studied substances, including Carbamazepine, was also observed in processes without carriers. To get an acceptable removal of most pharmaceutical compounds it seems more realistic to complement with a separation/degradation step, i.e. activated carbon or advanced oxidation with ozone etc.

Table 3.2. Analysed micropollutants group A and B.

Group	Pharmaceuticals	Mode of action
A	Atenolol	Antihypertensives
A	Carbamazepine	Sedatives
A	Ciprofloxacin	Antibiotics
A	Citalopram	Antidepressants
A	Diclofenac	Anti-inflammatories
A	Estradiol	Hormones
A	Etinylestradiol	Hormones
A	Hydroklortiazid	Antihypertensives
A	Ibuprofen	Anti-inflammatories
A	Irbesartan	Antihypertensives
A	Metoprolol	Antihypertensives
A	Oxazepam	Sedatives

Group	Pharmaceuticals	Mode of action
A	Propranolol	Antihypertensives
A	Risperidone	Antipsychotics
A	Sertralin	Antidepressants
A	Sulfametoxazole	Antibiotics
Biocides		
A	Atrazin	Herbicide
A	Mecoprop	Herbicide
Other micropollutants		
A	Benzotriazole	Corrosion inhibitor/ Drug precursor
A	Bisphenol A	Plastic monomers
B	PFOS	Surfactant
A	Sucralose	Sweetener

The extraction of micropollutants from wastewater using solid phase extraction (SPE) was modified based on a method previously described by Gros *et al.* (2006) for multi-residue analysis of pharmaceuticals. Aliquots of 50 to 500 ml thawed composite samples were spiked with the surrogate standards Carbamazepine-13C15N, Ibuprofen-d3, Ciprofloxacin-13C15N and M-PFOS (carbon 13 labelled). The final volume of a sample was defined based on its origin within the wastewater treatment process. Prior to extraction, the SPE cartridges (Oasis HLB, 6cc, Waters) were conditioned with methanol followed by MQ water. Thereafter, the samples were applied to the columns at a flow rate of two drops per second. The analytes were eluted from the SPE cartridges using 5 ml methanol followed by 5 ml acetone. The eluates were evaporated to dryness under nitrogen at 40° C. The samples were reconstituted in 1.0 ml in methanol:water (1:1) containing 0.1wt% ethylenediaminetetraacetate (EDTA-Na2) and centrifuged at 10 000 rpm for 10 minutes. The supernatants were transferred to vials for final determination on a high performance liquid chromatography- triple quadrupole mass spectrometer (HPLC-MS/MS).

The final determination of the amount of micropollutants A and B in the samples was performed on a binary liquid chromatography (UFLC) system with autoinjection (Shimadzu, Japan). The chromatographic separation of micropollutants A (pharmaceutical and pesticides etc.) was carried out using gradient elution on a C18 reversed phase column (dimensions 50 x 3 mm, 2.5 µm particle size, X Bridge, Waters, United Kingdom) at a temperature of 35° C and a flow rate of 0.3 ml / minute. The mobile phase consisted of 10 mM acetic acid in water (A) and methanol (B). The chromatographic retention of micropollutants B (Perfluorooctanesulfonic acid, PFOS) was carried out using gradient elution on a C8 reversed phase column (dimensions 50 x 3 mm, 5-µm particle size, Thermo Scientific, United states) at a temperature of 35° C and a flow rate of 0.4 ml / minute. Besides the column for the chromatographic retention of PFOS, an additional C8 column was attached prior to the injector in order to displace background levels of PFOS derived from the system. The mobile phase consisted of 2 mM ammonium acetate in water (A) and 2 mM ammonium acetate in methanol (B).

UFLC system was coupled to an API 4000 triple quadrupole mass spectrometry (MS/MS) (Applied Biosystems, Canada) with an electrospray ionization interface (ESI) performed in both positive and negative mode.

Perfluorooctane sulfonate (PFOS)
Since the 1960s, per- and polyfluoroalkyl substances (PFASs) have been used in many products, e.g. AFFFs (**A**qeuous **F**ire **F**ighting **F**oam) due to their surface active characteristics to enable the formation of an aqueous film and to resist heat, oil, and water. One of the main compounds, perfluorooctane sulfonate (PFOS) is an extremely persistent and toxic compound that biomagnifies in biota due to its protein binding properties. PFOS has received increasing public attention due to its possible adverse effects on humans and wildlife. Consequently, PFOS has been added to the persistent organic pollutants (POPs) list of the Stockholm Convention in May 2009, resulting in global restrictions on its use and production.

3.4.2.4 Estrogenic and androgenic activity (YES & YAS)

Several studies have shown that effluents from sewage treatment plants contain endocrine disrupting chemicals. Feminization in male fish including skewed sex ratios in exposed fish populations and oocytes in the gonads of males downstream from municipal sewage discharge has been linked to the occurrence of estrogenic compounds in the effluents. Natural estrogens such as those regulating the female reproductive cycle are excreted at a constant rate by both women and men in the population and occur in sewage. Estrogens used as contraceptives and pharmaceuticals are also excreted and have been found in municipal wastewater. In addition, synthetic compounds such as nonylphenol and its derivatives, and bisphenol A mimic estrogens, and have also been detected in wastewater. Skewed sex ratios in the offspring of fish and defects in primary and secondary sexual characteristics have also been reproduced in the laboratory after exposure to androgens. Anomalies have been related to interference with the function of androgenic hormones including sex ratios biased in favor of males and the development of male sexual characteristics in female fish.

The pH of 500 mL samples was adjusted to pH 2.9 – 3.1 with HCl. Extraction of samples was carried out using solid phase extraction (SPE) with prepacked columns (ENV+, Sorbent AB, Västra Frölunda) containing 0.2 g of polystyrene divinylbenzene copolymers according to a published procedure (Körner *et al.,* 1999; Svenson *et al.,* 2003). Before use, each SPE column was successively rinsed with two portions of 5 mL acetone and two portions of 5 mL 1 mM HCl. Samples were then passed through the columns by suction at flow rates of approximately 100 – 500 mL h-1. Then columns were washed twice with 5 mL HCl (1 mM) and dried under reduced pressure. Elution was performed with four portions of 2 mL acetone. Dimethylsulfoxide (100 µL, 99.5%, Sigma-Aldrich Sweden) was added and the eluate mixed and divided into four equal portions. The acetone in each portion was then evaporated with a gentle stream of nitrogen. The final extracts were stored at –18 ˚C until assay.

The assay of estrogenicity was performed with a recombinant yeast strain (Saccharomyces cerevisiae) transfected with the human estrogen receptor gene, using the procedure essentially as outlined by Routledge and Sumpter (1996). The assays were performed in triplicate on 96-wells microtitre plates. Each plate was filled with one row of a dilution series of 12 concentrations of 10 µL 17β-estradiol in ethanol, 0.1 – 500 ng L-1 final concentrations with a dilution factor of 1.8, as a positive control. One row of twelve wells contained uninduced assay medium (blank, negative control). The remaining six rows were used for assay of test samples with a dilution factor of 2 between each of the 12 concentrations. After adding 200 µL of assay medium containing the yeast strain and the chromogenic substrate to each well the plates were incubated for three or four days in darkness at 30 ˚C. Once daily, the plates were shaken for 30 s at 5 times per second. Absorbance was then measured using a plate reader (Spectracount, Packard) at 540 and 620 nm.

At higher concentrations of sample extracts, an inhibitory effect sometimes coincided with inhibition of cell growth due to toxicity. This was corrected by measurement of the turbidity at 620 nm. Values of A620 were examined and significant deviations from the average turbidity (calculated from average and standard deviations in medium controls) were located in the wells of the microtitre plates. Such wells, usually containing the highest concentrations of extracts were omitted in the following data collection and treatment.

Estrogenic and androgenic effects were calculated by a non-linear fit to the experimental data. Dose-response curves from the absorbance at 540 nm for the concentrations of positive controls and sample extracts were drawn, and a nonlinear exponential fit to the experimental data was carried out with the Solver program in Microsoft Excel (Microsoft, Redmond, USA). The median effective concentration values (EC50) and slopes of dose-response curves were derived from a minimization of the sum of deviations of the nonlinear fit and the experimental data calculated according to

$$A = A_{min} + (A_{max} – A_{min}) * (C_i/EC_{50})^s / (1 + (Ci/EC_{50})^s) \qquad \textbf{Eq. 3.1}$$

where A is the calculated absorbance, A_{min} was obtained from the average of uninduced wells at low concentrations of sample, or, if not available, the average of blank values in the negative control. A_{max} was obtained from the fully induced dilutions of the dose-response curve, or if not available from the positive controls of estradiol or dehydrotestosterone. C is the concentration of sample extract or positive control, and *s* is the slope of the dose-response curve. EC50 of positive controls were obtained and expressed in weight concentrations (ng L-1). Using dilution factors, sample volumes and the relation to the effect in the positive controls, the values for water samples were recalculated and expressed as estradiol respectively dehydrotestosterone equivalents in ng L-1.

Androgenicity in extracts of the samples was measured using a yeast strain (Saccharomyces cerevisiae) transfected with the human androgen receptor gene (Sohoni and

Sumpter, 1998). The assay was performed in triplicate in the same way as the estrogenic assay. 10-µL dehydrotestosterone in ethanol, 0.1 – 500 ng L-1 final concentrations with a dilution factor of 1.8, was used as positive control.

YES and YAS analyses were performed by IVL Swedish Environmental Research Institute.

3.4.2.5 Toxicity

To measure the toxicity of the water after the treatment in different steps the above-described YES/YAS test and Microtox toxicity tests were used. Specific toxic substances such as N-Nitrosodimethylamine (NDMA), Dioxane and Methyl tert-butyl ether (MTBE) were used for the evaluation for some tests.

Microtox

Microtox analyses were performed according to the ISO 11348-3:2008 (modified) method that utilizes the light emitting ability of the marine bacterium *Vibrio fisherii*. The light emission is recorded after 5, 15, and 30 min of incubation of the sample. The exposure of the sample provides a dose response relationship, which is used to calculate the 20% (EC20) or 50% (EC50) inhibition of the light emission. If the tested sample has low toxicity, a single concentration test (90% of the tested sample) is performed. The results are expressed as percentage light inhibition of the sample (inhibition at 90%).

Measurements of bacterial bioluminescence is a physiologically relevant method of testing of chemical substances acute toxic effects and often show good agreement with other test organisms as micro-algae, zooplankton and fish. However, results of Microtox cannot readily be extrapolated to other species, and particularly caution should be used in assessments for recipients. For that, result from Microtox must not be used alone, but results from other tests advocated, such as microalgae, zooplankton, and fish.

Microtox analyses were initially performed by Toxicon laboratory (Sweden) but later by IVL Swedish Environmental Research Institute.

N-Nitrosodimethylamine (NDMA)

NDMA is an industrial by-product or waste product of several industrial processes. It is water-soluble, colorless, and its taste and odor are weak or absent. It is toxic to the liver and other organs. NDMA's contamination of water is of particular concern due to the difficulty in removing it from water, as it does not readily biodegrade, adsorb, or volatilize.

NDMA was analyses by Toxicon laboratory (Sweden) according to the method 521 version 1 (GC/MS/MS). 500ml of the sample was filtered, then acidified with Hydrochloric acid and 100ml water, then filtered with a Solid Phase Extraction (2 g) and finally extracted with 1.5 ml Dichloromethane.

Dioxane

1,4-Dioxane is a heterocyclic organic compound and used as stabilizer. It is a colorless liquid with a faint sweet odor similar to that of diethyl ether. It is classified as ether. 1,4 dioxane toxicity is somewhat unspecified but short-term exposure to relatively high concentrations regardless of the route of exposure harm liver and kidney. The substance also has a weaker toxicity towards aquatic organism.

Dioxane was analyses by Toxicon laboratory (Sweden) according to the method 522 version 1 (GC/MS). 500 ml of the sample was acidified with Hydrochloric acid, then filtered with a Solid Phase Extraction (2g) and finally extracted with 1,5ml Dichloromethane.

Methyl tert-butyl ether (MTBE)
MTBE is mainly used as additives to fuels in order to enhance the combustion efficiency, and is one of the toxics substances found in wastewater. The removal of MTBE in WWTP is generally poor (Potter *et al.,* 2009). The determination of trace concentrations of MTBE in water was performed on water samples (100 ml) that were conditioned in a purge and trap flask (250 ml) in a water bath (37°C) for 40 minutes. The flask was connected to a water trap tube and an adsorption tube (Tenax TA) and purged with helium gas for 40 minutes at a flow rate of 100 ml/min. The adsorption tube was then analyzed using thermic desorption/GC-FID.

3.4.2.6 Heavy metals
Heavy metals in the water phase were determined by ALS Scandinavia with ICP-AES and ICP-SFMS after extraction with HNO_3.

In sludge
A fraction of the sample was dried at 105 °C for TS analyses according to the standard SS028113. For metal analyses, the sample was dried at 50 ° C and concentrations were TS-corrected. Resolution occurred with aqua regia and the analysis was done according to EPA methods (modified) 200.7 (ICP-AES) and 200.8 (ICP- QMS).

In water
Resolution and analysis of water samples, 12 ml of sample and 1.2 ml HNO_3 (Suprapur) was treated in an autoclave. For the analysis of Ag resolution with HCl in an autoclave was applied. In the analysis of As and Se with high resolution (ICP SFMSHRM) the sample was prepared, 0.2 ml of sample and 1 ml HNO_3 (Suprapur) in a microwave. Analysis were done according to EPA methods (modified) 200.7 (ICP -AES) and 200.8 (ICP- SFMS) Analysis of Hg with AFS has been carried out according to EN ISO 17852:2008.

Reference metals
According to standards
- SS 28113 1981, determination of dry matter and ignition residue in water, sludge and sediments,
- USEPA Method 200.7 Determination of Metals and Trace Elements in Water and Wastes by Inductively Coupled Plasma Atomic Spectrometry,
- USEPA Method 200.8 : Determination of Trace Elements in Waters and Wastes by ICP-MS
- ISO 17852:2008 Water quality - Determination of mercury - Method using atomic fluorescence spectrometry

3.4.2.7 Assimilable Organic Carbon (AOC)

Assimilable Organic Carbon (AOC) is the fraction of total organic carbon (TOC) in a water sample that can be used by bacteria as a nutrient source. It has been considered a good indicator of the potential for bacterial regrowth in water distribution and storage systems leading to water quality deterioration and violation of regulatory standards.

As the bio-available fraction of organic carbon cannot be distinguished from the recalcitrant fraction with existing analytical equipment, Assimilable organic carbon (AOC) analyses were carried out around the O3-BAF treatment units. The difficulty to analyze bio-available fraction of organic carbon is partially due to the immense amount of various individual carbon compounds that can occur in water at extremely low concentrations (< 1 μg/L), combined with a lack of knowledge on the biodegradability of various individual compounds.

Bio-available fraction is typically assessed with biological growth assays that consider the combined fraction of bio-available carbon rather than individual compounds. These usually fall within two main categories, namely biodegradable dissolved organic carbon (BDOC) assays and assimilable organic carbon (AOC) assays. The AOC and BDOC methods are conceptually similar: bacteria degrade the bio-available carbon. The fundamental difference in the two methods is that BDOC assays assess the concentration of DOC removed through microbial growth (usually biofilm related growth), while AOC assays usually assess the amount of cells produced through utilization of bio-available carbon. The present study used AOC assays only performed by Eurofins Eaton Analytical, Inc., the largest potable water-testing laboratory in the US.

3.4.2.8 Adenosine triphosphate (ATP)

Extracting and quantifying the amount of adenosine triphosphate (ATP) gives a measure of how much living biomass a sample contains. Quantification of ATP was carried out using a luminescent assay kit from BioThema AB. The assay was performed in two ways, one measuring the total amount of ATP in the samples and the second one just the microbial ATP.

To measure the total ATP, all the cells in the sample was first lysed, thus releasing all the ATP to the test matrix. After adding a reagent, the light emitted from the sample was measured then an ATP standard was added to the sample. The amount of ATP was then calculated.

The quantification of only microbial ATP was carried out in a similar fashion. However a few more steps were added, first all ATP not from a microbial source had to be degraded. This was achieved by lysing all mammalian cells before an ATP degrading reagent was added. The microbial ATP is then protected inside the intact microbial cells when all other ATP is degraded. When this is completed, another reagent was added to stop the degradation of ATP and lyse the microbial cells. Now the only ATP present in the matrix is from microbial sources and it can be measured the same way as total ATP.

ATP analyses were originally performed by SP Sweden. However, due to inconsistencies in the analyses results, a modified sample preparation with grinding the media and double extractions and ATP determinations, first extracellular ATP and then bacterial ATP, was suggested to the laboratory by the project. Results improved but additional test with a third laboratory (Micans, Sweden) using exactly the same analysis method but another sample preparation methods provided more realistic and comparable results to other studies.

3.4.2.9 Extracellular polymeric substance (EPS)

EPS establish the functional and structural integrity of biofilms and determine the physiochemical properties of a biofilm. Thus, EPS are important in biofilm formation and cells attachment to surfaces. The EPS is estimated by the carbohydrate quantity based on the protocol for neutral sugar estimation. The samples are mixed with a phenol solution, the carbohydrates are hydrolyzed, and a colorimetric reaction between the phenol and the neutral monosaccharides are initiated by the instant addition of concentrated sulphuric acid. The yellow color is proportional to the carbohydrate concentration. The quantity is determined by an external standard curve based on dilutions of glucose.

EPS analyses were performed by SP Sweden.

3.4.2.10 Proteins

Two methods of protein determination were tried. The bicinchonic acid (BCA) protein assay is a modification of the Lowry procedure and relies on the formation of a Cu^{2+}-protein complex under alkaline conditions followed by reduction of the Cu^{2+} to Cu^{+}. BCA forms a purple-blue complex with Cu^{+} in alkaline environment. The concentration of the purple-blue colour is proportional to the protein concentration. The second method is based on the Comassie blue protein detection by Bradford. The comassie dye binds protein in an acidic medium and shifts absorption maximum from 465 nm to 595 in proportion to the protein concentration. Both methods use an external standard curve based on dilutions of a BSA solution.

The Lowry based method was suggested by the project and performed by SP Sweden.

3.4.2.11 Environmental Scanning Electron Microscope (ESEM)

The Environmental Scanning Electron Microscope (ESEM) with its specialized electron detectors allows for the collection of electron micrographs of specimens that are "wet," uncoated, or both by allowing for a gaseous environment in the specimen chamber. Compared to normal SEM specimens can be examined faster and more easily which implies that biofilms in the BAFs could be studied without the artifacts introduced during SEM preparation. ESEM analyses were performed by ALS Scandinavia AB.

3.4.2.12 Bromate/Bromide

Bromide is analyzed on a Dionex anion-chromatograph. The sample is lead with a carbonate eluent through an anion exchange column where the ions are separated. The eluent conductivity is reduced by a suppressor and the anions are then detected with a conductivity detector.

Bromate was analyzed on a Dionex anion-chromatograph. The sample is lead with a potassium hydroxide eluent through an anion exchange column, where the ions are separated. Eluent strength increases gradually through a gradient generator to provide the best separation in the shortest time. The eluent conductivity is reduced by a suppressor and the anions are then detected with a conductivity detector.

Analyses were performed at the IVL laboratory in Gothenburg.

3.4.3 Other quality parameters and quality control

Colour analysis were performed according to the standard for "Deutsche Farbzahl". The sample was filtered and analyzed with a spectrophotometric method in the WTW photoLab 6600 15 Color (FB436) 0.5 – 250 m-1 DFZ Measurement at 436 nm.

3.4.3.1 Microscopic examination

The sludge characteristic in the secondary treatment was regularly analyzed using standard microscopic examination. This was performed at IVL. Especially the flock structure with filaments growing (e.g. Microthrix) and bridging them together was of interest during periods with sludge settling problems. Microthrix is common in Swedish municipal treatment plants and can cause sludge bulking and sometimes foaming/scum formation. Filaments can grow under anaerobic, anoxic, as well as aerobic conditions and are therefore hard to control. Certain types of filaments such as high numbers of spirochaetes often indicate lack of oxygen.

3.4.4 Nutrient balances

Nutrient balances were used to evaluate the nutrient removal capacity, adapt and optimize the removal efficiency in various processes and as base for the environmental and economic impact assessment.

3.4.4.1 Nitrogen balance

Nitrogen balances calculations were used to verify calculated nitrous oxide emission from performed measurements (see next section). Detailed nitrogen balances over the processes were further used for the set-up of the environmental impact models, and based on measured nitrogen fluxes into and out of processes.

Nitrogen mass balances were calculated for the ICEAS when operating in both NIT and NDN mode, with a focus on NIT mode. Figure 3.7 and points below describes the calculation method used.

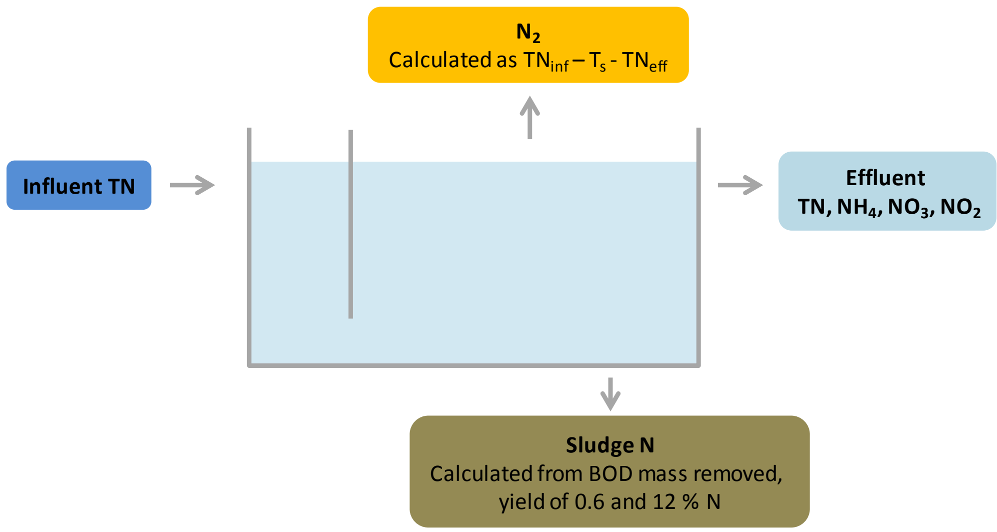

Figure 3.7. Calculation method used for Nitrogen mass balance.

- **Influent TN**: Mass flow per day calculated from daily composite samples of TN concentration in the influent water and the average measured influent flow from the same days. The daily composite samples were taken on average two to three times per week.
- **Effluent TN, NH4, NO3, NO2**: Mass flow per day calculated from daily composite samples of TN, NH4, NO3 and NO2 in the decanted water and the average measured influent flow from the same days. The daily composite samples were taken on average two to three times per week. Effluent organic and particulate nitrogen was further calculated as the difference between the total nitrogen and the NH4, NO3 and NO2 nitrogen.
- **Nitrogen in the wasted sludge:** Calculated based on the measured TN values in WAS and wasted sludge volume.
- **N2 gas produced**: The mass of N2 gas produced per day was calculated as the differences between the influent TN and the effluent TN and assimilated nitrogen.

3.4.4.2 Phosphorous balance

For the activated sludge process in the ICEAS, the mass balances for total phosphorous can be described as illustrated in Figure 3.8.

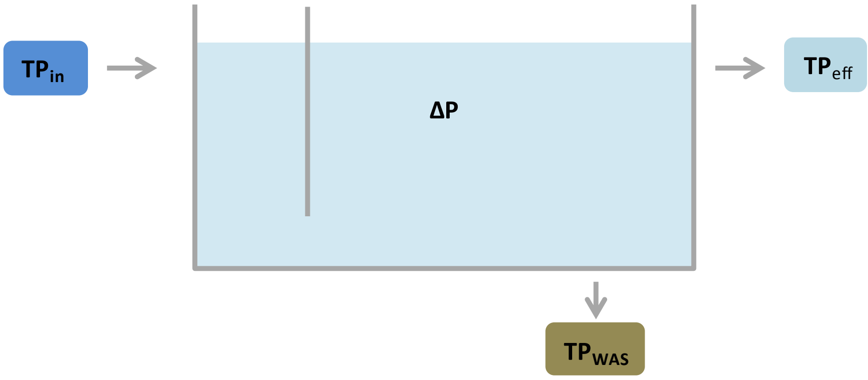

Figure 3.8. Phosphorus balance in ICEAS.

The corresponding mass flows are defined as:

$$TP_{inf} = TP_{WAS} + TP_{eff} + \Delta P \qquad \textbf{Eq. 3.2}$$

$$TP_{WAS} = P_{uptake\ in\ biomass} + P_{bio\text{-}P} + P_{precipitation} \qquad \textbf{Eq. 3.3}$$

TP_{inf} Total phosphorus in the influent to the system. Calculations were based on the lab analysis of 24-hour composite sample and daily flow rate to the mixing tank.

TP_{eff} Phosphorus lost in the effluent. Calculations were based on the lab analysis of 24-hour composite sample and daily flow rate to transfer tank.

TP_{WAS} Phosphorus lost in the wasted sludge. Calculations were based on the lab analysis of TP in the wasted sludge and daily wasted sludge flow rate.

ΔP Accumulation/consumption of phosphorus in the SBR per day - both in the liquid and sludge phases, i.e.:

$$\Delta P_{liquid} = (C_{TP_{eff},day1} - C_{TP_{eff},day0}) \times V_{SBR};$$
$$\Delta P_{MLSS} = (C_{MLSS,day1} - C_{MLSS,day0}) \times V_{SBR} \times 0.9 \times 0.02; \qquad \text{Eq. 3.4}$$
$$\Delta P = \Delta P_{liquid} + \Delta P_{MLSS}$$

with
0.9-the ratio of VSS/MLSS;
0.02- phosphorus content in the microorganism (2%) (This value is based on the daily chemical analysis from WAS-samples: TP/VSS ≈ 2%).

$P_{uptake\ in\ biomass}$ Phosphorus lost due to bacteria growth. The growth rate of the bacteria was calculated based on the soluble COD removal, biomass growth yield and cell phosphorus content. $sCOD_{in}$ and $sCOD_{eff}$ were determined from lab analysis of 24 hour composite samples. Biomass growth yield was set to 0.6 g COD in biomass/ g COD $_{(oxidized)}$. Dry biomass phosphorus content was calculated based on the TP and VSS value in wasted sludge sample, which was 2%. The ratio of s COD/BOD is 0.65.

$P_{precipitation}$ Phosphorus precipitation due to Fe in the influent. In the operational period, $P_{precipitation}$ was low and assumed as zero.

$P_{bio\text{-}P}$ – Phosphorus taken up due to bio-P.

Theoretically, when the SBR is under the stable condition, the changes of MLSS and phosphorus concentrations in the reactor should be close to zero. However, in practice after half a month of operation, there were changes of MLSS and phosphorus concentrations inside the ICEAS, which cannot be neglected. During the stable-state of each operational mode, the accumulation/consumption of phosphorus in the SBR (**ΔP**) should be in balance with the removal of WAS from the SBR and thus **ΔP** would be neglected in the mass balance.

P batch test

Batch tests to determine the phosphorous uptake capacity for different operation modes of the ICEAS were done. The principal experimental setup for two different scenarios with phosphorus addition and sampling is shown in the figure below. Each operational phase was 24 min according to the full-scale ICEAS operation. Oxygen was supplied intermittently to offer aerated /non-aerated phase. During the aerated phase, DO was tried to be maintained at 1.5-2 mg/L. Poly-phosphate was analyzed at each sampling points and COD was analyzed at the beginning and end of the non-aerated phase. Based on the concentration of VSS, the activity of PAOs was expressed as Δp (mgP/g VSS/phase). Positive values indicate poly-P released and negative values mean poly-P has been taken up.

Sample

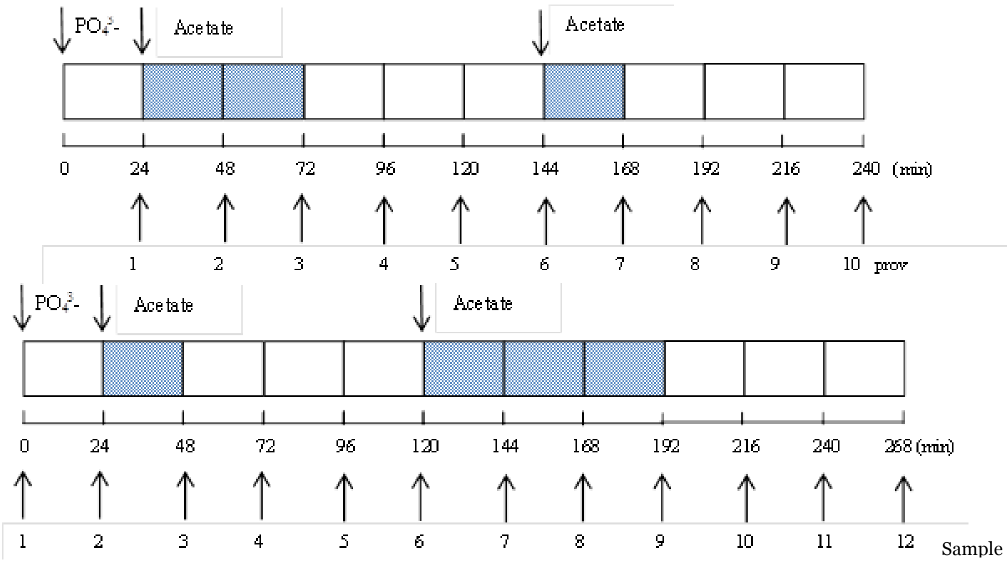

Figure 3.9. Principal setup of phosphorus-uptake batch-tests.

3.4.5 Nitrous oxide emissions

Nitrous oxide (N_2O) is an about 300 times stronger greenhouse gas than carbon dioxide and has been identified as the single most important ozone-depleting gas emitted in the twenty-first century (Ravishankara *et al.*, 2009) and is of special concern in wastewater treatment. Because of its high potential to affect the environment negatively, N_2O emissions will also influence the overall environmental impact assessment of the considered treatment trains. The actual measurement of emissions and consideration in the impact assessment is a significant contribution and one of the first studies including N_2O-emissions. Nitrous oxide is formed in biological wastewater treatment under both aerobic and anoxic conditions. Some of the reasons for N_2O emissions are low oxygen concentration during nitrification and low carbon/nitrogen-ratio during denitrification.

The setup of nitrous oxide measurement is shown in Figure 3.10. The PRZ was covered and isolated from the MRZ. Clean air was introduced above the water surface in the PRZ to maintain a constant airflow. The off-gas flow was continuously measured. A portion of the off-gas stream was pumped to the online instrument after cooling and drying. For measurement of N2O in the MRZ, a floated chamber was used to cover part of the water surface. The covered hood was also supplied with clean air to dilute and maintain a constant tested airflow when necessary.

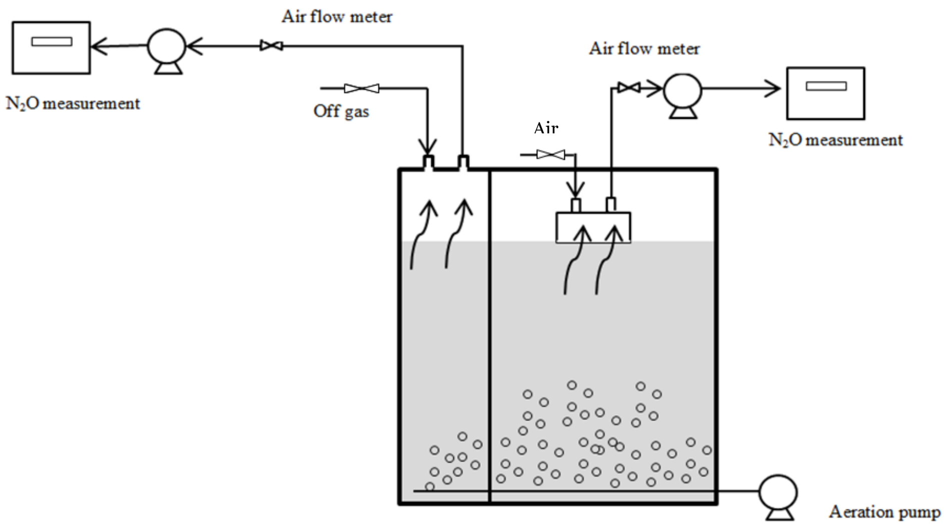

Figure 3.10. Principal setup of the off-gas measurements in the ICEAS.

The measurement results are expressed as N_2O-N/TN_{load}, N_2O-N/$TN_{removed}$, N_2O-N/TKN_{load} and N_2O-N/$TKN_{removed}$.

Measurements were solely carried out in the MRZ when the reactor was operated in NIT mode. In cases that only data from the PRZ was available for periods when the ICEAS has been operated in a different mode, total emissions from both MRZ and PRZ required assuming a certain ratio for the PRZ, initially assumed to be 0.2×MRZ.

The calculation of total emissions from the MRZ is shown as below:

$$\frac{C_{N_2O(MRZ)} \times Q_{airflow_{MRZ}} \times 1.94 \times \frac{28}{44} \times \frac{273}{293} \times 42.8 \times 1.2}{TN_{load}\left(TN_{removed,} TKN_{load,} TKN_{removed}\right)} \qquad \textbf{Eq. 3.5}$$

with

C_{N2O} – Concentration of the N_2O in the tested stream (ppm);

$Q_{airflow}$ – Air flow (11.4 m^3/d);

1.94 – converting factor from ppm to mg/m^3 at 0 °C (273K);

42.8 – the ratio of the areas between MRZ and the part covered by the hood.

1.2 – set value for N_2O emission from (PRZ+MRZ) compared with emission from MRZ

When the reactor was operated in NDN mode, N_2O measurements were carried out both in the PRZ and MRZ. Calculations are done according to below:

$$\frac{C_{N_2O(MRZ)} \times Q_{airflowMRZ} \times 1.94 \times \frac{28}{44} \times \frac{273}{293} \times 42.8 + C_{N_2O(PRZ)} \times Q_{airflowPRZ} \times 1.94 \times \frac{28}{44} \times \frac{273}{293}}{TN_{load}\left(TN_{removed,} TKN_{load,} TKN_{removed}\right)}$$

Eq. 3.6

with

$C_{N_2O(MRZ)}$	– Concentration of the N_2O in the tested stream (ppm) from the MRZ;
$Q_{airflowMRZ}$	- Air flow in MRZ (11.4 m^3/d);
$C_{N_2O(PRZ)}$	– Concentration of the N_2O in the tested stream (ppm) from the PRZ;
$Q_{airflowPRZ}$	- Air flow from PRZ (2.6 m^3/d);
1.94	- converting factor from ppm to mg/m^3 at 0 °C (273K);
42.8	- the ratio of the areas between MRZ and the part covered by the hood.

3.5 Sustainability Assessment Framework

Develop a sustainability assessment framework and evaluate both environmental and economic factors for the treatment trains piloted was the goal of this task. The "reference treatment trains" were assessed using Life Cycle Assessment tools that build on previous IVL work. The LCA results were evaluated to determine the most environmentally sound solutions which produce the required reclaimed water quality for 20 000, 100 000, 500 000 pe plant designs. In addition, a Life Cycle Cost (LCC) analysis was performed for the same configurations used in the LCA analysis. The results from the LCA and LCC analyses were combined to come with a recommendation for the "best" overall solution, which considers environmental, social, and economic factors.

3.5.1 Treatment Modelling

3.5.1.1 Preparatory work and Literature survey

A large number of environmental assessments of water treatment and water supply systems have been published. The scope of this survey was limited to environmental and economic assessment of water reclamation systems, alone or as parts of water supply systems. The goal of the survey was to obtain an overview of the methodologies used to assess the environmental and economic characteristics of water reclamation systems. The overview served as a guide for the ReUse project. The survey was also used to identify the most significant experts in this field for the ReUse reviewing group.

Implications and recommendations for the ReUse project were to use the selected LCA methodology for holistic sustainability assessment, based on specific modelling of the core processes and peripheral components described using input/output models. The data for the core processes must pertain to real full-scale applications and reflect normal and actual operating conditions. Upstream data for the supply of commodities can be collected from available literature or databases. The selection of this data should reflect the conditions and locations of the intended application.

