Deliverable D1.4

Pretreatment requirements and design guidelines for SAT technologies, and two SAT case studies

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Abstract	Soil aquifer treatment (SAT) is one of the most promising water reclamation and storage techniques in water reuse. This document summarizes the experiences gained in two full scale sites (Shafdan and El Port de la Selva) focused on overcoming the barriers associated with this low-cost technology.

Versioning and Contribution History

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Acronyms

AOP - Advanced Oxidation Process

AOC - Assimilable Organic Carbon

BZF - Bezafibrate

BDOC - Biodegradable Dissolved Organic Carbon

BAC - Biological Activated Carbon

BOD – Biological Oxygen Demand

BTC - Breakthrough Curves

CAPEX – Capital Expenses

CBZ - Carbamazepine

COD - Chemical Oxygen Demand

DCF - Diclofenac

DOC - Dissolved Organic Carbon

DO – Dissolved Oxygen

EC - Electrical Conductivity

ECD - Equivalent Circular Diameter

FMPN - Fecal coliforms count using MPN

EMPN - fecal streptococcus m-enterococcus count using MPN

GAC - Granular Activated Carbon

HRT - Hydraulic Retention Time

IPR - Indirect Potable Reuse

ICP - Inductively Coupled Plasma

IHX - Iohexol

IPDL - Iopamidol

IPRM - Iopromide

LoQ - Limits of Quantification

LT - Lower threshold

MAR - Managed Aquifer Recharge

MPN – Most Probable Number

MTF – Multiple Tube Fermentation

NDMA - N-nitrosodimethylamine

OPEX – Operational Expenses

PSD - Particle Size Distribution

POC - Points Of Compliance

PACl - Polyaluminum Chloride

PAH's - Polycyclic Aromatic Hydrocarbons

PCR - Polymerase Chain Reaction

PTFE - Polytetrafluoroethylene

pGAC - pulverized Granular Activated Carbon

SCADA - Supervisory Control and Data Acquisition

sSAT - short SAT

SAT - Soil Aquifer Treatment

SPE – Solid Phase Extraction

SUVA - Specific UV absorbance

SMX - Sulfamethoxazole

TOC – Total Organic Carbon

TOTB - Total Bacterial count

CMPN - total coliforms count using MPN

TSS - Total Suspended Solids

TrOCs - Trace Organic Compounds

UT - Upper threshold

UVA - UV absorbance at 254 nm

VLX - Venlafaxine

WWTP - Waste Water Treatment Plant

Executive Summary

This report deals with the process of Soil Aquifer Treatment (SAT). The natural treatment system SAT has been known for more than 40 years as an effective natural and relatively cheap tertiary treatment for water reuse. Within DEMOWARE, the objective was to demonstrate the feasibility of SAT as a safe and low cost treatment option.

After a short introduction to SAT (Chapter 1), the first part gives a generic overview about general pretreatment requirements and according technology selection from: i) existing legislation and ii) technical considerations (Chapter 2) and about design guidelines for the planning phase, implementation and operation & monitoring (Chapter3).

Subsequently, two case study sites (Shafdan pilot plant and El-Port de la Selva) from altogether nine demonstration sites within DEMOWARE are presented in detail in Chapters 4 and 5, in order to illustrate site-specific considerations and approaches and to give examples for pretreatment requirements, SAT design and monitoring.

The objective at the Shafdan pilot plant (Chapter 4) was to demonstrate the efficiency of a suggested hybrid pretreatment (flocculation-biofiltration-ozonation) prior to short SAT (~22 days hydraulic retention time, HRT) in obtaining high infiltration rate, while improving product quality. The pretreatment was suggested as a method for treating the Shafdan secondary effluents surpluses at a high infiltration rate using the current SAT fields, instead of constructing new fields or using alternative technologies for treating surpluses. The hybrid process included two main steps: (1) flocculation followed by biofiltration (~5 m/h media filter added with hydrogen peroxide to provide oxygen and promote nitrification). The objective of this step was to remove particles, ammonium, nitrite and some DOC (~20-25%) in order to reduce further oxygen and ozone demand, and (2) ozonation. The objectives of this step were to increase the biodegradability of organic matter, remove trace organic compounds (TrOCs) and microorganisms, and supply high amount of oxygen (~20-30 mg/L DO) to prevent reducing conditions and Mn dissolution in the SAT system. A Dug well (vadose zone infiltration), which was constructed in 2006 in the framework of RECLAIM project, was used for infiltration. Very good microbial quality, low DOC and UVA, low nutrients and low micropollutants containing water were obtained after short SAT using a much less infiltration area and relatively lower retention time in the aquifer.

El Port de la Selva (Chapter 5) represents a site, where SAT has been newly implemented within DEMOWARE with the objectives to supplement the local groundwater sources during the winter season for high demands during the summer and to be a countermeasure against sea water intrusion. Further, it incorporates SAT for indirect potable reuse without reverse osmosis, relying on natural treatment capacity of the soil and aquifer only. The scheme was designed based on given water quality data for the waste water treatment plant (WWTP) effluent, Spanish legislation for indirect potable reuse and hydrogeological and technical considerations. A risk assessment approach following the Australian Guidelines, as outlined in section 2.2, and field investigations and numerical modelling, as outlined in sections 3.1.2 and 3.2.3 were part of the planning and implementation phase and addressed the optimization of pathogen and trace organic removal.

Both case study sites present the results of intense monitoring campaigns and close with a summary of capital and operational cost. In Chapter 6, the lessons learnt are summarized and conclusions are given concerning recommended pretreatment and design guidelines.

1 Introduction

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Against the background of population growth, increasing industrialization, urbanization and climate change, water reuse is increasingly considered as alternative water source for diverse potable and non-potable uses. However, water reuse is currently restricted by low public confidence in available solutions, inconsistent approaches to evaluating costs and benefits of reuse schemes and to date no appropriate regulation at the European level. Task 1.2 within DEMOWARE had thus the objective to mitigate these barriers to water reuse implementation by demonstrating the technical feasibility for hybrid systems integrating improved pretreatment (advanced oxidation, disinfection) and seasonal storage for increased groundwater availability via SAT.

This chapter introduces SAT as a specific case of Managed Aquifer Recharge (MAR) and summarizes the currently applied schemes, advantages and limitations. Key water quality parameters from legislative and technical considerations and according pretreatment needs for implementing SAT schemes are addressed within the following Chapter 2, and design guidelines originating from hydrogeological and technical constraints within Chapter 3.

1.1 SAT as a specific case of MAR

Managed Aquifer Recharge (MAR) comprises a variety of systems in which water is intentionally introduced into an aquifer. In general terms, groundwater recharge can be performed in two ways: through soil and subsoil passage or by direct injection into the aquifer. Schemes using the favourable characteristics of soil, subsoil and aquifer for further (advanced) treatment of the infiltrated water are called SAT (Soil Aquifer Treatment).

SAT generally involves three steps: Surface infiltration, percolation through the unsaturated zone (vadose zone) and slow transport through the aquifer. SAT uses the soil and groundwater as treatment and seasonal or long-term storage (Bouwer and Rice, 1984). The vadose zone and aquifer act as natural, slow filters that effectively reduce the concentration of various pollutants due to physical, chemical, and microbiological processes. Suspended solids are filtered out; biodegradable organic compounds are decomposed; microorganisms are adsorbed, strained out, or die because of competition with other soil microorganisms; nitrogen concentrations are reduced by nitrification or denitrification processes; synthetic organic compounds are adsorbed and/or biodegraded; and phosphate, fluoride, and heavy metals are adsorbed, precipitated, or otherwise immobilized. The general outline of SAT schemes and the processes in the above named three steps are summarized in Figure 1 and Table 1, respectively.

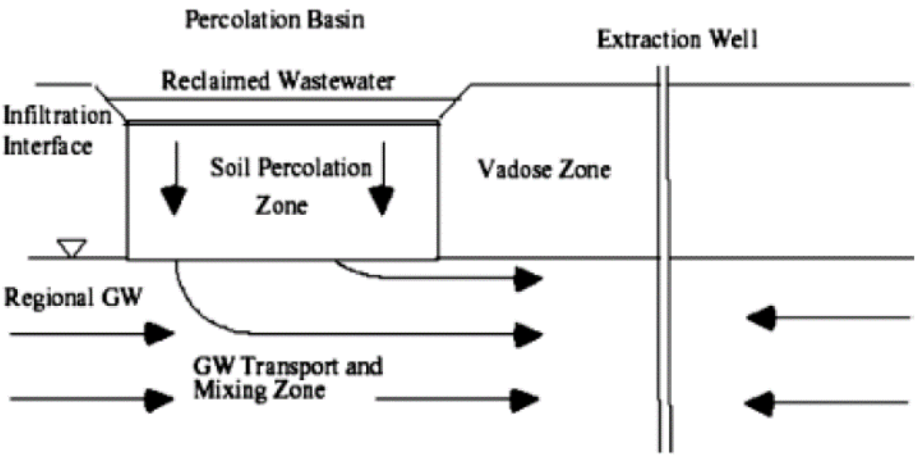

Figure 1 Schematic of a SAT system (Amy & Drewes, 2006)

Table 1 Comparison of typical SAT zones (Amy, 2009)

PROCESS/PARAMETER	INFILTRATION INTERFACE	SOIL-PERCOLATION	GROUNDWATER TRANSPORT
Treatment mechanisms	Filtration ✓, Biodegradation	Biodegradation ✓, Adsorption	Biodegradation, Adsorption, Dilution ✓
Transport	Saturated	Unsaturated	Saturated
Residence time	Minutes	Hours to days	Months to years
Travel distance	Centimeters/inches	3-30 m / 10-100 ft	Variable
Mixing	No	No	Yes (regional G. W.)
Oxygen (O_2) supply	Recharge water	Unsaturated zone	Regional G. W.
Biodegradable org. carbon availability	Excess	Excess/limiting	Limiting
Redox conditions	Aerobic	Aerobic to anoxic	Anoxic to aerobic

SAT is also used to store high quality water to replenish the diminishing groundwater supplies and to prevent sea water intrusion.

Up until now, SAT has been considered as anaerobic and aerobic oscillation. In the Shafdan case study (Chapter 4) we suggest an innovative concept of retaining aerobic conditions throughout the SAT process, which goes beyond the traditional approach to SAT.

1.2 Types of SAT

SAT can be achieved by: (i) surface spreading basins or (ii) vadose zone injection wells, depends on the local hydrogeological conditions and the quality of wastewater effluents available and product quality required (Sharma and Kennedy, 2016).

Where the top soil and aquifer conditions are favourable for using infiltration basins, a high degree of upgrading can be achieved by allowing WWTP effluent to infiltrate into the soil and move down to the groundwater. As outlined before, the unsaturated zone acts as a natural filter and can remove suspended solids, biodegradable materials, bacteria, viruses, and other microorganisms. Significant reductions in nitrogen, phosphorus, and heavy metals concentrations can also be achieved (Bouwer and Rice, 1984). Infiltration by surface spreading in basins can be operated only in sandy soils that should not contain too many clay layers or other soils that could restrict the downward movement of water.

Vadose zone wells are used where surface infiltration is hampered by hydrogeological properties of the soil and where available land is expensive. To prevent clogging, extensive pretreatment is required.

1.3 Advantages and limitations of SAT

SAT is a robust, natural, environment-friendly, sustainable system. It serves as a multiple contaminant removal technology, which removes suspended solids, microorganisms, organic matter, organic micro-pollutants, nitrogen species, phosphates and heavy metals by combined physical, chemical and microbial processes. SAT can be adapted to local conditions and intended use. When combined with appropriate pre/post treatments, its removal efficiencies can be further enhanced (Sharma and Kennedy, 2016).

One of the main advantages of SAT is that it can be applied in both developed and developing countries. It is highly attractive and suitable for arid and semi-arid regions of developing countries. In addition, it is cost-effective (relatively lower cost than above-ground treatment), simple to operate, does not require chemicals or expensive units/equipment, and require minimal operation and maintenance costs. Moreover, it allows seasonal and long-term storage, it can be used for non-potable and indirect potable reuse, and as a part of salt intrusion barrier in coastal aquifers.

On the other hand, limitations may arise from local hydrogeological conditions. Thus, it is not available everywhere. SAT requires extensive land area, especially for spreading basins application. In addition, SAT removal efficiencies are highly dependent on source water quality, local hydrogeological conditions, type of SAT system, and process conditions applied. It is limited in removal of certain contaminants (e.g. organic micro-pollutants), and its performance is site specific.

One of the main concerns of SAT is the possible clogging of infiltration basins, wells and aquifer by the accumulation of suspended matter or chemical precipitation. In addition, SAT can involve development of reducing conditions in the aquifer which may lead to leaching of aquifer materials (e.g. manganese, iron, arsenic, fluoride) to the recovered water. Moreover, SAT can potentially contaminate the aquifer (especially when improper design/pretreatment are applied). Hence, it requires proper monitoring and regulations to avoid hazards.

Generally, SAT is limited by land availability, topography, proximity to the WWTP, regulations and monitoring requirements.

1.4 Importance of SAT design and pretreatment

1.4.1 SAT design

SAT performance is usually difficult to predict. Hence, a feasibility analysis should be performed including relation to all technical, economical, institutional, legal, social and environmental aspects. Then, when the basic requirements for all of the abovementioned aspects are met, a preliminary design can be performed (Sharma and Kennedy, 2016). A proper SAT design will enable to obtain the required removal efficiencies.

Chapter 3 summarizes the main criteria that have to be taken into account when designing a SAT system avoiding long-term problems, minimizing maintenance operations and human risks and ensuring the environmental protection of the local natural resources.

1.4.2 SAT pretreatment requirements

The components of a typical SAT scheme are illustrated in Figure 2, including water source, pretreatment, SAT system, and post-treatment (Sharma and Kennedy, 2016). Pretreatment requirements for SAT may originate from: (i) legislation, (ii) risk-based approaches and/or (iii) technical considerations.

A proper pretreatment and design of SAT, together with appropriate monitoring, can prevent SAT main concerns e.g. clogging of infiltration basins, wells and aquifer; development of reducing conditions and leaching of aquifer materials into the recovered water; and contamination of the local aquifer.

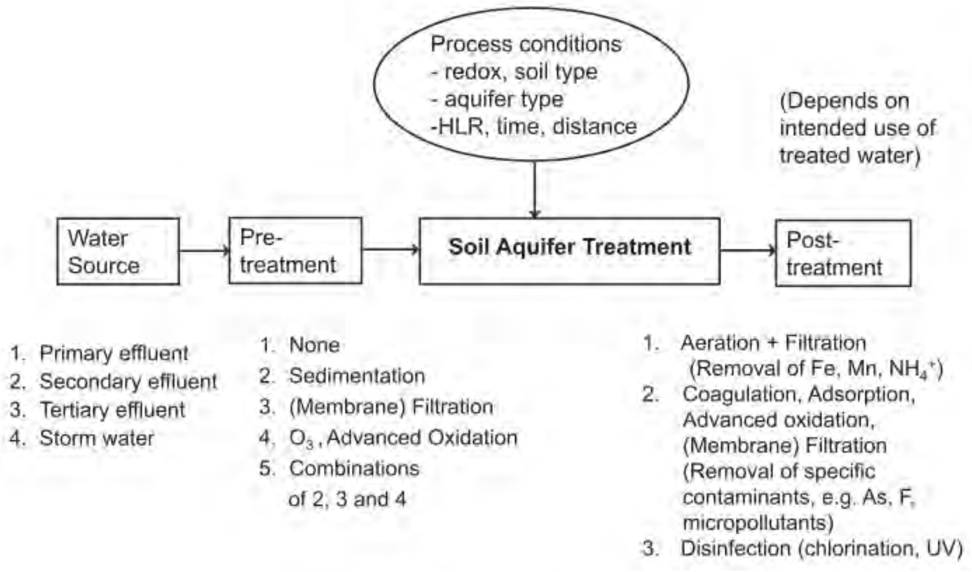

Figure 2 SAT system components (Sharma and Kennedy, 2016)

Chapter 2 summarizes existing legislation as well as relevant water quality parameters and technologies to address their removal.

1.5 Innovations in SAT and pre-treatment technologies

In recent years, new hybrid treatments have been proposed, including advanced pretreatment methods like:

1) Polishing of the secondary effluents before SAT by the use of UF, rapid infiltration of the UF effluents in a Dug-well and recovery of the water after a short SAT (15-20 m. travel distance, around 30 days residence time) to obtain very high quality water for unrestricted irrigation. EU RECLAIM WATER project (Gaus et. al., 2007; Cikurel & Aharoni, 2011).
2) Surface spreading infiltration of tertiary treated effluents (sand filtered secondary effluents) in a short SAT system (15-20 m. travel distance, around 30 days residence time) as pretreatment for NF to polish the SAT effluents to indirect potable reuse quality water. EU SWITCH project (Cikurel et al., 2010).

3) A hybrid pretreatment process of flocculation-biofiltration (~5 m/h media filtration added with hydrogen peroxide)-ozonation prior to dug-well infiltration and short SAT (7-8 m. travel distance and around 22-30 days residence time) for the degradation and removal of micropollutants and prevention of manganese solubilisation (Zucker et al., 2015; and the actual EU DEMOWARE project which further expends this idea, Chapter 4).

In the latter example, the preliminary treated secondary effluents were infiltrated by dug-well infiltration technique (in an intermittent 12 hrs flooding 12 hrs drying regime) at a high velocity (~4-6 m/d) and the water was pumped out after a relatively shorter than conventional retention time in the aquifer (one to two months instead of 6-36 months). Very good microbial quality, low DOC, low nutrients and low micropollutants containing water were obtained by this SAT system using a much less infiltration area and relatively lower retention times in the aquifer due to the pretreatment that was used (Chapter 4).

For both, SAT and well - recharge artificial recharge processes, experience has shown that within the limitations of the conventional toxicological testing, the recovered water does not pose greater health risks than currently acceptable potable water supplies (Bouwer, 1996). The effectiveness of micropollutant removal has been investigated in this project and results are presented within Chapters 4 and 5.

Other hybrid treatments that were also experimented at pilot stage are:

a. SAT pretreatment and NF polishing (Fernando et. al, 2009).
b. Effect of SAT pretreatment of effluents on performance of MF/UF/NF (Sharma et. al, 2009).

Typical residence times within the soil and aquifer passage before recovery lie at 3 to 12 months resulting in further purification and removal of microorganisms, precipitation of phosphates, adsorption of synthetic organics, etc. Thus, SAT is essentially a low-technology, advanced wastewater treatment system. In addition, it has an aesthetic advantage over conventionally treated sewage as water recovered from a SAT system is not only clear and odour-free, but it comes from a well, drain, or via natural drainage to a stream or low area, rather than from a sewer or sewage treatment plant.

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2 Pretreatment requirements for SAT

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Pretreatment requirements for SAT may originate from

1. legislation setting limit values depending on the intended scheme and water use,
2. risk-based approaches and/or
3. technical considerations.

This chapter summarizes accordingly existing legislation as well as relevant water quality parameters and technologies to address their removal.

2.1 SAT regulation

At the EU level, regulations for water reuse do not yet exist for reuse in general, nor specifically for reuse in SAT schemes. Recent publications point out the lack of harmony on the subject of water reuse within the EU, the need to rectify that, and also highlight the advantages and challenges associated with water reuse (BIO by Deloitte (2015); European Commission 2016a; European Commission 2016b; Paranychianakis et al. 2015; Petitguyot 2016; Sapiano et al. 2016).

At present, minimum requirements for the EU pertaining to wastewater reuse for irrigation and aquifer recharge are being drafted by the Joint Research Center of the European Commission (Alcalde Sanz et al. 2016). Guidelines for integrating water reuse into water planning and management schemes were recently finalized within the Water Framework Directive (European Commission 2016a). According to Wintgens et al. (2012) and similarly summarized by Miret et al. (2012), four European directives outline a broad guideline framework from which the following general aspects related to SAT can be derived:

1) Water Framework Directive (EU 2000a; EU 2008):
    - Artificial recharge is a possible measure to achieve ‘good chemical and quantitative status’ of groundwaters.

2) Groundwater Directive (EU 2006):
    - Managed aquifer recharge with reclaimed water must not introduce any hazardous substances into the groundwater.
    - Any other input of substances shall be limited so the groundwater quality does not deteriorate.
    - Processes in the vadose zone that result in attenuation or degradation of substances may be taken into account when considering measures to prevent or limit input into groundwater.
    - Authorized groundwater recharge may be exempted from measures required to prevent and limit inputs, provided appropriate monitoring of the groundwater is carried out.

3) Urban Wastewater Directive (EU 1991)
    - Treated waste water shall be reused whenever appropriate.

4) Drinking Water Directive (EU 1998):
    - A focus is set on the quality control of drinking water as an end-product.
    - Specific compliance values may be set for additional water quality parameters.
    - Member states may define more stringent compliance values.
    - It may be required to monitor additional substances or microorganisms in specific cases.

The respective scope for each of these four EU directives on SAT systems is illustrated in Figure 3 for various SAT scenarios with or without managed recovery of aquifer-treated water. Figure 3 also illustrates the points of compliance (POC), where parameters are typically monitored to assess the performance in SAT systems. The components of a SAT system in Figure 3 demonstrate that a comprehensive definition of SAT is theoretically straightforward, despite the lack of coherence between various national and international guidelines and regulations on the subject.

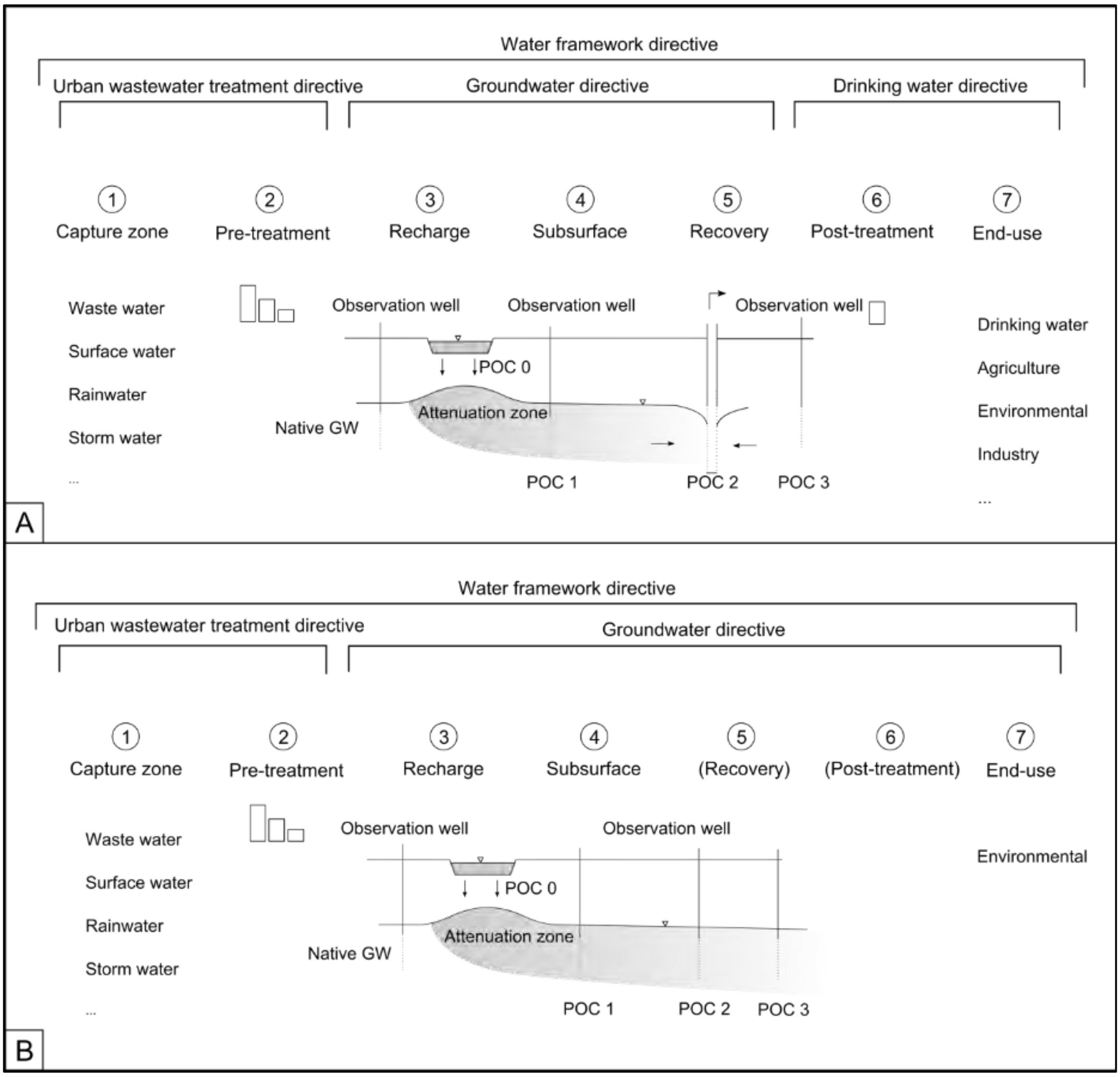

Figure 3 **Components of SAT systems and possible points of compliance (POC) — (A) with and (B) without managed recovery (Miret et al. 2012).**

At national level, specific requirements for SAT with treated municipal wastewater exist only in Greece and Spain (Greek Ministry of Environment, Energy and Climate Change 2011; Spanish Presidential Ministry 2007). According to Alcalde Sanz & Gawlik (2014), SAT is regulated in Cyprus, but Paranychianakis et al. (2015) and the US EPA (2012) do not list SAT as a regulated reuse option for this country. All of them address aquifer recharge as a non-potable application.

Regulations in Greece, Spain, and Cyprus define limit values for specific parameters. The number of regulated parameters varies by country, as outlined in Table 2.

Table 2 Allowable limit values for infiltrated water in SAT in EU countries where reuse for SAT is specifically regulated; note that Cypriot values are local/ site-specific.

Parameter	Unit	Greece	Spain	Cyprus
		Effluent quality standards according to Joint Ministerial Decree 145116/11 (Greek Ministry of Environment, Energy and Climate Change 2011; Paranychianakis et al. 2015; Stathatou et al. 2014)	Maximum acceptable values for indirect recharge according to Royal Decree 1620/2007 (Spanish Presidential Ministry 2007)	Maximum permitted values according to local discharge permission for the Ezousa/ Paphos site* (Yiannakou 2012)
Escherichia coli	[cfu/100 mL]	200 (median)	1000	50
Biological oxygen demand during 5 days (BOD_5)	[mg/L]	25		10
Chemical oxygen demand (COD)	[mg/L]			70
Total Suspended Solids	[mg/L]	35	35	10
Total Nitrogen	[mg N/L]		10	15
NO_3^-	[mg NO_3^-/L]		25	
Total phosphorous	[mg/L]			10
pH	[-]			6.5–8.5
Cl^-	[mg/L]			300
Conductivity	[µS/cm]			1,700
Residual Chlorine	[mg/L]			1
Fat & oil	[mg/L]			5
Intestinal worms	[eggs/L]			0
Boron	[mg/L]			1
Cadmium	[mg/L]			0.01
Chromium	[mg/L]			0.1
Copper	[mg/L]			0.1
Lead	[mg/L]			0.15
Mercury	[mg/L]			0.005
Nickel	[mg/L]			0.2
Zinc	[mg/L]			1

*In the case of the Ezousa aquifer in Cyprus which is recharged with effluent from the Paphos wastewater treatment plant, the water is subsequently reused for irrigation, a non-potable reuse application.

To date, no other EU countries have SAT specific regulations. It should be noted that reuse of treated municipal wastewater in SAT is not strictly forbidden in other EU countries. However, in virtually all cases, SAT with treated municipal wastewater is not allowable due to groundwater protect regulations. Discharge permits must be issued by the local authorities and the related constraints are usually significant.

For reuse applications other than SAT, for example, for irrigation, in sanitary appliances or for street cleaning, regulations do exist in many EU countries. A recent survey by the Joint Research Centre of the European Commission lists all countries with active regulations or guidelines regarding types of regulated water reuse (Alcalde Sanz & Gawlik 2014). Despite the fact that regulations for SAT do not exist in most EU countries, the techniques are utilized at several locations, for example, in Belgium and Germany (TYPSA 2013—Table 1; US EPA 2012).

In both Greece and Spain, recharge of aquifers with treated municipal wastewater is allowed using two possible methods: (i) aquifer recharge via surface percolation or basins (referring to SAT) and (ii) aquifer recharge by direct injection (Greek Ministry of Environment, Energy and Climate Change 2011; Paranychianakis et al. 2015; Spanish Presidential Ministry 2007). In both cases, wastewater reuse for human consumption is explicitly prohibited; however, Spanish regulations allow for it in emergency situations (Article 4.4 of the Spanish regulations). “Reuse for human consumption” may include both, direct potable reuse (e.g. after reverse osmosis + disinfection) or indirect potable reuse (after SAT). Hochstrat et al. (2010) interpret current regulations as *de facto* prohibition.Greek regulations explicitly prohibit both, implying that aquifers used for drinking water production cannot be recharged with treated municipal wastewater (Paranychianakis et al. 2015; Stathatou et al. 2014).

The strict interpretations of the regulations cited above would generally prohibit SAT activities where recharge aquifers are potentially in contact with aquifers used for drinking water production. Usually, drinking water production facilities and wastewater treatment plants (WWTP) are located within several kilometers of each other. Because of this, if treated wastewater is to be infiltrated through SAT, it is unlikely that the underlying aquifer will be separate from the aquifer used for drinking water production without extensive pumping to geographically separate the infiltration zone. In addition, even if infiltration to groundwater is not intended in a specific reuse application, for example with irrigation, reused water may still reach the aquifer via unintended percolation. This means that all applications of reused water have the potential to impact underlying aquifers. Therefore, prohibiting SAT whilst allowing other reuse applications is not a consistent approach for the comprehensive management of treated municipal wastewater and protection of groundwater.

In the light of these contradictions and to harmonize the inconsistency in regulations pertaining to SAT among EU member states, EU-wide guidelines and regulations for wastewater reuse are needed (BIO by Deloitte 2015; Paranychianakis et al. 2015). The general stigmatization of wastewater reuse and overprotective policies are obstructive to more widespread and efficient exploitation of SAT, especially in light of the fact that *de facto* wastewater reuse is already ubiquitously practiced. Indeed, drinking water production from any receiving water body downstream of a discharging WWTP is essentially partial wastewater reuse (Paranychianakis et al. 2015). Well-known examples are the urban water cycles in Berlin, Germany, and Orange County, USA (Pal et al. 2014).

Besides the regulations of certain EU member states, authorities in other regions of the world have established water reuse criteria for their countries. In particular, certain US states and Australia have produced detailed guidelines and regulations which are discussed briefly below. The regulatory approach varies by country. In California and other US states, water reuse regulations are mainly based on limit values and require specific wastewater treatment technologies depending on the intended water use. A comprehensive overview of the different types of water reuse regulation in US states is given in US EPA (2012).

In Australia, guidelines are provided, which outline procedures to assess and minimize risks associated with reuse to be applied to each reuse case. These risk-based water reuse regulations are introduced briefly in section 2.2.

The US EPA defines indirect potable reuse (IPR) as reuse schemes that utilize environmental buffers, consisting of surface water supply reservoirs and groundwater aquifers (Figure 3-7 in US EPA 2012). SAT as a water reuse strategy is thus contained in the second category. SAT limit values for regulated parameters in select US states are listed for both indirect potable reuse and non-potable reuse in Table 3. In addition to the states that prescribe specific limit values as listed in Table 3, other US states define that requirements for water reuse applications be established on a case-by-case basis. Several US states specify the basic treatment techniques to be used in non-potable or potable reuse scenarios.

In conjunction with which treatment techniques should be used, in most cases, specific treatment requirements are given. For example, minimum chlorine doses and contact times or documentation on best practices such as the UV guidelines of the National Water Reuse Institute—NWRI are summarized in Table 4.

The state of California outlines precise treatment results in terms of logarithmic pathogen reductions for various treatment technologies and their combinations that must be achieved for treated municipal wastewater to be used in SAT or groundwater replenishment (CDPH 2014). In addition, a maximum recycled wastewater contribution of 20 % of the total groundwater is defined in CDPH (2014), thus requiring a minimum dilution of factor 5, with possible exceptions. Dilution is also incorporated into calculation of the maximum allowable TOC of the recycled water, as indicated in Table 3. Furthermore, the state of California maintains a periodically updated list of technologies approved for water reuse, helping practitioners to identify potential treatment options (CDPH 2012).

Table 3 Allowable limit values for water to be infiltrated in SAT in different US states (adapted from US EPA 2012, Tables 4-15 & 4-16).

COBD—carbonaceous biochemical oxygen demand (≙ BOD without contribution of nitrogenous bacteria), †min. dilution factor of 5 (CDPH 2014).

		California	Florida	Texas	Washington (surface percolation)	Florida	Washington
Parameter	Unit	Indirect potable reuse	Indirect potable reuse	Indirect potable reuse	Indirect potable reuse	Nonpotable reuse	Nonpotable reuse
Escherichia Coli	[cfu/100 mL]			20 (30d geom) 75 (max) *or Fecal Coliforms*			
Total coliforms	[cfu/100 mL]	2.2 (7d med) 23 (exceeded only once /mon) 240 (max)	4 (max)		2.2 (7d med) 23 (max day)		2.2 (7d med) 23 (max day)
Fecal coliforms	[cfu/100 mL]			cf. E. Coli		200 (avg) 800 (max)	

Parameter	Unit	California Indirect potable reuse	Florida Indirect potable reuse	Texas Indirect potable reuse	Washington (surface percolation) Indirect potable reuse	Florida Nonpotable reuse	Washington Nonpotable reuse
Enterococci	[cfu/100 mL]			4 (30d geom) 9 (max)			
Total Suspended Solids	[mg/L]		5		30	20 (ann avg) 30 (mon avg) 45 (wk avg) 60 (max)	5
Turbidity	[NTU]	media filters: 2 (1d avg) 5 (5% of 1d) 10 (max) membrane filters: 0.2 (5% of 1d) 0.5 (max)	case-by-case (usually 2–2.5, continuous monitoring. required)	3	2 (avg) 5 (max)		2 (avg) 5 (max)
BOD_5 (or $CBOD_5$)	[mg/L]	-	$CBOD_5$ 20 (ann avg) 30 (mon avg) 45 (wk avg) 60 (max)	5	30	$CBOD_5$ 20 (ann avg) 30 (mon avg) 45 (wk avg) 60 (max)	5
Total Nitrogen	[mg N/L]	10 (avg of 4 consec. samples)	10 (ann avg)				Case-by-case
NO_3^-	[mg NO_3^-/L]					12	
TOC	[mg/L]	0.5 * dilution factor[†] of recycled wastewater	3 (mon avg) 5 (max)				Case-by-case
TOX	[mg/L]		0.2 (mon avg) 0.3 (max) alternate limits allowed				Case-by-case

Table 4 Standards for disinfection technologies for tertiary recycled water and high-level disinfection in California and Florida (adapted from US EPA 2012, chapter 6 and Table 6-7).

C_rT—contact time × total chlorine residual, CT—measured modal contact time × oxidant residual at end of contact period.

Technique	California	Florida
UV disinfection	100 mJ/cm^2 following sand or cloth filtration; 80 mJ/cm^2 following MF or UF; 50 mJ/cm^2 following RO	no uniform standard
Chlorine disinfection	450 mg min/L C_rT	25 mg-min/L if fecal coliform <1,000 MPN/100 mL 40 mg-min/L if fecal coliform 1,000 to <10,000 MPN/100mL 120 mg-min/L if fecal coliform >10,000 MPN/100mL
Ozone	1 mg min/L CT	No standard
Pasteurization	10 s at 81.67°C (179°F)	No standard

2.2 Requirements from risk-based approaches

Risk-based water reuse guidelines like those in place in Australia represent a substantially different approach. Here, the focus is put on risks associated with the particular case and the corresponding uncertainties. The Australian guidelines emphasize hazard identification and preventive measures, as well as operational management and provide adequate monitoring. They further differentiate between the intended direct use of recycled water, namely the augmentation of drinking water supplies (NRMMC-EPHC-NHMRC 2008), and indirect use by managed aquifer recharge (NRMMC-EPHC-NHMRC 2009).

The associated guidelines for augmentation of drinking water supplies provide an amplified risk management framework based on the following fundamental principles whose fulfillment should be critically and continuously assessed:

- Safety of public health,
- community acceptance and support,
- institutional oversight and control,
- technical robustness and reliability,
- capacity of designers, operators, and managers,
- management of industrial dischargers, and
- independent regulation and auditing.

Australian regulations provide minimum removal guidelines for drinking water augmentation for several microbial parameters as well as an extensive list of chemical contaminants with typical and guideline values (NRMMC-EPHC-NHMRC 2008).

The Australian risk-based guidance pertaining to managed aquifer recharge for all uses including augmenting drinking water supplies is laid out in NRMMC-EPHC-NHMRC (2009). Aspects such as documenting presumed benefits and feasibility assessment for SAT are outlined therein. Critically evaluating these fundamentals is prudent since it may save a lot of time in cases where benefits do not clearly outweigh investment in a project. Key hazards, as identified in the NRMMC-EPHC-NHMRC (2009) including those related to inadequate source water quality, unfavorable groundwater quality or aquifer composition, and hazards resulting from of MAR operation, along with preventive measures are listed in Table 5. The 2009 guidelines present impacts associated with each hazard in detail, along with mitigation techniques.

An extensive review on risk assessment and risk management strategies in water reuse schemes was made by Chen et al. (2013b). An introduction and comprehensive review of reuse schemes for potable water production providing examples of various treatment trains and specific local constraints encountered is provided by Gerrity et al. (2013).

Similarly, examples and case studies were summarized by Kazner et al. (2012). With respect to suitable treatment combinations for different end uses, a broad review is given in Chen et al. (2013a). Several epidemiologic and toxicological studies focusing on indirect potable water reuse projects are reviewed in Rodriguez et al. (2009) who conclude that no increased risk is posed by drinking water supplemented with recycled water. The high standards applied to technology and water quality control in water reuse schemes may guarantee even greater safety to the public health than common water sources with conventional treatment (Gerrity et al. 2013). This is underlined by a recent study by Tornevi et al. (2016) revealing that even in developed countries, microbial contamination of drinking water is still possible with conventional treatment_ENREF_117.

Table 5 Key hazards in source water, groundwater, and aquifer minerals, with examples and preventive methods (adapted from NRMMC-EPHC-NHMRC 2009).

Hazard	Origin	Examples	Preventive measures
Pathogens	source water, (groundwater)	viruses	adequate aquifer residence time
Inorganic chemicals	groundwater, aquifer minerals, source water	arsenic	control redox potential to avoid mobilization
Salinity (and sodicity)	groundwater, (source water)	salinity	Increase volume of freshwater recharged
Nutrients	source water, (groundwater)	nitrogen	pretreat water (e.g. activated sludge)
Organic chemicals	source water, (groundwater)	pesticides	exclude prone subcatchments
Turbidity and particles	source water, (groundwater)	suspended solids	pretreat water (e.g. wetland)
Radionuclides	ground water, aquifer minerals, (source water)	alpha radiation	pretreat to avoid increase in radioactivity of groundwater
Pressure, flow rates, volumes, and levels	source water	waterlogging	reduce injection pressure
Contaminant migration in fractured rock and karstic aquifers	source water, (groundwater)	polycyclic aromatic hydrocarbons	pretreat or exclude attenuation zone
Aquifer dissolution and aquitard and well stability	source water, aquifer minerals	excess sand recovery	control pH of source water
Impacts on groundwater-dependent ecosystems	source water, aquifer minerals	levels outside historical range	avoid proximity to groundwater-dependent ecosystems
Greenhouse gases	source water	excessive energy use	substitute passive treatments for active

Accordingly, the most common water quality parameters requiring additional treatment in wastewater reuse schemes and potential treatment techniques and their typical performance are described below to provide an overview of options to address these common issues. NRMMC-EPHC-NHMRC (2009) however recognise large differences in effective treatment of subsurface systems, so validation techniques are required.

2.3 Key parameters and pretreatment technologies

Meeting requirements for various water quality parameters is typically achieved by combining several treatment technologies. This is common practice in modern water treatment, and is in the context of water reuse often referred to as "multiple barrier concept" (Gerrity et al. 2013). An important aspect of wastewater reclamation is the location of the respective WWTP. Centralized and decentralized, or remotely-located plants, require different strategies in order to ecologically and economically harness the benefits of SAT (Asano et al. 2007).

As can be seen from the sections above, the regulatory approaches to water reuse in different regions vary considerably. However, the set of parameters deemed critical in water reuse projects is relatively consistent. These parameters encompass the following microbial indicator organisms for hygienic quality, as well as nutrients, and suspended and dissolved organic and inorganic components:

- Pathogens (bacteria, protozoans, nematodes, viruses),
- (total) suspended solids (TSS),
- dissolved organics,
    - dissolved organic carbon (DOC), often expressed via BOD or COD, potentially causing oxygen depletion in the underground,
    - organic micro-pollutants, such as pharmaceuticals and other chemicals,
- dissolved inorganic substances,
    - nutrients, such as nitrogen (ammonium, nitrate) and phosphorous (phosphate),
    - salts, such as Cl^-, Na^+, Mg^{2+}, Ca^{2+}, K^+, SO_4^{2-}, CO_3^{2-}, ...,
    - trace inorganic ions, such as various heavy metal species.

These fundamental water quality parameters are indicative of process performance and impacts on ecosystems and human health. Additional surrogate parameters that may be used for assessment of the water quality include:

- Turbidity, often used as a surrogate for the suspended matter/ TSS,
- color, usually measured at 436 nm, often used for inferring the amount of humic substances,
- transmittance or absorption, usually measured at 254 nm, often used for inferring the DOC,
- conductivity, often used to infer the salinity,
- temperature.

Depending on local circumstances, other parameters may require careful consideration, as discussed in section 2.2. For example, infiltrating low-salinity water can potentially lead to dissolution of aquifer-bound minerals, and ions present in the infiltrated water can substitute other bound ions, releasing them to the water. Underground retention time before water extraction and the permeability of the underground media should be assessed in light of this information as well.

2.3.1 Pathogens

With respect to acute hazards to human health, pathogens are usually deemed the most critical water quality indicator because of the potential for fecal-oral transmission in wastewater reuse scenarios.

Therefore, water reuse regulations for related parameters are in most cases very strict. The pathogen removal efficiencies of treatment techniques are normally measured in logarithmic ($\log_{10}$) reduction units. For an approximation of pathogen and indicator removals resulting from specific treatment techniques, Table 6 gives examples of hygienic improvements as determined in various studies. The first three columns indicate whether primary, secondary, and/or tertiary and advanced treatment stages are included in the given log removals. Given the log removal rates for single stages in Table 6, it is obvious that combinations of several techniques (e.g. activated sludge + sand filtration + disinfection) result in higher overall log removals. It should be noted that very high log removals are often not measurable due to results below the analytical limits of quantification (LoQ). This can be observed from the *seemingly* low log removals determined for some of the disinfection processes listed in Table 6. For example, log removals for the setup “UV after MBR” are comparatively low — which is due to the high log removal rate of the MBR step. Low feed concentrations going into the UV stage, make the high log removals achieved by UV analytically not detectable.

As a result of these considerations, it is advisable to provide pathogen/microorganism log removal rates together with the corresponding influent concentrations for each process stage. In accordance with regulations available for EU countries, the efficiencies of treatment trains to remove pathogens are generally determined by measuring absolute microorganism concentrations, for example, 1,000 organisms E. coli/100 mL as defined in the Spanish regulations (cf. Table 2). This approach avoids the pitfall of erroneously low log removals being measured for specific stages discussed above. Similarly, limit values also exist in the USA (shown in Table 3). In addition, pathogen log removals for specific technologies are defined, for example in California. These technology-specific log removals can then be added for all technologies combined in a treatment train (CDPH 2014; US EPA 2012) in order to reach the required log removals (12 for enteric viruses, 10 for Giardia and Cryptosporidium, in California).

It is strongly recommended to consult the original literature cited in Table 6 for study conditions with regard to water quality and process configurations for the treatment evaluations listed. A review on UV disinfection and required UV exposure for certain log removals in drinking water and treated wastewater is given in Hijnen et al. (2006). Song et al. (2016) review current advances in UV LED lamp technology for disinfection listing dose responses for various applied UV wavelengths. Rose & Rice (2014) review chlorination (concentration × time (ct) values for different log removals), UV radiation, and other disinfectants. Expected log removals of pathogens from Table 6 and with consultation of US EPA (2012), Table 6-3, are summarized below:

- activated sludge (+ filtration)
    - 1–4 log unit bacteria removal
    - 1–5 log unit virus removal
    - 0.5–4 log unit protozoa removal
- membrane filtration
    - ≥4 log unit bacteria removal
    - 2–≥6 log unit virus removal
    - 2–>6 log unit protozoa removal
- disinfection (excluding membranes)
    - 0–>6 log unit bacteria removal
    - 1–>6 log unit virus removals
    - 0–>2 log unit protozoa removal
    - efficiency strongly depends on preceding treatment and LoQs may be underrun frequently

Table 6 **Logarithmic removal rates of indicator organisms and pathogens in selected wastewater treatment processes.**

Modified and extended after Seis et al. (2015), verified original references: 1—Kistemann et al. (2008), 2—Seidel et al. (2013), 3—Fleischer et al. (2000) (not verified), 4—Flannery et al. (2012), 5—Francy et al. (2012), *6*—Zhang & Farahbakhsh (2007), 7—Fu et al. (2010), *8*—Houtte et al. (2012), 9—Huber & Popp (2005), *10*—Rose et al. (2004), *11*—Rose et al. (1996), *12*—Cikurel et al. (2012), *13*—Falsanisi et al. (2010), 14—Kuo et al. (2010), 15—Simmons et al. (2011), 16—Sima et al. (2011), *17*—Kraus et al. (2016), *18*—(Purnell et al. 2016), *19*—Gnirss et al. (2015), *20*—Stüber & Miehe (2016), *21*—the current study (cf. section 5) ; studies involving water reuse marked *italic*.

primary	secondary	tertiary	Technology/ WWTP AS—activated sludge, SF—sand filtration, F—flocculation FF—flocculation-filtration, RC—rotating contactor, C—chlorination, UF—ultrafiltration, MBR—membrane bioreactor MS—micro sieve PFA—performic acid (concentr. as pure PFA)	E. coli	Coliforms (MPN)	Total coliforms	Fecal coliforms	Enterococci	Fecal streptococcus	Clostridium perfringens	Campylobacter spp.	Salmonella spp.	Giardia lamblia / spp.	Cryptosporidium spp.	somatic coliphages	F-specific coliphages	Human adenovirus	Human Polyomavirus	Enterovirus	Enteric viruses	Norovirus	Culturable virus	REFERENCE
			Conventional stages + combinations																				
×	×		Trickling filter/ Heimerzheim (GER)	3.5	2.5				3	2.6			2.3										1
×	×		AS/ Loch (GER)	2.5	1.6				2.3	1.4			0.9										1
×	×		AS/ Hilberath (GER)	2.1	2.3				2.6	1.7			2										1
×	×		AS/ Duisburg Vierlinden (GER)	2				1.9									0.1	1.1	4.3				2
×	×		AS/ Bad Sassendorf (GER)	2.8				2.7									1.4	0.4	0.1				2
×	×		AS/ Schwerte (GER)	2.7				3.1									0.7		0.5				2
×	×		AS/ Reutlingen (GER)	2											2.1	2.4			2.1	1.5			3
×	×		AS/ Heidelberg (GER)	1.7											1.9	1.4			1.9	1.2			3
×	×		AS/ Tübingen (GER)	1.5											1.4	2			1.8	1.2			3
×	×		AS/ municipal WWTP (IRE)	1.5												2.1					0.8		4
×	×		AS/ several WWTP (USA)	3			2.9	3.1							2.5	4.2	2.6		1.5		>1.4	>1.8	5
×	×		AS/ Guelph (CAN)			2.3	2.8								2.3	2.7							*6*
×	×		AS/ (PRC)				2.9						1.9	1.6	2.3								7
×	×		Oxidation ditch/ (PRC)				2.2						2.6	2.2	2.7								7
×	×		Anaerobic-anoxic-oxic/ (PRC)				2.4						2	1.8	2.4								7
×	×		AS+F/ Torreele (BEL)	2		2		1.8		1.5													*8*
×	×	×	AS+SF/ Starnberg (GER)			2.7	3.3		2.3	2.5					1.8								9
×	×	×	AS+FF/ Flerzheim (GER)	3.3	3.2				3.4	2.7			3										1
×	×	×	AS+FF/ Rheinbach (GER)						3.6	3.1			3.5										1
×	×	×	AS+SF/ Miel (GER)	3.5	3				3.6	2.7			2.9										1
×	×	×	AS+SF/ 6 WWTP (USA)			4.1	3.5	3.3		2.8			3.3	2.2		3.4			3.1				*10*
×	×	×	AS+SF/ St. Petersburg (USA)			2.3	2.1						3.2	2.8					2.5				*11*
×	×	×	AS+RC+SF+C/ Guelph (CAN)			5.1	5								4.1	3.7							*6*
		×	SF/ Bad Tölz (GER)										1.8	1									9
		×	F+SF/ (PRC)				1.2						1.6	1.7	0.7								7

rimary	secondary	tertiary	Technology/ WWTP AS—activated sludge, SF—sand filtration, F—flocculation FF—flocculation-filtration, RC—rotating contactor, C—chlorination, UF—ultrafiltration, MBR—membrane bioreactor MS—micro sieve PFA—performic acid (concentr. as pure PFA)	E. coli	Coliforms (MPN)	Total coliforms	Fecal coliforms	Enterococci	Fecal streptococcus	Clostridium perfringens	Campylobacter spp.	Salmonella spp.	Giardia lamblia / spp.	Cryptosporidium spp.	somatic coliphages	F-specific coliphages	Human adenovirus	Human Polyomavirus	Enterovirus	Enteric viruses	Norovirus	Culturable virus	REFERENCE
			Membrane stages																				
		×	UF/ (PRC)				>4.6						>2.4	>1.8	>2.6								7
		×	UF/ Shafdan (ISR)	2.2		2.8	2.8	1.5		3.3									1				*12*
		×	UF+RO after AS/ Torreele (BEL)	>5.5		>6.1		>5.2		4.4													*8*
×	×	×	UF pilot/ Taranto (ITA)	4.2		4.2																	*13*
×		×	MBR pilot/ 4 WWTP, Ohio (USA)	>6.1			>6.7	6.3							3.2	5.1	3.7		3.4		3	2.5	5
×	×	×	MBR/ Guelph (CAN)			5.3	6.2								4.1	4.1							*6*
×	×	×	MBR/ Traverse City (USA)														5						14
×	×	×	MBR after AS/ Traverse City (USA)														3		3.6		4.7		15
	×	×	MBR/ Traverse City (USA)														5.5		5.1		3.9		15
×	×	×	MBR/ municipal WWTP (FRA)	5																	4.5		16
	×	×	MBR/ Sabadell (FRA)	>4.3		4.9				4.1			>3.5				>3.2				>3		*17*
×	×	×	MBR after AS/ Old Ford (UK)				>7	>6.2							5.1	>3.9	4.4				2.3		*18*
			Disinfection stages (excl. membranes)																				
		×	O_3 (0.7 mg/mg$_{DOC}$) aft. AS/ Duisburg V. (GER)	1.4				1.6									0.1	1.9	0				2
		×	O_3 (0.7 mg/mg$_{DOC}$) aft. AS/ Bad Sassen. (GER)	1				1.4									0	0.3	1.5				2
		×	UV after AS+SF/ Bad Tölz (GER)			4.3	4.6		3.6	0.8	1–3.5	1–3			4.5	2.1							9
		×	UV after MBR/ 4 WWTP, Ohio (USA)	0			0.3	0							2.2	0							5
		×	UV after AS/ 4 WWTP, Ohio (USA)	3.8			3.9	3.4							3	1.2	0.2						5
		×	C after AS/ 4 WWTP, Ohio (USA)	2.6			2.3	1.2							1.7	0.7	0.8						5
		×	C after AS+SF/ 5 WWTP (USA)			2.4	2.8	2.3		1.4			0.2	0.3		0.6			0.9				*10*
		×	UV after AS+SF/ 1 WWTP (USA)			3.2	2	2.1		2.3			0.3	0.3		1.1			0.1				*10*
		×	C after AS+SF/ St. Petersburg (USA)			4.2	5						0.7	0.4					1.5				*11*
		×	F+MS+UV (700 J/m^2) after AS/ Berlin (GER)	3.3				2.7		1.9			>1	>2.1	3.6	2							*19*
		×	UV (1100 J/m^2) after AS/ Berlin (GER)	2.8				2.2		0.5			0	0	3	1.9							*19*
		×	O_3 (0.5 mg/mg$_{DOC}$), after AS/ Berlin (GER)	1.4				1.7		0.1			>0.5	0	1.1	1.8							*19*
		×	O_3 (0.8 mg/mg$_{DOC}$), after AS/ Berlin (GER)	2.5				2		0.7			>1	0	2.2	1.4							*19*
		×	PFA (2.7 mg/L), after AS/ Berlin (GER)	3				2.6		0.2			0	0	1.7	0.1							*19*
		×	UV (800 J/m^2), after AS/ Braunschweig (GER)	3.7				2.7		0.4					3.7								*20*
		×	PFA (2.0 mg/L), after AS/ Braunschweig (GER)	3.1				2.6		0.5					2.4								*20*
		×	UV (800 J/m^2), after AS+SF, El Port d.l.S. (ESP)	3.1		2.9		2.6		2.8					2.5								*21*

2.3.2 Suspended solids (TSS)

The removal of total suspended solids (TSS) or turbidity aims at reducing undissolved material such as suspended particles (>0.45 µm). TSS are comprised of inorganic as well as organic particles, and therefore also include many pathogens such as multi-cellular parasites (e.g. nematodes), protozoa (and their cysts), bacteria, and viruses. In most cases, the majority of TSS is removed with sludge during secondary treatment. TSS concentrations of 5–25 mg/L after activated sludge processes and 4–10 mg/L after secondary treatment with nutrient removal and deep secondary clarifiers are typical, with turbidity normally in the range of 2–15 NTU (Asano et al. 2007).

Comparing Table 2 with Table 3 it can be observed that while typical European limit values for SAT (Table 2) are relatively easily reached, those in the USA (Table 3) are likely to require additional tertiary treatment. Despite the fact that tertiary TSS removal may not be required to reach SAT limit values in Greece and Spain (35 mg/L) or Cyprus (10 mg/L), the efficient removal of TSS is crucial for nearly all subsequent effluent polishing steps. The effectiveness of the multiple barrier concept relies heavily on the removal of suspended particles for efficient function of subsequent stages for the following reasons:

- Reduction of clogging/ fouling propensity in membrane filtration steps,
- decrease of oxidant consumption in advanced oxidation/ disinfection stages,
- mitigation of pathogen shielding in radiation-based disinfection processes,
- facilitation of smooth operation and sustainable functioning of SAT which are prone to clogging and may suffer from oxygen depletion in case of high particulate BOD.

It is important to note that any assessment of TSS removal strongly depends on the how TSS is measured. Several definitions exist and filters of different pore size (e.g. 0.45, 1 or 2 µm) may be defined as lower size references. Depending on the TSS content and volume of water filtered, *auto-filtration*, i.e. the additional filter effect of material collected on the filter, may occur, leading to separation of smaller material which would otherwise pass through the membrane. Therefore, the analytical technique used must always be reported, and any measurement procedures stipulated by the respective authority should be followed strictly.

While turbidity may be used as a surrogate parameter for TSS, there is no fundamental relationship between the two. Turbidity measurements are based on the scattering of light where intensity is influenced by particle sizes and optical properties of constituent material (characteristics of a specific wastewater). However, for a WWTP effluent with relatively little variation in particulate composition, turbidity can be a useful indicator for suspended solid content, especially since online turbidity measurement is relatively simple compared to measuring TSS directly.

Typical treatment schemes for particle and colloid removal from secondary effluents are given in Figure 4. Chemical coagulation/flocculation, often combined with sedimentation, typically constitutes the first step in secondary effluent polishing. In many cases, the coagulant is mixed directly into effluent from activated sludge aeration basins before settlement in secondary clarifiers. An additional effect of coagulant addition is the precipitation of phosphorous (cf. section 2.3.4). TOC/ DOC/ BOD/ COD may also be partially removed by coagulation/flocculation and sedimentation/filtration. For additional TSS removal after secondary treatment, multiple techniques exist. Among them are depth filtration, surface filtration, membrane filtration, and air flotation. Often, these techniques are combined with flocculation to increase particulate removal. Depending on the technique, effluent values of <8 mg/L TSS and <4 NTU turbidity should be obtained (Asano et al. 2007; often significantly lower, e.g. Sperlich et al. 2013). Most commonly, depth filtration or rapid filtration is used.

A variety of modifications exist which are classified by the driving force (gravity, pressure), direction of filtration (up, down-flow), backwash regime (dis-, semi-, continuous), filter bed (dual media, deep-bed single medium), and filter medium (sand, anthracite, pumice, synthetic). Selection will depend on water quality, available space, operational expenses, monitoring, automation and other site-specific considerations. In particular, a robust and reliable operation should be the goal — easily achievable with common deep-bed or dual-media rapid filters with sand or anthracite in suitable grain size fractions, operated in up- or down-flow mode. Spikes in TSS can be managed by online turbidity monitoring of the filter effluent, triggering backwashes "on-demand". Installation of frequency converters in the feed pumps is crucial to maintain reliable and continuous operation between backwash cycles and reduce TSS peaks. Relevant literature should be consulted for further information (Asano et al. 2007; Crittenden et al. 2012; Tchobanoglous et al. 2013).

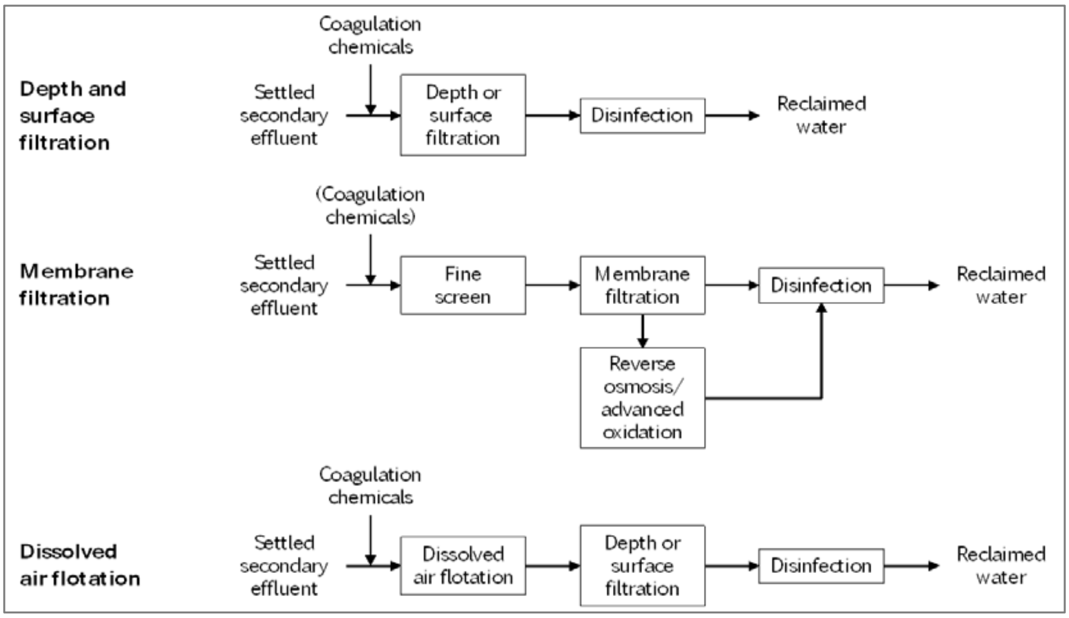

Figure 4 Typical flow diagrams for particle/ colloid removal from secondary effluents, adapted from Asano et al. (2007), Fig. 8-9.