The project ReUse is limited to the study of technologies to treat a given wastewater to a quality sufficient for non-potable reuse. The upstream boundary of the assessed system will thus be chosen as the wastewater at the point of intake to the water reclamation plant, i.e. water treatment and processes upstream of such a treatment are not included. The downstream boundary is the treated water including treatment of wastes, most importantly sludge, from the process but retentates etc., are also be included. The proposed methodology was the use of sludge as fertilizer.

Further it was decided to include at least a rough estimate of the construction phase in order to judge, whether the supply of construction materials contributes significantly to the environmental impacts or not.

Because normalization and valuation procedures may not be necessary to reach conclusions, it was decided to use these means only to communicate results internally. To the scientific public results are reported according to usual LCA standard procedures using a top-down approach. Average European reported values were used as a baseline for upstream processes (e.g. UCTE - Union for the Co-ordination of Transmission of Electricity for electricity) and normalizing used European or EU member states targets if available (such as Spain and Sweden), or the CML database (database that contains characterization factors for LCA).

3.5.1.2 Mass balance Models

The SBR was modelled using back calculations from an existing Xylem sizing tool within Microsoft Excel. For some parameters that are not included in this tool new models as described for the tertiary treatment steps were developed. The development work for the tertiary treatment models was done in MATLAB/Simulink (the treatment train models) and SIMCA-P environment (multivariate models to be implemented into the treatment train model). The SBR model and the tertiary treatment models were then connected within the MATLAB/Simulink environment.

3.5.1.3 Empirical MP models

The models consisted of mechanistically/physical models and/or multivariate regressions models depending on the results from the experiments, parameters considered and previously developed models. Prediction models built from the pilot scale test were validated using pilot and full-scale benchmark data. Then the models were scaled up to the three full-scale plants sizes (20 000 pe, 100 000 pe and 500 000 pe). When the model development was finalized, the results were used as input data for the LCA work.

3.5.1.4 Data quality – Selection of inventory data

Operational data for the core system is specific data from the experiments and the modelling in Matlab. Data on materials and construction of the equipment is specific design and engineering data from Xylem Inc. Data on supplied chemicals and energy wares is generic data from life-cycle inventory databases. Data is selected to meet the following criteria in the specified order:

1. Plausibility
2. Consistent with the geographical boundaries
3. Consistent with the temporal boundaries.

Missing data (data gaps) are filled in with analogues or approximations.

3.5.2 LCA

For the studied core processes, LCA needed the same data that was needed for the design of full-scale plants, i.e. the data that the pilot-plant experiments and the prediction models were supposed to deliver anyway. For modelled scenarios for which there was no experimental data, like operating different unit processes under different conditions, the prediction models was used to predict the required design and operation to reach a set of results. In addition, in order to calculate the KPI:s, also data was needed on direct emissions to air and water from the treatment plant, such as any emissions to air, e.g. from the SBR and from an ozone generator.

The ReUse system analysis (LCA and LCC) included all necessary upstream and downstream processes. A complete system description is displayed in Figure 2.1. Each treatment train is assessed by an attributional life-cycle assessment (ISO14044:2006, “Environmental management – Life cycle assessment – Requirements and guidelines”) which comprises the treatment from the influent water to the reclaimed water. The system is divided into three parts, namely the pilot-studied part of the system, the modelled part, which comprises also the pilot-studied part, and the peripheral part, which describes the supply of energy and other commodities.

The modelled part is the core process of the treatment train. The inflows (untreated wastewater, energy wares, chemicals, materials for construction, machine work) and outflows (reclaimed water, sludge, direct emissions from the site) are calculated by mathematical modelling from the pilot-plant data, supplemented with other data as necessary. The entire core process is then condensed into an input-output module, which is used as the core module of an assessment model in LCA software.

The LCA:s was carried out by exporting the primary results of the modelling of the core processes in MatLab/Simulink as an aggregated module to GaBi v. 6.3 (*PE International, Leinfelden-Echterdingen, Germany*). Then the final inventory of the system including upstream and downstream processes was compiled and the environmental profile calculated there.

The upstream and downstream processes supply energy and other commodities and dispose of waste. Applicable data, which describes the expenditure of resources and the emissions to the environment from these operations, was collected from the literature, usually in the form of modules from databases. These modules were the same throughout the project and only reacted to changing demands for energy and commodities. LCA thus provided a static description of the average performance of the treatment systems.

The data delivered to the LCA model related to real plants (the three studied plant sizes) designed for the basic setting of the scenarios. A data set based on full-scale installations and pilot-operation gave the design and operation parameters and the treatment result.

3.5.2.1 Functional Unit

The functional unit was defined as one (1) m^3 of reclaimed water delivered by the system for the intended purpose and meeting or exceeding the specified quality requirements for this purpose. Two systems, which deliver reclaimed water for the same purpose but of different quality, were thus considered to deliver the same function, as long as the reclaimed water fulfils at least the minimum requirements for the intended purpose.

3.5.2.2 System Boundaries

As a basis, the system boundaries are those laid out in Figure 2.1, i.e. upstream the untreated influent wastewater and the natural resources, which are necessary to generate energy, produce material commodities and construction materials and services, and to transport materials to the site of the plant. Downstream the boundaries are the reclaimed water at the outlet from the plant and the sludge after treatment on-site ready for transport to disposal. Decommissioning and disassembly of the plant and scrapping of the equipment are not considered in the system.

It follows from the definition of the functional unit, that the use of the reclaimed water is not part of the system. Differences in water quality may cause different environmental impacts in the use phase, but such differences are not considered, as long as the minimum requirements are fulfilled, i.e. a system, which does not meet the minimum requirements does not deliver the desired function, and is thus not considered at all.

In a sensitivity analysis to assess the importance of sludge disposal, the system boundaries of the treatment trains AG1 and AG2 were extended to comprise sludge disposal by use of the sludge as an agricultural fertilizer (see Section 4.2.6.2). The analysis also uses system expansion to calculate saved impacts from avoided fertilizing with mineral fertilizers. Avoided impacts from alternative sludge disposal methods are not taken into account, however.

3.5.2.3 Geographical Boundaries

The regions of focus for water reclamation projects are actually the Middle East, India, Latin America, and Australia. However, the European country Spain is a good proxy for these regions, and since region-specific inventory data is more easily available for Spain than for the above-mentioned regions, we have chosen Spain as a model country for implementation of the water reclamation systems. This means that electricity is modelled as supplied from the Spanish grid. Commodities like chemicals were assumed to be produced in Europe and the manufacture is as far as possible modelled with average European data. No specific site in Spain has been selected. Logistics are modelled by a standard assumption that materials are transported to the site 300 km by truck, with some exceptions. Details can be found in the section on inventory methodology.

The energy mix was in some cases replaced or compared to the Swedish energy mix and the one from the United States to investigate the significance of the energy origin.

To put the environmental impacts received in the ReUse project into a wider context, yearly emissions from EU25 and EU25+3 (including Norway, Switzerland and, Iceland) were used

3.5.2.4 Temporal boundaries

The core of the treatment system, i.e. the pilot-studied and modelled processes in Figure 2.1, is described by data, which pertains to best available technology in 2013. For the peripheral processes, most recent data has been selected. This means that the models of the peripheral processes reflect average technology during the period 2000 – 2010. The Spanish electricity is the average mix for the year 2012.

The assessment period is the survivable time, which is defined as 100 years. This has the implications described in the section on impact assessment boundaries.

3.5.2.5 Impact Assessment boundaries – Key performance indicators

As environmental key performance indicators maximum potential impacts (midpoint indicators) as they are defined and calculated in the life-cycle assessment methodology for selected impacts (see for instance Guinée *et al.,* 2002) were used. This means that selected performance indicators measure physical or chemical effects that have the potential to cause damage. The indicators do not describe the actual damages as such or the extent to which they actually occur at a given location.

Table 3.3 lists the impacts selected as key environmental effects. The selection is based on the findings of our literature survey (see 3.5.1.1). The table also briefly describes how these impacts are characterized at the midpoint level of the cause-effect chain. As characterization system, i.e. as method to calculate the key performance indicators from the inventory of emissions and flows of resources, we have selected the CML system (CML 2013). This database is widely used and geographically more generally applicable than the ILCD system (ILCD Handbook, 2011). The latest update (April 2013) of the characterization factors are used as available in the LCA software GaBi, v. 6.3. Table 3.3 specifies the impact assessment.

Table 3.3. Environmental impacts – Key performance indicators (KPI).

KPI	Unit	Calculation and contributing emissions (examples)
Global warming potential (GWP)	kg CO_2 equiv.	From an inventory of emissions of greenhouse gases; Greenhouse gases (CO_2, CH_4, N_2O)
Acidifying potential (AP)	kg SO_2 equiv. (or moles H^+)	From an inventory of emissions of acidifying compounds to air and water; (like SO_2, NO_x, NH_3, mineral acids)
Eutrophication potential (EP)	kg PO_4^{3-} equiv. (or kg NO_3^- equiv. or kg O_2 equiv.)	From an inventory of emissions of nitrogen and phosphorus compounds to air and water and of biodegradable organic compounds to water; BOD/COD, N and P compounds
Photochemical ozone creation potential (POCP)	kg ethylene equi.	From an inventory of emissions of volatile organic compounds to the air in the presence of NO_x.
Ozone depletion potential, destruction of the stratospheric ozone layer (ODP)*	kg CFC-11 equiv.	From an inventory of emissions of organohalogens to air. Difficult to get accurate values.
Depletion of abiotic resources ADP ▪ ADP elements (kg) ▪ ADP fossil (MJ)	kg Sb equiv./MJ of resource	Extraction of non-renewable material resources $ADP_i = [DR_i/(R_i)^2] \cdot [(R_{Sb})^2/DR_{Sb}]$ R_i = reserve of resource on earth (kg) DR_i = extraction rate of resource (kg/year) R_{Sb} = reserve of the reference resource antimony on earth (kg Sb) DR_{Sb} = extraction rate of the reference resource antimony (kg Sb /year)

Table 3.3 cont.

KPI	**Unit**	**Calculation and contributing emissions (examples)**
Ecotoxicity potential (inf.): ▪ MAETP - Marine Aquatic Ecotoxicity Potential; ▪ FAETP - Freshwater Aquatic Ecotoxicity Potential; ▪ TETP - Terrestric Ecotoxicity potential)	kg DCB equiv. (1,4 - dichlorobenzene) or m^3·days (CTU_e)	From an inventory of emissions of potentially ecotoxic compounds, e.g. “heavy” metals, pharmaceuticals, other organic micropollutants. The unit of measurement depends on the fate and effect model used. The time horizon was infinite (inf.)
Human toxicity potential	kg DCB equiv. or number of cases (CTU_h)	From an inventory of emissions of compounds potentially harmful to human beings. The unit of measurement depends on the fate and effect model used. The time horizon was infinite (inf.)

[*] ODP has been excluded from the considered KPI because of its present insignificance.

As the time horizon for the assessment was set to 100 years, the surveyable time is defined to the same time, except for toxicity potentials. This has the following implications for the impact assessment:

- Nitrous oxide (laughing gas) is regarded as purely a greenhouse gas. Its eutrophication impact is neglected.
- So called long-term emissions (used in the Ecoinvent database (www.ecoinvent.org, Swiss Centre for Life Cycle Inventories) are excluded.

For toxicity potentials, values directly available in the GaBi-tool (*PE International),* which are the potentials integrated to infinite time, were used. For persistent compounds in the environment, the difference between the calculated toxicity potentials at the surveyable time and at infinite time may be considerable. For example, the Aquatic Ecotoxicity Potentials (AETP) for marine waters may be significant depending on hydrogen fluoride (HF) emissions to air and on the time horizon to which the effect is integrated. An infinite time frame caused an anomalous impact of HF on Marine AETP that could not be supported by direct contact with CML (Huijbregts 2014) and its value was therefore modified to $4.1 \cdot 10^3$ kg DCB equivalents /kg HF. This was considered more realistic when considering the other potentials with a 100 years period of integration as fluoride has an extremely long residence time in the marine environment. Nonetheless, the fate and effects of fluorides are extremely difficult to predict in the aquatic environment with simple models, like USES-LCA (Huijbregts 2014).

3.5.2.6 Inventory methodology

Core processes include all the processes used directly in the treatment trains as described in section 3.2. The inventory of core process equipment comprises all construction materials as far as is practically possible at the time of the inventory. Operational data, such as use of energy, chemicals etc. when the equipment is operated as part of a treatment train, was not included. Such data was collected from the modelling of the operation of the treatment trains and included in the gate-to-gate unit-process modules of the entire treatment trains.

All data was provided by Xylem Inc. as design data for each type of equipment. From the design data, quantitative lists of materials were compiled. Wherever possible, missing data was supplemented by information, assumptions, and calculations of our own. Finally, the lists of materials are transformed into input-output lists per 1 piece of equipment for insertion into GaBi-modules.

Projected excavation work on the site of construction is included in the inventory, but otherwise assembling is not included, nor is decommissioning of the plant and the disposal of the construction materials after dismantling.

Required replacement of major parts of the equipment during its projected service life is included, but not material expenditure for daily routine maintenance, like lubricating oils, fuses, ordinary light bulbs, paint, putty etc. Replacement of UF membranes and of UV lamps is included. Transports of construction materials to the site are included by default assumptions (see the section about transports).

A service life of 20 years was assumed as a default for each assembled piece of equipment. For each part of the equipment with a specified service life the quantity of material for that part is multiplied by a factor = 20/(service life of that particular equipment part). This recalculated quantity of material is then entered into the input-output list. Replacement of major parts of the equipment during the service life is thus taken into account, as is the case where a constructional part outlasts the equipment. (This is the case for concrete tanks).

The inventories of equipment are inputs to the modules of the entire treatment trains. Each input is a fraction of number of pieces of equipment, calculated as 1/(m^3 of water delivered during the service life of 20 years).

The constructional materials are followed back to their origin in natural resources. Generic data from the databases ProfDB (GaBi, PE International, Leinfelden-Echterdingen, Germany) and Ecoinvent (www.ecoinvent.org, Swiss Centre for Life Cycle Inventories) is used for this. The basic rule was to choose average most current available European data. In some cases, average German data was used when this is considered to be of better quality or average global data when this was more appropriate. As far as possible, missing data is filled out with data for similar or analogous products.

More information about selected data sources and model examples are available in Appendix 9.1. For transports of construction materials to the site, default assumptions as defined in the system boundaries are used (for details see Table 9.3).

The electricity model is based on data collected from IEA Statistics on Spanish electricity for the year 2012 (IEA 2013), www.iea.org), supplemented with data from the database ProfDB (*GaBi*, *PE International, Leinfelden-Echterdingen, Germany*). The basic data is reported in Table 3.4.

Table 3.4. Basic data for Spanish electricity in 2012 (*data from IEA (2013),* www.iea.org).

Source of power	GWh	% of production
+Combustible fuels	146 840	50.85
+Nuclear	58 879	20.39
+Hydro	23 002	7.97
+Geothermal/Wind/Solar/Other	60 059	20.80
=Indigenous Production	288 783	100.0
+Imports	8 209	
-Exports	19 593	
Electricity supplied	277 399	
Grid losses (in 2009)		3.40

(*) Calculated from IEA data from 2009

Split of combustible fuels, % of total production(*)	
Coal and peat	11.265
Oil	5.749
Gas	32.565
Biofuels	0.808
Waste	0.461
Total comb.	50.85

Split of Geoth./Wind/Solar/Other, % of total production(*)	
Geothermal	0
Solar PV	2.835
Solar thermal	0.0104
Wind	17.792
Tide	0
Other sources	0.161
Total geoth. etc.	20.80

Imported electricity amounted to about 2.8 % of the indigenous production in 2012. If it is assumed that exported electricity is derived from the production mix + imports, i.e. that exported electricity has the same composition as the electricity supplied to the internal market, the supply mix can approximated with the production mix. The model of Spanish electricity is depicted in the Appendix (Figure 9.2). The data for the individual power plants was collected from the ProfDB. The figures for some minor contributions from coal gases and lignite were also estimated from the ProfDB. The reported flows were adjusted for grid losses.

3.5.2.7 Normalization and Aggregation

Normalization means that the value of each impact category indicator is divided by some kind of reference value. The reference value may for instance be the total value of the impact category for a given region, such as for a country, for Europe or for the world (EN ISO 14044:2006). Each KPI (total 5+5 different KPIs) was divided with a normalization reference (kg equivalents per year), and then multiplied with the flow as the example for GWP illustrates:

$$0.373 \text{ [kg } CO_2\text{-Equiv/m}^3\text{]} \mathbin{/} 5.2E{+}12 \text{ [kg } CO_2\text{-Equiv/year]} = 7.2E{-}14 \text{ [1/m}^3\text{]} \qquad \textbf{Eq. 3.7}$$

The Table below contains a set of normalization references from the Centre for Environmental Science, Leiden University, and The Netherlands (CML). The data set has been collected from the database of the LCA software GaBi. The references are total yearly emissions per year in the specified regions. In this project, normalization as performed with normalization reference for EU27+3.

Table 3.5. Normalization references.

Normalisation factors[1]	EU25+3[2]	World[2]	Unit
ADP elements	6.04E+06	2.1E+08	kg Sb-Equiv.
ADP fossil	3.51E+13	3.8E+14	MJ
AP	1.68E+10	2.4E+11	kg SO2-Equiv.
EP	1.85E+10	1.6E+11	kg Phosphate-Equiv.
FAETP	2.09E+11	2.4E+12	kg DCB-Equiv.
GWP	5.21E+12	4.2E+13	kg CO2-Equiv.
HTTP	5.00E+11	2.6E+12	kg DCB-Equiv.
MAETP	4.45E+13	1.9E+14	kg DCB-Equiv.
POCP	1.73E+09	3.7E+10	kg Ethene-Equiv.
TETP	1.16E+11	1.1E+12	kg DCB-Equiv.

[1] -from Gabi 6.3 CML2001 - Apr. 2013
[2] - year 2000. CML, IPCC, ReCiPe

If all environmental KPI:s are divided by the corresponding normalization reference, they will all be recalculated to the same unit of measurement, namely annual equivalents. They can thus arithmetically be added to one single indicator. This aggregation means that all normalized KPI are given the weight of 1 (equally important) and then summarized to a single number.

3.5.2.8 Important assumptions and simplifications

The German standard ATV –DVWK-A 131E (2000) was used as a base for the dimensioning of the biological treatment for the three selected full-scale plant sizes (20 000 pe, 100 000 pe and 500 000 pe). The ATV standard values for load per person (see Table 2.1) were used as a base for the full-scale design. In addition, peaking factors of 3 for the smallest size (20K pe), 2.5 for the middle size (100K pe) and 2 for the largest plant size (500K pe) were used.

All internal backwash waters from tertiary treatment as well as reject water from sludge dewatering are returned back to the influent of the ICEAS- SBR and are therefore included as internal loading to the plant. Note that in the ReUse-pilot such backwash streams were not returned to the inflow because of specific system setup requirements (see 3.1.1). Furthermore, the minimum temperature of wastewater of 10 °C was used for sizing the biological secondary treatment step because the full-scale plants are to be positioned in Spain. Performance parameters like energy consumption and chemicals consumption were collected from full-scale plants.

Decommissioning and disassembly of the plant and scrapping of the equipment were not considered in the system. The definition of the functional unit further implies that the use of the reclaimed water was not part of the system. Differences in water quality may cause different environmental impacts in the usage phase, but such differences are not considered, as long as the minimum effluent quality requirements (Table 2.2) are fulfilled. This simplification allows for the comparison of the different treatment options which otherwise would be affected by the environmental impact of the effluent (and sludge).

The fate and effects of fluorides are extremely difficult to predict in the aquatic environment with simple models, like USES-LCA (Huijbregts 2014). The impact of hydrogen fluoride (HF) emissions on Marine AETP was modified to $4.1 \cdot 10^3$ kg DCB equivalents /kg HF.

Nitrous oxide (N_2O) was only regarded as a greenhouse gas. In view of the selected time horizon of 100 years, its eutrophication potential was not considered.

3.5.3 **LCC**

A holistic economic evaluation of the various investigated treatment trains for wastewater reuse over their entire lifetime was performed. Life Circle Cost (LCC) analysis is an economic method of project evaluation in which all relevant costs arising over the lifetime of a project are considered. A fundamental aspect in the LCC analysis is the calculation of the total annual treatment costs, including both CAPEX (Capital Expenditure) and OPEX (Operating expense). Project costs typically arise over a longer time span including cost for owning, constructing, operating and maintaining a plant until the end of the facilities' useful life. Generally, the LCC analysis needs to address only those cost categories that are relevant to the scope of the project.

In order to calculate investment costs required for the construction of the facility, the collection and determination of process step specific cost values was performed for all treatment units and for all three evaluated plant sizes. Based on these specific cost values, investment costs of different plant sizes and configurations were calculated.

For the estimation of operational costs, i.e. all costs incurred to maintain and operate a treatment plant, the relevant process variables of the main treatment units (e.g. oxygen consumption, sludge deposits, etc.) were linked to specific costs.

3.5.3.1 Assumptions

The LCC analysis had the same system boundaries as the LCA, i.e. it included the costs of acquiring the necessary consumables and the costs and possible revenues of the sludge treatment (energy generation and fertilizer extraction). In addition, the system was expanded to include the costs of construction and installation as these cannot be neglected. The LCC resulted in a total cost (capital + operation) per unit of reclaimed water during the service life of the plants. The cost of decommissioning the plant was not included in LCC evaluation.

3.5.3.2 Cost model used

LCC per a discounted cash flow analysis was used to obtain net present value (NPV) for 20 years at 5.5% interest rate and 5% (and 2%) yearly increases in power cost (operating costs). The LCC model was structured according to DWA cost structure document from 2006 (DWA, 2006). LCC analysis was performed according to DWA guidelines from 2011 (DWA, 2011) for three wastewater treatment plant sizes - 20 000 pe, 100 000 pe and 500 000 pe - representing the various treatment scenarios as studied in the project.

To simplify the LCC analysis, the one-time investment costs are incurred at the end of the year in which they occur. All running costs incurred during the operation phase are expressed as annual expenses incurred at the end of each year. In this way, if individual cost items that are arising within one year are summed up into one end-of-year amount, an annual series of costs is created (see Figure 3.11).

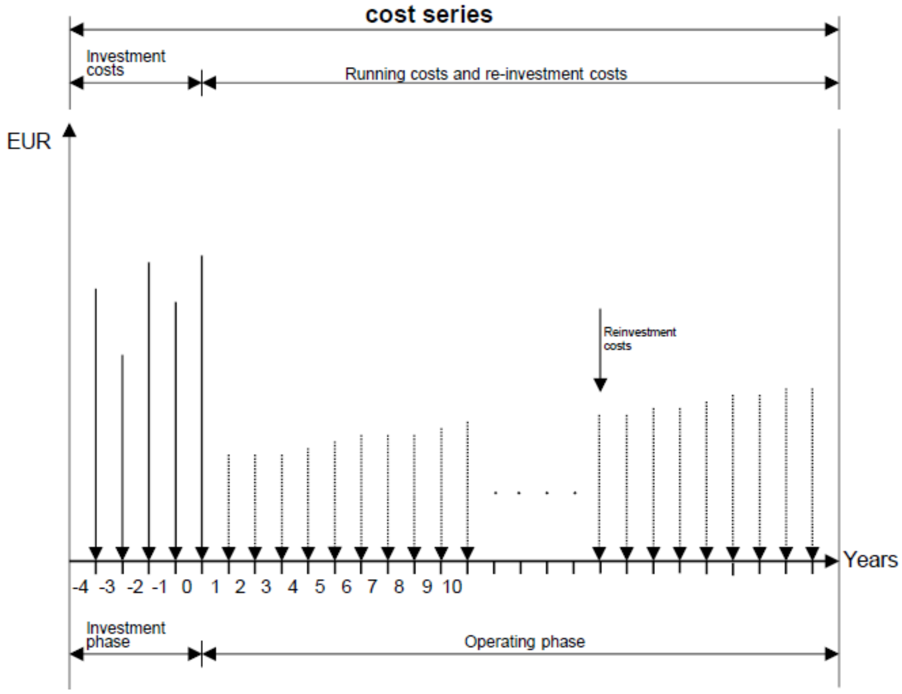

Figure 3.11. Basic terms of a time-based weighting of cost items (according to DWA, 2011).

The transformation of construction costs to equivalent annual costs can be done by using an annuity factor that is directly dependent on the economic lifetime and the discount rate. The following formula can be applied:

$$AN = INVEST \times \underbrace{\frac{i \times (1+i)^T}{(1+i)^T - 1}}_{Annuity\,factor} \quad ; i = \frac{p}{100} \qquad \textbf{Eq. 3.8}$$

where
AN = Annuity [€/a]
INVEST = Investment costs [€]
T = Economic lifetime [a]
p = Interest rate [%]

A distinction is often made between the economic lifetime of civil engineering parts and mechanical and electrical parts that usually have different life spans. In this project, the average economic lifetimes of 40 years was used for all civil engineering parts of the treatment plants, and 20 years for all mechanical and electrical engineering parts were used. This requires the process-related investment costs to be divided into the three cost parts:

1. Civil engineering
2. Mechanical engineering
3. Electrical engineering

Civil cost consists of excavation cost, concrete cost and installation cost. Mechanical cost comprises all equipment costs: diffusers, mixers, decanters, pumps, blowers, UV, RGSF, UF, Ozone, centrifuges, etc. including installation costs. It should be noted that for comparing the cost between eight different ReUse lines, Xylem intercompany prices were used for Xylem equipment, and not a final customer price. Spares and planed re-investments (due to shorter lifetime of equipment) are included in mechanical costs. In addition, piping and valves within the tanks are included in mechanical costs, while pipes between the tanks are excluded. Electrical costs consist of instruments, control and automation as well as installation cost.

The LCC was calculated for 20 yrs-life length of the plants, using interest rate of 5.5 %. The economic KPI's Service Life, Capital expenditure, Capital Cost, Operating costs (costs in $US) were used for a given functional unit as $/m^3 of treated water/year = $/(Average Dry Weather Flow)×365. They are specified by including the following items:

CAPEX

- Site specific costs excluded (HVAC is excluded from CAPEX)
- Balancing tank included in the ICEAS cost (mainly civil cost)
- Feed pumps needed for RGSF, pUF, permeate from sUF and for ozone were selected according to the hydraulic profile of the plant.

OPEX

- Maintenance of mechanical and civil calculated as % of CAPEX (1% civil, 1.5% Maintenance and operation , M&E)
- Energy cost =0.2 $/kWh
- Chemicals cost: polymer =4 $/kg polymer, NaOCl = 0.4 $/kg, FeCl3 = 0.3 $/kg, NaOH = 0.45 $/kg, HCl = 0.25 $/kg , citric acid =0.7 $/kg, LOX =0.2 $/kg
- Sand and gravel = 50 $/ton; Anthracite =257 $/ton

4 Results, discussion and conclusions

The tests with various treatment unit configurations and process parameters within the ReUse-project aimed at providing data for a thorough assessment of the total environmental and economic impact of wastewater reclamation for different purposes. The pilot tests were further used to identify optimal process configuration and operation and improvements of various techniques for implementation in full-scale. The following sections provide first results from the general performance of the treatment, improvements and limitations identified and if relevant realized. Then, results from the environmental and economic assessment are provided.

4.1 Pilot-system performance

4.1.1 ICEAS - Secondary treatment

The evaluation of the ICEAS system as secondary treatment step was based on different modes for partial (AG-NIT) and complete (NIT or NDN) nitrogen removal. Within each mode, variations of different operational parameters such as aeration time, settling time, and biomass content etc., were used to map their impact on the overall treatment efficiency in order to optimize the systems performance.

4.1.1.1 Optimization of the NDN operation mode

The NDN mode was operated through the complete project period (2012 - 2014) with different configurations (Figure 4.1) including 8h to 14h of aeration per day and with normal and 25% shorter cycle in 2014. The goal with the NDN modes with 10 h and 8h of aeration per day was to reach 10 and 5 mg/L of TN in the effluent, respectively. The goal with NDN 12 and 14 h of aeration was to reach 15 mg/L of TN. The goal with a short cycle was to increase the suspended solids out from ICEAS to 20 mg/L for ICEAS + pUF evaluation.

Except for 2012, when average ammonium (measured as TKN) and BOD loads were in the design range, a higher influent load was treated than the system was designed for initially.

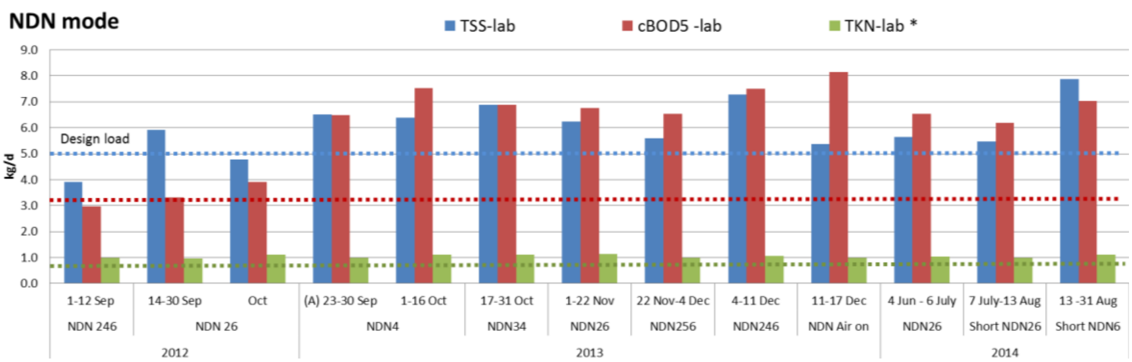

Figure 4.1. Monthly average loads in the influent compared to design load.

The performance during NDN modes was compared with target effluent quality in Table 4.1. It can be concluded that in NDN mode, the target of 10 mg/L of TN has been reached in 2012 with 10h of aeration per day (for example NDN26 that has two anoxic periods, second and sixth, that are each 24 min long). The target, however, was exceeded in both 2013 and 2014 during four months of operation. Target of < 5 mg/L TN was exceeded during one month of operation in 2014 with 8h of aeration per day (for example NDN246 that has three anoxic periods, second, fourth and sixth, each 24 min long). The reached low effluent quality of 3 mg/L TN has been confirmed with full-scale benchmark data.

It can be also seen that during 2012 and 2013, the target TP of 1 mg/L was reached without addition of chemicals but only with biological-P removal (Bio-P).

During 2012, the pilot treated wastewater from a different source (Sickla inflow to Henriksdal WWTP) with a lower load that implied a lower cBOD/TKN ratio and therefore lower denitrification and nitrogen removals. In 2013, the ratio between cBOD/TKN in the influent increased from 3.3 to 6.6, on the average, due to a switch to the Henriksdal wastewater that is more representative for urban sewage.

In July-Aug 2014, even with a 25% shorter cycle (last column in Table 4.1), an effluent nitrogen concentration of 5 mg/L was achieved. During the short cycle, a 90% higher BOD load and a 40% higher nitrogen load were treated due to higher inflow to the plant (23 m^3/d compared with designed 17 m^3/d). These excellent results were obtained with 2 to 4 times higher biomass in the process than design mass (MLSS) and with a low DO of around 1 mg/L.

Table 4.1. Monthly average effluent concentrations compared to target effluent quality during NDN mode.

NDN	Target 10h / 8h	2012 10h	2013 10h	2013 8h	2014 10h	2014 Short 10h
Flow (m^3/day)	17	24	21	19	19	23
$cBOD_5$ (mg/L)	10	4	5	4	5	9
TN (mg/L)	10 / 5	10	5	3	5	5
NH4 (mg/L)	1	1.5	0.5	0.9	0.8	2
TP (mg/L)	1	1	1.4	1	2.5	2
TSS (mg/L)	10	5	6	5	10	8

In spite of higher load than design, the ICEAS could reach 5 mg/L TN by increasing MLSS and operation at 1 mg/L DO. Target TP of 1 mg/L could not always be reached without chemical precipitation.

4.1.1.2 Optimization of the NIT nitrification and denitrification capabilities

The goal of the NIT operation was varying during the test period. In 2012, it was defined as to reach 15 mg/L of TN and 1 mg/L of ammonium in the effluent. In 2013, different attempts were made to inhibit nitrification and to reach 5 mg/L of ammonium for agriculture application in the mode called AG-NIT. In 2014, the goal for typical NIT mode was set to reach 10 mg/L of TN. The process was then operated at the design MLSS/SRT ratio during March 2014 and at a higher MLSS/SRT ratio during April-May 2014.

In Figure 4.2, monthly average influent loads (based on 24h composite samples) are compared to the design load. In general, the design NIT load was 20% higher than the design load for the NDN-mode in order to compensate for a larger basin that was originally designed for NDN-mode. All values except for flow and temperature are laboratory data based on the 24 h composite sample. It can be seen that in July 2012 the plant was under-loaded. In August 2012, the plant was overloaded in terms of TSS and ammonium. From Jan to Sep 2013, the pilot received in average a 32% higher BOD load and a 3% lower TKN load (lowest in July). During 2014, in the average a 36% higher BOD load and an 8% higher TKN load were treated.

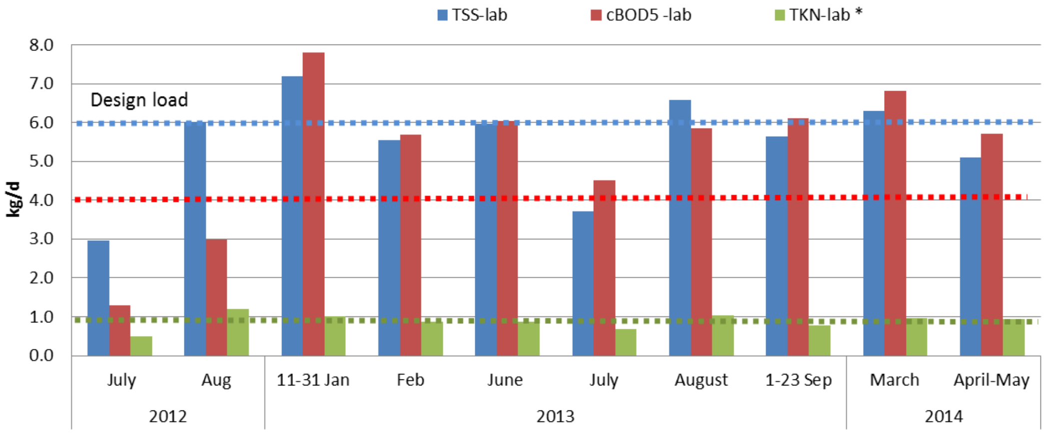

Figure 4.2. Monthly average loads in the influent compared to design load.

In Table 4.2, reached effluent quality during NIT modes has been compared with the targeted monthly average quality. It can be seen that in 2012, the target of 15 mg/L of TN has been reached during the two months of operation. During this time, phosphorus has been kept at the average value of 1.2 mg/L without addition of chemicals. During 2014, a new target of 10 mg/L of TN has been reached with 86% of total nitrogen being removed. As NIT modes do not have anoxic periods during the react phase for denitrification to occur, the hypothesis that most of denitrification occurred during settling and decant periods was further evaluated. With increased MLSS (only 700 mg/L above design) and SRT compared with design, in April-May 2014, the same effluent TN was reached compared to March 2014 when 1500 mg/L less MLSS was used to treat a higher load than in April 2014. During this period, lower SRT was also used comparing with design SRT.

Table 4.2. Monthly average effluent concentrations in NIT modes compared to the target effluent quality.

NIT	Target	2012	2013	2014
Flow (m^3/day)	20	17	17	19
Influent load				
$cBOD_5$ (kg/day)	4	2	6	7
TKN (kg/day)	0.9	0.8	0.9	0.9
TP (kg/day)	0.1	0.1	0.1	0.1
TSS (kg/day)	6	4	6	6
Effluent concentration				
$cBOD_5$ (mg/L)	10	4	6	6
TN (mg/L)	15-10	12	9	7
NH4 (mg/L)	1	0.3	1	1
TP (mg/L)	2	1	2	3
TSS (mg/L)	10	3	6	8

It can be concluded that during NIT mode (in 2014) the more stringent effluent target of 10 mg/L of TN has been reached at all time (with 100% of the values below the target!). This performance has also been confirmed by full-scale benchmark data.

In January and February 2013 during AG-NIT mode when an effluent ammonium of 5 mg/L was targeted for agriculture reuse, the maximum of 1.4 mg/L of ammonium could be reached. This was reached despite a highly overloaded process and a decreased DO to 1.5 mg/L. The average TN in this period was 7 mg/L with 88% of TN being removed. This proved that the ICEAS is a high performing process and that both nitrification and denitrification is difficult to inhibit. Therefore, during July-September 2013 the nitrification was optimized by decreasing the targeted dissolved oxygen from 1.4 to 1 mg/L, defining the mixed liquor concentration to 1600 - 1400 mg/L, and setting the SRT to 8 days in order to reach effluent ammonium of 5 mg/L in AG-NIT mode. In August 2013 (15^{th} to 31^{st}), the DO set point was 0.8 mg/L and air was turned off during the last 5 min of each of the three 40 min reaction periods. Due to DO probe failure, the actual DO was much higher than the set point so ammonium decreased to 1 mg/L.

During all NIT periods (except for AG-NIT that had a different target), in average 80% of total nitrogen was removed indicating the possibility for denitrification even in NIT-mode. In addition, the target effluent of 15 and 10 mg/L total nitrogen was reached on the 100%-ile basis. In addition, 66% of total phosphorus was removed (the highest TP removal was 77% during chemical P removal in June 2013 with in the average 1.5 mg/L of TP in the effluent achieved by addition of $FeCl_3$).

The ICEAS had a good nitrogen removal even in NIT mode, indicating active denitrification during settling and decant periods. It was difficult to avoid nitrification in order to keep high NH4-N for agriculture use.

4.1.1.3 Optimization of ICEAS controller

The ICEAS pilot has been used to develop and optimize several control features of the process (controller named OSCAR™):

- The SIMS (Solids Inventory Management System) SRT mode logic was improved in stability and accuracy compared to the previously used logic. This was observed at the pilot and confirmed by tests in full-scale. The accuracy of the controller was within the set criteria.
- Two new control modes for SIMS, the MLSS control and Smart SRT mode, have also been developed and optimized at the plant (Henriksson and de Kerchove, 2015).
- A large set of safety nets (safety algorithms) have been developed and tested, which detects invalid sensor readings or process upsets within a controller. When detected, the control switch to a safe mode and an alarm is sent to the operator.
- Initial testing has been conducted of the ICEAS ammonium control. The trials indicate an energy savings potential of 20-25 %, but full development of the controller require further testing.

- Significant testing has also been conducted on improving the ICEAS DO control. Although the targeted control stability has not been reached, the testing has provided learnings about DO control and operation.

Control systems for MLSS and SRT have been improved, and safety algorithms developed and tested. Control of NH4-N and DO has been improved but still not perfect.

4.1.1.4 Nitrogen removal capacity

The ICEAS nitrogen removal capacity was evaluated using mass balance studies (see 3.4.4.1). The objective was to determine the nitrogen mass balance of the reactor as well as the nitrification and denitrification rates during different operating modes and to use these results to evaluate and improve the current design method.

Focus was especially on the basins capacity to denitrify. During the whole project, the ICEAS showed significant denitrification capability when running in NIT mode. Similar denitrification capability has been noted at full-scale ICEAS plants in NIT mode. This indicates a capacity to denitrify which is today not included in the design methods of such plants and therefore provides a significant improving potential. To quantify this capacity, the denitrification capacity of different cycle modes was studied and defined and the results were validated with full-scale data.

Table 4.3 below shows the result of the nitrogen mass balance. The values illustrate the fate of nitrogen as percentage of influent TN. The ICEAS was operated in NIT mode during January, February and June 2013 and during March and April 2014, and in different NDN modes (with 10 hours of aeration per day and with 8 h of aeration per day) during October to December 2013. Only periods of stable operation was used.

Table 4.3. Results from nitrogen mass balance of the ICEAS basin (as percent of influent TN).

	NIT[1]	NDN (10h)[2]	NDN (8h)[3]
Nitrogen denitrified	53 %	65 %	66 %
Nitrogen assimilated*	30 %	27 %	29 %
Nitrogen released to the effluent	17 %	8.7 %	5.5 %
- *As NH4*	*2 %*	*0.9 %*	*1.6 %*
- *As NO3*	*9.5 %*	*3.5 %*	*2.0 %*
- *As NO2*	*0.4 %*	*0.9 %*	*0.7 %*
- *As organic and part. nitrogen*	*5.1 %*	*3.3 %*	*1.1 %*

[1] - Periods 130120 to 130213; 130221 to 130306; 130601 to 130631; 140301 to 140323; 140411 to 140508

[2] - Periods: 131017 to 131030; NDN34 131031 to 131121; NDN26

[3] - Period: 131112 to 131210

The following conclusions can be drawn from the observed data on nitrogen removal:

- When operating in NIT mode, the average measured effluent TN concentration was 8 mg/L. Full-scale data confirm that the NIT mode reaches effluent TN concentrations around 10 mg/L. This indicates a significant denitrification capacity, despite the lack of anoxic periods in the reaction periods.
- A nitrogen mass balance showed that on average 53 % of the influent nitrogen was denitrified in NIT mode. When adding 4 and 6 hours per day anoxic time during reaction periods, this number is increased to 65 % and 66 %, respectively. The high denitrification during NIT and the relatively small difference be-

tween NIT and NDN confirm that a large part of the denitrification in the ICEAS occur outside of the allocated anoxic time during react.

Based on these findings, a new design approach for denitrification in the ICEAS was developed which provides lower cost designs through reduced basin sizes and CAPEX with up to 15-20 % (Henriksson *et al.*, 2015).

The denitrification during non-react periods can be used to change the design criteria and save reactor volume.

4.1.1.5 Phosphorous removal capacity

Two studies have been carried out in order to understand and find out how the bio-P process in ICEAS works. In detail, we investigated:

1) the bio-P process performance and efficiency in ICEAS under different operational modes;
2) the activities of Polyphosphate Accumulating Organisms (PAOs).

The evaluation of the bio-P process performance and efficiency in ICEAS was analyzed based on the operational data obtained from duration March 7 - May 15, 2013 and October 17 - 31, 2013. During this time, ICEAS was operated under four operational modes. Batch tests were carried out to assess the activities of PAOs under different operation conditions. The scenarios of PAOs activity at different combinations of the non-aerated/aerated phases were discussed, which can give recommendation for future ICEAS operation.

The TP mass balance results in the ICEAS are shown in Table 4.4. The mass balance holds well when the operation mode was NDN34, which corresponds to 10h of aeration per day (NDN34 has two anoxic periods, third and fourth, each 24 min long). When the operational mode was NDNP1 (first 24 min period is anoxic) and NDNP135 (three anoxic periods leading to 8h of aeration per day), 15-18% of the phosphorus is missing. Possible explanations could be:

a) Errors in the measurement of flow rates;
b) Online MLSS readings (however, only values of ΔMLSS/day were used in the calculations, not the absolute values);
c) Probably there is some precipitation on the walls of the reactor during the operational period NDNP1 and NDNP135.
d) Impact of assumption in ΔP calculation, like the sludge yield.

Table 4.4. Summary of bio-P under different operational modes.

Operational mode	Name	TP removal (%)	Bio-P (% of influent)	Bio-P (% of removed TP)
■□□□□□□	NDNP1	70	0	0
■□■□■□□	NDNP135	71.3	11	15.4
■□□□■□□	NDNP15	67.5	26	38.2
□□■■□□□	NDN34	80.3	20	24.9

■ - non-aerated period of cycle
□ - aerated period of cycle

Table 4.5 summarizes the bio-P process performance and efficiency. NDNP1 did not show any bio-P performance. Operational mode of NDNP15 obtained the highest bio-P efficiency. Bio-P process also showed a high removal efficiency when the operation mode was NDN34.

Table 4.5. TP balance in ICEAS (as percent of influent TP).

Mode	Days	P balance				P in sludge		No. of data
		WAS	TP eff%	ΔP/d	P missing	P activated sludge	bio-p	
NDNP1	12	27.2	30.0	27.0	15.7	27.2	0.0	5
NDNP135	36	40.2	28.7	12.5	18.6	28.4	11.9	20
NDNP15	19	61.8	32.5	-1.5	7.2	35.4	26.4	7
NDN34	15	54.0	19.7	25.1	1.2	33.3	20.7	9

Different scenarios were calculated based on the batch test results (Table 4.6). It shows that operational modes with first anoxic period followed by either two or three aerated periods were reasonable operational modes, which give high P removal. During the batch tests, PAOs released poly-P very fast in a short period of non-aerated phase, which is because acetate solution was added in the batch tests. In the real ICEAS, VFA can be a limiting factor for PAOs.

Table 4.6. Scenarios of different operational modes.

Scenarios	Batch 1 ΔP (mgP/g biomass)	Batch 2 ΔP (mgP/g biomass)
■□	0.09	0.87
■□□	-0.18	0.06
■□□□	-0.32	-0.1
■■□		1.57
■■□□		0.83
■■□□□		0.06
■■■□	2.4	
■■■□□	0.2	
■■■□□□	-0.2	

■ - non-aerated period of cycle
□ - aerated period of cycle

The obtained values indicate that the ICEAS-system was to our understanding not in a steady state operation mode during any of these modes and P-removal rates should be used with care. In addition, the unclosed mass balance indicates that measured flows or/and concentrations include errors. We therefore recommend a detailed study of P removal during a limited number of cycles with intensive sampling and analyses at all significant locations (including PRZ and MRZ) and cycle times.

More tests are needed concerning Bio-P during different operation modes. Results indicate a lack of VFA.

4.1.1.6 GHG-emissions

Emissions of N2O from the pilot scale ICEAS were measured between July and December 2013. The ICEAS was operated in AG-NIT and NDN modes.

AG-NIT-mode (Sep, 2013)

In the period between July – September, 2013, the ICEAS was operated in the AG-NIT mode. The goal of this operational mode was to maintain a high ammonium concentration in the effluent (around 5 mg/L) needed for the agriculture reuse of the water. Since it was difficult to stop nitrification, especially given the high temperature in the water, different control strategies were tested. This included decreasing MLSS in the tank, decreasing DO to below 1 mg/L and lowering SRT to 3 days. Before a stable operation was achieved, measured N_2O-emissions showed high variations. The stable operation and steady state (2-3 SRT) was reached in the period between 1st of Sep and 15th of Sep. Only the results from this period were used for final emission calculations as shown in Table 4.7.

The results show that during the operation period under AG-NIT mode and with the simplifying assumption of 0.2MRZ for emissions from the PRZ, there was on average 1.68% of incoming total nitrogen load converted to nitrous oxide with a standard deviation of 0.75 (2.62% of TKN load, Table 4.7). The total nitrogen removal efficiency in AG-NIT mode was around 81%. Most of nitrogen in the effluent was in NO_3-N form. With AG-NIT operation mode, 2 hours continuous aeration enhances the N_2O production by AOB and gives no chance for N_2O consumption. All the produced N_2O in 2 hours aerated period immediately stripped out in the air.

Table 4.7. Summary of N_2O results when the reactor was operated under AG-NIT operational modes.

	Parameter	Number of measurement point	Average*	Std.v
1st Sep - 14th Sep	**N_2O-N % (TN)**	6	1.68	0.75
	N_2O-N % (TKN)	6	2.62	1.17
	N_2O-N % (removed TN)	6	2.09	0.90
	N_2O-N % (removed TKN)	6	3.64	1.54

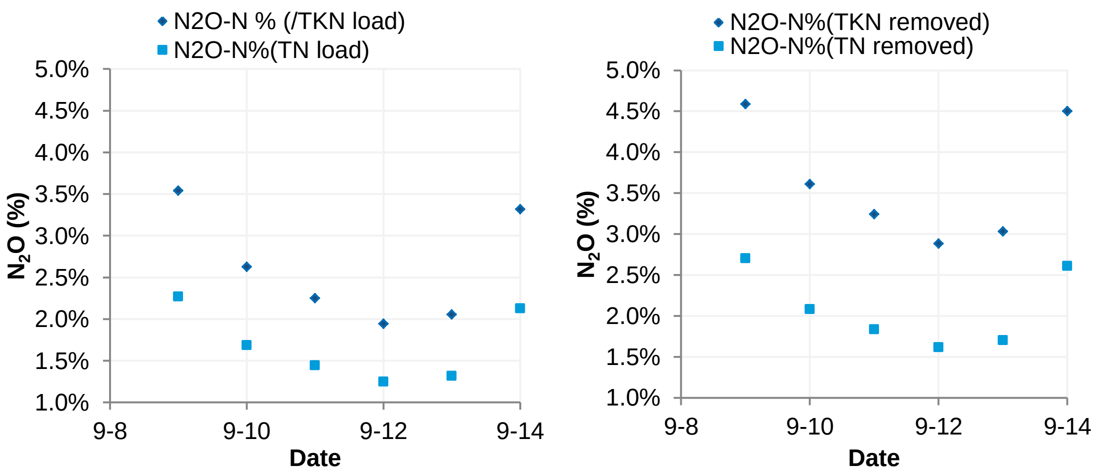

Figure 4.3. Example of nitrous oxide emissions in AG-NIT mode.

NDN-mode (Oct-Dec, 2013)

In the NDN operational condition, different modes were tested, which were NDN34, NDN26, NDN256 and NDN246.

During the operational time, the average of the total nitrogen removal reached 84%. From the end of October, Bio-P process was achieved and a high phosphorus removal was observed. Emission of N_2O-N (% TN) was 0.1% (0.18% of TKN load) on average, in other words, 0.22% of the removed TKN and 0.16% of the removed total nitrogen was converted to nitrous oxide (Figure 4.4). When the ICEAS was operated in NDN mode, part of the produced N_2O can be consumed by denitrification process and the emissions was therefore mostly less than 0.3% of the total nitrogen load. More details for different operational modes are shown in Table 4.8.

Table 4.8. Summary of N_2O results when the reactor was operated under NDN operational modes.

<table>
<tr><th>Operation mode</th><th>Parameter</th><th>Number of measure-ment points</th><th>Average</th><th>Std.v</th></tr>
<tr><td rowspan="4">NDN34 (18th-28th, oct)</td><td>N_2O-N % (TN)</td><td>11</td><td>0.097</td><td>0.047</td></tr>
<tr><td>N_2O-N % (TKN)</td><td>11</td><td>0.213</td><td>0.105</td></tr>
<tr><td>N_2O-N % (removed TN)</td><td>1</td><td>0.105</td><td>-</td></tr>
<tr><td>N_2O-N % (removed TKN)</td><td>1</td><td>0.299</td><td>-</td></tr>
<tr><td rowspan="4">NDN26 (8th-20th, nov)</td><td>N_2O-N % (TN)</td><td>10</td><td>0.057</td><td>0.016</td></tr>
<tr><td>N_2O-N % (TKN)</td><td>10</td><td>0.102</td><td>0.050</td></tr>
<tr><td>N_2O-N % (removed TN)</td><td>2</td><td>0.064</td><td>0.028</td></tr>
<tr><td>N_2O-N % (removed TKN)</td><td>2</td><td>0.100</td><td>0.035</td></tr>
<tr><td rowspan="4">NDN256 (27th nov-3rd dec)</td><td>N_2O-N % (TN)</td><td>7</td><td>0.209</td><td>0.292</td></tr>
<tr><td>N_2O-N % (TKN)</td><td>7</td><td>0.278</td><td>0.351</td></tr>
<tr><td>N_2O-N % (removed TN)</td><td>2</td><td>0.524</td><td>0.799</td></tr>
<tr><td>N_2O-N % (removed TKN)</td><td>2</td><td>0.624</td><td>0.666</td></tr>
<tr><td rowspan="4">NDN246 (5th-10th dec)</td><td>N_2O-N % (TN)</td><td>6</td><td>0.075</td><td>0.012</td></tr>
<tr><td>N_2O-N % (TKN)</td><td>6</td><td>0.125</td><td>0.016</td></tr>
<tr><td>N_2O-N % (removed TN)</td><td>1</td><td>0.111</td><td>-</td></tr>
<tr><td>N_2O-N % (removed TKN)</td><td>1</td><td>0.066</td><td>-</td></tr>
</table>

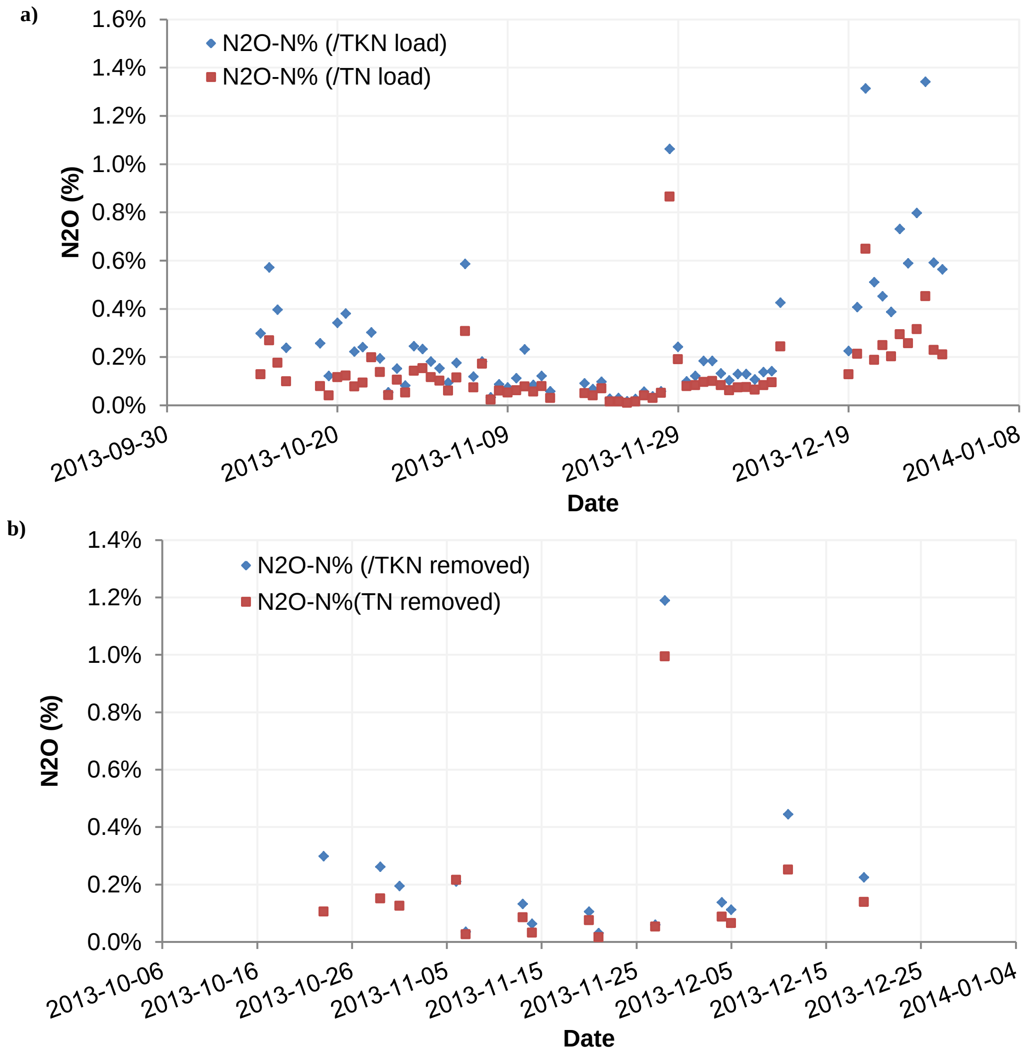

Figure 4.4. Nitrous oxide emissions when ICEAS operated under NDN mode.

In the nitrogen biological removal process, nitrous oxide emissions are significantly influenced by production rate and stripping. AOB and denitrifiers are the main sources for nitrous oxide production. Moreover, denitrifiers can consume N2O under complete anoxic condition. High nitrous oxide production can be due to not high enough DO concentration for AOB under aerobic condition or too high free oxygen for denitrifiers under anoxic conditions. Therefore, concentrations of DO and activities of AOB and denitrifiers are the main factors influence nitrous oxide production. However, higher production rate of nitrous oxide will not completely lead to a high emission if there is no stripping. In our measurement results, the emissions of nitrous oxide in the non-aerated phase were close to zero due to no stripping. It gives a direct proof that the duration of air supply would also influence the nitrous oxide emissions.

ICEAS was operated under different modes. The activities of the microorganisms can be indirectly evaluated by process performance. The stripping ability of the system can be simply checked by the aerated time (Table 4.9). Theoretically, when the nitrogen removal efficiency is high (which means that denitrifiers are active and can consume produced N_2O) and less aeration hours, there would be low emissions of nitrous oxide.

Table 4.9. Aeration hours and nitrous oxide emissions in each operational mode.

Operational mode	Aeration hours /day	N_2O-N % (/TKN load)	N_2O-N % (/TN load)	N_2O-N % (/TKN removed)	N_2O-N % (/TN removed)
AG-NIT	12	2.62	1.68	3.64	2.09
NDN xx	10	0.16	0.08	0.15	0.07
NDN xxx	8	0.21	0.15	0.45	0.37

The measurement results from our study showed that N_2O-N emissions are less than 2% of TN load, (3% of TKN load). This is comparable with results obtained in the other WWTPs we measured and within commonly estimated emissions of N_2O-N mostly lay between 0.01-5% of TKN (Global Water Research Coalition report, 2011). Sun *et al.* (2013) measured N_2O emission from full scale SBR in municipal wastewater treatment plant and it was around 6.52% of the nitrogen load transformed to the emitted N_2O. Rodriguez-Caballero *et al.* (2013) reported that 0.8% of the removed nitrogen converted to N_2O.

Limitations

The chemical analysis of different nitrogen components in the influent and effluent was normally carried out twice per week. The measurement results of N_2O were daily average. The resolution of the emission measurements and nitrogen loads/removal do not match and used average nitrogen loads/removal rates induce an uncertainty in the total emission estimations. During the calculations, there was not enough nitrogen removal data available; therefore, there was few data on N_2O emission related with nitrogen removal rates.

The period of some operational mode was very short. For example, in AG-NIT mode, only one chemical analysis result of influent was available. Due to this reason, the nitrogen loads of the system were estimated based on the online ammonium sensor in the influent municipal wastewater. This would bring uncertainty in the results.

For the AG-NIT mode, the assumed emission ratio between PRZ and MRZ induces also an uncertainty in the total emission estimation.

Emission of N2O-N was about 1.7 % and 0.1 % of influent TN in the AG-NIT and the NDN modes respectively. This is in the normal range of emission from biological nitrogen removal systems.

4.1.2 **Tertiary treatment processes**

4.1.2.1 Disc filter (DF)

Average turbidity results monitored online at the inlet and effluent of the disc filter are shown in Table 4.10 for each of the operation periods. Results show that the DF was operated under similar influent conditions for each of the two test-periods. The analysis of the effluent turbidity showed that both the 10 and 18-µm meshes could reduce the turbidity of the feed water by 45%. The fractionation of the results for ranges of increasing feed turbidity showed that at the feed turbidity below 2 NTU, the effluent turbidity is as low as 0.9 NTU. As the feed turbidity increases, the effluent turbidity is increased and reaches 3 to 4 NTU for feed water of turbidity above 4 NTU.

These results suggest that the performance of the disc filter operated without polymer is directly related to the feed quality. Despite these variations in performance, the average effluent turbidity remained below 2 NTU, which matches the requirements for agriculture and GWR application. Maximal effluent turbidity was 6 and 10 NTU for the 10 and 18 µm meshes, which is superior to the maximal turbidity tolerated for the selected application. It is, however, likely that such target can be reached for both mesh filter if ionic polymer is added to the feed for flocculation.

Table 4.10. Turbidity monitoring for DF operation with mesh of 10 and 18-µm pore sizes, respectively.

	IN	OUT	*Diff %*	0 to 2 NTU	*Diff %*	2 to 5 NTU	*Diff %*	> 4 NTU	*Diff %*
at 10 µm mesh									
Average	3,9	2	*52*	0,6	*66*	1,7	*53*	2,8	*47*
St Dev	0,9	1		0,1		0,9		1,2	
Max	6,2	6		0,9		4,7		6,0	
Min	1,3	0		0,3		0,3		0,6	
at 18 µm mesh									
Average	4,19	2,24	*45*	0,9	*42*	1,9	*46*	4	*44*
St Dev	3,77	1,70		0,3		0,9		2	
Max	47,788	9,853		2,0		4,6		10	
Min	0,874	0,107		0,1		0,2		1	

Water quality parameters such as carbon-based parameters, solid content and bacterial content, were monitored based on the collection of 24-hour composite samples of the DF-feed and -effluent at a frequency of one to three times a week. Results indicate that both mesh sizes have similar filtration performances. The disc filter reduced the COD, BOD and TOC by 10, 30 and 10% respectively. Total dissolved solids and color were not affected by filtration. Total suspended solids and microbial content were reduced by up to 61%.

The performances of the filter demonstrate a good removal of suspended material and suggest that the addition of cationic polymer for the flocculation of dissolved organics and the destabilization of colloids could improve the removal of carbon-based contaminants.

In conclusion, the operation of the disk filter demonstrated the reliable performances of this conventional filtration technology. The effluent quality observed during operation met the process guarantee requested by the supplier. Increase of the performance in solid removal could be obtained by dosing a polymer upstream for the flocculation of particles. The automatism of the backwash and easy access to the filtration mesh to the operator are offering robustness, practicality and simplicity to the operation and maintenance. The disk filter technology can be recommended for reuse application requiring an effluent turbidity less than 2 NTU only (such as irrigation for agriculture) and where the operation and maintenance could be limiting factors.

The disk filter removed about 50 % of NTU and TSS with both 10 and 18 µm mesh. Operation of the filter was simple and robust, and the removal efficiency could probably be increased with upstream flocculation.

4.1.2.2 Ozonation (O3)

The comprehensive pilot studies conducted intended to better define the role of ozone in tertiary treatment processes combined with and without upstream and downstream processes. Besides the classic water parameters such as COD, BOD, TOC, Color, UV Transmission, Pathogens, Total Suspended Solids (TSS), advanced trace analytic methods have been used to observe the reduction rate of micro-pollutants such as Carbamazepine, Sulfamethoxazole, Ibuprofen, Metoprolol, Benzotriazole, etc. by ozonation. The oxidation by-products, such as Bromate, Assimilable Organic Carbon (AOC), have been observed over testing period as well.

DF-O3-BAF: Testing has been intensively conducted over the period from Jan 21, 2013 to May 03 2013. The ozone dosages ranged between 4-10 mg/L. The hydraulic retention time (HRT) was about 23-27 minutes. In this treatment line, the Disc Filter (DF) served as pre-filtration for ozone. The different ozone dosages have been examined combined with different downstream media filter EBCTs (5-20 minutes).

The water quality from ICEAS was quite stable with an average COD concentration about 41 mg/L, an average TOC concentration about 12 mg/L and an average BOD concentration about 6 mg/L over this period. The Disc Filter (DF) did not change most of the water parameters except the efficient removal for TSS. The TSS has been efficiently removed from 7.9 mg/L to 2.4 mg/L by DF. The average bromide level in the raw water was about 58 µg/L and no bromate formation has been detected (< 6µg/L) in the ozonated water. The optimized ozone dosage was in the range 6-8 mg/L (ozone/TOC ratio 0.5-0.7 mg/mg) in terms of COD reduction, BOD increase and UV-Transmission improvement. The COD could be decreased optimally from 38.9 mg/L down to 32.4 mg/L, the BOD could be increased optimally from 3.9 mg/L up to 5.4 mg/L, the UV-Transmission could be significantly improved from 51.6%/cm to 63.2%/cm at 254nm. At the same ozone dosage, the color can be removed more than 70%. The disinfection credits ranged between 1-2 LOG in terms of total coliform reduction and faecal coliform reduction.

O3-BAF: Testing has been intensively conducted over the period from Feb 28, 2014 to April 11, 2014. The ozone dosages ranged between 6-18 mg/L (Ozone/TOC Ratio 0.6-1.2 mg/mg). The hydraulic retention time (HRT) was about 15 minutes. In this treatment line, the Disc Filter (DF) was out of operation compared to treatment line 1. Both downstream media filters had fixed EBCTs at 15 minutes.

The water quality from ICEAS was quite stable with an average COD concentration about 41 mg/L and an average BOD concentration about 7 mg/L over this period. As mentioned, the DF was out of operation, the average TSS in the ozone inlet was 7.9 mg/L averagely. It is interestingly noticed the TSS has been reduced from 7.9 mg/L to 4.0 mg/L after ozonation. This phenomenon has not been observed in the treatment line 1 with low TSS concentration in the ozone inlet. In this treatment line, the COD concentration varied between 40.7 ±6.84 mg/L in the ozone inlet and 31.7±6.18 mg/L in the ozone outlet. The higher COD removal indicated there are more easily oxidized organics portion remaining in the water without DF as pre-filter. There was no clear plateau to show an optimized ozone dosage in terms of COD reduction, BOD increase and UV-Transmission improvement by ozone alone. However, in the combined evaluation with BAFs together, the optimized ozone dosage has been found in the ozone/TOC ratio of 0.8 mg/mg in terms of COD reduction, which achieved 50% by "ozone + BAF" together. Meanwhile, the AOC concentration has been found to increase with an increasing ozone dosage. That would lead to elevated food source in the filter.

pUF-O3-BAF: Testing has been intensively conducted over the period from April 15, 2014 to May 8, 2014. The ozone dosages ranged between 5-11 mg/L (Ozone/TOC Ratio 0.6-1.0 mg/mg). The hydraulic retention time (HRT) was about 15 minutes. In this treatment line, pUF served as pre-filtration for ozone. Both downstream media filters had fixed EBCTs at 15 minutes.

The water quality from pUF was quite stable with an average COD concentration about 30.4 mg/L and an average BOD concentration about 2 mg/L over this period. The TSS was at a very low level about 0.4 mg/L. In this treatment line, the COD concentration varied between 30.4±3 mg/L in the ozone inlet and 27.7±3.7 mg/L in the ozone outlet. The low COD removal indicated there are more "hard" oxidized organics portion remaining in the water after pUF as pre-filter. The pUF was very effective to remove Coliforms from the water, no Coliforms have been detected in the pUF permeate. There was no clear plateau to show an optimized ozone dosage in terms of COD reduction, BOD increase and UV-Transmission improvement by ozone alone. However, in the combined evaluation with BAFs together, the optimized ozone dosage has been found in the ozone/TOC ratio of 0.8 mg/mg in terms of COD reduction, which achieved 30% by "ozone + BAF" together. Meanwhile, the AOC concentration has been found to be increased by ozonation but no clear trend with an increasing ozone dosage. Anyhow, that would lead to elevated food source in the filter.

MBR-O3-BAF: Testing has been intensively conducted over the period from Jan 22, 2014 to Jan 31, 2014. The ozone dosages ranged between 3-5 mg/L (Ozone/TOC Ratio 0.3-0.6 mg/mg). In this treatment line, MBR water served as feed water for ozone. Both downstream media filters had fixed EBCTs at 15 minutes.

This MBR treatment line testing was an additional test trial for a short period compared to the original project schedule. The water quality from MBR was quite stable with an average COD concentration about 21 mg/L and an average BOD concentration about only 1 mg/L over this period. The TSS was at a very low level about 0.4 mg/L. In this treatment line, the COD concentration varied between 21±3 mg/L in the ozone inlet and 21.8±2.75 mg/L in ozone outlet. That indicated the remaining COD in the MBR effluent is hard COD, it is more resistant to ozone than the water from ICEAS effluent as secondary treatment. No AOC testing has been done for this treatment line, because the COD and BOD did not have obvious change, we may deduce that the increase of AOC in this treatment line may be lower than the other treatment lines.

As conclusions, the performed ozonation tests gave a preliminary evaluation about the water quality difference among different upfront treatment processes for Ozone (MBR, SBR-DF, SBR-pUF, and only SBR). Selected ozone COD ratios (or ozone/TOC ratios) were in common application ranges; however, it is important that optimum ozone dosage need to be evaluated with the filter data and micro-pollutants data together. Further investigation should be performed on the water characterization difference between membrane water and biological treated water. For example, which upfront treatment line benefits the “O3-BAF” application economically? Which upfront treatment line does not economically benefit the “O3-BAF” application, for example resulting in high ozone dosage demand but slow biological growth in filter?

Ozone doses between 6 and 8 mg/L, or 0.5-0.8 mg ozone/mg TOC, gave 10-20 % decrease in COD, a small increase in BOD and substantial increase in UV transmission. Eventually formed bromate was below detection limit. There was also a disinfection effect. The total treatment effect was dependent on filter pre-treatment and the following BAF.

4.1.2.3 Disinfection: UV

All the UV Collimated Beam Device (CBD) tests within the project have been performed on different water qualities from various treatment lines of combined technologies as described in this report. The table below gives an overview about the water qualities when CBD testing was conducted.

Table 4.11. Overview of Water Quality Parameters during UV-tests.

		Water Parameter								
	Upstream	Turb.	SS	UVT 254nm	COD	Fe Tot	Fe Sol.	pH	T	Alk.
Date	process	NTU	mg/L	%/cm	mg/L	mg/L	mg/L	-	°C	mg/L
30.07.2012	ICEAS+RGSF	2.08	1.2	61.9	28	-	-	-	-	-
10.10.2012	ICEAS+DF+ 2ppmO3+GAC	0.46	-	82.0*	20	0.06	0.04	7.10	18.9	90
16.10.2012	ICEAS+DF+ 2ppmO3+RGSF	1.27	-	66.9	27	0.15	-	7.36	19.7	106
06.02.2013	ICEAS+DF+ 8ppmO3+GAC	0.84	0.4	71.3	14	0.05	0.04	7.35	17.0	82
21.02.2013	ICEAS+DF+ 8ppmO3+GAC	0.09	0.0	79.9	24	0.14	0.05	7.17	16.8	81
02.05.2013	ICEAS+DF+ 10ppmO3+GAC	0.25	0.2	79.0	19	0.06	0.06	7.14	19.0	101
29.05.2013	ICEAS+RGSF (no Fe)	-	0.6	57.1	26	0.07	0.05	7.43	-	100
26.06.2013	ICEAS+RGSF (with coag)	-	1.6	61.6	16	0.15	0.09	7.11	22.4	82
24.07.2013	ICEAS+DF	-	2.6	53.3	38	0.15	0.10	7.20	-	95
14.08.2013	ICEAS+RGSF (with coag)	0.41	1.2	61.5	32	0.14	0.13	6.90	21.0	66
11.09.2013	ICEAS+DF+pUF	0.99	0.4	54.4	36	0.35	0.29	7.39	22.5	67
09.10.2013	ICEAS+DF+sUF	0.47	2.4	57.5	31	1.10	0.28	7.31	23.3	-
31.10.2013	ICEAS	0.86	6.0	53.8	41	0.11	0.09	7.75	11.0	99
11.12.2013	ICEAS+DF+GAC	0.74	0.4	66.2	22	0.10	0.09	7.18	17.6	113
18.12.2013	ICEAS+DF+GAC	0.59	0.8	71.3	5	0.07	0.07	7.15	17.0	100
08.01.2014	ICEAS+DF+GAC	0.91	2.6	66.3	30	0.15	0.07	7.18	19.0	104
15.01.2014	ICEAS+DF+GAC	0.46	0.2	64.6	28	0.10	0.08	7.08	15.5	104
09.04.2014	ICEAS+ 7ppmO3+GAC	0.57	0.4	74.8	24	0.16	0.04	6.98	15.6	94

*UV-T is quite high. It is caused by wrong measurement or new GAC media

The best UV Transmission has been achieved by the treatment line with ozonation, for example at 8 ppm ozone followed by GAC. As the water qualities varied from case to case, the initial total coliform concentrations in the to-be-radiated water also varied (Table 4.12). For example by using ICEAS effluent on Oct. 31, 2013, the total coliform bacteria with 250 000 CFU/100 mL was at the highest level. At the same time, the UV Transmission with 53.8 %/cm at 254nm was almost the lowest one, with one exception on Jul. 24, 2013 for the combination ICEAS+DF. Therefore, for ICEAS-effluent or ICEAS+DF-effluent, a higher UV dose is required to achieve certain coliform reduction than the other treatment lines combined with more advanced treatment technologies.

Table 4.12. Overview of CBD Test Data – Total and faecal Coliform Bacteria Reduction.