2.3.3 Organic micro-pollutants

Concern over the detection of organic micro-pollutants such as pharmaceuticals and industrial chemicals in WWTP effluent, surface water, drinking water, and groundwater has been frequently raised in recent years. With analytical methods getting ever more sensitive, very low concentrations of these pollutants can now be detected. Potential effects on human health and the environment are currently being explored and are an important aspect of ongoing discussions. Although consequences regarding the toxicity of multiple organic micro-pollutants being present in water are still not clear, the precautionary principle requires addressing the issue.

A recent study elucidates the potential to optimize conventional treatment stages for organic micro-pollutant removal. The study also shows that a broad variety of substances cannot be effectively removed with conventional, i.e. secondary aerobic or anoxic, treatment (Falas et al. 2016). With the number of substances that are resistant to conventional treatment techniques likely to be increasing, advanced tertiary treatment will be the necessary to control organic micro-pollutant concentrations in WWTP effluents (Eggen et al. 2014).

Such advanced technologies for organic micro-pollutant removal include at present: adsorption using activated carbon, oxidation (ozone, advanced oxidation processes), and dense membrane filtration (nanofiltration, reverse osmosis. Removal rates of various organic micro-pollutants under aerobic and anoxic conditions are given in NRMMC (2008).

2.3.3.1 Adsorption onto activated carbon

In contrast to oxidizing techniques, adsorption does not transform organic micro-pollutants but removes them by attachment to the inner surface of the adsorbent material. For the removal of organic micro-pollutants the adsorbent is typically activated carbon. Because pollutants are removed, waterborne toxicity can be assumed to decrease after adsorption stages (e.g. Margot et al. 2013; Prasse et al. 2015). Since the inner surface of activated carbon generally possesses aromatic characteristics, interactions with aromatic, non to moderately-polar substances are favored. Accordingly, substances that are susceptible to adsorption and ozonation share similar properties. The efficiency of treatment with activated carbon depends on a variety of factors. High concentrations of dissolved organic carbon can reduce efficiency due to competitive adsorption. In particular, the quality of activated carbon used can have substantial impacts on efficiency and thorough testing is often necessary. Recently, it was shown that the micro-pollutant removal potential can be satisfyingly derived from the removal of the UV absorption at 254 nm under various process conditions (e.g. Anumol et al. 2015; Zietzschmann et al. 2014; Zietzschmann et al. 2016b).

Activated carbon is typically applied in one of two methods. Using granular activated carbon where treated water flows through packed, fixed-bed adsorbers; or in a pulverized form which is suspended in the water and later removed via coagulation + sedimentation/filtration. Granular activated carbon is preferred where the flow volume remains constant and where peak concentrations of the substances of concern are modest. One advantage of granular activated carbon is that it can be thermally re-activated, reducing overall carbon consumption. If flow volumes or organic micro-pollutant concentrations vary considerably, powdered activated carbon can be more suitable, as dosage is more easily adjustable. Recent research demonstrates that direct addition of powdered activated carbon to rapid filtration influents is a simple and effective alternative to separate contactor basins (Altmann et al. 2015b; Löwenberg et al. 2016). For detailed documentation on water treatment using activated carbon, the general literature is recommended (e.g. Crittenden et al. 2012; Sontheimer et al. 1988; Worch 2012).

2.3.3.2 Ozonation (& advanced oxidation)

Ozonation is an effective means for oxidizing electron-rich moieties in compounds such as those containing aromatic structures, π–electron bonds, and deprotonated amines. Depending on water properties (e.g. pH, nitrite, and background organic matter concentrations), the efficiency of ozone to oxidize organic micro-pollutants can vary greatly. Alkaline conditions cause faster decomposition of ozone to form OH-radicals which react nonspecifically with background organic matter and are thus easily scavenged. Due to its oxidative nature, ozone typically does not mineralize compounds but leads to their transformation. Normally the concentration of dissolved organic carbon (DOC) decreases only marginally due to ozonation, and may even increase due to dissolution of particulate matter).

The application of ozone is typically reported as the concentration of applied ozone per concentration of dissolved organic carbon (DOC), in mg O_3/mg DOC. This is also referred to as ozone consumption. Usually, ozone consumption has a range between 0.4 and 1 mg/mg. In this range, removals of >80 % of compounds like carbamazepine, diclofenac, and metoprolol which are problematic with conventional treatment, can be obtained (e.g. Margot et al. 2013).

In contrast, compounds lacking π-/aromatic electrons or de-activated aromatic systems are less effectively removed, e.g. iodinated x-ray contrast agents (e.g. Altmann et al. 2014).

In many cases, removal of organic micro-pollutants with ozone leads to reduced toxicities but adverse effects have also been reported, e.g. the formation of aldehyde groups (e.g. Magdeburg et al. 2012; Magdeburg et al. 2014; Prasse et al. 2015). Particular problems can arise from synthesis of carcinogenic nitrosamines, especially N-nitrosodimethylamine (NDMA). Furthermore, ozonation can cause the formation of bromate in waters with elevated bromide concentrations. Although these negative impacts may only occur in some circumstances, ozonation requires close monitoring of formed by-products using elaborate equipment. The potential for adverse effects from ozonation has led to the practice of using biological filters to reduce potential effluent toxicity as a post-treatment

Caution must also be taken when handling ozone because of the toxicity of gaseous O_3. Increased safety protocols are required. For detailed documentation on ozonation processes and applications, the relevant literature is recommended (e.g. Crittenden et al. 2012; Tchobanoglous et al. 2013; von Sonntag & von Gunten 2012).

Several full-scale plants employing advanced oxidation processes (AOP) are now in operation around the world (Australian Government 2009; Pal et al. 2014). AOP typically act by creating hydroxyl radicals which are unspecific with regard to target substance and have the potential to attack a broader range of compounds than other advanced treatment techniques. This lack of specificity can also be a constraint as it can decrease process efficiency in cases with high concentrations of background organic matter which easily scavenge hydroxyl radicals (Prasse et al. 2015). Similar to ozonation, advanced oxidation processes normally do not mineralize organic micro-pollutants and thus, potential adverse effects of transformation products must be considered.

2.3.3.3 Dense membrane filtration

Dense membrane filtration technologies such as nanofiltration and reverse osmosis are used for the removal of dissolved inorganic compounds such as salts but these techniques can also remove many organic substances. All molecules larger than the nominal pore diameter of the membrane are removed from the filtrate. Besides the molecular size cutoff of the membrane, the properties of the membrane itself, including the type of material strongly influence particle rejection. Charged compounds are especially well retained by membrane filters due to electrostatic repulsion and steric hindrance. Retention of neutral compounds is often considerably poorer. Adsorption of compounds to the membrane surface, interactions with other water constituents (e.g. natural organic matter), and diffusion hindrances due to fouling layers also influence the rejection rates of membranes. Unlike the removal of organic micro-pollutants with oxidative techniques, the risk of hazardous by-product creation does not exist using membrane processes. However, several oxidation by-products resulting from pre-chlorination, such as NDMA, are able to pass even dense membranes. Often, AOP are used after membrane filtration to address these substances (Pal et al. 2014). Additional drawbacks of dense membrane filtration include the concentration of rejected substances in the retentate generating brines with high concentrations of undesired compounds. The disposal of these brines poses a major challenge and represents high economic and environmental costs. Also, energy consumption for membrane treatment is considerably higher than in most adsorption and oxidation processes. Because of the higher cost, dense membrane filtration is most appropriate to treat water containing inorganic substances as well as organic constituents. A short introduction to dense membrane filtration for the removal of dissolved inorganic constituents is given in section 2.3.4.

For more details, the relevant literature is recommended (e.g. AWWA 2007; Crittenden et al. 2012; Peinemann & Nunes 2010; Tchobanoglous et al. 2013).

2.3.3.4 Comparison of techniques for organic micro-pollutant removal

A comparison of the advanced treatment techniques for organic micro-pollutant removal discussed above is given in Table 7. The table shows which technologies are appropriate for specific properties of substances targeted for removal. A single technology is presently not available to remove all possible organic micro-pollutants. Currently, more than 65 million chemicals are commercially available and the number continues to increase (Snyder 2014). Thus, the selection of suitable technologies is highly case specific. Choosing the appropriate advanced treatment technology is a complex process where available resources and the potential side effects of each treatment technology, both positive and negative, must be evaluated in the context of the entire treatment train.

Consideration of economic and environmental impacts of the various treatment alternatives further complicates selection (Bui et al. 2016). Within the recent German projects ASKURIS and IST4R, cost comparisons and life cycle assessments were conducted for multiple treatment configurations using ozone and activated carbon. The results of these projects demonstrate that with regard to cost, neither technology is clearly superior. (Jekel et al. 2016a; Jekel et al. 2016b). With respect to environmental impacts, ozonation is preferable to activated carbon treatments if the raw material for activated carbon is fossil based (e.g. lignite), even if a higher percentage of renewable energy sourced electricity is assumed for the future. However, adsorption configurations in which activated carbon is sourced from renewable raw materials (e.g. coconut husk) show environmental impacts similar to ozonation.

Table 7 Rough classification of advanced techniques for the removal of organic micro-pollutants, according to target substance properties and process characteristics

NDMA—N-nitrosodimethylamine.

	Ozonation	Activated carbon	Nanofiltration	Reverse osmosis
Target substance size		<10000 g/mol	>1000 g/mol	>100 g/mol
Target substance chemistry	aromatic, electron-rich	aromatic, non-polar, hydrophobic	ionic, multiple-charged	ionic
Overall toxicity	(generally) reduced	reduced	reduced	reduced
Beneficial side-effects	disinfection, increase of dissolved organic carbon biodegradability	removal of dissolved organic carbon, biologic enhancement; filtration (granular activated carbon); increased sludge heating value (powdered activated carbon)	salinity reduction, disinfection	salinity reduction, disinfection
Adverse side-effects	transformation products requiring post-treatment, toxic by-products (bromate, NDMA)		change of water matrix composition—potential effects on water chemistry	change of water matrix composition—potential effects on water chemistry
Benefits from pre-treatment	high, e.g. from prior reduction of suspended/ dissolved organic carbon, nitrite	medium, e.g. from prior reduction of dissolved organic carbon	high, e.g. from prior reduction of suspended/ dissolved inorganic/ organic material	high, e.g. from prior reduction of suspended/ dissolved inorganic/ organic material

	Ozonation	Activated carbon	Nanofiltration	Reverse osmosis
Efforts	electricity consumption, O_2 consumption	mining, (re)activation, transport	energy consumption, membrane production, cleaning chemicals	energy consumption, membrane production, cleaning chemicals
Waste streams		increased sludge amount (powdered activated carbon)	brines	brines
Safety issues	gaseous ozone	only when handling dry powdered activated carbon (explosion risk)	cleaning chemicals, brines, high pressures	cleaning chemicals, brines, high pressures

2.3.4 Dissolved inorganic constituents

2.3.4.1 Nutrients

Nutrient removal is one of the key aspects of wastewater treatment and effective processes are crucial when subsequent SAT is intended. In general, the secondary treatment stages of WWTPs are responsible for removal of nutrients, particularly ammonia and nitrate. To this end, sufficient aerobic and anoxic conditions, and SRTs are essential. Phosphorous is typically removed via chemical precipitation. More detail can be found in the relevant literature (e.g. Asano et al. 2007; Crittenden et al. 2012; Tchobanoglous et al. 2013).

2.3.4.2 Salts

Dissolved salt constituents in water are often referred to as total dissolved solids (TDS). Due to their high concentrations in WWTP effluents relative to most groundwaters, partial removal may be necessary. High concentrations (e.g. Cl^-) can be toxic for some species and high Na^+ concentrations can cause replacement of Mg^{2+} and Ca^{2+} ions at the cation exchange sites in soils.

In many cases, simple dilution of WWTP effluents may be sufficient to obtain decreased TDS concentrations. If TDS are to be removed, dense membrane filtration processes, namely nanofiltration, reverse osmosis, and electrodialysis, are suitable. Due to several factors, the application of dense membrane filtration for WWTP effluent polishing should be scrutinized in detail for each case: Investment and operational costs are comparatively high, the accruing brines have to be treated and disposed off, and the water produced may still require post treatment. Asano et al. (2007) concluded that, based on cost, electrodialysis may be best suited for reducing the salinity of low-to-medium saline waters. An advantage of nanofiltration and reverse osmosis is that they target several problematic parameters in parallel, namely suspended matter, TDS, nutrients, organic pollutants, and pathogens. Table 8 compares common dense membrane filtration technologies for treatment of WWTP effluent.

Table 8 Comparison of nanofiltration, reverse osmosis, and electrodialysis for salt removal from WWTP effluent (modified after Asano et al. 2007; Kazner et al. 2012; Orchard 2007).

Parameter	Unit	Nanofiltration	Reverse osmosis	Electrodialysis
Operating pressure (WWTP effluent)	bar	3.5–5.5	12–18	-
Energy consumption (WWTP effluent)	kWh/m^3	0.6–1.2	0.75–1.8	1.1–2.6
Total dissolved solids removal	%	20–95	85–99	50–94
Advantages	-	organics removal, disinfection	organics removal, disinfection	fouling/ scaling controllable via periodical pole reversal
Disadvantages	-	scaling, fouling, water composition change	scaling, fouling, water composition change	no organics/ pathogen removal, not suitable for very high TDS

2.3.4.3 Trace inorganic ions/ heavy metals

In most cases, the effluents of modern WWTP contain only low concentrations of inorganic trace pollutants, as most of these are adsorbed in the solid phase/sludge (Fuchs et al. 2010). However, in some cases concentrations of heavy metals, e.g. Zn, B, Al, Mn, up to several µg/L, have been reported (Choubert et al. 2011). The cited study concluded that conventional secondary treatments stages are capable of removing the majority of these metals, in particular via adsorption to the solid phase and separation during secondary sedimentation. Removals of >75 % were reported for Ti, Cr, Cd, Cu, Zn, Sn, Pb, Fe, Ag, and Al. Moderate removals (~50 %) were reported for intermediately adsorbing metals: Ni, Se, Ba, and U. Poor removals <20 % were reported for Co and As as well as for the weakly adsorbing metals Li, Rb, Mo, Sb, V and the metalloid B (Choubert et al. 2011). Similar removals were reported by Karvelas et al. (2003) who further reported removals of Mn at >75 %. Thus, for most heavy metals, increased removal rates can be achieved by optimizing secondary treatment and/or adding a tertiary filtration stage.

To assess the relevance of heavy metals dissolved in WWTP effluent, Table 9 compares typical (conservative, i.e. comparatively high) emission values for large-scale German WWTPs with threshold values for groundwater, below which no considerable negative impacts are expected (modified after Seis et al. 2015). According to the calculated ratios, the heavy metal concentrations in WWTP effluents generally comply with the groundwater negligibility criteria. The ratio, and thus the relevance decreases in the order Zn > Cu > Cr > Ni > Cd > As > Pb > Hg. For the metalloid B, no emission factors are defined by UBA (2015), but from values reported in Choubert et al. (2011), it can be assumed that concentrations in WWTP effluents are <250 µg/L for both municipal and rural WWTP.

Table 9 **Emission factors for typical German size 5 WWTP with >100,000 PE (UBA 2015) versus German negligibility threshold values for groundwater (LAWA 2004) for different heavy metals (modified after Seis et al. 2015) and Boron (Choubert et al. 2011).**

Compound	Unit	Emission factor	German negligibility threshold	Ratio [%]
As	µg/L	0.326	10	3
B	µg/L	<250 (Choubert et al. 2011)	740	<34
Cd	µg/L	0.06	0.5	12
Cr	µg/L	2.36	7	34
Cu	µg/L	7.61	14	54
Hg	µg/L	0.0016	0.2	0.8
Ni	µg/L	3.88	14	28
Pb	µg/L	0.19	7	2.7
Zn	µg/L	51.6	58	89

In cases where additional heavy metal or boron removals from WWTP effluents are required, there are numerous technologies available. Table 10 gives an overview of potential methods. Consulting the cited literature is recommended as the removals stated in Table 10 are combined from various studies. For example, there are countless variations of clay minerals and further chemical modifications, each of which may have specific adsorptive affinities for listed pollutants. Also, other factors such as temperature, pH, and the ion speciation of a compound must be taken into account since they can significantly impact the removal efficiencies of different treatments (Sen Gupta & Bhattacharyya 2012; Sen Gupta & Bhattacharyya 2014). Note that boron removal by reverse osmosis is generally high, but most studies focus on seawater desalination where *very* high removals of boron are necessary to comply with drinking water standards (e.g. Kim et al. 2009).

Table 10 **Heavy metal and boron treatment efficiencies of different techniques: +high, ◦medium, -low**
(combined from Argun 2008; Asano et al. 2007; Athanasiadis & Helmreich 2005; Aziz et al. 2008; Barakat & Schmidt 2010; Bouguerra et al. 2009; Chen et al. 2009; Crittenden et al. 2012; Ferella et al. 2007; Fu & Wang 2011; Genc-Fuhrman et al. 2007; Huang et al. 2010; Ipek 2005; Kabay et al. 2008; Karahan et al. 2006; Kim et al. 2009; Korus & Loska 2009; Okay et al. 1985; Öztürk et al. 2008; Renman et al. 2009; Sen Gupta & Bhattacharyya 2012; Sen Gupta & Bhattacharyya 2014; Tu et al. 2013).

Compound	Precipitation/ coagulation + sedimentation/ filtration	Lime softening	Activated carbon	Clay minerals	Activated alumina	Anion exch.	Cation exch.	Ultrafiltration + surfactant/ polymer	Reverse osmosis
As	+	+	◦	-	+	+	-	◦	+
B	-			◦	◦, +(MgO)	+			+
Cd		+		◦				+	+
Cr (+3)	◦	+	◦	◦	-	-	+	+	+
Cr (+6)	◦	-	◦	◦	-			+	+
Cu	+	+	◦	+		-	◦	+	+
Hg	◦	◦	◦	◦	-	-	◦		+

Compound	Precipitation/ coagulation + sedimentation/ filtration	Lime softening	Activated carbon	Clay minerals	Activated alumina	Anion exch.	Cation exch.	Ultrafiltration + surfactant/ polymer	Reverse osmosis
Ni		+		◦	+		+	+	+
Pb	+	+	◦	+	-	-	◦	+	+
U	+	+	-		+	+	+		+
Zn	+	+		+	+		+	+	+

2.4 Common problems and matrix for technology selection

A brief guide to identifying pretreatment issues using water quality parameters is given in Table 11 along with possible solutions in the secondary and tertiary treatment stages. This information is based on the most common problems encountered in wastewater treatment and that specific cases may be more complex. Comprehensive treatment train assessment can be achieved with a risk-based evaluation of all potential shortcomings and by identifying the most relevant weak points (cf. section 2.2). In parallel, the economic and environmental costs should be assessed in detail during the planning of WWTP upgrades.

Table 11 Indicators, associated problems, and possible solutions in secondary or tertiary treatment of wastewater prepared for SAT

SRT—solids retention time

Indicator	Potential issue	Potential solutions Secondary treatment	Potential solutions Tertiary treatment
Total suspended solids (TSS)/ turbidity	inefficient secondary clarification	reduce flow/ flow variability, avoid dead spaces and wind circulation cells, deepen clarifiers	
	additional filtration needed		install tertiary filtration/ membrane filtration
BOD, COD, TOC, DOC	inefficient aeration	increase aeration and/ or SRT	install tertiary (biological) filtration (e.g. biologic granular activated carbon)
	inefficient TSS removal	see TSS	see TSS
Ammonia nitrogen	inefficient nitrification (aeration)	increase aeration and/ or SRT	Biofiltration (e.g. Shafdan site)
Nitrate nitrogen	inefficient denitrification	increase anoxic phases/ SRT	
Nitrite nitrogen	inefficient nitrification (aeration)	increase aeration phases/ SRT	
Total nitrogen	inefficient nitrification and/ or denitrification	evaluate aerobic/ anoxic phases, SRT	
Total phosphorous	inefficient precipitation	evaluate precipitant dose/ dosage/ mixing, floc separation	additional precipitation; evaluate precipitant dose/ dosage/ mixing, floc separation

Indicator	Potential issue	Potential solutions Secondary treatment	Potential solutions Tertiary treatment
Total dissolved solids (TDS)	high saline influent		install dense membrane filtration
Heavy metals, boron	inefficient secondary clarification	reduce flow/ flow variability, avoid dead spaces and wind circulation cells, deepen clarifiers	
	inefficient precipitation/ coagulation	evaluate precipitant dose/ dosage/ mixing, floc separation	additional precipitation; evaluate precipitant dose/ dosage/ mixing, floc separation
	soluble species in effluent		install advanced stage for trace metal/ metalloid removal (cf. Table 10)
Volatile organic compounds (VOCs)	inefficient aeration and/ or mixing	increase aeration and/ or mixing	
	additional oxidation or adsorption needed		install ozonation/ AOP or activated carbon
Organic micro-pollutants	inefficient aeration (/ sludge activation)	increase aeration and/ or SRT	
	recalcitrant organic substances		install advanced technique (ozonation, activated carbon adsorption, dense membrane filtration)
Pathogens (e.g. total coliforms, protozoan cysts/ oocysts, viruses)	inefficient TSS removal	see TSS	see TSS
	higher pathogen removal required		install disinfection or membrane filtration

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3 Design guidelines for SAT systems

E. Vilanova, M. Bayer (Amphos 21)

SAT is essentially a low-technology, advanced water treatment system using the soil, vadose zone and groundwater as treatment and seasonal or long-term storage (Bouwer and Rice 1984). As a result, SAT has an excellent capacity for removing a wide range of contaminants from the effluent by a variety of processes (Aharoni *et al*., 2011). With proper operation and maintenance and adequate monitoring, SAT should be considered as attractive and reliable method for effluent reclamation and reuse in areas where suitable conditions exist for groundwater recharge via spreading basins (Idelovitch et. al, 2003). Consequently, it is important to identify these suitable conditions that allow the proper and safe long-term performance of these systems and incorporate them into design guidelines and recommendations.

The SAT treatment is done in three main steps: surface infiltration, percolation along the non-saturated zone and slow transport during the aquifer. The main reactions take place in soils and in the first centimeters of the vadose zone.

This chapter summarizes the main criteria that have to be taken into account in order to design a SAT system avoiding long-term problems, minimizing maintenance operations and human risks and ensuring the environmental protection of the local natural resources.

3.1 Planning phase

The successful implementation of a SAT system is based on the fulfilment of different technical, legal and social criteria which need to be assessed before SAT project construction. Before the project implementation, two stages of previous studies are recommended:

1) Pre-feasibility stage. In this stage, the main elements are qualitatively evaluated before any economic investment.
2) Feasibility stage. In this phase, different site parameters are quantified in order to properly design the project. These tasks correspond to an implementation phase.

It has to be taken into account that a parallel risk analysis is needed especially when dealing with reclaimed or reused water. Additionally, the Australian Guidelines for Water Recycling (NRMMC, 2009) state that some risks cannot be fully understood before managed aquifer recharge is implemented at full scale, due to uncertainties associated with aquifer processes. However, with adequate system characterization and assessment, it is possible to adopt preventive measures and operational procedures (including incident and emergency management).

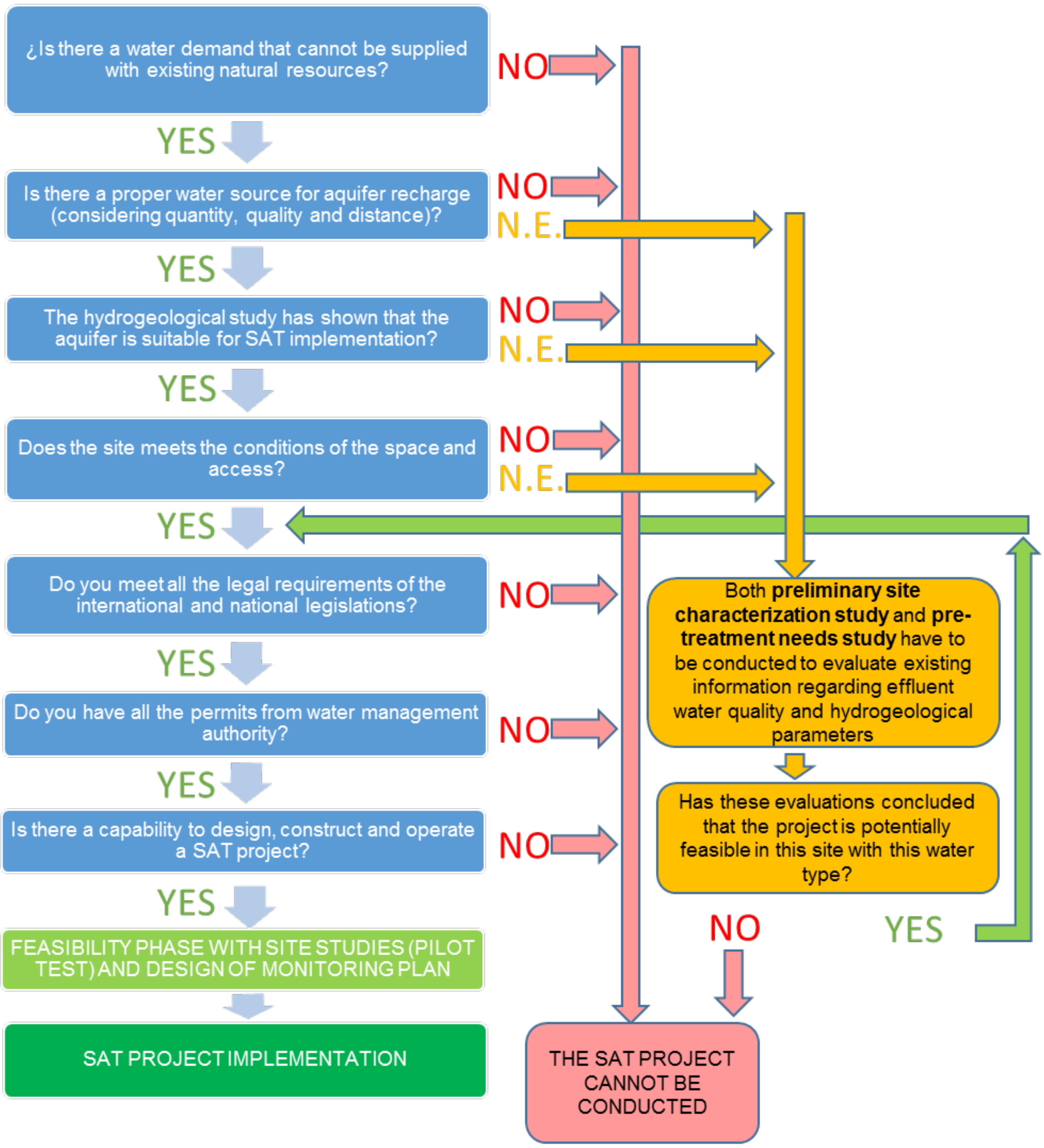
3.1.1 Pre-feasibility phase of a SAT project

In this stage the successful implementation of a SAT system is evaluated through a qualitative approach. The main elements that have to be addressed in this stage are according to Dillon (2009) (Figure 5):

1) A sufficient demand for recovered water.
2) An adequate source for recharge.
3) A suitable aquifer in which to store the water.
4) Sufficient land to build the project.
5) Capability of effectively manage the project.

Additionally, the project has to fulfil international and national legislations and count with all required permits.

A demand that cannot be supplied with existing resources is the driver for investment in a SAT project with reclaimed water and therefore this is the first question that needs to be considered (Figure 5). The remaining questions address entitlements to source water location, quantity, quality and the physical capability of the aquifer for groundwater storage, which require a detailed knowledge based on existing and collected information. It has to be pointed out that effluent water needs to be characterized at this stage as this can be a limiting factor. Positive answers suggest that the project is potentially viable and then a feasibility phase can be initialized with the collection of local present data and a potential pilot plant project.



N.E.= Not evaluated

Figure 5 A list of considerations to decide whether to undertake a SAT project (modified from Dillon, 2009; Parsons et al 2012)

The implementation of a SAT project requires a site characterization study with a different degree of hydrogeological features knowledge depending of the feasibility phase. This preliminary site characterization should address at least the storage capacity, the presence of impermeable layers or preferential conducts, and potential contamination problems and should allow identifying the complexity of the system and the additionally required studies. A suitable aquifer provides enough storage capacity, allows high recovery efficiency and does not lead to a significant deterioration of recovered water quality (Steinel, 2012)

3.1.2 Feasibility phase: hydrogeological requisites

Hydrogeological features determine the final feasibility of a SAT project. The design of the system will depend on the values of different hydrogeological parameters, if the pre-feasibility study has been positive.

Within the DEMEAU project (EU grant agreement no. 308339), based on Dillon et al (2008), investigations identified nine parameters with their corresponding objective criteria to adequately evaluate the suitability of a site for man-made aquifer recharge. These parameters that are also described in Table 12 are:

1) Aquifer confinement: SAT cannot be conducted in a confined aquifer, where an impermeable layer separates the target aquifer from the non-saturated zone.
2) Target aquifer permeability. Best recharging aquifers are those with high permeability values.
3) Saturated thickness. This is relevant in case of water recovery as thin saturated thickness can sometimes allow higher potential recovery rates, subject to groundwater salinity.
4) Thickness of unsaturated zone. If this depth is less than 10 m the storage potential is low. Depths higher than 100 m impede the management of the project and the water recovery (Steinel, 2012)
5) Aquifer type: porous aquifers are the most suitable for groundwater recharge and SAT processes.
6) Homogeneity of hydraulic parameters is advisable for recharging purposes.
7) Quality of native water. Change in redox conditions can lead to the dissolution of some metals in water. Non-saline aquifers are generally easier to manage in recharge projects.
8) Low hydraulic gradients are preferred as they promote conservation of recharged water near recharging areas and increase the residence times.
9) Area with high topographic slopes are not suitable for SAT systems (recharging ponds)

Table 12 Hydrogeological pre-requisites and design criteria parameters for a SAT project (based on de la Loma & Sprenger 2015)

Parameter	Criteria	Assessment
Aquifer confinement	Confined	Not viable for SAT systems
	Unconfined	Storage capacity depends on depth-to water table and effective porosity High vulnerability to surface contamination
Target aquifer permeability, K_f (m/s)	$< 10^{-6}$	Very low, limited suitability
	$10^{-6} – 10^{-5}$	Low, limited suitable
	$10^{-5} – 10^{-4}$	Medium, suitable
	$10^{-4} – 10^{-3}$	High, suitable
	$>10^{-3}$	Very high, suitable

Parameter	Criteria	Assessment
Saturated Thickness in target aquifer	<10	Thin, high potential recovery rate
	10-50	Medium, medium potential recovery rates
	>50	Thick, high protection (low vulnerability) of fresh aquifers
Thickness of unsaturated zone	<5	Limited potential storage potential
	10-30	Good potential storage potential (Aharoni et al., 2011)
	>30	High potential storage potential, (groundwater mounding can be neglected). But increase of complexity over groundwater recharge. More than 100 m higher recovery costs.
Aquifer pore type and consolidation	Porous	Very suitable
	Porous and fractured (mixed)	Suitable (limited)
	Fractured / Karstified and a combination of both	Suitable with limitations. Deeper hydrogeological characterization is needed to ensure storage and required travel times.
Uniformity of hydraulic properties	Homogenous (e.g. variance of K ≤ 0.5 log10)	Minimal mixing and higher recovery rates for well injection if native groundwater is brackish
	Heterogeneous (e.g. variance of K ≥ 0.5 $\log_{10}$)	Lower recovery rates for well injection if native groundwater is brackish In karstic and fractured aquifers limited ability to store recharged water
Change in existing redox state	Aerobic aquifer and aerobic recharging water	Higher inactivation rates for pathogens and some endocrine disruptors (EDCs) Higher removal for some pharmaceutical active compounds (PhACs)
	Sub-oxic aquifer and aerobic recharging water	Higher removal for some pharmaceutical active compounds (PhACs) Redissolution of some metals in the aquifer Clogging processes
Intrinsic aquifer salinity	Fresh (TDS <1000 mg/l)	Fresh aquifers are suitable for recharge of fresh water. Together with TDS conservative tracers can be analyzed (e.g. Cl) to trace and differentiate the recharged water from the native one.
	Brackish/saline (TDS >1000 mg/l)	Mixing with native groundwater should be minimized and a specific buffer volume is recharged for this purpose. Saline groundwater can have a buoyancy effect on the recharged water due to differences in density. This can result in abstraction of saline water by upconing from lower parts of the aquifer.
Hydraulic gradient	Gentle (<0.1%)	Small lateral (and vertical) hydraulic gradients promote conservation of the recharged water in the recharge zone. Avoid flood-prone areas. Higher residence times
	Moderate to steep (> 0.1 %)	Lateral flow can cause the injected fresh water to move outside of the recharge zone , where it cannot be recovered
Topographic slope	Gentle (<5%)	Suitable
	Moderate to steep (> 5 %)	Not suitable

3.1.3 Water final use

The design of the project has further to take into account the intended use of the recovered water and the present use of the aquifer.

The objective of intended recharge can be environmental with ecological objectives, for water quality management (e.g. to reduce saline intrusion), physical management of the aquifer or to maximise natural storage. Once implemented, different final uses can take place in any site: environmental, agriculture, industrial domestic or drinking. The two later cases are those that need specific studies to ensure that human health is not affected. National legislations can have different levels depending on the final use.

1) If the aquifer is connected with natural ecosystems, quality of water in this output has to be properly addressed and evaluated.
2) When water is recovered for agricultural uses, the quality of water will define the irrigation type (if disinfection is not complete, spray irrigation methods are not allowed).
3) Industrial uses cover a wide range of application from cleaning purposes to process water. A specific study needs to be carried out in each case in order to evaluate the need of potential post-treatments.
4) If there are productive wells and drinking wells close to the recharge area, a risk analysis is mandatory to ensure human health. This study has to be based on travel times between recharging area and abstraction point considering both the quality of the recharging water and the dilution factor along the aquifer, as it has been explained before in Chapter 2. Numerical models have been proven as a powerful tool to estimate this value. National legislations have to be considered in these cases.

3.2 Design phase

If the feasibility study based on hydrogeological parameters has resulted in a positive conclusion regarding the construction of a SAT system, then the system has to be designed taking into account the previously identified relevant hydrogeological characteristics, the water availability, the water demand and the risk analysis. This phase requires a deeper characterization study as a well as a pilot phase. Field survey and pilot testing should always be done to ensure a proper site performance and system management. Parameters to be addressed in this stage include:

- The physical design of the construction
- Wastewater quality: the need of additional pretreatments (addressed in previous chapters)
- (Residence) Time and travel distance
- Chemical reactions: Redox conditions (oxidation-reduction)
- Maintaining infiltration (hydraulic loading)

Altogether, they determine the requirements and conclusions for the location, geometry and depths of the recharge and recovery facilities as will be outlined below.

3.2.1 Shape and location of recharging facilities

Shape and location of the recharge facilities are the main factors for residence time, travel distance, resulting redox conditions and maintenance needs. When constructing a recharge basin it is recommended that the elongated side should be perpendicular to the regional groundwater flow direction. This allows the best lateral flow away from the recharge site and ensures best infiltration performance (de la Loma & Sprenger, 2014).

Additionally, Massmann et al. (2003) stated that ponds with large aspect ratios (defined as pond length divided by pond width) have higher infiltration rates than ponds with lower aspect ratios, based on numerical simulations.

Increased pond depth increases the hydraulic gradient across the bottom layer, which tends to increase the infiltration rate if all others factors remain constant. However, this effect competes with that of the surface clogging effect of the bottom layer. Therefore, the increase in depth can either result in a higher or lower infiltration rate (Houston et al., 1999). This effect has to be evaluated locally and, to avoid undesired consequences, lower depths are preferred in order to facilitate management procedures and to have short wet-dry cycles of recharge (eg Pavelic et al 2011).

In case of treating primary, secondary or tertiary effluents, the recharging area should be placed at certain distance from the production wells in the same aquifer. This distance depends on different factors but the most influent are the pretreatments conducted prior to recharge and hydraulic aquifer characteristics. If no pretreatments are applied, the quality of infiltrating water needs to be assessed very carefully as the removal of DOC, micropollutants, heavy metals and phosphate depends mainly on residence time /travel distance (Sharma et al., 2008). The residence times/travel distances are more related to horizontal residence time within the aquifer and the distance of the recovery well from the infiltration point than to the infiltration rates and the residence time and vertical travel distances in the vadose zone. Longer travel times allow high removal efficiency.

For the removal of effluent organic matter hydraulic loading rate and redox conditions are important, too (Saroj et al., Harun, 2007, Miret et al.2012).

A typical distance that has been applied in different systems to ensure a complete disinfection when dealing with reclaimed water is a distance that results in a travel time of 6 months to reach the wells. Nevertheless, the DEMOWARE project has demonstrated that this rule of thumb cannot be applied everywhere especially if bacteria and viruses are present. Additionally, other studies have pointed out that infiltration percolation efficiency in disinfection has been demonstrated to be variable and mainly depending on water detention time (related to pore velocities and media heterogeneity) and oxidation achievement (Brissaud et al., 1998).

Harun’s study (2007) concluded that:

- The removal of contaminants during SAT is case specific and may not be presented using one kinetic model.
- Travel time has more influence on DOC removal than travel distance.
- Generally average bacteria removal was >4 log at travel distance <10m and virus removal was >2 log at travel distance >10 m. Not all SAT systems are able of completely remove microbes.

Therefore, a generally applicable distance between infiltrating area and production wells cannot be formulated. The main tools to determine the distance and the corresponding travel time are numerical models and risk analysis approaches. Confidence on numerical modelling will depend on the available information for its construction. Numerical modelling tasks and risk analysis approaches are further developed in chapter 5.7 and deliverable D3.2 of the DEMOWARE project, respectively.

Validation of log removal using diffusion cells (Toze and Hanna 2002) or using pathogen surrogates at field scale would improve confidence in quantitative micriobial risk assessment, and in some countries (eg Australia) are required if the aquifer is relied on for pathogen removal within an attenuation zone (NRMMC-EPHC-NHMRC 2009).

3.2.2 Filtration layer (bottom of the pond)

A technical sand layer within the ponds ensures spatially constant infiltration rates, decreases maintenance efforts and is thus highly recommended. Clogging, which is in most cases unavoidable, will develop mainly on the surface and allows that filter layer material can be removed, washed and filled back in case of infiltration rates decreasing below a certain threshold.

The bed of the infiltration basins is usually filled with two layers of quartz material of minimum 80% SiO_2 content. The filter layer on top acts as a filter for cleaning the source water. The supporting bottom layer below aims to avoid “sweeping” of filter layer material to the aquifer. Grain size should be about three times larger than filter layer material. Grain size distribution for both layers are characterised by a steep slope, expressed by the uniformity coefficient. An example of technical specifications of the filter layer material is shown in Table 13 (Sprenger and de la Loma, 2014). If ambient soil meets these criteria this avoids the need to import sand.

Care should be taken not to compact the underlying soil when levelling, placing filter media or scraping and replacing. Use of large soft types on lightweight vehicles, or hand labour in developing countries, is preferred.

Table 13 Filter layer thickness and grain size distribution according to DIN EN 12904.

	Filter layer	Supporting layer
Thickness, min-max (m)	0.4 – 2	0.1 – 0.5
Grain size, d10-d90 (mm)	0.15-0.3	×3
Grain size, min-max (mm)	0.1-0.35	×3
Uniformity, d60/d10 (-)	~2.3	~2.3

3.2.3 Residence time and dilution factors: Numerical models

In contrast to analytical solutions, numerical models can be adapted to a wide range of site-specific conditions and problem statements. A large number of numerical models have been used to analyse various SAT systems. It is important to keep in mind that numerical models try to simulate aquifer behaviour based on field data. On the other hand, numerical modelling deals with a high degree of complexity and a high demand of site-specific data and is therefore a tool to obtain a range of values to estimate the general performance of the system.

Modelling helps to improve the selection of the proper operational scenario, and to decide on the amount of recovery wells and their distance from the infiltration area. Therefore, the model has to be developed in the feasibility phase and has to include data from an initial monitoring period to help in system design. Furthermore, it is necessary to calibrate the model using pilot test data. Once calibrated, the model should be able to reproduce, with some degree of uncertainty, the system performance.

The type of the numerical model will depend on site characteristics such as e.g. water salinity. A transient numerical 2D groundwater flow model with transport has been developed within DEMOWARE for the case study site El Port de la Selva to evaluate travel time to production wells (cf. section 5.8). It illustrates the constraints linked to this approach.

In the case of El Port de la Selva (Mediterranean climate), as travel times are highly dependent on short and intense rainfall events, different rainfall scenarios have been simulated to estimate a probability density function of travel time from the infiltration basins to the abstraction wells used for drinking water production.

As a result, the limiting parameter to estimate travel time here is not the hydrogeologic parameters but the precipitation pattern. The estimation of travel times has to take into account the uncertainty in precipitation and the resulting values given with a confidence interval.

3.3 Implementation phase: SAT system operation

3.3.1 Wet-dry circles to reduce clogging

The main operational problem in SAT systems (both infiltration basins and wells) is clogging. The vadose zone wells cannot be pumped or rehabilitated after clogging (in some cases even chemical and mechanical treatments are not efficient) (Aharoni et al., 2011).

In surface spreading SAT systems, the problems with clogging appear in the surface layer. The sedimentation of this clogging layer at the bottom of the ponds alters the original hydraulic properties of the upper layers of the basin due to hydrophobicity that develops (Arye et al., 2010). Usually, mechanical graving methods are applied to remove the top clogging layer (Aparicio and Carrera, 1998). A proper operation of short wet-dry cycles can reduce the formation of this low conductivity clogging layer. The objectives of this procedure are:

1) to provide oxygen in the spreading basins,
2) wet-dry cycles help to improve redox conditions. During wetting periods, oxygen is depleted as it is consumed by organics. In drying periods, oxygen enters soil as wetted front. Then ammonia can consume oxygen resulting in nitrate production (Aharoni et al., 2011).
3) to control the passage of the effluent through the unsaturated zone and part of the aquifer,
4) to disminish the clogging of the infiltration surfaces,
5) to allow the dryness of the bottom of the infiltration zone and the posterior formation of dessecation mud cracks on sedimented material,
6) to clean the bottom of the infiltration surfaces, if needed, during dry periods.

Reintroducing water to the basin on a windy day additionally helps to drive dessicated sediment to one end of the basin and to refresh the bulk of the basin area.

3.3.2 Monitoring network and hydrogeological tests

In order to recognize any deviation from designed criteria and initial operational considerations, the design of a SAT system has to include the design of the monitoring network and the definition of a monitoring plan to ensure that water quality is not affected. Determining quantitative recharge aspects also helps to undertake a better cost-benefit analysis and make informed decisions. Due to site-specific characteristics, each scheme will be different.

Depending on the scale of the recharge scheme and the density of existing wells, it might be necessary to drill new wells. If drilling is done, it will additionally allow getting a detailed lithological description and material from deeper unsaturated layers for permeability assessment as well as analysis of geochemical components to judge possible interactions between the recharged water and the aquifer matrix. Different monitoring wells have to be placed at different distances of the recharging areas and multi-tube piezometers are recommended to verify the homogeneity of the media. More than one monitoring well is especially required in fractured or karstic aquifers. They should be placed at least at the margin of the attenuation zone.

If reclaimed wastewater is used, lab tests or geochemical modelling with supply water, groundwater and aquifer matrix helps to identify possible reactions (especially precipitation or dissolution) that might pose a risk to the SAT scheme.

Groundwater level measurements over a larger region should be repeated over at least one year to assess groundwater flow direction and seasonal variations, especially if previously available data are scarce. If flow paths are unknown, tracer tests could help identifying flow direction and velocity (e.g. Käss, 1998; Kalbus et al., 2006; Luhmann et al., 2012). This would further allow determining the best recovery location for highest recovery efficiency after the required retention time (Steinel, 2012).

Groundwater samples should be taken at a number of wells over the year in the upstream and downstream area to assess the beneficial use potential starting with a baseline monitoring i.e. the investigation of the current situation before the SAT scheme is implemented. For quality monitoring, it is further important to characterize the quality of recharging water with continuous measurements, especially if reclaimed water is used as this can be different along a year. In these cases, an automatic system to shut down the systems if certain quality parameters are not fulfilled (suspended solids, salinity, N compounds, etc.) is advisable and exemplarily outlined for the case study site El Port de la Selva in Chapter 5.

Conservative tracers in source and groundwater can help determine the amount of mixing. Tracers are those compounds with different concentrations in native groundwater and recharging water. Calculating the amount of mixing is a way to estimate the increase of recharge in the aquifer. The most common conservative tracers in that context are Cl, B, Br and F. Minor components as Li or Sr can be good indicators, too. Stable isotopes of water can also be a good tool to quantify recharge processes. Additionally, if reclaimed water is used, anthropogenic contaminants as persistent organic micropollutants can be studied to evaluate the arrival of recharging water. However, the behaviour of some of these organics is still not well understood and interpretations have to be done carefully. As typically recharging water comes from water sources that are close to the aquifer and chemical characteristics are thus similar in both water types, using multi tracer approaches to characterise the different water types is highly recommended.

Operational monitoring finally includes the definition of both the sampling parameters and the frequency. The objective of operational monitoring is to ensure that the operation is not negatively impacting human health. These aspects will depend on water type and aquifer characteristics. If reclaimed water is recharged, higher sampling frequencies are recommended (e.g monthly).

The monitoring plan has to idenfy the sampling points (new or existing water wells or piezometers) and to set up the sampling parameters. Groundwater level has to be measured in all monitoring points.

In Port de la Selva, 15 points are being monitored including water in ponds, drinking water well and output form the WWTP. In all the points, water level is measured and chemical parameters are evaluated (Figure 6). The parameters are different in each campaign and point. They have been grouped in 12 categories (Table 14). In situ data (group 0), anions (group1) cations (group 29 and backup samples (12) are collected and analyzed in all campaigns which are beign done every 2 weeks. The rest of parameter are analyzed monthly in those points located between infiltration points and drinking water well. Water level and salinity data measured continuously in water that is going to be recharged have been implemented in the SCADA system. In case that the EC value is higher than 1500 µS/cm, the system is automatically shut down.

At the present, 1 year after the first infiltration, the monitoring has not shown evidence of recharging water in drinking well. A detailed description of all monitoring and sampling activities is further given in the according sections in chapter 5.

Table 14 **Groups of monitoring parameters in Port de la Selva.**

Groups	
0	In situ
1	Anions
2	Cations
3	Metals
4	Doc, uv254
5	Aox
6	Isotopes
7	Priority substances
8	Trace organic compounds
9	Bacteria
10	Phages
11	Viruses
12	Backup

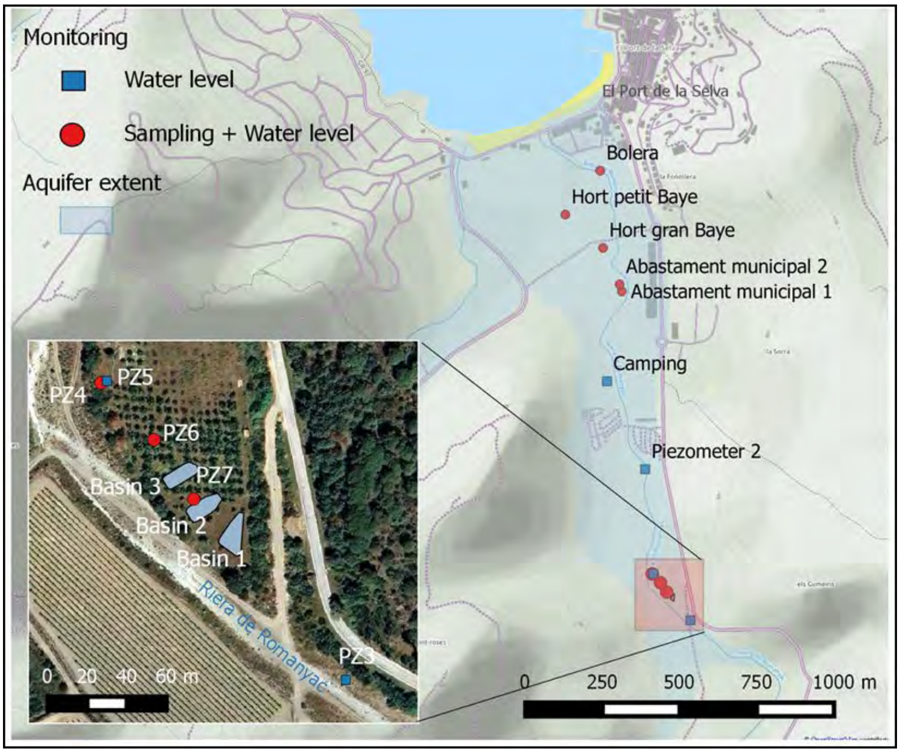


Figure 6 **Monitoring points in Port de la Selva site**

In the Shafdan pilot site the following sampling points were monitored periodically along the process: (1) secondary effluents, (2) after coagulation (mainly for particle analyses), (3) after biofiltration, (4) after ozonation, (5) ceramic cups (1.5 m under the dug well, in the upper layer of the vadose zone), (6) observation well OW1 (~22 days HRT, 7.3 m distance from the dug well), (7) observation well OW2 (~113 days HRT, 17.6 m distance from the dug well).

The parameters that were monitored: (1) Daily measurements: turbidity (at sampling points 1-4), pre-treatment operational parameters (e.g. flows, pressures, ozone and chemical dozes, etc.); (2) Monthly campaigns: field measurements: turbidity, pH, conductivity, temperature, DO, ORP; Kit measurements: H_2O_2; Lab measurements: DOC, UVA, BOD, COD, DO, NH_4, NO_2, NO_3, Kjeldahl-N, PO_4, Fe, Mn, Chlorides, bacteriology (TOTB, CMPN, FMPN, EMPN), TrOCs (Tel-Aviv University), particle analysis (Tel-Aviv University); (3) Season measurements: alkalinity, hardness, Ca, Mg; (4) Special measurements: viruses, Cryptosporidium, Giardia, bromide/bromate.

Sampling procedures follow the Standard Methods requirements (Rice et al., 2012). The measuring equipment has to be calibrated in each sampling campaign while continuous measuring probes need a calibration every two months. An external calibration is recommended additionally every 2 years.

The review of monitoring results allows the adjustment of parameters and sampling frequencies (NRMMC-EPHC-NHMRC, 2009) and it is vital that monitoring results are collected, evaluated and distributed to interested stakeholders. The compilation and acceptance of annual operation and monitoring reports should be a license condition for the SAT operation. The responsible government agency should retain these electronic records in a library to provide a research resource to allow future refinement of guidelines and allow effects of changes in operating practices to be documented.

3.3.3 Reporting of monitoring results

The objective of the monitoring tasks is to ensure that the operation is beign conducted without risk for local population. As a result, the monitoring results have to be continuously revised and analysed taking into account the site characteristics. Permission documents usually contain requirements for documentation and reporting of operational and monitoring results.

For results interpretation standard hydrogeological methods and graphs should be applied, such as piper diagrams, Schoeller diagrams or boxplots comparing native groundwater, infiltrating water and resulting mixted groundwater along the flow paths from the recharge area towards the point of abstraction. Additionally, the characterization has to be accompanied by tracer compounds identification and analysis, as it has been explained before. Main target is to identify the arrival of the infiltrating water at the sampling points. To evaluate the expected concentration in sampling points, mixing models and reactive transport processes can be calculated using softwares as Phreeqc. These tools are useful to evaluate the reactive processes of those non-conservative tracers and should be regularly compared to real measured values in mixed groundwater samples.

In order to control operation and recognize and mitigate any deterioration as fast as possible, the operator has to fix two types of quality indicators:

- Indicators in infiltrating water: if these limit values are exceeded the infilatration has to be stopped
- Indicators in monitoring points:
    - Trends: if the defined parameters show a trend that is related with recharge and can lead to water quality deterioration, the recharge system has to be revised.
    - Thresholds: if the defined tresholds are exceeded, the recharge system needs an inspection. These tresholds have to be lower than drinking water limits in order to be ale to use them as an alert indicator.

Sometimes monitoring tasks are managed by non hydrogeology experts. In these cases, a check-list can be used to assess the results of the monitoring together with the previously fixed indicators.

Table 15 provides an example that can be used to asses the changes or impacts in the native groundwater and to identify potential impacts.

Table 15 Proposal of check-list to evaluate results of monitoring campaigns.

	Conservative compound			Non-conservative compound		
Compound						
Range of concentration in native water						
Range of concentration in infiltration water						
Data in the sampling campaign /date						
Indicator value in sampling point						
Limit value in drinking legislation						
Data in the previos campaign						
Has the data changed significantly from the previous campaing? Y/N						
Is there some trend in the data evolution? (increase, decrease, stable, radom…)						
If this trend expected by mixing or reactive models?/ is this trend defined as impact indicator?						
Is this value below the indicator value?						

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4 Shafdan

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4.1 Site description

The Shafdan is the largest WWTP in Israel, treating approximately 370,000 m^3/d of wastewater, contributed by the entire Dan (Greater Tel-Aviv) Region (2.5M p.e.). The wastewater is treated by primary mechanical treatment (bar screen and grit removal), conventional activated sludge (secondary) treatment, and tertiary Soil Aquifer Treatment (SAT, Dan region Sewage reclamation project).

The Shafdan WWTP produces about 130 Mm^3/yr of secondary effluent of which 125 Mm^3/yr are further treated by SAT. During SAT, the secondary effluent is recharged into the local aquifer through the vadose (unsaturated) zone, by means of infiltration basins (Figure 7), conducted in an intermittent plan of flooding/drying cycles. The SAT includes five infiltration basins having a total area of about 1 Mm^2. Each basin is divided into several ponds which are flooded alternately. The recharge regime is based on 3-5 days cycles of flooding and drying of the basins (usually 1 day flooding and 2 days drying. In winter drying could be longer) and their bi-monthly plowing. This management regime was adopted in order to increase oxidation along the SAT system and to maintain aerobic conditions (Icekson-Tal et al., 2003). The thickness of the vadose zone below the infiltration basins is 30-40 m. Recovery of the water is done, following a retention period (6-36 months) in the aquifer, through wells or drains surrounding the recharge area. The production wells around each infiltration basin are arranged in circles, aimed at creating a hydrologic trough between the effluents and the surrounding freshwater, thereby minimizing leakage into the main body of the Coastal Plain aquifer. By the time the effluents are reclaimed in these production wells, their quality is greatly improved and they are qualified for unrestricted irrigation and incidental drinking. The reclaimed effluents are then piped for irrigation to the Negev desert in southern Israel (Goren, 2008). At Shafdan, ~90% removal of DOC has been observed (Elkayam et al., 2015), which is high in comparison to other SAT systems worldwide, likely due to the Israeli climate conditions and long hydraulic retention times (HRTs) that enable effective biodegradation of organic matter. Besides, the SAT provides efficient physical pathogen removal due to long retention times. This cost-effective technology, which is mostly used in arid and semiarid regions, results in efficient removal of many wastewater contaminants based on the combined effect of biological and physicochemical processes (Amy and Drewes, 2007).

Figure 7 **Infiltration basins at Yavne, Dan Region.**

4.1.1 Specific challenges

Removal of organic matter is a critical parameter in SAT since it also influences the removal of other contaminants like nitrogen compounds, micropollutants and microorganisms. Most of the oxidation processes of organic compounds as well as of nitrogen compounds take place in the unsaturated zone (vadose zone). In the Shafdan SAT system (6-36 months HRT), which treats ~140 MCMY at intermittent natural aeration of the vadose zone (1 day infiltration and 2 days drying), the consumption of oxygen in the vadose zone is 42.5 ± 12 mg/L (Elkayam et. al, 2015). This very high oxygen consumption agrees well with the high purification efficiency (~90% removal of DOC and complete removal of ammonia) of the Shafdan SAT. During infiltration, high levels of ammonium and DOC result in a rapid consumption of DO by aerobic biodegradation processes (Lance, 1972). Once the DO has been consumed, microorganisms utilize alternate electron acceptors, commonly nitrate ($NO3^-$) and manganese/ferric oxides, which are present in the soil (Curtis, 2003). This may cause anoxic conditions and mobilization of dissolved manganese Mn^{2+} from the soil. Indeed, in several of the Shafdan production wells, high Mn concentrations were observed (2-40 μmol/L which are 110-2200 μg/L), which caused precipitation of Mn oxides in the distribution pipelines, and clogging of drippers in agricultural irrigation systems (Goren, 2008).

Another problem of the Shafdan SAT is that it is limited in treating the increasing amounts of effluents for infiltration due to limited infiltration rate in the existing fields and lack of land for more available infiltration area. Hence, the present infiltration fields are gradually operating in more stressed conditions where the HRT in the soil is shortening, and there is much less available DO in the upper layer of the SAT (vadose zone). This also causes a lack of oxygen in the following groundwater horizontal flow regime up to the production wells. Thus, decreasing the SAT efficiency.

Additional concern is the presence of TrOCs such as pharmaceutical residues, which partially persist during conventional wastewater treatment and can be detected in receiving surface water (Kolpin et al., 1999-2000). Although many of these TrOCs are efficiently removed in SAT systems by sorption and/or biodegradation, some polar and non-biodegradable compounds persist and can be detected in reclaimed water (Kinney et al., 2006). The use of reclaimed water for irrigation is thus an important route for the introduction of TrOCs into the environment (Chefetz et al., 2008).

4.1.2 Selection of pretreatment (pilot)

Several pilot- and full-scale studies on alternatives to conventional SAT by multi-stage tertiary treatments based on ozonation with subsequent (bio)filtration were conducted and presented in the literature. Ozonation followed by soil columns to simulate the SAT process (Schumacher and Jekel, 2004; Tiwari et al., 2014), quick biological sand filtration (Hollender et al., 2009) or biological activated carbon (BAC) treatment (Gerrity et al., 2011; Reungoat et al., 2012) demonstrated improved biodegradation of DOC and elimination of ozonation byproducts and TrOCs.

The current case study suggests a pretreatment to SAT based on biologically active dual-media filtration (referred to as "biofilter") and ozonation, to allow better SAT performance in a short HRT (~22 days of short SAT, sSAT). Biofiltration was chosen to remove the excess ammonia in the Shafdan secondary effluents that consume the oxygen in the upper layer SAT and cause anoxic conditions in the aquifer. Biofiltration is a relatively economic means for ammonia removal and is also a clean process (as compared to ammonia evaporation that could require air pollution control).

Biofiltration was operated with the addition of hydrogen peroxide (H_2O_2) for oxygen supply to remove ammonium and nitrite, as well as particles and some DOC, in order to reduce oxygen and ozone demand in subsequent treatment steps. At the current stage hydrogen peroxide was chosen as a rapid oxygen provider that can easily be soluble through the biofilm on the filter media as compared to all sorts of gaseous oxygen. Due to the high cost of hydrogen peroxide, alternatives like reuse of off-gases from ozonation and/or return part of the ozonated stream (rich with high ~20-30 mg/L DO) back to the biofilter for better biodegradation and oxygen supply will be further investigated.

In addition to its major objective to remove TrOCs and microorganisms, ozonation was applied in order to increase the biodegradability of DOC before SAT and to supply oxygen for the enhancement of aerobic conditions during SAT, in order to control Mn^{2+} mobilization. The overall hybrid SAT pretreatment was suggested to enable reduction of physical footprint of existing SAT by increasing infiltration rate.

4.2 Experimental pilot setup

4.2.1 Operation modes

The Shafdan pilot plant was fed by 144 m^3/d treated wastewater from the secondary clarifier output. The pilot included a hybrid biofiltration-ozonation–short SAT system (Figure 8), with a biologically active high-rate filtration unit (~5 m/h, section 4.2.2), an ozonation unit (section 4.2.3) and a short SAT facility (sSAT, section 4.2.4). The sSAT facility included a Dug well (i.e. Recharge Borehole, RBH) and two functional observation wells: (1) *OW1* of ~22 d HRT; (2) *OW2* of ~113 d HRT (Figure 11).

The first stage of the research included optimization of chemical dosages: PACl, H_2O_2, and O_3. The second stage of the research included a long-term continuous operation in *Mode 1*: infiltration of biofiltered effluents, without addition of ozonation (blank experiment, March 2014-July 2015, Figure 8). The third stage of the research included a long-term operation in *Mode 2*: infiltration of biofiltered–ozonated effluents (August 2015-September 2016, Figure 8). Due to operational problems in the ozone system, several shutdowns of the ozone system were obtained during *Mode 2* operation. From June until September 2016, the ozone system worked continuously. Thus, enough data was collected on the effect of ozonation on biofiltered effluents. However, the effect of ozonation on SAT performance, which require continuous operation of several months (in order to see the effect on OW1 and especially on OW2), was obtained at the last months of *Mode 2* operation, between June and September 2016 (total of 111 days), when the ozone system was operated continuously.