		Total (1st row) and Faecal Coliform (2nd row) in CFU/100mL at UV Dose (mJ/cm²)								
Date	**Upstream process**	**0 (Inlet)**	**5**	**10**	**15**	**20**	**25**	**30**	**35**	**40**
30.07.2012	ICEAS+RGSF	327 778	-	957.7	-	5	1	-	-	1.0
		4758	-	105	-	1	1	-	-	1
10.10.2012	ICEAS+DF+	27 400	3267	70.7	-	1.0	-	1.0	-	1.0
	2ppmO3+GAC	566.7	229	3.7	-	1	-	1	-	1
16.10.2012	ICEAS+DF+	12 847	3193	444.7	-	1.0	-	1.0	-	1.0
	2ppmO3+RGSF	2080	378	52	-	1	-	1	-	1
06.02.2013	ICEAS+DF+	10 403	1910	170	-	0.7	-	0.7	-	0.5
	8ppmO3+GAC	1592	156	21.7	-	0.5	-	0.5	-	0.5
21.02.2013	ICEAS+DF+	9.7	2.2	0.5	0.5	0.5	0.5	-	-	-
	8ppmO3+GAC	0.7	0.5	0.5	0.5	0.5	0.5	-	-	-
02.05.2013	ICEAS+DF+	756.7	57.7	14.0	0.5	0.5	-	-	-	-
	10ppmO3+GAC	3.7	0.5	0.5	0.5	0.5	-	-	-	-
29.05.2013	ICEAS+RGSF	2420	2130	1986	318.7	43.7	46.0	-	-	-
	(no Fe)	2420	610.3	155	34	3.3	0.7	-	-	-
26.06.2013	ICEAS+RGSF	9900	2950	320.3	48.3	6.5	6.3	-	-	-
	(with coag)	2005	360.7	61	6.3	1	1	-	-	-
24.07.2013	ICEAS+DF	241 960	142 213	2420	1538	154.7	74.5	-	-	-
		19 501	9026	1133	75	7	5.7	-	-	-
14.08.2013	ICEAS+RGSF	51 987	13 003	1253	123	8.3	4.7	-	-	-
	(with coag)	6436	1272	127.3	5	0.7	0.8	-	-	-
11.09.2013	ICEAS+DF+pUF	ND	ND	ND	ND	ND	ND	ND	ND	ND
		ND	ND	ND	ND	ND	ND	ND	ND	ND
09.10.2013	ICEAS+DF+sUF	3.3	0.7	0.7	0.5	0.5	0.5	-	-	-
		0.5	0.5	0.5	0.5	0.5	0.5	-	-	-
31.10.2013	ICEAS	248 433	-	5539	-	340.3	-	149.7	106.7	41.3
		37 983	-	342.3	-	26.7	-	10.7	5.7	6
11.12.2013	ICEAS+DF+GAC	21 910	-	420	-	9.3	5	5.3	-	1.8
		2685	-	27.3	-	1.2	0.5	0.7	-	0.5
18.12.2013	ICEAS+DF+GAC	13 380	-	6603	-	5	0.8	0.5	-	0.5
		2142	-	23.7	-	0.5	0.5	0.5	-	0.5
08.01.2014	ICEAS+DF+GAC	142 213	-	3203	-	19	1.8	0.7	-	0.8
		19 863	-	216	-	1.3	0.5	0.5	-	0.5
15.01.2014	ICEAS+DF+GAC	11 617	-	326.3	-	3	0.8	1.3	-	0.7
		2158	-	13.5	-	0.5	0.5	0.5	-	0.7
09.04.2014	ICEAS+	630	-	50	0.5	0.5	-	0.5	-	0.5
	7ppmO3+GAC	10	-	6.7	0.5	0.5	-	0.5	-	0.5

Note: ND is non-detected, - is used for non-applied dose

Similar to total coliform, the initial faecal coliform concentrations in the to-be-radiated water were also varied from case to case. For example by using ICEAS effluent on Oct. 31, 2013, the faecal coliform bacteria with 38 000 CFU/100 mL was at the highest level.

In general, the total coliform concentrations in the to-be-radiated water are approximately 1-LOG higher than the faecal coliform concentrations.

The detailed evaluation in the following section is based on the treatment scenarios that we have investigated.

Treatment train SBR(NDN) + pUF + UV

As observed in Table 4.12, no total coliform and faecal coliform have been detected in the pUF permeat and hence no UV test data is available for this treatment line. Despite this, it is recommended to have an UV process in full-scale applications as the UV-process could serve as backup process in case of pUF maintenance or shutdown.

It is recommended to layout the full scale UV-reactor with following preconditions:

- 2 LOG reduction of total coliform as treatment goal
- The design UVT can be ≥ 60% (The UV-T was 54,4 % during the pilot testing. However, it is very possibly caused by wrong measurement)
- The design SS can be ≤ 2 mg/L

Treatment train SBR(NDN) + sUF + UV

Compared to SBR(NDN) + RGSF, SBR(NDN) + sUF can remove approximately 5-LOG total coliform. In real application, the sUF supplier gives normally 4-LOG reduction as process guarantee. Thus, it is recommended to have an UV process in full scale STP with the same design parameters as above.

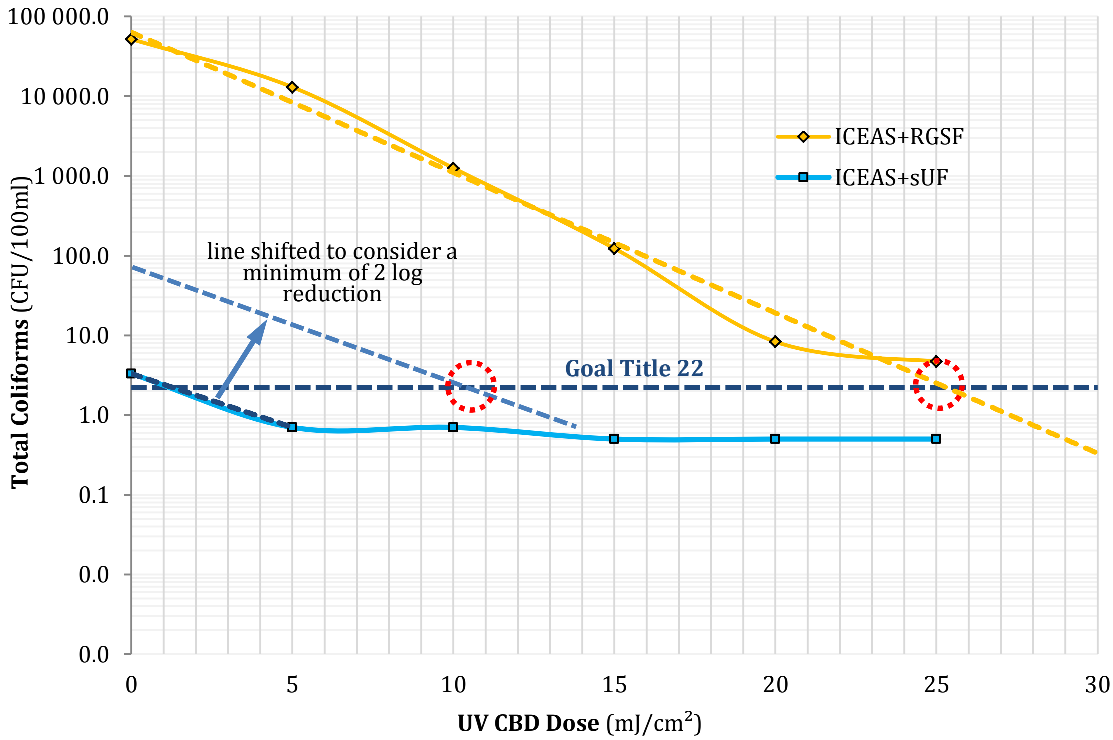

Figure 4.5. Comparison of Treatment Lines with/without sUF.

Treatment train SBR + RGSF + UV

Comparing the performed tests shown in Figure 4.6, the total coliform concentration ranged between 104-105 CFU/100mL regardless with/without coagulation in front of the sand filtration. To achieve the targeted reduction of 2.2 CFU/100mL, the CBD dose has to be at 25 mJ/cm². It is recommended to layout the full-scale UV reactor with following preconditions:

- 5-LOG reduction of total coliform bacteria as treatment goal
- The design UV-T can be ≥ 55%
- The design SS can be 2-5 mg/L

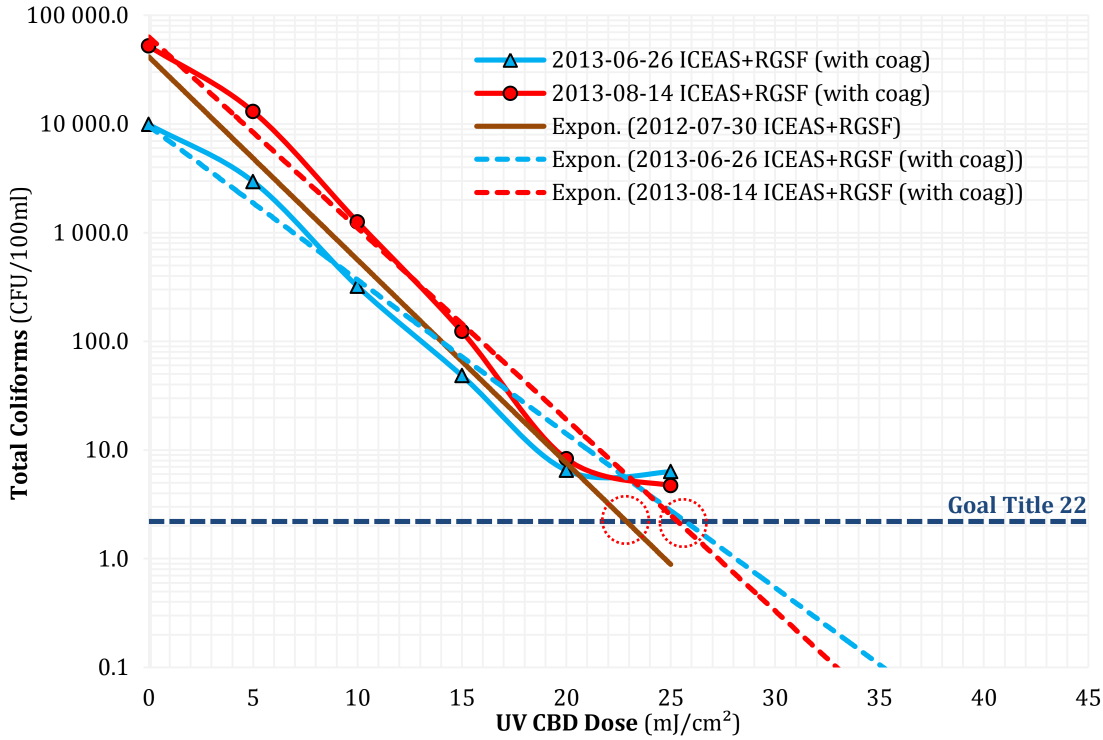

Figure 4.6. Comparison of Treatment Lines with & without coagulation.

Treatment train: SBR(NIT) + DF + UV

The figure below shows that if DF is used, the required CBD Dose is significantly lower than without DF in order to achieve the goal of total coliform 2.2 CFU/100mL. The calculated CBD dose is 32 mJ/cm² with and 52 mJ/cm² without DF, respectively. The Suspended Solid (SS) concentration in the water is 2.6 mg/L and 6 mg/L, respectively. It is not preferred to use UV directly after ICEAS to achieve the treatment goal, as the UV energy consumption would be rather high. On the other hand, it be would suitable to use UV directly after the secondary treatment to achieve less stringent treatment goals as for example Escherichia coliform of 500 CFU/100mL according to European Bathing Water Directive 2006/7/EC.

It is recommended to layout the full-scale UV reactor with following preconditions:

- 6-LOG reduction of total coliform as treatment goal

- The design UVT can be ≥ 50%, the design SS can be 2-5 mg/L with DF
- The design UVT can be ≥ 50%, the design SS can be 5-10 mg/L without DF (*Note: That's not the preferred selection if the treatment goal is strict.)

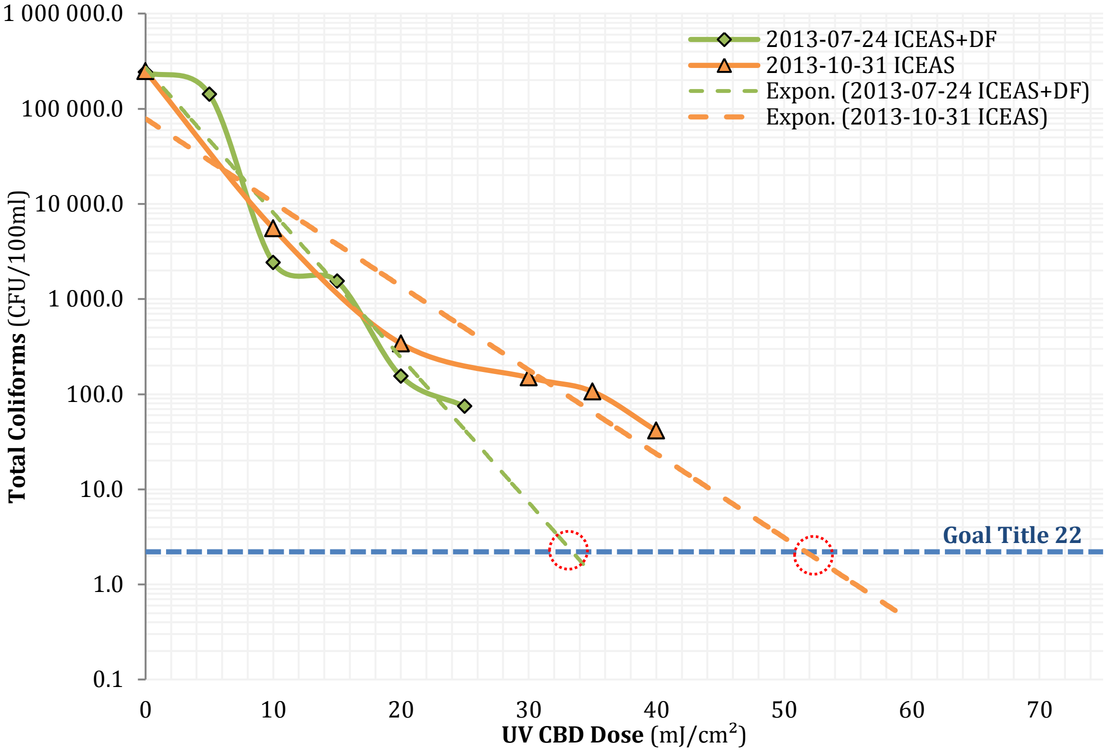

Figure 4.7. Comparison of Treatment Lines with & without of DF.

Treatment train: SBR(NDN) + DF + O3 + BAF (GAC) + UV

Figure 4.8 shows the total coliform concentration variation after the treatment of Ozone + BAF. It varied between 10-104 CFU/100mL and no stable data have been obtained. To have a close look at the water quality data in Table 4.12, actually the overall water quality on Feb. 06, 2013 was the best one among the three tests. For example the COD value was only 14 mg/L on that day, but the total coliform is as high as 10 000 CFU/100mL. No reasonable explanation could be found so far.

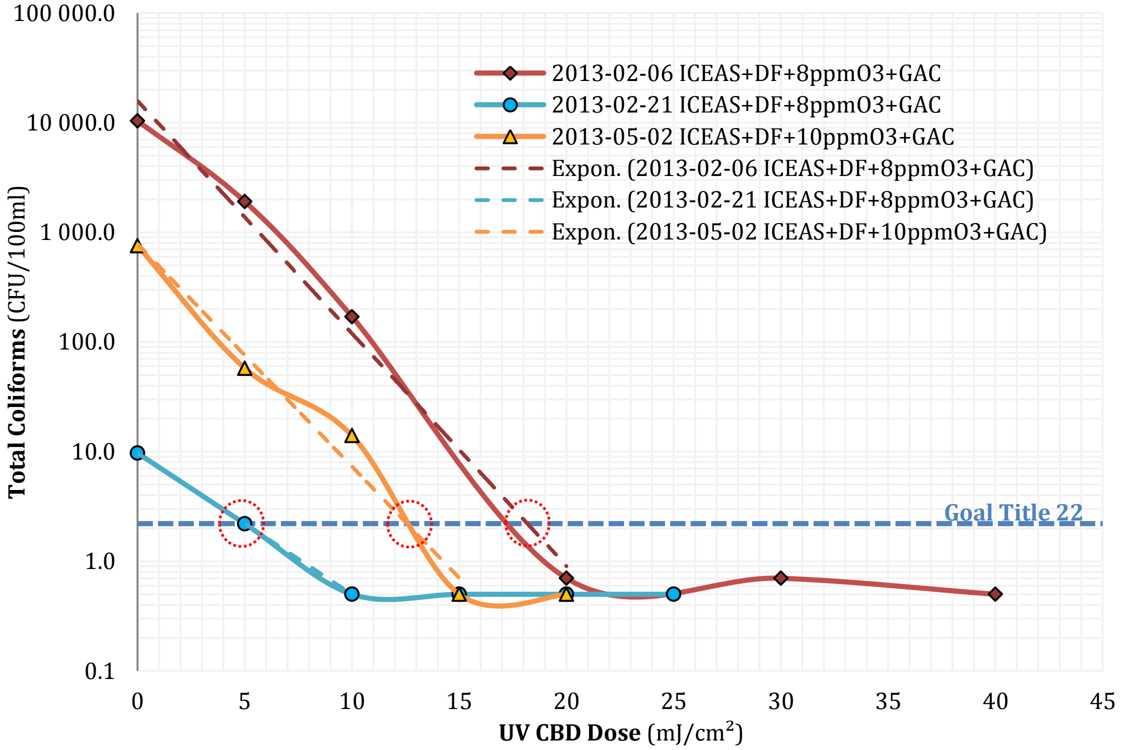

Figure 4.8. Comparison of Treatment Lines with "DF + Ozone + BAF".

Excluding one of the best curves (blue one) on Feb. 21, 2013, the calculated CBD dose ranged between 13-18 mJ/cm² to achieve the targeted effluent concentration. It is recommended to layout the full-scale UV reactor with following preconditions:

3-LOG reduction of total coliform as treatment goal
- The design UV-T can be ≥ 70% with O3 + BAF,
- The design SS can be <1 mg/L with O3 + BAF

Treatment train: SBR(NDN) + O3 + BAF + UV The two lines with and without DF were studied for disinfection by UV. CBD tests showed that addition of the DF had no impact on disinfection (data not shown). Therefore, the same design parameters as for the previous treatment line are recommended.

Additional Treatment train Evaluation: SBR(NDN) + DF + GAC + UV
In this treatment line, there was no disinfection process like ozonation in front of the GAC. The total coliform concentration in the GAC effluent varied between 10^4-10^5 CFU/100mL. To achieve 2.2 CFU/100mL, the calculated CBD dose would be 22-26 mJ/cm². Thus, it is recommended to layout the full-scale UV reactor with following preconditions:

- 5-LOG RED of total coliform as treatment goal
- The design UV-T can be ≥ 60%
- The design SS can be < 2 mg/L

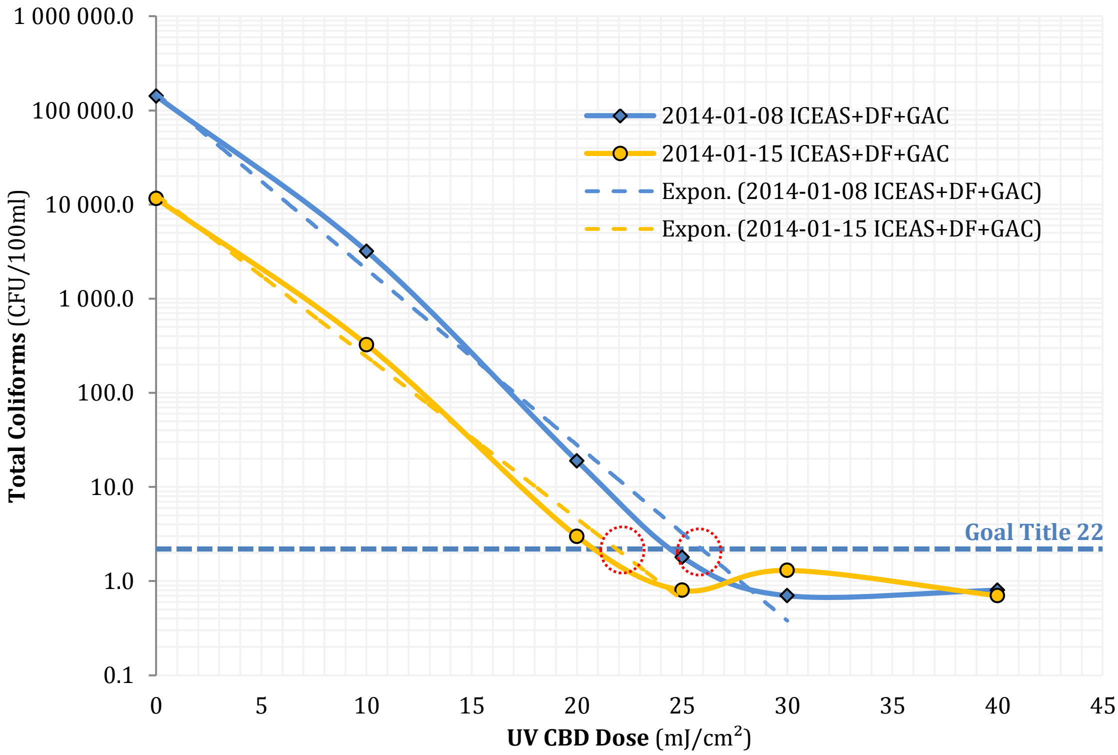

Figure 4.9. Comparison of Treatment Line with/without DF in front of GAC.

Required dose of UV to reach the goal varied with pre-treatment. UF should remove all bacteria, but a low dose of UV (10 mJ/cm²) is recommended for safety. Lines with RGSF demanded about 25 mJ/cm², while for lines without filter or just DF needed higher UV intensity. Ozone steps decreased the demand below 25 mJ/cm², due to both some disinfection and increased UV transmission. Design parameters for different cases are suggested.

4.1.2.4 Sand filter (RGSF)

Figure 4.10 shows the operational periods of the filter. Three scenarios were carried out; constant flow, diurnal flow, and storm simulation without addition of chemical for coagulation. The constant flow was maintained at 7.3-7.5 m/h; the diurnal swing was maintained between 5 to 9 m/h; and during storm simulation, the hydraulic loading was maintained at 22-23 m/h (9 gpm/ft²).

The influent turbidity was from 2 to 6 NTU most of the time and occasionally the influent turbidity was spiked due to the stirring of the slow mixing tank. The effluent turbidity was averaged less than 2 NTU throughout this phase. When the column pressures reached 2.75 m backwash was initiating.

There were eight cycles during constant and diurnal flows. Each cycle was defined as the filtration period from the finishing point of a backwash to the beginning of the next backwash. The filter runtimes were all maintained between 24 to 48 hours. Generally, the minimal acceptable runtime was 24 hours at the design flow.

The average turbidity in each cycle was around 1 NTU and this was confirmed by the composite sample measurement. This indicates that California Title 22 criteria can be satisfied without the need of coagulation using this media configuration.

During the 24-hour storm, the filter experienced three complete cycles as shown in Figure 4.10. The effluent turbidity was slightly higher than that during the other two events but still much lower than 2 NTU. The required filtration rate by the California Title 22 of < 5 gpm/ft² (12.2 m/h) was satisfied during storm event even if the backwash rate was higher. Filter runtime was around 6 hours. Actually, the storm event was also a constant flow event.

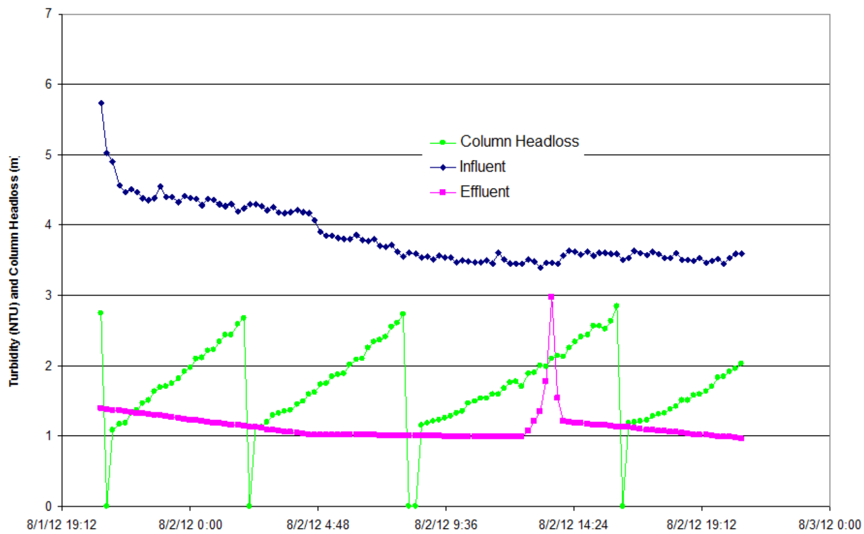

Figure 4.10. Turbidity profiles and filter column pressures during the 24-hour storm.

Table 4.13 summarizes Xylem process expectation of dual media filters after a secondary biological treatment system. In the last two columns of the table, composite samples results during the test are listed for comparison. Only few parameters for some dates were chosen for analysis. However, results show that BOD, COD, and TSS were within expectation (design criteria). The effluent TSS was un-detected. BOD and COD were only reduced slightly after filtration. As expected, nitrogen species remained unchanged after filtration. Total phosphorus was higher than design criteria due to the absence of coagulation. Additionally, total coliform counts in the effluent were also higher than that in the influent.

Table 4.13. Lab analysis results as compared with design expectation of dual media filters after a secondary biological treatment system.

Parameters	Unit	Max Feed Criteria	Design Target	Pilot results Influent Max/Average	Pilot results Effluent Max/Average
Organics and Inorganics					
BOD5	mg/L	10	5	5/3.89	5/2.78
COD	mg/L	60	50	22/20	34/21.57
TOC	mg/L	20			
Total nitrogen	mg/L	10	-		
Ammonia nitrogen	mg/L	1	-	1.51/0.51	
Nitrate nitrogen	mg/L	5	-	17/10.6	
Organic nitrogen	mg/L	4	-		
Total phosphorus	mg/L	0.5	0.2	1.39/1.39	1.35/1.36
Dissolved oxygen	mg/L	4		7.9/7.3	7.9/7.9
pH		6-9	6-9	7.1/6.8	
Solids and turbidity[1]					
TSS	mg/L	15	5	3.7/2.45	UD/UD[2]
Average turbidity	NTU	10	5	3.4	
Maximal turbidity	NTU	20		4.15	
Microbiology					
Total coliform	/100 mL			6300	17100
Fecal coliform	/100 mL			6300	17100

[1] The results in this period were the results of composite samples except turbidity numbers were online daily averages.
[2] UD – undetectable

The RGSF reached the target for NTU reduction both at normal (7.5 m/h) and high (23 m/h) hydraulic load. For high load backwash was needed every 6 hours, while at normal load every 24 hours was enough. COD, BOD, TN and TP were not significantly reduced. Bacterial counts increased over the filter.

4.1.2.5 Pressurized membrane filtration (pUF)

The pUF was fitted with two membrane elements from X-Flow, type AQF each with 6.2m² of membrane area. The feed to the plant came from the EQ tank. Provision was made for a 100µm screen to be in line. During most of the first trial, this was inadvertently bypassed. When it was used, it was found to blind too rapidly to be used. Subsequent trials used the disk filter fitted with the same screen size. The general operating sequence was as advised by the membrane supplier: Filtration was carried out in a dead end mode. At the end of a defined time, the membrane was backwashed with permeate, while a feed flow was maintained. Simultaneously, the waste valve was opened allowing the membrane to be flushed of retentate. This cycle was repeated for a defined number of times, after which two chemically enhanced backwashes (CEB) were applied. The first was with alkaline sodium hypochlorite (Sodium Hydroxide: 525 mg/L; Sodium hypochlorite: 200 mg/L) to remove organics, and the second was with hydrochloric acid (450 mg/L) to remove scale.

The plant performed well from a mechanical point of view, and the overall performance from a permeate quality point of view was as expected of a membrane. The permeability was clearly affected by large particles blocking the entry to the fibre lumens. It is unclear if this was an issue from the start of the tests or if this was only problematic in the second half of the test period, as there appears to have been an increase of large material in the influent to the SBR. The current filter was unable to cope with the solids loading and became blocked within 12 hours. This issue will need to be addressed before the next campaign.

Figure 4.11 shows a plot of permeability over the period of the trials. This has not been adjusted for temperature as this varied little with a value of 20.7°C ± 1 °C over the duration of the trails. Extreme data points have been removed from the graph to aid clarity. These points occur when data is recorded during transition between backwash and filtration, and can result in unrealistically high, or low apparent permeabilities.

Reading the graph from left to right shows an initial high permeability followed by a drop and then a recovery. This is normal behaviour for new membranes. The curve then drops to a steady saw tooth pattern for about four days. (10 August to 14 August). The saw tooth is indicative of reversible fouling occurring, which is then removed by the CEB. There then follows a period where the permeability showed a slight increase for a couple of days before declining around the 17th to 19th August. The failure of the CEB to recover the permeability indicates a potential change in the nature of the foulant. The following period of instability is a reflection of a number of shutdowns caused by high feed pressure. On August 29, a number of cleanings were carried out, without improving the performance, leading to the conclusion that the face of the elements could be blocked. This was confirmed by removing the elements form the rig for visual inspection. The waste material was removed and the elements inverted. The pre filter was also brought on line at this time. This resulted in an immediate increase in performance.

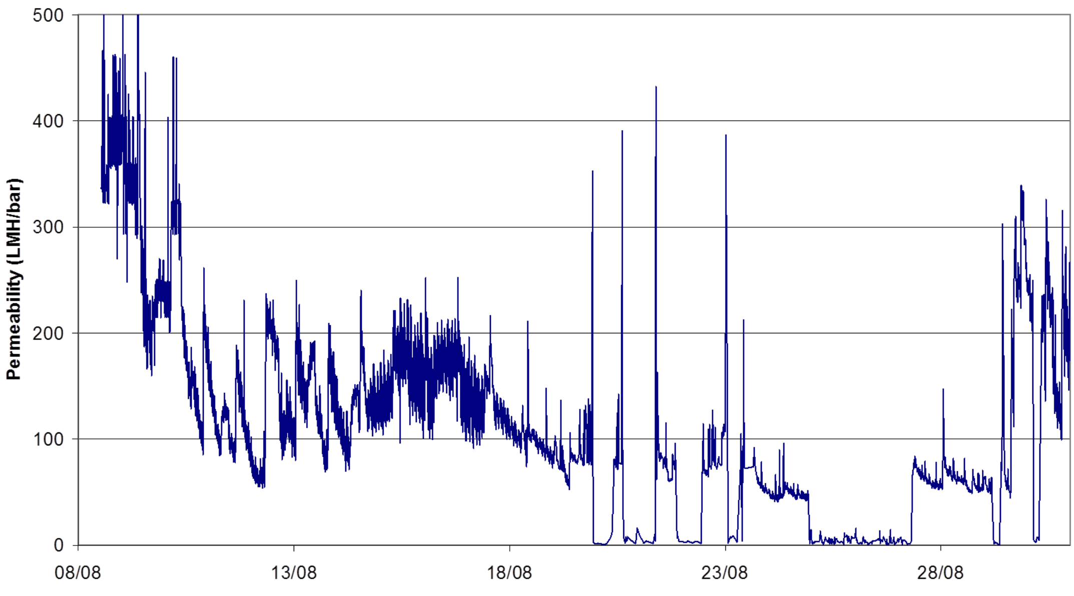

Figure 4.11. Permeability over the period of the trials.

It is clear that under nominally similar conditions of flow and influent quality the plant exhibited different performance, sometimes stable, and sometimes showing rising TMP. Under these conditions, the critical flux is between 32 LMH, and 52 LMH, which match the membrane supplier requirements. The parameter of permeability can be seen to have limitations, as it is clearly a function of flow rate. However, it remains a useful tool to get an overall impression of performance.

In conclusions:

1. The TMP was recovered by CEB providing there was not excessive material deposited on the face of the modules.
2. Excessive material deposited on the face of the membrane element could not be removed by the backwash or CEB. Thus, the pre-filter is essential, but the ability of the current device to cope requires attention.
3. Turbidity was, as expected, removed well by the membranes at 98.8% average removal. Permeate turbidity is maintained at 0.1 NTU
4. COD removal by the pUF membrane depends on the feed content in soluble COD.
5. The pUF membrane is optimal for solid removal as the permeate TSS concentration is permanently at the detection limit of method.
6. There was little attenuation of color.
7. The pUF membrane offer a disinfection credit of 4 to 6 logs in terms of total coliform removal.

4.1.2.6 Submerged membrane filtration (sUF)

The membrane area of the sUF was 34 m^2, the minimum membrane area available from GE Zenon. This gave a theoretical capacity of about 1100 L/h. This is more than the line design flow of 600L/h average, so the membranes were under-loaded. Additionally, because of the minimum clearances around the membranes in the tank, the ratio between the membrane tank volume and the membrane area is higher in the pilot than it would be in a full-scale plant. This means that the time to reach a recovery of 90% (i.e. time required for the filtration of 9 times the membrane tank volume) is higher in the pilot than in a full-scale plant. Accordingly, all results have to be interpreted with this in mind.

No coagulant was dosed during these trials.

The plant was started up with an initial flux of about 9 lmh, giving a flow of 300 L/H. The graphs show a steep rise in TMP from the start of operation, with the matching decline in permeability. As the TMP reached a maximum limited by the plant, the feed flow was reduced as the permeability declined. This is consistent with fouling of the membranes, with limited recovery from the backwash.

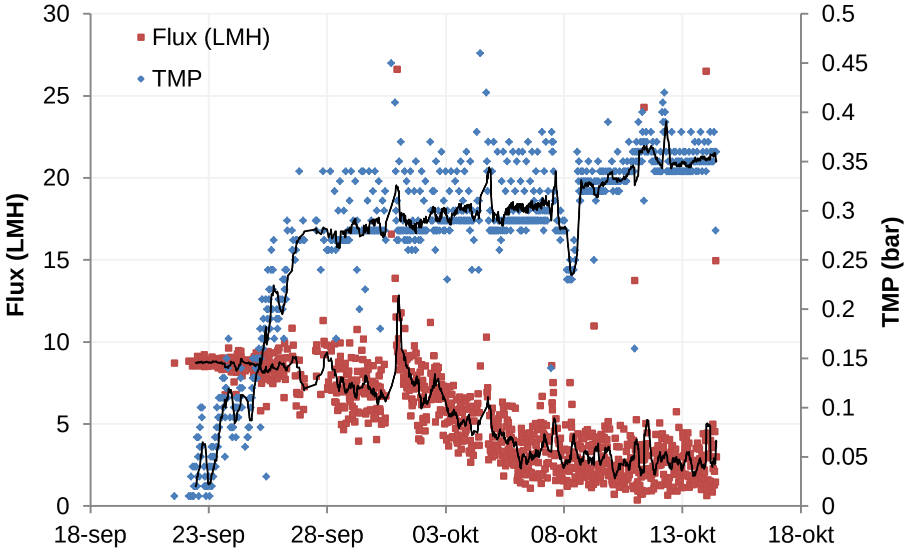

Figure 4.12. Flux and TMP for the sUF-operation.

In summary, the membranes appeared to foul badly and they were not recovered by cleaning. The separation performance, however, remained very good. The submerged ultrafiltration membranes demonstrated an ability to consistently remove particulate material, with all four tests showing an average TSS in the permeate of 1mg/L or less. The averages of the influent TSS ranged from 8 to 60 mg/L.

Performance on Total coliform removal was general log four or better. The few data that were less than this were attributed to permeate side contamination from flush water used to charge the permeate tank for operational reasons. This conclusion was supported by the lack of any indication of membrane integrity failure in any other parameters.

The COD removal was one of the most variable parameters, with average removals of 30%to 90% on the pUF and the sUF. Both of these runs suffered the most fouling and the feed UVT for the sUF was as low as 33%/cm. These results indicate the high dependence of the COD removal by UF on the feed content in dissolved COD. This is worthy of further review, especially the mode of operation of the ICEAS at that time.

When operated at stable conditions, the average permeability of the pressurised membranes is about 200 lmh/bar. The rapid fouling of the sUF may be a result of the compromises that had to be made on plant design and operation protocol due to the minimum membrane area available. However, it should also be noted that the plant had a more severe challenge than the pUF. Indicators are the high TSS, and the low UVT.

In conclusion, the polymeric membranes are capable to achieving a stable permeate quality, suitable for use with reuse applications. Fouling, as in all membrane applications, needs to be considered and appropriate application of coagulant and prescreening is necessary to ensure the performance of the membranes.