During operation of the designed pilot system, 16 grab-sampling campaigns (in each unit of the pilot) were conducted from March 2014 to September 2016 (Table 16). In addition, on May 2015, three ceramic cups were installed at a 1.5 m depth under the Recharge Borehole, in order to better understand the hydro-chemical mechanisms involved in the vadose and saturated zones (Figure 13) and were sampled starting from June 2015.

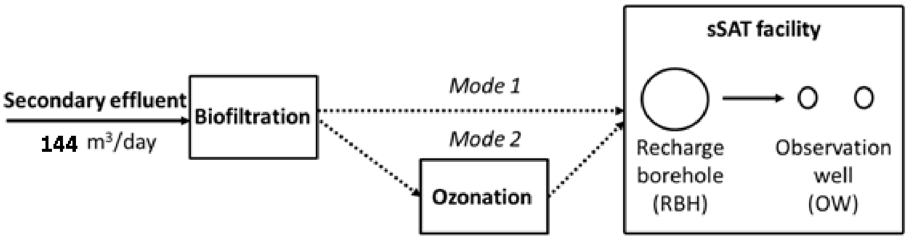

Figure 8 A general scheme of the pilot system at Shafdan, including operation in two modes: Mode 1 without or Mode 2 with ozone treatment.

Table 16 Overview of sampling campaigns

Experiment	Sampling campaigns	Sampled stations	Sampled parameter group
Chemical dosing Optimization		Secondary effluents, After Biofilter, after Ozonation	Turbidity, pH, NH_4, DOC, UVA, TrOCs
Mode 1 – Infiltration of Biofiltered effluents	7 campaigns from March 2014 to April 2015	Secondary effluents, After Biofilter, OW1, OW2	Field measurements, N-compounds, DOC, UVA, DO, Mn, TrOCs, Other
	12 campaigns from May 2014 to October 2015	Secondary effluents, Before Biofilter, After Biofilter	Particle analysis
	3 campaigns from January to April 2015	Secondary effluents, After Biofilter, OW1, OW2	Bacteriology
	1 campaign on February 2015	Secondary effluents, After Biofilter, OW1	Cryptosporidium, Giardia
	1 campaign on May 2015	Secondary effluents, After Biofilter, OW1	Viruses (human and bacteriophage)
	2 campaigns on June-July 2015	Ceramic cups	Field measurements, N-compounds, DOC, UVA, DO, Mn, TrOCs
Mode 2 – Infiltration of Biofiltered-Ozonated effluents	9 campaigns from August 2015 to September 2016	Secondary effluents, After Biofilter, After ozonation, OW1, OW2	Field measurements, N-compounds, DOC, UVA, DO, Mn, TrOCs, Other
	6 campaigns from May 2014 to October 2015	Secondary effluents, After Biofilter, After ozonation	Particle analysis
	5 campaigns from August 2015 to August 2016	Secondary effluents, After Biofilter, After ozonation, OW1, OW2	Bacteriology
	5 campaigns from January to September 2016	Ceramic cups	Field measurements, N-compounds, DOC, UVA, DO, Mn, TrOCs

4.2.2 Biofiltration unit

The biofiltration unit was fed continuously with 6 m^3/h (144 m^3/d) 500-µm-filtered secondary effluent from the Shafdan WWTP. The unit included coagulation/flocculation, which was carried out by injecting polyaluminum chloride (PACl, 18% Al_2O_3) diluted with water in-line by peristaltic pump to achieve a final concentration of ~2.7-3.6 mg/L as PAC in a 1.3 m^3 flocculator tank with ~15 min HRT. Following flocculation, ~27 mg/L H_2O_2 was added to the filter influent to provide oxygen for microbial processes (catalase enzyme converts peroxide to oxygen inside the filter). H_2O_2 was chosen as an oxygen source due to the high efficiency in supplying readily available oxygen throughout the biofilter volume with minor losses.

High-rate biofiltration was operated in a modified active dual-media filter (1.2 m diameter tank, 1 m media height) with ~5 m/h filtration velocity (~10 min HRT) and a backwash cycle of 12 h. Periodic backwashing included combined air/water followed by water backwashing and adjusted to avoid clogging of the filter concurrently with maintaining a steady and active biomass, as observed by the stable performance with low effluent turbidity (less than 0.8-1.0 NTU at the biofilter outlet throughout the whole year; in summer less than 0.6 NTU). The filter consisted of 1 m of multilayer media: a 10-cm basalt support layer (two 5-cm sub-layers of 2.36-4.75 and 0.60–1.18 mm particle size range), a 30-cm basalt layer (0.16–0.50 mm), and a top 60-cm layer of anthracite (0.80-1.50 mm). No chlorine was used during the backwash to maintain bioactivity in the filter. The microbial community of the biomass is indigenous therefore not artificially inoculated, and developed over time by feeding the biofilter with secondary effluent wastewater.

4.2.3 Ozonation unit

The biofiltration outlet was fed into an ozonation system (Figure 9). The ozonation unit (100 g/h, OCS GSO 30, Xylem Services GmbH, Germany) was operated in continuous mode with an ozone consumption of 10 mg/L (1.0–1.2 mg O3/ mg DOC), calculated by:

$$Z = \frac{\left(c_{o3,\,in} - c_{o3,\,out}\right) \cdot Q_{gas}}{Q_W}, (\text{mg L}^{-1})$$

Where Z represents the ozone consumption (mg/L), $C_{O3,\,in}$ and $C_{O3,\,out}$ represent the inlet and outlet ozone concentration in the gas phase (g/m^3) respectively, and Q_{gas} and Q_w represent the gas and the water flows respectively.

Ozone was produced from pure oxygen (~12 L/min, Q_{gas}) and introduced by venturi injection into the effluent stream (~4.3 m^3/h, Q_w). A 500-L reaction tank provided an HRT of ~7 min. In-gas and off-gas ozone was measured using an ozone gas analyzer (BMT 963, Germany) to set up a complete mass balance. The system included monitoring of pH/ORP (oxidation–reduction potential) and dissolved ozone (Hefer systems and controls, Israel). The ozonated effluents flowed into a 10-m^3 reservoir before infiltration to ensure complete ozone depletion. The ozone consumption (Z), calculated from the equation above was generally set on 9-11 mg/L.

At the final stage, the oxygen-rich treated effluents from the ozonation pilot unit were further infiltrated into the short SAT facility.

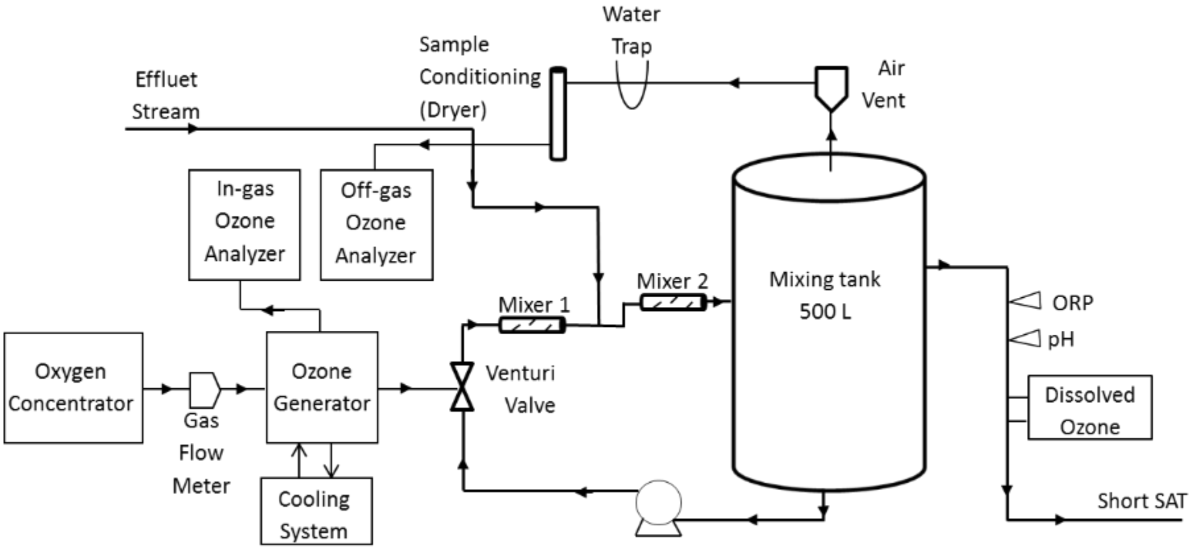

Figure 9 Ozone pilot unit scheme.

4.2.4 Short SAT facility

4.2.4.1 Study area and hydrogeology

The Shafdan main facilities and the pilot plant are located in the coastal aquifer which serves as one of the main water resources in Israel. This aquifer is divided to two main sub-aquifers: (1) Upper, a perch aquifer, which is mainly composed from calcareous sandstones interbedded below local sands, and (2) Lower aquifer, which is composed with calcareous sandstone and serves as the regional aquifer. These sub-aquifers are separated by a brownish clay layer (see Figure 10).

A pilot "Dug well" site including three observation wells in the saturated zone and three ceramic (porous) cups for water sampling in the unsaturated zone were constructed in 2006 as part of RECLAIM WATER EU project (Figure 10). The "Dug well" is a cylindrical perforated tube (3.6 m diameter, 10 m^2 surface area) drilled into the unsaturated zone of local sands and calcareous sandstone. The depth of the Dug well is 2.5 m, so that an unsaturated zone of about 12-13 m was left.

Three observation wells (R-1, R-2, and R-3) were set up downstream of the Dug well (Figure 10, Figure 11, Figure 12). Well R-1 was drilled to a depth of 27.5 m in a distance of 7.3 m from the Dug well. A deeper well, R-2, was located 5.1 m downstream to well R-1 and was drilled to a depth of 54 m. Up to a depth of 32 m the lithological section is composed of calcareous sandstone with white sand similar to the section in well R-1. Below 32 m the section is composed of clay (sandy clay and black clay). This clayey layer ends at a depth of 46 m and separates two sand sequences. Below 46 m the section is composed of calcareous sandstone. Well R-2 was drilled to see if there is a clayey layer between depths 27-46 m. Due to an impermeable clay layer which ended at a depth of 46 m, a separation of the upper-most relevant sub-aquifer was observed, which prevented the infiltration of the treated effluent in this well. Thus, well R-2 was not used in the previous and the current experiments. A third well, R-3 was located 10.3 m downstream to well R-1 and was drilled to a depth of 28.3 m, same layers as in well R-1 (Figure 10, Figure 11, Figure 12).

In the current project, Shafdan secondary effluents after pretreatment (*Mode 1* - biofiltration alone; or *Mode 2* - biofiltration followed by ozonation) were infiltrated at the Dug well at a regime of 12h infiltration (at 5 m3/h) and 12h drying (intermittent infiltration, similar to Shafdan SAT regime).

The observation wells R-1 and R-3 (termed in the current project *OW1* and *OW2*, respectively) were used to assess the performance and the resulting water quality at the end of the suggested process. OW1 represented the targeted final product of short SAT, in which the objective was to get good chemical and microbial quality, including TrOCs removal and minimum Mn^{2+} dissolution, in a short HRT (~22 days), and high infiltration rate (~4-6 m/d). OW2 was investigated as a control of longer SAT (~113 days HRT), and in order to understand the hydro-chemical processes in the aquifer at the area. This well is much more affected by the surrounding non-treated secondary effluent flows and the reaction time in this well is relatively long. In addition, in May 2015, three ceramic cups were installed at a 1.5 m depth under the Dug well, in the unsaturated zone, to examine the hydro-chemical processes occurring at the vadose zone (Figure 13). The HRT in the ceramic cups (total of 60 m^3/day infiltrated effluents, 10 m^2 area dug well, 1.5 m depth of ceramic cups under the dug well) is about 6 hours.

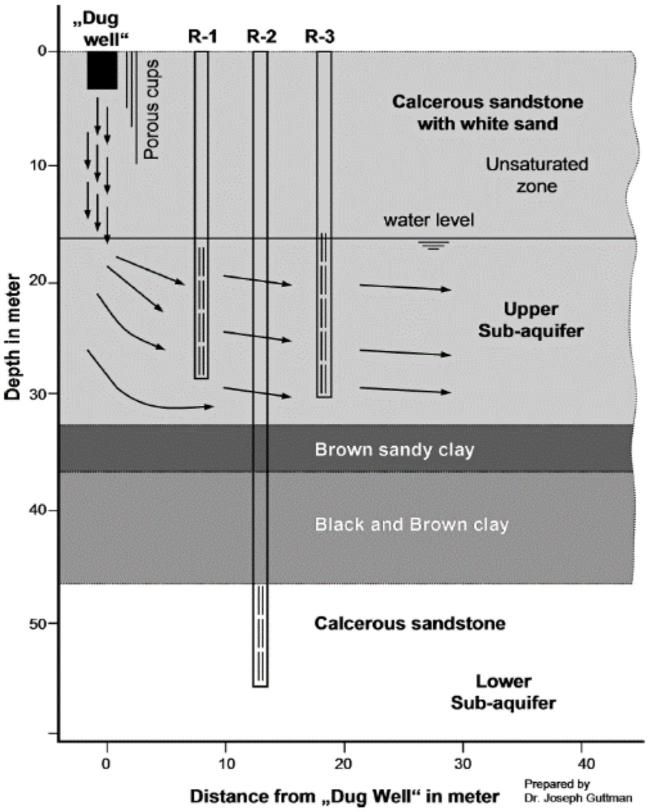

Figure 10 A technical geological sketch describing the dug well and the monitoring wells at the Shafdan pilot site (sources: Gaus et al. 2007; Cikurel et al. 2012)

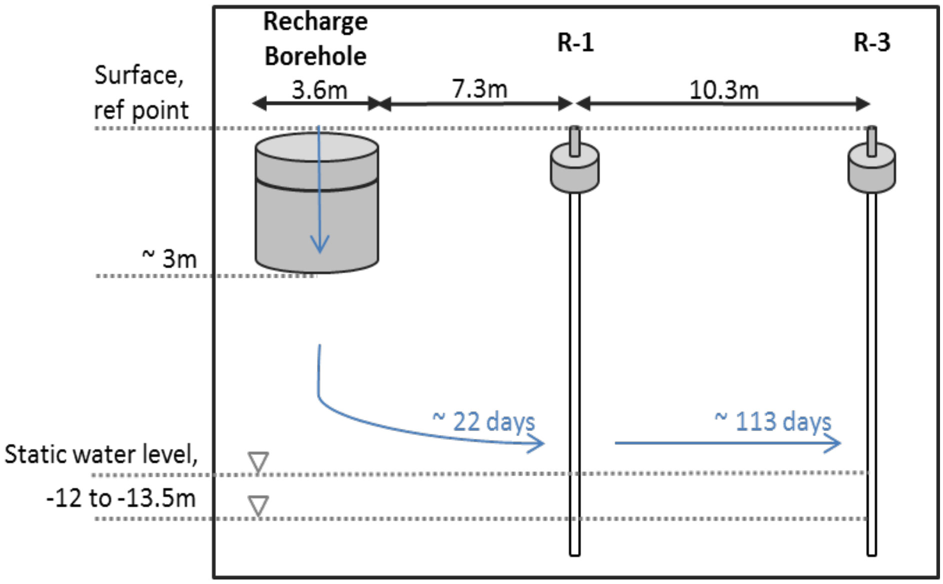

Figure 11 Recharged effluent flow path through SAT facility (unsaturated zone).

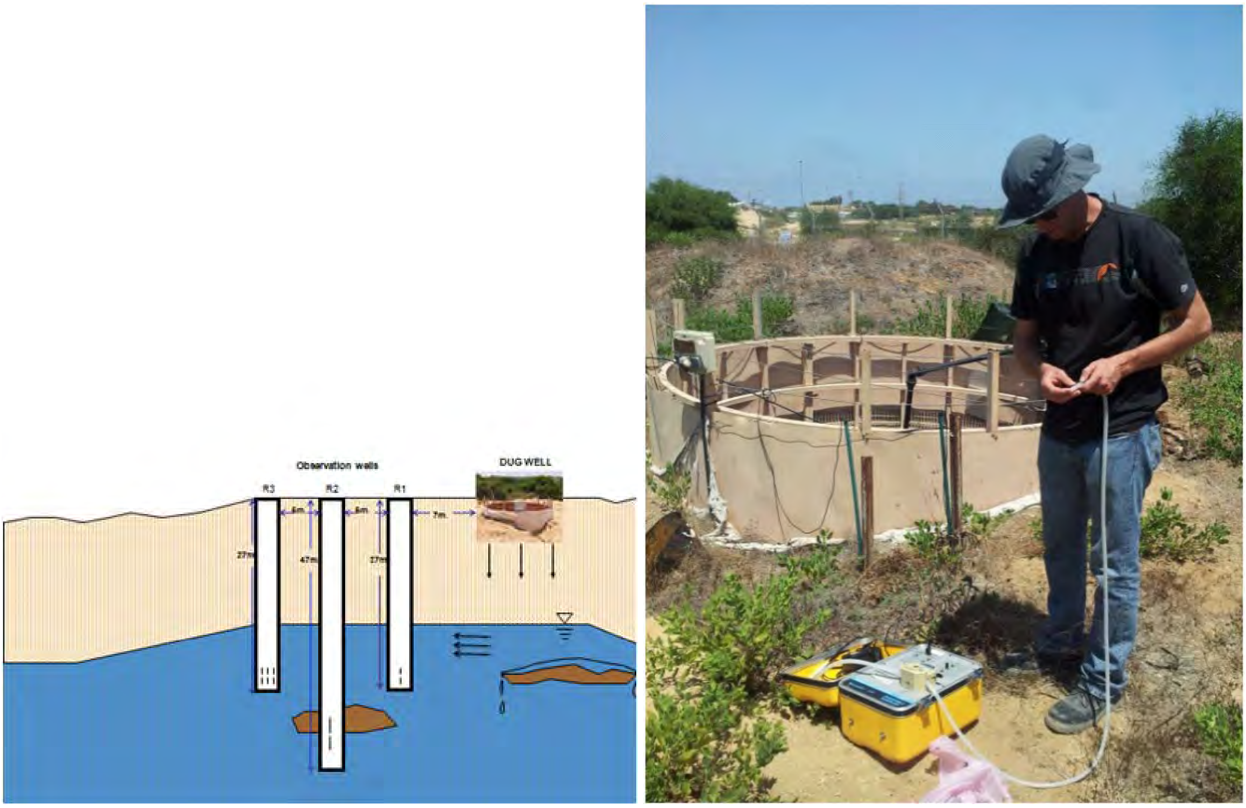

Figure 12 Dug well (right) and three observation wells (left, R-1, R-2, R-3).

Figure 13 A porous ceramic cup.

4.2.5 Bromide tracer test

The hydraulic retention time of the recharged pretreated secondary effluents during infiltration was determined using a bromide salt tracer. The bromide (Br) was injected at a 12h injection 12h drying regime for 10 days. The tracer was recharged into the Dug well to characterize the vertical flow velocities of the infiltrating water. The injected concentration was around 30 mg/L Br. The concentration of bromide was measured periodically in the observation wells. The curves of the concentration in OW1 and OW2 were typical breakthrough curves. The breakthrough point in OW1 and OW2 were around ~22 and ~113 days, respectively (Figure 14). Hydrologically, as the infiltrate effluents moves away from the Dug well, it is more likely to be mixed with natural background and therefore reduce the tracer concentration. However, bromide concentration in OW2 was found to be similar to that in OW1, indicating plausibly flow in one direction. The results show that there was little dilution since almost 80-90% of the tracer showed up after the above mentioned retention times in the wells.

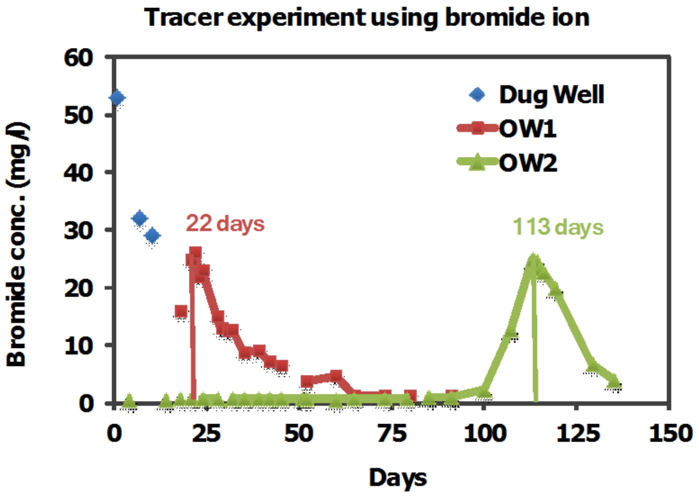

Figure 14 Bromide tracer concentrations (mg/L) at the Dug well, OW1, and OW2.

4.2.6 Microbial analytical methods

4.2.6.1 Bacteria

Total Bacterial count (TOTB) was conducted using Standard Method 9215B (Standard Methods, 22nd Edition, APHA, AWWA, WEF). Total coliforms, fecal coliforms, and fecal streptococcus m-enterococcus counts were conducted by a multiple-tube fermentation (MTF) technique, where results are reported in most-probable number (MPN) (quantification) of organisms present. This simple, compact and statistical method based on lactose fermentation includes two main selection stages (presumptive and confirmed). The presumptive stage is common for both total and fecal coliforms (i.e. CMP, SM 9221), while the second stage used for identifying indicators bacteria, total coliforms (i.e. CMPN, SM 9221B), fecal coliforms (i.e. FMPN, SM 9221E) and fecal streptococcus m-enterococcus (i.e. EMPN, SM 9230B).

4.2.6.2 Virus

Virus analysis included total active cell cultures based on 3 cell line cultures: (a) human embryonal rhabdomyosarcoma – RD (Bell and Cosgrove, 1980), (b) buffalo green monkey – BGM (Barron et al., 1970), and (c) HUKI – a sensitive tissue for cell cultures. While the first two lines are well established ATCC methods recommended by the EPA, the third line is a local method developed by the central virology laboratory. Real time PCR molecular methods were used to identify four groups of human viruses (Adeno, Noro, Entero, and Parecho) following concentration of viruses from large volumes (SM 9510C). Total coliphages were measured based on common assay method employs the inoculation of a susceptible E. coli strain (USEPA, 2001). This analysis was carried out by the Ministry of Health main, Central Virology Laboratory at Tel-HaShomer, Israel.

4.2.6.3 Protozoa

Cryptosporidium and Giardia counts were conducted using USEPA1623.1 method.

4.2.7 Chemical analytical methods

4.2.7.1 Field measurements

- Turbidity measurements were conducted using Hach 2100.
- DO, temperature and ORP measurements were conducted using MULTI 3420.
- pH, conductivity and temperature measurements were conducted using 532CON.

4.2.7.2 Kit measurements

H_2O_2 measurements were conducted using NOVA 60 kit

4.2.7.3 Lab measurements

The following lab measurements were conducted using Standard methods (Standard Methods, 22nd Edition, APHA, AWWA, WEF) as follows:

- Ammonium: Selective Electrode 4500-NH3 D.
- Nitrite: 4500-NO2-B Colorimetric Method.
- Nitrate: 4110B Ion Chromatography with Chemical Suppression of Eluent Conductivity.
- Kjeldahl N: 4500-Norg B Macro-Kjeldahl Method.
- Phosphate: Stannous Chloride Method 4500-P D.
- UVA: 5910 B Ultraviolet Absorption Method.
- DOC: 5310B High-Temperature Combustion Method.
- Alkalinity: titration method 2320 B.
- Chloride: 4500-Cl-D Potentiometric Method.
- Manganese, Ferric, Calcium, Magnesium, Total phosphorous: 3120B Inductively Coupled Plasma (ICP) Method.
- Hardness: 2340B Hardness by Calculation.
- Bromide: 4110B Ion Chromatography with Chemical Suppression of Eluent Conductivity.
- Bromate: 4110D Ion Chromatographic determination of Oxyhalides and Bromide.
- Dissolved oxygen: 4500-O C Azide Modification.
- COD: 5220B Open Reflux Method.
- BOD_5: 210B 5 Day BOD Test.
-

4.2.7.4 Trace Organic Compounds (TrOCs), chemicals and standards.

TrOCs analyses were conducted at the Hydrochemistry lab, Tel-Aviv University, headed by Prof. Avisar.

The TrOCs monitored in this study were selected based on their rate of reaction with molecular ozone (k_{O3}) and their relevance for SAT systems. TrOCs can be categorized into three groups: rapidly ($k_{O3} > 10^4$ $M^{-1}s^{-1}$), moderately (10 $M^{-1}s^{-1} < k_{O3} < 10^4$ $M^{-1}s^{-1}$) and slowly ($k_{O3} < 10$ $M^{-1}s^{-1}$) reacting compounds. All standards (purity > 99%) were obtained from Sigma-Aldrich (with the exception of iopromide, provided by Holland-Moran, Israel), and used as purchased. TrOC classes and rate constants for the reaction with ozone and hydroxyl radicals (kOH) are summarized in Table 17.

Table 17 Class, chemical characteristics and rate constants at pH 7 for reaction of the selected TrOCs

Name	Class	Rate constants		References
		k_{O3} [$M^{-1}s^{-1}$]	k_{OH} [10^9 $M^{-1}s^{-1}$]	
Iopromide (IOP)	Contrast media	0.8	3.3	(Huber et al., 2003)
Iohexol (IHX)		1.4	3.3	(Baus et al., 2004)
Iopamidol (IPDL)		1.4	2.8	
Bezafibrate (BZF)	Lipid regulator	590	7.4	(Huber et al., 2003)
Venlafaxine (VLX)	Antidepressant	$3.3{\cdot}10^4$	8.8	(Lester et al., 2013; Wols et al., 2013)
Carbamazepine (CBZ)	Anti-epileptic	$3{\cdot}10^5$	8.8	(Huber et al., 2003)

Name	Class	Rate constants		References
		k_{O3} [$M^{-1}s^{-1}$]	k_{OH} [10^9 $M^{-1}s^{-1}$]	
Diclofenac (DCF)	Anti-inflammatory	$1 \cdot 10^6$	7.5	(Huber et al., 2003)
Sulfamethoxazole (SMX)	Antibiotic	$2.5 \cdot 10^6$	5.5	(Huber et al., 2003)

4.2.7.5 TrOCs sample extraction and analysis

To detect the presence of TrOCs at low concentrations in a pilot-scale application (without spiking) following ozonation and infiltration, samples were pre-concentrated by solid-phase extraction (SPE), using 500-mg, 6-mL Oasis-HLB cartridges connected to a 12-port SPE manifold and vacuum pump. The SPE cartridges were conditioned with 10 mL methanol and 10 mL DI water. After loading the samples, the cartridges were dried under a 5/9 nitrogen stream for 10 min. The analytes were eluted with 6 mL methanol, collected in 10 mL glass tubes and dried to zero under a nitrogen stream. Finally, the extracts were reconstituted with 1 mL HPLC-grade water (with 10% v/v methanol) at pH 3 and transferred to HPLC vials.

TrOCs were detected by HPLC (Agilent 1100 series; ACE-RP phenyl column 2.1 mm × 250 mm) coupled with a mass spectrometer (Q-Tof MS, Waters Premier). The HPLC flow rate was 0.5 mL/min and the injected volume was 100 µL. The mobile phase consisted of water (A) and methanol (B), adjusted to pH 3 with formic acid. The eluent gradient started with 10% eluent B for 1 min, followed by a 4-min linear gradient to 90% B, a 5-min isocratic elution at 90% B, and a 2-min linear gradient back to 10% B, maintained for 4 min for equilibration. The Q-Tof MS was operated in positive mode, with limit of quantification (LOQ) of 0.1 µg/L.

CBZ was also measured several times at Mekorot laboratory by SPE and GC/MS using EPA 525 method.

4.2.8 Particle Analysis.

Particle analyses were conducted at the laboratory of Water Technologies, Tel-Aviv University, headed by Prof. Mamane.

Effluent particles were analyzed by Micro Flow Imaging technology (DPA 4100, Protein Simple Inc., Canada). This apparatus employs a digital camera with an illumination and magnification system to capture in-situ images of suspended particles in a flowing sample. A detailed description of this analysis is published elsewhere (Lakretz et al., 2014). In brief, a fluid sample is drawn through a flow cell, illuminated with a light-emitting diode at 470 nm wavelength, and the magnified image is captured by digital camera. This image was automatically analyzed to determine the particles' equivalent circular diameter (ECD) which represents the diameter of a sphere that occupies the same two-dimensional surface area as the particle. Analysis was conducted on particles of between ~2 and 400 µm.

4.3 Results - technology Performance

4.3.1 Optimization of chemical dosages

4.3.1.1 PACl dosing optimization

PACl was added to Shafdan secondary (500 µm filtered) effluents at doses of 0.0-7.2 mg/L. Modified Jar-test included direct filtration via coagulation, flocculation and Buchner filtration. Experimental set up included fast mixing (1 min, 300 rpm), slow mixing (20 min, 30 rpm), and then filtration on 11µm Whatman filter paper. The optimal concentration found was 3.6 mg/L as PACl when combining results both for filtrate turbidity and the optimal filtered volume after various times of filtration (Table 18).

Table 18 Turbidity, pH and filtered volume as a function of PACl addition

PACl, mg/L	0	2.7	3.6	5.4	7.2
Turbidity after Buchner	0.83	0.39	0.29	0.22	0.12
pH with PACl addition	7.66	7.41	7.36	7.27	7.25
Vol – 8 min post filtration	530	450	550	410	370
Vol – 50 min post filtration	1000	860	820	780	580

4.3.1.2 H2O2 dosing optimization

H_2O_2 was added to supply oxygen demand for the biological process in the biofilter. Ammonia was measured as a control parameter to the nitrification process. A set of H_2O_2 concentrations was applied ranged between 0-27 mg/L. The minimal H_2O_2 concentration which supplied oxygen demand for full nitrification and elimination of up to ~4-5 mg/L ammonia at the biofilter was found to be ~27 mg/L (Figure 15). This value depends on ammonia concentrations in the biofilter influent (i.e. secondary effluents) which varies daily and the on the weather which may affect the biofilter biomass.

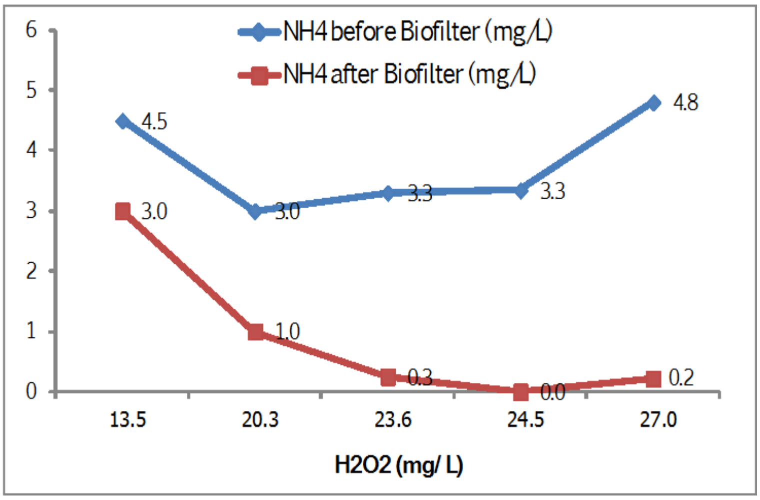

Figure 15 NH4+-N removal as a function of H2O2 addition.

4.3.1.3 Ozone dosing optimization

An experiment was set in the pilot plant to determine optimal ozone consumption (Z) values. Dissolved ozone, UV absorbance (UVA), dissolved organic matter (DOC) and TrOCs were measured.

Turbidity, pH, DOC and temperature were not significantly affected by ozonation. At the dose of 6.93 mg/L ozone, UVA was reduced by 50% (Figure 16). Dissolved ozone was also changed with ozone dose higher than 6.93 mg/L from negligible values to 0.1 mg/L.

Among the different TrOCs that were measured, the fast and moderate reacting compounds were degraded to below LOQ at 6.93 mg/L ozone dose (Figure 17), and the slow reacting compounds showed low degradation during ozonation, with a significant degradation rate following 6.93 mg/L ozone dose (Figure 18).

Although ~7 mg/L ozone doze was found to be the optimal dose for the current experimental set-up, in order to get significant results in OW1 and OW2 in the limited time of the project, ~10 mg/L ozone dose was used for the long-term infiltration trial (*Mode 2*).

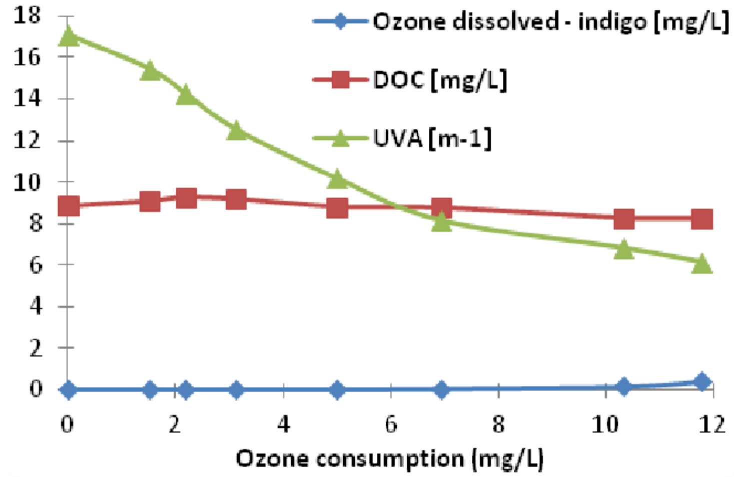

Figure 16 Dissolved ozone, DOC and UVA as function of ozone consumption.

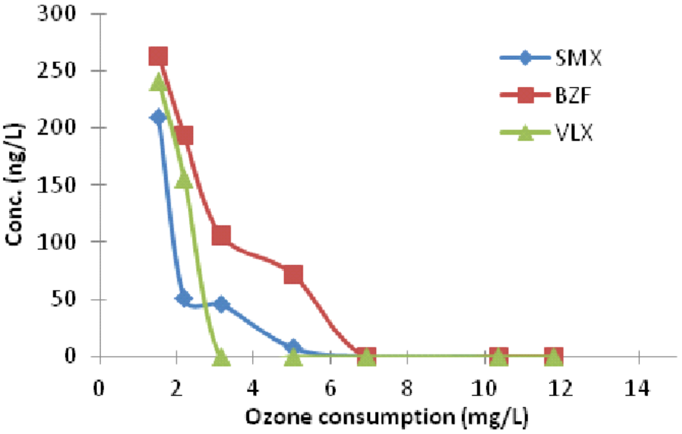

Figure 17 Concentrations of fast and moderate reacting compounds versus ozone consumption.

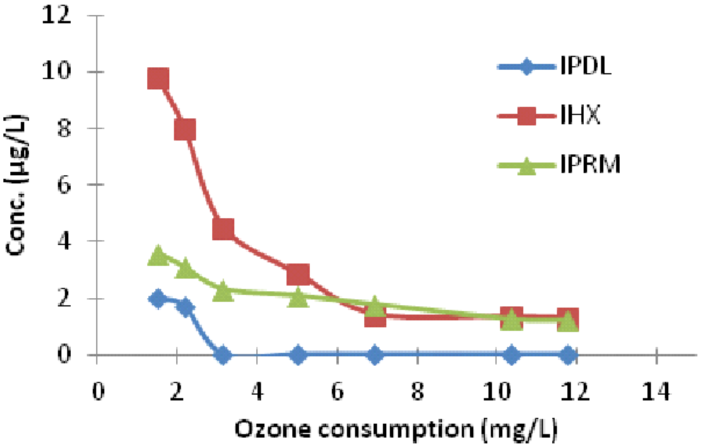

Figure 18 Concentrations of slow reacting compounds versus ozone consumption.

4.3.2 Mode 1: Infiltration of biofiltered effluents (March 2014-July 2015)

The major objectives for addition of biofiltration to the pretreatment process prior to ozonation and short SAT were to reduce clogging by wastewater particles and to minimize oxygen demand during infiltration by removing ammonium and some organic matter. In addition, the removal of nitrite and particles by the biofilter decreased ozone demand.

In general, *Mode 1* operation (infiltration of biofiltered effluents without ozonation) was conducted as a control experiment to see the effect of the biofilter alone on the product quality, without the effect of ozonation. However, it is important to note that the biofiltration stage is required only when there is a high residual of ammonium after secondary treatment (CAS), which highly consumes oxygen in the vadose zone and thus may lead to reduced SAT efficiency and mobilization of Mn^{2+} (Elkayam et al., 2015). Since the addition of hydrogen peroxide (~27 mg/L) increases the operational cost, and operation in ~5 m/h filtration rate increases the footprint and the capital cost of the process, in case of good nitrification during CAS, a conventional media filter of ~10 m/h, added with in-line coagulation, could be enough as a pretreatment prior to ozonation in order to remove particles and turbidity (see section 4.4, capital and operational costs).

4.3.2.1 Particle analysis

Particle Size Distribution (PSD) of Secondary effluents is presented in Figure 19 and includes results from May 2014 to October 2015. The distribution of particles by size was similar for both winter and summer samples, with more than 99% of the particles smaller than 50 µm, and the highest fraction at ECD = 2–3 µm. Particles smaller than ~2 µm were below the detection limit of the analytical instrument and could not be quantified. In general, particle count was higher in the winter than in summer, for all ECD ranges (with some minor exceptions). This trend is probably due to inhibition of the biological treatment by the cold weather (average winter temperature = 14±2 ˚C; average summer temperature = 24±4 ˚C), resulting in low sludge-settling efficiency. Based on the PSD analysis, the study focused on particles smaller than 50 µm.

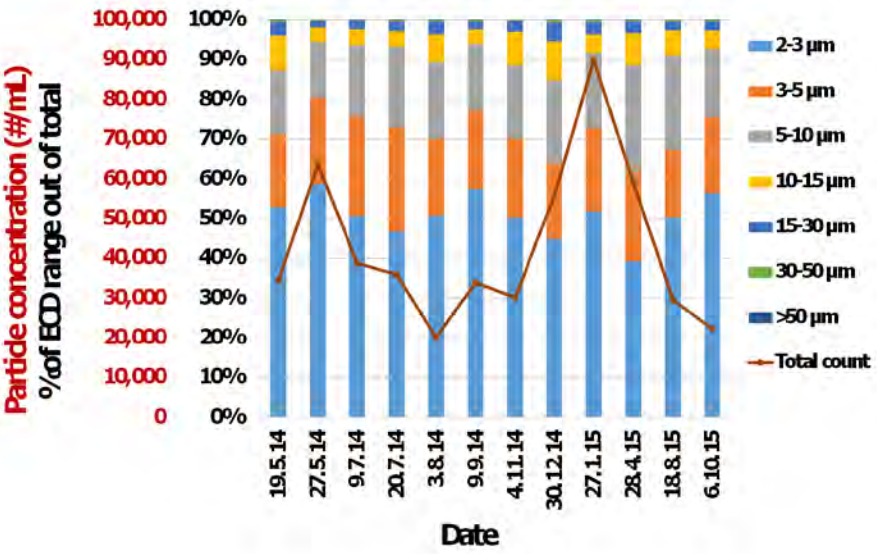

Figure 19 Particle Size Distribution of Secondary effluents.

Typical PSD analysis and turbidity values (conducted in 04/11/2014) are presented in Table 19 and in Figure 20. The points sampled were: secondary effluents, before biofilter (after flocculation) and after biofilter. It can be seen that flocculation before biofiltration increased particle counts. As expected, particle concentration decreased dramatically after coagulation/flocculation and biofiltration for all ECDs. The total particle removal after biofiltration was found to be 95±2% for all sampling campaigns.

Table 19 Typical Particle Size Distribution (PSD) of Secondary Effluents, before and after Biofilter (04/11/14)

Particle size µm	Secondary effluents	Before Biofilter	After Biofilter
2-3	15,171	19,367	1,170
3-5	5,957	8,064	606
5-10	5,559	5,657	446
10-15	2,516	1,531	63
15-30	820	996	20
30-40	73	109	3
40-50	19	42	1
50+	8	32	3
Total	30,122	35,798	2,313
Turbidity	1.98		0.46

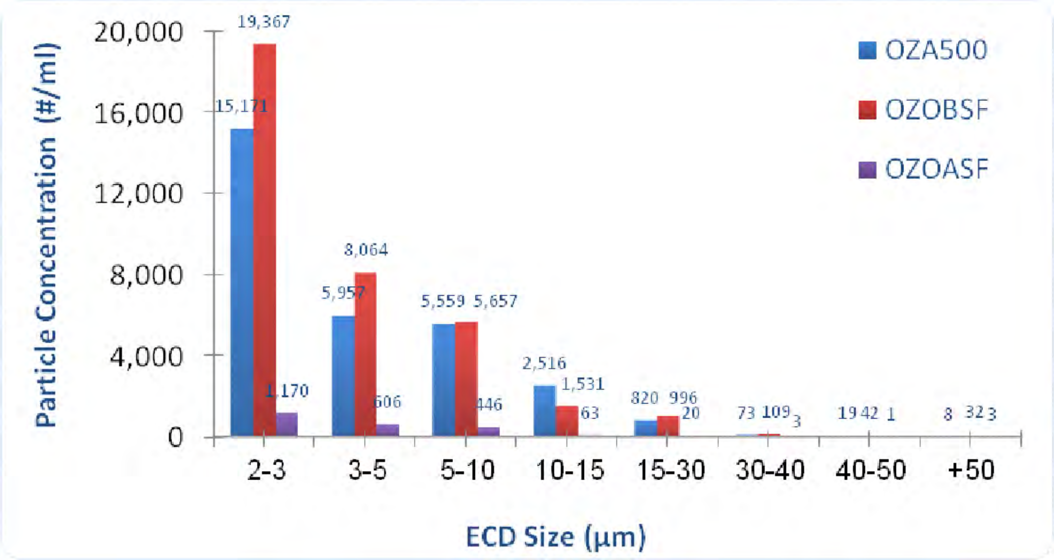

Figure 20 Typical PSD of Secondary Effluents (OZA500), before (OZOBSF) and after (OZOASF) Biofilter (04/11/14).

4.3.2.2 Bulk chemistry

Table 20 presents average values of chemical parameters measured at the different sampling points (secondary effluents, after biofilter, OW1, OW2) during *Mode 1* (infiltration of biofiltered effluents, without ozonation). It can be seen that turbidity was removed by 78% after biofiltraion, and by additional 33% after infiltration in short SAT (OW1, ~22 days HRT). Average DO concentrations of 3.7 mg/L in the biofilter effluent indicates oxic conditions throughout the filter. However, it can be seen from the standard deviation that there were fluctuations in these concentrations.

Although DO concentration fluctuated due to variations in effluent quality, in most cases it was more than 2–3 mg/L, which is commonly suggested as the threshold value for denitrification or anaerobic bio-activity (Kremen et al., 2005).

In OW1, it can be seen that during *Mode 1*, the average DO was higher than 2 mg/L most of the time (2.2 ± 1.1 mg/L in average). This could be explained by the effective removal of all ammonia (up to 4-5 mg/L, section 4.3.2.3) and removal of around 24% DOC by the biofilter pretreatment (section 0). As mentioned earlier, ~10 mg/L DOC and 4-5 mg/L ammonia theoretically consume 40-50 mg/L DO, while ammonia would consume between 20-25 mg/L DO (Elkayam, 2015). Hence, it seems that more than 50% of the DO consumption was reduced and thus enabled the entrance of more DO to the groundwater, reaching OW1. In OW2, however, the DO was lower than this observed in OW1 (1.2 ± 0.4 mg/L in average, lower than 1.8 mg/L at all measurements).

It is important to note that the aquifer under the Shafdan pilot site is unconfined. Thus, secondary effluents from the surrounding reservoirs could flow into the two observation wells (OW1 and OW2). From the current results it seems that infiltration of ~60 m3/day of treated effluents (which are free of ammonium and rich with DO) can significantly affect the closer perimeter (OW1, ~22 days HRT) and less affect the farer perimeter (OW2, ~113 days HRT). Results during *Mode 1* infiltration showed lower DO concentration and higher Mn concentration in OW2 compared to OW1. This could be explained by either natural minerals in aquifer causing oxygen decline or by other sources of organic carbon (e.g. secondary effluents in the area) that may intrude OW2, and less likely intrude OW1. In addition, the redox conditions in the soil are known to be changed with distance. Thus, Mn could be precipitated and dissolved while DO is consumed in the traveling distance between OW1 and OW2.

Secondary effluents were characterized by low BOD (5.3 ± 1.2 mg/L) as compared to 41.3 ± 5.5 mg/L COD, indicating that during Activated Sludge treatment most of the biodegradable matter was successfully removed. However, the ratio of ~0.1 BOD/COD shows that there was still non-biodegradable material to treat. Biofiltration successfully removed the rest of the BOD (~88%) while some removal of COD also occurred (~38%). Thus, the BOD/COD ratio after biofiltration further decreased from 0.13 ± 0.01 to 0.02 ± 0.02 after biofilter, meaning that almost all biodegradable matter was removed (leaving non-biodegradable matter that will be biodegraded by ozonation in *Mode 2*). After short and longer SAT (OW1, ~22 days HRT), BOD was completely removed, while the hardly biodegradable matter measured by COD decreased by ~85, due to the longer retention times in the unsaturated and saturated zones, involving two mechanisms- biodegradation and adsorption. BOD and COD obtained in OW2 (~113 days HRT) were similar to those obtained in OW1.

Phosphate concentrations (mg/L as P) in secondary effluents were relatively low (~0.7 mg/L), and were significantly removed after biofiltration followed by short/longer SAT (OW1/OW2). As expected, parameters like alkalinity, chlorides, calcium, magnesium and hardness did not significantly change along the process, since they were not affected by it. Table 20 presents a presentative measurements from September 2014 (Alkalinity, Hardness, Mg, Ca) and April 2015 (Alkalinity, chlorides).

Table 20 Chemical parameters at different sampling points during Mode 1 (infiltration of biofiltered effluents

Mode 1				
	Secondary Effluents	After Biofilter	OW1 (~22d HRT)	OW2 (~113d HRT)
Turbidity (NTU)	2.7 ± 0.6 (n=6)	0.6 ± 0.1 (n=6)	0.4 ± 0.3 (n=7)	0.4 ± 0.5 (n=7)
DO* (mg/L)	3.7 ± 1.0 (n=3)	3.7 ± 3.6 (n=3)	2.2 ± 1.1 (n=6)	1.2 ± 0.4 (n=6)

Mode 1				
	Secondary Effluents	After Biofilter	OW1 (~22d HRT)	OW2 (~113d HRT)
BOD (mg/L as O2)	5.3 ± 1.2 (n=3)	0.7 ±0.6 (n=3)	<1.0 (n=4)	<1.0 (n=4)
COD (mg/L as O2)	41.3 ± 5.5 (n=3)	25.7 ± 4.2 (n=3)	3.9 ± 0.1 (n=2)	3.55 ± 0.8 (n=2)
BOD/COD	0.13 ± 0.01	0.02 ± 0.02		
BOD/DOC	0.45 ± 0.05	0.07 ± 0.06		
PO4 (mg/L as P)	0.7 ± 0.2 (n=2)	0.6 (n=1)	<0.5 (n=2)	<0.5 (n=2)
Alkalinity as CaCO3 (mg/L)	226.5 ± 3.5 (n=2)	207.0 (n=1)	233.0 ± 14.1 (n=2)	230.0 ± 5.7 (n=2)
Cl^- (mg/L)	216.0 (n=1)	216.0 (n=1)	180.0 (n=1)	206.0 (n=1)
Ca (mg/L)	64.3 (n=1)		79.0 (n=1)	89.0 (n=1)
Mg (mg/L)	16.6 (n=1)			
Hardness as CaCO3 (mg/L)	228.9 (n=1)		265.0 (n=1)	348.0 (n=1)

* Measured in the lab by 4500-O C Azide Modification

4.3.2.3 N-compounds

Figure 21 shows the profile of the nitrogen compound concentrations analyzed at the different sampling points during *Mode 1*. Secondary effluents contained average concentrations of 3.3 ± 2.1 mg NH_4^+-N/L, 0.9 ± 0.7 mg NO_2^--N/L, and 0.4 ± 0.2 mg NO_3^--N/L, indicating the incomplete nitrification process in the Shafdan secondary treatment. Results after biofiltration showed high evidence of nitrification including 73% removal of NH_4^+-N and 97% removal of NO_2^--N, and increase in nitrate up to 3.6 ± 0.4 mg NO_3^--N/L. Towards the end of *Mode 1*, two sampling campaigns of the ceramic cups (1.5m depth under the RBH, in the upper layer of the SAT) were performed. Further nitrification was observed at the vadose zone, with complete removal of ammonium and nitrire, and a slight increase in nitrate (up to 4.1 ± 0.7 mg NO_3^--N/L). No significant changes were observed after short (OW1) or longer (OW2) SAT. As was shown by Elkayam et al., (2015), most of the oxidative processes in the Shafdan SAT occur in the unsaturated zone, while ~54% of oxygen demand is consumed by ammonia and organic nitrogen species. Thus, apparently all ammonia was transformed to nitrate in the vadose zone. Thus, there was no much difference between the upper layer vadose zone and the ~22 days HRT observation well (OW1).

From organic N measurements it was observed that there was some loss of N in the biofilter plausibly due to anoxic–anaerobic regions since not all the pores in the media are free of solids (data not shown). The periodical backwash process succeeded to return the head loss to its initial value, but channeling could still exist. In the SAT system (OW1 and OW2), however, there was no significant loss of N. Thus, it seems that OW1 and OW2 were sufficiently aerobic to inhibit denitrification. These results are in accordance with the DO results presented in section 4.3.2.2. In addition, the high nitrate levels in OW1 and OW2 would slaw denitrification processes.

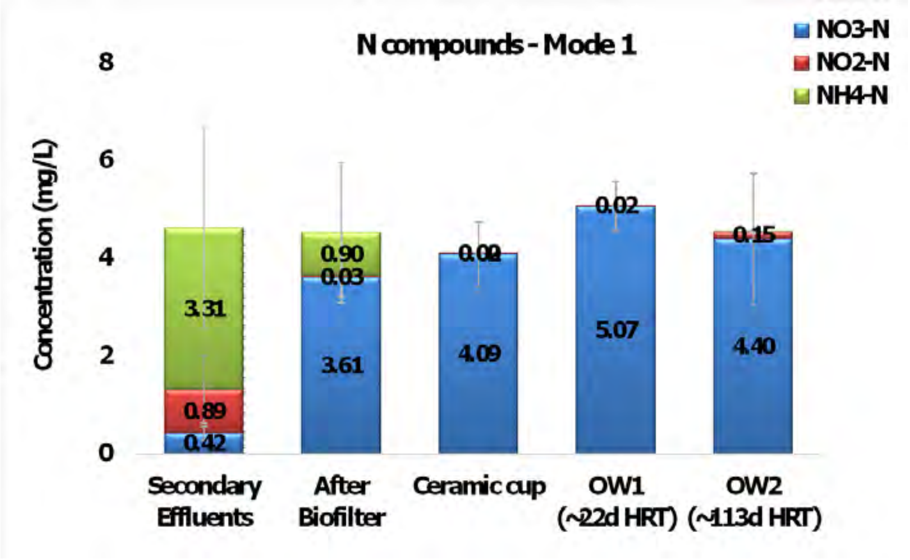

Figure 21 **Nitrogen compound concentrations at the different sampling points during Mode 1.**

4.3.2.4 DOC, UVA, SUVA

Figure 22 shows average DOC, UV absorbance at 254 nm (UVA), and Specific UV absorbance (SUVA) values obtained at the different sampling points during *Mode 1*. SUVA was determined as the ratio between UVA and initial DOC concentration. Trends in SUVA, as an index of aromaticity, could be detected along the different pilot stages.

It was shown that Shafdan secondary effluents contained average values of 11.8 ± 1.1 mg/L DOC and 22.4 ± 1.3 1/m UVA. Biofiltration decreased DOC and UVA by 24% and 20%, respectively, to 8.9 ± 1.1 mg/L, and 17.9 ± 1.5 1/m. Infiltration in the vadose zone (ceramic cups) further decreased DOC and UVA by 49% and 35%, respectively, to 4.5 ± 1.0 mg/L, and 11.6 ± 0.4 1/m. This decrease can be attributed to the biodegradation of organic matter in the soil. Further and more pronounced removal of DOC and UVA was obtained after short SAT (OW1) by further 69% and 63%, respectively, to 1.4 ± 0.1 mg/L, and 4.3 ± 0.2 1/m. No significant changes were observed after longer SAT (OW2, Figure 22). It is assumed that due to the lack of oxidation by ozone, there were still non-biodegradable aromatic compounds after biofiltration. The remained DOC after the upper layer of the SAT, which was mostly (slowly biodegradable) COD, was further decreased in the aquifer plausibly due to the combined effect of biodegradation due to high HRT and adsorption (Bouwer, 1996; Gerrity, 2011a).

Similar values of SUVA were obtained following biofiltration, meaning that although reducing DOC, biofilter did not change the DOC composition and its aromatic content. After infiltration at the vadose zone, there was some increase in SUVA from 2.0 ± 0.2 to 2.6 ± 0.7, and further increase after short SAT (OW1) up to 3.1 ± 0.2. No significant change was observed after longer SAT (OW2). The increase in SUVA during SAT, at the vadose zone and after retention time in the aquifer, indicates increase in aromatic compounds content of the DOC during degradation of organic matter. This could be explained by preferential removal of nonaromatic (non-humic) components (Amy and Drewes, 2007).

In general, Soil Aquifer Treatment (SAT) is done in three steps: surface infiltration, percolation through the unsaturated zone (vadose zone) and slow transport through the aquifer. In the SAT method, dissolved organic matter is removed by combined biological, chemical, and physical processes mainly in the vadose zone (the unsaturated zone). The vadose zone and aquifer act as natural, slow filters that effectively reduce the concentration of various pollutants.

Suspended solids are filtered out; biodegradable organic compounds are decomposed; microorganisms are adsorbed, strained out, or die because of competition with other soil microorganisms; synthetic organic compounds are adsorbed and/or biodegraded; and phosphate, fluoride, and heavy metals are adsorbed, precipitated, or otherwise immobilized. (Aharoni et. al, 2011).

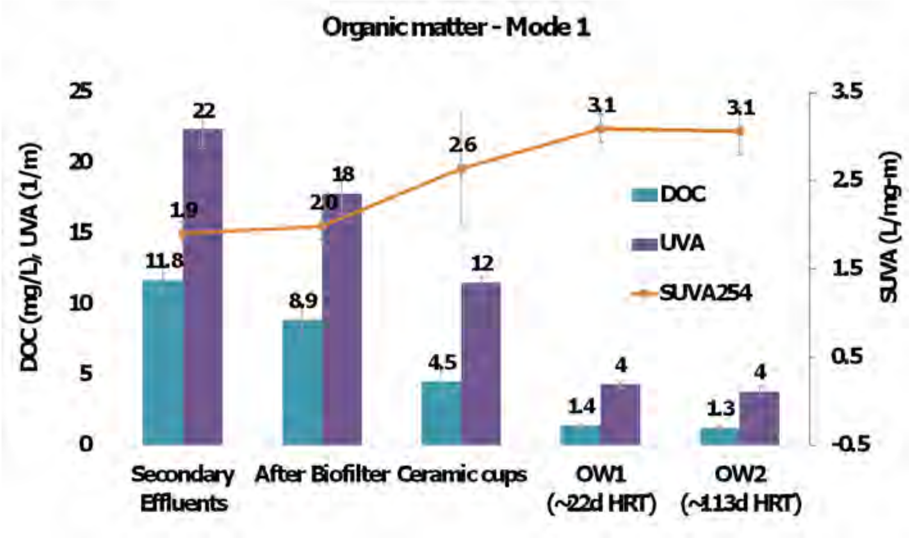

Figure 22 Organic matter-related parameters at the different sampling points during Mode 1 (HRT in biofilter: ~10 min., HRT in ceramic cups: ~6 hours).

4.3.2.5 Inorganic traces (Mn, Fe)

Figure 23 presents average concentrations of Mn (total) and Fe (total) at the different sampling points during *Mode 1*. In general, Mn concentrations in secondary effluents and after biofilter were relatively low (21.4 ± 3.8 and 16.2 ± 7.0 µg/L, respectively) with no significant removal by biofilter. Fe concentration in secondary effluents was 68.5 ± 15.3 µg/L, with ~50% removal by biofilter (to 32.3 ± 13.7 µg/L). Processes in the vadose zone showed removal of Mn up to 3.0 ± 4.2 µg/L (Fe measurements were not conducted in the vadose zone). After short SAT (OW1), some Mn was observed (average of 41.6 ± 9.7 µg/L), and in much higher concentrations after longer SAT (OW2, average of 456.4 ± 136.4 µg/L). No Fe was found in both OW1 and OW2. The results indicate some Mn^{2+} dissolution, especially in OW2, conceivably due to oxygen depletion and development of reducing conditions.

The Mn concentration found in OW1 after biofiltration maintained concentrations lower than 52 µg/L, as compared to the high Mn concentrations in OW2, or to the initial concentration in OW1 at 2012 (before applying any pretreatment, i.e. 839 µg/L). These results indicate Mn dissolution minimization in OW1, probably due to the efficient removal of ammonium, nitrite and DOC by the biofilter, which strongly reduced oxygen demand for biological processes in the short SAT. The calculated oxygen demand in the biofilter (using Elkayam et al., 2015 calculation) would be ~17 mg/L[1], out of a total of ~39-40 mg/L[2] oxygen demand for the overall process from secondary effluents to recovery wells OW1/OW2.

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[1] OD in the biofilter (Elkayam et al., 2015) = $\Delta[DO] - (\Delta[NO_2^-])*SF_{NO2-} - (\Delta[NO_3^-])*SF_{NO3-} + (\Delta[DOC])*SF_{DOC} = 0 - 0.86 * 3.43 + 3.19 * 4.57 + 2.85 * 2 = 17.33$ mg/L

[2] OD in OW1 = 1.5 – 0.87 * 3.43 + 4.65 * 4.57 + 10.35 * 2 = 40.46 mg/L, OD in OW2 = 2.5 – 0.74 * 3.43 + 3.98 * 4.57 + 10. 5 * 2 = 39.15 mg/L

The value of Mn concentration at OW2 in 2012 was 617 μg/L. The Mn concentrations found in OW2 during *Mode 1* infiltration fluctuated between 188-572 μg/L, with most of the cases concentration higher than 500 μg/L. As mentioned earlier, OW1 was used to represent the targeted final product of short SAT, in which the objective was to get good chemical and microbial quality in a short HRT (~22 days), and high infiltration rate (~4-6 m/d). OW2 was investigated as a control of longer SAT (~113 days HRT), and in order to understand the hydro-chemical processes in the aquifer at the area. It can be concluded from the results that OW2 is much more affected by the surrounding non-treated secondary effluent flows. In addition, the reaction time in this well is relatively long. The high Mn in OW2 could be explained by residual Mn in the aquifer, and likely presence of anoxic microniches, either by previous organic loadings or intrusion of secondary effluents from the surrounding. If sorption of carbon occurs, then subsequent degradation could contribute to ongoing Mn release.

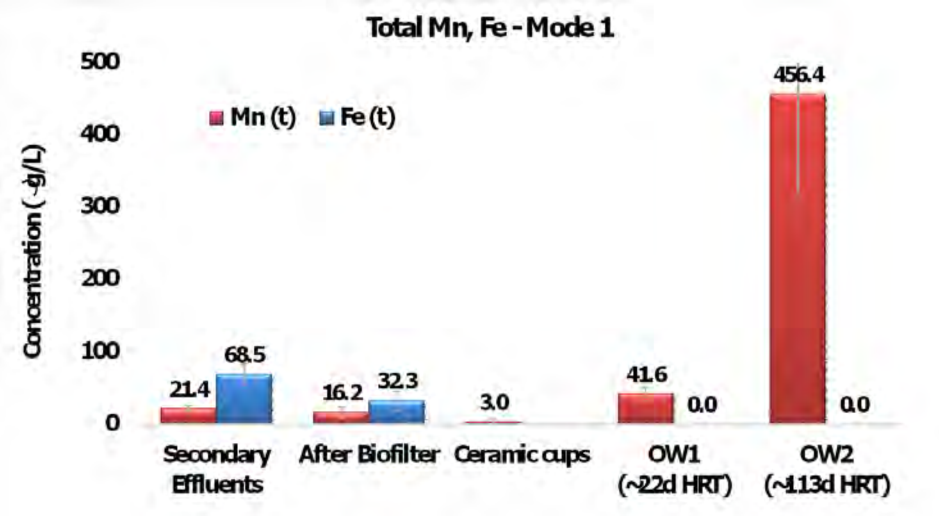


Figure 23 Averaged concentrations of Mn (total) and Fe (total) at the different sampling points during Mode 1.

4.3.2.6 Microbiology

Figure 24 presents average concentrations of microbial counts: Total bacteria (TOTB), Total coliforms (CMPN), Fecal coliforms (FMPN) and Fecal streptococcus m-enterococcus (EMPN) at the different sampling points during *Mode 1*. Biofiltration showed bacterial removal of 0.2 log TOTB, 0.7 log FMPN and EMPN, and 0.8 log of CMPN. Processes in the vadose zone (observed in the ceramic cups) significantly decreased CMPN, FMPN and EMPN by 2.5 log, 2.4 log, and completely, respectively. After short (OW1) and longer (OW2) SAT, all CMPN, FMPN and EMPN counts were below 10 MPN/n per 100mL. TOTB counts were in the range of ~1-2*10^4 cfu/mL.

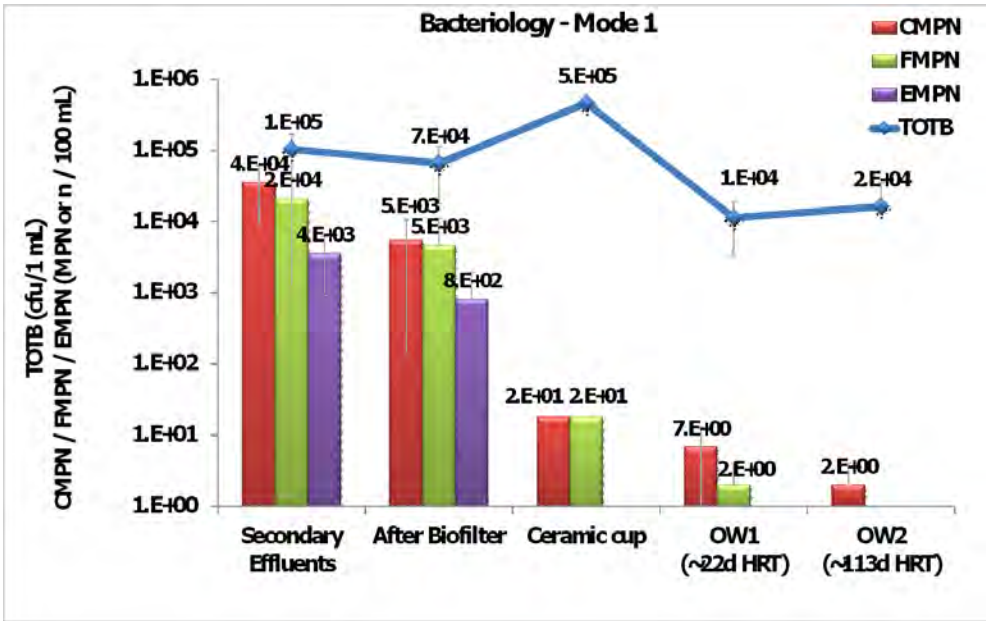


Figure 24 Averaged concentrations of Total bacteria (TOTB), Total coliforms (CMPN), Fecal coliforms (FMPN) and Fecal streptococcus m-enterococcus (EMPN) at the different sampling points during Mode 1.

Table 21 presents virus analysis results in secondary effluents, after biofilter, and in OW1 which were conducted during operation in *Mode 1*. Norovirus, Enterovirus and Arechovirus were not found in any of the samples examined. Adenovirus and coliphages were found in both secondary effluents and after biofilter (Table 21). No virus (human or coliphage) were found in OW1, suggesting that short SAT after biofiltration eliminates viruses. However, these results should be verified with more measurements in future research.

Table 21 Virus analysis results at secondary effluents, after biofilter, and OW1 during Mode 1

Sampling point	Norovirus	Enterovirus	Arechovirus	Adenovirus	Bacteriophage
	(n/L)	(n/L)	(n/L)	(n/L)	(n/L)
Secondary Effluents	Negative	Negative	Negative	540	960
After Biofilter	Negative	Negative	Negative	675	1080
OW1	Negative	Negative	Negative	Negative	0

Table 22 presents Cryptosporidium and Giardia count results conducted at secondary effluents, after biofilter, and at OW1 during *Mode 1*. Cryptosporidium and Giardia were detected in secondary effluents at values of 59 and 908 in 10L, respectively. As expected, less than 1 n/10L of both Cryptosporidium and Giardia were found after biofilter (0.1 and 0.4, respectively, Table 22). In addition, neither Cryptosporidium nor Giardia was found in OW1. This could be explained by the effective filtration chain involved in the current treatment - first through the biofilter, then through the upper layer SAT, and further polishing in the groundwater where almost no food and no suitable conditions for microorganisms prevail.

Table 22 Cryptosporidium and Giardia count results at secondary effluents, after biofilter, and OW1 during Mode 1.

Sampling point	Cryptosporidium (n/10L)	Giardia (n/10L)
Secondary Effluents	59.0	908.0
After Biofilter	0.1	0.4
OW1	0.0	0.0

In general, soil aquifer treatment (SAT) enables effluents purification by means of flow through unsaturated soils and aquifers. Due to this slow sand treatment, the upper layers (1-2 m) of the vadose zone can be an effective and low energy step in the SAT technology to obtain water for reuse.

The bacteria and viruses are inactivated as a function of retention time and temperature in the soil, as well as removed through net attachment, which is strongest when divalent cations are present. Bacteria and protozoa are removed by filtration along with other mechanisms. Viruses can possibly survive six months at low temperatures.

Table 23, adapted from Metcalf and Eddy, 2004, presents typical pathogen survival types in various environments, including in the soil (bolded). It can be seen that bacteria, viruses, and protozoa usually survive up to 50, 20, and 10 days, respectively, in the soil. However, pathogen survival may in some cases exceed values shown in the ranges of

Table 23, depending on many factors such as temperature, oxygen status, ionic strength, sodium adsorption ratio, mineralogy, texture, and even velocity changes in the aquifer. In situ pathogen fate studies are recommended where reliance is placed on aquifer treatment for pathogen removal.

One of the reasons for applying long SAT for secondary effluents treatment is to enable at least 6 months of retention time in the aquifer, in order to completely remove viruses (Fox, 2010). However, in our case, pretreatment using biofiltration only (*Mode 1*) succeeded to eliminate all viruses examined (human or coliphage) after ~22 days HRT, without addition of ozonation (*Mode 2*). The elimination of various bacteria and viruses could be attributed also to the high sunlight (UV) irradiation to which the effluents are exposed to during infiltration. Overall, the results indicate a very good microbial quality at the final product after short SAT (OW1).

Table 23 Typical pathogen survival times at 20-30oC in various environments (Metcalf and Eddy, 2004)a

		Survival times, days		
Pathogen		Fresh water and Wastewater	Crops	Soil
Bacteria:	Fecal coliforms[b]	<60 but usually<30	<30 but usually<15	<120 but usually<50
	Salmonella spp. [b]	<60 but usually<30	<30 but usually<15	<120 but usually<50
	Shigella [b]	<30 but usually<10	<10 but usually<5	<120 but usually<50
	Vibrio cholerae [c]	<30 but usually<10	<5 but usually<2	<120 but usually<50
Protozoa:	E. histolytica cysts	<30 but usually<15	<10 but usually<2	<20 but usually<10
Helminths:	A. Lumbricoides eggs	Many months	<60 but usually<30	<Many months
Viruses: [b]	Enteroviruses[d]	<120 but usually<50	<60 but usually<15	<100 but usually<20

[a] Adapted from Feachem et al. (1983).

[b] In seawater, viral survival is less, and bacterial survival is very much less than in fresh water.

[c] *V. cholera*e survival in aqueous environments is a subject of current uncertainty.

[d] Includes polio, echo, and coxsackie viruses.

4.3.2.7 Organic micro-pollutants

Figure 25 summarizes the TrOC concentrations observed during *Mode 1*. The concentrations in the secondary effluents ranged from several micrograms per liter for the compounds IPDL, IPRM, BZF, CBZ, DCF and SMX up to 21.9 ± 11.2 and 64.1 ± 260.9 for IHX and VLX, respectively. Concentrations were fluctuated between campaigns, probably due to weather or operational conditions, or even to the sampling or analysis protocols.

As expected, biofiltration did not significantly reduce the concentrations of most TrOCs. Only IPDL, IHX, IPRM, and VLX were removed by approximately 24%, 7%, 25%, and 99%, respectively. Although no differentiation was made between sorption and biodegradation, based on the physicochemical properties and literature data, the elimination of these compounds can be attributed mostly to biodegradation (Ternes et al.,2007; Buerge et al., 2011). Rapid degradation of IPRM has been previously described (Batt et al., 2006).

Two campaigns were involved sampling of infiltrated effluent at 1.5 m depth via the ceramic cups. Most of TrOCs were reduced following first 1.5 m depth infiltration at the range of ~79-100%. After short SAT (OW1, ~22d HRT), most of the TrOCs were removed to below 50 ng/L. These results are in line with results from other bank filtration and artificial recharge sites (Grunheid et al., 2005; Ternes et al., 2007). The concentrations of IPDL, CBZ, and SMX in OW1 were increased compared to those found in the ceramic cups, and may be originated from native water or desorbed from soil. However, it should be noted the ceramic cups results based only on 2 campaigns, and further assessment of the biodegradation contribution to TrOC reduction during SAT should be addressed.

CBZ concentration after short SAT was found to be 840 ± 675 ng/L. High persistence of CBZ in biological water treatment and aquifer recharge systems is well established, and it has been proposed as a marker for anthropogenic influences (Arye et al., 2011). Under strong reducing conditions, however, CBZ removal might occur (Wiese et al., 2011). Significant but incomplete removal was observed also for SMX. Slow degradation of SMX in oxic and anoxic soil systems is well known (Baumgarten et al., 2003).

After longer SAT (OW2, ~113d HRT), there was some removal of the remaining TrOCs IPDL, CBZ and SMX, to values of 785 ± 302, 495 ± 262, and 27 ± 61 ng/L, respectively.

The elimination of TrOCs by passage through the soil can be attributed to either sorption or biodegradation (Hiscock and Grischek, 2002). Therefore, the sorption properties of the TrOCs to soil and the types of mineral and organic sorbents are essential for assessing their potential to leach into the groundwater and be transported in aquifers and surface water. Some separation between the two mechanisms involved in the vadose and saturated zones can be achieved via observation of TrOC concentration at different depths of the vadose zone (e.g. ceramic cups, Arye et al., 2011).

4.3.2.8 Summary of Mode 1 results (March 2014-July 2015)

The overall results during *Mode 1* operation demonstrate the benefits of the tested biofilter operation for the overall treatment scheme. Aside from the positive aspects of particle removal in preventing clogging in the short SAT and reducing ozone demand (Zucker et al., 2015), the efficient removal of ammonium, nitrite and DOC strongly reduced oxygen demand for biological processes in the short SAT, by approximately 50%, and above 2 mg/L DO in OW1 in most cases. In addition, nitrite removal reduced ozone demand by approximately 3.0 mg/L (calculated with 3.4 mg O_3 per mg NO_2-N).

Short SAT performance (OW1, ~22 days HRT) after biofiltration exhibited efficient removal of DOC and UVA up to levels of 1.4 mg/L and 4.3 1/m, respectively. Total Mn detected in OW1 maintained concentration lower than 52 μg/L throughout the overall *Mode 1* operation, as compared to initial concentration of 839 μg/L in 2012 (before application of pretreatment to SAT).

This result of Mn dissolution minimization was attributed to the fact that the biofilter significantly reduced oxygen demand in the SAT due to efficient removal of ammonium, nitrite and DOC, and thus controlled development of reducing conditions.

Although the Israeli water reuse standards allow 200 µg/L dissolved Mn^{2+} in treated effluents for unrestricted irrigation (Goren, 2008), 50 µg/L was suggested herein as an upper limit for Mn^{2+} due to operational clogging problems along the dripper distribution pipelines. Operation of the pilot in *Mode 1* almost met this enforceable standard for recharged water with a HRT of ~22 days.

Microbial quality at OW1 after biofiltration was found to be very high (<10 MPN/100mL total coliforms, <10 n/100 mL fecal coliforms, no fecal streptococcus, no Cryptosporidium/Giardia, no human/coliphage viruses). Moreover, most of the TrOCs measured in OW1 were significantly removed (<50 ng/L), except for IPDL, CBZ and SMX.

Results after longer SAT (OW2, ~113 days HRT) pretreated with biofilter showed similar DOC, UVA, N-compounds results as in OW1. Remaining TrOCs were somewhat reduced in OW2, however IPDL and CBZ concentrations were still high (785 and 495 ng/L, respectively). In addition, Mn concentration in OW2 was much higher (in an order of magnitude) compared to OW1, presumably due to residual carbon sources originated from either natural minerals in the aquifer or external sources of organic carbon (e.g. secondary effluents in the area) that may intrude OW2. The lower DO concentrations obtained in OW2 is in accordance with the Mn results indicating evidence of oxygen depletion processes in the aquifer and development of anoxic/reducing conditions.

As previously discussed, OW2 is much more affected by the surrounding secondary effluents in the groundwater compared to OW1. Reactions like Mn^{2+} dissolution and to a certain extent TrOCs desorption are a function of travel distance in almost anoxic conditions. Thus, it is advantageous to try to reach the highest water quality at a relatively short HRT, as was demonstrated in the current pilot (OW1).

In the current study, oxic filtration in the biofilter was established by the addition of H_2O_2. In future experiments, the use of ozonation off-gas or part of ozone stream (rich in DO) as a supplementary oxygen supply will be examined in order to reduce treatment costs. As previously noted, the biofiltration stage is required only when not enough ammonium and DOC are removed in the secondary treatment.

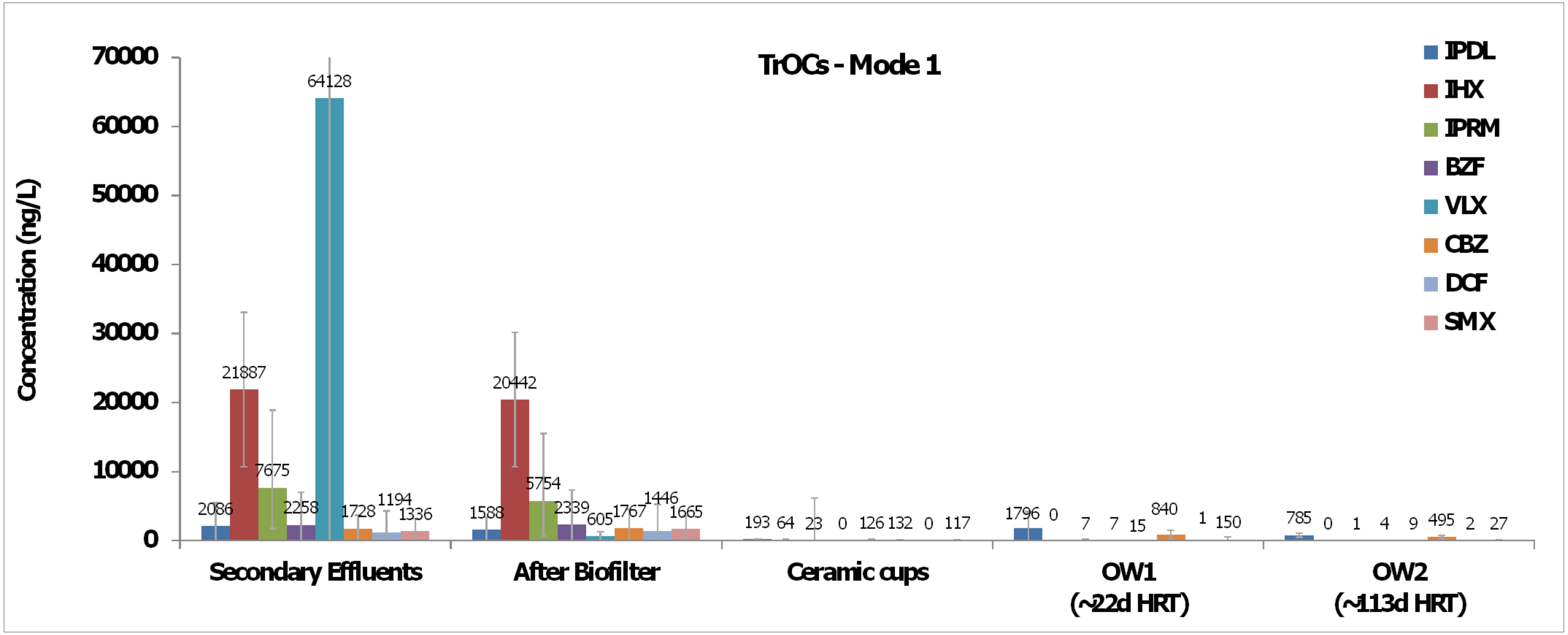


Figure 25 TrOCs concentrations at the different sampling points during Mode 1.

4.3.3 Mode 2: Infiltration of Biofiltered-Ozonated effluents (August 2015-September 2016)

The ozonation stage had several objectives in addition to the biofiltration stage: (1) break down of the remaining organic matter, in order to make it more biodegradable for further removal in the short SAT, (2) addition of significant amount (~20-30 mg/L) of dissolved oxygen to the infiltrated effluents, in order to prevent anoxic/reducing conditions and Mn^{2+} dissolution, (3) removal of TrOCs and microorganisms and improve product quality.

As mentioned earlier, due to operational problems in the ozone system, several shutdowns of the ozone system were obtained during *Mode 2* operation. From June until September 2016, the ozone system worked continuously. Thus, the effect of ozonation on biofiltered effluents was summarized for the overall *Mode 2* period in section 4.3.3. The effect of ozonation on the observation wells OW1 and OW2, which require continuous long-term infiltration, was obtained at the last months of *Mode 2* operation, between June and September 2016 (total of 111 days, when the ozone system was operated continuously), and was summarized in section 4.3.3.2.

4.3.3.1 Mode 2 - the effect of ozonation on biofiltered effluents (Aug. 2015-Sep. 2016)

4.3.3.1.1 Particle analysis

Particle size distribution is an important analysis for treatments combined with ozonation since: (1) particle presence in effluents significantly affects ozone reactions (Zucker et al., 2014); and (2) particle distribution following ozonation can indicate the different mechanisms that occur in the oxidized effluent.

Table 24 and Figure 26 represent a Typical PSD analysis (conducted in 05/01/2016) of secondary effluents, before biofilter (after flocculation), after biofilter, and after ozonation. It can be seen that flocculation before biofiltration increased particle counts. As observed during *Mode 1*, particle concentration decreased dramatically after coagulation/flocculation and biofiltration for all ECDs. The phenomena of highly reactive particles due to ozonation was observed in these samples as in Zucker et al (2014). The concentration of particles decreased following ozonation mostly for particles in the range of 3-15 µm. Particle analysis also revealed that particle formation (in the range of 2-3 µm and above 15 µm in this case) and breakage occurs simultaneously during wastewater ozonation.

Table 24 Typical PSD of Secondary Effluents, before and after Biofilter, and after Ozonation (05/01/16)

Particle size µm	Secondary effluents	Before Biofilter	After Biofilter	After Ozonation
2-3	18,357	41,538	2,329	3,580
3-5	6,123	16,194	1,271	1,263
5-10	11,184	12,103	2,075	1,929
10-15	7,344	3,320	324	376
15-30	2,187	2,228	19	122
30-40	232	208	3	16
40-50	47	58	0	4
50+	38	37	1	1
Total	45,512	75,686	6,023	7,292

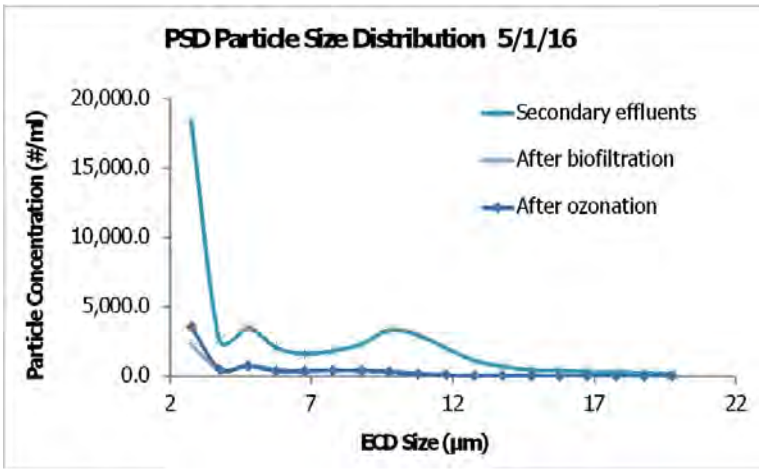


Figure 26 Typical PSD of Secondary Effluents, after Biofilter, and after Ozonation (05/01/16).

Table 25 represents the campaigns in which ozone dose of ~10 mg/L was applied, and particle analysis was conducted. A correlation between PSD, ozone dose and UVA removal was found. UVA was decreased by 50-63% in all of the experiments. Removal of UVA is indicative of ozone reaction with conjugated systems in the dissolved organic matter. Particle count decreased following ozonation except in the experiments conducted at July 2014, October 2015 and January 2016 (Table 24). Zucker et al., 2014 showed that particle image analysis reveals particle formation/aggregation and particle disintegration occurring simultaneously during wastewater ozonation. The concentration of particles presented in Table 25 decreased following ozonation for all ECDs smaller than 15 µm. larger particle concentrations were increased indicating on flocculation effects during ozonation.

Table 25 Correlation between PSD, ozone dose and UVA removal

		PSD total (#/mL)			UVA (m^{-1})		
UVA removal	After ozonation	After biofilter	PSD removal	After ozonation	After biofilter	O_3 consumption	Date
%55.11	8.21	18.29	27.73%	11,429	15,814	10	27.5.14
%63.28	5.56	15.14	28.29%	2,413	1,881	10.44	20.7.14
%57.21	6.5	15.19	28.17%	889	1,237	10.19	3.8.14
%61.93	6.7	17.6	49.70%	1,015	2,018	10.3	9.9.14
49.71%	8.7	17.3	38.92%	5,227	8,558	~10	18.8.15
-	-	-	178.57%	3,389	1,217	~10	6.10.15

4.3.3.1.2 Bulk chemistry

Table 26 presents average values of chemical parameters measured at the different sampling points during *Mode 2* (infiltration of biofiltered-ozonated effluents). It was shown that turbidity was removed by ~75% after biofiltration (similarly to *Mode 1*). No effect of ozonation on turbidity was observed.

In general, it is important to note that due to internal process changes in Shafdan WWTP (addition of sludge treatment and consequently introduction of nutrients back to the feed wastewater), there was a higher ammonium concentration at the Shafdan secondary effluents during *Mode 2* (2015-2016, section 4.3.3.1.3) compared to *Mode 1* (2014-2015, section 4.3.2.3), with average concentration of 5.9 ± 2.9 mg NH_4^+-N/L, and a maximum concentration of 12 mg NH_4^+-N/L (Figure 27, section 4.3.3.1.3).

This affected the efficiency of the nitrification process in the biofilter, which was found to remove up to ~4 mg NH_4^+-N/L in its current configuration, resulting in lower DO and higher ammonium concentrations at the biofilter outlet.

Consequently, DO concentration after biofilter during *Mode 2* was found to be 1.8 ± 1.0 mg/L (Table 26), much lower than this found during *Mode 1* (Table 20), due to the higher loads of ammonium (and other nutrients) that was introduced to the biofilter during *Mode 2* and consumed oxygen. After ozonation, the DO concentration increased to 19.1 ± 2.1 mg/L (Table 20) and up to ~30 mg/L, since ozone is produced from pure oxygen which only part of it is converted to ozone. This oxygen excess was highly important for the SAT process (and was one of the reasons for adding ozone to the pretreatment scheme) since it enabled infiltration of highly oxygenated effluents to the upper layer of the SAT, providing enough DO for the biochemical reactions, despite the lower efficiency of the biofilter. The excess of DO was supposed to prevent anoxic/reducing conditions and thus to minimize Mn^{2+} dissolution (discussed in section 4.3.3.2).

Similarly to *Mode 1*, BOD and COD were reduced by ~75% and ~32%, respectively, after biofiltration. After ozonation, BOD increased from ~1 to ~2 mg/L, probably due to the breakage of organic matter compounds and increase in its biodegradability. In general, ozonation has been demonstrated to improve biodegradability of DOM. This effect is well known from drinking water treatment, where an increase of assimilable organic carbon (AOC) necessitates subsequent biological treatment to prevent bacterial regrowth in distribution systems. An increase in BOD, BOD/COD ratio, and biodegradable dissolved organic carbon (BDOC) was reported for ozonation of various types of waters (Hubner et. al., 2015). The increase of biodegradability can be attributed to the transformation of organic matter into smaller and more polar compounds and the formation of readily degradable functional groups, such as aldehydes and carboxylic acids (Hubner et. al., 2015). COD after ozonation decreased by ~19%.

Phosphate concentrations found in secondary effluents were relatively low (Table 26). Thus, no significant differences were observed between *Mode 1* and *Mode 2* in this regard.

Table 26 Chemical parameters at different sampling points during Mode 2 (infiltration of biofiltered-ozonated effluents).

Mode 2			
	Secondary Effluents	After Biofilter	After ozonation
Turbidity (NTU)	2.4 ± 0.8 (n=7)	0.6 ± 0.2 (n=7)	0.6 ± 0.2 (n=7)
DO* (mg/L)	3.7 ± 0.7 (n=5)	1.8 ± 1.0 (n=5)	19.1 ± 2.1 (n=5)
BOD (mg/L as O_2)	4.0 (n=1)	1.0 (n=1)	2.0 (n=1)
COD (mg/L as O_2)	39.0 ± 21.9 (n=2)	26.5 ± 5.5 (n=2)	21.5 ± 8.4 (n=2)
BOD/COD	0.13 (n=1)	0.04 (n=1)	0.13 (n=1)
BOD/DOC	0.42 (n=1)	0.13 (n=1)	0.27 (n=1)
PO4 (mg/L as P)	0.62 (n=1)	0.36 ± 0.05 (n=2)	0.38 ± 0.02 (n=2)

* Measured in the lab by 4500-O C Azide Modification

4.3.3.1.3 N-compounds

Figure 27 presents nitrogen compound concentrations measured at the different sampling points during *Mode 2*. Secondary effluents contained average concentrations of 5.9 ± 2.9 mg NH_4^+-N/L, 0.9 ± 0.2 mg NO_2^--N/L, and 0.5 ± 0.3 mg NO_3^--N/L, indicating incomplete nitrification process at the Shafdan WWTP.

Results after biofilter showed evidence of nitrification (incomplete, compared to *Mode 1*) including ~67%, and ~98% removal of NH_4^+-N and NO_2^--N, respectively, accompanied with increase in nitrate up to 3.8 ± 1.1 mg NO_3^--N/L. These results suggest that the biofilter in its current configuration is able to reduce ~4 mg NH_4^+-N/L. When planning a commercial plant, maximum ammonium concentrations in the feed should be taken into account when designing the biofilter dimensions and media depths. However, a better (and most economical) approach will be to improve the nitrification process during the Activated Sludge treatment.

During *Mode 2*, ozonation was conducted at a constant ozone consumption of 1.0–1.2 mg O_3/mg DOC. No significant changes in N-compounds were observed after ozonation. A slight unsignificant decrease in ammonium and a slight unsignificant increase in nitrate were observed after ozonation. This could not be explained by oxidation of ammonium and nitrite residues. The possible increase in nitrate might be partially attributed to oxidation of organic nitrogen. In one measurement of nitrogen compounds at 1.5m depth of the vadose zone (ceramic cups), a complete removal of ammonium and nitrite, accompanied with increase in nitrate were observed, indicating complete nitrification in the upper layer of the SAT, similarly to the observation in the ceramic cups during *Mode 1* (data not shown).

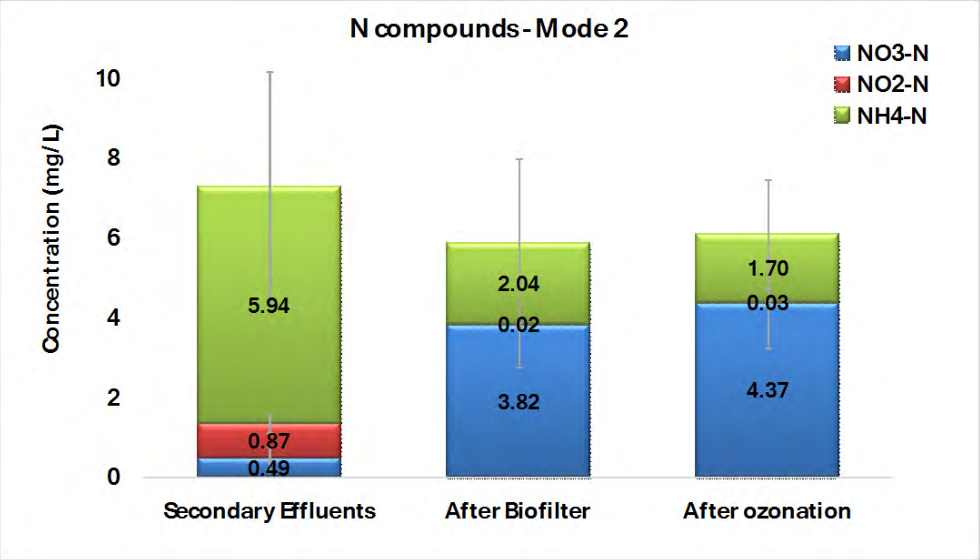


Figure 27 Nitrogen compound concentrations at the different sampling points during Mode 2

4.3.3.1.4 DOC, UVA, SUVA

Figure 28 shows average DOC, UVA, and SUVA values measured at the different sampling points during *Mode 2*. Similarly, to *Mode 1*, Shafdan secondary effluents consisted 10.1 ± 1.1 mg/L DOC and 19.7 ± 3.2 1/m UVA, while biofilter reduced DOC and UVA by ~23% and ~19%, respectively.

There was no significant change in DOC after ozonation. Limited DOC removal (~3%) by ozone is in line with the literature for similar ozone dosages (Scheurer et al., 2012). In contrast, efficient reduction of UVA (~51%) to 7.9 1/m was observed due to selective attack of the conjugated systems in DOC. It should be noted that for large-scale ozonation units, the monitoring of UVA is mentioned as an easily managed control parameter that provides a good indication of oxidation performance (Bahr et al., 2007).

Infiltration in the vadose zone (ceramic cups) further decreased DOC and UVA by ~73% and ~47%, respectively, to 2.1 ± 0.1 mg/L, and 4.2 ± 0.2 1/m, much higher rates than those obtained in *Mode 1*.

This could be explained by the higher activity in the upper SAT layer during *Mode 2* which received more available organic matter due to bond cleavage by ozonation, compared to that during *Mode 1*. In addition, *Mode 2* effluents after ozonation had higher DO compared to *Mode 1* effluents after biofiltration alone, thus this could also have contributed to the DOC and UVA reduction in the vadose zone.

As in *Mode 1*, similar values of SUVA were obtained following biofiltration, whereas much lower values (~48% reduction) were seen following ozonation, indicating removal of aromatic components. SUVA value increased again after infiltration in the vadose zone, and was similar or slightly lower than this obtained during *Mode 1* (2.0 ± 0.2 in *Mode 2* compared to 2.6 ± 0.7 in *Mode 1*). As previously discussed, the increase in SUVA during infiltration at the vadose zone could be explained by preferential removal of nonaromatic (non-humic) components (Amy and Drewes, 2007).

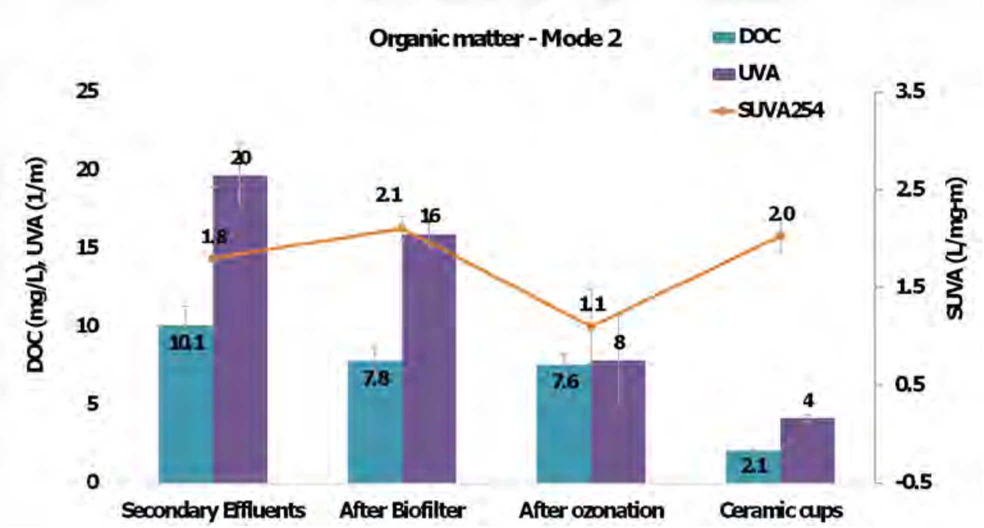


Figure 28 **Organic matter-related parameters at the different sampling points during Mode 2.**

4.3.3.1.5 Inorganic traces (Fe, Mn)

Figure 29 shows average concentrations of Mn (total) and Fe (total) at the different sampling points during *Mode 2*. Mn concentration in secondary effluents (21.3 ± 5.0 μg/L) was similar to this observed during *Mode 1*, while Fe concentration was somewhat higher (~55 μg/L). Similarly, to *Mode 1*, biofiltration did not remove Mn, but removed Fe by ~51% to ~27 μg/L. Ozonation did not change Mn and Fe concentrations. Processes in the vadose zone (obtained in 3 campaigns) removed Mn down to ≤3 μg/L (Fe measurements were not conducted in the vadose zone).

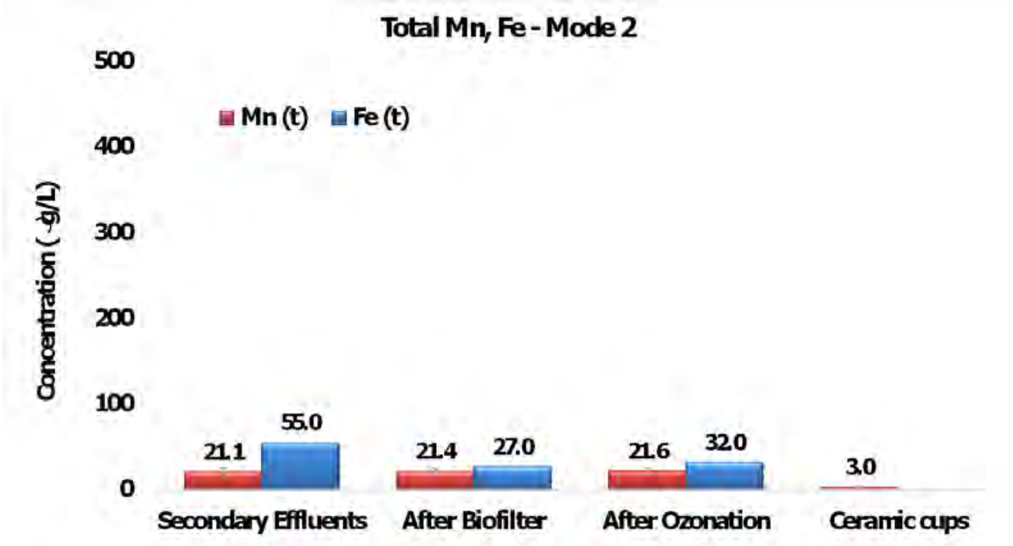


Figure 29 Averaged concentrations of Mn (total) and Fe (total) at the different sampling points during Mode 2.

4.3.3.1.6 Microbiology

Figure 30 presents average concentrations of microbial counts (TOTB, CMPN, FMPN, and EMPN) at the different sampling points during *Mode 2*. Biofiltration showed similar bacterial removals to those obtained during *Mode 1*, with ~0.1, ~1.1, ~0.6, and ~0.4 log removals of TOTB, CMPN, FMPN, and EMPN, respectively. Ozonation showed high removal rates of all types of bacteria, with ~1.0, ~2.7, ~3.5, and complete log removals of TOTB, CMPN, FMPN, and EMPN, respectively. Microbial measurements were not conducted in the ceramic cups during *Mode 2*.

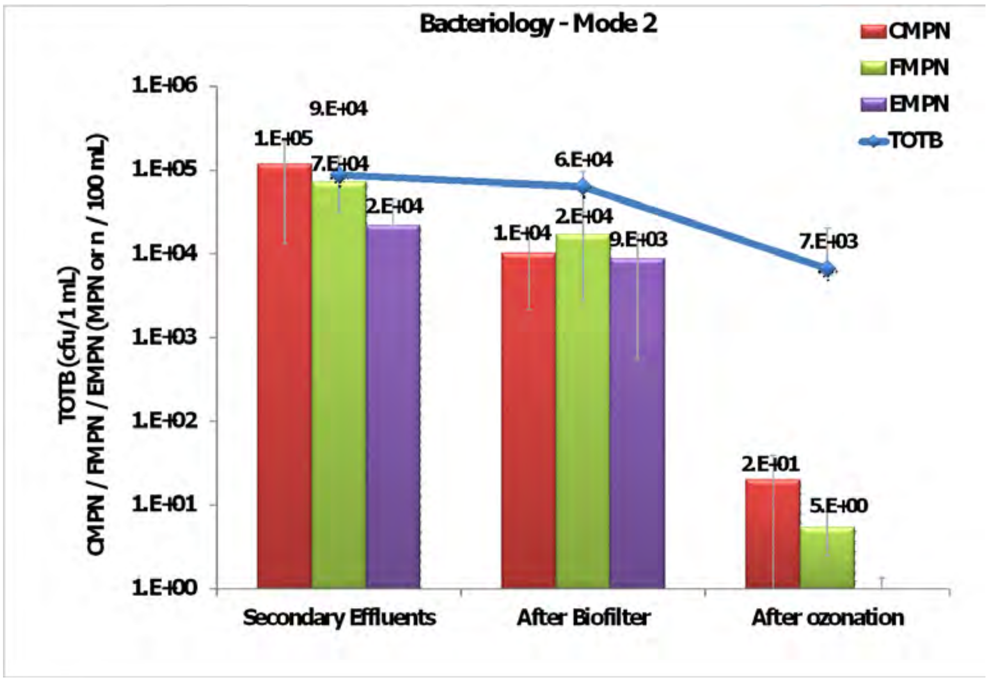


Figure 30 Averaged concentrations of Total bacteria (TOTB), Total coliforms (CMPN), Fecal coliforms (FMPN) and Fecal streptococcus m-enterococcus (EMPN) at the different sampling points during Mode 1.

4.3.3.1.7 Organic micro-pollutants

Oxidation with ozone is a promising method for the removal of many relevant TrOCs from secondary wastewater effluents. During ozonation, a mineralization of TrOCs is typically not achieved and compounds are only transformed. Several studies demonstrated that the primary attack by ozone is sufficient to reduce specific effects, such as endocrine disruption, antibacterial and antiviral activity (Hubner et. al., 2015).

Figure 31 summarizes the average TrOC concentrations observed during *Mode 2*. Removal of TrOCs by ozonation strongly depends on their rate constants with ozone (Table 17). Observed removal rates were in line with literature data (Bahr et al., 2007; Hollender et al., 2009; Hubner et al., 2012). The fast-reacting compounds VLX, CBZ, and SMX were efficiently removed by ~94%, ~97%, and ~97%, respectively. Previous studies have indicated complete removal of these compounds at the much lower ozone consumption of 0.4–0.6 mg O_3/mg DOC (Hollender et al., 2009). The relatively low removal rate of DCF (~64%) was not expected. High removal of the moderately reacting compound BZF (~95%) was observed, probably due to relatively low concentrations in the feed. Concentrations of the ozone-resistant compounds IPDL, IHX, and IPRM were decreased by ~50%, ~64%, and ~73%, respectively, confirming OH-radical formation during ozonation of secondary effluent (Hubner et al., 2013). For the transformation of compounds with low reactivity (i.e. ozone resistant compounds) the indirect reaction via hydroxyl (OH) radicals plays an important role. These highly reactive and nonselective radicals can be formed from the reaction of ozone with natural organic matter during any ozonation process, but especially during wastewater ozonation, with yield oxidation conditions similar to ozone-based AOPs in drinking water treatment (Acero and Von Gunten, 2001; Buffle and Von Gunten, 2006; Pocostales et al., 2010).

Processes in the vadose zone (ceramic cups) showed further removal of most TrOCs, by ~64%, ~95%, ~93%, ~96%, and ~60% for IPDL, IHX, IPRM, BZF, and SMX, respectively.

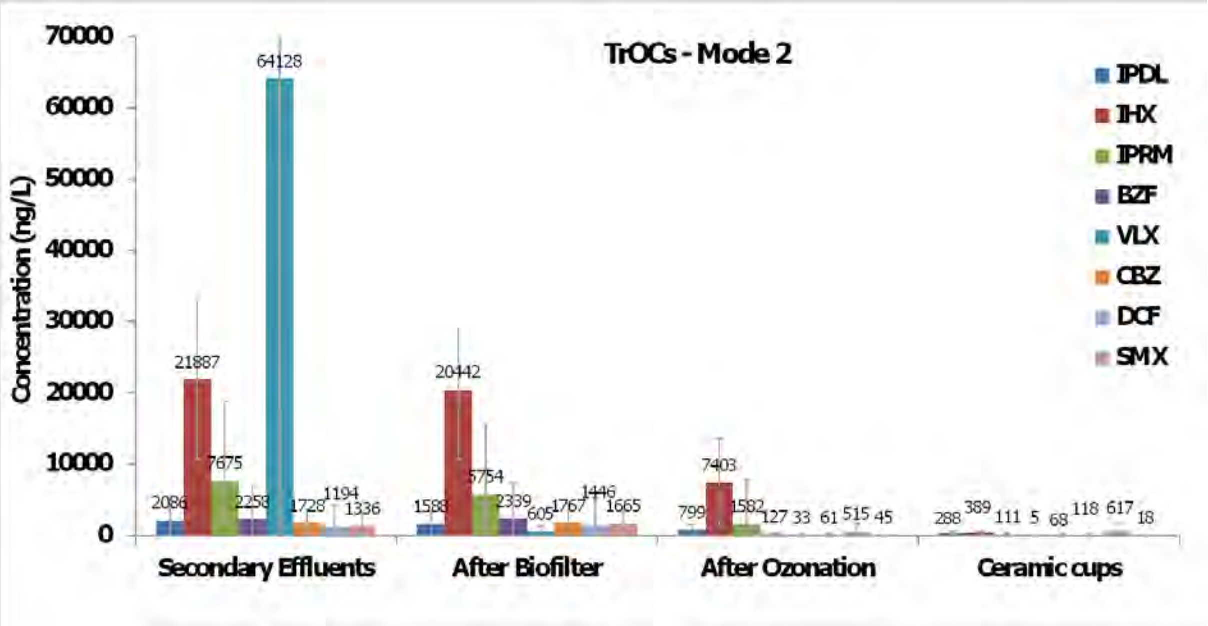


Figure 31 TrOCs concentrations at the different sampling points during Mode 2.

4.3.3.2 Mode 2 – effect of ozonation on SAT (111 days of continuous ozone operation)

In order to see the effect of ozonation on the observation wells OW1 and OW2, a continuous long-term infiltration of ozonated effluents is required. After overcoming the operational challenges, the ozone system started to work continuously from June until September 2016 (total of 111 days). The overall results of this period are summarized in the following section, showing the effect of *Mode 2* on OW1 (~22 days HRT), which represents the targeted short SAT product.

Observation well OW2 (~113 days HRT according to the tracer test), which was monitored mainly for control, require longer period of continuous operation, thus it was not expected to be affected at the 111 days period. Nevertheless, the results for OW2 are still presented. All of the results in the following section are presented by dates and not in average, since the processes in the SAT are progressive, thus it is better to look at the results as a function of time.

4.3.3.2.1 N-compounds

Table 27 presents N-compounds concentrations along the process during *Mode 2* continuous operation. It can be seen that ammonium concentration in secondary effluents fluctuated and were sometimes higher than 5 mg NH_4^+-N/L. In these cases, the biofilter did not succeed to perform a full nitrification, and residual ammonium was observed.

Ozone did not have a significant effect on N-compounds (as explained in *Mode 1*, section 4.3.2.3). In OW1 and OW2 no ammonium was detected at all sampling campaigns, accompanied with increase in nitrate, indicating complete nitrification in the SAT system.

Table 27 N-compounds along the process during 111 days of continuous Mode 2 operation.

Date	No. of days from continuous ozonation startup	Sampling point	NO3-N (mg/L)	NO2-N (mg/L)	NH4-N (mg/L)
27/06/2016	27	Secondary Effluents	0.68	0.97	2.82
		After Biofilter	3.16	0.01	0.57
		After ozonation	4.07	0.01	1.55
		OW1	6.55	0.03	<0.03
		OW2	8.58	0.09	<0.03
26/07/2016	56	Secondary Effluents	0.45	0.72	11.72
		After Biofilter	2.94	0.01	6.91
		After ozonation	4.07	0.01	4.23
		OW1	6.33	0.04	<0.03
		OW2	6.55	0.09	<0.03
16/08/2016	77	Secondary Effluents			5.07
		After Biofilter			2.75
		After ozonation			2.46
		OW1			<0.03
		OW2			<0.03
13/09/2016	105	Secondary Effluents			1.87
		After Biofilter			0.33

Date	No. of days from continuous ozonation startup	Sampling point	NO3-N (mg/L)	NO2-N (mg/L)	NH4-N (mg/L)
		After ozonation			0.46
		OW1			<0.03
		OW2			<0.03

4.3.3.2.2 DOC, UVA, SUVA

Table 28 shows TrOCs, DOC, UVA, SUVA and Mn measurements at all sampling points throughout the process during the continuous operation of *Mode 2*. It can be seen that DOC and UVA at OW1 and OW2 were reduced down to 0.8 mg/L and, 2.2-2.6 1/m, respectively, lower values than those obtained at *Mode 1*. The lower UVA means less aromaticity which is a result of ozonation.

SUVA decreased after ozonation down to 0.8 L/mg-m, increased up to 2.4 L/mg-m at the ceramic cups, and further increased to 3.0-3.3 L/mg-m at OW1 and OW2 (Table 28), similarly to *Mode 1*. The increased biodegradability of DOM following ozonation is well known (Hubner et al., 2012; Lee et al., 2013). Results from the current study confirmed that the combination of ozonation and SAT provides a good way to benefit from this effect. A significant reduction in DOC following a treatment combining ozonation and sand columns was also demonstrated by Schumacher et al. (2004).

4.3.3.2.3 Inorganic traces (Mn)

It can be seen from Table 28 that Mn concentration at OW1 (~22 days HRT according to tracer test) maintained 29-35 µg/L, which is lower than the Israeli water reuse standards (allowing 200 µg/L Mn for unrestricted irrigation), but higher than the concentration observed in the ceramic cups (≤3 µg/L). This could indicate that there was still Mn dissolution in the short SAT system, as was observed in *Mode 1*, probably due to the high ammonium residuals after the biofilter. Nevertheless, the ozone addition succeeded to minimize/control it, due to its high oxygen contribution (~20 mg/L).

These results suggest that ozonation could be advantageous also in cases where the nitrification process at the secondary treatment does not work ideally. In these cases, ozone addition compensates the oxygen depletion in the soil that may occur due to ammonium and DOC residuals. It seems that the ozonation technology does not necessarily require biofiltration added with H_2O_2 to remove ammonium residuals. It may effectively work with conventional media filtration prior to it, added with in-line coagulation only. Replacing the biofilter with a media filter prior to ozonation can significantly reduce the overall process costs (further discussed in section 4.4).

Results in OW2 showed higher Mn concentrations than those obtained in OW1 (138-170 µg/L), indicating much high rates of Mn dissolution. These values were lower than those obtained during *Mode 1*. However, since the measured HRT of OW2 in the tracer test was ~113 days, it is assumed that the ozone effect on OW2 was still not obvious. In addition, OW2 is suspected to be more affected by the surrounding than OW1. Thus, a longer infiltration period is required in order to better understand the results in this well.

4.3.3.2.4 Organic micro-pollutants

As shown in Table 28, ozonation effectively removed all of the examined fast-reacting compounds VLX, CBZ, DCF and SMX down to < 50 ng/L (98-100% removal). The moderately-reacting BZF was found in low concentrations in the secondary effluents and was effectively removed by ozonation to a non-detectable level.

Concentrations of the ozone-resistant compounds IPDL, IHX, and IPRM were decreased by ~60±17%, ~72±7%, and ~72±4%, respectively (to average levels of 535±279, 7781±1950, 1588±429, respectively), probably due to the formation of OH^- radical in the ozonation process.

Three campaigns of ceramic cups showed that all TrOCs measured were reduced by more than 95%, except for IPDL that was reduced by 72±26%. In the last campaign all TrOCs measured at the ceramic cups were undetectable (Table 28).

Concerning OW1, in the range of 56-77 days from the continuous *Mode 2* startup, it was shown that all TrOCs were reduced down to <100 ng/L. In the last TrOCs campaign (105 days) all TrOC values were below detection limit, except for VLX which was 41 ng/L (Table 28). Similar phenomenon was not observed during *Mode 1* operation, and is attributed to the combination of ozonation and SAT. Based on the tracer experiment, it was expected to observe reduction in TrOCs after ~22 days HRT from the startup of the continuous operation of *Mode 2*. Arye et al., (2011) studied the transport and fate of CBZ in Shafdan SAT system. Soil samples were taken from seven subsections of soil profiles (150 cm) in the upper layer infiltration basins. Results of this study indicated adsorption, and consequently retardation in CBZ transport through the top soil layer (0–5 cm), to a lesser extent in the second layer (5–25 cm), but not in deeper soil layers (25–150 cm). The soluble and adsorbed fractions of CBZ obtained from the two upper soil layers comprised 45% of the total CBZ content in the entire soil profile. This behavior correlated to the higher organic matter content observed in the upper soil layers (0–25 cm) in the SAT (bioactive layer). Thus, the delay in the effect of *Mode 2* pretreatment (biofiltration-ozonation) on TrOCs removal in OW1 could be explained by the retardation of CBZ displacement in the upper soil layers of the pilot SAT (0-25 cm). Meaning, when CBZ free water after ozonation were infiltrated through the CBZ rich layer (adsorbed CBZ at the previous *Mode 1* operation), there was a slow displacement of this excess CBZ that retarded the total CBZ removal. IPDL removal retardation might be explained by a similar mechanism. However, this assumption should be further investigated.

Pre-ozonation might affect biodegradation of TrOCs during aquifer recharge by: (i) reducing the initial compound concentration before infiltration, (ii) changing the amount and composition of biodegradable organic matter as a primary substrate for co-metabolic TrOC degradation, and (iii) increasing redox conditions in the subsurface. During ozonation, complete mineralization is not anticipated and transformation of organic compounds usually occurs. The removal of these products in short SAT was not addressed in this study, but recent studies revealed an increased biodegradability of most major products from ozone reactions with the olefinic compounds CBZ (Hubner et al., 2014) and ACS (Scheurer et al., 2012) in biological post-treatment. In addition, results from a literature study indicate that products persistence in biological processes is related to the ozone-reactive sites within the compound (Hubner et al., 2015). Enhanced removal efficiencies can be expected for olefins and aromatic compounds, whereas ozonation has probably a minor effect on biomineralization of amines (e.g. VLX).

Regarding OW2 (~113 days HRT according to the tracer test), 111 days of continuous *Mode 2* operation were not expected to affect this observation well. Indeed, IPDL and CBZ concentrations in this well were around 400 and 800 ng/L, respectively, similarly to *Mode 1*. All of the other TrOCs were under detection limit. A longer infiltration period is required in order to understand the possible effect of ozonation on this well.

4.3.3.2.5 Profile sampling along the perforated part of wells OW1 and OW2

In order to better understand the flow patterns along the observation wells OW1 and OW2, a more detailed sampling campaign was conducted on day 111 of *Mode 2* along the perforated part of the wells (starting from -14m of the static water level down to -26m at the bottom of the wells).

This sampling was conducted using a liquid sampler connected to an electric contact meter which enables to obtain liquid samples from any desired depth (KLL-S, 2" tube diameter, SEBA , Figure 32).

DOC, UVA and CBZ results showed homogeneity along the overall depth of OW1, showing very low values of 0.7-0.9 mg/L DOC, 2.1-2.2 1/m UVA, and <10 ng/L CBZ (). These results proved that the ozonated water completely covered the area around this observation well and positively affected the quality of the groundwater.

Regarding OW2, 0.7-0.9 mg/L DOC, 2.1-2.3 1/m UVA, and 635-688 ng/L CBZ were observed along the well, except for the lower depth (-26 m) which showed higher DOC and UVA values of 2.0 mg/L and 3.9 1/m, respectively (Figure 33). This result indicates, as was previously mentioned, that OW2 was still not completely affected by the ozonation of *Mode 2*. In addition, it is assumed that this well is more affected by the surrounding compared to OW1.

It is important to note, however, that all of the results presented in section 4.3.3.2 were obtained during a short period of 111 days during summer (June-September) of continuous infiltration of biofiltered-ozonated effluents. The results show a good feasibility of applying biofiltration and ozonation as a SAT pretreatment to reduce organic matter including TrOCs, while minimizing Mn dissolution. However, this process should be further investigated and demonstrated for a longer period of time (at least 1 year as was originally suggested), including winter time, in order to confirm the results and to see the effect of these processes on infiltration in a long-term period.

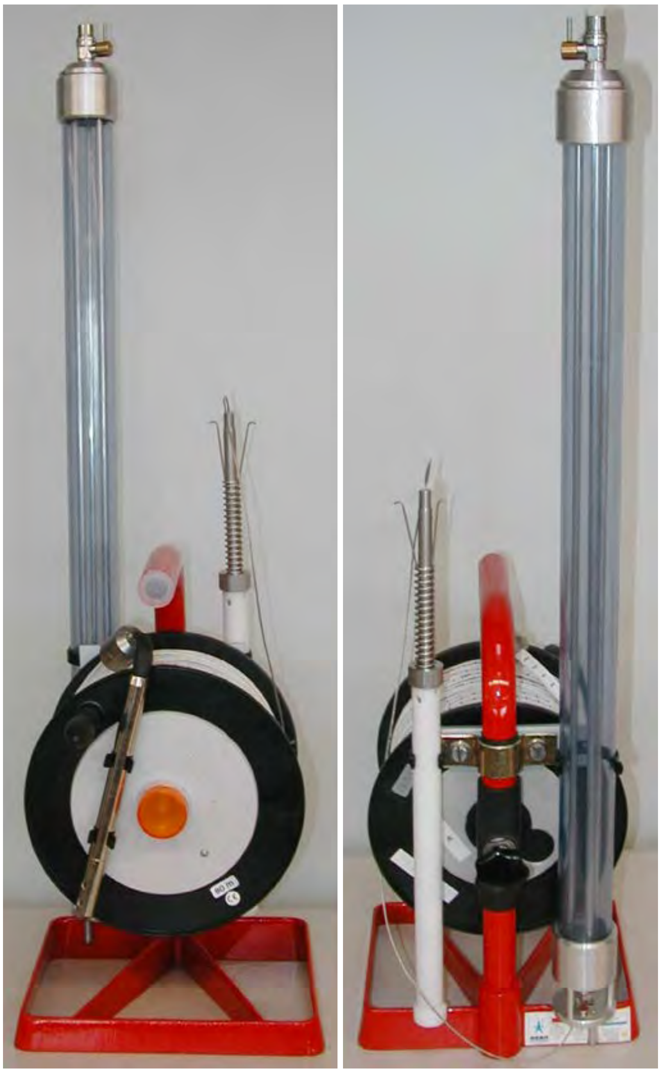


Figure 32 A KLL-S liquid sampler (4" diameter tube) connected to an electric contact meter (SEBA).

Table 28 Removal of organic matter, Mn and selected TrOCs during 111 days of continuous Mode 2 operation (Requirements: TrOCs <100 ng/L; Mn < 50 µg/L; DOC < 1 mg/L).

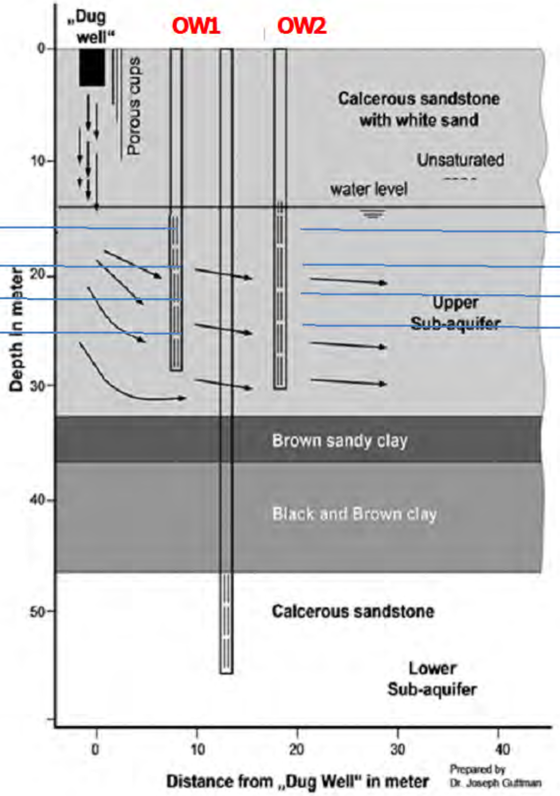
Date	No. of days from cont. ozonation startup	Sampling point	TrOCs (ng/L)								DOC	UVA	$SUVA_{254}$	Mn (total)
			IPDL	IHX	IPRM	BZF	VLX	CBZ	DCF	SMX	(mg/L)	(1000/cm)	(L/mg-m)	(µg/L)
27/06/2016	27	Secondary Effluents	1449	21952	4066	ND	246	878	337	211	9.5	203	2.1	18
		After Biofilter	1662	20732	3313	ND	392	1218	630	450	7.7	168	2.2	19
		After ozonation	573	5499	992	ND	ND	36	ND	ND	7.3	62	0.8	21
		Ceramic cups	199	143	83	ND	ND	44	ND	ND	2.1	40	1.9	<3
		OW1	315	ND	ND	ND	69	602	ND	139	1.3	35	2.7	43
		OW2	440	ND	ND	ND	ND	456	ND	123	0.9	26	2.9	167
26/07/2016	56	Secondary Effluents	1163	37074	8274	156	286	1037	517	380	10.4	148	1.4	23
		After Biofilter	1269	28609	6383	ND	337	1112	708	577	7.3	149	2	23
		After ozonation	587	8309	1582	ND	ND	43	ND	ND	8.1	72	0.9	23
		Ceramic cups									2	43	2.2	<3
		OW1	406	ND	ND	ND	ND	994	ND	ND	1	26	2.6	46
		OW2	115	ND	ND	ND	92	174	ND	124	1	26	2.6	167
14/08/2016	75	OW1						ND						
		OW2						630						
16/08/2016	77	Secondary Effluents	778	37049	9366	99	275	870	995	306	9.6	193	2	22
		After Biofilter	837	36859	7385	119	313	901	1023	375	7.8	151	1.9	22
		After ozonation	155	7178	1801	ND	ND	ND	ND	ND	7.4	67	0.9	24
		Ceramic cups	77	488	121	ND	4	ND	7	ND				
		OW1	68	ND	ND	ND	53	ND	ND	13	0.8	24	3	29
		OW2	419	ND	ND	ND	ND	934	ND	ND	0.8	26	3.3	138
04/09/2016	96	OW1						ND						
		OW2						710						

Date	No. of days from cont. ozonation startup	Sampling point	TrOCs (ng/L)								DOC	UVA	SUVA$_{254}$	Mn (total)
			IPDL	IHX	IPRM	BZF	VLX	CBZ	DCF	SMX	(mg/L)	(1000/cm)	(L/mg-m)	(µg/L)
13/09/2016	105	Secondary Effluents	1268	42026	9559	ND	241	901	557	403	8.7	194	2.2	20
		After Biofilter	1404	28697	6039	ND	254	1021	664	400	7.3	168	2.3	24
		After ozonation	827	10139	1976	ND	ND	ND	ND	ND	7.2	69	1	23
		Ceramic cups	ND	ND	ND	ND	ND	ND	ND	ND	1.6	38	2.4	3
		OW1	ND	ND	ND	ND	41	ND	ND	ND	0.8	22	2.8	35
		OW2	404	ND	ND	ND	ND	808	ND	ND	0.8	25	3.1	170
19/09/2016	111	OW1						ND			0.8	22	2.8	
		OW2						710			0.8	26	3.3	

ND – Non-detectable. Detection limit <10 ng/l. All samples from the observation wells OW1 and OW2 were taken after pumping around 3-5 m^3 of water from the wells so it is a dynamic sampling; Ceramic cups - static sampling of small quantities.