4.1.3 Removal of Micropollutants

Six substances from the list provided in Table 3.2 were not detected once during the project because all concentrations were below their specific detection limits. These substances were 17-ß-estradiol, 17α-ethynylestradiol, mecoprop, risperidone, atrazine and Bisphenol A. In different studies (Abegglen and Siegrist, 2012; Arge 2013; Schaar and Kreusinger, 2011; Ternes and Joos, 2006) these substances were detected during the treatment of municipal wastewater. Based on literature, substances characteristics and the process behavior in this project, following conclusions can be provided:

- Both hormones (17-ß-estradiol and 17α-Ethynylestradiol) should be completely reduced by ICEAS (approx. 80-90%) and ozone (approx. 70-100%)
- Risperidone should be completely removed by the secondary treatment
- Mecoprop should be removed by ozone by 70-80%
- Atrazine should be removed by ozone by 20-45%
- Bisphenol A should be reduced by the secondary treatment (approx. 80-90%) and ozone (more than 70%)

Different treatment processes have different removal efficiencies. Based on the molecular structure and the chemical or physical behavior of single substances the removal rate during different processes is important. The removal efficiency of single substances by different treatment trains is shown in the following figure. The figure includes average removal rates for the following treatment trains and selected substances:

- SBR (NDN) + DF + ozone + BAF 1 (anthracite)
- SBR (NDN) + DF + ozone + BAF 2 (GAC)
- SBR (NDN) + pUF + Ozone
- SBR (NDN) + RGSF + disinfection
- SBR (NDN) + DF
- SBR (NDN) + pUF
- SBR (NDN) + sUF

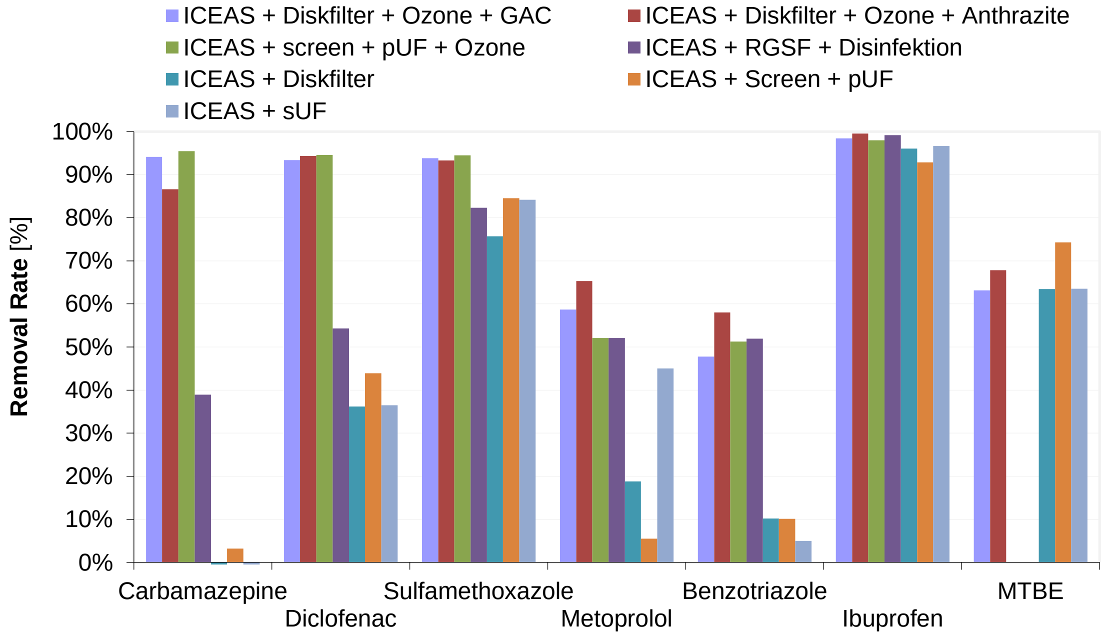

Figure 4.13. Removal efficiency (given as average values) for different treatment trains.

The figure shows that for most of the targeted substances ozonation is required to achieve removal rates > 80 %. Without oxidation or adsorption, removal rates are < 50 %. Substances like Ibuprofen or MTBE show high removal rates without any influence by the selected technologies. This is mainly caused by removal processes in the secondary treatment process, i.e. Ibuprofen is known as biodegradable.

Of the different treatment-trains that were tested only results for systems with oxidation or adsorption are considered in the following, as described target values otherwise are not achieved. During the period from December 2013 to May 2014, the following processes were investigated:

- BAF (anthracite) / BAF (GAC)
- Ozone + BAF (anthracite) / BAF (GAC)
- Ozone + BAF (anthracite) / BAF (GAC)

The removal efficiencies were analyzed for some selected substances. The bars in each figure show the average values of the specific concentration and the indicators show the maximum and minimum concentration. Indicated target values are based on Table 2.3.

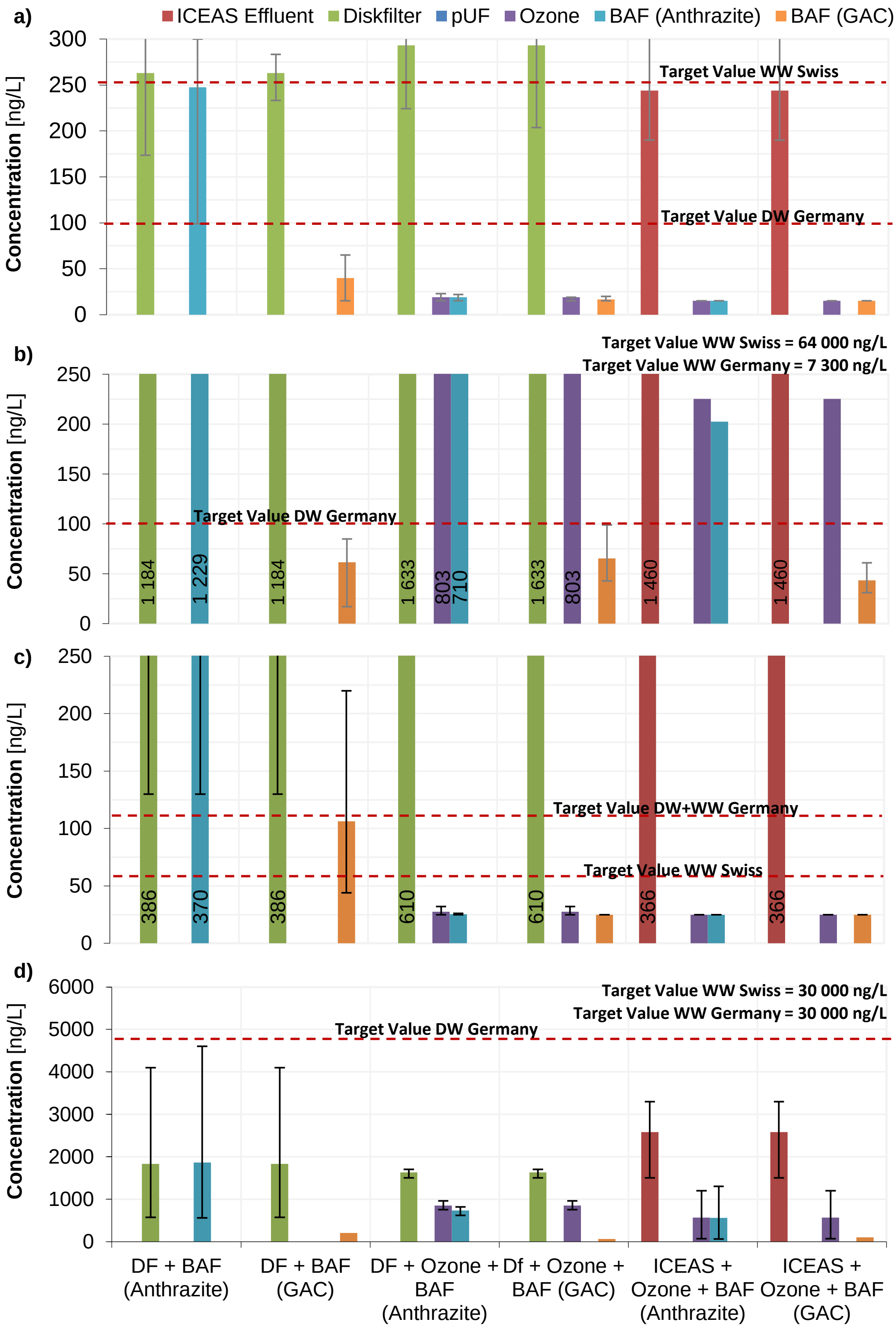

Figure 4.14. Removal of a) Carbamazepine; b) Metoprolol, c) Diclofenac; and d) Benzotriazole by different treatment processes and some target values for drinking (DW) and wastewater (WW).

The inlet concentrations of Carbamazepine are close to target value for Carbamazepine in Swiss WW. BAF (anthracite) is not able to reduce the concentrations significantly. BAF (GAC) reduces Carbamazepine more efficient under DW limit value. Ozone reduces Carbamazepine to detection limits. Carbamazepine is a substance with high reaction kinetics for ozone. For Metopronol inlet concentrations are below WW target values. BAF (anthracite) has no impact on the reduction of Metoprolol. BAF (GAC) show good adsorption and results in values below DW target value. Ozone reduces Metoprolol but the DW target can only be reached by additional adsorption by GAC. Depending on the ozone concentration, the removal of Metoprolol can be increased. Inlet Diclofenac concentrations are varying between 350 – 600 ng/L and are above the known target values for this substance. BAF (anthracite) cannot reduce Diclofenac significantly. BAF (GAC) adsorbs a lot of Diclofenac but not below the Swiss target value. Ozone reduces Diclofenac due to its high reaction kinetics efficiently down to detection limit. The inlet concentrations of Benzotriazole are below DW target value. BAF (anthracite) shows minor removal. BAF (GAC) shows significant adsorption of Benzotriazole. Ozone shows different removal rates depending on ozone dosage.

The following conclusions were drawn from the tests:

1. *Anthracite as a filtration media should not be used for micropollutant removal in general as target effluent concentrations could not be reached. Further investigation should focus if the biological activity can be improved to reduce higher loads of biodegradable substances. Further, transformation products from the ozonation should be measured to see effects of removal of these specific substances.*
2. *GAC removes most of the investigated substances very efficient but breakthrough potential of specific substances or desorption of specific substances has to be considered. The EBCT (empty bed contact time) has no impact on the removal of substances at least in the investigated configuration.*
3. *Ozonation provides good removal rates for most substances depending on their specific reaction kinetics and the ozone dosage. Ozone treatment can also improve the removal performance of GAC because loads to the filter are reduced, which improves the run time of GAC filtration.*
4. *Disk filters (DF), submerged (sUF) or pressurised (pUF) ultrafiltration have a low removal efficiency regarding micropollutants.*

4.1.4 **Ecotoxicity**

The used YES and YAS tests are standardized tests to measure the estrogenic and androgenic effects of water samples. These effects are related to concentrations of natural or industrial hormones (known as endocrine disrupting compounds (EDC)) in the water. In general, these substances will be easily removed by ozone (at lowest ozone dosages) or GAC. Table 4.14 shows the results of YES for the treatment trains including ozone and BAF.

Table 4.14. Results of YES for treatment trains including ozone and BAF (anthracite and GAC).

	Inlet	ICEAS	DF	pUF	sUF	Ozone	BAF1	BAF2	Disinf.
SBR+DF+O3+BAF1 (anth.)									
SBR+DF+O3+BAF2 (GAC)									
SBR+Screen+pUF+O3									
SBR+RGSF+Disinfection									
SBR+DF									
SBR+Screen+pUF									
SBR+sUF									

Red - high estrogenic effects; Yellow - medium estrogenic effects; Green - low estrogenic effects

The raw water shows high estrogenic effects, which are reduced during the biological treatment. Ozone reduces the estrogenic effects significantly. The effect of BAF or disinfection cannot be seen here.

Table 4.15 shows the results of YAS for the treatment trains including ozone and BAF. The androgenic effects are removed by the biological system. The effect of UF, ozone, BAF and disinfection cannot be seen here.

Table 4.15. Results of YAS for treatment trains including ozone and BAF.

	Inlet	ICEAS	DF	pUF	sUF	Ozone	BAF1	BAF2	Disinf.
SBR+DF+O3+BAF1 (anth.)									
SBR+DF+O3+BAF2 (GAC)									
SBR+Screen+pUF+O3									
SBR+RGSF+Disinfection									
SBR+DF									
SBR+Screen+pUF									
SBR+sUF									

Red - high estrogenic effects; Yellow - medium estrogenic effects; Green - low estrogenic effects

The principle of Microtox is based on a reduction in luminescent ability during exposure to contaminants or pollutants (water samples). This reduction is taken as a measure of toxicity. Table 4.16 shows the result of the Microtox test. All values are for EC20 / 5 minutes (lowest effect and contact time). The effects measured by Microtox are reduced after secondary treatment. Further treatment has no impact in this method.

Table 4.16. Results of Microtox for treatment trains (as EC20/5min).

	Inlet	ICEAS	DF	pUF	sUF	Ozone	BAF1	BAF2	Disinf.
SBR+DF+O3+BAF1 (anth.)									
SBR+DF+O3+BAF2 (GAC)									
SBR+Screen+pUF+O3									
SBR+RGSF+Disinfection									
SBR+DF									
SBR+Screen+pUF									
SBR+sUF									

Red - high effects; Yellow - medium effects; Green - low effects

ICEAS and ozone reduce estrogenic effects effectively. The effect of BAF or disinfection could not be shown due to low incoming concentrations. Androgenic effects were so effectively reduced in ICEAS that the effect of other treatments could not be evaluated.

Microtox was not sensitive enough to show any remaining toxicity after ICEAS.

4.1.5 Stabilization by chlorination

The first test aimed at defining the impact of water quality on the chlorine demand. Water samples after treatment including a secondary treatment by ICEAS, a filtration by media filter (RGSF) and full disinfection by UV exposure at 20mJ/cm² were analyzed for ammonia, COD, UVT and the color. The low quality sample was characterized by a higher content in ammonia and COD and a low color adsorbance.

Table 4.17. Water quality pre-chlorination.

	NH4 mg/L	COD mg/L	UVT %	Colour cm^{-1}
Low Quality	3.5	52	50	0.05
High Quality	0.8	35	58.9	1.7

The chlorine demand for these samples is presented in the Figure 4.15. The results show that the low and high quality water requires a chlorine demand of 22 and 7.3 mg/L, respectively. The detected ratios between the total chlorine to dose and the ammonia concentration are 6.3:1 and 9.1:1, respectively. These calculated ratios are close to the expected ratio of 7.5:1. The collected data could not specifically correlate the chlorine demand to the COD content, UVT, or watercolor. It is likely that these factors play a role on the require chlorine demand, and these parameters should be subjected to further investigations.

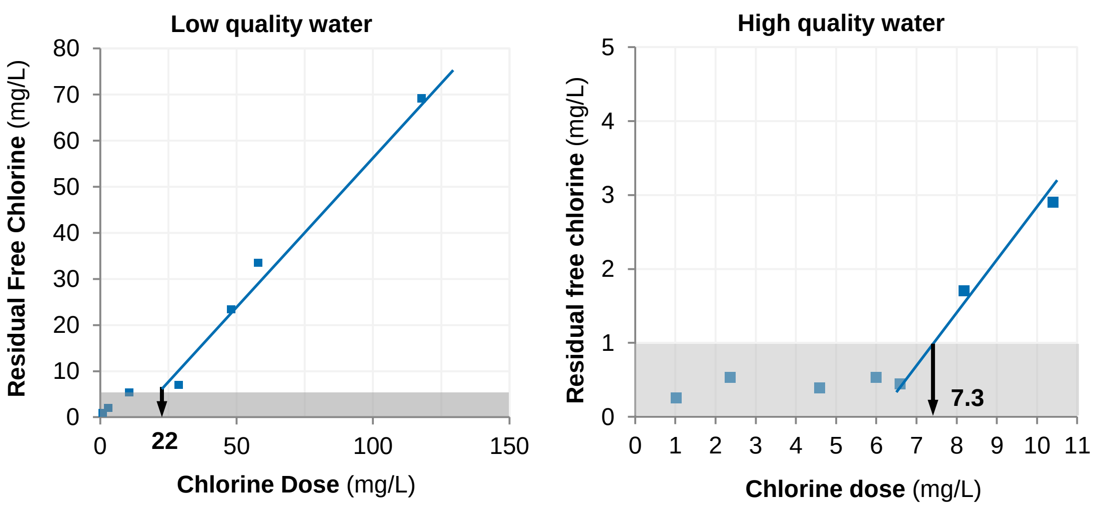

Figure 4.15. Chlorine demand as a function of the ammonia content.

The effect of pre-disinfection was tested on the chlorine demand for stabilization. Three disinfection technologies were evaluated. The tertiary effluent used was the disc filter effluent after full denitrification in ICEAS.

- The first disinfection used was oxidation by chlorination. According to the Collins model, a full disinfection to 2.2 CFU total coliform/100mL is obtained with less than 10 mg/L of free chlorine and a contact time of 30 min. The current experiment used a free chlorine concentration of 20 mg/L. The excess free chlorine was degassed by maintaining the sample refrigerated at 4 degrees C overnight.

- The second disinfection was oxidation by ozone. An ozone dose of 10 mg/L for a reaction time over 30 min was applied by running the DF effluent on the ozone pilot. The used dose was demonstrated as greater than the required ozone dose for a full total coliform disinfection (see detail in specific Ozone Report for Reuse Application)
- The third disinfection technology used was UV. The UV was applied using a column-beamed device on a 500 mL batch sample of the DF effluent. A UV intensity of 20 mJ/cm^2 was used as it was demonstrated to induce full disinfection of total coliform (see detail UV/CBD report for Reuse application).

The effluent quality after filtration and disinfection is listed in Table 4.18. The effect of disinfection on the water quality is shown as a percentage of removal of the specific contaminant. The values given in the table are specific to the experiment and aim at demonstrating the effect of oxidation during disinfection. Results show that disinfection by chlorination and ozone reduced the COD content and improved the UVT and color. Chlorination also reduced the ammonia content to form chloramines. Disinfection by UV did not affect any of the quality parameters tested in this experiment. The disinfection by oxidation generally improves the general water quality, which can then reduce the chlorination demand for stabilization.

Table 4.18: Water quality and impact of the disinfection technology on the quality.

	NH4 mg/L	NH4 removal	COD mg/L	COD removal	UVT %	UVT increase	Colour m^{-1}	Colour removal
Tertiary effl.	0.25		35		60.5		1.7	
+ disinf. Cl_2	0.10	60%	25	29%	70.2	16%	0.4	76%
+ disinf. Ozone	0.25	0%	29	17%	75.5	25%	0.4	76%
+ disinf. UV	0.25	0%	39	0%	58.9	0%	1.7	0%

The results of the tests on chlorination demand are represented in Figure 4.16. A dose of 0 to 12 mg/L of free chlorine was applied on each of the four water samples. The residual free chlorine was measured after a contact time of 30 min. The breakpoint chlorination is detected for each chlorination curve. Result show that the tertiary effluent and UV disinfection have similar chlorination demand at breakpoint of 7.3 to 8 mg/L of free chlorine. The applications of ozone and chlorination for disinfection reduced the chlorination demand for stabilization to 6 and 2 mg/L of free chlorine, respectively. The oxidation of organic carbon observed for the disinfection by oxidation favors the reduction of chlorine demand. The addition removal of ammonia by disinfection by chlorination further decreases the chlorine demand for stabilization.

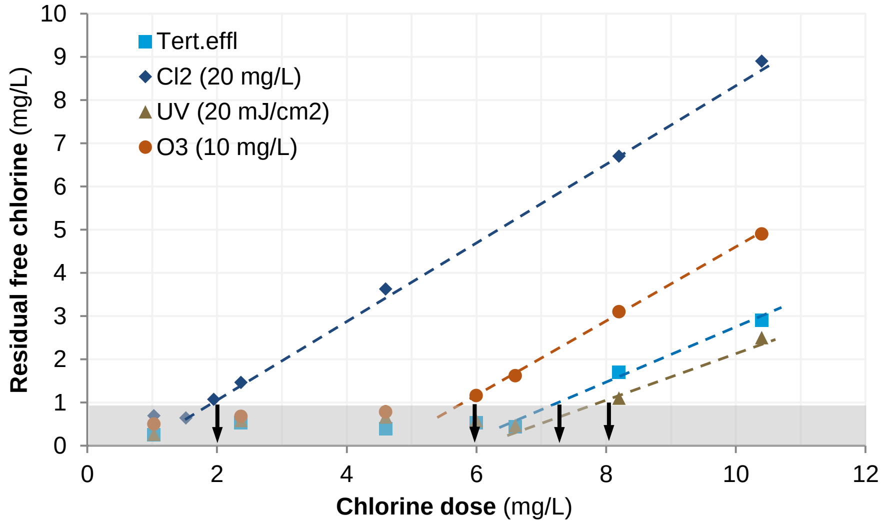

Figure 4.16: Chlorine demand as a function of the disinfection technology.

The technology used for disinfection is highly affecting the chlorination demand for stabilization. Oxidation technologies, such as chlorination and ozonation, kill microorganisms by oxidation of the organic material that keeps the integrity and viability of the living cells. The oxidation leads to the disintegration of this organic material in smaller fragments and eventually to elimination of a fraction of the carbon load in the water by carbon dioxide degasing. The lower carbon load in the water reduces the available organic matter that consumes the free chlorine available before reaching breakpoint and reduces then the chlorination demand for stabilization.

The removal of ammonia during disinfection was shown to reduce the most the chlorine demand. A full nitrification of the water is then advisable in order to reduce the chemical consumption downstream the treatment and optimize the stabilization of the final effluent.

The disinfection by UV was shown to have little effect on the stabilization of the final effluent. The UV treatment deactivates the genetic material of microorganisms that is light sensitive. The organisms cannot reproduce anymore, but their cell structure remains intact and their organic content remains the same. Despite being deactivated, the microorganisms remain in the final effluent and are available to react with the free chlorine before breakpoint.

The following conclusions were drawn from the tests:

1. *Nitrification is crucial to reduce the chlorine demand for both disinfection and stabilization and*
2. *UV disinfection has no reduction on the chlorine demand for stabilization due to the lack of oxidation potential.*

4.1.6 **Treatment train performances**

All treatment trains met the required effluent quality as defined in Table 2.2. As these target effluent qualities were minimum monthly average requirements, the treatment systems did perform better during most of the test period. Figure 4.17 presents average effluent qualities for the different investigated ReUse treatment systems and most common parameters.

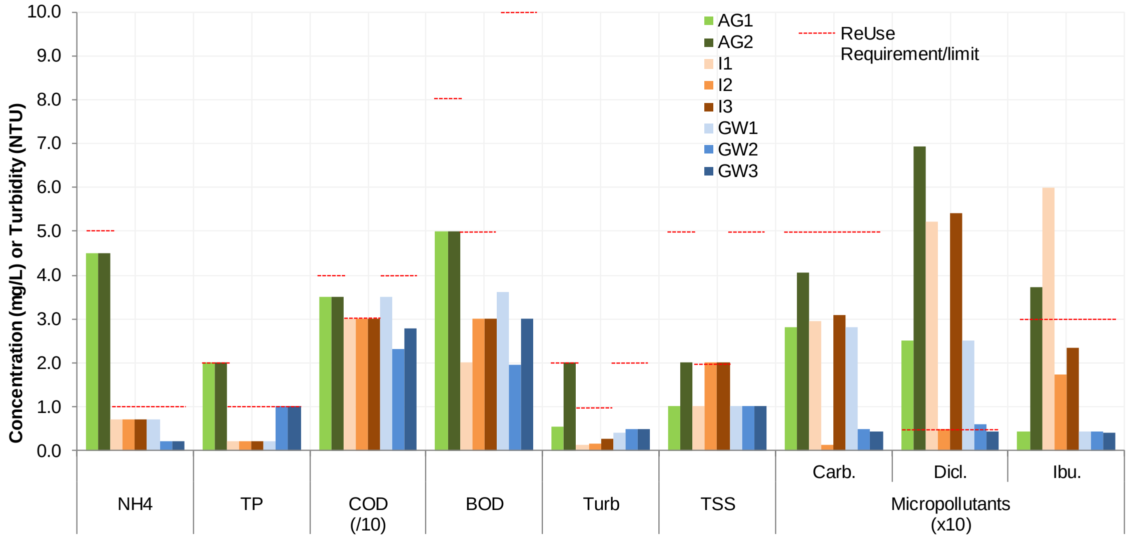

Figure 4.17: Effluent quality results for the different treatment trains and effluent quality requirements for the three reuse categories AG, GW and I (see section 3.3, and removal of micropollutants (here exemplified by three pharmaceuticals Carbamazepine, Diclofenac and Ibuprofen, and recommend maximum concentrations according to Table 2.3).

It can be observed that effluent qualities within the same targeted water reuse application may differ. This is mainly explained by distinct treatment step characteristics included and not in the various treatment train setups.

4.1.7 Data quality and uncertainties

The collected data from the pilot tests must generally be judged as high quality data. This is because online measurements were completed and backed up with both onsite and external laboratory analyses. All data was checked and discussed by the project group and in certain cases reviewed by external experts for quality control. All external analytical results were checked by an IVL analytical expert for reliability and consistency and repeated analytics required if mistakes were detected. Method improvements have also been initiated by the ReUse project in order to improve detection limits of some analytical methods.

The limits of detection for single substances vary depending on the water matrix, the sample preparation and the analytical method applied. In order to minimize the risk of treatment assessment on false analytical results, several external laboratories have been used to analyze substances in the same sample water. This, together with results from other projects enables the project to select the most relevant and robust methods. Several problems such as far too low or inconsistent analytical results have been observed during these tests.

In order to obtain reliable online-readings of observed parameters, weekly validation measurements were performed and if necessary adjustments of the probes done. Analyses with an expected high uncertainty, such as when new analysis methods or parameters were included, triplicate test were performed until a high reliability of the analyses could be guaranteed. To guarantee high data quality of external analyses, test samples and blinds were send simultaneously to several labs and results evaluated before selecting the most reliable laboratory.

4.2 Life Cycle Assessment of investigated reuse applications

4.2.1 **General**

For agriculture reuse application, the lowest environmental impact and best effluent quality has the line with RGSF. Addition of nutrient removal and chlorination to achieve groundwater recharge qualities increases the overall environmental impact. For industrial reuse, the lowest environmental impact was achieved with the line including sUF and UV. The industrial reuse line that produced the best effluent quality with lowest concentrations of micropollutants has the highest environmental impact. For Groundwater Recharge, the lowest environmental impact has the treatment system including RGSF. The other GW-treatment systems are actually reaching industrial effluent quality but at a higher total environmental impact.

4.2.2 **Reuse application: Agricultural use**

The two treatment trains for agricultural wastewater reuse are intended for irrigation and the only technological difference between the trains is the substitution of the RGSF with a disk filter. Hence, the divergence in environmental impact between the trains will depend on the differences between the RGSF and the disk filter.

The results were regarded due to change in technology and influence of plant size. The treatment trains were also compared to each other.

The disk filter requires an additional polymer dosage, however, the RGSF is more energy demanding. In addition, the two techniques differ in their material components and composition. Figure 4.18 shows the GWP for the AG-treatment trains categorized in electricity, chemicals, materials, and off-gas from the secondary treatment (N_2O).

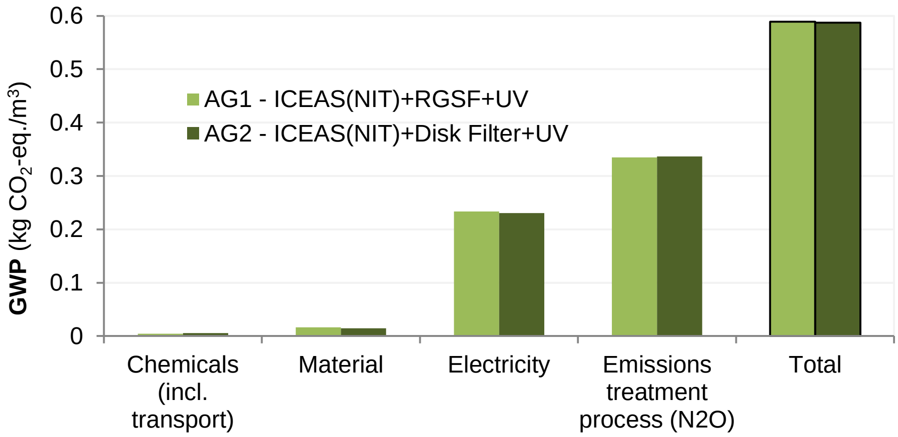

Figure 4.18. Global warming potential (GWP) for treatment trains AG1 and AG2, 20 000 pe.

The major contributing factors to each impact category for the treatment trains are summarized in Table 4.19. Clearly, electricity is the dominating factor for all impact categories except for GWP and POCP.

Table 4.19. The major contributing factors to each impact category for train AG1 and AG2.

Impact category	Train AG1	Train AG2
Global Warming Potential (GWP)	Off-gas (N_2O)	Off-gas (N_2O)
Acidification Potential (AP)	Electricity	Electricity
Eutrophication Potential (EP)	Electricity	Electricity
Photochem. Ozone Creation Potential (POCP)	Polymer flocc.	Polymer flocc.
Human Toxicity Potential (HTP inf.)	Electricity	Electricity
Freshwater Aquatic Ecotoxicity Pot. (FAETP inf.)	Electricity	Electricity
Marine Aquatic Ecotoxicity Pot. (MAETP inf.)	Electricity	Electricity
Terrestric Ecotoxicity Potential (TETP inf.)	Electricity	Electricity
Abiotic depletion (ADP), elements	Electricity	Electricity
Abiotic depletion (ADP), fossil	Electricity	Electricity

Error! Reference source not found. in the Appendix provides the environmental impacts for all trains, plant sizes and each category. Replacing the RGSF with a disk filter has almost no influence on the environmental performance. The difference is however most significant for POCP and AD (elements). For POCP it is due to the additional polymer dosage required for the disk filter, which is dominating for POCP. For AD (elements) it is due to the higher impact from the materials required for the disk filter (mainly glass fiber reinforced plastic), which outweighs the somewhat higher use of electricity for the RGSF used in train AG1.

Table 4.20 illustrates the impact of the off-gas (N_2O) from SBR as the largest contributor to the GWP. The second largest contributor is electricity. The SBR and the sludge treatment on site are the dominating users of electricity in the present treatment designs.

Table 4.20. Relative contribution of various impacting items to GWP for different plant sizes.

	AG1			AG2		
	20 000 pe	100 000 pe	500 000 pe	20 000 pe	100 000 pe	500 000 pe
Off-gas (N_2O)	56.8 %	63.6 %	65.4 %	57.2 %	64.0 %	65.6 %
Electricity for water treatment (SBR)	12.5 %	13.3 %	15.5 %	12.5 %	13.4 %	15.4 %
Sludge handling onsite*	19.2 %	14.8 %	12.6 %	19.3 %	14.8 %	12.6 %

* thickening, dewatering and aerobic digestion

The effect of plant size on the environmental impact is 20 000 pe> 100 000 pe ≥500 000 pe. As the plant size increases, the environmental impact due to equipment decreases, also, the electricity use per cubic meter treated water decreases, except for the secondary treatment, and for the disk filter, illustrated in Figure 4.19. These two observations can be regarded as a general trend, for all treatment trains as well as for all impact categories studied.

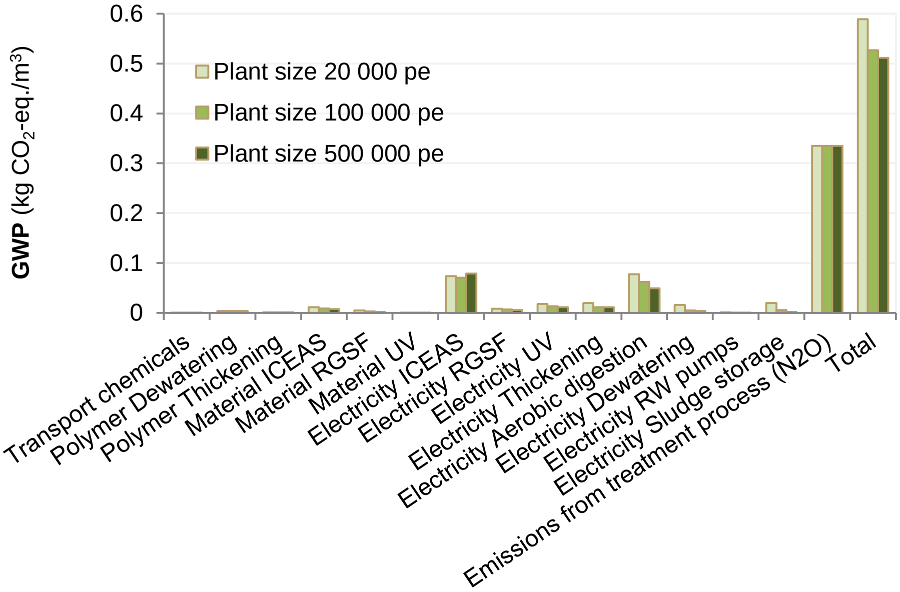

Figure 4.19. Impact of plant size to global warming potential (GWP), train AG1.

The normalized environmental impacts are shown in Figure 4.20.

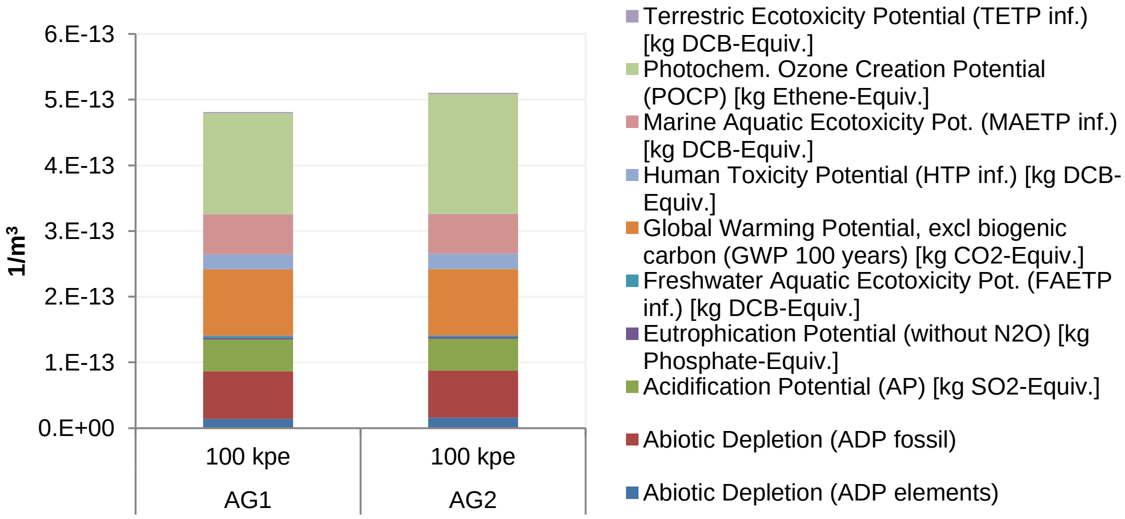

Figure 4.20. The environmental impact normalized to EU25+3 yearly emissions for train AG1 and AG2.

In conclusion, the major contributors to the environmental impact over the entire trains are; use of electricity from fossil fuel sources, emissions of N_2O from the treatment process (off-gas) and use of polymer for some impacts. The SBR and the aerobic digestion onsite are the dominating users of electricity in the present treatment designs. As the plant size increases, the environmental impact due to equipment decreases, also, the electricity use per cubic meter treated water decreases, except for the secondary treatment, and for the disk filter.

The exchange of RGSF for a disk filter does not efficiently address a major environmental issue as gauged within the assessment boundaries. The difference between the trains is most significant for POCP due to the additional polymer dosage required for the disk filter, which is dominating for the POCP.

4.2.3 Reuse application: Industrial reuse

The three treatment trains are intended for industrial wastewater reuse and the technological difference between the trains is the use of sUF vs. pUF as well as ozonation vs. UV. Hence, the divergence in environmental impact between the trains will depend on the differences between the energy, material and chemicals required for sUF/pUF and UV/ozonation.