OW1

DOC (mg/l)	UVA (m^{-1})	CBZ (ng/L)	Depth (m)
0.9	2.1	<10	14
0.8	2.1	<10	18
0.9	2.2	<10	22
0.7	2.1	<10	26



OW2

DOC (mg/l)	UVA (m^{-1})	CBZ (ng/L)	Depth (m)
0.9	2.3	635	14
0.8	2.2	688	18
0.7	2.1	687	22
2.0	3.9	666	26

Figure 33 In depth analysis of organic matter (UVA, DOC) and CBZ along the perforated part of OW1 and OW2 conducted at day 111 of Mode 2 continuous operation.

* From 13 m. depth the pipes are perforated so that the in-depth samples represent the composition of the water in and surrounding the pipe. The samples were taken by a KLL-S liquid sampler that enables static sampling at the indicated levels with no need to pump out water from the pipe (Figure 32)

4.3.3.2.6 Bacteriology

Table 29 shows microbial counts (TOTB, CMPN, FMPN, and EMPN) measurements during July-August 2016 at the different sampling points along the process. It can be seen that TOTB in OW1 and OW2 were in the range of 10^3-10^4 cfu/mL, while all CMPN, FMPN and EMPN counts were negative. These counts were lower than those obtained during *Mode 1*, plausibly due to the disinfection effect of ozonation.

Table 29 Bacteriology measurements during 111 days of continuous Mode 2 operation.

Date	No. of days from cont. ozonation startup	Sampling point	Total count (TOTB, cfu/1mL)	Total coliforms (CMPN, MPN/100mL)	Fecal coliforms (FMPN, n/100mL)	Fecal streptococcus (EMPN, MPN/100mL)
26/07/2016	56	Secondary Effluents	8.E+03	2.E+04	2.E+04	5.E+02
		After Biofilter	1.E+05	>16000	>16000	>16000
		After ozonation	810	49	8	<1.8
		OW1	3.E+03	negative	Negative	<1.8
		OW2	4.E+03	negative	Negative	Negative
16/08/2016	77	Secondary Effluents	1.E+05	2.E+05	1.E+05	5.E+04
		After Biofilter	3.E+04	>16000	>16000	>16000
		After ozonation	880	22	4	<1.8
		OW1	2.E+04	negative	negative	negative
		OW2	6.E+03	negative	negative	negative

4.3.3.2.7 Bromide, Bromate

Two bromate measurements were conducted during the last 111 days of *Mode 2* operation. Bromate was formed as a by-product following ozonation in concentration of ~15 µg/L as a result of high bromide concentrations in Israel (~0.5 mg/L in Shafdan secondary effluent) and relatively high applied ozone dosage of ~10 mg/L. Bromate concentration following short SAT was 9-13 in OW1 and <5 µg/L in OW2. Bromate removal was not observed in slow and aerobic filtration (Zimmermann et al., 2011; Hubner et al., 2012). Bromate reduction can be attributed to biological process in anoxic zones where bromate reacted as an electron acceptor (Kirisits et al., 2001). So far, the final bromate concentration is close to the drinking water standard (10 µg/L) and below the proposed eco-toxicological threshold value of 3 mg/L (Hutchinson et al., 1997). However, further investigation and measurements are required. Establishment of oxic conditions throughout the aquifer will probably necessitate measures for bromate minimization, e. g. reduction of ozone dosages.

4.3.3.3 Summary – Mode 2

It was shown that ozonation decreased UVA by ~50%, decreasing subsequent DOC and UVA after short SAT (OW1, ~22 days HRT) down to 0.8 mg/L and 2.2 1/m, respectively. These values were lower than those obtained during *Mode 1*, and could be attributed to the contribution of ozonation in increasing biodegradability of DOC and in contributing high DO concentration.

In addition, it was observed that Mn concentration after short SAT (OW1, ~22 days HRT) during *Mode 2* was reduced to 29-35 µg/L, lower than the Israeli water reuse standards allowing 200 µg/L, but still higher than the concentration observed in the ceramic cups (<3 µg/L, located at a 1.5 m depth under the recharge borehole).

These results indicated some Mn dissolution in the short SAT system, as was observed in *Mode 1*. This was attributed to the high ammonium residuals that were obtained after the biofilter, which could not perform complete nitrification when ammonium concentrations were higher than 5 mg NH_4^+-N/L. Nevertheless, the ozone addition succeeded to minimize and control the Mn dissolution, presumably due to its high oxygen contribution (~20 mg/L). In general, operation in *Mode 2* met the enforceable Mn^{2+} standard for recharged water with a HRT of ~22 days.

Ozonation showed high removal of all bacteria types, giving ~$7*10^3$ cfu/mL total bacteria and <100 MPN or n/100mL for all other bacterial types (total and fecal coliforms, fecal streptococcus). Results after short SAT (OW1) showed lower bacterial counts than those obtained in *Mode 1* including ~$4*10^3$ cfu/mL total bacteria, no total and fecal coliforms, nor fecal streptococcus.

Regarding TrOCs, ozonation effectively removed all of the examined fast and moderate-reacting compounds VLX, CBZ, DCF, SMX, and BZF down to < 50 ng/L (98-100% removal), and decreased the slow-reacting compounds IPDL, IHX, and IPRM by ~60-72% due to the formation of OH^- radical. This resulted in reduction of all TrOCs in OW1 (~22 days HRT) after 56-77 days down to <100 ng/L, and further reduction down to <50 ng/L of all TrOCs at day 105 of continuous *Mode 2* operation.

In summary, biofiltration-ozonation pretreatment prior to SAT succeeded to address the goal of using a short SAT (~22 days) system, operating at high infiltration rate (~4-6 m/d), while maintaining and even improving SAT product quality (enhanced DOC, N-compounds, TrOCs, and microorganism removal, while controlling Mn dissolution).

However, it is important to note that the results obtained at OW1 during the continuous infiltration of biofiltered-ozonated effluents were acquired at a limited period of time of 111 days during summer (June-September). These results show a good feasibility of applying biofiltration-ozonation pretreatment prior to short SAT. However, this process should be further investigated and demonstrated for a longer period of time of at least 1 year (as was originally planned), including winter time, in order to confirm the results and to better understand the effect of these processes on SAT at a longer infiltration period. In addition, the issue of bromate formation should be further investigated and might require reduction of ozone dosage. Furthermore, the issue of ozone degradation products that may persist after SAT was not investigated in the current study and should be further investigated together with appropriate toxicity tests. In order to reduce process costs, it is suggested, based on the current results, to investigate the use of a conventional media filter instead of a biofilter prior to ozonation, and to see if this could still enable obtaining the desired product quality.

4.4 Capital and operational costs

The capital (CAPEX) and operational (OPEX) costs required for the different treatments was estimated for a scenario where 1000 m^3/h Shafdan secondary effluents surpluses will have to be treated in the conventional SAT system. Since the actual conventional SAT is at its full capacity and the actual infiltration rates are relatively low (~1 m/d), in order to treat additional amount of effluents, construction of new SAT ponds will be required.

Alternatively, the current study suggests infiltrating pretreated secondary effluents (including the surpluses) in the existing ponds at a higher infiltration rate. The considered pretreatments are: (1) biofiltration at lower filtration rates (~5 m/h) using PACl coagulant, flocculation tank, and hydrogen peroxide to remove ammonia from secondary effluents, (2) coagulation-biofiltration (as described in (1)) followed by ozonation; (3) Another option that was suggested based on *Mode 2* results, is using Media filter at higher rates (~10 m/h) added with in-line coagulation using PACl (without addition of H_2O_2 and ammonia removal) followed by ozonation.

Both filtration modes can significantly remove particles and facilitate good ozonation performance prior to short SAT. Due to the higher infiltration rates expected in both cases, there will be no need to construct new SAT ponds. However there would be potential for remobilising sorbed Mn at least in the short term and would need to understand the constraints on ozone in being sufficient to maintain aerobic conditions in the subsurface in the absence of H_2O_2 while not exceeding acceptable bromate concentrations.

The following four treatment possibilities were compared for 1000 m^3/h surpluses of Shafdan secondary effluents by calculating the CAPEX and OPEX costs:

1) No pretreatment. In that case, installation of new SAT ponds will be required.
2) Pretreatment by PACl coagulation and biofiltration (5 m/h filtration rate biofilter using hydrogen peroxide to remove ammonia) prior to the existing SAT system (was demonstrated in ***Mode 1*** at the current study)*.
3) Pretreatment by PACl coagulation and biofiltration (5 m/h filtration rate biofilter using hydrogen peroxide to remove ammonia) followed by ozonation prior to the existing SAT system (was demonstrated in ***Mode 2*** at the current study)*.
4) Pretreatment by in-line PACl coagulation followed by 10 m/h media filter and ozonation prior to existing SAT system (was not demonstrated in the current study). No ammonia is removed during the pretreatment but excess oxygen supplied by the ozonation is supposed to counteract the oxygen scavenging effect of ammonia in the upper layer vadose zone of the SAT system*.

*In treatments 2-4 no new ponds are required.

Basic Assumptions for the calculations are:

- Interest Rate 5%
- Economic lifetime: 25 years
- Manpower costs: USD 61,100
- Energy cost: 11.5 c/kWh
- Investment cost includes: treatment plant, site expenditures, management expenditures, civil work.
- Planning, management and supervision was estimated at 25%.

As can be seen fromTable 30, treatment no. 4 has the lowest capital and operational costs compared to all other treatments. In case of new SAT ponds, the main costs are the very expensive new land, and in case of biofiltration it is the hydrogen peroxide costs and the relatively low filtration rate that requires more area and equipment investment for the filtration stage. In case of biofiltration alone (added with coagulation, treatment no. 2) no all TrOCs are removed. Thus, treatment no. 4 has a high potential to be used as a cost-effective pretreatment prior to SAT. However, this process should be demonstrated in a pilot scale to ensure that it gives the desired product quality.

Table 30 CAPEX and OPEX comparison of four treatment for 1000 m³/h excess Shafdan secondary effluents.

[cent euro/m^3]	1.New SAT ponds	2. Coagulation-Biofiltration (Mode 1)	3. Coagulation-Biofiltration-Ozonation (Mode 2)	4. In-line coagulation-Media filtration-Ozonation (modified *Mode 2**)
Capital Cost (CAPEX)	14.8	3.6	4.7	2.8
Operational Cost (OPEX)				
Fixed	4.7	2.1	2.4	1.9
Variable				
Energy	1.3	6.3	10.2	7.4
Chemicals		4.6	4.6	1.2
Total (CAPEX + OPEX)	20.8	16.6	21.9	13.4

* Treatment no. 4 was not demonstrated in a pilot scale, and should be further investigated.

4.5 Summary and conclusions

Table 31 presents a comparison between the product quality obtained in OW1 at *Mode 1* and in *Mode 2* operation. *Mode 1* results include averaged results from the overall period (more than 1 year operation). *Mode 2* results include results between days 56-111 of continuous operation, in which ozone effect was already observed.

In *Mode 1* (April 2014-July 2015) the pretreatment before sSAT included biofiltration only during more than one year. During this treatment, ammonia, nitrite and phosphate were completely removed, and all pathogens were highly reduced. In addition, besides the persistent Carbamazepine (CBZ) and Iodine-organic contrast media IPDL, all measured TrOCs were effectively removed after sSAT. Mn concentrations found after sSAT during *Mode 1* were lower than 52 µg/L, indicating controlled Mn dissolution.

In *Mode 2* (August 2015-September 2016) the pretreatment included biofiltration followed by ozonation (~10 mg/L ozone dose, ~1.0-1.2 mg ozone/mg DOC) prior to sSAT. When *Mode 2* was operated continuously for almost 4 months (June-September 2016), values of DOC and UVA after sSAT were reduced down to 0.8 mg/L and 2.2-2.6 1/m, respectively, while Mn dissolution was controlled and maintained down to 29-35 µg/L at the sSAT well. No ammonia or nitrite were found, and good bacterial quality was obtained.

Concerning TrOCs, in the range of 56-77 days from continuous *Mode 2* startup, it was shown that all TrOCs and specially Carbamazepine (CBZ) were reduced down to <50 ng/L. In the last TrOCs campaign (105 days) all TrOC values were below detection limit, except for VLX which was 41 ng/L. The delay in the effect of the pretreatment on TrOCs removal (>56 days instead of ~22 days) was explained by the retardation of CBZ displacement in the upper soil layers of the pilot SAT (0-25 cm).

In-depth sampling in the sSAT observation well after 111 days of continuous *Mode 2* operation, showed (by low DOC, UVA, and CBZ along all sampled depths) homogeneity along the overall perforated section of the sSAT well (from -14 to -26 m), likely due to vertical flow outside the well screen from non-puckered layers. This result may suggest that the ozonated water completely covered the area around the observation well and positively affected the quality of the groundwater.

Generally, it was shown that a hybrid pretreatment process based on coagulatio-biofiltration-ozonation prior to short SAT (~22 days HRT) can effectively maintain good SAT product quality including removal of organic matter, microorganisms, and micropollutants, while controlling Mn^{2+} dissolution and maintaining high infiltration rate (~4-6 m/d). Up until now, SAT has been regarded as anaerobic and aerobic oscillation. The innovation of the current study is in suggesting a new concept of maintaining aerobic SAT.

Residual impacts of past operations and effect of the surrounding need to be accounted for. In addition, the suggested process should be further demonstrated at a longer period of time (at least 1 year) including winter time. Moreover, issues such as bromated/other byproduct formation and ozone degradation products should be taken into account. In order to reduce process costs, it is suggested, based on the current results, to investigate the use of a conventional media filter instead of a biofilter prior to ozonation, and to examine if this process can obtain the desired SAT product quality at a lower cost.

Table 31 Comparison between short SAT (OW1, ~22 days HRT) product qualities during *Mode 1 and *Mode 2*** continuous operation.**

	Results in OW1 during *Mode 1**	Results in OW1 during *Mode 2***
DOC (mg/L)	1.4	0.8
UVA (1/m)	4.3	2.2 – 2.6
Mn (μg/L)	42	29 – 35
NH_4-N (mg/L)	<0.03	<0.03
NO_2-N (mg/L)	0.03	0.03 – 0.04
NO_3-N (mg/L)	5.78	6.33 – 6.55
TOTB (cfu/mL)	$1*10^4$	$3*10^3 – 2*10^4$
CMPN (MPN/100 mL)	7	Negative
FMPN (n/100 mL)	2	Negative
EMPN (MPN/100 mL)	<1.8	<1.8
IPDL (ng/L)	1796	<10 – 68
CBZ (ng/L)	840	<10 or <50 ***
SMX (ng/L)	150	<13

* Average results of more than one year *Mode 1* operation.

** Results after >56 days of *Mode 2* continuous operation, in which ozone effect was observed.

*** CBZ concentration was measured during days 75-111 of *Mode 2* continuous operation in two different labs using two methods: GC-MS (10 ng/L detection limit, Tel-Aviv University) and LC-MS (50 ng/L detection limit, Mekorot lab). In both cases CBZ was found to be under detection limit.

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5 El Port de la Selva

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5.1 Site description and specific challenges

El Port de la Selva is located near Cap de Creus in the northern part of Catalonia (Spain). In this coastal village the population during the summer months is ten times the 1,000 permanent residents present in winter.

The village is not connected to the regional water distribution network and relies on local groundwater as its only water source. The municipal wastewater treatment plant (WWTP) was last refurbished in 1997 and has a capacity of 10,500 p.e. (secondary effluent). Due to dry periods in the first decade of the 21st century, El Port de la Selva, like other settlements along the Costa Brava, added a tertiary treatment step consisting of double-filtration, UV disinfection and residual chlorination to the WWTP. The additional treatment step has a capacity of 25 m^3/h. A dual-pipe network for reclaimed water was constructed in order to make use of reclaimed water during the summer for urban purposes. From the WWTP, the water is pumped up to an elevated storage tank, from which it can be distributed by gravity-driven flow to the city's reclaimed water network. A concession for urban use of reclaimed water has however not yet been obtained.

Average annual drinking water abstraction in El Port de la Salva is about 400,000 m^3, while the average annual volume of treated wastewater is in the range of 220,000 m^3. Both abstraction and wastewater volumes have large fluctuations between summer and winter (Figure 34). Furthermore, chloride concentrations in drinking water abstracted from the municipal well were above the drinking water limit of 250 mg/L during the autumn months (Figure 34).

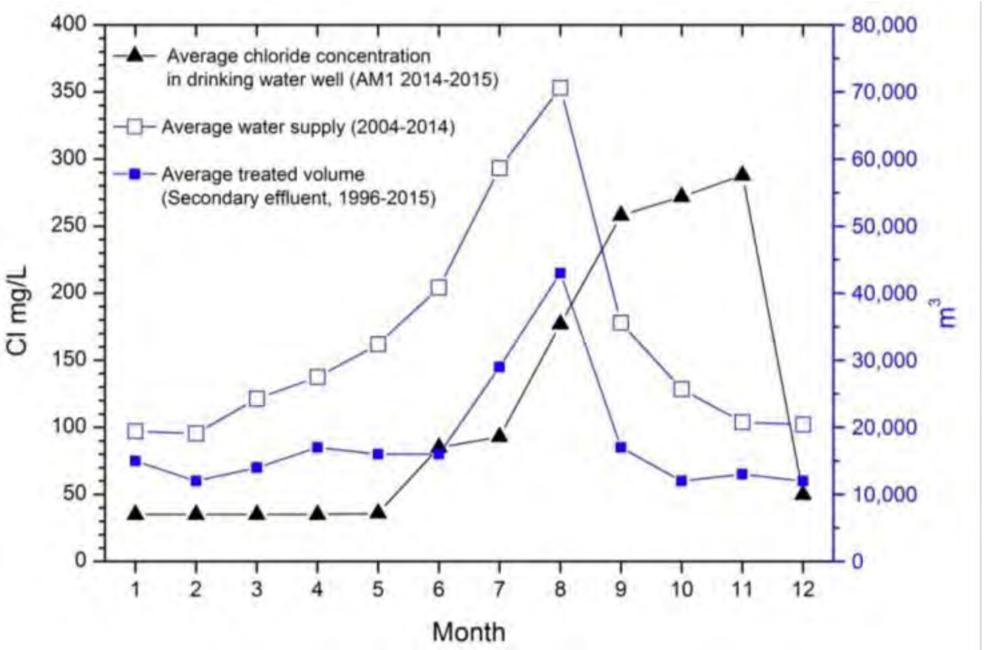


Figure 34 **Seasonal fluctuations of treated and abstracted water volumes and measured chloride concentration in drinking water well AM1.**

Water reclamation by aquifer recharge was intended to counteract these seasonal salinity peaks and to generally improve groundwater availability. Following a pre-feasibility study within the KWB-Veolia project “OXIMAR” (2011-2013), the operator envisaged to directly distribute reclaimed water for non-potable uses such as gardening or street cleaning during summer, and to recharge the local aquifer during winter. A site approximately 950 m upstream of the local drinking water wells, along the Riera de Rubies (or Riera de Romanyac) was preassigned as the recharge site prior to the start of the DEMOWARE project, and it was planned to infiltrate 200 m^3/d of tertiary treated within 200-240 d/yr, resulting in 40,000 – 48,000 m^3/yr (about 10% of the abstracted groundwater). Objective within DEMOWARE was to design infiltration structures with low maintenance requirements taking into account local hydrogeology and availability and quality of reclaimed water, and to optimize pretreatment to meet all regulatory demands for indirect potable re-use.

Planning started with the compilation of aquifer and water quality data for infiltration pond design and a review of the tertiary treatment process including options for alternative disinfection methods based on an entry level risk assessment as formulated in the Australian Guidelines for Water Recycling (NRMMC-EPHC-NHMRC 2009). Supported by the municipality and a local engineer, KWB and VWSI evaluated the performance of pretreatment and implemented the infiltration technologies. Amphos 21 was, in parallel, responsible for creating a conceptual model of the aquifer, setting the network of observation wells, monitoring hydraulic heads and evaluating groundwater recharge under various boundary conditions.

5.2 Entry-level assessment

The Australian Guidelines for Managed Aquifer Recharge (NRMMC 2008) were applied to the reuse site El Port de la Selva before constructing ponds and pipeline in order to demonstrate the feasibility from a risk-based approach. Data were compiled from personal communication with the local partners and several reports made available by them. They included site and aquifer description, source water characterization and defining the components of the SAT scheme (Table 32).

Table 32 Components of planned SAT scheme in El Port de la Selva

Component	System in El port de la Selva
Capture zone	WWTP El Port de la Selva
Pre-treatment	Secondary treatment + coagulation filtration + UV-disinfection + Chlorination
Recharge	Infiltration via constructed basins
Subsurface storage	Unconfined aquifer
Recovery	Extraction from aquifer via public and private wells
Post treatment	Chlorination
End use	Drinking water supply

While the Australian Guidelines summarize 18 attributes to describe the viability and degree of difficulty, a more detailed questionnaire of altogether 46 questions addressing water demand, site description, hydrochemistry, legal and social framework, economic considerations and necessary field investigations was developed within the EU-FP7 project DEMEAU (grant agreement no. 308339) and applied for El Port de la Selva within DEMOWARE. Both approaches yielded the identification of data gaps and necessary extent of field campaigns as well as the identification of main risks to be addressed by subsequent monitoring and evaluation during the pilot phase.

The resulting preliminary risk assessment matrix is summarized in Table 33. Data gaps, addressed in subsequent sampling and drilling campaigns, included the oxidation/ reduction potential and some trace inorganics (heavy metals; i.e. I, F, As) in tertiary effluent and native groundwater, the confirmation of aquifer thickness and presence/ absence of clay lenses (low kf) and determination of soil parameters (infiltration rate, loss on ignition). Further, the selection of pretreatment and WTTP upgrade based on the entry-level assessment. Generally, hazards towards the environment were considered to be higher than those towards humans. Transfer of trace organics to the aquifer and increase of DOC were identified as major hazards and thus closely monitored in subsequent sampling campaigns. At that stage, the project was also presented to the Catalan Water authority (ACA) and health authority. A full risk assessment including quantitative microbial and chemical approaches (QMRA/ QCRA) was part of work package 3 and will thus not be further reported here.

Table 33 Results of preliminary risk assessment based on available data prior to design and implementation

Hazard	Environmental endpoint - Aquifer	Human endpoint - Drinking water	Comments/ Action
Pathogens	Low	Low	> 2 log removal from UV disinfection > 1 log removal from infiltration > 5 log removal expected from Aquifer
Inorganic chemicals	Medium	Low: Heavy metals	
		Medium: Fe/ Mn	Mainly technical risk: Turbidity due the Fe/Mn + precipitation in distribution network
Salinity/ Sodicity	Medium	Medium	Strong seasonal fluctuation of EC
Nutrients	Medium: Transfer of N/P into the aquifer	Medium (10 mg/L NH_4-N) <40 mg/L nitrate-ammonia can reduce redox potential in the aquifer	Ammonia to be reduced to < 2 mg N/L for the WWTP
Organic chemicals	High: Transfer of micro-pollutants	Low: Transfer of micro-pollutants	
		High: Increase of DOC (increase in formation of chlorination by-products)	
Turbidity/ particulates	Low	Low	

5.3 Planning and design of recharge facilities

As demanded by the Catalan Water Authority (ACA) after presenting the project to them, the infiltration area was divided into three infiltration ponds of 200 m² each to manage expected clogging of the infiltration layer by wet-dry cycles and facilitate low maintenance operation. The design was based on the following assumptions:

- Flow rate: 200 m^3/d
- Turbidity of tertiary effluent: < 10 NTU
- Minimal infiltration rate: 1 m/d (expected after > 2-5 years of operation due to clogging)
- Only one pond is in operation, while one is drying and the third is dry for at least 7 d

Pond depth was set to 1.5 meters with a 50 cm thick layer of technical sand at the bottom of the ponds. The technical sand serves as a filter layer to homogenize infiltration rates, and facilitate maintenance. Final pond design is shown in Figure 35.

Pond construction followed a careful procedure and included connecting the storage tank with the ponds via a pipeline, as well as excavating, paving, and filling the three pond basins. During construction, the following adaptations were made on site:

- alignment of breakwater rocks along the pond perimeter to secure the slope stability
- reduced sand thickness (40 cm)
- adjusted water inlet to the ponds

Table 34 summarizes the final effective recharge areas. Some details of the construction works are shown in Figure 36.

Table 34 Effective recharge areas for the infiltration ponds 1-3

Basin ID	Recharge Area (m^2)
Basin 1	166
Basin 2	142
Basin 3	131
Sum	439

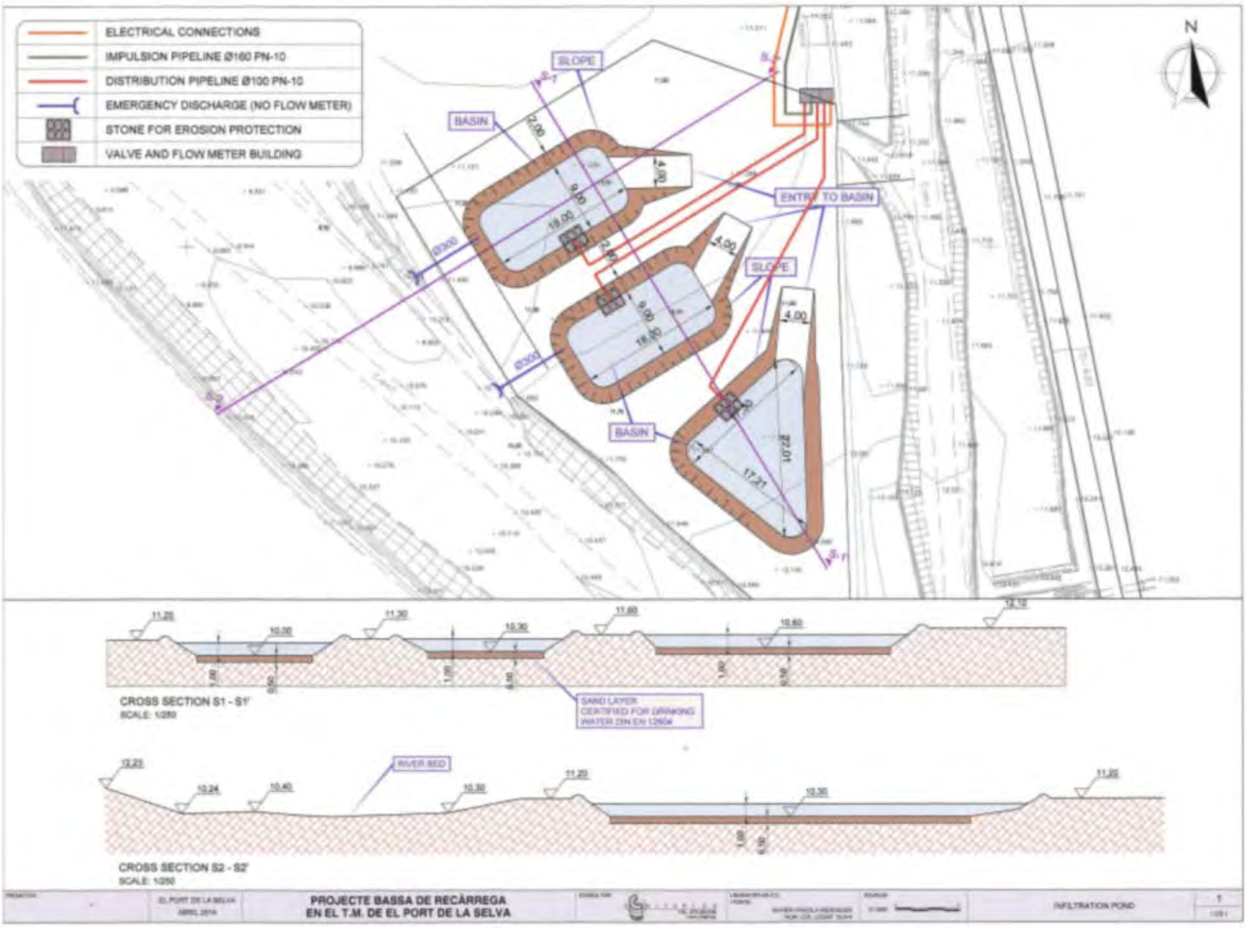


Figure 35 Pond design as in the tender procedure

Figure 36 **Work in progress.**
Top left: excavation work for infiltration pond; top right: slope stabilization with rocks; middle left: infiltration pond with layer of technical sand and water intake structure; middle right: excavation work for the pipeline; bottom left: distribution pipes to the three infiltration ponds; bottom right: housing at recharge site with distribution pipes and control devices.

As shown in the photos (Figure 36), embankments of the basins have been flattened and boulders used to stabilize the slopes. The bed of each infiltration basin was filled with a layer of quartz sand of minimum 80% SiO_2. The filter layer aims to ensure spatially constant infiltration rates, decreased maintenance efforts, and act as an additional filter for cleaning the source water.

Clogging will develop mainly on the surface of the technical sand, which can be removed, washed, and filled back in case infiltration rates decrease below a certain threshold. Grain size distribution for the sand layer is characterised by a steep slope, expressed by the low uniformity coefficient. Technical specifications for filter layer material requirements and final realization are shown in Table 35.

Table 35 Filter layer specifications

Filter layer	Requirement (according to DIN EN 12904)	Realization
Grain size (mm)	d_5 = 0.4 d_{95} = 0.8	d_5 = 0.3* d_{95} = 0.95*
Uniformity d_{60}/d_{10} (-)	<1.5	1.9**
Bulk Density (t/m³)	~1.55 (1.4 – 1.7)	~1.55***

*from grain size distribution; **calculated; ***estimated

5.4 Selection of pretreatment (pilot)

The aim of upgrades made at the El Port de la Selva WWTP under the DEMOWARE project was to identify an appropriate pretreatment for SAT. Low cost/low energy reuse schemes in particular should be considered to optimize pathogen and organic micro-pollutant removal, and reduce clogging. Prior to the recent upgrade of the WWTP in El Port de la Selva, the treatment train consisted of conventional primary and secondary stages, secondary settlers, and a tertiary stage with rapid granular filtration followed by UV disinfection and chlorine dosage (sodium hypochlorite). The advanced tertiary treatment was conceived to improve water quality for reuse application in private irrigation, via a dual-pipe distribution system. In addition to reuse for private irrigation, the planned WWTP effluent reuse was later extended to incorporate SAT. Operation would be for private irrigation in summer, and for SAT in winter. Despite E. coli and TSS regulations being more strict for private irrigation purposes than for SAT (Spanish Presidential Ministry (2007)), the addition of SAT implied increased water quality standards for nutrients. To address this, additional modifications to the system and operations were made within the DEMOWARE project.

Table 36 lists water quality parameters identified as crucial for successful water reuse. Associated operational concerns, target parameter values, and related measures taken or planned for mitigation are also listed. As noted in Table 36, chlorination was deemed unfeasible during SAT operation due to concerns about the formation of disinfection by-products from DOC and bromide being present in the effluent (cf. Table 33). After replacement of UV lamps in March 2014, all values were < 1000 eColi/ 100 ml, thus meeting the limit value for aquifer recharge for indirect potable reuse. Design transmittance of 40% (although actual values are mostly between 50 and 70%) further provided very high disinfection of virus indicator somatic coliphages, which are < 10 PFU/100 ml after UV.

Table 36 also shows the relatively ambitious target values set for nutrients (phosphorous, ammonium, total nitrogen). The measures to improve the treatment process included:

- installation of frequency converters to regulate tertiary filtration in order to reduce TSS/ turbidity peaks,
- assessed and performed tuning or calibration of the filtration and UV stages to improve E. coli removal,
- improved secondary control with online ammonium probe and increased aeration for ammonium reduction, resulting in NH_4^+ values < 1 mg/L under most operating conditions,

- improved secondary control and increased retention times within the WWTP for improving nitrate reduction,
- installation of online phosphate probe and addition of iron precipitant for phosphorous removal:
    - Addition before tertiary filters (stopped due to high head losses at relatively low doses of 4 mg Fe/L),
    - addition to aeration basins in a flow-proportional manner (~20 mg Fe/L), resulting in typical total phosphorous concentrations < 2 mg P/L,
- installation of online conductivity measurement and automated shut off of tertiary stages in case of exceeding 1,500 µS/cm threshold, with redirection of secondary effluent to the sea,
- shutting down tertiary chlorination during periods of SAT to avoid the formation of potentially toxic chlorination by-products,
- installation of a granular activated carbon stage for organic micro-pollutant adsorption.

Table 36 Essential WWTP water quality parameters in El Port de la Selva, associated concerns, targeted values for improved pretreatment, and taken/ planned measures.

Parameter	Associated concerns	Target	Measures taken/ planned
TSS (/ turbidity)	SAT clogging, oxygen depletion	as low as possible	Install frequency converters for improved filtration (+ additional benefits from precipitation)
E. coli	hygienic water quality	< 1,000/100mL	Improve filtration & UV disinfection schemes
Ammonium	microorganism growth, oxygen depletion	< 1 mg/L	Increase aeration
Total nitrogen	microorganism growth, oxygen depletion	< 10 mg/L	Improve control, increase WWTP retention time
Total phosphorous	microorganism growth	< 2 mg/L	Improve control, install precipitation
Conductivity (salinity)	salt concentration	< 1,500 µS/cm	Install online conductivity probe & provide system shut down in case of exceedance
Halogenated organics	Toxicity of disinfection by-products	as low as possible	Switch off chlorination during infiltration periods
Organic micro-pollutants	(Potential) toxicity	as low as possible	Install adsorption stage (granular activated carbon)

Wastewater volume and composition in El Port de la Selva are subject to strong seasonal variation due to highly variable volumes of the wastewater produced and variable nutrient loads (high in summer, low in winter, further discussed below). It is generally difficult to maintain constant WWTP effluent qualities throughout the year. Seasonal fluctuations since 2015 of essential water quality parameters of the WWTP effluent, excluding halogenated organics and organic micro-pollutants (as these were not frequently monitored in routine operation), are evident in Figure 37. The start of infiltration and operation of the SAT system in November 2015 is indicated in the conductivity graph of Figure 37 (bottom). Optimization of the wastewater treatment process occurred continuously over the whole period.

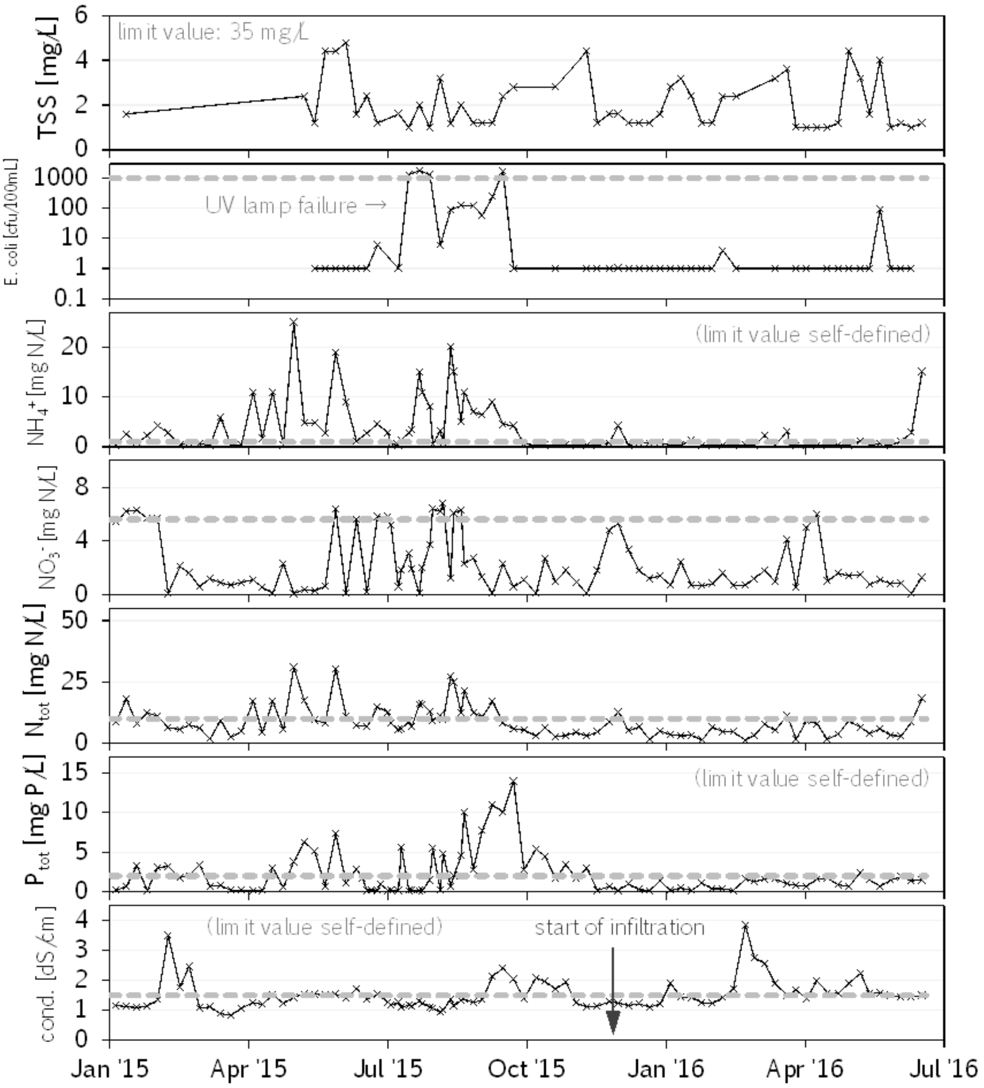


Figure 37 **Concentrations of total suspended solids (TSS), E. coli, ammonium (NH_4^+), nitrate (NO_3^-), total nitrogen (N_{tot}), total phosphorous (P_{tot}), and conductivity since 2015**

Limit values as dashed lines, start of infiltration period indicated in bottom graph; note that the limits for TSS, E. coli, nitrate, and total nitrogen correspond to the limit values defined in Spanish Presidential Ministry (2007), whereas the limits for ammonium, total phosphorous, and conductivity are self-defined.

Figure 37 demonstrates that most treatment steps appear to be working effectively since the start of infiltration. For TSS, effluent quality could not be improved further, but all values are below the limit value of 35 mg/L and only occasionally exceed 4 mg/L. E. coli as the hygienic indicator organism did not show any limit exceedances since the start of the infiltration. Some measurements above the self-defined limit value were observed for ammonium, but compared to the first half of 2015, this parameter is generally more stable in 2016 with only a few high values. Nitrate shows only one exceedance, still close to the limit value. Similar to ammonium, the fluctuations and levels of total nitrogen were clearly reduced with the process improvements. The increase of ammonium and total nitrogen towards June 2016 indicates higher incoming loads to the WWTP due to the increase of El Port de la Selva residents in summer. Installation and optimization of the phosphorous precipitation scheme resulted in better phosphorous removal. Although the source of moderate fluctuations in conductivity was not resolved, the fluctuations likely result from saltwater intrusion to the sewerage system/WWTP influent from the sea.

Where values exceed limits for some parameters shown in Figure 37, especially total nitrogen and E.C., Spanish regulations may require countermeasures be taken. Appendix I.B of the Spanish regulations may require additional monitoring and Appendix I.C temporary stops of the SAT infiltration until countermeasures have been taken and discussed with local authorities (Spanish Presidential Ministry 2007).

Sampling prior to the implementation of the SAT scheme focused on micro-pollutants in effluent compared to native groundwater. Three campaigns prior to infiltration yielded the analysis of trace inorganics and trace organics including the full list of priority substances (EU 2000b) as demanded by the Catalan Water authority and pharmaceuticals and pesticides selected based on literature and practical experience from other SAT sites (e.g. Berlin). Figure 38 shows concentrations of several organic micro-pollutants in a grab sample from WWTP effluent in February 2016 in El Port de la Selva. Concentrations of nearly 4 µg/L were obtained for valsartan acid, candesartan and olmesartan ranged around 2 µg/L, and gabapentin and venlafaxine around 1 µg/L. Concentrations between ~0.5–1 µg/L were observed for iopromide, methylbenzotriazole, diclofenac, 4-formylaminoantipyrine, and sulfamethoxazole.

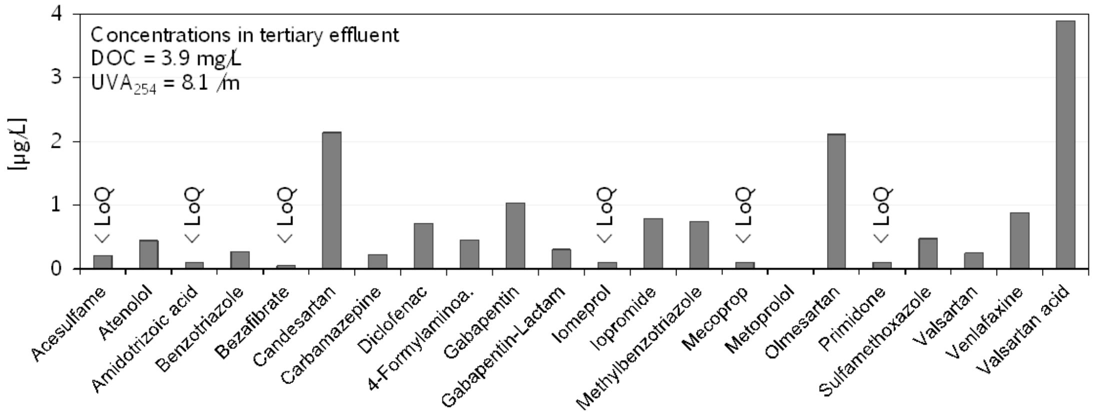


Figure 38 Organic micro-pollutant concentrations in a grab sample from the WWTP effluent in El Port de la Selva on 11 Feb 2016, with DOC concentration and UV absorbance at 254 nm.

As noted in Table 36, an advanced granular activated carbon stage for the removal of organic micro-pollutants has been planned. Ozonation was regarded not feasible given the high concentrations of bromide in the WWTP effluent that could lead to formation of toxic bromate. The dissolved organic carbon (DOC) concentration of ~4 mg/L is low compared to other SAT sites. Additional samples (data not shown) confirm that the DOC concentration of the WWTP effluent in El Port de la Selva can be assumed to lie in the range of 4–6 mg/L.

A granular activated carbon (GAC) adsorber pilot has begun. Using experience from the recent German projects ASKURIS and IST4R (Jekel et al. 2016a; Jekel et al. 2016b) a generalized alignment method from a recent publication (Zietzschmann et al. 2016b) and new knowledge about relatively high overall comparability of adsorption phenomena in different WWTP effluents (Zietzschmann et al. 2016a), preliminary lab tests with the water characterized in Figure 38 were conducted and the potential performance of an activated carbon stage estimated.

Lab tests show that the organic micro-pollutant and DOC removals at given DOC-proportional dosages of pulverized granular activated carbon (pGAC) are similar to those determined in Berlin, cf. Figure 39. Assuming that the WWTP effluent DOC concentrations in El Port de la Selva are ~50 % of those of the tested Berlin WWTP effluent, and given their similar DOC compositions (data not shown), the performance of GAC adsorbers is likely to be similar at the two locations (Zietzschmann et al. 2016b).

With the actual GAC adsorber dimensions (bed height 2 m, hydraulic loading rate 6 m/h, GAC grain size 0.4-1.7 mm) and a stop criterion of ~50 % breakthrough of gabapentin in the adsorber effluent (without biological degradation), an operation cycle of ~7,000 bed volumes (equivalent to ~97 d) can be assumed for the El Port de la Selva WWTP.

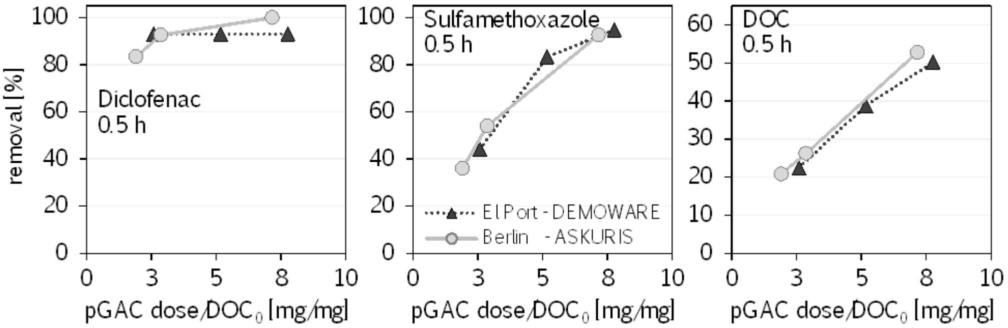


Figure 39 Diclofenac, sulfamethoxazole, and DOC removals in El Port de la Selva and Berlin after 0.5 h of adsorption at variable DOC-proportional dosages of pulverized granular activated carbon (pGAC)

Jacobi Aquasorb 5000; note the different y-axis scaling; apparently constant diclofenac removals in El Port de la Selva result from limit of quantification being underrun.

5.5 Monitoring

5.5.1 Overview of online monitoring system (WWTP, storage tank level and pond inflow)

In order to continuously control the infiltration, pond inflow controls were integrated into the existing Supervisory Control and Data Acquisition (SCADA) system of the WWTP (Figure 40).

The infiltration ponds are depicted in the lower left corner: *Llacuna 1-3*. Water inflow volume is not measured individually for each basin because ponds are not intended to be operated in parallel. Control points and threshold values have been set in the SCADA system to meet legal (Spanish Presidential Ministry (2007)) and self-defined thresholds for infiltration as described in Table 37.

Control devices are typically operated in hysteresis mode with an upper and lower threshold to ensure robust operating conditions.

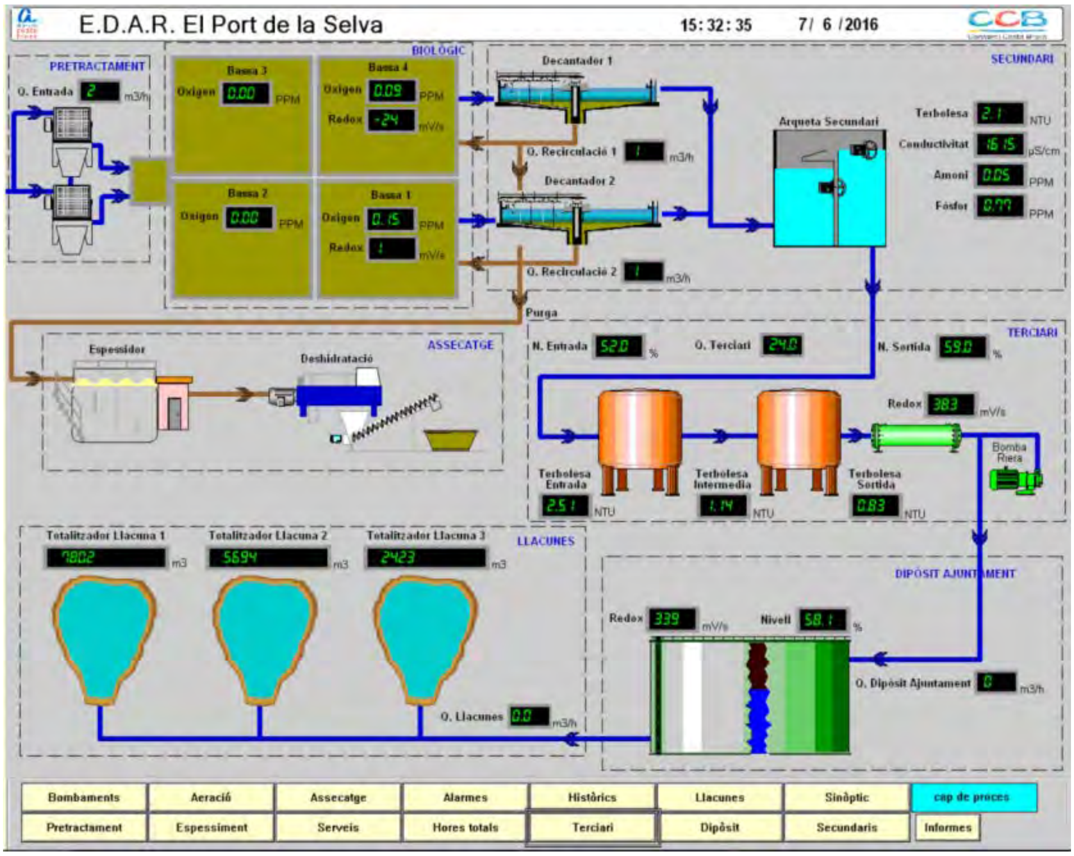


Figure 40 Overview of the SCADA control system in El Port de la Selva.

Table 37 Overview of parameters and threshold values implemented in the SCADA system for controlling the flow to the infiltration basins.

Parameter	Upper threshold (UT)	Lower threshold (LT)	Unit	Monitoring point	Goal	Action
Electrical conductivity	1500	1450	µS/cm	secondary effluent	salinity control	stop pumping to storage tank if above UT, start if below LT
Orthophosphate*	1.5	1.3	mg/L	secondary effluent	nutrient control	stop pumping to storage tank if above UT, start if below LT
Turbidity	2	1.8	NTU	2nd filter, tertiary	clogging control	stop pumping to storage tank if above UT, start if below LT
Ammonium*	1	0.8	mg/L	secondary effluent	nutrient control	stop pumping to storage tank if above UT, start if below LT

*logger installed 8/4/2016

5.5.2 Setup of the groundwater monitoring network (SAT performance in groundwater)

The monitoring network comprised the complete treatment chain from raw sewage, to secondary and tertiary treatment, infiltration ponds to some groundwater observation wells and finally the drinking water wells (AM1, AM2). Some groundwater monitoring wells are situated at the recharge site and pre-existing observation wells along the flow path to the drinking water wells and on the coastline were also used (Figure 41).

The groundwater monitoring network includes newly drilled sampling stations and pre-existing wells. Monitoring stations were used for various purposes according to their spatial position and hydraulic or logistical limitations (Appendix Table 60). Some monitoring stations were not suitable for sampling. For example, the Camping well water shows very high turbidity due to the presence of iron oxides. Because of this high suspension load in the Camping well, it was not possible to sample the well by submersible pump for risk of pump damage. PZ2 had a welded top cap through which only small items, for example, a dip meter, could be inserted. On the 3rd of June 2016 the top cap was removed, allowing for regular sampling and monitoring. Performance of the infiltration system was assessed with observation wells close to the ponds: PZ7, PZ6, and PZ4 (Figure 41). Other wells are not influenced by recharge activities and therefore serve as controls.

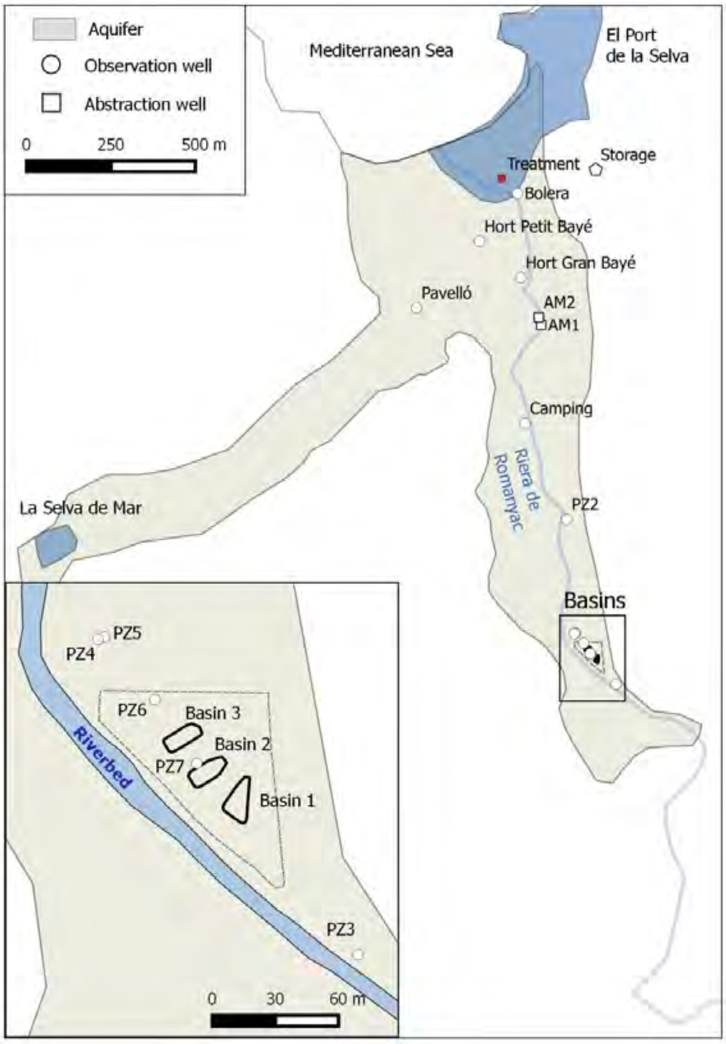


Figure 41 Overview of monitoring network.

An overview of parameters analyzed during monitoring is given in Table 38. Monitoring covered a wide range of hydrochemical parameters from standard hydrochemistry over inorganic trace compounds to various sets of microbial parameters.

Table 38 Sampling specification for each group of parameter

Parameter group	Specification
In-situ	Temperature, pH, Dissolved Oxygen, Electrical Conductivity, Alkalinity as HCO3 (sometimes measured as CaCO3)
Main compounds	Na, K, Ca, Mg, Cl, NO2, NO3, orto-PO4, SO4, NH4, HCO3, Br, F
Inorganic trace compounds	Al, Fe, Mn, Cu, As, B
Other	DOC, UVA254
Organic trace compounds	Different sets of parameters, analyzed by different methods and laboratories see 5.5.5.1 for more details
Microbiology I	Adenoviruses, Enteroviruses, Noroviruses, Rotaviruses
Microbiology II	Somatic phages, E.coli, Enterococci, Clostridium perfringens, Total coliforms

The sampling campaigns were structured in three phases: a screening phase prior to infiltration, operational monitoring, and a final campaign after infiltration (Table 39).

Table 39 Overview of sampling campaigns

Sampling campaigns	Sampled stations	Sampled parameter group	Comments/Purpose
3 campaigns from May 2014 to October 2015	TE, HPB, PZ2, PZ3, PZ4, PZ5, Roqueta*, AM1, AM2	In-situ, Main compounds, Inorganic trace compounds, Other, Organic trace compounds	Screening of tertiary effluent and native groundwater composition to fill gaps from entry-level risk assessment
10 campaigns from November 2015 to July 2016	RW, SE, TE, Pond, PZ7, PZ6, PZ5, AM1, HGB, HPB, Bolera	In-situ, Main compounds, Metals, Inorganic trace compounds, Other, Microbiology I and II, Organic trace compounds	Operational monitoring, performance assessment
1 campaign end of July		In-situ, Main compounds, Metals, Inorganic trace compounds, Other, Microbiology I and II	Post-infiltration

RW=raw sewage; SE=secondary effluent; TE=tertiary effluent; HPB=Horts Petit Bayé; HGB=Horts Grand Bayé; PZ=Piezometer; *not shown in overview map, sampled only once

Due to financial and logistical limitations, not all sampling campaigns covered all parameter groups. Detailed information for each sampling campaign can be found in Appendix Table 59.

In summary: three out of ten sampling campaigns focused on microbiology, inorganic trace compounds were analyzed during five campaigns, and organic trace compounds were sampled during five campaigns.

In-situ parameter and main compounds were analyzed for all campaigns. Sampling was executed by Elisenda Taberna (VWSI) and Joan Sanz (VWSI), with support by Christoph Sprenger (KWB), Martí Bayer (A21) and other KWB and A21 staff.

5.5.3 Native groundwater composition (Aquifer baseline)

Monitoring of native groundwater aimed to assess spatial and temporal variability of groundwater hydrochemistry and hydraulics to describe aquifer conditions without anthropogenic impacts. The aquifer baseline study included monitoring carried out prior to infiltration operations and also existing hydrochemical information from other studies.

There are basically three methods available to distinguish between a natural and an anthropogenically influenced groundwater composition: i) statistical component separation (Kunkel et al. 2004); ii) data preselection (Mueller et al., 2006); and iii) historical data analysis (Griffioen et al. 2008). In this study, the hydrochemical composition of native groundwater was calculated based on the pre-selection method (Mueller et al. 2006). The pre-selection method is based on the idea that certain chemical compounds which indicate exclusively anthropogenic influence can be used as tracers. When these substances are detected or exceed certain thresholds, the groundwater sample is regarded as anthropogenically influenced and excluded from further evaluation. Tracers include pharmaceuticals and pesticides, which are exclusively of anthropogenic origin, or tracers that usually occur in very low concentrations such as nitrate and heavy metals. After this selection procedure, the upper and lower threshold of the regarded compound is then expressed with confidence intervals (usually 10th and 90th percentile).

In this study, samples with EC > 600 µS/cm have been excluded (e.g. some samples from AM1 and AM2). The Bolera well was excluded because it is subject to large fluctuations in EC due to its location close to the shoreline. Native groundwater composition was calculated from the remaining 24 samples as shown in appendix Table 61.

5.5.4 Field methods

5.5.4.1 Hydrochemistry

Pond water was collected directly into the sampling bottles. Groundwater was abstracted with a submersible pump (Geotech SS Geosub). On-site parameters (pH, temperature, electrical conductivity and dissolved oxygen) were measured with HACH HQ40 device in a flow-through cell. After on-site parameters were stable and at least three static water volumes of the observation wells had been pumped out, samples were taken and stored in polypropylene or glass bottles. All samples for ion determination were filtered on site with 0.2 µm acetate cellulose filters. The sample for cation measurements was acidified to pH 2 with ultra-pure HNO_3 and one bottle of each sample was not acidified and kept for anion determinations. Alkalinity, as HCO_3, was determined by HCl titration in the field using a Merck (Darmstadt, Germany) acidity test. Percentage error of ion balances and saturation indices were calculated with PHREEQC. Values for bicarbonate were added for samples without measured value. Samples with ion balance error ≥10 % were excluded from further evaluation.

5.5.4.2 Microbiology

Microbial sampling in groundwater (Microbiology I+II) was performed with disinfected pumping equipment. One day before sampling, the pump, hose and electrical cable were washed in a solution of 1% sodium hypochlorite.

In order to neutralize this disinfection solution, the pumping equipment was then washed in a solution of 1% thiosulphate for at least two hours. In each step the pump was operated for at least 30 minutes to sterilize then to neutralize and flush out possible contamination residues from the inner parts. Latex gloves were used for all sampling activities.

5.5.5 Lab methods

5.5.5.1 Organic trace compounds

Groundwater sampling for TrOC´s was performed with a polytetrafluoroethylene (PTFE) hose connected to the GeoSub pump in order to minimize secondary contamination. Samples were sent to different laboratories for subsequent analyses as detailed below.

TUB

All samples for the Technical University of Berlin (TUB) were filtered through 0.45 mm membranes prior to chemical analysis. Organic trace compounds were quantified by HPLC-MS/MS using direct injection of 25 mL on a TSQ Vantage (Thermo Scientific, USA). Detailed information can be found in Altmann et al. (2015a)

CSIC

All samples were for Consejo Superior de Investigaciones Científicas (CSIC) filtered through 0.45 mm membranes prior to chemical analysis. The Spanish National Research Council (CSIC) used LLE and Gas Chromatography coupled to Mass Spectrometry in Tandem (GC-MS/MS) for TrOC´s analytics.

BWB

All samples for Berliner Wasserbetriebe (BWB) were filtered through 0.45 mm membranes prior to 115chemical analysis. The certified BWB laboratory analyzed all samples according to DIN 38407-F36.

JRC

The Joint Research Centre (JRC) provided their “Mariani box” to the El Port de la Selva WWTP staff. This field kit consists of filters in a specially designed portable box where the sample water is pumped through. Sample volumes were 0.5L for effluent and 5L for groundwater samples. The filters were subsequently sent to the JRC lab and were analyzed using LC-MS/MS.

5.5.5.2 Somatic coliphages

Somatic coliphages were measured according to ISO 10705-2 by Empresa Mixta d'Aigües de la Costa Brava S.A. (ISO 2000b). Secondary effluent and raw sewage samples were analyzed with 0.1 mL or 0.01 mL of sample volume, resulting in a LOQ of 1000 pfu/100 mL and 10 000 pfu/100 mL, respectively. Groundwater samples were tested with 10 mL of sample volume resulting in a LOQ = 10 pfu/100 mL. Please note that after the first microbial campaign in February 2016, LOQ was decreased by increasing the sample volume, resulting in a LOQ of 5 pfu/100 mL in pond and 2 pfu/100 mL in groundwater samples.

5.5.5.3 Total coliforms

Total coliforms were isolated and enumerated according to the methods described in Environment Agency (2002a) by Empresa Mixta d'Aigües de la Costa Brava S.A..

5.5.5.4 E. coli

Escherichia coli were quantified according to APHA et al. (1989) and Environment Agency (2002a) by Empresa Mixta d'Aigües de la Costa Brava S.A..

5.5.5.5 *Enterococus faecalis*

Enterococcus faecalis was quantified according to the methods reported in APHA et al. (1989), Environment Agency (2002b), and ISO (2000a) by Empresa Mixta d'Aigües de la Costa Brava S.A..

5.5.5.6 *Clostridium perfringens*

Clostridium perfringens is a ubiquitous indicator for fecal contamination and produces spores resistant to disinfection techniques. It was quantified as described in Environment Agency (2002c) by Empresa Mixta d'Aigües de la Costa Brava S.A..

5.5.5.7 Viruses (Rotavirus, Enterovirus, Norovirus, Adenovirus)

The virological parameters of Rotavirus, Enterovirus, Norovirus, and human Adenovirus were measured at the laboratory of “Virus contaminants of water and food” at the University of Barcelona. The detection of viruses in the environment requires the concentration of the viral particles into small volumes. All water samples were concentrated with the skimmed milk flocculation protocol (Calgua et al. 2013a; Calgua et al. 2013b). 140 µl from the skimmed milk flocculated concentrates, were used in a nucleic acid extraction (QIAmp® Viral RNA kit (QIAgen, Inc.)) and were recovered in a total of 90 µl. The viral concentrations were quantified by specific qPCR assays for HAdV (Bofill-Mas et al. 2006; Hernroth et al. 2002), NoV GGI (da Silva et al. 2007; Svraka et al. 2005), NoV GGII (Kageyama et al. 2003; Loisy et al. 2005), Entero Virus (Allard et al., unpublished) and RoV (Zeng et al. 2008). To establish a limit of detection (LOD), dilution series for different concentrations of genome copies of each virus were tested. From these curves, the theoretical limit of detection with a 95% probability and 95% confidence level were determined (Table 40). Some lab results were reported slightly below LOD (e.g. Human Adenovirus in groundwater), but these results were also considered as positive by University of Barcelona.

Table 40 Limit of detection for viruses

	Human Adenovirus (GC/L)	Norovirus type I (GC/L)	Norovirus type II (GC/L)	Rotavirus (GC/L)	Enterovirus (GC/L)
LOD	105 (75-176)	100 (77-171)	742 (585-1021)	950 (725-1477)	1015 (745-1674)

5.6 Results

5.6.1 Hydrogeology

Based on the piezometer drillings at the recharge area, the shallow geology is a relatively thin (13-14 m) aquifer composed of poorly sorted and poorly rounded metamorphic rocks in gravel and block size, embedded in a matrix of sand and silt. At the bottom of the aquifer, compact plastic clays were encountered of about 5-6 m thickness in PZ7 and PZ6. These alluvial and colluvial deposits of young Pleistocene to Holocene age are characterized by a high unconformity and overlay a metamorphic series of low permeability with local flow through fractures. A hydrogeological cross section based on the drill logs at the recharge site is shown in Figure 42.

Grain size analysis of sediment samples from drill cores at the recharge site yielded hydraulic conductivities in the range of 4-600 m/d (Appendix Table 58). During drilling, groundwater was encountered at 6-8 m below ground level. Groundwater mounding below infiltration ponds at an inflow rate of 8 m^3/h was about 1 m.

Organic carbon content of sediment samples is 0.1-0.2 weight % measured in drilling samples and slightly elevated in samples from shallow excavations (0.14-0.23 wt. %) (Appendix Table 58).

In general, the C_{org} content is relative low and it can be concluded that particular organic carbon in sediments will not contribute to reactive processes to a large extend.

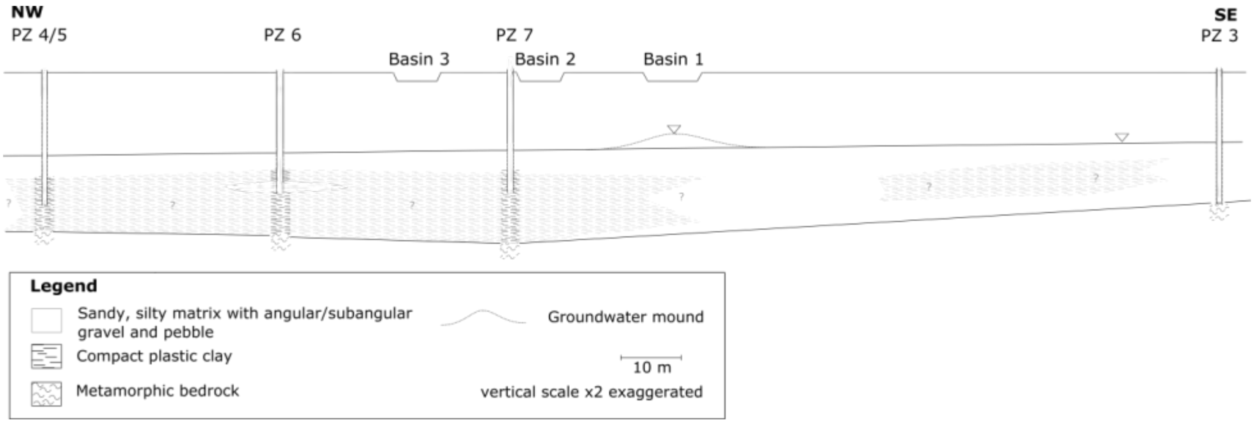


Figure 42 **Geological cross-section based on drill logs from observation wells (PZ3-PZ7) and approximate height of groundwater mounding below infiltration ponds.**

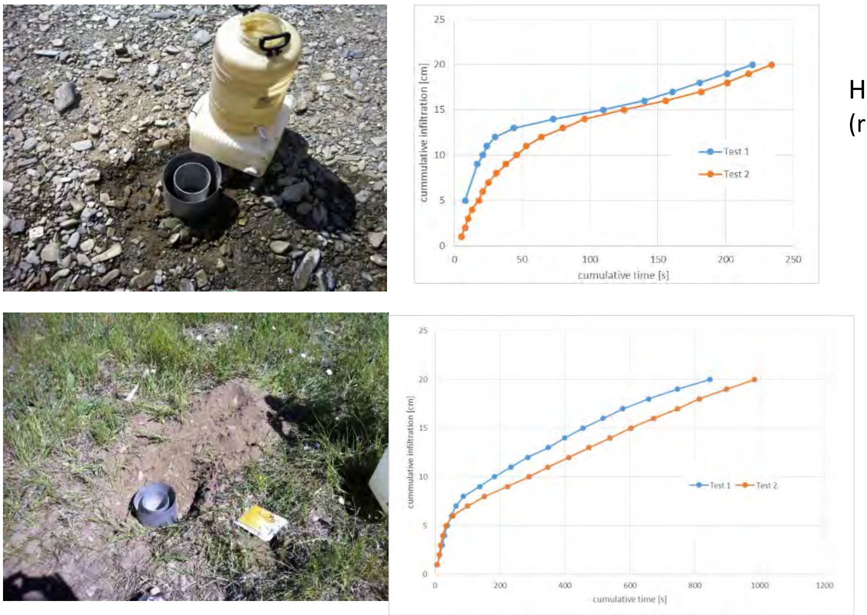
The transmissivity was estimated by pumping tests in the Bolera well with 825 m²/d and a hydraulic conductivity of 60 m/d (Eptisa 2010). These pumping tests were performed with low pumping rates (5 and 10 l/h) and drawdown could not be observed in external piezometers so interpretation was done using drawdown at the pumping well. Pumping in well AM1 for water supply causes drawdown in wells AM2 and HGB. Using these measurements, we could estimate a value of hydraulic conductivity of 218 m/d by applying Dupuit equation for unconfined aquifers:

$$h^2(r_2) - h^2(r_1) = \frac{Q}{\pi K} Log\left(\frac{r_2}{r_1}\right)$$

where $h(r)$ [L] is head for steady state conditions at a distance r [L] from the pumping well, Q [L^3 T^{-1}] is pumping rate and K [L T^{-1}] aquifer hydraulic conductivity. From this equation, hydraulic conductivity can be obtained from observations of head at AM2 and HGB (two different locations r_1 and r_2) through:

$$K = \frac{Q}{\pi\left(h^2(r_2) - h^2(r_1)\right)} Log\left(\frac{r_2}{r_1}\right)$$

In addition, two infiltration tests were conducted in the upper soil layer at the planned infiltration site and the river bed near the basins location providing estimates of the vertical hydraulic conductivity of 35 m/d and 13 m/d (Figure 43).



Hydraulic conductivity
(river bed)

$$K = 34.6\ m/d$$

Hydraulic conductivity
(organic soil)

$$K = 13\ m/d$$

Figure 43 Infiltration tests conducted in river bed and soil at the recharge site.

Grain size analysis of sediment samples from drill cores at the recharge site yielded hydraulic conductivities in the range of 4-600 m/d (Appendix Table 58). During drilling, groundwater was encountered at 6-8 m below ground level. Groundwater mounding below infiltration ponds at an inflow rate of 8 m^3/h was about 1 m.

5.6.2 Water level fluctuations and electrical conductivity

Water level fluctuations of selected wells at the drinking water abstraction site, cf. Figure 41, with rainfall from “Nautic” station situated in El Port de la Selva and measured electrical conductivity are shown in Figure 44. Well locations are shown in Figure 41.

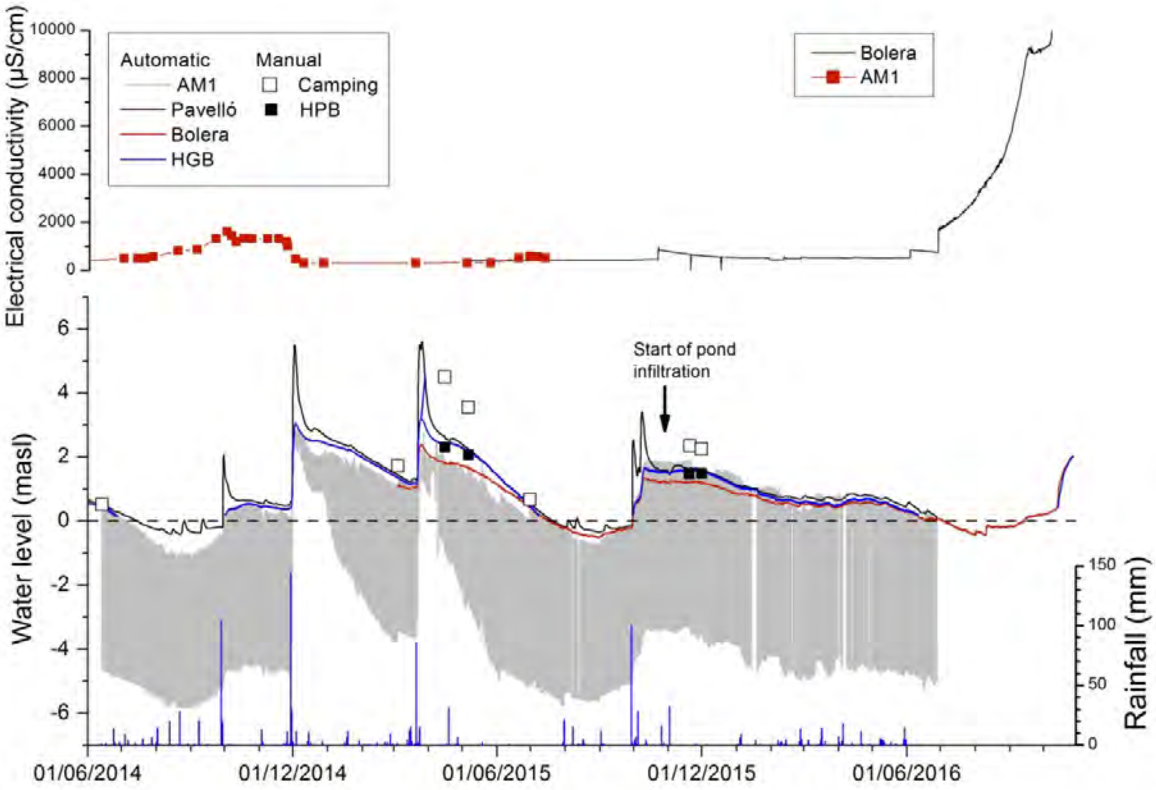


Figure 44 Water level fluctuations and electrical conductivity of selected wells further down-gradient of the recharge zone

Drinking water well AM1 (also AM2, not shown in Figure 44) was operated for few hours a day, usually about 10 h/d in winter and up to 20 h/d in summer (details are shown in annex Figure 71). Measurement interval of automatic loggers in AM1 was 15 minutes, resulting in a broad range of water levels showing the drawdown in AM1 of about 5 m. When AM1 was pumping water level was below sea level during most of the time, while other monitoring wells (Pavelló and Bolera, not used for pumping) show water levels below sea level only in autumn.

In 2014 water level below sea level was observed for about 3 months from July to September (AM1, Pavelló), while in 2015 water levels dropped below sea level for about 2 months from August to September (AM1, Pavelló, Bolera). In HGB (Horts Grand Bayé) the automatic recording was interrupted when water level dropped below sea level, because the well went dry during these times. However, it is very likely that the water level in the aquifer at the location of HGB was following the general trend and dropped also below sea level in autumn. The manual measurements in Camping followed the general trend observed by automatic water loggers and were (almost always) 1-2 m above Pavelló measurements. It is thus not clear if the water level in this part of the aquifer was below sea level.

The manual measurements in HPB (Horts Petite Bayé) were slightly below HGB and it can be concluded that the water level in this part of the aquifer is also below sea level in autumn.

Water level fluctuations show a fast response to rainfall events. Time lag between rainfall events >50 mm and increase of water levels in AM1, Pavelló, Bolera and HGB were often about 1 day. The highest increase in water level was observed in Pavelló, e.g. 124 mm rainfall event of two days (28th and 29th of September 2014) resulted in 2 m rise of water level, and 175 mm rainfall event of three days (from 28th – 30th of December 2014) resulted in 5 m rise in Pavelló and more than 2 m rise in HGB.

Short and intensive rainfall events are typical for the study area and it can be concluded that intense rainfall events contribute to natural groundwater recharge to a large extent. Moreover, these strong rainfall events usually occur after the hot and mostly dry summer period, when water levels are below sea level, and represent the main groundwater recharge period. The dependency of natural groundwater recharge on the occurrence of erratic rainfall events in combination with phreatic water levels below sea level make the aquifer highly vulnerable to seawater intrusion. The approximate aquifer area where seasonally occurring water levels below sea level may induce lateral inflow of seawater and that is highly vulnerable to seawater intrusion is shown in Figure 45.

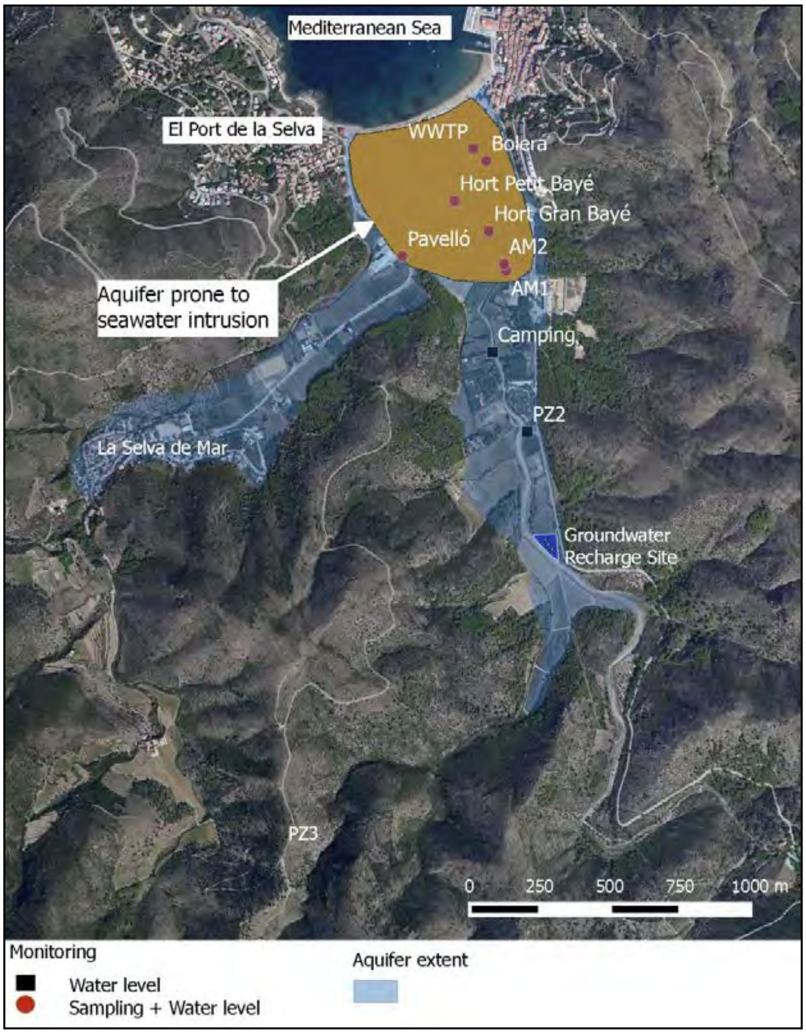


Figure 45 Approximate aquifer area prone to seawater intrusion based on seasonally occurring water levels below sea level.

Water level and electrical conductivity (EC) measured in groundwater observation wells at the recharge site are shown in Figure 46.

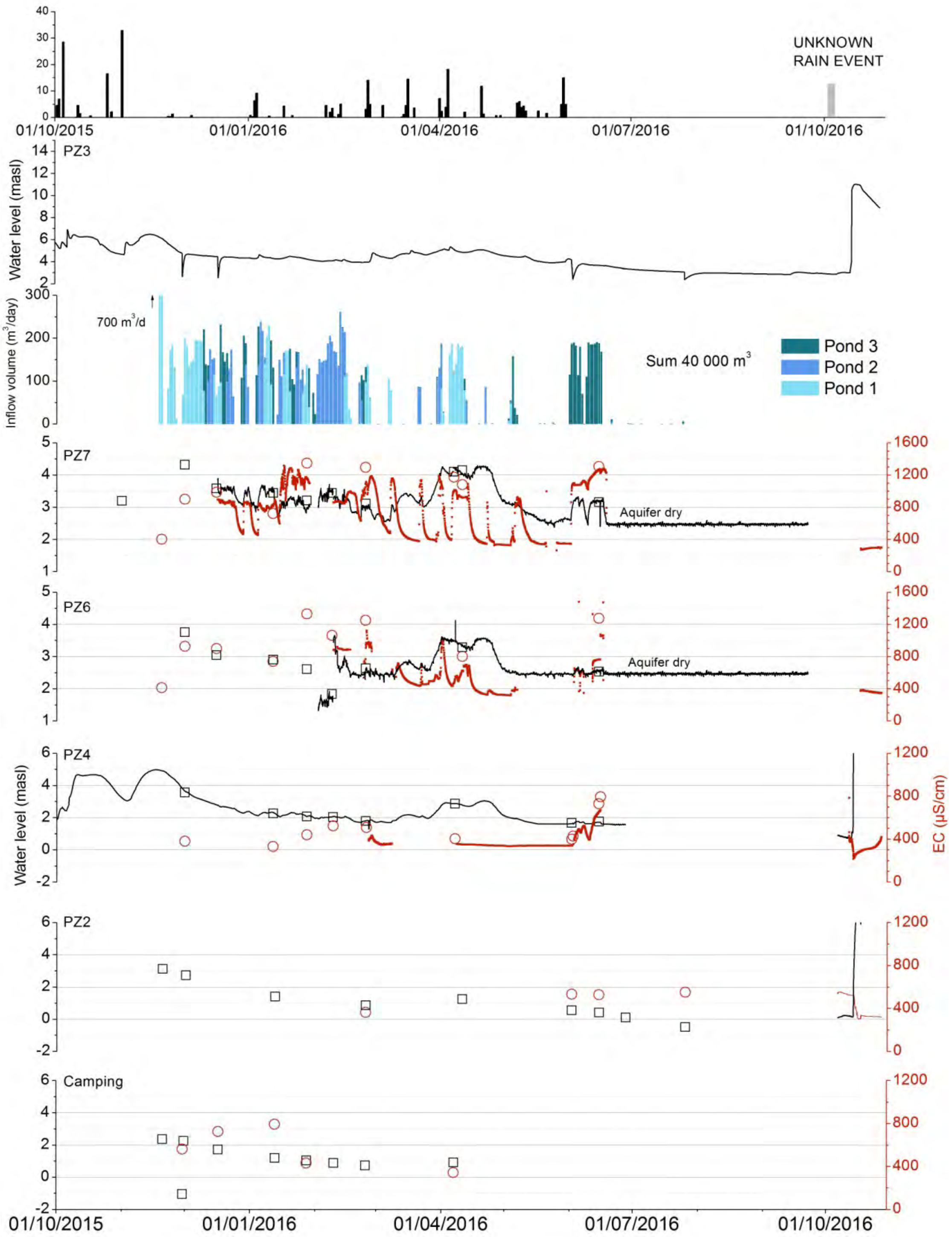


Figure 46 **Water level (masl) and electrical conductivity (µS/cm) measurements in groundwater with rain fall (mm) from "Nautic" station.**
Groundwater observation wells are ordered along flow path from PZ3 (top) up-gradient of infiltration basins to Camping (bottom) down-gradient. Red circles are manual EC measurements, black square manual water level measurements.

Observation wells close to the recharge site (PZ7 and PZ6) show a clear response on infiltration by an increase of electrical conductivity and, if infiltration stopped, rapid decrease to background values. Water levels in PZ7 and PZ6 respond to infiltration in a similar way.

Both EC and water level fluctuations are superimposed by natural groundwater recharge due to rainfall. Rainfall was measured at the "Nautic" station at the port in El Port de la Selva (top graph in Figure 46). Strong rainfall events measured at "Nautic" are reflected by rapid water level increases, e.g. Oct. –Nov. 2015 water level increase in PZ4 and PZ3. In October 2016, after infiltration was stopped, water levels as observed in PZ3, PZ4 and PZ2 increased by several meters due to a strong rain fall event (no rainfall measurement was available for this event).

Manual measurements of electrical conductivity (red circle) and automatic measurements (red line) were in good accordance. Depending on which infiltration basin was in operation, EC peaks measured in PZ7 and PZ6 responded differently. When basin 3 alone was in operation (e.g. June 2016) EC measured in PZ7 and PZ6 responded in a similar way, with similar peak values (~1200 µS/cm) and similar, but delayed temporal response of breakthrough curve. Also, PZ4 responded on basin 3 infiltration by increased EC, but lower peak value (~800 µS/cm) indicated dilution effects.

When basin 1 alone was in operation (e.g. April 2016) EC in PZ7 was higher (~1100 µS/cm) compared to EC in PZ6 (~800 µS/cm), indicating dilution of the infiltrate with native groundwater. EC increase in PZ4 was not observed during basin 1 operation. From this observation it can be concluded that the infiltrate forms a plume in the aquifer and, depending on the spatial position of the operating basin, it is observed in the observation wells with different degrees of dilution. The approximate extent of infiltrate in the aquifer and main flow direction depending on the spatial position of the operating basin is shown in Figure 46.

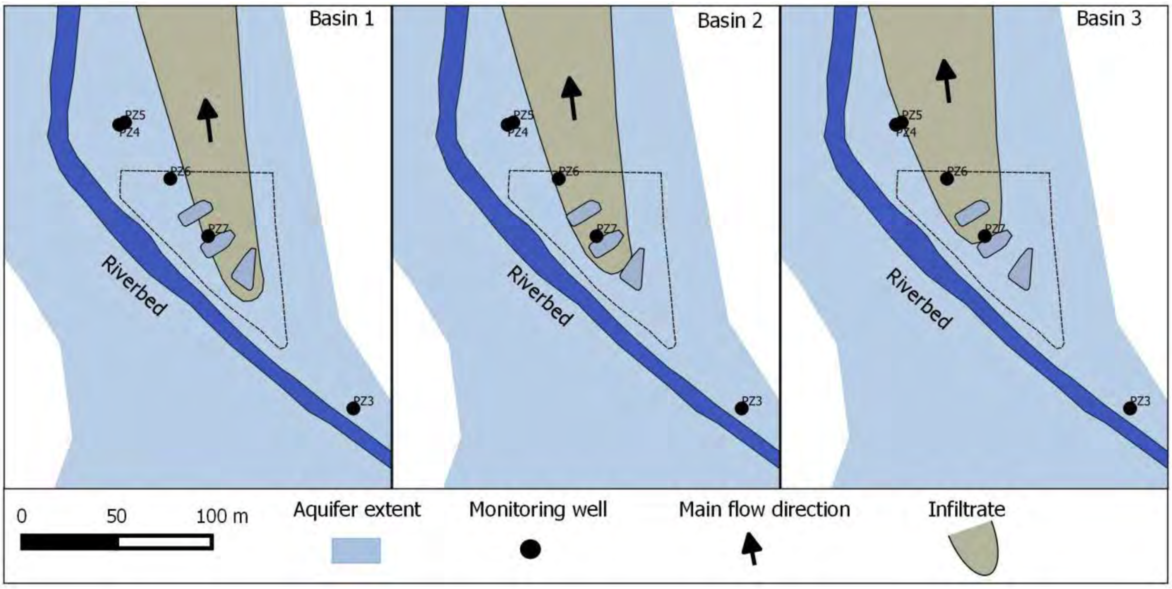


Figure 47 Approximate extent and flow direction of infiltrate in the aquifer at the recharge site.

While PZ6, PZ7 and PZ4 responded on basin infiltration to different extents, the propagation of the infiltrate further down-gradient is unclear.

5.6.3 Approximation of travel time by electrical conductivity

The large contrast between native groundwater (~500 µS/cm) and infiltrate (~800-1500 µS/cm) together with the relatively low reactivity in the subsurface allowed for the use of electrical conductivity as a tracer. Travel time estimations of infiltrate during subsurface passage at the recharge zone are based on breakthrough curves (BTC) of electrical conductivity.

Normalized BTCs measured in PZ7 from infiltration basin 1 are shown in Figure 48 (right) and in PZ6 and PZ7 from infiltration basin 2 in Figure 48 (left). During the inflow experiment from infiltration basin 1, PZ6 was not correctly equipped with a logger and no BTC could be measured.

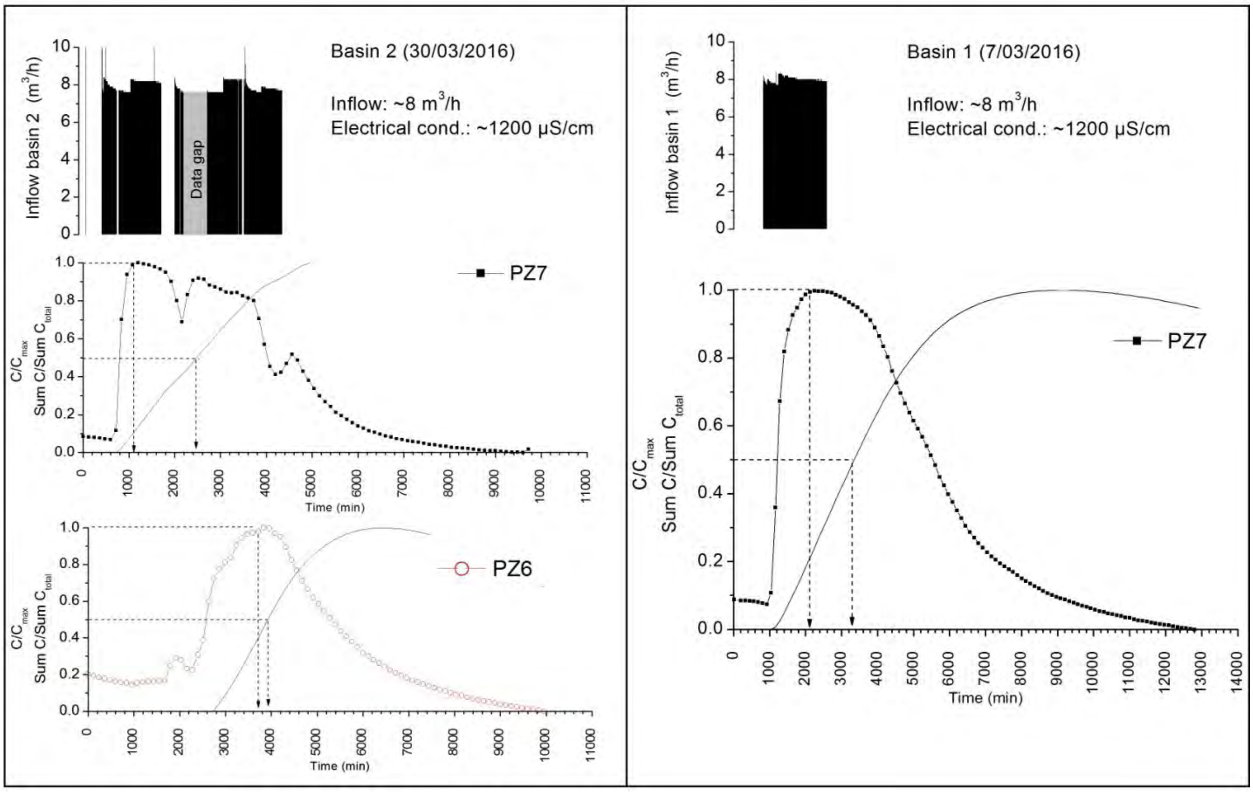


Figure 48 **Normalized breakthrough curves electrical conductivity between infiltration pond 1+2 and observation wells PZ6 and PZ7.**
Solid lines without symbols indicate the sum curves (left: measured on the 30th of March 2016; right: measured on 7th of March 2016).

BTC of electrical conductivity indicating travel time between basin 1 and PZ7 was measured on the 7th of March 2016. Inflow to the infiltration pond started at t= 643 min with a relatively constant rate of 8 m^3/h for a period of 1383 min. First response of electrical conductivity in PZ7 was detected at t= 1080 min and peak concentration of the tracer (C/C_{max} = 1) was detected after t= 2240 min. Arrival of peak concentration C/C_{max} = 1 indicates the dominant travel time, while the median flow velocity is defined by breakthrough of 50 % of the tracer mass. Median flow velocity was determined by sum C/sum C_{total} = 0.5 and was detected after t = 3200 min. The average flow velocity (V_{avg}) characteristic for the infiltration site is defined by $V_{dom} > V_{avg} > V_{med}$.

In a second tracer test BTCs from basin 2 to observation wells PZ7 and PZ6 were tested. Normalized BTCs for PZ7 and PZ6 for a recharge event in infiltration pond 2 are shown in Figure 48 (left). Unfortunately, inflow data retrieved from the SCADA system was incomplete and the data gap is shown as a grey area in Figure 48 (left). Infiltration began at t= 508 min, first arrival of the tracer was detected in PZ7 at t=650 min and peak concentration of the tracer (C/C_{max} = 1) was reached at t= 1200 min. In PZ6 first arrival of tracer was detected at 1795 min and peak concentration of the tracer was reached at t= 3835 min. Sum C/sum C_{total} = 0.5 was detected at t = 3955 min. The resulting residence times, dominant and median flow velocities for the two tracer events are shown in Table 41.

Table 41 Travel time and dominant flow velocity calculated by breakthrough curves of electrical conductivity

	Distance from pond edge (m)	Dominant residence time (h)	Dominant flow velocity, V_{dom} (m/h)*	Median residence time (h)	Median flow velocity, V_{med} (m/h)**
Pond 1 – PZ7	22	26.6	0.8	42.6	0.5
Pond 1 – PZ6	57	-	-		
Pond 2 – PZ7	3	11.5	0.3	33.2	0.1
Pond 2 – PZ6	23	55	0.4	57	0.4

*based on maximal concentration; **based on 50% of tracer breakthrough

It must be noted that the dominant flow velocities are valid only for the recharge zone and cannot be used for extrapolation, e.g. to calculate travel times from infiltration ponds to the drinking water wells. This is due to an elevated hydraulic gradient caused by groundwater mounding below the infiltration ponds.

5.7 Technology Performance

5.7.1 Native groundwater (aquifer baseline)

Characterization of native groundwater composition (aquifer baseline) is important during risk assessment to define pre-existing conditions before managed aquifer recharge activities take place. Native groundwater composition can then be compared with samples influenced by MAR activities (cf. section SAT performance).

Hydrochemistry (EC, pH, HCO3, SO4, Cl, Na, K, Ca, Mg) of native groundwater was previously evaluated by ACA (Year unknown) for an aquifer close to El Port de la Selva in Llança. Most of the values fall in the 10th to 90th percentile range of those calculated in this study (appendix

Table 62), except for sulphate, which appears higher in the ACA (Year unknown), while chloride and sodium are higher in samples from the present study. Inorganic trace compounds and other parameters (Br, NO3, DOC, orto-PO4, F, SIO2, Al, Cd, Fe, Mn, Cu, As, B, Ni, Zn, Pb) characteristic of the native groundwater are shown in appendix

Table 62. Most of the inorganic compounds in native groundwater have been detected in typical geogenic concentrations. The native groundwater is low in DOC, Iron, Manganese and Arsenic. Fluoride concentrations were below detection limit. Nitrate is the dominant N-species and was found in concentrations which indicate an anthropogenic impact, but for a limited extent. Bromide is of meteoric origin and typical for coastal aquifers under the local climatic conditions (Alcala & Custodio 2008). Sodium and Aluminum concentrations are typical for silicate weathering. Dissolved oxygen was measured in concentrations between 4.7 – 8.8 mg/L in native groundwater. Redox potential (E_h) was not measured during this study.

Four samples classified as native groundwater have been analyzed for organic trace substances. Out of 45 analyzed compounds, none were above the limit of quantification (LOQ) in native groundwater samples. Metoprolol and Phenazone were detected once (!) in AM1, but were not found in any other native groundwater sample. Based on these results it can be concluded that the native groundwater in the El Port de la Selva aquifer is oxic and of almost pristine quality. Table 42 gives an overview about trace organic analyses for groundwater samples.

Fifteen out of sixteen analysed polycyclic aromatic hydrocarbons (PAH´s) have been detected in PZ3 in June 2016, but only naphthalene in concentration > 5 ng/L. The sum of all PAH´s in PZ3 is 40 ng/L, with naphthalene of almost 25 ng/L.

Out of nine analysed polybrominated diphenyl ethers only BDE-47 and BDE-209 were detected above LoQ. Both compounds occur in low concentrations of <0.2 ng/L. Out of 20 organochlorine pesticides, three were detected above LoQ, inclusive β- Hexachlorocyclohexane, γ- Hexachlorocyclohexane and p,p'-DDT, but all with detections <1 ng/L. Out of 23 pesticides, seven were detected >LoQ. The sum of all detected pesticides was 20 ng/L, where diuron and simazine show highest concentration of 10.6 and 4.3 ng/L, respectively.

Out of seven volatile compounds none were detected above LoQ in PZ3. Out of 11 perfluorinated compounds none were detected above LoQ.

None of the 51 substances analysed in PZ4 by BWB were detected above LoQ..

None of the 53 substances analysed in AM1 by BWB were detected above LoQ.

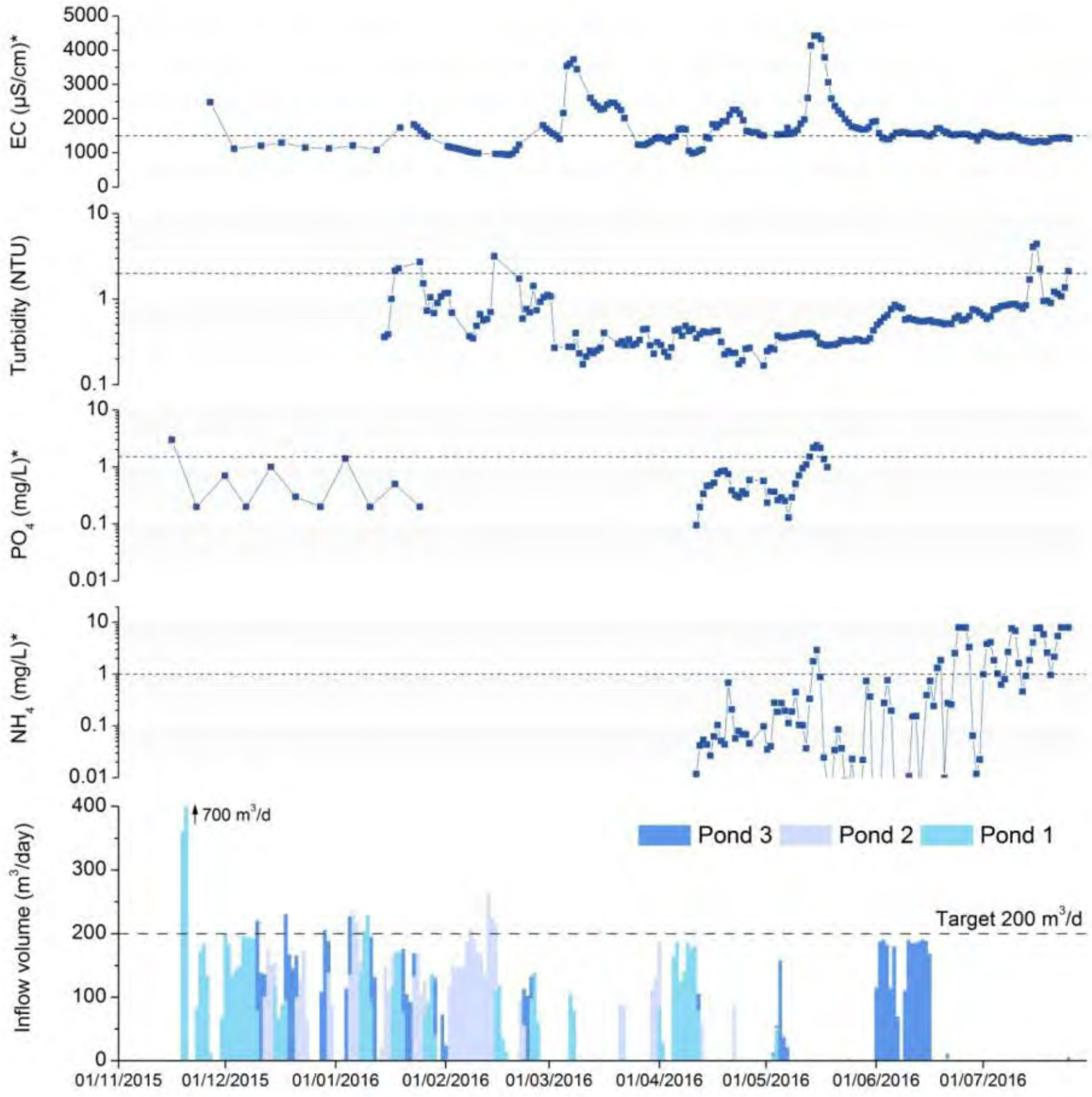
In PZ7 none of the measured substances was found in concentrations above LoQ.

Table 42 Overview of trace organic analysis in native groundwater

Monitoring station	Sampling date	Analysed substances	Laboratory	Comment
PZ3	02.06.2016	97 substances (Polycyclic aromatic hydrocarbons, Polybrominated diphenyl ethers, Organochlorine pesticides, Pesticides, Volatile compounds, Perfluorinated Compounds)	CSIC	Up-gradient of recharge area, not influenced by infiltration
PZ4	30.10.2014	51 substances (pharmaceuticals, some pesticides)	BWB	Down-gradient of recharge area, at time of sampling not influenced by infiltration
AM1	19.11.2015	53 substances (pharmaceuticals, some pesticides)	BWB	Down-gradient of recharge area, not influenced by infiltration
PZ7	20.11.2015	53 substances (pharmaceuticals, some pesticides)	BWB	Down-gradient of recharge area, at time of sampling not influenced by infiltration (sampling one day after start of infiltration)

5.7.2 SAT performance

The pond infiltration started on the 19.11.2015 with 360 m^3/d. The last day of infiltration was the 25.07.2016. In total about 18,200 m^3 were infiltrated, which is about half of the planned volume. Infiltration volumes and control parameters from the SCADA system are shown in Figure 49.



*measured in secondary effluent, dotted lines indicate threshold values for the respective parameter

Figure 49 Infiltrated volumes and control parameter (Electrical conductivity, Turbidity, Phosphorous, Ammonium).

The most problematic parameter in terms of exceeding threshold values was Electrical Conductivity (EC). Several times, EC was measured above 1500 µS/cm during the recharge period. When this occurred, pumping to the storage tank was stopped. Hence, EC was the most limiting parameter for infiltration. The reasons for the salinity spikes measured in secondary effluent could be leaky sewers or sea spray effects.

Turbidity was measured after the second filter in tertiary treatment and was mostly below the threshold value of 1.5 NTU. At the end of the recharge period, turbidity increased and was above the threshold value a few times.

Phosphate was mostly below the threshold of 1 mg/L, but was not measured continuously. After July 2016 ammonium concentrations made the effluent unfit for infiltration according to a threshold value of 1 mg/L.

5.7.2.1 Bulk chemistry, DOC, nitrate, phosphate

During groundwater recharge, a series of processes in the subsurface may occur that alter composition of the infiltrate. These processes include e.g. mixing with native groundwater, biodegradation, ion exchange, and sorption and are often called attenuation processes. In order to evaluate these processes, concentrations of main ions of the infiltrate in the pond, the infiltrate in the groundwater (PZ-7, PZ-6 and PZ4), and the native groundwater concentrations are shown in Figure 50.

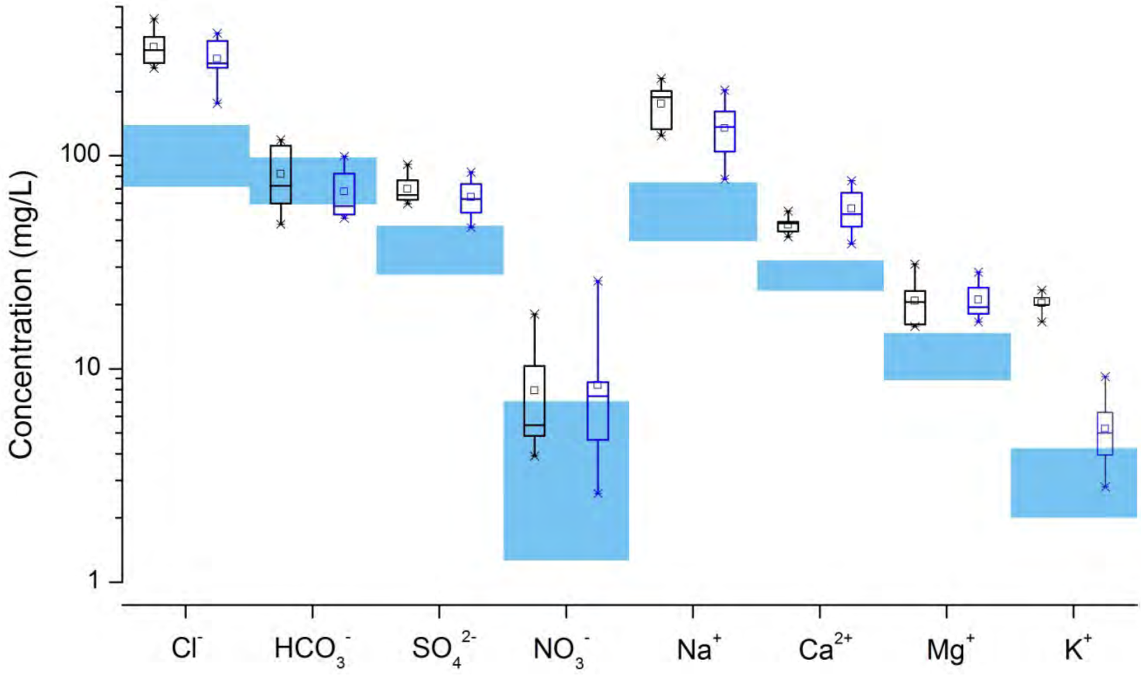


Figure 50 **Main compounds of infiltrate measured in the infiltration basins (black box plot, left) and during subsurface passage in PZ7 and PZ6 (blue box plot, right) compared to native groundwater composition 10th to 90th percentile range (blue band).**
Infiltration basin n = 8; Infiltrate groundwater n = 17.

Only groundwater samples where the influence of infiltrate was clearly observable (i.e. elevated EC) were used. In total, 16 samples, mostly from PZ6 and PZ7, meet this criterion. At the end of the recharge period in July 2016 the infiltrate reached PZ4. Therefore, one sample from PZ4 was also taken into account.

Compared to native groundwater composition, the infiltrate, as measured in the pond, is higher mineralized, mostly due to elevated Sodium and Chloride. Although infiltration is restricted to water with below the self-defined threshold of 1500 µS/cm, native groundwater concentrations of Na and Cl were exceeded by approx. 130 mg/L and 199 mg/L, respectively. Salinity control is very important for this study and it is recommended to further decrease the threshold for electrical conductivity.

Chloride is a non-reactive anion and the difference in chloride concentration between the infiltrate measured in the pond and in groundwater indicates dilution effects. In this case, the proportion of native groundwater in the groundwater samples was between 8-12 %. Therefore, changes in the hydrochemistry observed between the pond and the groundwater were mostly due to attenuation processes other than dilution.

Ions showing increasing concentration in groundwater as compared to pond samples result from processes that release these ions from the geological media. Dissolution of ions, for example, is a common process during groundwater recharge and may be observed for Mg and Ca.

Both ions are usually available in abundance and lead to increasing concentrations along the flow path. These dissolution effects are then accompanied by an increase of HCO3. An increase in HCO3 was however not observed and thus it seems to be more likely that Ca and Mg are subject to ion exchange processes. Other cations, such as Na and K, show decreased concentrations in groundwater. These ions were, in addition to dilution, likely subject to ion exchange processes, too.

Other compounds such as nitrate were measured in concentrations within the range of the native groundwater. Therefore, these compounds do not pose a risk to groundwater quality.

Sulphate is found in elevated concentrations in pond water samples (71.3 mg/L) and concentrations in the groundwater (63.8 mg/L) can be attributed entirely to dilution effects with native groundwater. Therefore, sulphate is stable under the given conditions and will be transported beyond the attenuation zone.

Phosphate is not shown in Figure 50, but was detected in pond water in two out of eight samples at concentrations of 0.4 mg/L. Out of 16 samples taken from groundwater phosphate was always below the detection limit (<0.1 mg/L) in groundwater. Phosphate appeared to be attenuated during infiltration and was not released to the aquifer beyond the zone of attenuation.

Compared to DOC concentration in native groundwater (~1.2 mg/L), DOC concentration in pond and groundwater was elevated with concentrations of 4.2 mg/L and 2.1 mg/L, respectively. Dilution with native groundwater alone would decrease the concentration to approximately 3.7 – 4 mg/L. Hence, the remaining decrease to 2.1 mg/L was attributed to degradation processes.

Bromide in pond and groundwater was found slightly elevated as compared to native groundwater composition. Bromide is a non-reactive anion and only subject to mixing effects.

5.7.2.2 Inorganic traces (Fe, Al, B, Cu, Mn, As, Br)

Inorganic trace compounds (Fe, Al, B, Cu, Mn, As, Br) measured in pond and groundwater and the range of native groundwater compositions are shown in Figure 51. It should be noted that inorganic trace compounds were measured only a few times and the results are therefore less meaningful than main inorganic measurements.

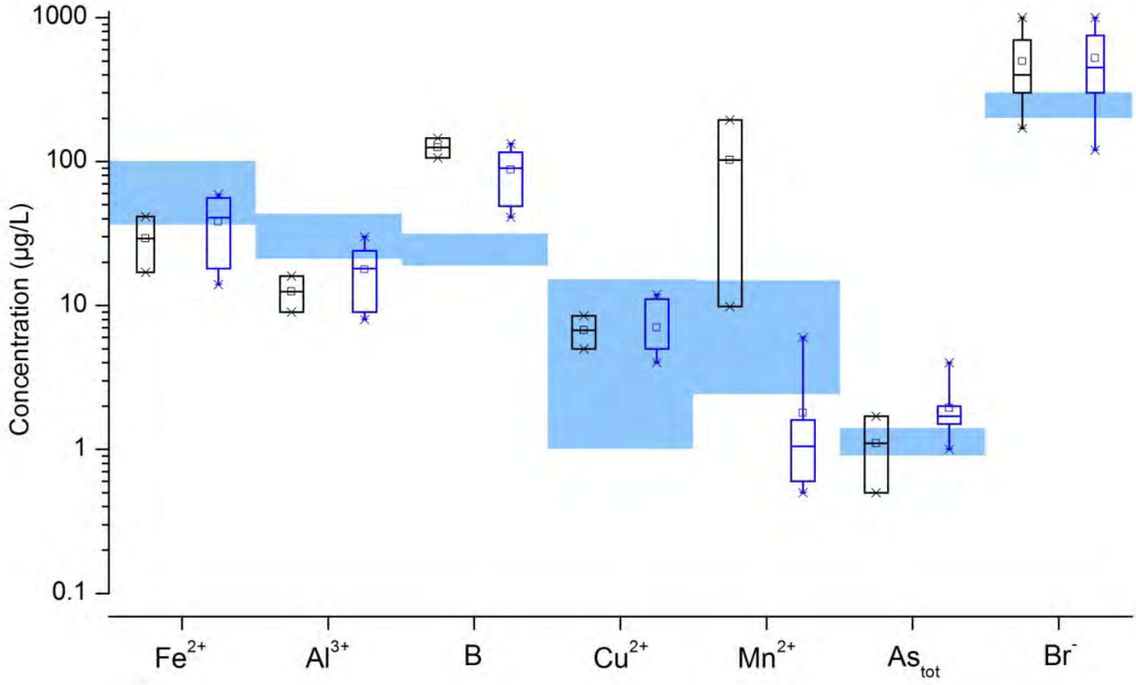


Figure 51 **Inorganic trace compounds of infiltrate measured in the infiltration basins (black box plot, left) and during subsurface passage in Pz7 and Pz6 (blue box plot, right) compared to native groundwater composition 10th to 90th percentile range (blue band).**
Infiltration basin n = 2; Infiltrate groundwater n = 6.

Iron and aluminum concentrations were measured below native groundwater levels in the pond and show an increase in groundwater. Mixing with native groundwater alone would result in 30-34 µg/L iron and 13-15 µg/L for aluminum. Hence, the remaining increase to the measured concentrations in groundwater (Fe=38.1 µg/L; Al=17.8 µg/L) can be attributed to dissolution.

Boron is a typical wastewater indicator, used for example as washing powder additive, and not removed in the WWTP. Boron was found in elevated concentrations of 125 µg/L in pond water. Mixing with native groundwater (based on Cl as conservative tracer) would result in concentration between 113 -116 µg/L. Measured concentrations in groundwater were much lower (87.7 µg/L) and it remains unclear how to explain these concentrations. However, drinking water standard for Boron in the EU is 1 mg/L and thus concentrations present at El Port de l Selva do not pose a risk in terms of drinking water standards.

Copper was found within native groundwater concentrations, both in pond and groundwater samples. Manganese was measured in concentrations below native groundwater concentrations. Differences between pond and groundwater samples for Mn and Cu were marginal and do not allow for interpretation.

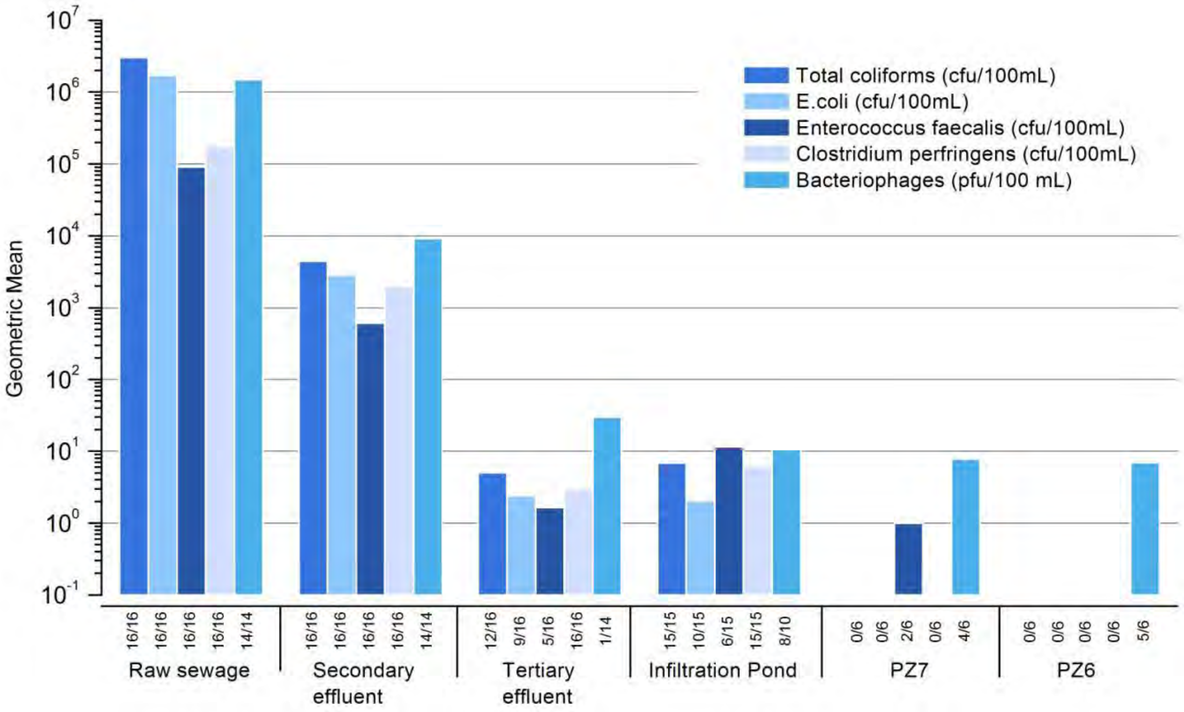
Arsenic in pond water was measured in concentrations similar to native groundwater, while groundwater samples show an increase above native groundwater composition with average concentrations of 1.9 µg/L. This increase may be attributed to minor dissolution effects in the aquifer. Arsenic concentrations are far below drinking water thresholds (10 µg/L) and it seems unlikely that under the given oxic redox-conditions a further concentration increase must be expected.

5.7.2.3 Microbiology

Geometric means of Total coliforms, *E. coli*, *E. faecalis*, *Clostridium perfringens* and Bacteriophages measured in raw sewage, secondary effluent, tertiary effluent, infiltration pond, PZ7, and PZ6 show decreasing concentrations along the treatment train (Figure 52).

Analytical results for all measured microbiological parameters (bacteria and virus) can be found in the appendix: Table 64,

Table 65, Table 66, Table 71, Table 72, and Table 73.



4/6 = four positive out of six samples; cfu = colony forming units; pfu = plaque forming units

Figure 52 **Geometric means of total coliforms, E.coli, E.faecalis, clostridium perfringens and bacteriophages measured during treatment train.**

It is noteworthy that chlorination in the WWTP of El Port de la Selva was not in operation during the sampling period; the removals reported here were achieved only by tertiary filtration and UV disinfection. Overall, the results show effective WWTP performance regarding microbial contamination and high compliance with the limit values as defined in the Spanish Ministerial Decree for water reuse. Hence, high continuity and robustness of the advanced wastewater treatment stages can be assumed.

Removal of microbial indicator organisms achieved by the advanced tertiary treatment is around 3 log units for total coliforms, E. coli, and Clostridium perfringens, and around 2.5 log units for E. faecalis and Bacteriophages (Table 43). On the path from the tertiary effluent to the infiltration ponds, concentrations of Total coliforms, E. faecalis and Clostridium Perfringens show a slight increase. The recharge site is protected a fence, but on the river bed side it was not allowed to construct a proper fence. This increase is attributed to contamination by animals that can easily access the open infiltration ponds. In the groundwater, all bacteriological parameters were below detection limits except for PZ7 where E. faecalis was detected in two out of six samples. It should be noted that these positives were close to the detection limit of 1 cfu/100 mL and further along the flow path (PZ6) E. faecalis was no longer detected.

The $\log_{10}$ reductions calculated from percentage changes of geometric means (Log_{10} = 100/(100-%change)) of measured concentrations are shown in Table 43.

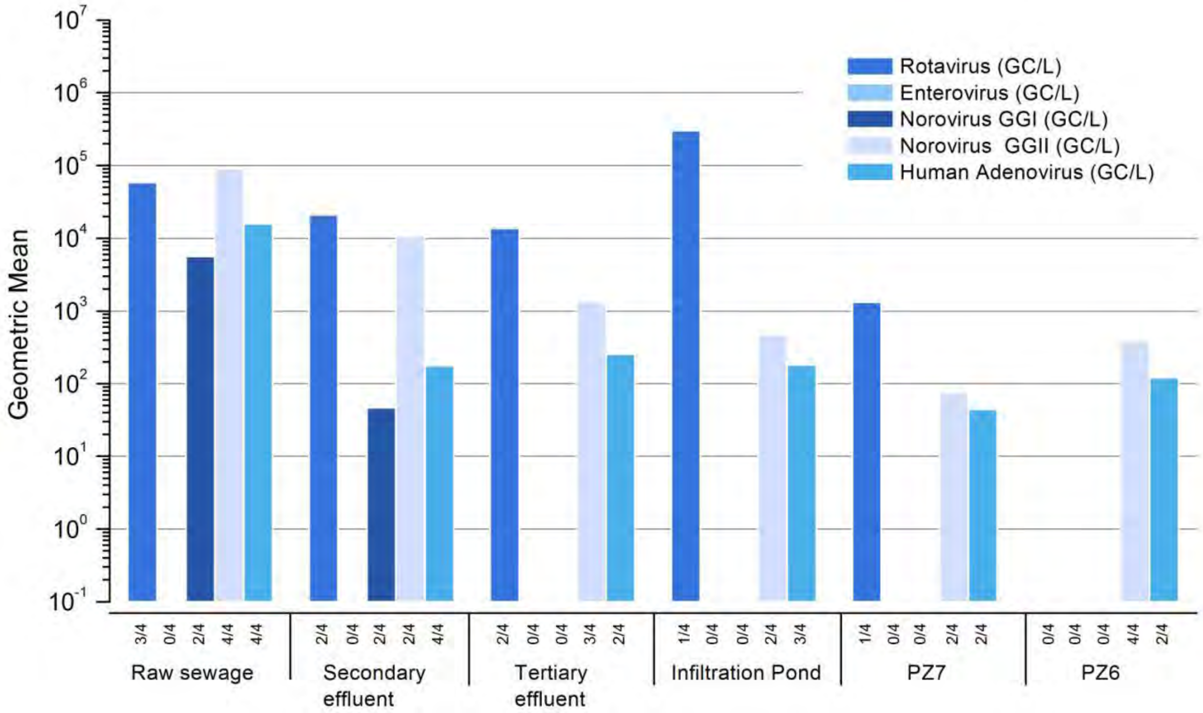
If microbial indicators were not detected, the detection limit was used to calculate the reduction, resulting in minimum $\log_{10}$ reduction indicated by the 'greater than' sign.