Comparison between the lines on regard to GWP is seen in Figure 4.18. The N2O emissions from the treatment process are not as dominating as for the AG lines due to the NDN mode instead of the NIT mode with much less GHG-emissions from the biological process. It is instead the electricity, which is most dominating for GWP. The table below summarizes the most dominating factor for each impact category.

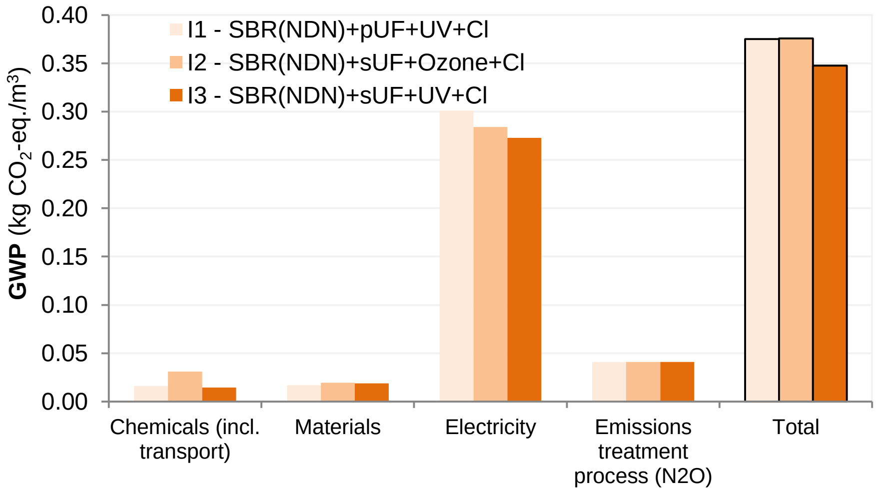

Figure 4.21. Global warming potential (GWP) for the industrial treatment trains at 20 000 pe.

As can be seen in Table 4.21, electricity is the most dominating cause for all impact categories except POCP and ADP elements. For POCP it is instead the use of polymer in sludge treatment and for ADP of elements it is the use of sodiumhypochlorite for process cleaning.

Table 4.21. Dominating resource/effect for the various impact categories for industrial reuse lines.

Impact category	Train I1	Train I2	Train I3
Global Warming Potential (GWP)	Electricity	Electricity	Electricity
Acidification Potential (AP)	Electricity	Electricity	Electricity
Eutrophication Potential (EP)	Electricity	Electricity	Electricity
Photochem. Ozone Creation Potential (POCP)	Polymer flocc.	Polymer flocc.	Polymer flocc.
Human Toxicity Potential (HTP inf.)	Electricity	Electricity	Electricity
Freshwater Aquatic Ecotoxicity Pot. (FAETP inf.)	Electricity	Electricity	Electricity
Marine Aquatic Ecotoxicity Pot. (MAETP inf.)	Electricity	Electricity	Electricity
Terrestric Ecotoxicity Potential (TETP inf.)	Electricity/ NaClO	Electricity/ NaClO	Electricity/ NaClO
Abiotic depletion (ADP), elements	NaClO	NaClO	NaClO
Abiotic depletion (ADP), fossil	Electricity	Electricity	Electricity

For TETP, both electricity and sodiumhypochlorite are dominating. Here, electricity is most dominating for the 20 000 pe, for all lines. However for 100 000 pe and 500 000 pe it is sodiumhypochlorite, this trend is illustrated for train I3 in the figure below.

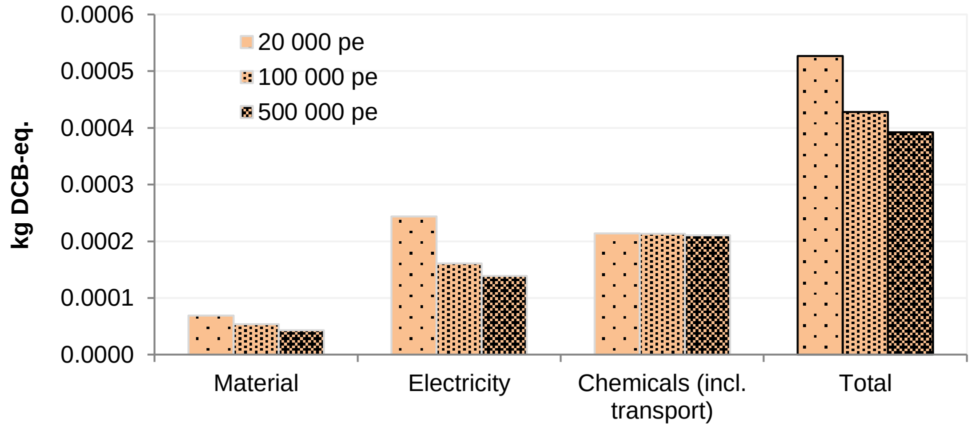

Figure 4.22. Impact of various resources used on the terrestric ecotoxicity potential exemplified by industrial reuse treatment Train I3.

The impact of electricity from the different modules is shown for AP in the graph below. The SBR (ICEAS) and the sludge treatment on site are the process units with the highest electricity use and thus acidification potential. It is also seen that the pUF is more electricity demanding than the sUF, and that ozonation requires somewhat more electricity than UV.

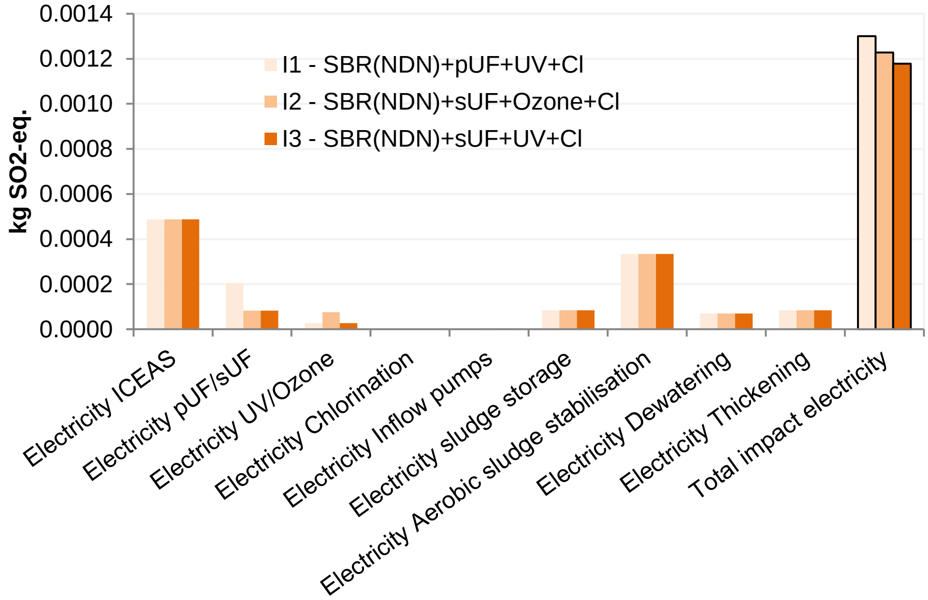

Figure 4.23. Impact of electricity from various processes on the acidification potential for industrial reuse treatment trains (at a size of 20 000 pe).

Figure 4.24 shows the impact of the materials on FAETP as another example. It can be seen that the impact from ozonation is somewhat higher than from the UV. This is mainly because of the larger amount of stainless steel required for the ozone reactor, which has a higher impact on FAETP. Also illustrated in Figure 4.25 are the impacts on FAETP from the different materials required for the sUF and pUF. It is mainly the polyethylene pipes, PVDF and stainless steel used for sUF, which has the largest impact on FAETP.

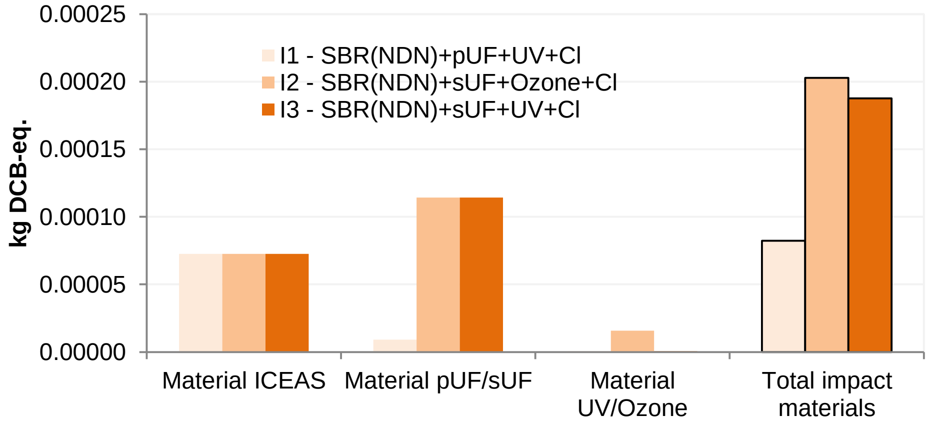

Figure 4.24. Impact of material use for various processes on the freshwater aquatic ecotoxicity for industrial reuse treatment trains (at a size of 20 000 pe).

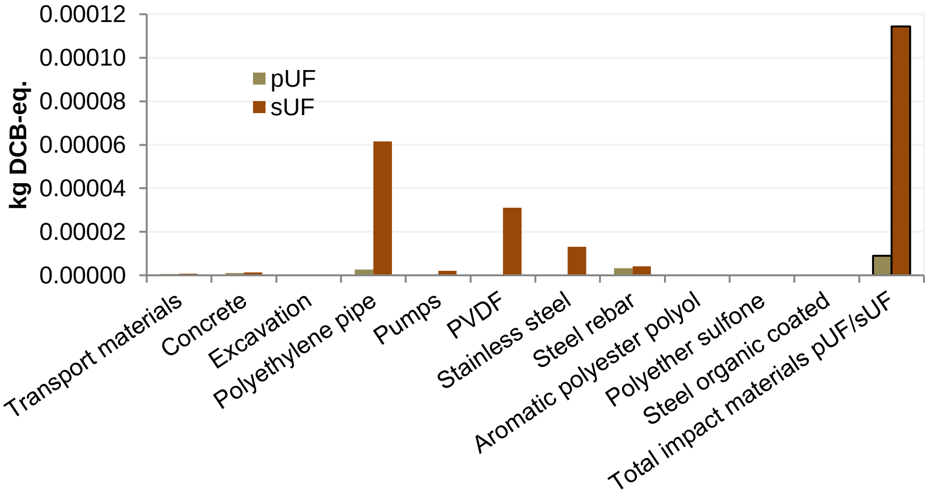

Figure 4.25. Impact of different materials required for the sUF and pUF on the freshwater aquatic ecotoxicity (at a size of 20 000 pe).

The materials for the sUF consistently have a higher impact than for the pUF on all KPIs. This is true except for ADP elements, where the copper in the pumps used for providing the pressure for the pUF increases the impact on ADP.

As seen in Table 4.21, the impact from chemicals is of major importance for POCP, TETP and ADP elements. In Figure 4.26, the impact on TETP from the different chemicals used for the I-lines is shown. The trend for ADP elements is similar to the graph below, but for POCP the impact is mainly due to the polymer.

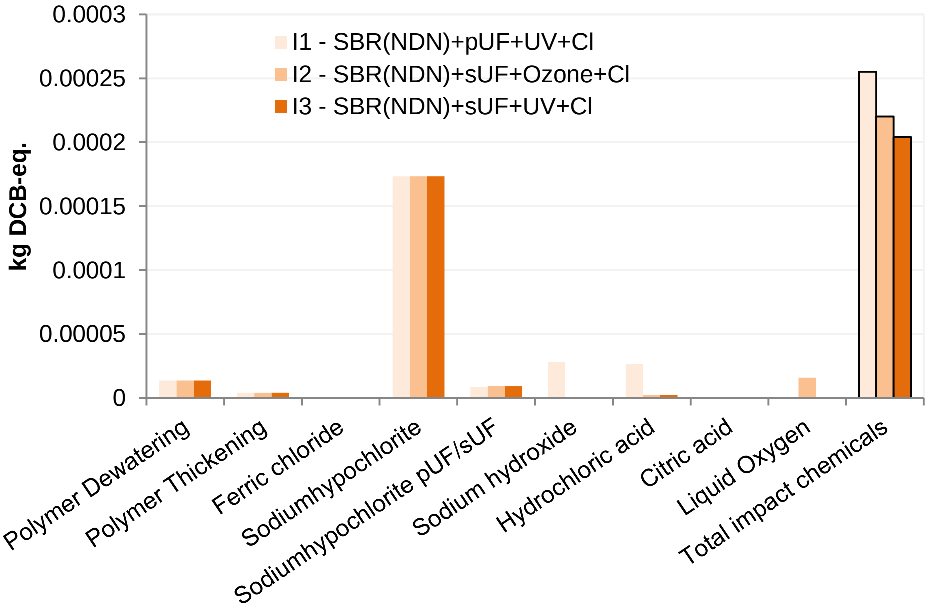

Figure 4.26. Impact of different chemicals used in the industrial reuse treatment trains on the terrestic ecotoxicity potential (at a size of 20 000 pe).

In total, treatment train I3 has the smallest impact, which is true for all KPIs and all plant sizes, except for FAETP for plant size 20 000 pe. The reason for this is mainly that less energy is required for operation of the sUF than for the pUF. At the same time, ozonation is not required for this treatment train, which increases the impact for the other two lines due to an increased material use ozonation than for UV. In addition, no additional chemicals are required as is for the treatment system with pUF (sodiumhydroxide) and the treatment system with ozonation (liquid oxygen).

The exception for train I3 when it comes to FAETP is due to the relatively large influence of materials on FAETP compared to the other KPIs. The use of sUF creates thus a higher impact from materials than for other lines. That this only applies for the smallest plant size of 20 000 pe is because the total impact per treated cubic meter water is larger the smaller the plant size as illustrated with an example in Figure 4.27.

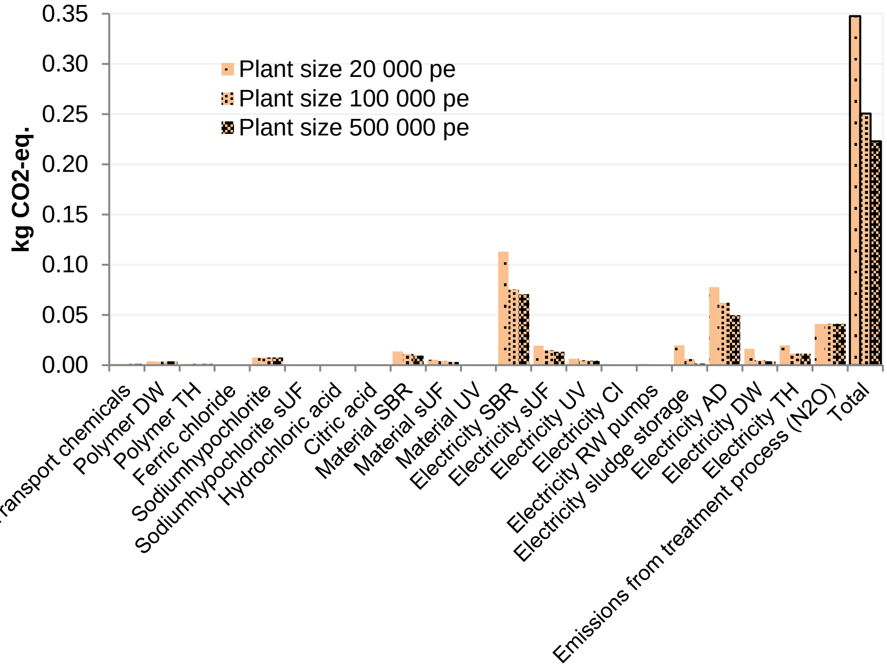

Figure 4.27. Impact of materials, chemicals and electricity uses exemplified by industrial reuse treatment Train I3 on the global warming potential.

All results are provided in the appendix (**Error! Reference source not found.**). The difference between I1 and I2 is relatively small, although, for plant size 20 000 pe, train I2 has higher environmental impacts on all KPIs except for the three KPIs: POCP, TETP and ADP elements, for which I1 instead has a higher impact. However, as the plant size increases, I1 becomes the industrial reuse treatment system with the highest environmental impact for more and more KPIs. At 100 000 pe, I1 impacts are highest for seven KPIs, at a plant size of 500 000 pe this number includes all nine KPIs.

The influence of plant size within each train is that the environmental impacts per treated m^3 wastewater are 20 000 pe> 100 000 pe ≥500 000 pe. With increasing plant size, the electricity use per m^3 treated water decreases and the environmental impact due to the required equipment decreases too.

Normalization of the calculated impacts to EU25+3 yearly emissions was also performed and is shown in Figure 4.28 for a plant size of 100 000 pe.

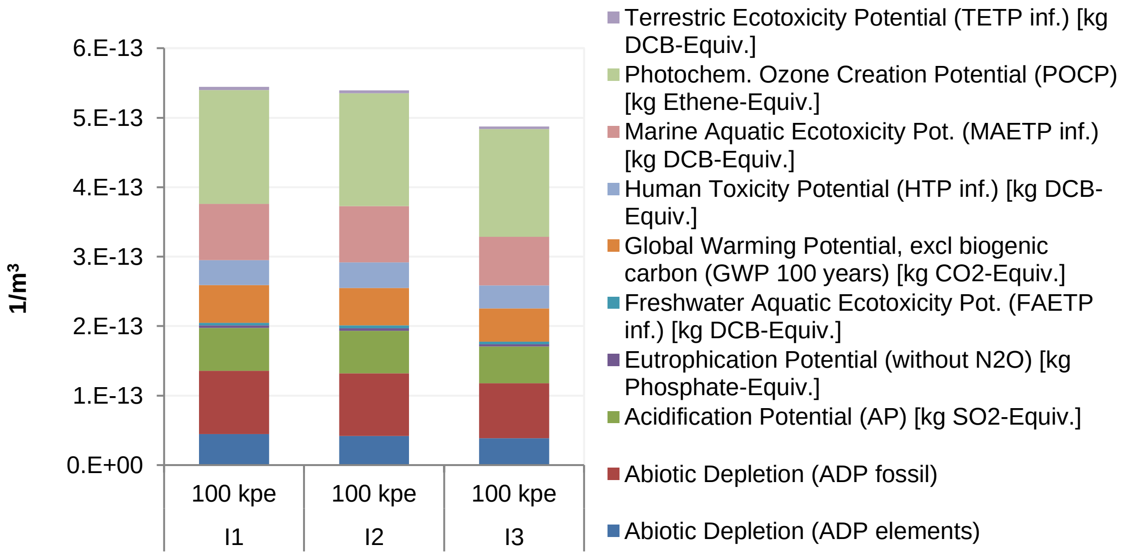

Figure 4.28. The environmental impact normalized to EU25+3 yearly emissions for industrial reuse treatment trains and a plant size of 100 000pe.

4.2.4 Reuse application: Groundwater recharge

Results for all three investigated plant sizes and treatment trains intended for groundwater recharge with reclaimed wastewater are provided in **Error! Reference source not found.** (Appendix). The table below provides information about which of the resources/effects of the system that are dominating the various impact categories.

Table 4.22. Dominating resource/effect for the various impact categories for industrial reuse lines.

Impact category	Train GW1	Train GW2	Train GW3
Global Warming Potential (GWP)	Electricity	Electricity	Electricity
Acidification Potential (AP)	Electricity	Electricity	Electricity
Eutrophication Potential (EP)	Electricity	Electricity	Electricity
Photochem. Ozone Creation Potential (POCP)	Polymer flocc.	Polymer flocc.	Polymer flocc.
Human Toxicity Potential (HTP inf.)	Electricity	Electricity	Electricity
Freshwater Aquatic Ecotoxicity Pot. (FAETP inf.)	Electricity	Electricity	Electricity
Marine Aquatic Ecotoxicity Pot. (MAETP inf.)	Electricity	Electricity	Electricity
Terrestric Ecotoxicity Potential (TETP inf.)	Electricity/ NaClO	Electricity/ NaClO	Electricity/ NaClO
Abiotic depletion (ADP), elements	NaClO	NaClO	NaClO
Abiotic depletion (ADP), fossil	Electricity	Electricity	Electricity

The use of electricity (mainly for the ICEAS and aerobic digestion) produced from fossil sources and N2O-emissions (valid for GWP) are dominating the impacts categories. For some KPI:s, also the use of chemicals (polymer and sodium hypochlorite) becomes important.

The differences between GW2 and GW3 are relatively small for most of the KPIs. The only technological difference between the trains is the Disc filter, which is introduced for GW2. GW3 uses more electricity for the ozonation than GW2. GW3 uses also more liquid oxygen than GW1, which "compensates" for the extra process step (disk filter for GW2). POCP is an exception of this trend, where a more significant difference can be observed. This is because chemicals are the main contributor, and hence the additional polymer dosage required for the disk filter for GW2, implies an increased impact. Likewise for ADP elements due to impact of the material for the disk filter.

Both, GW2 and GW3 have more process steps, than GW1, AND the environmental impact for GW2 and GW3 is higher than for GW1. GW2 has most process steps; however, this is outweighed by the lower electricity consumption for the ozonation and the lower use of liquid oxygen than in the other lines .

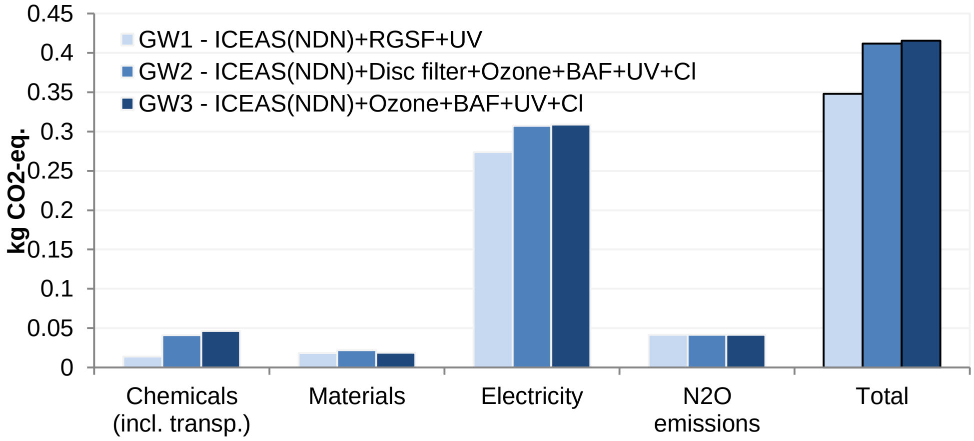

Figure 4.29. Impact of various resources used on the global warming potential for groundwater recharge treatment trains and a plant size of 20 000pe.

As for the agriculture and industrial reuse treatment trains, normalization was performed. Figure 4.30 shows the environmental impact normalized to EU25+3 yearly emissions for train GW1, GW2 and GW3, 100 000 pe.

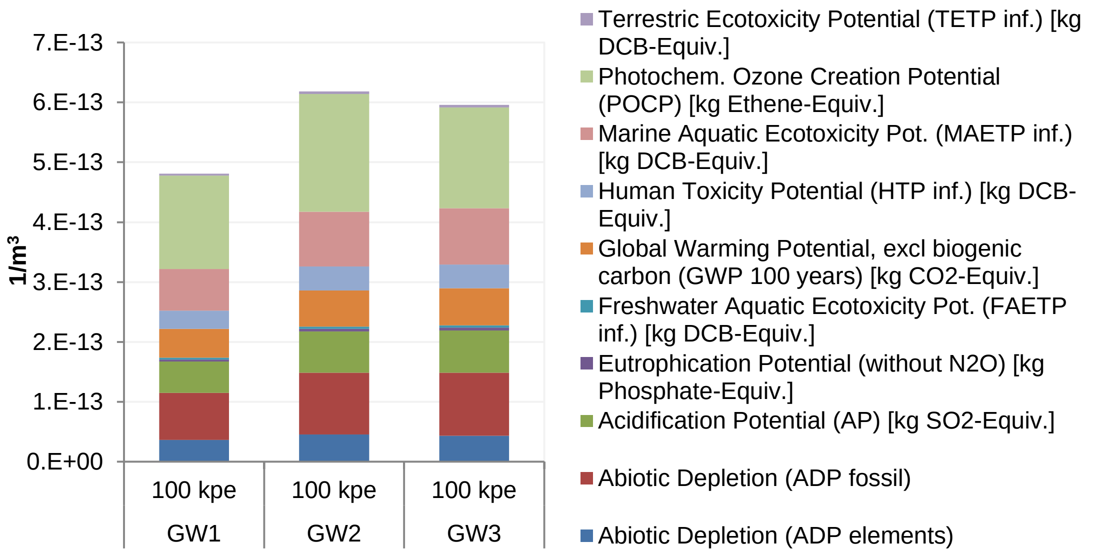

Figure 4.30. The environmental impact normalized to EU25+3 yearly emissions for groundwater recharge treatment trains and a plant size of 100 000pe.

4.2.5 Recognized uncertainties

In the LCA model, the recognized main uncertainties and limitations include:

- uncertainties in the core modules of the model such as for instance electricity and chemical usage. Input to these core modules are based on actual tests, models and full-scale installations. Hence, the uncertainty of these figures, as they represent an average plant design, is induced by natural variations and uncertainty in this data, which will also affect the uncertainty of the LCA results.
- the choice of life cycle inventory data used for the upstream modules, such as for instance chemicals, building materials and electricity, and how well these actually represent the "real" situation.
- uncertainties within the inventory database data. Sometimes standard deviations are reported for database data, which can be in the order of magnitude of 100 %.
- data gaps, are always associated with uncertainty.
- uncertainties in the characterization models, it is for instance known that the uncertainties within the toxicity potential models are quite large. This is due to both the uncertainty in the modelling of the fate of the emitted substances in the nature and uncertainties in toxicity data for different organisms.

The most dominating factors in the LCA study are electricity, off-gas (N2O), polymer and sodium hypochlorite. The dosages of sodium hypochlorite and polymer as well as electricity requirements are determined parameters from pilot-plant tests and full-scale installations. Thus, the uncertainty was tried to be kept at a minimum. Nitrous oxide emissions were based on actual measurements using the pilot and mass balance calculations. The use of this data in the LCA implies uncertainties.

4.2.6 Decisive system parameters – Selection of scenarios and parameters for sensitivity analyses

4.2.6.1 Sludge stabilisation (aerobe vs. anaerobe)

The main sludge stabilization method was selected as aerobic as this has been identified as being the most relevant handling method in the regions of reuse focus in the pre-study review. As results from the environmental impact assessment (see 4.2.2 to 4.2.4) showed, sludge handling is one of the main contributing processes especially as it requires much energy. To evaluate the impact of sludge handling on the overall environmental performance of the treatment trains, anaerobic sludge stabilization was replaced by anaerobic sludge stabilization in one of the modelled scenarios and presented by Dahlgren *et al*. (2014). Sludge disposal was not investigated for this case but a comparative impact analyses for one treatment system consisting of SBR(NDN)+sUF+UV+Cl and a plant size of 100 000 pe was performed. GHG-emissions may be higher from the anaerobic sludge treatment due to methane leakage during digestion and handling, however, the produced energy is assumed usable to cover parts of the energy need in the train processes. A methane leakage of 2% of the total methane produced was assumed in the evaluation. This conforms to commonly reported values and experiences by the project group from full-scale installations.

Following assumptions were used:

- For both
  - polymer dose of 2kg/tDS for thickener, 8 kg/tDS for dewatering
  - same amount of reuse water
  - Thickener from 0.85% TS to 4% TS with 95% of TS to sludge phase
  - Dewatering to 18% TS with 93% of TS to sludge phase
  - Addition of polymer and backwash water has not been allowed to affect the flow or TS in the model.
- Anaerobic digestion at 35°C (temperature in sludge from thickener 14.9 °C).
- 2% methane slip for anaerobic digestion
- 43% efficiency for electricity generation from biogas

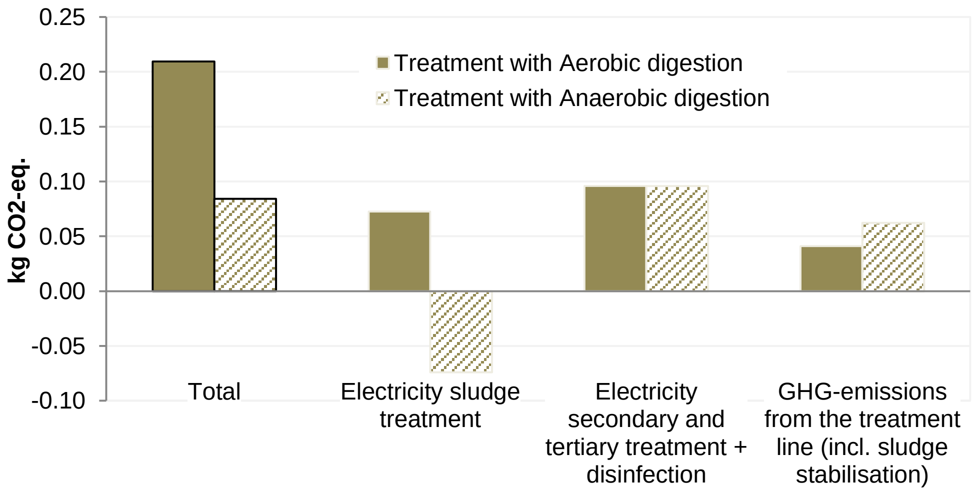

Figure 4.31. Global warming potential (GWP) for the entire treatment train with aerobic or anaerobic digestion (modified from Dahlgren *et al.*, 2014).

Results presented in Figure 4.31 show that using anaerobic digestion, instead of aerobic stabilization decreased the total GWP with 60%. This is mainly because anaerobic digestion will produce energy and not consume even so emissions of methane during anaerobic stabilization implies a negative impact.

4.2.6.2 Sludge fertilizing in agriculture

Considering the measured nutrient concentrations in the ReUse-sludge from the agricultural reuse treatment trains, nitrogen could be identified as the limiting factor for how much sludge that can be applied without exceeding the limits Total P and Total N in sludge. This has been discussed and presented by Baresel *et al.* (2015a).

When disposal of bio-sludge by using it as an agricultural fertilizer is included in the system, the approach is to calculate the acreage of land that can be fertilized with the quantity of sludge generated from the production of 1 m^3 of reclaimed water. The environmental impacts of fertilizing the same acreage with mineral fertilizers are then deducted. The deducted impacts include production and delivery of the NP-fertilizer diammonium hydrophosphate. The only emissions considered are greenhouse gases, acidification, and eutrophication compounds.

Considering the measured nutrient concentrations in the ReUse-sludge from the AG-system, and the presence of restrictions only for nitrogen, the total amount of sludge from the reclamation trains that could be used on agriculture land is limited to 8036.5 kg dry solids /(ha·yr). In other words, supporting phosphorous fertilizing may be required if more phosphorous fertilizing is desired than the maximum amount of sludge that can be spread per hectare provides. The N:P-ratio can be affected in the sludge before fertilizing or by additional phosphorous fertilizing after sludge application. The total agricultural area that could be fertilized with sludge produced from the ReUse-train amounts to 184.4 ha when meeting nitrogen limits. With the current composition of the biosludge as produced in the study and with a dose that meets the required amount of plant-available nitrogen; about 0.26 kg phosphorous per kg nitrogen is broad on the fields. This implies that enough phosphorus is supplied if considering recommended N:P ratios of commercial NP-fertilizer for wheat (NPK (ratio of elemental N, P and K) 8-24-8 plus NAC 27; see Bellido 2010).

The result of the environmental impacts evaluation of using the sludge as fertilizer shows that the major difference for considering sludge fertilizing on GWP is due to N_2O emissions during the sludge distribution on field. The major avoided impact is due to avoided emissions of CO_2 during manufacturing of the mineral fertilizer and avoided N_2O emissions during fertilization, originating from the mineral fertilizer.

Table 4.23 shows the impacts also on acidification potential (AP) and eutrophication potential (EP). The major impact of fertilization on AP is due to ammonia emissions to air emitted from the sludge during fertilization. For EP it is instead nitrate to freshwater. The avoided emissions for AP are mainly SO_2 to air during manufacturing of the mineral fertilizer, likewise for EP, it is emissions of phosphate to seawater.

Table 4.23. Comparison with and without system expansion, 100 000 pe.

Impact category	Without sludge fertilizing	With sludge fertilizing
Acidification potential [kg SO_2-Eq.]	8.15E-04	3.60E-04
Eutrophication Potential (without N_2O) [kg Phosphate-Eq.]	5.42E-05	1.27E-03
Global Warming Potential, excl biogenic carbon (GWP 100 years) [kg CO_2-Eq.]	5.27E-01	5.52E-01

The largest negative impact of reusing sewage sludge as a fertilizer is, however, for the terrestric ecotoxicity and that is mostly due to the emission of Cr to agriculture soil (62 % of contribution). Because the quantification of the impact of fertilization with sludge on terrestric ecotoxicity is not a complete due to the lack of data on the content of metals in mineral fertilizers (except for cadmium), it was not included in Baresel *et al.* (2015a). It was not possible to calculate avoided impacts properly but sewage sludge specific data is presented in the section below.

4.2.6.3 Heavy metal content in sludge

Based on pilot-plant measurements and recalculated to consider an increase in concentration due to the VSS digested in the aerobic digester (e.g. assuming a decrease in amount of TS without any loss of metals), an investigation of potential limitations of sludge application in agriculture was performed with one AG treatment train as basis (Baresel *et al.*, 2015a). As reference, Spanish (RD 1310/1990) and Swedish (SEPA 1998) limits were used.

The analysis of average concentrations of heavy metals in the sludge recalculated to relevant units showed that these were well below the Spanish limits but also Swedish limits. However, when applying nitrogen limiting amount of sludge to be used per hectare), amount of cadmium, copper, nickel and chromium exceed the Swedish limits and would thus require a reduced sludge quantity to be distributed on agriculture land per hectare (Baresel *et al.*, 2015a).

4.2.6.4 Nitrous oxide emissions

The presented environmental impact assessment for the various treatment trains indicated significant differences in GWP caused by emissions of greenhouse gases from biological treatment process. The significance of emissions of nitrous oxide (N_2O) from the biological nitrogen removal was investigated using measured data for different operation modes of the secondary treatment system and a simplified sensitivity analyses (Baresel *et al.*, 2015a; Yang *et al.,* 2015). Emissions of nitrous oxide (N2O) used in the calculations are mainly based on actual measurements during a period of 6 month within the ReUse-project accounting for 2.09% of N2O per TN being removed for AG-line and 0.2% of N2O per removed TN for I-line. However, higher emissions have been reported in literature and are accounted for in one scenario (Global Water Research Coalition report, 2011; Rodriguez-Caballero *et al.,* 2013; Sun *et al.,* 2013). Studies of various emissions of nitrous oxide (N2O) from the biological nitrogen removal process at constant energy consumption and the selected Spanish energy-mix presented by Baresel *et al.* (2015b) showed that the overall environmental impact would be affected significantly. High emissions of greenhouse gases from the biological treatment may even dominate the total environmental impact of a treatment system in a similar way as terrestic ecotoxicity from metals in sludge used as fertilizer.

4.2.6.5 Energy mix

Due to the significant impact of electricity, in specific from the secondary treatment process (ICEAS), a sensitivity analysis was performed with an agriculture reuse treatment trains as example (Baresel *et al.,* 2015a). Studies on the influence of the type of electricity supply were performed by reducing the used energy by 10 % and 20 %, respectively. Further, the Spanish average electricity mix was replaced by in turn two other electricity sources, namely Swedish average electricity and US average electricity mix (Baresel *et al.,* 2015a).

Results showed that a reduction of the energy use within the reuse treatment system has only minor impact on the GWP for the complete system. Whereas external factors such as the site location and therefore the used energy mix composition may have significant impact on the overall global warming potential of the wastewater reclamation process. As the percentage of green energy increases, going from Spanish to Swedish electricity mix, GWP decreases by 60% for the same energy consumption. Higher percentage of fossil fuels like hard goal as it is used in US mix increased the GWP by 50%. Nuclear power is commonly considered as green energy at a time horizon of 100 years. As the Swedish energy mix consists of almost 40% nuclear power, different assessment of this energy source would of course significantly alter the outcome of the performed evaluation. The decrease in GWP due to Swedish electricity mix is due to its large amount hydropower and nuclear power. However, even with the Swedish mix, electricity remains the dominating factor for most impact categories.