Table 43 Log$_{10}$ reduction based on geometric means for each step and microbial parameter (Total coliforms, E.coli, E. faecalis, Clostridium Perfringens, Bacteriophages)

Log_{10} reduction	Raw sewage – Secondary effluent	Secondary effluent – Tertiary effluent	Tertiary effluent – Infiltration Pond	Infiltration Pond – PZ7	PZ7 – PZ6
Total coliforms	2.84	2.94	-0.13	>0.84	-
E.coli	2.77	3.07	0.07	>0.31	-
E. faecalis	2.17	2.57	-0.85	1.06	
Clostridium Perfringens	1.95	2.83	-0.34	>0.80	-
Bacteriophages	2.21	2.49	0.45	0.13	0.05

Somatic coliphages were used as surrogates for human enteric viruses. These non-pathogenic viruses attack E. coli and have similar physical properties as human pathogenic viruses. Compared to average dimensions of bacteria, somatic coliphages are an order of magnitude smaller and considered to be more mobile during subsurface passage. Somatic coliphages were detected in pond water in the range of 10 pfu/100 mL and show decreasing concentration along the flow path from PZ7 to PZ6. After a residence time of ~55 h in the aquifer about 7 pfu/100 mL were detected in PZ6.

Geometric means of Rotavirus, Enterovirus, Norovirus GG I + GG II, and Human Adenovirus measured in raw sewage, secondary effluent, tertiary effluent, infiltration pond, PZ7, and PZ6 do not show a steady decrease along the treatment train for all virological parameters (Figure 53).



4/6 = four positive out of six samples; GC = genome copies

Figure 53 Geometric means of Rotavirus, Enterovirus, Norovirus, Human Adenovirus measured during treatment train.

Substantial removals in tertiary treatment were determined for the monitored norovirus types. Similarly, to observations with technical treatment, no clear trend can be observed during subsurface passage. On the flow path from infiltration pond to PZ7 the three detected viruses (human adenovirus, norovirus GGII and rotavirus) show a certain removal, but were detected in elevated concentrations in PZ6, resulting in negative $\log_{10}$ reductions for norovirus type II and human adenovirus. A possible reason for this observation could be that adenovirus detections in PZ7 and norovirus GG II in all groundwater samples were below limit of detection (see Table 40). Moreover, the analytical method, which is based on polymerase chain reaction (PCR) cannot differentiate between infectious and inactivated/ non-infectious virus particles. If the PCR-amplified fragment of the respective genome is present in a sample, the fragment is detected, independent of it being contained in an infectious or non-infectious virus (or being contained elsewhere in the sample). This shortcoming of PCR-based virus quantification has been reported in the literature (Lim et al. 2010). Also, the relatively low number of samples does not allow for conclusions regarding these parameters. These apparently poor results obtained for the viruses should be further assessed in more detail. This is underlined by the fact that UV disinfection has been reported in the literature as effective for the removal/ inactivation of the viral agents quantified here (Hijnen et al. 2006; Song et al. 2016), cf. also section 2.

Table 44 Log10 reduction based on geometric means for each step and virological parameter (Rotavirus, Enterovirus, NoV GGI, NoV GGII, HAdV)

Log_{10} reduction	Raw sewage – Secondary effluent	Secondary effluent – Tertiary effluent	Tertiary effluent – Infiltration Pond	Infiltration Pond – PZ7	PZ7 – PZ6
Rotavirus	0.4	0.2	-1.3	2.4	
Enterovirus	-	-	-	-	-
NoV GGI	2.1		-	-	-
NoV GGII	0.9	0.9	0.5	0.8	-0.7
HAdV	1.9	-0.2	0.1	0.6	-0.4

5.7.2.4 Organic micro-pollutants

As shown in section 5.5.5, four laboratories with different analytical methods and different sets of compounds were involved in this study. To check the consistency between the laboratories parallel samples were taken. The comparison between the involved laboratories for compounds which were detected in concentration above the given limit of quantification is shown in Figure 54.

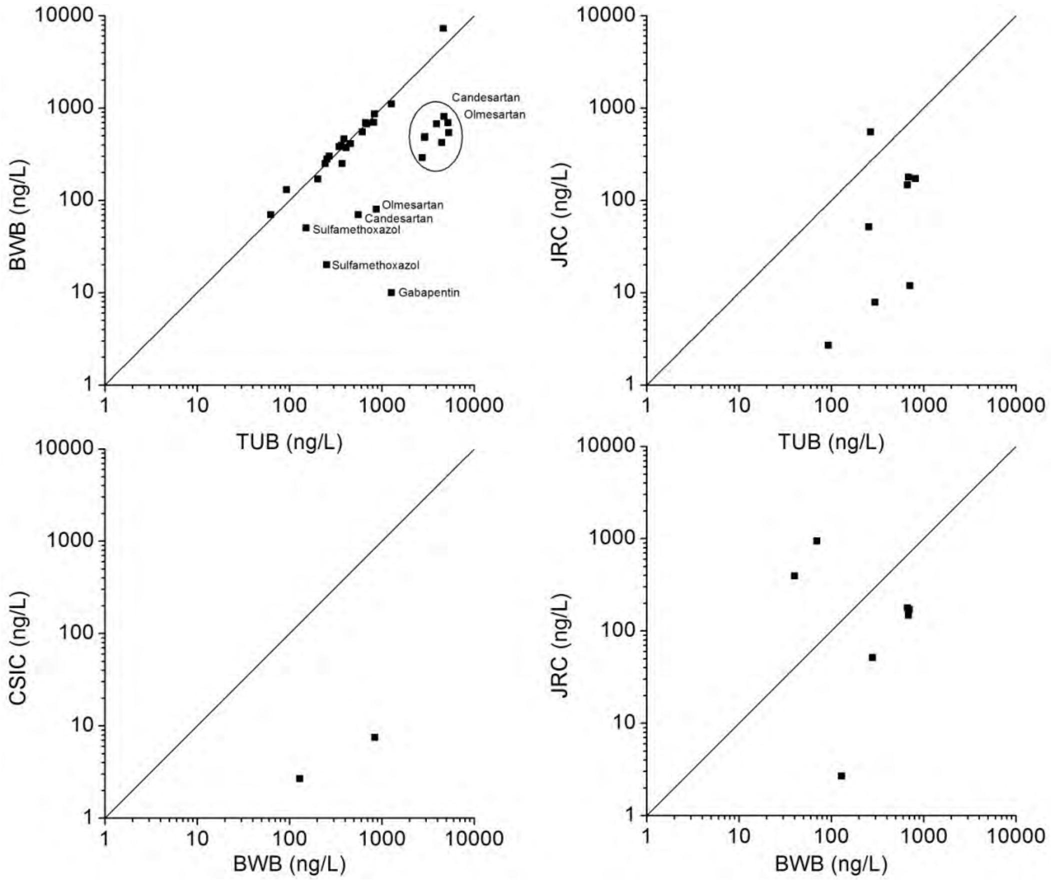


Figure 54 Comparison of parallel samples between laboratories.

BWB and TUB show a good correlation for most compounds, except for olmesartan, candesartan, sulfamethoxazol. Correlation between CSIC and BWB, JRC and TUB, JRC and BWB are poor. Hence only BWB and TUB samples were used for further evaluation.

The sampling campaigns were used to obtain trace organic compound concentrations indicative for anthropogenic impacts from infiltration on groundwater composition. Prior to the start of infiltration, samples were obtained from the WTTP effluent and native groundwater incl. drinking water well. Results from this prior-infiltration sampling are shown in section 5.7.1. Effluent concentrations were also used in risk management and reported in a separate deliverable (work package 3).

Min and max concentrations of organic trace compounds measured in tertiary effluent are shown in Figure 55. Compounds measured below LoQ are shown as LoQ/2 in the following figures.

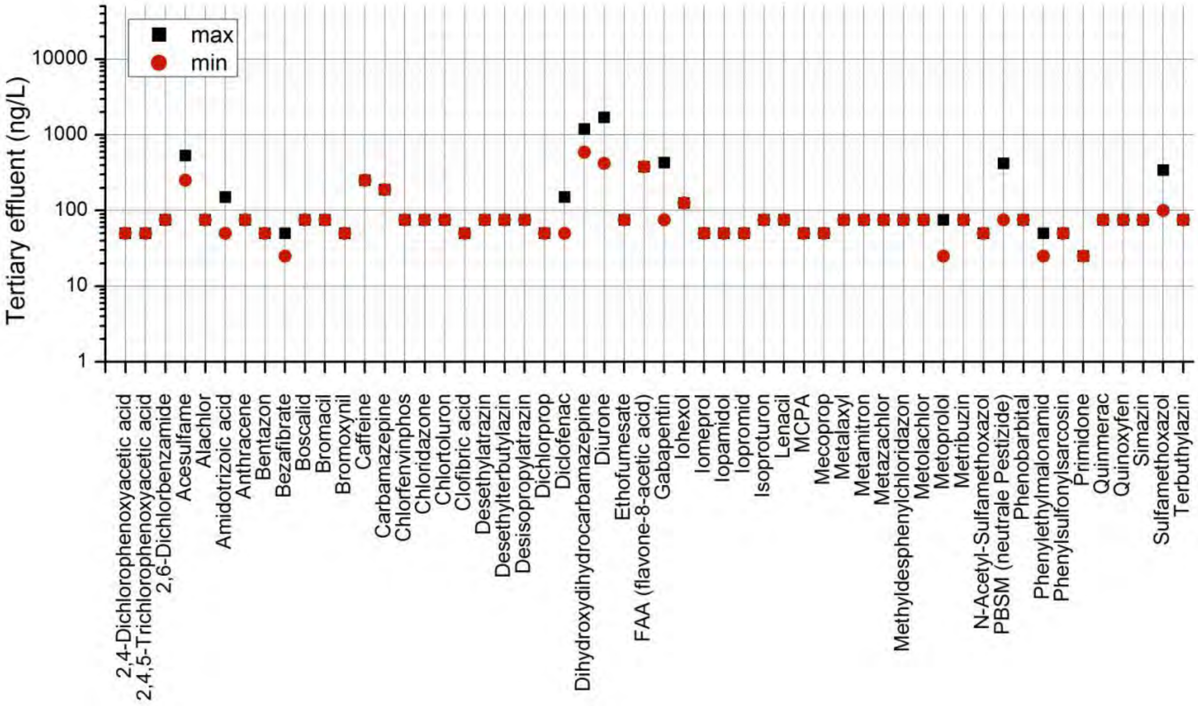


Figure 55 Min and max concentration of organic trace compounds measured in tertiary effluent (n=2-4).

In total 54 compounds were analysed in tertiary effluent. Metalaxyl, Lenacil, Alachlor, Iohexol, Phenobarbital, Quinoxyfen, 2,4,5-Trichlorophenoxyacetic acid, Chlorfenvinphos, and Bromoxynil were always below LoQ. Min and max concentrations of organic trace compounds measured in the infiltration basins are shown in Figure 56.

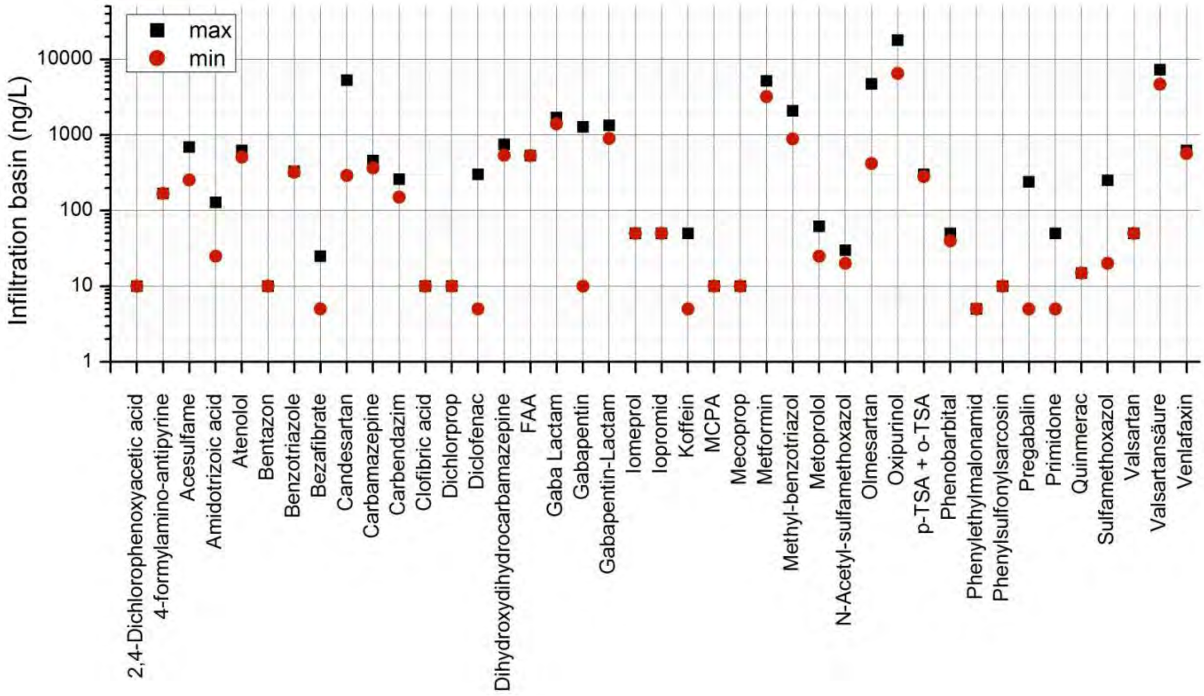


Figure 56 Min and max concentration of organic trace compounds measured in infiltration basins (n=2-4).

In total 41 compounds were analysed in infiltration basins. Highest concetration were found for candesartan, melformin, olmesartan, oxipurinol, and valsartan acid.

Min and max concentrations of organic trace compounds measured in PZ6 are shown in Figure 57.

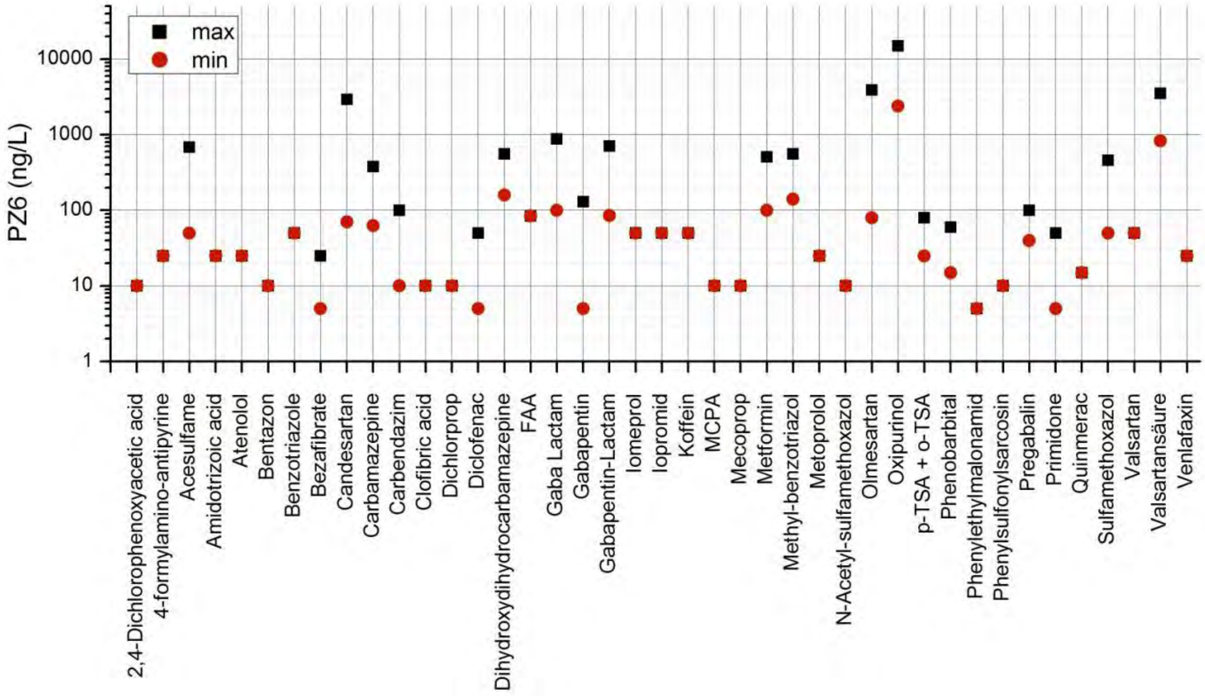


Figure 57 Min and max concentration of organic trace compounds measured in PZ6 (n = 2-4).

Iomeprol, Valsartan, Primidone, Venlafaxin, Diclofenac, Bezafibrate, Benzotriazole, Gabapentin, Atenolol, Amidotrizoic acid, Metoprolol, 4-formylamino-antipyrine, Iopromid were not detected above LoQ in PZ6. Highest concentrations were found for oxipurinol, candesartan and olmesartan and valsartan acid.

In PZ7 71 compounds have been analyzed as shown in (Figure 58). Alachlor, Lenacil, Diclofenac, Bromoxynil, Metoprolol, Metalaxyl, Phenobarbital, Iopromid, Iomeprol, Iohexol, Primidone, Chlorfenvinphos, Quinoxyfen, Bezafibrate, Atenolol, Amidotrizoic acid, Valsartan, 2,4,5-Trichlorophenoxyacetic acid and Venlafaxin were detected below LoQ.

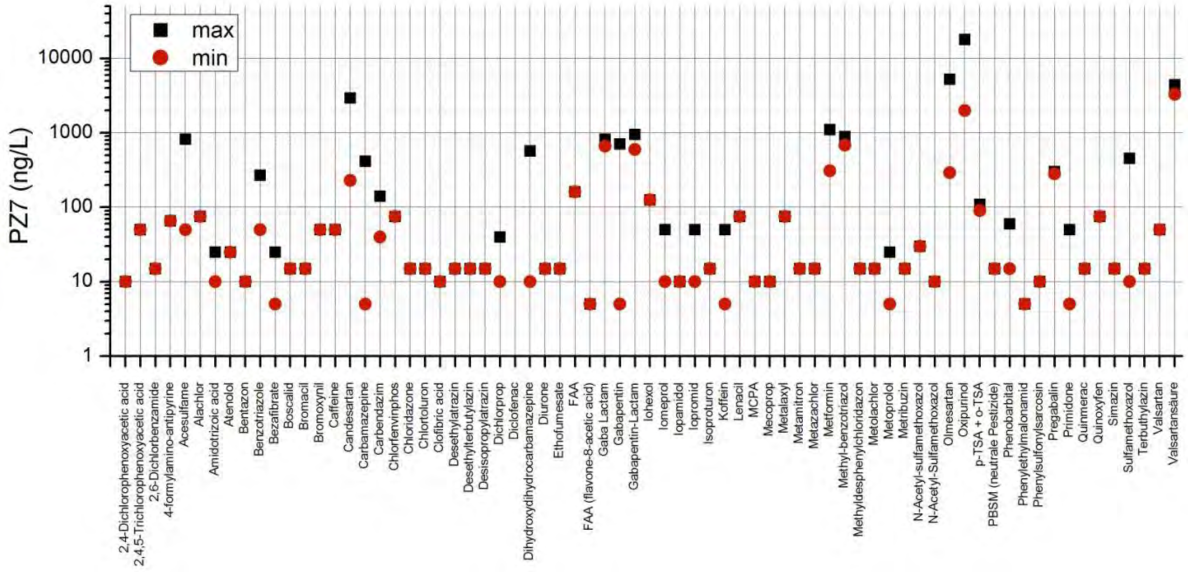


Figure 58 Min and max concentration of organic trace compounds measured in PZ7 (n= 2 - 4).

However, because of the quite different method, the high range of LoQs and LoDs and the limited availability of comparable data final evaluation is pending and monitoring is continued.

5.8 Flow and transport modelling

5.8.1 Conceptual model, objectives and scope

Key variables in the design of SAT systems are the travel time from infiltration ponds to abstraction wells and dilution of infiltrate with native groundwater. Using a flow and transport model, the migration of the plume of infiltrate in the aquifer can be simulated to analyze the sensitivity of travel times and dilution factors related to different rainfall scenarios, infiltration rates, pumping rates in abstraction wells, aquifer porosity or hydraulic conductivity. In the present work, the model has been used to analyze the influence of different rainfall scenarios.

The conceptual model implemented in the numerical model is depicted in Figure 59. Water inflow to the aquifer is infiltration from rainfall (p), lateral inflow from hill-slope run-off (f) and recharge from the infiltration basins (r_{SAT}). Outputs in the system are due to pumping in abstraction wells (Q) and discharge to the sea (d). All these variables are highly variable in time and differences in/out at a given time result in increase/decrease of aquifer storage. Dry periods with low infiltration rates and high pumping rates result in sea water intrusion i.e. negative d.

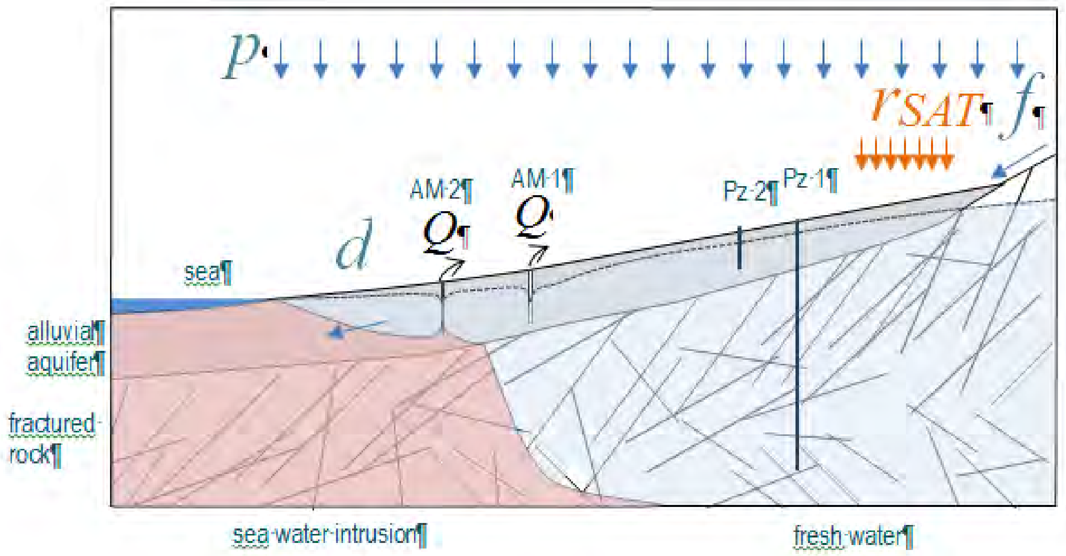


Figure 59 **Conceptual model for the SAT infiltration in El Port de la Selva**

The objective of the numerical model was to estimate travel time from the infiltration basins to the water supply wells as well as dilution rates in extracted water (percentage of infiltrate water and native water). The flow and transport model aimed to reproduce the transient behavior of the aquifer at regional scale (300 to 1000 m) assuming conservative transport of infiltrated water (non-reactive transport).

The numerical model has been updated several times during the project starting with a steady state flow model, then observations in several wells allowed to gain sufficient insight on the temporal evolution of hydraulic heads to develop a transient flow model with areal recharge and lateral inflow being functions of daily rainfall. At this stage, the basins were represented in the model by a rectangle with a constant-in-time inflow of 200 m^3/day to obtain estimates of travel time from the basins to the water supply wells. Finally, in the last model update, the basins were implemented with the real geometry and infiltration rates implemented in the model using data of the actual flow rates to each basin. At this final stage, measurements of E.C. in observation wells located near the basins were used for calibration of the transport model. Travel times and dilution factors have been updated during the project as more data and observations were incorporated in the numerical model. At the time of writing this document, the numerical model has been used to simulate different rainfall scenarios to estimate travel times from the infiltration basins to the abstraction wells used for drinking water production.

5.8.2 Rainfall and pumping rates in water supply wells

The climate in El Port de la Selva is typically Mediterranean, with heavy rainy events at the beginning of autumn and/or spring. Average rainfall is around 550 mm/year but rain events can be up to 100 mm in just a few days.

Figure 60 shows daily rainfall for the period 2010-2015 (average between two available meteorological stations located in Roses and Portbou) and available data of daily pumping rates in water supply wells for 2014 and first months in 2015. Table 45 indicates the monthly volumes extracted by water supply wells in 2014 and 2015 (AM1 and AM2). As expected, the summer season corresponds to the dry months of the year and the highest water demand. Seasonality of both rainfall events and pumping rates in water supply wells result in changes in hydraulic heads in observation wells as it is discussed in section 5.6.2.

Figure 60 Daily pumping rates for 2014-2015 (top, data available until 12/4/2015) and rainfall for the period 2010-2015 (bottom, mm/month).

Table 45 Volumes of water extracted from water supply wells AM1 and AM2 in m³ (Data provided by SOREA)

Month	AM1 + AM2 2014	AM1 + AM2 2015
January	15850	18440
February	14220	17820
March	16600	18570
April	20850	24450
May	24980	27950
June	32530	35350
July	45620	53440
August	61990	60720
September	35150	33480
October	23040	23430
November	18910	20550

Month	AM1 + AM2 2014	AM1 + AM2 2015
December	18090	18070
TOTAL	327830	352270

5.8.3 Model domain, discretization and boundary conditions

We developed a 2D finite element numerical model to simulate groundwater flow and conservative transport through the alluvial formation using FEFLOW (Diersch 2014). The geometry of the alluvial formation was implemented with uniform hydraulic conductivity and variable thickness, resulting in aquifer transmissivity variable in space but constant in time (i.e. transmissivity is not affected by changes in hydraulic head). Boundary conditions include constant head at the shore line (Dirichlet boundary condition), lateral inflow from hill slope run-off (Neumann boundary condition), pumping rates in water supply wells, areal rainfall recharge and infiltration in basins.

Details on model domain, discretization and calibration of the flow model for steady-state and transient conditions are given in Bayer-Raich & Schwarzmüller (2015) and Bayer-Raich et al. (2015). Discretization and boundary conditions are shown in Figure 61 and Figure 62.

Figure 61 Finite Element discretization and model domain.

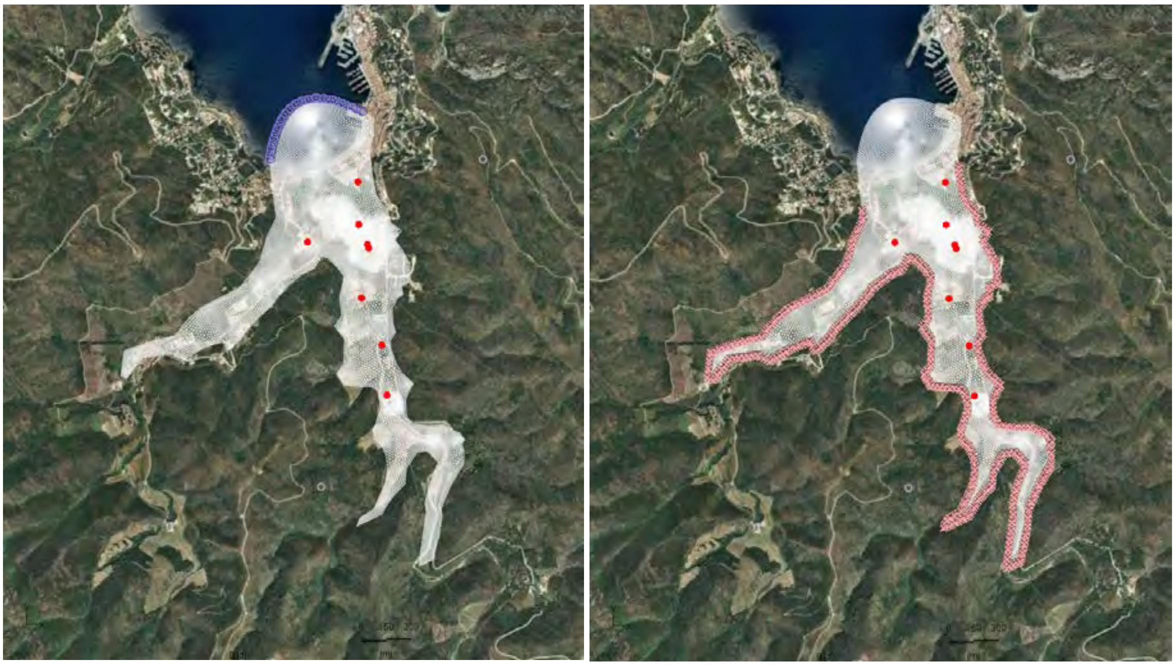


Figure 62 Boundary conditions: left: fixed heads with Dirichlet boundary condition (Sea level); right: hill slope inflow (Neumann boundary condition). Red dots: network of monitoring wells (Pz3,4,6,7 not shown)

The recharge due to rainfall has been implemented in the model through both areal recharge and flux boundary condition. We separate the rainfall in the whole basin in two terms:

1) Rainfall directly on the extension of the alluvial aquifer is implemented as areal recharge
5) Rainfall within the basin (outside of the alluvial aquifer) is considered as surface runoff along the hill slope that ends up infiltrating near the boundaries of the aquifer and is implemented as specified flux.

Both 1) and 2) are temporal series computed from rainfall data, area of the whole basin and area occupied by the alluvial formation as depicted in Figure 63.

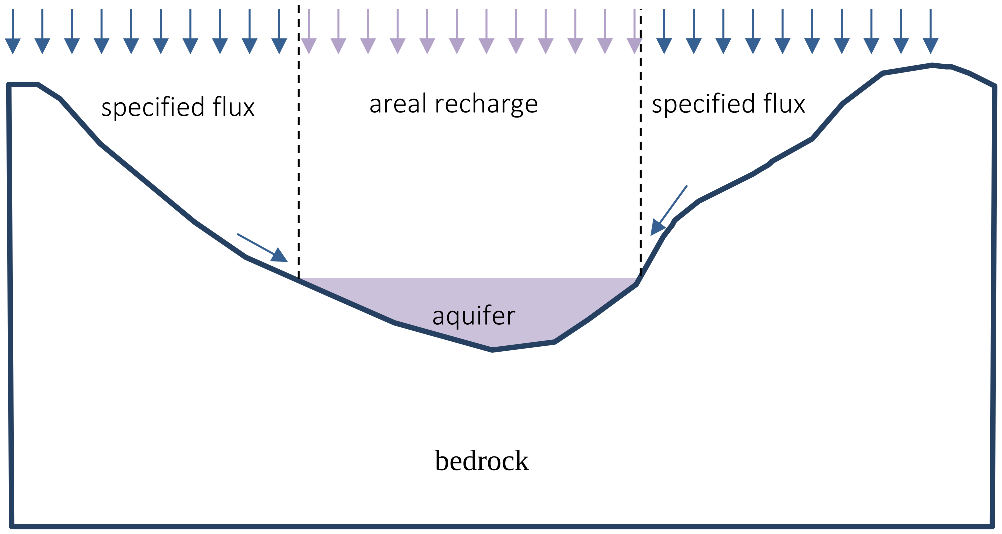


Figure 63 Implementation of inflows in the model from rainfall data

5.8.4 Calibration and global mass balance for steady-state conditions

At a first stage, the flow model was calibrated for steady-state conditions using average pumping rates in water supply wells, average head in available monitoring wells and average annual rainfall.

Comparison of computed and measured hydraulic heads and global mass balance within the steady-state model are shown in Figure 64 and Table 46.

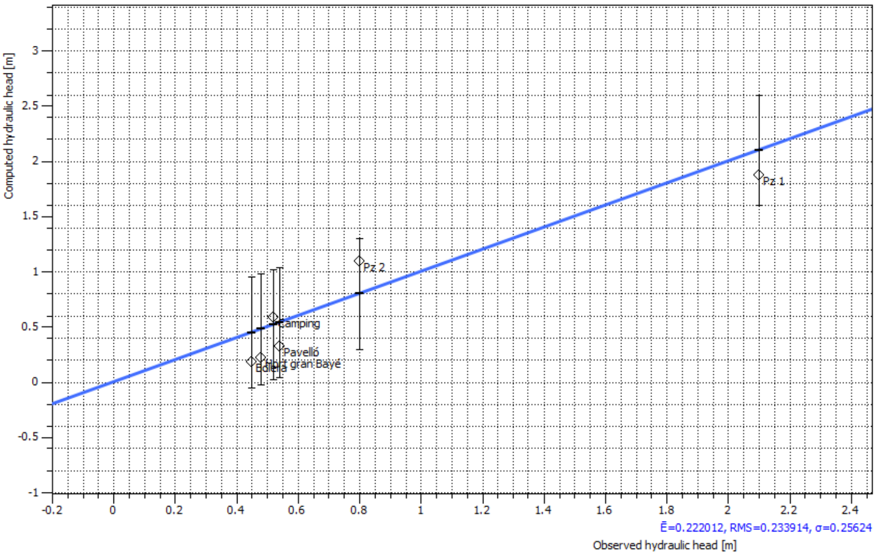


Figure 64 Calibration of measured vs computed data for steady-state conditions (Bayer-Raich & Schwarzmüller, 2015).

Table 46 Mass balance in steady-state model (Bayer-Raich & Schwarzmüller, 2015).

In	1/6/2012-1/6/2013
Infiltration from basins: 200 m^3/d	Pumping wells: 1060 m^3/d
Areal recharge: 408.68 m^3/d	Discharge to the sea: 813.25 m^3/d
Lateral Inflow from hill slope: 1264.3 m^3/d	

The total amount of water inflow from rainfall within the whole basin outside the alluvial formation (implemented as specified flux) is larger (1264 m3/day) than direct areal recharge (408 m3/day) because the area of the alluvial aquifer is smaller than the area of the whole basin. The proportion between these water inflows is difficult to quantify based on field measurements and is the result of the model calibration for steady-state conditions (Figure 64).

5.8.5 Calibration for transient conditions

When sufficient data on temporal evolution of hydraulic heads was available, the flow model was further calibrated for transient conditions. Measured and computed heads for the transient flow model for the period 1/1/2014 to 1/6/2016 are shown in Figure 65 to Figure 68.

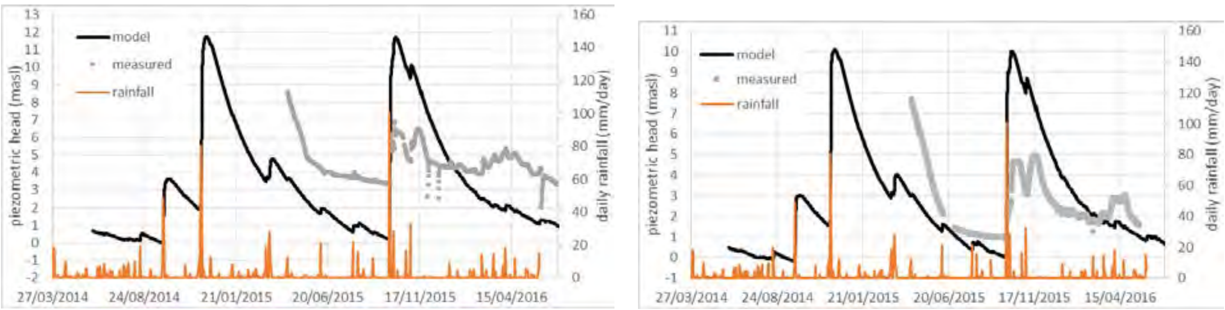


Figure 65 Model vs measured head in observation well Pz3 (left) and Pz4-Pz5 (right)

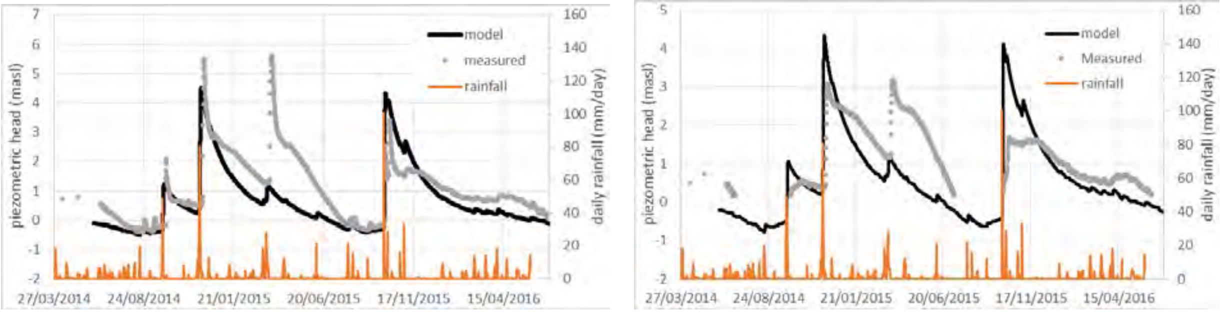


Figure 66 Model vs measured head in observation well Pavelló (left) and HGB (right, during summers, water level is not measured as falls below diver location).

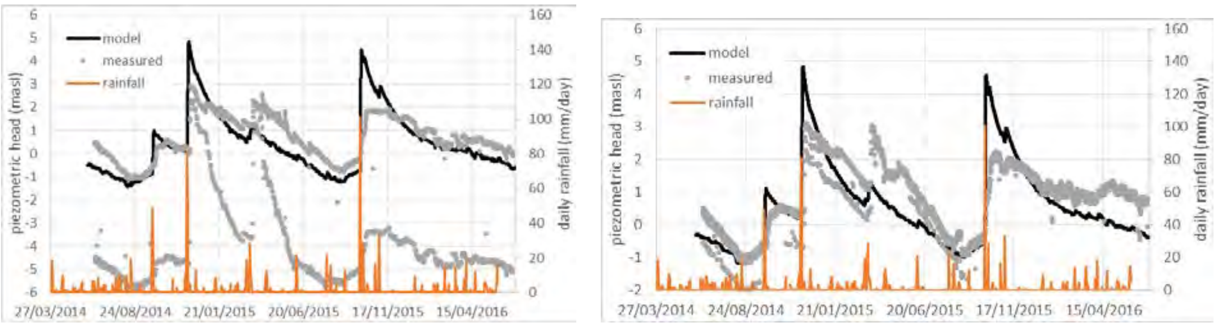


Figure 67 Model vs measured head in pumping well AM1 (left) and AM2 (right). Pumping rate in model uses daily values while in reality well operates during just a few hours a day.

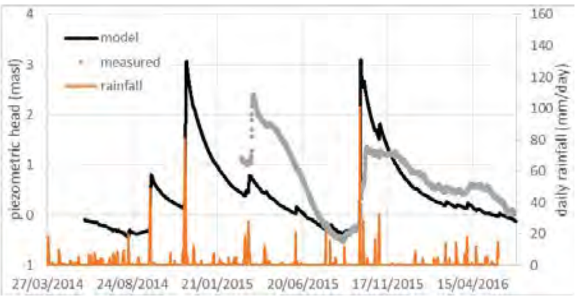


Figure 68 Model vs measured head in observation well Bolera

The transient flow model reproduces the fast response to rainfall events in most cases with reasonable accuracy.

After calibration of the flow model, the only data available (with temporal evolution of solute concentration) to calibrate the transport model were the E.C. measurements in Pz6, Pz7 and Pz4. The data measured in Pz6 and Pz7 were very valuable because they gave insight on the transport parameters of the aquifer. However, the travel times and distances were too small to be reproduced in detail by this numerical model. The numerical model at present stage partially reproduces the E.C. measured in Pz6 and PZ7 but fails to reproduce the measured E.C. in Pz4. To simulate the transport near the basins, it would be desirable to develop a 3D model at a local scale for the aquifer near the basins (100 to 200 m) including the unsaturated zone. This would require a better understanding of the processes ongoing but the efforts have been put in developing the regional model more than focusing at the local scale.

Modelling is still ongoing at the time of writing this report because of the large amount of data collected after summer 2016. At present stage the model partially reproduces peaks of E.C. measured in Pz6 and Pz7 (Figure 69) using a porosity of 20%.

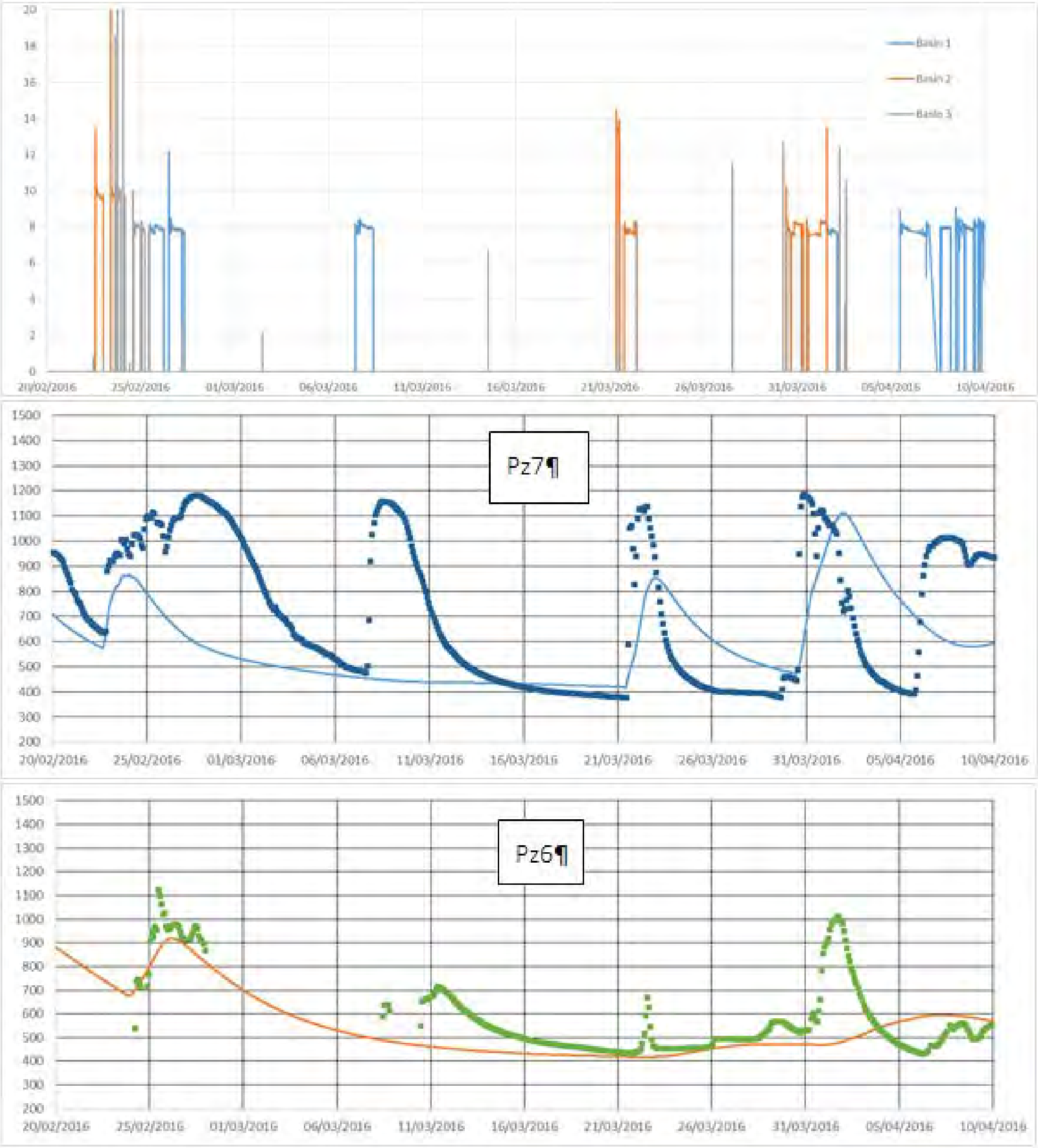


Figure 69 **Top: Infiltration rates in Basins. Middle: E.C. in Pz7. Bottom: E.C. in Pz6. (dots: measured E.C. with diver, line: simulated E.C.).**

The simulation does not reproduce the measured responses to infiltration in Basin 1 but gives acceptable results in Pz6 when infiltration takes place in Basin 3 and in Pz7 when infiltration is done in Basin 2. The explanation for this is that concentration at a given location is very sensitive to plume direction at the local scale and using a regional model does not allow for enough resolution at this small scale.

In view of the results of model calibration for flow and transport, it must be accepted that the numerical estimates obtained with this model will have a considerable degree of uncertainty. To obtain a more satisfactory fit of the measured data would require increasing the complexity of the numerical model, for instance, developing a full 3D model of the alluvial formation accounting for the processes in the unsaturated zone with more detailed information about the spatial variability of model parameters.

5.8.6 Flow and transport modelling for different rainfall scenarios

Using the calibrated transport model, the transport of infiltrated water in the basins (started on November 19th 2015 and operated until July 25th 2016) was simulated for a 3-year period from June 1st 2015 to June 1st 2018. Daily pumping rates in wells AM1 and AM2 from the year 2014 were considered for all 3 years. This simulation includes 1 year in the past (1/6/2015 to 1/6/2016) and 2 years in the future (1/6/2016 to 1/6/2018).

Two data sets were available for daily rainfall, the first are measurements taken in El Port de la Selva (station located in “club Nautic”) for the period 1/1/2013 to 1/6/2016 and the second is the average between two stations located in Portbou and Roses (“Portbou-Roses”), available for the period 1/1/2007 to present. The measured rainfall data from El Port de la Selva (station “club Nautic”) was used for the first year of the simulation (the period 1/6/2015 to 1/6/2016) and different scenarios were considered for the future rainfall in the second year (1/6/2016-1/6/2017) and third year (1/6/2017-1/6/2018) of the simulation using the available data sets. Average annual rainfall is 550 mm and annual rainfall for each year of both data sets is shown in Table 47. The table also indicates the probability of rainfall being above this value (for wet conditions above 550 mm) and probability of rainfall being below this value (for dry years with less than 550 mm). For example, probability <7.5 for annual rain 334 mm (*dry year*) indicates that 82.5% of the years will have a rain *above* 334 mm and a probability <9% for annual rain 750 mm (*wet year*) means that 91% of the years will have annual rainfall *below* 750 mm.

Table 47 Annual rainfall data sets available

Data set	1/6/2012-1/6/2013	1/6/2013-1/6/2014	1/6/2014-1/6/2015	1/6/2015-1/6/2016
“Port de la Selva”	Not available	334 mm (very dry) (Probability<7.5%)	750 mm (very wet) (Probability<9 %)	460 mm (average dry) (Probability<27%)
“Portbou-Roses”	582 mm (average wet) (probability<42%)	348 mm (very dry) (Probability<8.9%)	674 mm (average wet) (Probability<20.4%)	494 mm (average dry) (Probability<34%)

The different years in the data sets of measured rainfall are used to define 5 rainfall scenarios; in each scenario a single data set was implemented for the 2nd and 3rd years of the simulation as indicated in Table 48.

Scenario 1 is an example of extremely wet conditions in the 2nd year of the simulation (750 mm with 144 mm measured in a single day on 29/11/2014) while scenario 3 represents the other extreme: extremely dry conditions (no rain from 1/6/2016 to 1/6/2018). The other three scenarios represent more common situations with annual rainfall between 334 mm (average dry) and 674 mm (average wet).

Table 48 **Rainfall scenarios implemented for the 3-year simulations**

Scenario	1^{st} year (1/6/2015 to 1/6/2016)	2^{nd} year (1/6/2016 to 1/6/2017)	3^{rd} year (1/6/2017 to 1/6/2018)
1	460 mm (average dry) ("Port de la Selva" 1/6/2015 to 1/6/2016)	750 mm (very wet) ("Port de la Selva" 1/6/2014 to 1/6/2015)	460 mm (average dry) ("Port de la Selva" 1/6/2015 to 1/6/2015)
2		674 mm (average wet) ("Portbou-Roses" 1/6/2014 to 1/6/2015)	494 mm (average dry) ("Portbou-Roses" 1/6/2015 to 1/6/2016)
3		No rain (very dry)	
4		334 mm (very dry) ("Port de la Selva" 1/6/2013 to 1/6/2014)	750 mm (very wet) ("Port de la Selva" 1/6/2014 to 1/6/2015)
5		582 mm (average wet) ("Portbou-Roses" 1/6/2012 to 1/6/2013)	348 mm (very dry) ("Portbou-Roses" 1/6/2013 to 1/6/2014)

Breakthrough curves of reclaimed water in water supply well AM1 for the five scenarios are shown in Figure 70. Dilution of reclaimed water in native water is above 86% in all cases (concentration of reclaimed water remains below 14%) and travel times are 350 days for scenario 1, 392 days for scenario 2, >925 days for scenario 3, 441 days for scenario 4 and 447 days for scenario 5 (defining the travel time as the time to reach 5% of reclaimed water in AM1).

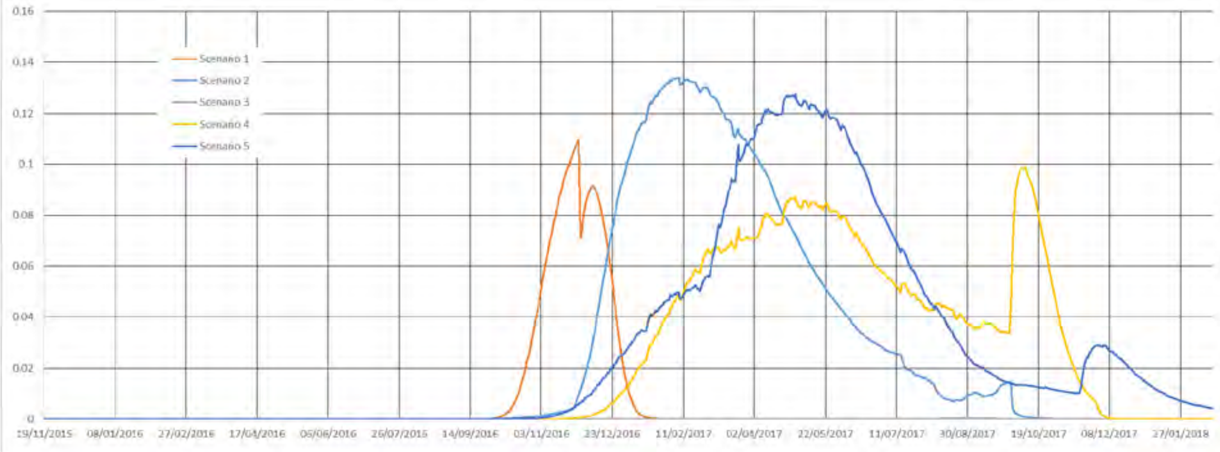


Figure 70 **Breakthrough curves of reclaimed water in well AM1 for all considered scenarios.**
Note that in scenario 3 reclaimed water has not reached the well. Origin of time axis 19/11/2015 (start of infiltration) and vertical grid lines with 50-day separation.

The travel times (defined here as the arrival of 5% of reclaimed water) for the scenarios considered range from a minimum of 350 days (for scenario 1 with very wet conditions in the second year with 750 mm rainfall) to more than 925 days (for extremely dry conditions, assuming that there is no rain from 1/6/2016 to 1/6/2018). For the three scenarios with rainfall closer to average, travel time equals 392 days, 441 days and 447 days (Table 49).

Table 49 Travel time (arrival of 5% of reclaimed water in abstraction well AM1) for the different rainfall scenarios

Scenario	Travel time [days]	Comments
1	350	High annual rainfall (750 mm) during the second year of the simulation (probability <9% for rainfall above 750 mm)
2	392	High annual rainfall (674 mm) during the second year of the simulation (probability <20.4% for rainfall above 674 mm)
3	>925	Test for analyses of water velocities for dry conditions (no rainfall for 2 years)
4	441	Low annual rainfall (334 mm) during the second year of the simulation (probability <7.5% for rainfall below 334 mm)
5	447	Annual rainfall during the second year of the simulation 582 mm (close to average 550 mm)

The results of the different scenarios indicate that for wet conditions, travel time is shorter and dilution rates are larger and for dry conditions travel times are larger but dilution rates are lower. In all cases, dilution rates are below 14% (more than 86% native water). To obtain a probability density function for the travel time, it would be necessary to run a large number of scenarios with historical rainfall distribution for the 3 years of the simulation. At the time of writing this report the efforts have been put in considering two worst case scenarios (1 for high annual rainfall and 3 for dry conditions) and three scenarios with average rainfall closer to average conditions. Since probability of scenarios 1 and 3 is below 20%, it can be estimated that the expected travel time from a weighted average from the five scenarios with weight 0.2 for scenarios 1 and 3 is 470 days.

5.8.7 Conclusions of travel times obtained from the numerical model

The numerical model developed in El Port de la Selva within the Demoware project was calibrated with hydrological measurements taken from summer 2014 to summer 2016 and Electrical Conductivity data from automatic measurements by divers located in observation wells near the basins.

As expected in a Mediterranean environment, the groundwater flow is controlled by short and intense rainfall events. The aquifer response to rainfall events is very fast, hydraulic heads can rise several meters in just a few days after strong rainfall events > 100 mm. The numerical model reproduces the measured hydraulic heads reasonably well in most observation points and is also capable of reproducing the aquifer response to pumping in the water supply wells.

To simulate the migration of the reclaimed water infiltrated in the basins, a transport model was developed simulating three years from June 1st 2014 to June 1st 2018 and the infiltration rates in the basins from November 19th 2015 to July 25th 2016. Rainfall measured in El Port de la Selva for the period 1/6/2015 to 1/6/2016 was implemented in the numerical model (with a total rainfall of 485 mm) and different scenarios for the (future) period June2016-June2018 with annual rainfall ranging from 750 mm to 0 mm were assumed.

In the most conservative scenario (rainfall 750 mm in the period 1/6/2016 to 1/6/2017), the model predicts arrival of reclaimed water at concentration of 5% after 350 days (with probability of such wet conditions being below 20%). The expected travel time estimated from the five scenarios is 470 days. In all scenarios, concentration of reclaimed water in water supply wells is below 14% (water extracted in water supply wells contains more than 86% of aquifer native water and less than 14% of reclaimed water).

5.9 Capital and operational costs

Capital and operational cost stated here consider the design, implementation and start of operation of a new SAT system as done for El Port de la Selva within DEMOWARE. Starting point was an existing dual-pipe system within the village comprising a hill deposit storing treated wastewater from the local Waste Water Treatment Plant (WWTP).

This section summarizes the capital and operating cost and contains the efforts required to improve pre-treatment at the WWTP to meet the regulatory limit values, as well as the installation of ponds, their connection to the uphill deposit and the following operation period incl. monitoring efforts. Cost values and efforts were provided by the operator for the life cycle assessment carried out within WP3 or were taken from actual costs that occurred within the DEMOWARE project as stated in the description of work (project proposal).

5.9.1 Pre-treatment

Pre-treatment includes all actions needed to ensure that the WWTP effluent quality complies with the Spanish regulations for SAT with treated municipal wastewater and self-defined thresholds. For El Port de la Selva, it included

- investments in online monitoring devices to automate filling of the hill deposit tank,
- additional effort for coagulation at the activated sludge state,
- the addition of a dual-media filter with online monitoring and automatization,
- investments for a Granulated Activated Carbon (GAC) filter

In total about 30,000 € were invested for upgrading online monitoring and automatization at the WWTP (Table 51). Consulting costs as stated in Table 51 include consultancy for the the whole WWTP upgrade, estimated by the approximate work load invested by KWB. The GAC filter would cost about 28,632 €, including valves and control system for backwashing the filter. The GAC holds approximately 1000 kg of activated carbon.

Table 50 Capital cost at the WWTP for upgrade and consulting.

Capital costs (€)	Machinery and electrical		Consulting	
Activated Sludge	25,854 €	incl. probes for Ammonia 15,512 €; Ortophosphate 10,342 €	30,000 €	KWB (estimated)
Dual Media	4,200 €	incl. 3,000 € Frequency converter, 500 € Turbidity probe, 700 € EC probe		
GAC filter	28,632 €	Installation at the WWTP, valves and all devices for automation of the filter performance, including backwashing		

Operating costs include electricity consumption and demand in chemicals per treated m3 of effluent. Most relevant for the operation of the SAT in El Port de la Selva are the costs for FeCl3 coagulant, which is applied in winter only during the operation of the SAT. In summer time, the effluent is chlorinated and discharged into the river. Chlorinated water is not used for the SAT in order to avoid issues with disinfection by-products.

The operation of the granulated activated carbon requires about 970 kg of GAC in total with an estimated lifetime expectancy of 30 years assuming regular maintenance and regeneration. Recommended regeneration interval is 95 days, based on 240 d/year of operation (wintertime only). Electricity consumption for the treatment train as applied in the winter months for the operation of the SAT sums up to 0.45 kWh per treated m3 of effluent (Table 51).

Table 51 Energy consumption and demand in chemicals for the operation of the WWTP.

Operating costs	Electricity	$FeCl_3$ (40%) w water	NaOCl (15 %) w water	GAC	regeneration intervall GAC
	kWh/m³	g/m³	g/m³	kg	d
Activated Sludge	0.02	94 (13 ppm Fe) only in winter			
Dual media	0.36 (per m³ filter influent water)				
GAC	0.03 (5 m lifting & backwash)			970	95
UV	0.04 (assumed)				
Chlorination			113 (8 ppm Cl)		

5.9.2 Infrastructure and infiltration

Cost of infiltration in the basins include for the El Port de la Selva case study:

- pipeline construction to connect the infiltration site to the (pre-existing) uphill deposit
- consultancy for design, planning, tender procedure and implementation
- construction of infiltration basins
- monitoring facilities and equipment and automatization of infiltration start & stop (and implementation in pre-existing SCADA, see section 5)
- cost of external laboratories for qualitative monitoring (regular sampling)
- electricity consumption for pumping to the uphill deposit

In total about 270,000 € were invested in pipeline and recharge facilities construction. Major cost items were the construction of the infiltrations ponds and the pipeline with almost 150,000 € and the technical sand for the infiltration basins with roughly 24,000 €. About 14,500 € were invested in monitoring devices such as level loggers and sampling equipment (submersible pump, multiparameter kit etc.). About 45,000 € were spent for planning and design (including modelling efforts), as well as for consultancy and supervision during the initial infiltration phase. Land was purchased for the pipeline and the infiltration basins for 6 €/m^2. Costs for the drilling, development and assembly of two groundwater monitoring wells was about 6,000 € (Table 52).

Table 52 Capital cost for infrastructure and recharge facilities

Capital costs (€)	Pipeline		Recharge facilities	
Land	2,092 €	6 €/m2	15,144 €	6 €/m2, 2,524 m2 of land
Consulting & engineering services			45,000 €	Groundwater modeling, supervision of regular monitoring
Machinery and electrical			14,500 €	equipment for SAT, e.g. pumps, field kits, etc. 5.500 Solinst EC logger, 6,700 € GeoSub submersible pump, 1,700 € Hach HQ40 multimeter, plus flow through cell etc.
Constructions	146,453 €	incl. excavation and pipeline (800 m length) + infiltration basins (~ 500 m2 infiltration area)	30,000 €	incl. technical sand: 24,315 € (420 t), monitoring wells: 6,000 € (in total ca. 22 drilled meter + well assembly)
Regulatory and operational testing			16,550 €	incl. 16,000 €: feasibilty study, sampling before infiltration starts, 550 €: Levelling of 20 points (observation wells, wells)

Labour and electricity consumption for the recharge facilities are shown in Table 53. Electricity consumption increases by 0.69 kWh per m³ by lifting the water about 116 m to the deposit tank. Electricity consumption for the operation of the WWTP (see Table 51) and the lifting to the deposit tank sums up to 1.14 kWh/m^3. Taking the electricity price for households in Spain of 0.237 €/kWh (http://ec.europa.eu), total electricity costs including operation of activated sludge, dual media filter, GAC, UV and lifting to the deposit tank would cost 0.27 €/m^3.