4.2.7 Evaluation - Aggregation of LCA results

Figure 4.32 shows absolute values for each of the investigated reuse treatment-trains (AG1 to I3) for all of the considered environmental impact categories (KPI) and for the three different plant sizes, 20 000 pe, 100 000 pe, and 500 000 pe as presented by Baresel *et al.* (2015b). The cell coloring further provides information about how the environmental impacts of each treatment train changes relative to the first train, i.e. AG1, shown for each plant size.

Plant size: 20 000 pe

KPI	Reuse treatment train							
	AG1	AG2	GW1	GW2	GW3	I1	I2	I3
GWP (10^{-1})	5.89	5.87	3.48	4.12	4.16	3.75	3.76	3.48
AP (10^{-3})	1.08	1.07	1.30	1.57	1.59	1.42	1.43	1.31
EP (10^{-3})	0.07	0.07	0.09	0.11	0.11	0.10	0.10	0.09
POCP (10^{-3})	0.29	0.34	0.30	0.37	0.32	0.31	0.31	0.31
HTP (10^{-3})	15.11	15.48	21.47	26.91	26.56	23.66	24.96	22.52
FAETP (10^{-3})	0.72	0.74	0.91	1.17	1.16	1.00	1.12	1.01
MAETP (10^{-1})	36.3	36.0	45.1	55.7	56.4	49.7	50.9	45.8
TETP (10^{-3})	0.28	0.28	0.52	0.61	0.60	0.59	0.57	0.53
AD, elements (10^{-6})	0.12	0.12	0.27	0.35	0.33	0.33	0.31	0.28
AD, fossil (10^{-1})	33.4	33.0	40.1	48.6	49.1	43.7	43.8	40.2

Plant size: 100 000 pe

KPI	Reuse treatment train							
	AG1	AG2	GW1	GW2	GW3	I1	I2	I3
GWP (10^{-1})	5.27	5.25	2.50	3.14	3.21	2.83	2.81	2.51
AP (10^{-3})	0.81	0.81	0.88	1.16	1.19	1.03	1.02	0.89
EP (10^{-3})	0.05	0.05	0.06	0.08	0.08	0.07	0.07	0.06
POCP (10^{-3})	0.27	0.32	0.27	0.34	0.29	0.28	0.28	0.27
HTP (10^{-3})	11.39	11.79	15.36	20.09	19.99	18.02	18.58	16.51
FAETP (10^{-3})	0.54	0.55	0.62	0.86	0.87	0.74	0.82	0.72
MAETP (10^{-1})	27.3	27.0	30.7	40.5	41.7	36.1	36.3	31.4
TETP (10^{-3})	0.21	0.21	0.41	0.50	0.50	0.50	0.47	0.43
AD, elements (10^{-6})	0.08	0.10	0.22	0.27	0.26	0.27	0.25	0.23
AD, fossil (10^{-1})	25.5	25.2	27.6	36.2	36.9	31.9	31.7	27.7

Plant size: 500 000 pe

KPI	Reuse treatment train							
	AG1	AG2	GW1	GW2	GW3	I1	I2	I3
GWP (10^{-1})	5.11	5.12	2.21	2.84	2.90	2.57	2.52	2.23
AP (10^{-3})	0.75	0.75	0.77	1.03	1.06	0.93	0.90	0.78
EP (10^{-3})	0.05	0.05	0.05	0.07	0.07	0.06	0.06	0.05
POCP (10^{-3})	0.26	0.31	0.26	0.33	0.28	0.27	0.27	0.26
HTP (10^{-3})	10.4	11.0	13.6	18.0	17.8	16.4	16.4	14.6
FAETP (10^{-3})	0.49	0.51	0.53	0.76	0.76	0.65	0.71	0.62
MAETP (10^{-1})	25.1	25.1	26.6	36.0	37.0	32.4	31.8	27.4
TETP (10^{-3})	0.19	0.20	0.38	0.46	0.46	0.47	0.42	0.39
AD, elements (10^{-6})	0.07	0.09	0.21	0.25	0.24	0.26	0.23	0.21
AD, fossil (10^{-1})	23.6	23.6	24.0	32.4	33.1	28.7	28.0	24.3

Legend:
Difference compared to AG1

- <50 %
- ≥50 - <80%
- ≥80 - <95%
- 100 ± 5%
- >105 - ≤120%
- >120 - ≤150%
- >150 - ≤200%
- >200%

Key:
- GWP - Global Warming Potential; AP - Acidification Potential; EP - Eutrophication Potential; POCP - Photochemical ozone creation; HTP - Human Toxicity; FAETP - Freshwater Ecotoxicity; MAETP - Marine Ecotoxicity; TETP - Terrestric Ecotoxicity; AD - Abiotic Depletion (elements; fossil)
- AG – Agriculture reuse; GW – Groundwater recharge reuse; I – Industrial reuse

Figure 4.32. Environmental KPIs for all investigated treatment trains, absolute values (Baresel *et al.*, 2015b).

Higher effluent qualities (GW and I) do generally imply an increased environmental impact. For Global warming potential (GWP), however, a significant decrease is observed when going from agricultural (AG) to industrial (I) reuse or groundwater recharge (GW) (from green colour to blue). This is because significantly larger amounts of nitrous oxide (N_2O) are emitted from the process in the less advanced AG-treatment mode (see chapter 0) due to incomplete nitrogen removal compared with a complete nitrification/denitrification for groundwater recharge and industrial applications. The emission of N_2O outweighs lower energy consumption required to accomplish partial nitrogen removal from the wastewater. At lower N_2O emissions, energy consumption becomes the dominating effect resulting in an increased GWP with increasing effluent quality (GW and I lines). However, for some KPIs as e.g. Terrestric Ecotoxicity (TETP) and Abiotic Depletion (AD) of elements, higher effluent quality implies a very significant increase of impacts from treatment processes for all plant sizes, in many cases more than 200% (marked with red colour).

Furthermore, most KPIs increase with increasing effluent water quality while the plant size is kept constant, while most environmental impacts are reduced as the plant size increases. For more detailed information and discussion of aggregated results, see Baresel *et al.* (2015b).

In the evaluation of the results, also a normalization against two common datasets was done; the EU25+3 (Figure 4.33) and World (Figure 4.34) average emissions Worth noticing, the impact category POCP has the largest impact for all lines when normalized to EU25+3 average, mostly caused by manufacturing of polymer and electricity consumption. However, when normalized to world average emissions, Marine Aquatic Ecotoxicity has the largest impact for all lines.

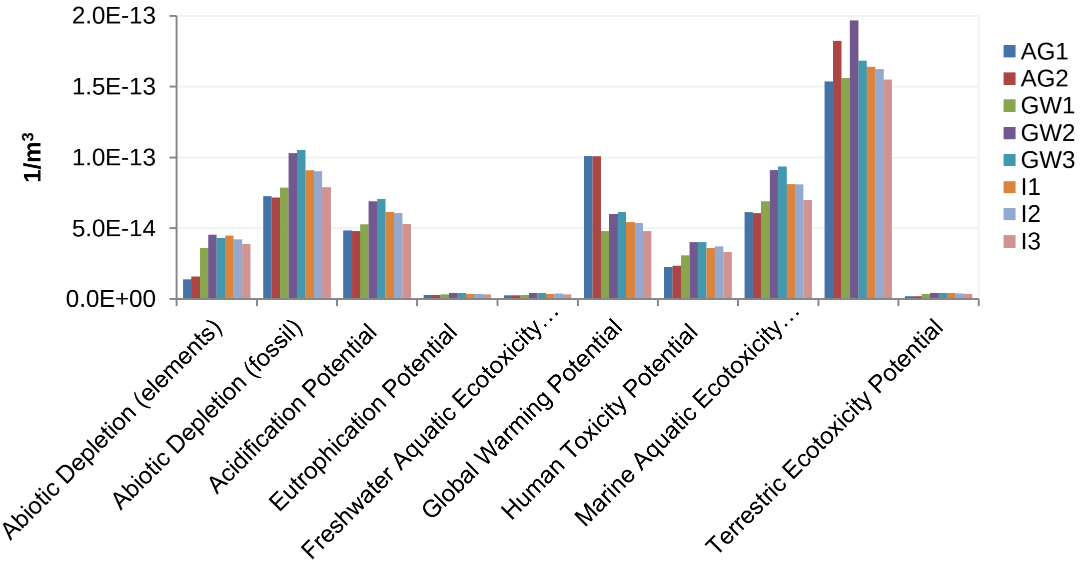

Figure 4.33. Environmental impact of the investigated treatment trains for a plant size of 100 000 pe normalized to the EU25+3 average emissions data set.

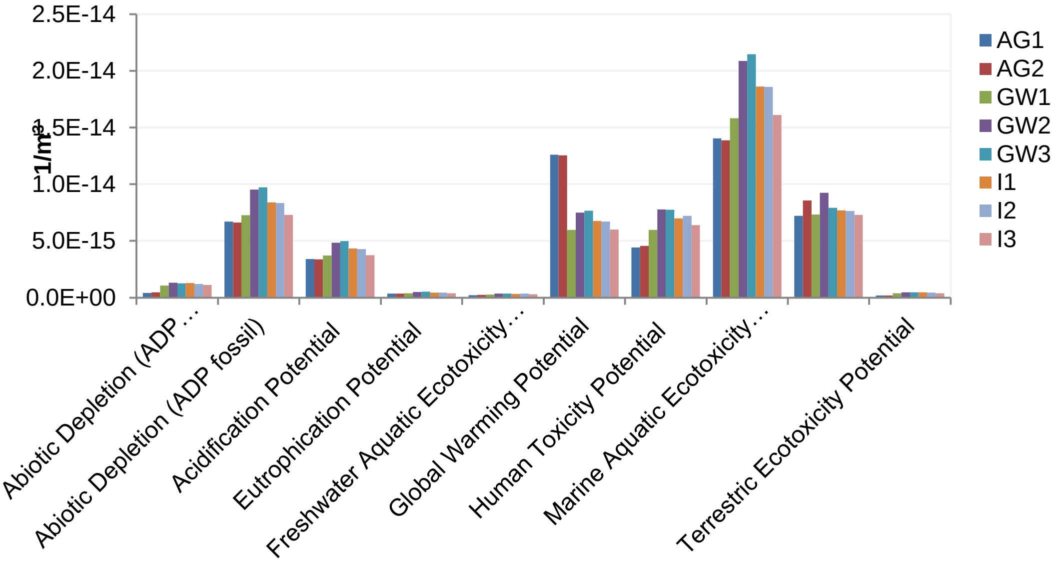

Figure 4.34. Environmental impact of the investigated treatment trains for a plant size of 100 000 pe normalized to the World average emissions data set.

4.2.8 **Comparison to traditional treatment**

The focus of this project was the comparison of environmental impacts of different treatment systems to achieve various reuse effluent water qualities. As these systems are designed for achieving reuse water qualities, they all include more complex treatment processed compared to traditional wastewater treatment. Because of that and because of the fact that environmental impact assessment of traditional wastewater treatment plants are rare in the same comprehensive way as done here, a comparison with traditional treatment plants is difficult. However, a reported study on a WWTP by Pasqualino *et al.* (2009) that has both a comparable size (114 000pe), that is located in the same region as considered here, i.e. Spain, and that includes several of the impact factors as used here may be used as a baseline scenario for comparisons.

Considering only the operation of water treatment processes, i.e. excluding onsite sludge handling, and a total estimated annual energy use of about 2 580 MWh was reported including operation of screen, sand/grease separator, primary and secondary clarifier, and the active sludge process. This can be compared to e.g. the AG1 train of the ReUse-project that requires about 2 025 MWh for the complete treatment system. Environmental impacts may also be compared, but due to limitations in reported studies, the impact of construction has to be excluded.

Table 4.24 provides a comparison of AG1 with the baseline case for some KPI:s including or including GHG-emissions from the treatment process.

Table 4.24. Comparison of selected environmental impact factors for AG1 and a baseline scenario.

	Baseline	**AG1**	**AG1**
	GHG-emissions excluded	GHG-emissions excluded	GHG-emissions included
Acidification Potential (AP) [kg SO_2-Equiv.]	1.58E-03	3.91E-04	4.20E-04
Eutrophication Potential (without N_2O) [kg Phosphate-Equiv.]	8.92E-05	3.91E-04	2.82E-05
Global Warming Potential, excl. biogenic carbon (GWP 100 years) [kg CO_2-Equiv.]	1.76E-01	9.06E-02	4.38E-01
Depletion of abiotic resources [kg Sb-eq.]	1.28E-03	2.85E-08	5.18E-08

It becomes clear that the total environmental impact of the optimized ReUse-system AG1 is lower than for the baseline scenario representing traditional treatment. Even if considering emissions of greenhouse gases during the biological nitrogen removal, which is highest for AG1 system (see section 0), the traditional treatment system has a higher environmental impact for all KPI:s except for GWP. The latter is an effect of the considered greenhouse gas emissions that were not considered in the baseline scenario.

4.3 Life Cycle Cost Assessment of investigated reuse applications

4.3.1 Overall results

In Figure 4.35, CAPEX, OPEX and overall LCC results for the 8 investigated treatment trains at a size of 100 000 pe are summarized. All results are calculated assuming a 20 years lifetime of plants and presented as $/m³ of treated wastewater and then normalized to (divided by) Agriculture line 1 (AG1) to present a difference in the cost from AG1.

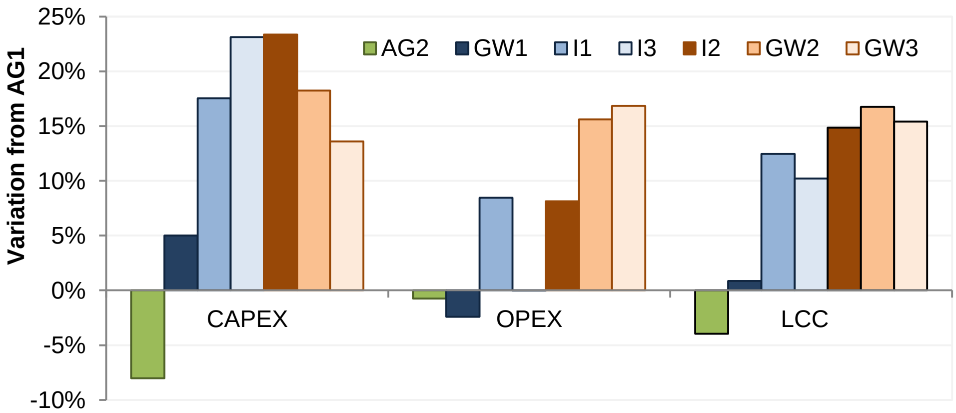

Figure 4.35. LCC results for all treatment lines, normalized to AG1 for a plant size of 100 000 pe (with baseline AG1 costs of CAPEX = 0.13 $\$/m^3$, OPEX = 0.17 $\$/m^3$ and LCC = 0.3 $\$/m^3$).

As it can be seen, investment costs (CAPEX) are not directly a function of the increased effluent quality the different ReUse-treatment lines imply. As the effluent quality increases from AG to lines for Industrial reuse (I), and Groundwater recharge (GW), CAPEX changes in different manner. For example, the removal of micropollutants achieved with lines GW2 and GW3 (highest water quality as discussed in Chapter 4.1.6) can be obtained at a CAPEX lower than that of I2 involving membranes that is also achieving removal of micropollutants. On the other hand, operation costs (OPEX) are generally a function of the effluent quality, as there is a trend of increased OPEX with increased effluent quality as for example Industrial and Groundwater recharge lines illustrate. This is most likely as the energy consumption needed to obtain higher effluent quality is higher. Combining CAPEX and OPEX gives the Life Cycle Costs (LCC), here for the selected lifetime of 20 years. It can be seen that an increase of LCC is followed with increasing effluent quality requirements.

4.3.2 Reuse application: Agriculture use

Using the LCC methodology, results for the two agriculture reuse lines at all three studied full-scale sizes were calculated and are presented in Table 4.25 and Figure 4.36. It can be seen that for the smallest size (20 kpe) AG2 (with DF) has a lower LCC cost (8% difference) than AG1 (with RGSF). For the middle size (100 kpe) and the largest size (500 kpe), the difference between AG1 (with RGSF) and AG2 (with DF) has been reduced to almost zero (from 4% to 0.4%).

Table 4.25. Costs of line AG1 per m^3 of treated wastewater.

Costs ($/m^3)	20 000	Plant size pe 100 000	500 000
CAPEX	0.25	0.13	0.09
OPEX	0.24	0.17	0.12
LCC	0.49	0.30	0.21

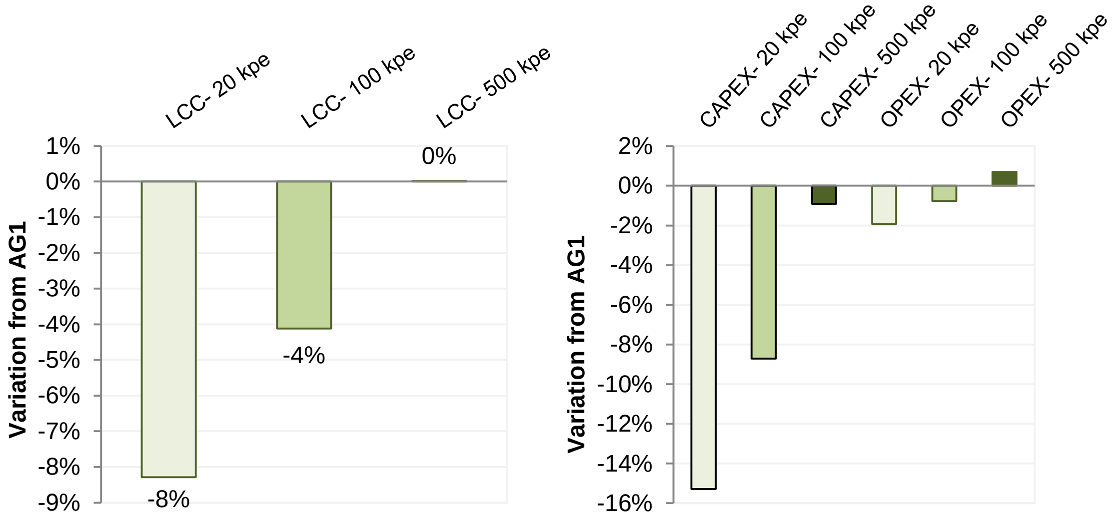

Figure 4.36. LCC (left), and CAPEX and OPEX (right) results for the two lines achieving agriculture effluent quality, normalized to AG1.

Generally, AG2 has both lower OPEX and CAPEX than AG1 (Figure 4.36). The difference in OPEX and CAPEX between the AG-lines decreases as the plant size increases. Differences between the two AG-lines in OPEX costs are very small but a shift can be seen for the largest plant size where AG2 has higher OPEX than AG1.

4.3.3 Reuse application: Industrial reuse

Investment cost (CAPEX), operating cost (OPEX) and the overall LCC for industrial reuse lines I1-I3 are shown in Table 4.26 and Figure 4.37 after being normalized with industrial I1 (with pUF). It can be seen that for the same effluent quality in terms of solid removal, I3 (with sUF) has lower LCC only for the medium size of the plant. Both 20 kpe and 500 kpe have lower LCC when using line I1. In addition, I3 has higher CAPEX for all sizes, while it has lower OPEX for all sizes comparing to I1.

I2 that has ozonation instead of UV is achieving micropollutants removal. Figure 4.37 indicates that to reach this higher effluent quality implies a higher LCC cost. This is only due to higher investment costs. Operating costs of the line are almost identical to I1, which is because an increased energy consumption of ozone for micropollutants removal compared to UV is offset by lower energy consumption of sUF compared to pUF.

Table 4.26. Costs of line I1 per m^3 of treated wastewater.

Costs ($/m^3)	20 000	Plant size pe 100 000	500 000
CAPEX	0.26	0.15	0.11
OPEX	0.26	0.18	0.13

LCC	0.52	0.33	0.25

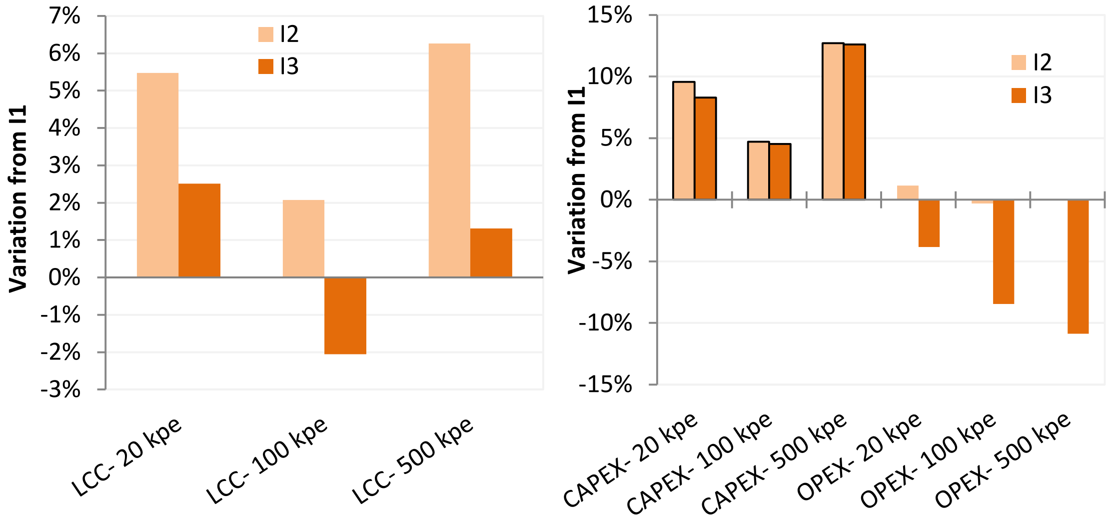

Figure 4.37. LCC (left), and CAPEX and OPEX (right) results for the three lines achieving industrial effluent quality, normalized to I1.

4.3.4 Reuse application: Groundwater recharge

Reuse lines achieving groundwater recharge effluent quality GW1-GW3 are also, similarly to industrial lines, divided in two different effluent target groups. One achieving "only" groundwater recharge limits (GW1) and the other group achieving groundwater recharge quality with lower COD levels AND micropollutants removal (GW2 and GW3).

In Table 4.27 and Figure 4.38, results for LCC, CAPEX and OPEX for these three groundwater reuse lines for all three investigated plant sizes are summarized. As in the previous industrial example, higher effluent quality, here accomplished using ozone and BAF, requires higher investment and operational cost. The overall LCC cost for GW3 has 8 to 14% higher cost compared with GW1 and for all three sizes. Comparing GW2 and GW3 it can be concluded that adding of a DF (GW2) in order to decrease load to the down flow equipment did not paid off as capital investment is higher (increase of 5%) than the gain in operating cost (only 1% decrease).

In addition, it has been found (see Chapter 4.46) that for all three GW-lines and for the smallest size, CAPEX is the dominating cost, up to 52% of the total LCC. As the size of the plant increases OPEX in $/m³ is dominating, up to 58% of LCC.

Table 4.27. Costs of line GW1 per m^3 of treated wastewater.

Costs ($/$m^3$)	**Plant size pe** 20 000	100 000	500 000
CAPEX	0.26	0.14	0.10
OPEX	0.25	0.16	0.12
LCC	0.51	0.30	0.21

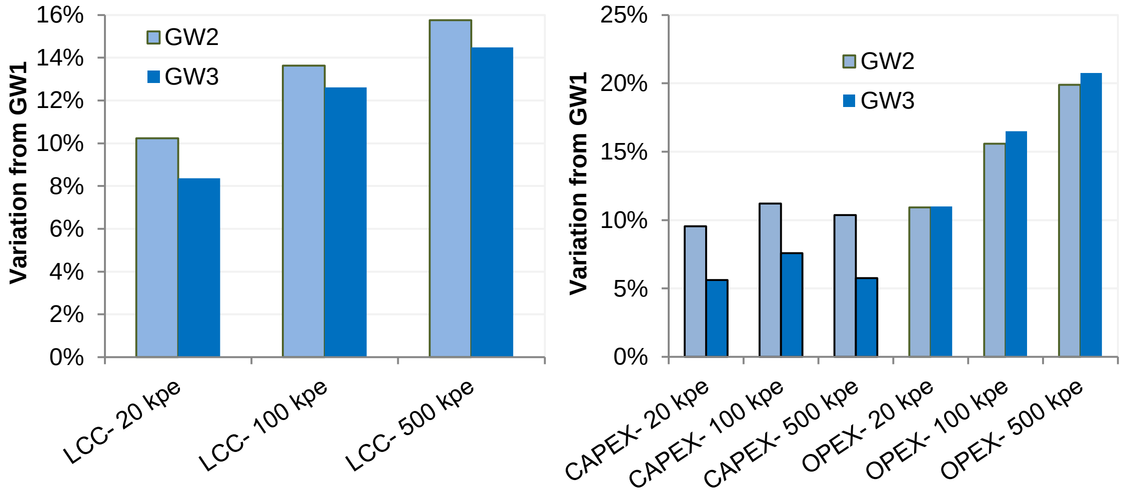

Figure 4.38. LCC (left), and CAPEX and OPEX (right) results for the three lines achieving groundwater recharge effluent quality, normalized to GW1.

4.3.5 **Impact characteristics**

In order to understand the reasons for the cost trends presented in the chapters above, the distribution of CAPEX and OPEX costs has been studied in more detail for an example of agriculture reuse line AG1 and for the medium plant size of 100 000 pe. AG1 CAPEX distribution per civil, mechanical and electrical including both water and sludge line is presented in Figure 4.39. It can be seen that the largest fraction, 57%, of Capital cost is due to the civil cost, and only 24% is due to mechanical (equipment) cost. Electrical cost accounts for 3% and 16% are due to other costs (overheads, buildings, preliminaries, commissioning and training). This means that the largest fraction of Capital investment cost of the whole wastewater treatment plant is due to civil cost of secondary treatment step.

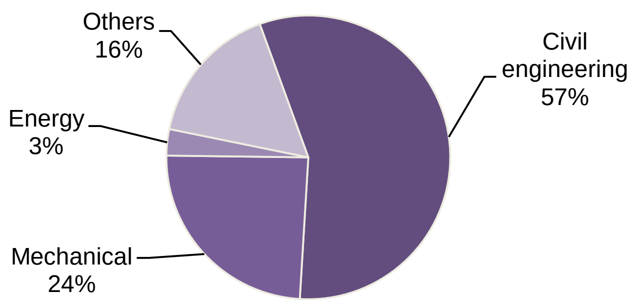

Figure 4.39. CAPEX distribution per civil, mechanical and electrical cost for AG1 and a plant size of 100 000 pe with a CAPEX = 0.14 \$/m^3.

On the Operating cost side (OPEX), it can be seen (Figure 4.40) that for this example, the two largest costs are energy consumption, that accounts for 51% of the Operating cost, and manpower cost (labor cost for operating the plant for 20 years) that is 29% of OPEX. In addition, secondary treatment (SBR) accounts for 50% of overall energy consumption, while sludge treatment (mostly due to aerobic stabilization) consumes 40% of overall energy.

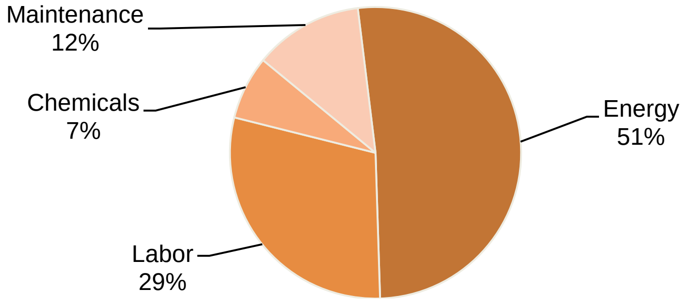

Figure 4.40. OPEX distribution per energy, labor, chemical and maintenance cost for AG1 and a plant size of100 000 pe with an OPEX = 0.16 $/m^3$.

4.3.6 **Plant size**

An example of the impact the increasing plant size has on the overall CAPEX and OPEX cost is provided in the Figure 4.41. As the size of the plant increases, the cost per m³ of treated wastewater decreases. CAPEX cost function decreases by 44% and 26% with increasing plant size. OPEX cost function decreases by 33% and 26% with increasing plant size. The reason for this lies in several factors. As the size of the plant increases, energy consumption per m³ of treated water decreases. As discussed previously, this is due to an optimization of energy consumption for larger plants and better efficiency of larger equipment. In addition, the civil cost per m³ decreases as well due to better optimization of the civil work for larger plants. Since energy consumption and civil cost are the two major contributors to the LCC cost as shown in Chapter 4.4.5, the overall LCC cost will follow the same trend.

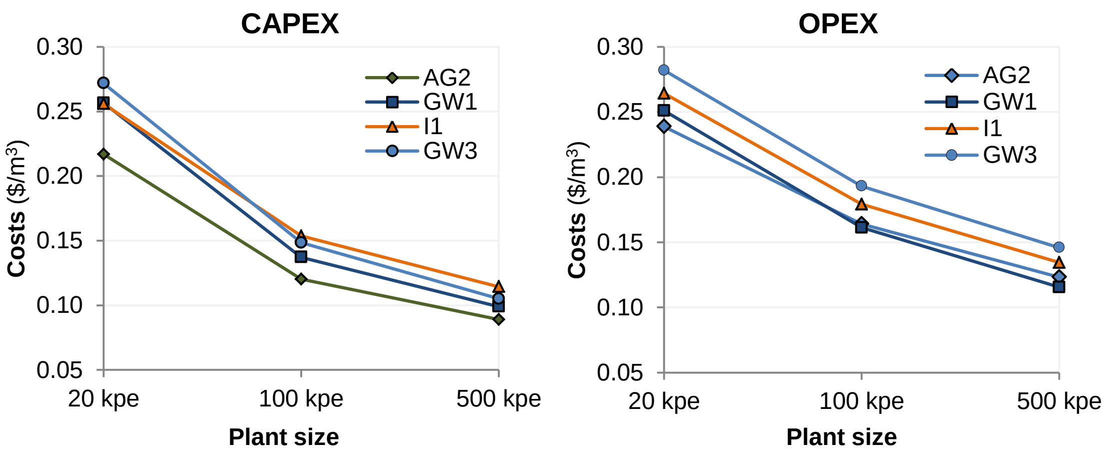

Figure 4.41. Impact of plant size on CAPEX (left) and OPEX (right).

In Figure 4.41 can be also seen that the OPEX cost is larger than CAPEX for the sizes 100 000 pe and 500 000 pe and for all evaluated lines. For AG-lines this is true even for the smallest size of 20 000 pe. This implies that the OPEX determines the trend of the overall LCC and not the CAPEX. This finding indicates that it is more sustainable to focus on the overall LCC costs for the whole lifetime and not focus too much on capital investment costs.

4.3.7 **Uncertainties and Sensitivity analysis**

As discussed previously in methodology part, intercompany prices were used for Xylem equipment, as the goal of the project was to compare LCC cost of our eight treatment trains and not to compare with other suppliers.

In addition, all prices are based on the Spanish prices, as the plant was situated in Spain. Therefore European energy prices, chemicals prices, person-hours price, etc. were used. These costs will be very different from region to region.

4.3.8 **Comparison to traditional treatment**

Comparing costs of different treatment systems is difficult as considered costs items included in CAPEX and OPEX usually differ significantly. Reported values in grey literature vary and include sometimes onsite sludge handling, transport of wastewater and waste disposal. Considering Gryaab WWTP, a Swedish large size treatment plant (approx. 700 000 pe) costs for water treatment are determined as 0.7 $/m^3$ (www.gryaab.se). Assuming that this also would include the CAPEX, total costs for this traditional treatment system would be higher than or equal to any of the costs of the presented ReUse-systems. Considering that presented costs here are net-costs provided by technology providers, real costs may increase with <100 percent. Comparing the 500 000 pe plant size of the ReUse-project, costs for producing water for different reuse applications will still be lower than reported costs for existing sewage treatment at e.g. Gryaab WWTP with lower effluent water quality.

4.4 Aggregation of treatment, sustainability and cost performance

Figure 4.42 describes the methodology used in the ReUse project; sustainable wastewater reuse requires a proper evaluation of treatment systems regarding their treatment efficiency (see Chapter 4.1), total environmental impacts (se Chapter 4.2), and total lifetime costs (se Chapter 0).

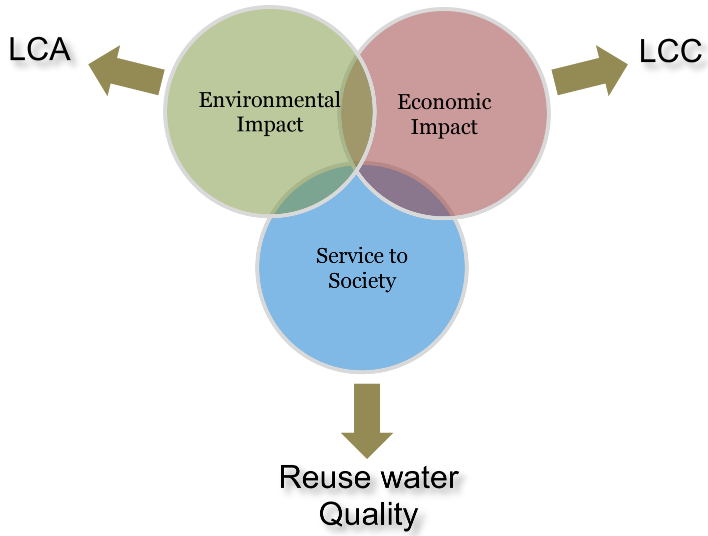

Figure 4.42. Main aspects to consider for wastewater treatment systems for reuse applications as investigated here.

However, sustainable treatment systems for wastewater reuse further requires an aggregation of the individual outcomes from the different evaluations of environmental impacts, costs and achieved water quality. This means to view wastewater treatment systems in the context of the sum of society's demands and needs. The most straightforward approach is to consider various aspects and results from each of the three evaluations and try to identify the treatment system that meets most of the predefined demands and requirements on an anticipated system.

Lazic *et al.* (2015) provides a discussion on how to aggregate results from the various evaluations and Figure 4.43 provides an example of the aggregated environmental impacts for all investigated treatment systems and information about key water quality differences. Out of this, preferred treatment systems may be selected for further analyses as shown in Figure 4.44. Based on the evaluation of the main impact causes and information on LCC (OPEX and CAPEX) as for example shown in Figure 4.41, the treatment train that best corresponds to conceptions may be selected.

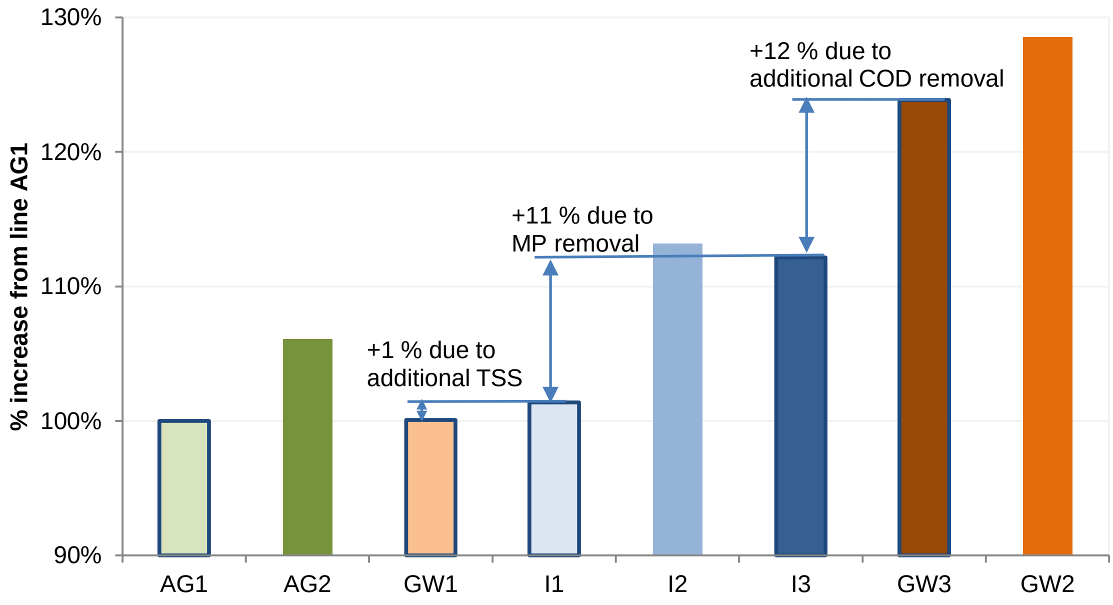

Figure 4.43. Aggregated environmental impacts of all investigated treatment systems for a plant size of 100 000 pe and normalized to EU25+3 average emissions.

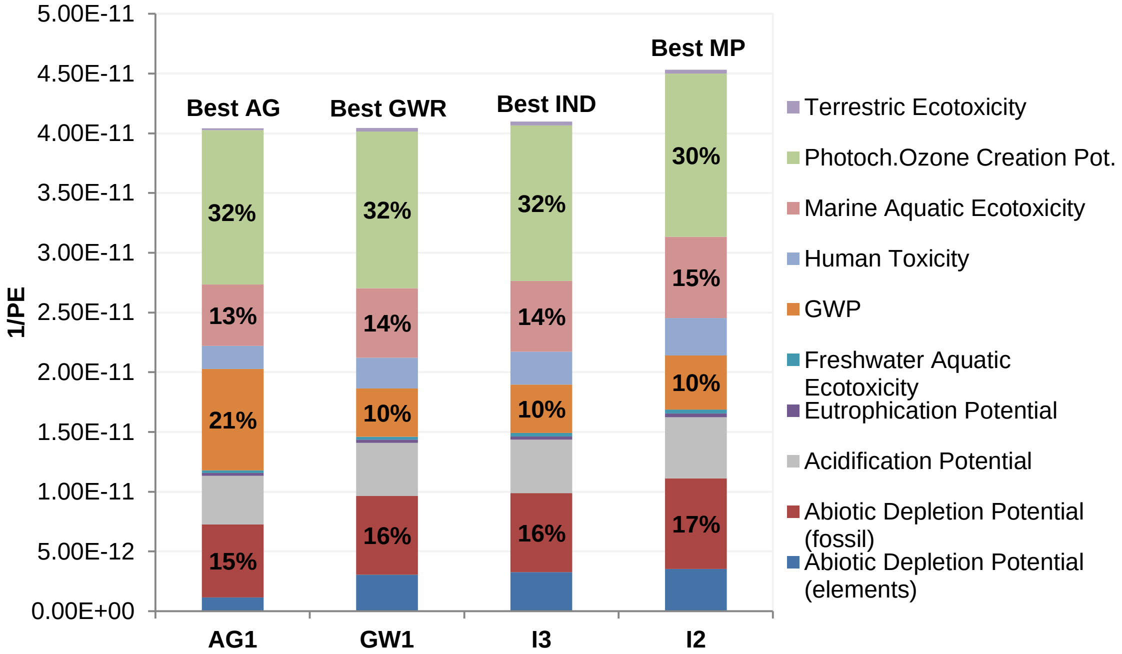

Figure 4.44. Aggregated impacts for preferred treatment systems and main causes of various impact categories (based on Figure 4.43).

The individual evaluation of environmental impacts, costs and achieved reuse water quality is the recommended approach as aggregation of results always implies loss of details, which may lead to biased or wrong decisions. However, even the approach presented above starts with an overall aggregation of main results in order to filter out the most relevant alternatives to go more into details with. Considering the enormous amount of results and detail, this is a necessary step in the evaluation process. Aggregation also implies combining various results and this becomes difficult when dealing with total different methodologies such as when talking about LCA, LCC and water quality.

Figure 4.45 provides an example of how the results of all three evaluation-categories, water quality, environmental impact and costs, could be combined for a first and non-scientific results aggregation.

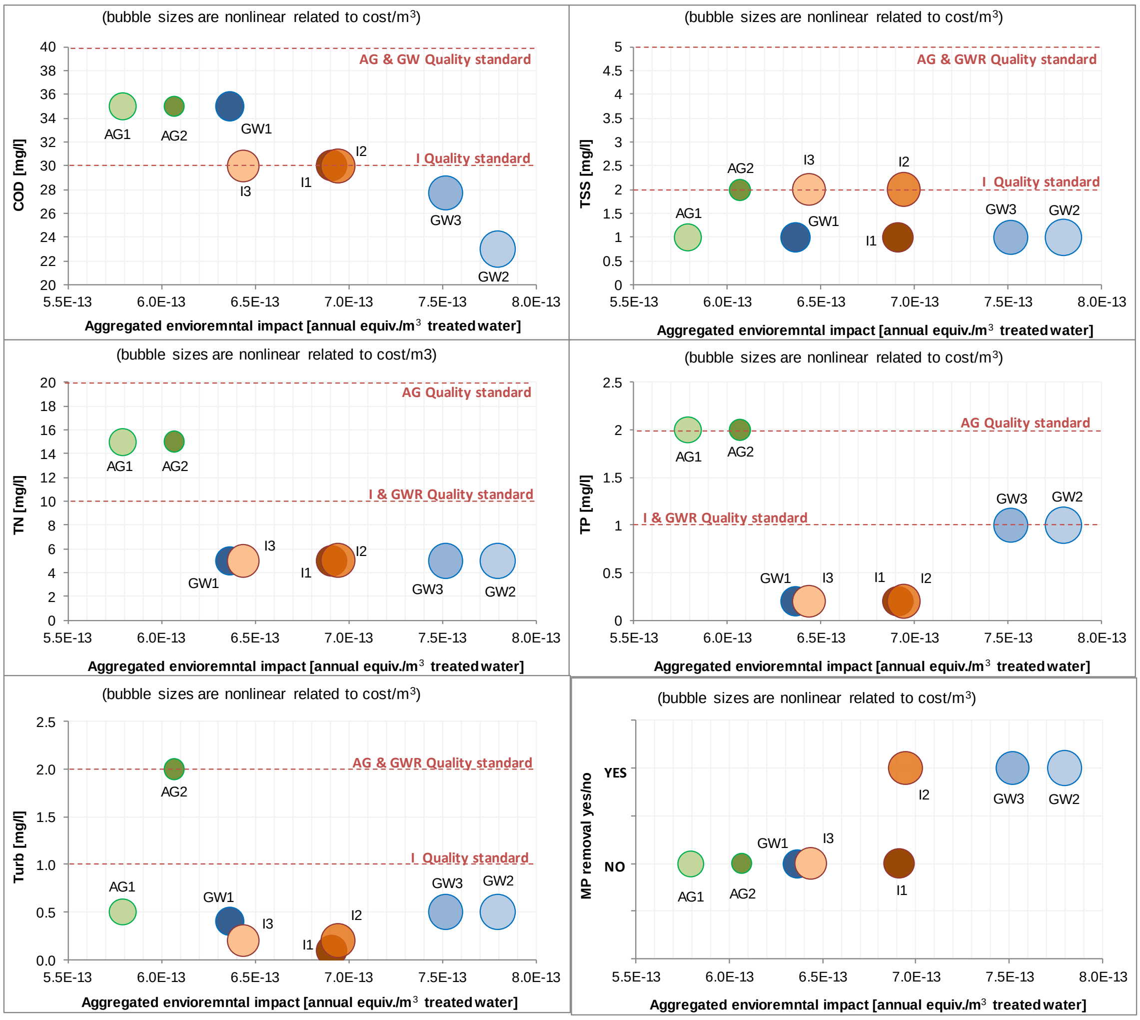

Figure 4.45. Aggregated LCA and LCC results for all investigated treatment systems for a plant size of 20 000 pe for various water quality parameters.

5 Optimization and process developments within the ReUse project

The presented evaluation and assessments of the various treatment trains are done on optimized treatment systems. These optimizations were within the scope of the project and based on the high number of tests with different process configurations within a whole train or just some process steps.

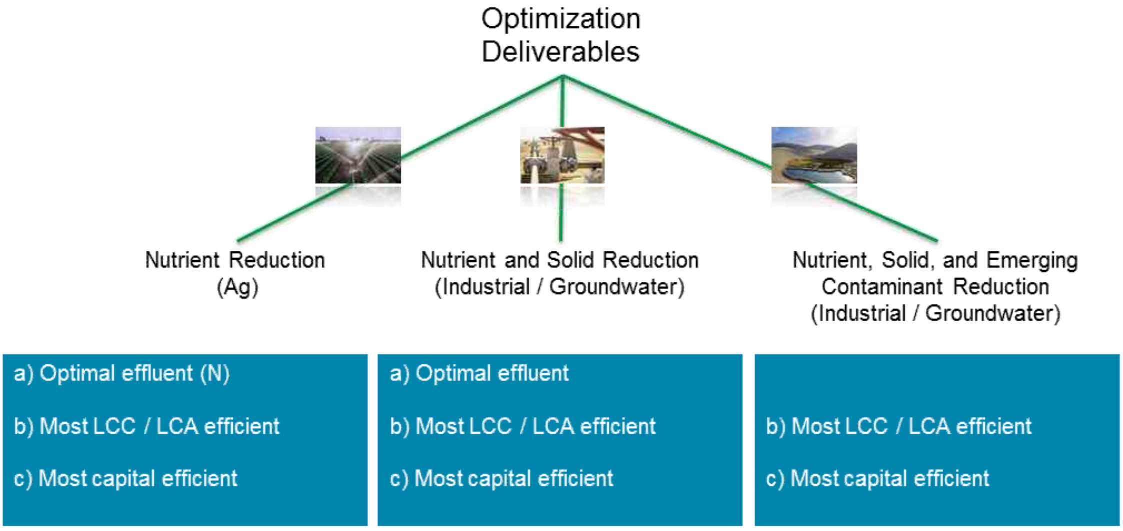

Figure 5.1. Optimization strategy used in the ReUse project.

An optimization may for example imply that the effluent quality of a certain process step may differ from the standard design if the following process can accomplish the quality requirements of the final effluent more resource efficient than if both processes would have been operated as in normal design mode. This becomes clear taking the example of different filtration technologies such as DF and RGSF to reduce effluent turbidity, which has an impact of the required energy used for UV disinfection as described by de Kerchove *et al.* (2015). Different treatment processes combined fulfill the same final effluent requirements at altering process operation and thus environmental impacts and costs.

Investigated treatment trains within the ReUse-project were established based on common design criteria for construction and operation for each treatment process. The combination of comprehensive pilot testing and the environmental impact and life cycle cost assessment facilitated to identify potential improvements to either reduce the overall environmental impact or cost at maintained effluent water quality.

One example illustrating a successful refined solution is shown in Figure 5.2 where the impact was a 9 % operating cost decrease and 5 % capital cost decrease when system solution thinking was applied to the overall treatment train. In this case due to over performing of individual unit processes during the entire tested period, the solution was found in optimization of both unit processes when operated together. This resulted in decreasing of the balancing tank size by 40 % when ICEAS is followed by RGSF. In addition, from the hydraulic profile and from the OPEX profile it was found that by lifting the RGSF feed pump and therefore decreasing the pump head, energy consumption will decrease and therefore the operating cost as well.

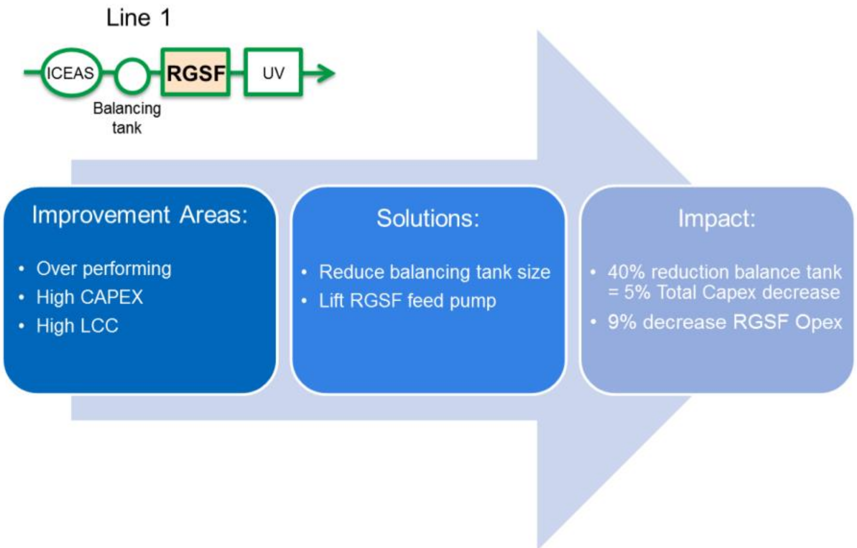

Figure 5.2. Optimization example.

Besides already included optimizations of treatment-systems described here, results from treatment performance, LCA and LCC point out most processes and equipment that further optimization may focus on. This may include technical improvements, changes in upstream processes, use of more environmentally friendly alternatives, etc. Much focus should e.g. be on the secondary treatment as results indicate a dominating impact on the overall treatment system impact. Any reduction in the ICEAS size or energy consumption would strongly affect both LCA and LCC (see also Chapter 4.2 and 4.3).

5.1 Oxelia - Ozone Enhanced Biological Active Filtration

The investigation of operating parameters for ozonation and BAF processes for efficient removal of total organic carbon or chemical oxygen demand (COD), and micropollutants (MP), after the process units of a sequential batch reactor and a pre-filtration unit provided comprehensive information for the development of a combined process unit for these two techniques. Xylem's Oxelia system combines the treatment synergy of ozone oxidation and biologically active filtration into a single process solution (Figure 5.2).

Comparing with no pre-filtration, we identified that pre-filtration with DF has no effects on the process performance for the ozone demand and biofilter performance. Even though UF as pre-filtration reduced the absolute ozone demand because of the reduced COD concentrations after UF, the levels of final COD and the removal efficiencies of trace organic compounds were similar to that without pre-filtration. Generally, in an ozone enhanced biofiltration system, ozone shows high efficiencies for the removal of trace organic compounds and BAF (loaded with GAC) exhibited polishing effects.

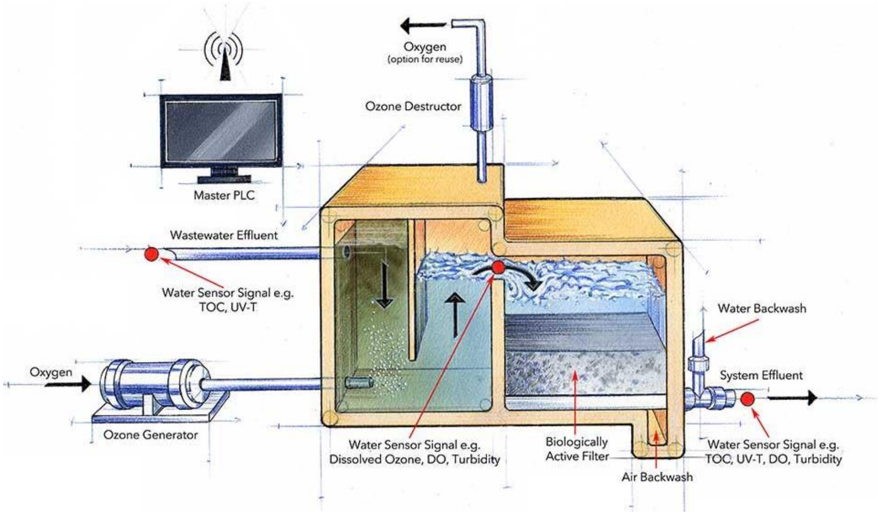

Figure 5.2. Ozone Enhanced Biofiltration system.

5.1.1 Ozone Enhanced Biofiltration with DF as Pre-filtration or no Pre-filtration

Generally, an increasing O3/TOC ratio increased the COD removal efficiency. It appeared that an O3/TOC ratio of 0.8 was the optimal ozone dose for COD/TOC removal with the combined ozone and biofiltration without pre-filtration. The overall removal was about 50% on the average. GAC and anthracite behaved similarly for COD removal and UVT removal. AOC was found to increase with an increasing ozone dosage. UVA removal increased with an increasing ozone dosage. Adenosine triphosphate (ATP) and extracellular polymeric substances (EPS) contents in GAC media were significantly higher than that in anthracite. It was confirmed that the top layer of the media harbored majority of the biomass. Anthracite was not effective for the removal of MPs while GAC effectively polished the residual MPs.

5.1.2 Ozone Enhanced Biofiltration with UF Pre-filtration

The optimal O3/TOC ratio was 0.8 for the removal of COD of the combined O3 and BAF with UF as pre-filtration. The removal efficiency was about 30% on the average for the combined ozone and BAF system. AOC did not increase with an increasing ozone dosage. For some compounds such as Ibuprofen, Citalopram, Ciprofloxacin, hydrochlorothiazide, Metoprolol, and Oxazepam, the removal efficiencies with UF as pre-filtration appeared negatively affected. No obvious advantages were observed with UF as pre-filtration for the removal of TOC and MPs.

5.1.3 Adsorption Contactor

It took about one month (Bed Volume about 5000) to exhaust the absorbtion capabilities of the brand new GAC with typical wastewater in terms of TOC/COD saturation. GAC was better than anthracite for the removal of MPs. GAC may need frequent replacement if high removal rates of the TOC and MPs is required by absorption effect on the activated carbon.

6 Overall conclusions and recommendations on reuse sustainability

The main overall conclusion from the presented combined assessment of treatment performances, environmental impact and life cycle cost of a number of treatment systems of different plant sizes based on state-of-the-art technologies and archiving various wastewater reuse quality requirements is that only a consideration including all of these factors can be the way forward to sustainable wastewater reuse systems.

6.1 Treatment performance

Treatment performance in terms of reaching a certain effluent quality by removal of a number of common but also emerging contaminants, and process stability was a starting point and prerequisite for sustainability evaluation. This project has demonstrated that the right technologies can be efficiently combined to meet local regulations and requirements, and guarantee that the solutions work.

The novelty in the ReUse-project, however, is the approach to shift focus from individual processes to treatment systems (here called trains) while not losing single process importance. This implies that the risk for sub-optimizations is avoided. This means overall system optimization based on the whole system assessment that guaranty best value-for-money. This may then for example include optimization of one single process, under- or over-performance of single processes, or changed process design depending on the assessment results.

6.2 Environmental impacts

The project further illustrated that increased environmental impacts, caused by more stringent effluent quality targets that require more advanced treatment processes, become less significant with increasing plant size. This implies that higher quality targets do not automatically imply an increase of environmental impacts. Instead, poorer water treatment can increase the total environmental impact if considering the treatment process and up- and downstream effects. Agriculture reuse of treated wastewater, for example, may not be the most favoured reuse approach with the selected technology. Despite its less stringent effluent quality to recycle nutrients, the impact of the treatment as such can imply a higher environmental impact for some impact categories than treatment systems for reuse applications requiring higher effluent qualities such as groundwater recharge and industrial reuse.

It became also clear within the environmental assessment that it would be desirable to consider all downstream impacts, i.e. after the actual water and sludge treatment, which have not been considered in this project. These may add additional impacts, both negative (toxicity, heavy metals etc.) or positive (nutrient, decreased utilization of fresh water) to the assessment. In a wider context, not only comparing technical systems as done in this study but also all emissions to land, water and atmosphere, a complete sustainability assessment could be performed. This would then also include avoided impacts by, e.g. reduced need of artificial fertilizers, use of clean natural water sources and possibly use of bio-energy. However, such impacts may not be quantifiable in the same way as for the technical system and they would thus make a comparison of different technical systems more difficult, which is also the main reason why they were not included in this study.

The main contributing factor identified for most of the investigated treatment systems was the use of energy for nutrient removal and sludge stabilisation. This study, using actual measurements of nitrous oxide emissions from biological nitrogen removal processes in environmental impact assessment, indicates that the energy savings when targeting lower effluent criteria may be outweighed by the increased release of nitrous oxide to the atmosphere. However, high nitrous oxide emissions and their impact may potentially be reduced by an optimisation of the ICEAS mode, which has not been done within this study.

The study further showed that external conditions could have a larger impact on the overall environmental performance of reuse treatment systems than internal optimizations can compensate. These results imply that reuse schemes could become less environmental impactful if the wider context or system boundaries is considered and possible impact-reducing measures are taken.

Compared to tradition wastewater treatment systems, using one baseline scenario, the investigated reuse-systems are strongly competitive and at the same time obtaining reuse water quality. Unfortunately, complete environmental impact assessments of existing treatments are rare.

6.3 Life cycle costs

Economic evaluation of eight studied reuse solutions showed that investment costs (CAPEX) of different treatment systems are not directly related to an increased effluent quality. This means that higher water quality does not necessarily implies higher investment cost. On the other hand, operating costs (OPEX) are generally increasing with increasing effluent quality as the energy needed to remove additional contaminants will increase as the quality increases. In addition, it was also shown that the sum of investment and operating costs over a whole lifetime, i.e. the Life Cycle Costs per m^3 of treated wastewater, decreases as the size of the plant increases. The impact of size of the plant lies in higher efficiency of large systems both from the performance standpoint and from the construction stand point.

Only when evaluating the overall LCC it becomes apparent that for some plant sizes the operating cost is the dominating cost over the whole 20 years of the plant's lifetime and not the investment (capital) cost. This finding strengthens the need for LCC evaluation and LCC approach when funding a new project in order to take a more sustainable solution over the life length of the plant.

It was also shown that individual processes can have a significant impact on the overall treatment train costs and the LCC assessment provides a helpful tool to identify specific components or processes with high costs. For example, the secondary biological treatment step is the largest fraction of both CAPEX and OPEX. The reason behind this lies in high civil cost for concrete basins on the CAPEX side, and high-energy demand and thus energy cost on the OPEX side. This means that optimizing secondary treatment step in terms of decreasing its footprint and energy consumption will have a huge impact on the overall LCC of the whole Reuse line. This is equally true for new builds (green filed) as well as for retrofitting of existing plants.

LCC evaluation of different ReUse trains also revealed that both tertiary filtration and disinfection steps contribute only by few % to the overall LCC of a treatment train, even though these steps are responsible for increasing the water quality to the Reuse effluent quality standards. In this evaluation the economic benefit of reusing of the water was not taken into account.

6.4 Sustainable reuse solution

The sustainable and optimal reuse solution is defined as an intersection of the environmental, economic and social dimensions. However, the results show that there can be more than one optimal solution when constraints such as capital are introduced. The results also show that the optimal solution varies both by application, and by plant size. Figure 6.1 shows that for a 20 000 pe agriculture quality effluent the target can be reached but there is a 6% higher environmental impact for the capital efficient solution relative to the more sustainable solution which produced a higher quality of water without the use of chemicals. The same comparison between the most sustainable solution and the most capital efficient solution will give different results when the size of the plant increases to 100 000 pe and 500 000 pe.

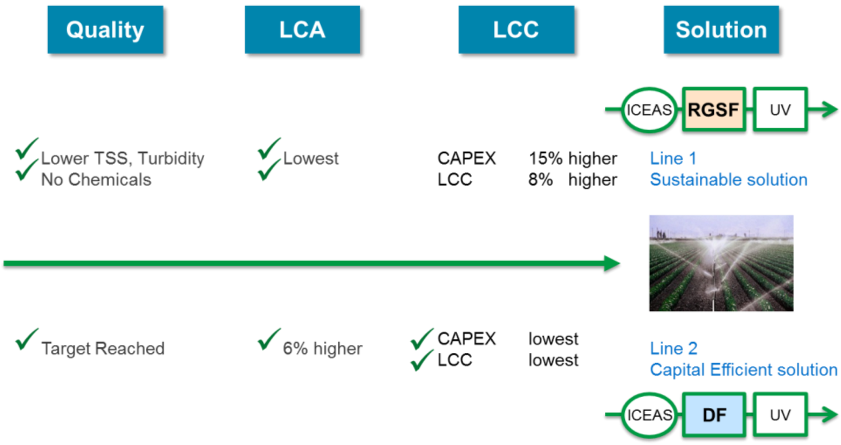

Figure 6.1. Best Overall Solution

When taking an overall holistic view the optimal reuse solution alternatives for a particular situation vary based on plant size and flexibility desired. The tradeoffs between performance, economic, and environmental impacts can be quantified and evaluated. The solution alternatives can provide high quality effluents, which protect public health and the environment, while being optimized for lowest capital or life cycle cost.

Holistic Wastewater ReUse solutions require the simultaneous evaluation of treatment system quality, environmental impacts and cost analysis.

7 Future work

The goals of the ReUse project were achieved and water treatment processes and systems for sustainable reuse of treated wastewater in the community were further developed and optimized. With the help of a pilot plant that includes more than 10 state-of-the-art technologies, extensive testing for removal of unwanted substances, modeling of treatment processes and life cycle analysis for each of the treatment lines has resulted in a comprehensive basis for selection and dimensioning of purification techniques that are tailored to regional needs and circumstances.

The ReUse project also identified a number of dominant processes that can both be affected and that can provide much better resource utilization and reduced environmental impact. A continuation of the Reuse project will thus focus on these items using the existing but adapted pilot equipment on the R&D-facility Hammarby Sjöstadsverk. Based on experiences and results from the ReUse project, focus will primarily be on the optimization of secondary wastewater treatment. The main objectives are to optimize the ICEAS treatment processes for nitrogen reduction targeting non-potable water reuse applications worldwide, and to develop an empirical method to predict the performance. Further, to assess the treatment process alternatives in terms of sustainability and to define the performance of the suggested reuse offering in terms of phosphorus removal using all capabilities of each treatment when associated to conventional and emergent coagulation technologies.

In specific future work would have to include:

- Low/Ultra-low P technology assessment to determine limits of technology at bench scale for a) Ferric Chloride, and b) Ferric Chloride + Ozone, in order to achieve an effluent of less than 0.05 mg/L total P.
- Evaluate ICEAS as two reactors (PRZ and MRZ) to determine capacity of pre-react zone (PRZ) and main react zone (MRZ).
- Establish Mass balance and nitrification and denitrification rates for PRZ and MRZ to validate nitrification and denitrification capacities of baseline in pre-react and main react zones. This has been a problem in the ReUse project due to continuous adaptations of the treatment system.
- Optimize overall performance of the secondary treatment by evaluating changes in food, biomass, and air on nitrification and denitrification capacity.
- Evaluate operating cost and capital cost savings potential to confirm if savings in operating and/or capital cost can be achieved for new installations by identifying and process design alternatives based on the aforementioned work.
- Validate long term Performance of Optimized Nutrient (N and P) Removal Process to determine the robustness of the optimal solution obtained by running the optimized process for an extended period to generate requisite date to establish statistical averages for effluent and sludge quality (i.e. 95% vs. monthly average).
- Life Cycle Assessment of Optimized Nutrient (N and P) Removal Process to assess the optimized process treatment train using previously developed IVL Sustainability Assessment Framework and LCA models to assess considers social, environmental, and economic sustainability factors.

- Evaluate High-Rate Bio Processes to determine if a high-rate small-footprint biological process can be implemented in continuous inflow SBR which would add nutrient removal capacity.
- This study provided guidance for process design for indirect and direct potable water reuse of treated wastewater, and wastewater discharge into sensitive water bodies where trace organic compounds are to be regulated such as in some regions of Germany and Switzerland. Other media such as pumice, expanded clay, and synthetic media should also be investigated to compare with GAC, aiming to provide a cost effective alternative. While the combined ozone and BAF system is a cost effective alternative to reverse osmosis, an intergraded control strategy and platform during design and process operation should be considered.

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9 Appendices

9.1 Inventory methodology

Table 9.1. Data sources for construction materials and excavation.

Material	Data source	Area	Ref. year	Further comments
Ceramics (ozonator)	Ecoinvent	Germany	1999	Data for refractory bricks, high in aluminium oxide
Concrete, C30/37	ProfDB	Germany	2001	Density used: 2400 kg/m^3
Electronics	Ecoinvent	Europe	2005	Electronics for control units (46% steel (housing), 32% plastics, 14% printed wiring boards and 8% cables (various types)).
Excavation work	Ecoinvent	Europe	2001	Hydraulic digger, diesel powered (uncertain data quality)
Glass (ozonator)	Ecoinvent	Germany	2001	Modelled as borosilicate glass tube
Glass reinforced plastic (GRP)	Ecoinvent	Europe	2000	Data for glass fibre reinforced plastic, polyester resin (based on assumptions) [1)] Density used: 1.7 kg/dm^3 [2)]
Indium (UV lamps)	Ecoinvent	Europe	2005	As by-product from zinc production.
Iron, ductile	Ecoinvent	Europe	2001	35 % scrap, 65 % raw iron as iron input, EAF process assumed.
Manganese dioxide	No data found.			Assumed catalyst in the ozone destructor
Mercury (UV lamps)	Ecoinvent	Global	2000	Liquid mercury. Large uncertainty due to weak data on the production process.
Polyester (filtration cloth, membrane casing)	ProfDB (PU Europe)	Europe	2008	Aromatic Polyester Polyol (APP) (European average, without flame retardant, based on DMT, PET, PA, PG and DEG) [3)] Assumed porosity of cloth: 50 % Assumed density: 1.4 kg/dm^3 [4)]
Polyether sulphone (pUF membranes)	ProfDB (Plastics Europe)	EU-25	2007	Density of PES: 1.37 kg/dm^3. Assumed porosity: 50 % Modelled as bisphenol-A. No data found for polyether sulphone. [5)]
Polyethylene pipes	ProfDB (Plastics Europe)	Europe	2005	Data for HDPE pipe.
Polyvinyl chloride parts	ProfDB (Plastics Europe)	Europe	2005	Injection moulded PVC.

[1)] In some cases (SBR and UV) GRP was modelled as glass fibre (Ecoinvent, average European data). In these cases this is judged to be of minor significance but should be corrected.
[2)] Assuming the density of polyester and 55 % E-glass.
[3)] DMT = dimethyl terephthalate, PET = polyethylene terephthalate, PA = phthalic acid, PG = polyethylene glycol, DEG = diethylene glycol.
[4)] Density of unreinforced PET.
[5)] Polyether sulphone is synthesised from a dihydroxybenzene, like bisphenol A or 4-hydroxyphenol, and bis(4-chlorophenyl)sulphone. No data has been found on polyether sulphone.

Table 9.2. Data sources for construction materials and excavation (continued).

Material	Data source	Area	Ref. year	Further comments
Polyvinyl chloride tube	ProfDB (Plastics Europe)	Europe	2005	Data for PVC pipe.
Polyvinylidene fluoride (sUF membranes)	Ecoinvent, ProfDB, USLCI	Europe	1979 - 2011	Density of PVDF: 1.78 g/cm^3. Assumed porosity: 50 %. No data found for PVDF. Modelled as PVF film (see figure 3.4.3.1).
Pumps, ≤ c. 20 kW	Flygt	Sweden	2010	Data from an EPD of pump 3153.181.
Pumps, ≥c. 20 kW	Flygt	Sweden	2010	Data from an EPD of pump 3301.180.
Quartz glass	A glassworks	Europe		No data found. Modelled as melting of quartz sand. Energy as for soda glass melting [6)]. Energy from natural gas.
Quartz sand, extraction	EDIP			Comprises use of explosives and of diesel oil in a small engine.
Reinforcement steel	ProfDB (Worldsteel)	Global	2007	Data for steel rebar. Mix of ore- and scrap-based production.
Stainless steel, 304 and 316	ProdDB (Eurofer)	Europe	2008	Cold-rolled coil, scrap-based production (EAF process).
Steel sheet	ProfDB (Worldsteel)	Europe	2007	Organic coated, ore-based (blast furnace)production.

[6)] Should be an underestimate. However, a theoretical calculation of the heat demand to heat quartz from 20°C to 2000°C including fusion yielded a lower value.

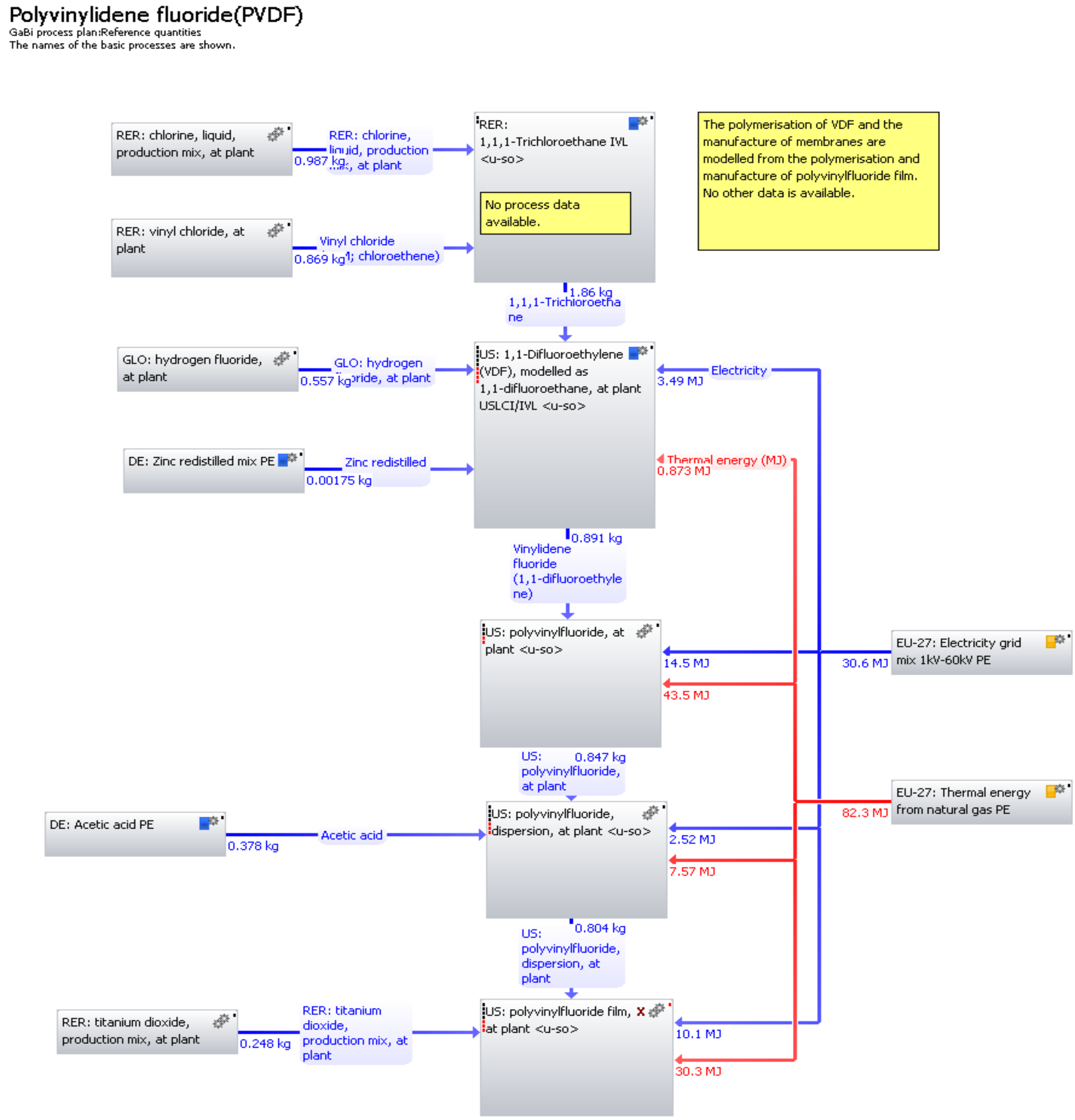

Figure 9.1. Modelled synthesis and manufacture of polyvinyl fluoride (PVF) film. The model is used to approximate the manufacture of polyvinylidene fluoride membranes.

Table 9.3 Transport assumptions.

Material	Distance km	Mode of conveyance	Payload capacity used % of max. load (weight)	Transport pattern
Concrete	30	Truck [1)]	50 (empty return)	Urban roads 100%
Electronics	7400	Freight aircraft [2)]	66	
	50	Truck [1)]	85	Motorways 50% Main roads 10% Urban roads 40%
Pumps	2400 [3)]	Truck [1)]	85	Motorways 90% Main roads 5% Urban roads 5%
All other materials	300	Truck [1)]	85	Motorways 70% Main roads 23% Urban roads 7%

[1)] Diesel truck, Euro 4, 5 % biogenic carbon and 10 ppm sulphur in the fuel, 14 – 20 t gross weight / 11.4 t payload capacity.
[2)] 400 ppm sulphur in the fuel (default value), 65 t payload capacity.
[3)] Emmaboda – Barcelona.

Data has been collected from the following sources:
Truck: ProfDB, global data, reference year 2011.
Freight aircraft: ProfDB, global data, reference year 2011.
Diesel oil: ProfDB, EU-27, diesel mix with 10 ppm sulphur and 5,75 % w/w biofuels at refinery, distribution transports have been neglected, reference year 2009.
Aircraft fuel: ProfDB, EU-27, jet A1 with 480 ppm sulphur at refinery, distribution transports have been neglected, reference year 2009.

Figure 9.2. Model of average Spanish electricity in 2012. Flows per MJ delivered electricity after grid losses.

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