Table 53 Labour and electricity consumption for the operation of the recharge site.

Operating costs	Labour	Electricity
	h/y	kWh/m³
Pipeline		0.69
SAT	480 (10 campaigns, 2 persons, 2 days field work and 1 pre-/post work each campaign)	

About 480h were spent for sampling and monitoring activities by the operator during the initial infiltration period. Extensive sampling activities, as described in the previous chapters, caused laboratory cost of ~25.000 € (700 € per sample without trace organics, for trace organics prices vary between 350 and 650 € depending on extent and laboratory), having some of the parameters covered by pre-existing routine monitoring at WTTP level (Table 54).

Table 54 Operating cost for monitoring of different parameters per sample.

Monitoring cost, €	Unit	Cost	Description
Basic wastewater parameters	€/sample	30 €	routine measurements (incl. Cl-, NO2, NO3, PO4, NH4+, NTK, pH, Turbidity, UVA254, Conductivity, Oxygen, Temperature, Redox, Suspended solids)
Groundwater quality	€/sample	330 €	incl. main ions, stable water isotopes, metals, DOC, UV254
Microbiology I (Viruses)	€/sample	300 €	incl. qPCR analysis of Rotavirus, Enterovirus, Norovirus GG I + GG II, and Human Adenovirus
Microbiology II (Bacteria, Phages)	€/sample	40 €	incl. total coliforms, E.coli, E.faecalis, Clostridium perfringens and Bacteriophages
Trace organic monitoring	€/sample	≥ 350 €	trace organic prices depend on parameters set, laboratory & applied methods and range between 350 and 650,- €

The resulting CAPEX and OPEX are shown in Table 55. The per m^3 costs are calculated assuming 48,000 m^3 of infiltrated water volume. Total capital costs are reported as annual costs taking into account the linear depreciation of investment costs over time with a cost depreciation factor based on the interest rate and lifetime expectancy according to:

$$Capital\ recovery\ factor = \frac{i\ \times (1+i)^n}{(1+i)^n - 1}$$

Lifetime expectancy (n) of 30 years is assumed and an interest rate (i) of 2%. Capital costs are calculated according to:

$$Total\ capital\ costs = \sum investment\ costs\ \times CRF$$

And the resulting annual costs are calculated by:

$$Annual\ costs = total\ operating\ costs + total\ capital\ costs.$$

Infrastructure & consulting and machinery & electrical equipment investment for pipeline and infiltration ponds are costs items as shown in Table 52. Infrastructure & consulting and machinery & electrical for the WWTP upgrade are shown in Table 50. Electricity OPEX is the sum of operating costs for the activated sludge, dual media and GAC operation multiplied by 0.237 €/kWh (http://ec.europa.eu). Personnel OPEX are calculated based on monitoring efforts during pilot phase, assuming 480 h of workload and 50 €/h labour cost. Monitoring OPEX are calculated assuming 1050 €/sample (incl. basic wastewater parameters, groundwater quality, microbiology I, microbiology II and trace organic monitoring) of analytical costs multiplied by 20 samples.

Table 55 CAPEX and OPEX for WWTP upgrade, pipeline and infiltration ponds

	Cost item	Unit	WWTP Upgrade	Pipeline + Infiltration ponds
Investment	Infrastructure + consulting	€	30,000	255,239
	Machinery + Electrical eqp.	€	58,686	14,500
	Total CAPEX	€/a	4,406	12,044

	Cost item	Unit	WWTP Upgrade	Pipeline + Infiltration ponds
	Specific CAPEX	€/m³	0.08	0.25
OPEX	Electricity	€/m³	0.10	0.16
	Personnel	€/m³	-	1
	Monitoring	€/m³	-	0.44
	Specific OPEX	€/m³	0.10	1.10
Annual cost	Specific CAPEX + OPEX	€/m³	0.18	1.35

5.9.3 Discussion and conclusions

Assuming a life-time expectancy of approx. 30 years, the depreciated specific CAPEX for infiltration basins and associated infrastructure (pipeline, monitoring wells, control devices) is 0.25 €/m^3. Specific OPEX sum up to 1.60 €/ m^3, with 1 €/m^3 for personnel costs alone and monitoring costs of 0.44 €/m^3. These cost estimations are based on the rather intensive monitoring carried out during first year of implementation. Costs for monitoring during regular operation can be substantially reduced by focusing on less but more meaningful parameters. Monitoring requirements for SAT systems for potable re-use are under discussion.

Khan et al. (2008) reports capital costs for the construction of infiltration basins in Australia of 690,000 € (1 AU$ = 0.69 €) for 2.4 ha, which makes approx. 29 € per m^2 compared to 340 € per m^2 in the case of El Port de la Selva. Costs in Spain appear to be much higher, but include a layer of technical sand and the excavation costs for pipeline construction (800 m length). This example illustrates the difficulty of comparing cost between sites.

Capital and operating costs of the SAT scheme in El Port de la Selva, as stated here, are preliminary figures derived from the pilot phase of implementation. In a later phase, when system understanding has improved, some cost items (e.g. monitoring costs) will decrease leading to lower operating costs. On the other hand, after one season of operation, no experience on the development of clogging in the infiltration basins and sufficient self-cleaning by wet-dry cycles has been gained. Regular cleaning of the technical sand layer may be required in order to keep high infiltration rates (at other sites in intervals of months to years) which would increase the long-term operating costs, but is still more cost-efficient than an irreparably clogged upper soil layer.

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6 Summary and conclusions

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The main objective of the current report was to demonstrate the feasibility of SAT as a safe and low cost treatment of reclaimed water for its reuse.

In order to address this goal, two different examples of SAT were demonstrated in the DEMOWARE project: (1) Shafdan pilot plant, and (2) El-Port de la Selva, illustrating site-specific considerations and approaches, and giving examples for pretreatment requirements, SAT design and monitoring.

It is important to note that the objectives and motivations for implementing SAT at the two sites were different, and thus the pretreatment requirements and design. Table 56 summarizes the key characteristics of the two case study sites.

Table 56 Characteristics of SAT treatment in Shafdan pilot plant and El Port de la Selva sites.

		Shafdan pilot plant	El Port de la Selva
WWTP	Capacity (volume treated)	144 m^3/d	600 m^3/d
	Tertiary treatment	Flocculation-biofiltration-ozonation	Flocculation, Double-filtration, UV disinfection
SAT	Total Basins area/infiltration	Pilot "Dug well" 3.6 m diameter	Basins with 439 m^2
	Infiltrated volume	120 m^3/d	200 m^3/d (9 months per year)
	Final water use	Irrigation and incidental drinking	Indirect potable reuse
Hydrogeology	Travel distance	7.3 m and 17.6 m	950 m
	Thickness of vadose zone	12 - 13 m	2 – 6 m
	Hydraulic retention times (HRT)	Observation wells at ~22 and ~113 days travel time (estimates from a Bromide tracer test)	Water supply wells at 350 to 500 days travel time (estimated from-modelling based on different rainfall scenarios)
Capital and operational costs	CAPEX (cent euro/m3 product)	4.7	8* + 25**
	OPEX (cent euro/m3 product)	17.2	10* + 110**
	Total (CAPEX+OPEX, cent euro/m3 product)	21.9	18* + 135**

* WWTP upgrade, ** Pipeline + Infiltration ponds

6.1 Objectives, motivations & SAT pretreatment requirements and design

6.1.1 Shafdan pilot plant

Objectives and motivation: The objective at the Shafdan pilot plant was to demonstrate the feasibility of a suggested hybrid pretreatment (flocculation-biofiltration-ozonation) prior to short SAT (~22 days HRT) in obtaining high infiltration rate, while maintaining/improving product quality and minimizing Mn dissolution. The motivation to develop such a pretreatment in this site originated from the essential need of the Shafdan SAT reclamation project to find a solution for the increasing amounts of secondary effluents and surpluses. The attempt was to demonstrate a pretreatment which will enable treating the Shafdan secondary effluents at a high infiltration rate using the current SAT fields, instead of constructing new fields or using alternative technologies.

Pretreatment requirements: The targeted product quality after SAT was of unrestricted irrigation, as in the Shafdan conventional SAT, in order to target the treatment to enable supply of SAT treated effluents to the Negev desert irrigation. Another aspect that had to be taken into account was the minimization of Mn dissolution which is known to occur in certain recovery wells of the Shafdan SAT, causing operational problems and clogging of irrigation systems.

Process design: In order to achieve this goal, the pretreatment included two stages. The first stage included PACl coagulation followed by biofiltration (media filter of ~5 m/h added with H_2O_2) to reduce particles and suspended solids and prevent clogging (section 2.3.2). In addition, the biofilter was aimed at removing oxygen consuming compounds such as ammonium and nitrite (through nitrification), and some DOC (~20%), in order to prevent the development of anoxic conditions in the soil (Table 11, section 2.4). The second pretreatment stage included ozonation which supplied high amount of dissolved oxygen (~20 mg/L) to the SAT in an attempt to maintain aerobic conditions in the soil. The other benefits of ozonation is its ability to increase the biodegradability of organic matter (expressed in UVA decrease, improving infiltration rate and organic matter removal) as well as its efficient removal of pathogens and TrOCs, which contribute to the enhanced product quality (Table 7, section 2.3.3.4).

Innovation: The innovation of the pretreatment process designed at the Shafdan pilot plant was in its new approach of maintaining aerobic conditions in the SAT, compared to the common approach of SAT being regarded as anaerobic and aerobic oscillation.

6.1.2 El Port de la Selva

Objectives and motivation: In contrast to the Shafdan pilot plant, the objective at the El Port de la Selva site was to demonstrate implementation of SAT in application of water reclamation by aquifer recharge. The motivation to design such a process at this site was the local need for improved groundwater availability to the El Port de la Selva village, which relies on local groundwater as its only drinking water source. In addition, SAT was intended to prevent seasonal salinity peaks. The approach was to supplement the local groundwater sources using SAT during the winter for the high demands during summer, while counteracting seawater intrusion.

Pretreatment requirements: The SAT treatment designed in El Port de la Selva had to meet all regulatory demands for indirect potable reuse (IPR). Pretreatment requirements were given by the existing Spanish regulation setting limit values for aquifer recharge for IPR ((Spanish Presidential Ministry 2007). Concerning infiltration volumes, constraints were set by the capacity of tertiary treatment determining the size and capacity of infiltration. On the other hand, one of the targets was to design a SAT process with low maintenance requirements.

According to this objective of designing a low technology/low cost hybrid system, nutrient removal, turbidity and UV disinfection were the main issues addressed for optimization at the WWTP level within WP1.

Process design: A site approximately 950 m upstream the local drinking water wells was chosen as the recharge site for infiltration of 200 m^3/d tertiary effluents (~10% of the abstracted groundwater). It needs to be noted that site selection was not a part of the DEMOWARE project, but decided before by the WWTP operator and local municipality against the findings of a predecessor project. The SAT pretreatment included double-filtration and UV disinfection. The pre-feasibility and feasibility stages for designing the SAT scheme and according monitoring and risk mitigation (section 3.1) targeted first of all the aquifer characterization from available documents. The general approach within DEMOWARE is summarized in Table 57 with major issues highlighted in bold. These refer mostly to the sound characterization of hydrogeological and geochemical soil and aquifer parameters as these determine the recharge and recovery rates (travel time, dilution) and conditions for removal of potentially hazardous substances such as pathogens or persistent organic micro-pollutants identified during entry-level risk assessment (section 5.2). The setup of a numerical model to determine seasonal dynamics and dilution rates of recharged water was one of the main objectives within DEMOWARE and started together with the initial assessment. Assumptions made for the model concerning aquifer porosity, permeability and resulting travel time estimates were fed back to the risk assessment (carried out in WP3).

Innovation: The innovation and importance of this study was to demonstrate SAT for IPR without the need for advanced treatment such as ozoanation or reverse osmosis, relying on natural treatment capacity of the soil and aquifer only.

Table 57 Draft project plan presented at kick-off for the design and implementation of the SAT scheme at the case study site El Port de la Selva within DEMOWARE with site-related specific challenges highlighted in bold

Stage (Duration)	Pre-feasibilty (M1-3)	Feasibility (M3-6)	Implementation (M6-30)	Operation & maintenance*
Purpose	*Collection of available information; qualitative assessment*	*Collection of additional necessary information; duration depending on availability of data and complexity of setting*	*Upscaling from pilot to full scale*	*Regular operation*
Issues to be addressed	• Legislation • **Social Acceptance**** • Budget • **Existing structures (wells)** • Water quality • End-use • Volume • Climatic conditions • **Aquifer properties (spatial occurrence of Kf, thickness,** lithology, **aquifer geometry, storage, water table fluctuations, drill logs, …)**	• Assessment of **near-surface hydraulic conductivity** (0 – 3 (max 5) meters): infiltration tests (field), grain size analysis, moisture retention curves (lab) • **Assessment of Kf at various scales**: pumping tests (large to medium scale); slug and bail tests (small scale) • Water quality sampling • Hydrogeological investigations of deeper aquifer: drilling, sediment sampling,	• Design • Construction • Definition of monitoring requirements • Initial operation with close monitoring	• Monitoring • Optimization • Maintenance procedures

Stage (Duration)	Pre-feasibilty (M1-3)	Feasibility (M3-6)	Implementation (M6-30)	Operation & maintenance*
	• Environmental issues: risks to the environment: groundwater dependent ecosystems: wetlands, hydraulic connection to river and stream close	• Analytical (in any case) and modelling tools (if necessary) • Definition of monitoring program to: control mounding (if any), water quality, impact on native groundwater		

* outside DEMOWARE; ** part of risk management (WP3), thus not further addressed in the deliverable

6.2 Water quality management and monitoring

As mentioned in chapter 1, SAT uses the soil, vadose zone and groundwater as treatment and seasonal or long-term storage. For a proper management of SAT systems, and in order to obtain the desired water quality, monitoring is mainly required in three points:

1) Infiltrated water quality monitoring – according to Groundwater directive EU 2006 - Managed aquifer recharge with reclaimed water must not introduce any hazardous substances into the groundwater. Any other input of substances shall be limited so the groundwater quality does not deteriorate.
2) Vadose zone or active upper SAT zone monitoring – accrding to Groundwater directive EU 2006 - Processes in the vadose zone that result in attenuation or degradation of substances may be taken into account when considering measures to prevent or limit input into groundwater.
3) End point groundwater monitoring – according to Water Framework Directive 2000, 2008 - Artificial recharge is a possible measure to achieve 'good chemical and quantitative status' of groundwaters.

SAT around the world is being mainly used for three purposes: unrestricted irrigation, prevention of seawater intrusion and indirect potable reuse (IPR). In all cases SAT is used as seasonal reservoir and treatment system. Usually in Europe, due to Groundwater Directive regulations, UF-RO treated effluents are infiltrated to drinking water aquifers.

6.2.1 Shafdan pilot plant

The Shafdan pilot plant demonstrated an approach for a case where secondary effluents were pretreated and infiltrated to a local aquifer surrounded by secondary effluent reservoirs (not to a drinking water aquifer). In this case, the pretreated effluents are not suspected to deteriorate the local aquifer water conditions (oxic-anoxic prevailing due to secondary effluents intrusion), but on the contrary, to improve its quality and to improve the redox conditions which cause Mn dissolution, by improving mainly the biochemical reaction rate in the upper vadose zone which subsequently improves the overall water quality.

The infiltrated water complies with the EU Ground water directive since in the areas closer to the infiltration fields the groundwater quality is poorer than the infiltrated water which is more oxic, has less micropollutants and microorganisms, and more oxygen.

Thus, the area surrounding the infiltration well is gradually cleaned at a short hydraulic retention time, showing almost indirect potable reuse quality water which is free of organics, micropollutants and microorganisms.

6.2.2 El Port de la Selva

The El-Port de la Selva SAT system demonstrated the case of infiltration of tertiary effluent to a drinking water aquifer implying long hydraulic retention times, attenuation, degradation and dilution. DOC, bromide, EC and fate of pathogens and organic micro-pollutants (pesticides/ pharmaceuticals) were of special interest for determining pretreatment requirements, recharge conditions and aquifer and human health impacts.

Basin infiltration started in November 2015 and was operated until July 2016 accompanied by extensive monitoring. The monitoring network included the complete treatment chain from raw sewage to secondary and tertiary treatment, infiltration ponds, groundwater observation wells, and finally the drinking water wells. The vadose zone was not monitored at this stage. In order to continuously control the infiltration, pond inflow controls were integrated into the existing Supervisory Control and Data Acquisition (SCADA) system of the WWTP. Control points and threshold values were set in the SCADA system to meet legal (Spanish Presidential Ministry (2007)) and self-defined thresholds for infiltration. At the WWTP, monitoring devices (e.g. turbidity sensor, EC logger) were installed to control treatment and basin inflow. Among others, a self-defined threshold value of 1.500 µS/cm was established to shut down infiltration in case of exceedance. Also, disinfection was switched from chlorination to UV disinfection in order to avoid the infiltration of water potentially containing chlorination by-products.

The monitoring program covered a wide range of hydrochemical parameters from standard hydrochemistry over inorganic trace compounds to various sets of microbial parameters. Monitoring campaigns were divided into three phases: a screening phase (pre-infiltration), operational monitoring, and post-infiltration campaigns. During the screening phase the native groundwater composition (aquifer baseline) was assessed to define pre-existing conditions before SAT activities took place. Aquifer baseline was measured as being oxic, with dissolved oxygen in concentrations between 4.7 – 8.8 mg/L. Electrical conductivity in native groundwater was 400 to 500 µS/cm and (almost) no pesticides or pharmaceuticals were present apart from one single detection of Metaprolol and Phenazone (both not confirmed in subsequent samplings). An entry-level risk assessment had further shown the general water demand and principle feasibility. The next steps in project development were thus the technical upgrading of the WWTP, construction of the required infiltration infrastructures and the initial setup of a numerical groundwater model.

Hydraulic conductivities at the recharge site were determined from drill samples taken from different depths by grain size analyses, resulting in k-values distributed over three orders of magnitude ($5x10^{-5}$ to $7x10^{-3}$ m/s, cf. Appendix, Table 58). Aquifer transmissivity close to the coastal line was estimated by pumping tests (Bolera well) with 825 m^2/d and a hydraulic conductivity of 60 m/d. Drawdown measurements in the drinking water well (AM1) and surrounding observation wells led to an estimation of hydraulic conductivity of 218 m/d by applying Dupuit equation for unconfined aquifers. Despite of grain size analysis, pumping tests and small-scale infiltration tests at the recharge site on aquifer scale some uncertainties of the spatial distribution of hydraulic transmissivity and hence travel time estimates remained.

Prior to the start of infiltration, samples were obtained from the WTTP effluent and the drinking water well. The effluent concentrations were used for initial risk assessment. Risk assessment was carried out in parallel and reported separately (cf. D3.2 Show case of the environmental benefits and risk assessment of reuse schemes).

Mobilization of undesired substances due to changed redox conditions were considered unlikely as i) groundwater at the recharge area was measured to be oxic and ii) concentrations of measured redox-sensitive parameters such as Fe, Mn or As were similar to native concentrations and showed no significant increase during operation.

Travel time estimations of infiltrate during subsurface passage close to the recharge zone were based on breakthrough curves of electrical conductivity. Median travel times based on 50 % breakthrough of tracer were in the range of few days measured in observation wells close to the infiltration basins (PZ6 and PZ7).

Microbiological monitoring showed that UV disinfection was sufficient for the tertiary effluent under all operating conditions since the Spanish Royal Decree allows up to 1000 *E. coli*/100 mL and thus chlorination was deemed unnecessary. Bacteriological parameters (Total coliforms, *E. coli*, *E. faecalis*, *Clostridium perfringens*) were not detected in groundwater (PZ6) after few days of subsurface travel time, while somatic bacteriophages and two (Norovirus GG type II and Human Adenovirus) out of four detected pathogenic viruses showed a breakthrough after few days of travel time as observed in PZ6. It must be noted that concentrations of Norovirus GG type II and human Adenovirus were not detected in all samples and if so concentrations were sometimes below the limit of quantification. Microbial monitoring demonstrated the complete removal of bacteriological indicator organisms in the aquifer and gave evidence for an efficient removal of virological parameters. However, some uncertainties regarding the removal of human pathogenic viruses associated to analytical procedures remained.

The sampling campaigns were also used to obtain trace organic compound concentrations indicative for anthropogenic impacts from infiltration to the groundwater composition. As reported before trace organics remain the most important issue requiring attention.

6.3 Main results and conclusions

6.3.1 Shafdan pilot plant

It was shown that a pretreatment consisting of flocculation-biofiltration-ozonation prior to short SAT (~22 days HRT) obtained high infiltration rate (to ~4-6 m/d) and effectively improved the final water quality including removal of organic matter, microorganisms, micropollutants, and Mn^{2+} dissolution control (chapter 4).

In the first stage of the experiment, the pretreatment before short SAT included coagulation and biofiltration only (March 2014-July 2015, *Mode 1*). This mode of operation was used as a control experiment, in order to elucidate the effect of ozonation in the second stage. During this treatment, ammonia, nitrite and phosphate were completely removed in the short SAT, and all pathogens were highly reduced. In addition, besides the persistent Carbamazepine (CBZ) and Iodine-organic contrast media (IPDL), all measured trace organic compounds (TrOCs) were effectively removed after short SAT. Mn concentrations were minimized up to 52 μg/L.

In the second stage (August 2015-September 2016, *Mode 2*), the pretreatment included coagulation and biofiltration followed by ozonation (~10 mg/L ozone dose, ~1.0-1.2 mg ozone/mg DOC) prior to short SAT. When *Mode 2* was operated continuously at the last period of the experiment (June-September 2016, total of 111 days, almost 4 months), values of DOC and UVA after short SAT were reduced down to 0.8 mg/L and 2.2 1/m, respectively, while Manganese (Mn) dissolution was controlled and maintained down to 29-35 μg/L. No ammonia or nitrite were found, and good bacterial quality was obtained.

Concerning TrOCs, in the range of 56-77 days from continuous *Mode 2* startup, it was shown that all TrOCs and especially CBZ were reduced down to <50 ng/L. In the last TrOCs campaign (105 days) all TrOC values were below detection limit (10 ng/L), except for VLX which was 41 ng/L. The delay in the effect of the pretreatment on TrOCs removal (>56 days instead of ~22 days) was explained by the retardation of CBZ displacement in the upper soil layers of the pilot SAT (0-25 cm).

In-depth sampling in the short SAT observation well (OW1) after 111 days of continuous *Mode 2* startup, showed (by low DOC, UVA, and CBZ along all sampled depths) homogeneity along the overall perforated section of the short SAT well (from -14 to -26 m). This result proved that the ozonated water completely covered the area around the observation well and positively affected the quality of the groundwater.

The efficient SAT pretreatment process that was obtained in the current study at the Shafdan pilot plant, which combines high infiltration velocity, good organic matter and TrOC removal, and Mn dissolution control, produces a technology that can be applied universally even though the anoxic conditions due to excess of ammonia in secondary effluents is a specific case of Shafdan SAT.

The biofiltration is subsequently considered to be an optional step prior to ozonation, which was applied mainly to reduce ammonium residuals in the Shafdan secondary effluents. Based on the experimental and cost analysis results, it is suggested, in order to reduce the overall process costs, to examine the possibility of replacing the coagulation-flocculation-biofiltration (~5m/h added with ~27 mg/L hydrogen peroxide) with in-line coagulation-conventional media filtration (~10m/h).

The ozonation step, which was the main component in the suggested pretreatment scheme, showed improvement of biodegradability of non-biodegradable organics and subsequently high removal of organic matter in the soil, as compared to the common Bioactive Carbon filtration. In addition, it was shown that the high oxygen supply of the ozone system succeeded to control Mn dissolution. Moreover, ozonation combined with short SAT showed removal of most TrOCs under 50 ng/L and the extra retention time (around a month) was able to polish the water and remove all analyzed TrOCs, bacteria, viruses and organic matter (down to 0.7-0.8 mg/L).

As a result, a safe and relatively low cost process for even Indirect Potable Quality water (according most international standards) was obtained. The environmental benefits are mainly that besides hydrogen peroxide as chemicals (with a lesser degree of PACl coagulant) and ozone as energy consuming product, the rest of the process is completely natural and no sludge or brine is produced to be further treated.

6.3.2 El Port de la Selva

In the case of El Port de la Selva pretreatment requirements resulted from i) Spanish legislation for indirect potable reuse, ii) human health protection and iii) technical considerations for keeping maintenance efforts low. The scheme was implemented without advanced treatment such as ozonation or reverse osmosis. It was found that in order to be on the safe side, an addition of GAC filtration following media filtration would be required in the future for micropollutants and other organic residues removal. An according pilot stage is still on-going at the end of DEMOWARE project and beyond.

With the WWTP upgrades, target effluent quality values were met nearly all times. Ammonium and electric conductivity (EC) turned out to be the most critical parameters limiting basin inflow. It was found that EC values were sometimes > 4000 µS/cm in the effluent for a short period of time. The reasons for these salinity spikes measured in secondary effluent remain unclear need to be addressed in future.

The numerical model developed in El Port de la Selva was calibrated by water level measurements and electrical conductivity.

At the time being, the numerical model failed to reproduce the measured hydraulic heads reasonably well in all observation points but was capable of reproducing the localized aquifer response to pumping in the water supply wells. Monitoring results, i.e. EC, were also incorporated in model calibration, but the numerical model was not capable to finally determine travel time and dilution rate with acceptable reliability.

The given capital and operating costs of the SAT scheme in El Port de la Selva are preliminary figures derived from the pilot phase of implementation. In a later phase, when system understanding has improved, some cost items (e.g. monitoring costs) will decrease leading to lower operating costs. On the other hand, after one season of operation, no experience on the development of clogging in the infiltration basins and sufficient self-cleaning by wet-dry-cycles and/or associated cleaning costs of the technical sand layer has been gained. Regular cleaning of the technical sand layer (at other sites in intervals of months to years) would increase the long-term operating costs by an amount to be determined through a sensitivity analysis for 1 and 2 year cleaning frequencies.

Altogether, the El Port de la Selva SAT system demonstrated the case of infiltration of treated effluent coplying with the Spanish regulation for aquifer recharge for indirect potable reuse, but being not in compliance to the EU Groundwater directive which requires to infiltrate water that does not degrade the aquifer quality status. At current stage, it can not be neglected that groundwater quality deterioration occurs. Monitoring will thus be continued by the site operator to validate subsurface travel times and degradation and dilution rates from filed observations at the point of recovery.

6.4 Future recommendations for SAT pretreatment and design based on the experience gained in the two case studies sites

Although the two demonstrated case studies were quite different in their approaches and constraints, there are still main points and steps that can be recommended from both sites experience to be followed when designing SAT and SAT pretreatment:

- Define the objectives of the SAT system, the pretreatment requirements (originated from legislation, risk-based approaches and/or technical considerations, see sections 2.1 and 2.2), and thus the main issues that should be addressed in the pretreatment and the SAT design.
- Accordingly, define the best pretreatment SAT technology (see section 2.4) and the important parameters to be monitored on-line or periodically (see section 2.3).
- Set up a risk management plan including critical control points, defintion of parameters to stop operation and mitigation measures and responsibilities.
- Design a detailed monitoring plan.
- Evaluate geochemical conditions for metal release, including redox conditions prevailing in the SAT area vadose zone and aquifer.
- In case of anoxic condtions or evidence of metal dissolution, it is recommended to improve the removal of oxygen consuming compounds such as ammonium, nitrite and organic matter in order to reduce oxygen depletion. In addition, it is recommended to consider the addition of an oxidation step to the pretreatment.
- In case of applying oxidation, add monitoring of byproducts and degradation products in the final product, and conduct appropriate toxicity tests.
- Design the SAT system using pre-feasibility, feasibility, design and implementation phases (see chapter 3) and including operation and maintenance (see example in Table 57).
- Conduct at least one year (preferred a few years) of a pilot study.
- Implement on-line monitoring and frequent analysis to track product quality and oxic conditions in the SAT at all seasons.
- Design monitoring at the three main points: (1) infiltration area, (2) upper layer of the vadose zone, and (3) groundwater.
- Implement monitoring (EC, chloride/boron as conservative tracers) to verify travel times further down-stream of the recharge site.

- Include trace organic compounds in regular continued monitoring, e.g. Acesulfame and Carbamazepine (persistent, not degraded, allowing thes determination of dilution rate).
- Implement enough observation wells towards the drinking water abstraction wells.
- Implement a numerical model in which to define a baseline sceanrio with and without recharge and compare to seasonal effects, drought/ wet scenarios etc., once the model is sufficiently calibrated (ideally before finally constructing the recahrge facilities).
- Conduct periodic tracer tests in order to understand the flow directions and rates in the SAT, as a basis for hydrogeological models.
- Consider fencing and coverage to protect infiltration area from microbial deterioration (animals/ wild birds).

APPENDIX 1 (El Port de la Selva)

Table 58 Organic carbon content, hydraulic conductivity and humidity of sediment samples (analysed by CTM)

Observation well	Depth (m below ground level)	Humidity (%) loss at 105°C	Organic matter (%) > 375°C (400ºC, 16h)	Hydraulic conductivity (m/d)
PZ-6	2	3.7	0.1	608
	5	2.7	0.1	608
	7-8	11.4	0.2	na
PZ-7	2	2.6	0.1	50
	4	2.6	0.1	4.5
	7.5-8	11.0	0.2	na
	13-14	12.6	0.2	na
Pond 1	Shallow excavation	0.53		180
Pond 2	Shallow excavation	0.49		77.5
Pond 3	Shallow excavation	0.62		6

Table 59 Details of sampling campaigns

Sampling Date	Raw waste water	SE	TE	PZ3	Pond	PZ7	PZ6	PZ5	PZ4	PZ2	Camping	AM1	HGB	HPB	Bolera
19.11-20.11	1,2,4,5,7,8,12	1,2,4,7,8,12			0,1,2,3,4,6,7,8,12	0,1,2,3,4,6,12					0,1,2,3,4,6,7,8,12	0,1,2,3,4,6,12	0,1,2,3,6	0,1,2,3,4,6,12	
30.11-01.12				0	0,1,2,12	0,1,2,12	0,1,2,12		0		0	0	0	0	0,1,2,12
16.12-17.12		12	12	0	0,1,2,12	0,1,2,12	0,1,2,12	0.12			0.12	0.12	0.12	0	0,1,2,12
12.01-13.01					0,1,2,4,8,12	0,1,2,4,8,12	0,1,2,4,8,12		0,1,2,4,12		0	0	0	0	0,1,2,6
28.01					0,1,2,12	0,1,2,12	0,1,2,12		0		0	0	0		0,1,2
09.02-11.02	9,10,11	1,2,4,9,10,11,12	1,2,4,7,8,9,10,11,12		0,1,2,3,4,7,8,9,10,11,12	0,1,2,3,4,7,8,9,10,11,12	0,1,2,3,4,7,8,9,10,11,12		0,1,2,4,12		0	0	0	0	0,1,6
24.02-25.02		5	7, 8		0,1,2,6,7,8	0,1,2,6,7,8	0,1,2,6,7,8		0,1,2,6		0	0,1,2,6	0,1,2,6	0,1,2,6	0,1,2,6
06.04-07.04 / 11.04	9,10,11	7,8,9,10,11,12	7,8,9,10,11,12		0,1,2,4,7,8,C,9,10,11	0,1,2,3,4,7,8,C,9,10,11	0,1,2,3,4,5,7,8,C,9,10,11		0,1,2,3,4		0	0,1,2,7,8			0,1,2,3,4
02.06	3	3	3	C					0,1,2,3,4,7,8	0,1,2,3,4,7,8,C		0,7,8			0, 6
15.06-16.06	3,9,10,11	3,9,10,11	3,9,10,11		0,1,2,3,4,7,8,9,10,11	0,1,2,3,4,7,8,9,10,11	0,1,2,3,4,7,8,9,10,11		0,1,2,3,4	0,1,2,3,4,7,8		0			
25.07-26.07				0, 1						0, 1,					

0=IN SITU; 1=ANIONS; 2=CATIONS; 3=METALS; 4=DOC, UV254; 5=AOX; 6=ISOTOPES; 7=PRIORITY SUBST. - MARIANI BOX; 8=TRACE ORGANICS – KWB; C=TRACE ORGANICS. – CSIC; 9=BACTERIA; 10=PHAGES; 11=VIRUSES;12=BACKUP

Table 60 **Description of monitoring stations.**

Monitoring station	Construction date	Well head (masl)	Total depth (mbgl)	Filter screens (mbgl)	Monitoring purpose	Comments
Secondary effluent (SE)	-		-	-	Sampling	
Tertiary effluent (TE)	-		-	-	Sampling	
Ponds 1-3	August/ September 2015		~2	-	Sampling	
PZ3	October 2014		11	2-11	limited	Hydraulic connection to aquifer limited, sampled occasionally only
PZ7	October 2015	12.6	10	6-10	Sampling + Water level (automatic)	
PZ6	October 2015	12.067	10	6-8 and 9-10	Sampling + Water level (automatic)	
PZ5	October 2014	11.583	9	4-9	limited	Observation well often dry, sampled occasionally only
PZ4	October 2014	11.614	13	9-13	Sampling + Water level (automatic)	
PZ2	Pre-existing	7.155	11	NA	Water level (manual), sampling after June 2016	Access to well modified in June 2016, welded cap removed
Camping	Pre-existing	5.917	5.4	NA	Water level (manual)	Iron precipitation in well water, sampled occasionally by bailer only
AM1/AM2	Pre-existing	1.394/1.8	14	NA	Sampling + Water level (automatic)	
Hort Gran Bayé (HGP)	Pre-existing	4.038	4	NA	Sampling + Water level (automatic)	Mainly to assess native groundwater
Hort Petit Bayé (HPB)	Pre-existing	3.11	3.1	NA	Sampling + Water level (automatic)	Mainly to assess native groundwater
Bolera	Pre-existing	2.77	18	12-16	Sampling + Water level (automatic)	Mainly to assess seawater intrusion, more information see Eptisa (2010)
Pavelló	Pre-existing	8.167	13	NA	Water level (automatic)	Mainly to assess seawater intrusion,

Water level = manual or automatic water level measurements; sampling = manual water sampling; NA = not available

Table 61 Samples used for native groundwater calculation.

AM1
16/12/2015
06/05/2014
19/11/2015
30/11/2015
22/10/2015
AM2
22/10/2015
Camping
30/11/2015
Hort Gran Bayé
19/11/2015
22/10/2015
30/11/2015
17/12/2015
Hort Petit Bayé
19/11/2015
17/12/2015
30/11/2015
PZ-3
21/10/2015
17/12/2015
PZ-4
21/10/2015
12/01/2016
01/12/2015
PZ-5
22/10/2015
16/12/2015
PZ-6
20/11/2015
PZ-7
20/11/2015
PZ-2
13/01/2016

Table 62 Comparison of main hydrochemistry of native groundwater.

Parameter	Unit	Native groundwater concentration from (ACA (Year unknown))	Native groundwater this study median (10th – 90th percentile)
EC	µS/cm	400	417 (372 – 513)
pH	-	7.4	6.9 (6.7-7.5)
HCO3	mg/L	60	64 (57-100)
SO4	mg/L	60	32 (27-38)
Cl	mg/L	30	82 (64-91)
Na	mg/L	25	45 (36-48)
K	mg/L	4	2.2 (1.9-3.4)
Ca	mg/L	30	25 (21-33)
Mg	mg/L	9	10 (8.4-12.7)

Table 63 Calculated inorganic trace concentration in native groundwater.

Parameter	Unit	Native groundwater concentration from this study, median concentration (10th – 90th percentile)
Br	mg/l	0.2 (0.2-0.3)
NO3	mg/L	3.7 (1.7-7.1)
DOC	mg/L	0.9 (0.5-1.9)
orto-PO4	mg/l	0.9 (0.3-1.5)
F	mg/l	<0.2 (always below detection limit)
SIO2	mg/l	17 (16-20)
Al	µg/l	26 (21-43)
Cd	µg/l	<1 (always below detection limit)
Fe	µg/l	63 (36-100)
Mn	µg/l	4.0 (2.4-14.8)
Cu	µg/l	2.3 (1-16)
As	µg/l	1.0 (0.9-1.4)
B	µg/l	27 (19-32)
Ni	µg/l	1.4 (0.9-2.4)
Zn	µg/l	19.6 (3.2-86.4)
Pb	µg/l	0.7 (0.6-1.1)

Table 64 Total coliforms, E.coli, E. faecalis, Clostridium Perfringens, Bacteriophages measured in February 2016.

	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6
Raw sewage	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	3.30E+06	4.60E+06	5.10E+06	6.30E+06	5.60E+06	1.50E+06
E.coli	9.00E+05	1.30E+06	3.10E+06	4.00E+06	2.70E+06	7.00E+05
E. faecalis	2.00E+05	2.00E+05	2.70E+05	2.90E+05	1.90E+05	2.30E+05

	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6
Clostridium Perfringens	1.83E+05	1.86E+05	2.13E+05	1.80E+05	2.03E+05	1.80E+05
Bacteriophages	5.0E+05	6.8E+05	8.0E+04	1.5E+06	1.6E+06	9.8E+05
Secondary effluent	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	1.90E+03	1.60E+03	3.20E+03	1.80E+03	2.00E+03	3.10E+03
E.coli	1.40E+03	1.00E+03	2.40E+03	1.20E+03	1.20E+03	1.30E+03
E. faecalis	3.70E+02	2.10E+02	2.20E+02	2.10E+02	7.00E+01	1.20E+02
Clostridium Perfringens	3.46E+03	3.80E+03	2.47E+03	2.43E+03	2.05E+03	3.45E+03
Bacteriophages	1.7E+04	6.8E+03	8.7E+03	1.2E+04	3.6E+03	1.2E+04
Tertiary	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	1	1	3	1	<1	<1
E.coli	1	<1	3	<1	<1	<1
E. faecalis	<1	<1	<1	<1	<1	<1
Clostridium Perfringens	1.5	3	2	2	2.5	1
Bacteriophages	<10	<10	<10	<10	<10	<10
Pond2	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	2	4	6	1	1	1
E.coli	1	3	3	<1	<1	1
E. faecalis	<1	<1	<1	<1	<1	<1
Clostridium Perfringens	10	3.5	9	3.5	2	2
Bacteriophages	70	10	10	<10	20	<10
PZ7	ufc/100ml	ufc/100ml				
Total coliforms	<1	<1				
E.coli	<1	<1				
E. faecalis	<1	1				
Clostridium Perfringens						
Bacteriophages	40	<10				

	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6
PZ6	ufc/100ml	ufc/100ml				
Total coliforms	<1	<1				
E.coli	<1	<1				
E. faecalis	<1	<1				
Clostridium Perfringens						
Bacteriophages	10	80				

Table 65 Total coliforms, E.coli, E. faecalis, Clostridium Perfringens, Bacteriophages measured in April 2016.

	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6
Raw sewage	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	1.40E+06	1.20E+06	6.20E+05	2.00E+06	1.46E+06	5.30E+05
E.coli	1.30E+06	9.20E+05	4.30E+05	1.80E+06	3.70E+05	4.00E+05
E. faecalis	1.00E+05	1.00E+04	4.00E+04	1.00E+04	6.00E+04	2.00E+04
Clostridium Perfringens	5.00E+04	1.27E+05	7.00E+04	9.60E+04	3.20E+05	7.70E+04
Bacteriophages	8.6E+05	1.3E+06	1.2E+06	1.7E+06		
Secondary effluent	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	4.60E+03	4.60E+03	4.70E+03	6.00E+03	6.40E+03	5.90E+03
E.coli	3.30E+03	3.00E+03	3.10E+03	3.80E+03	4.20E+03	4.30E+03
E. faecalis	5.20E+02	7.90E+02	7.90E+02	6.80E+03	7.10E+02	5.00E+02
Clostridium Perfringens	2.70E+03	1.70E+03	2.20E+03	2.50E+03	2.50E+03	2.50E+03
Bacteriophages	6.7E+03	6.9E+03	5.3E+03	6.9E+03		
Tertiary	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	23	11	5	14	12	16
E.coli	10	4	1	3	4	2
E. faecalis	<1	6	<1	1	2	1

	Sample 1	Sample 2	Sample 3	Sample 4	Sample 5	Sample 6
Clostridium Perfringens	3.5	8	1.5	4	2	2
Bacteriophages	<10	<10	<10	<10		
Pond2	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml	
Total coliforms	10	4	9	6	11	
E.coli	3	1	4	3	4	
E. faecalis	<1	<1	5	<1	1	
Clostridium Perfringens	27.5	8	7.5	14	11.5	
Bacteriophages*	4	8				
PZ7	ufc/100ml	ufc/100ml				
Total coliforms	<1	<1				
E.coli	<1	<1				
E. faecalis	<1	1				
Clostridium Perfringens						
Bacteriophages**	6	2				
PZ6	ufc/100ml	ufc/100ml				
Total coliforms	<1	<1				
E.coli	<1	<1				
E. faecalis	<1	<1				
Clostridium Perfringens						
Bacteriophages**	6	6				

*5 pfu/100 mL; **2 pfu/100 mL

Table 66 Total coliforms, E.coli, E. faecalis, Clostridium Perfringens, Bacteriophages measured in June 2016.

	Sample 1	Sample 2	Sample 3	Sample 4
Raw sewage	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	1.36E+07	9.70E+06	1.09E+07	5.40E+06
E.coli	1.21E+07	6.20E+06	6.60E+06	2.80E+06
E. faecalis	9.00E+04	3.10E+05	7.00E+04	1.70E+05
Clostridium Perfringens	5.30E+05	3.80E+05	3.40E+05	2.40E+05
Bacteriophages	1.2E+07	5.4E+06	7.3E+06	3.8E+06
Secondary effluent	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	1.27E+04	1.05E+04	1.04E+04	6.50E+03
E.coli	1.06E+04	8.10E+03	5.90E+03	3.20E+03
E. faecalis	3.10E+03	1.80E+03	1.20E+03	2.20E+03
Clostridium Perfringens	1.00E+02	3.20E+03	1.40E+03	1.30E+03
Bacteriophages	1.3E+04	1.6E+04	1.6E+04	1.1E+04
Tertiary	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	<1	4	7	<1
E.coli	<1	1	<1	<1
E. faecalis	<1	<1	1	<1
Clostridium Perfringens	4	3	10	7
Bacteriophages	<10	<10	<10	30
Pond2	ufc/100ml	ufc/100ml	ufc/100ml	ufc/100ml
Total coliforms	37	58	47	31
E.coli	<1	<1	1	<1
E. faecalis	19	13	43	46
Clostridium Perfringens	5	7	4	6
Bacteriophages*	7.5	5		
PZ7	ufc/100ml	ufc/100ml		

	Sample 1	Sample 2	Sample 3	Sample 4
Total coliforms	<1	<1		
E.coli	<1	<1		
E. faecalis	<1	<1		
Clostridium Perfringens	0	0		
Bacteriophages**	<2	<2		
PZ6	ufc/100ml	ufc/100ml		
Total coliforms	<1	<1		
E.coli	<1	<1		
E. faecalis	<1	<1		
Clostridium Perfringens	0	0		
Bacteriophages**	2	2		

*5 pfu/100 mL; **2 pfu/100 mL

Table 67 In-situu, hydrochemistry and calculated ion balance for samples classified as infiltrate groundwater.

StationID	Sampling Date	pH	Temperature	Conductivity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion balance
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
PZ 7	01/12/2015	na	13	900	85.40	186	0.1	7.9	<0.1	46	<0.1	0.3	0.1	77.5	3	46	18.1														-6.78
PZ 6	16/12/2015	6.7	17.1	899	57.95	253.2	<0.1	21.6	<0.1	50.9	0	0.3	<0.1	100.5	3.7	61	22.5														-6.78
PZ 7	16/12/2015	6.71	15.5	986	57.58	271.3	0.1	25.8	<0.1	57	<0.1	0.4	<0.1	125.5	5	60	21.8														-6.43
PZ 6	12/01/2016	6.62	15	740	54.90	263.3	<0.1	9.1	<0.1	58.4	<0.1	0.8	<0.1	105.9	3.6	54.1	21.9	1.8	0.032												-5.26
PZ 7	12/01/2016	6.77	15.7	721	53.19	270.3	<0.1	9.1	<0.1	59	<0.1	0.8	<0.1	108	4.3	55.3	18.8	1.8	0.041												-6.63
PZ 6	28/01/2016	6.93	17.6	1331	52.60	360	<0.2	7.5	<0.2	81.6	<0.1	0.7	<0.2	164	5.1	76	25.7		0.008												-0.34
PZ 7	28/01/2016	6.75	14.5	1346	52.90	371	<0.2	7.4	<0.2	83.5	<0.1	0.7	<0.2	173	6	76.5	28.4		0.032												0.83

StationID	Sampling Date	pH	Temperature	Conductivity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion balance
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
PZ 6	09/02/2016	6.91		1062	58.10	264	<0.1	5.5	<0.1	63	<0.1	0.4	<0.1	135	6	51	19	2.3	0.045	24	49	6	5	1.5	90						0.12
PZ 7	09/02/2016	6.93			60.00	268	<0.1	4.9	<0.1	62	<0.1	0.4	<0.1	141	8	47	17	2.5	0.05	9	18	1	5	1.5	99						-0.52
PZ 6	25/02/2016	na	14.9	1251	51.00	347	<0.1	8.2	<0.2	72	<0.1	1.3	<0.2	137	5	73	27														-3.20
PZ 7	25/02/2016	na	13.5	1296	53.00	375	<0.1	7.9	<0.2	72	<0.1	1.3	<0.2	143	5	76	27														-4.58
PZ 7	11/04/2016	na		1079	98.60	289.3	<0.1	4.5	<0.1	64.8	<0.1	1.2	<0.1	158.6	6.5	48.8	18.2	1.8	0.04	8	14	0.5	7.2	2	86	2.3	6	0.2	0.5	NA	-2.09
PZ 6	11/04/2016	na		803	78.60	187	<0.1	3.3	<0.1	51.1	<0.1	0.7	<0.1	103.5	4.2	42.5	16.6	1.1	0.031				5	4	49	2.8	11	<0.1	1.9	<0.1	1.44
PZ 7	15/06/2016	na	24.4	1306	99.40	342.8	<0.2	4.6	<0.2	77.2		1	<0.2	202.9	9.2	44.6	18.2	3.1	0.057	<5	32.6	0.6	11.9	1.9	133	4.1	15.7	0.2	0.6		-1.8
PZ 6	15/06/2016	na	22.8	1275	93.80	335.4	<0.2	4.7	<0.2	75.5		1	<0.2	189.9	6.5	52.3	20	2.7	0.049	30	56	1.1	11.1	1.7	116	3.5	14.7	0.2	0.7		-0.9
PZ 4	15/06/2016	na	17.4	729	79.00	175.7	<0.2	2.6	<0.2	46.1		0.5	<0.2	87.7	2.8	38.5	16.7	1.4	0.021	18	59	1.6	4	1	41	1.5	21.1	<0.1	0.5		-1.50
PZ 7	01/12/2015	na	13	900	85.40	186	0.1	7.9	<0.1	46	<0.1	0.3	0.1	77.5	3	46	18.1														-6.78
PZ 6	16/12/2015	6.7	17.1	899	57.95	253.2	<0.1	21.6	<0.1	50.9		0.3	<0.1	100.5	3.7	61	22.5														-6.78

Table 68 In-situ, hydrochemistry and calculated ion balance for samples from the infiltration ponds.

StationID	Sampling_Date	pH	Temperature	Conductivity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion balance (%)
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
Pond 1	01/12/2015	na	14.3	1284	59.78	272	0.5	18.1	0.4	59.5	1.5	0.4	0.2	127.5	16.6	49	16.4														-6.9
Pond 1	17/12/2015	6.97	13.7	920	47.70	272.4	0.5	14.2	0.4	60.9	<0.1	0.3	0.1	138.1	20	48.3	15.8														-2.6
Pond 3	12/01/2016	7.26	13.2	1204	64.29	381.1	<0.1	4.3	<0.1	75.7	<0.1	0.8	<0.1	186.4	19.8	48.9	21.7	3.3	0.0743												-3.3
Pond 1	28/01/2016	7.21	na	1280	0.00	337	<0.2	5.5	<0.2	67.2	<0.1	0.6	<0.2	189.4	20.3	48	24.8														7.8
Pond	09/02/201	6.8	12.7	1068	72.2	257	<0.	5.4	<0.	63	<0.	0.4	<0.	124	20	46	16	3.3	0.078	9	17	19	5	0.5	10						-3.2

StationID	Sampling_Date	pH	Temperature	Conductivity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion balance (%)
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
2	6	9			0		1		1		1		1									5			6						
Pond 3	25/02/2016	na	14.3	1600	100.00	439	<0.1	5.4	<0.2	91	<0.1	1.7	<0.2	230	20	55	31														-1.6
Pond 1	07/04/2016	7.74	16.2	1254	118.80	288.9	<0.2	6.4	<0.2	63.4	<0.1	0.3	<0.2	195.7	22.8	42.1	21.5	4.3												<0.1	4.5
Pond 1	15/06/2016	na	21	1342	111.20	339.1	<0.2	3.9	<0.2	77.7		1	<0.2	206.6	23.4	41.5	19.7	5.7	0.13	16	41.5	9.8	8.5	1.7	145	7.6	67	<0.1	0.1		-0.1

Table 69 In-situ, hydrochemistry and calculated ion balance for samples classified as native groundwater.

StationID	Sampling Date	pH	Temperature	Conductivity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion balance (%)
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
PZ 6	01/12/2015	na	14.7	500	87.84	76	<0.1	1.6	<0.1	35.1	0.2	0.2	0.1	43	2	22.6	9.2														-7.8
AM1	06/05/2014	7.42	21.7	386	69.30	85	0.01	1.5	1.6	28	0.28	0.2	<0.1	44.7	<2	19.7	8.6	<0.70		<20	102	<5								<0.04	-8.2
PZ 3	21/10/2015	7.8		517	*72.00*	77.9	<0.2	3	<0.2	24.2	0.1	0.2	<0.2	44.3	4.4	39.3	10.9	1.8	0.033	42.1	77.1	85.5	2.3	1.3	23	3.1	2.7	<0.1	0.7		9.2
PZ 4	21/10/2015	6.8		408	72.71	69.4	<0.2	3.9	<0.2	30.9	<0.1	<0.2	<0.2	39.5	1.9	23.4	9.1	0.9	0.014	38.1	115.9	14.8	2	1	18.8	1.5	4.6	<0.1	1.2		-5.0
AM1	22/10/2015	7.1		487	*70.00*	89.4	<0.2	5	<0.2	32.9	<0.1	0.3	<0.2	46.4	2.2	27.3	10.4	0.6	0.013	26.9	60.4	2.8	16.4	1.0	27.7	1.7	102.9	<0.1	0.7		-4.7
AM2	22/10/2015	7		479	*70.00*	86.8	<0.2	4.9	<0.2	32.4	<0.1	0.2	<0.2	45.1	2.0	26.9	10.3	0.5	0.01	23.9	63.0	2.4	15.1	1.0	26.7	1.3	70.0	<0.1	1.0		-4.8
Hort Gran Bayé	22/10/2015	7		590	*90.00*	114.6	<0.2	7.3	<0.2	38.7	<0.1	0.3	<0.2	52.9	2.9	35.2	13.4	0.9	0.016	25.8	49.6	4.5	4.9	0.9	31.4	1.2	34.6	<0.1	0.5		-7.2
PZ 5	22/10/2015	6.8		371	67.83	60.7	<0.2	3.5	<0.2	27.2	<0.1	<0.2	<0.2	35.0	1.5	20.1	8.1	0.9	0.015	24.7	35.3	2.3	1.6	1.0	16.5	0.6	3.7	<0.1	0.7		-6.0
AM1	19/11/2015	7.1	17.4	426	75.64	77	<0.1	4	<0.1	27.1	<0.1	0.2	<0.1	39.5	2.0	24.0	9.3	0.5	0.018	17.0	45.0	4.0	9.0	0.9	31.0						-6.7
Hort Gran Bayé	19/11/2015	6.7	18.2	501	*90.00*	91	<0.1	7.5	<0.1	32.8	<0.1	0.2	0.1	47.7	3.0	25.6	11.3	1.2	0.027	25.0	82.0	7.0	1.0	0.7	31.0						-9.5
Hort Petit Bayé	19/11/2015	6.9	17.5	386	70.00	62	<0.1	6.6	0.2	31.7	<0.1	0.2	0.1	35.2	3.0	23.0	8.3			43.0	63.0	4.0	1.0	1.2	32.0						-8.1
PZ 6	20/11/2015	6.9	18.7	434	78.08	74	<0.1	2.6	<0.1	32.4	<0.1	0.1	<0.1	44.6	2.0	24.2	10.0	2	0.033	85.0	81.0	9.0	5.0	2.1	26.0						-2.4
PZ 7	20/11/2015	6.9	17.5	437	84.18	75	<0.1	2.5	<0.1	32.1	<0.1	0.2	<0.1	44.0	2.0	23.6	9.6	0.9	0.024	21.0	24.0	3.0	1.0	1.4	25.0						-5.0

Sta-tionID	Sampling Date	pH	Tempera-ture	Conductiv-ity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion bal-ance (%)
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
PZ 4	12/01/2016	6.77	16	330	***90.00***	86.1	<0.1	2.3	<0.1	37.3	<0.1	0.3	<0.1	47.2	2.2	26.2	10.9	1.1	0.019												-4.2
PZ 4	09/02/2016	6.93	15.9	524.7	54.90	103	<0.1	1.4	<0.1	40	<0.1	0.2	<0.1	57.0	3.0	32.0	14.0	0.7	0.009												6.8
PZ 4	25/02/2016		14.5	465	76.00	87	<0.1	1.6	<0.2	39	<0.1	0.3	<0.1	45.0	2.0	28.0	11.0														-2.4
AM1	06/04/2016			432	75.00	82.9	<0.2	4.2	<0.2	27.7	<0.1	<0.2	<0.2	50.1	2.3	26.4	10.8														3.5
PZ 4	07/04/2016		16.3	402	71.10	70.4	<0.1	1	<0.1	37	<0.1	0.3	<0.1	48.3	2.5	24.6	10.2	4.3	0.011	22.0	31.0	0.7	1.0	1.0	20.0	0.6	2.0	<0.1	0.2		3.9
PZ 2	02/06/2016		16.1	531	66.40	110.8	<0.1	1.7	<0.1	30.1		0.4	<0.1	62.2	2.6	27.9	10.9	1.4	0.016	18.0	460	120	1.5	0.2	25.0	0.9	9.0	<0.5	0.6		2.4
PZ 4	02/06/2016		19.7	398	61.60	70	<0.1	1.1	<0.1	39.2		0.2	<0.1	44.4	1.9	21.6	9.7	0.9	0.015	24.0	63.0	1.6	1.9	1.2	22.0	1.3	14.0	<0.5	0.4		0.8
PZ 2	15/06/2016		22.4	528	77.40	115.8	<0.2	0.6	<0.2	27.9		0.4	<0.2	61.8	2.6	29.3	11.8	1.1	0.01	13.0	123.0	246.8	1.1	0.2	18.0	0.7	16.7	<0.1	0.5		0.8

Red, bold and cursive values = calculated not measured

Table 70 In-situ, hydrochemistry and calculated ion balance for Bolera samples (seawater influenced).

Sta-tionID	Sampling Date	pH	Tempera-ture	Conduc-tivity	HCO3	Cl-	NO2	NO3-	PO4	SO4	NH4+	Br	F	Na+	K+	Ca2+	Mg2+	DOC	UVA254	Al	Fe	Mn	Cu	As	B	Ni	Zn	Cd	Pb	AOX	ion bal-ance (%)
		-	°C	µS/cm	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	1/m	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	mg/L Cl	(%)
Bolera	21/10/2015	7		1170	***75***	286.9	<0.2	5.5	<0.2	54.5	<0.1	1	<0.2	138.6	6.5	39.6	26.9	0.6	0.014	41.1	138.0	16.0	3.0	1.0	43.0	1.0	17.0	<0.1	1.7		-0.4
Bolera	19/11/2015	7.1	17.6	883	***75***	200	<0.1	5.6	0.5	49.6	<0.1	0.5	0.5	103.6	5.0	29.2	20.4	0.5	0.031	15.0	194.0	17.0	1.0	0.8	45.0						-1.1
Bolera	30/11/2015	na	18	605	***75***	163	<0.1	6.2	0.1	46.2	<0.1	0.4	0.1	89.4	4.4	23.9	16.3														-2.0
Bolera	16/12/2015	6.71	16.8	627	***75***	155.1	<0.1	6.1	0.1	46.9	<0.1	0.4	0.1	101.5	3.9	24.3	15.4														3.0
Bolera	12/01/2016	6.83	16.2		***75***	156.8	<0.1	6.1	<0.1	53	<0.1	0.7	<0.1	91.6	4.6	23.2	15.6														-1.7
Bolera	28/01/2016	6.85	16.6	633	81.8	146	<0.2	5.3	<0.2	46.3	<0.1	0.4	<0.2	101.0	4.4	24.2	16.1														4.6
Bolera	25/02/2016		19.2	545	75	131	<0.1	5.3	<0.2	42	<0.1	0.5	0.1	72.0	4.0	24.0	14.0														-2.0
Bolera	11/04/2016			693	77.4	153.5	<0.1	4.8	<0.1	46.2	<0.1	0.7	<0.1	106.7	4.9	23.8	16.9	0.4	0.005	8.0	193.0	37.0	0.9	1.0	45.0	0.6	15.0	<0.1	0.3		5.7

Red, bold and cursive values = calculated not measured

Table 71 Rotavirus, Enterovirus. Norovirus and Adenoivirus sampled in February 2016

Samples	Date	Concentration of virus (GC/L)				
		Rotavirus	Enterovirus	NoV GGI	NoV GGII	HAdV
PZ6-1		ND	ND	ND	1.97E+02	2.34E+02
PZ6-2		ND	ND	ND	1.99E+01	7.77E+01
PZ7-1		ND	ND	ND	ND	ND
PZ7-2		ND	ND	ND	1.14E+02	ND
Raw	09/02/2016	1.39E+03	ND	ND	5.14E+04	1.94E+03
Pond		ND	ND	ND	1.27E+03	3.78E+01
Secondary		ND	ND	6.19E+01	5.11E+03	6.42E+01
Tertiary		ND	ND	ND	9.91E+02	ND
PZ6-1		ND	ND	ND	1.62E+02	ND
PZ6-2		ND	ND	ND		ND
PZ7-1		ND	ND	ND	1.97E+02	1.13E+02
PZ7-2		ND	ND	ND	ND	ND
Raw	11/02/2016	3.15E+04	ND	4.08E+01	3.53E+04	6.30E+02
Pond		ND	ND	ND	1.77E+02	ND
Secondary		2.43E+03	ND	3.56E+01	2.18E+04	3.76E+02
Tertiary		5.42E+02	ND	ND	1.98E+03	ND

Table 72 Rotavirus, Enterovirus. Norovirus and Adenoivirus sampled in April 2016

Samples	Date	Concentration of virus (GC/L)				
		Rotavirus	Enterovirus	NoV GGI	NoV GGII	HAdV
PZ6-1		ND	ND	ND	4.90E+04	ND
PZ6-2		ND	ND	ND		ND
PZ7-1	07/04/2016	1.28E+03	ND	ND	ND	ND
PZ7-2		1.35E+03	ND	ND	ND	ND
Raw		4.58E+06	ND	1.12E+04	1.07E+06	9.33E+05
Pond	06/04/2016	3.01E+05	ND	ND	1.36E+03	3.17E+03
Secondary		1.84E+05	ND	ND	ND	2.47E+02
Tertiary		3.45E+05	ND	ND	1.27E+03	1.38E+03

Table 73 Rotavirus, Enterovirus. Norovirus and Adenoivirus sampled in June 2016

Samples	Date	Concentration of virus (GC/L)				
		Rotavirus	Enterovirus	NoV GGI	NoV GGII	HAdV
PZ6.1	15/06/2016	ND	ND	ND	9.07E+01	1.25E+02
PZ6.2		ND	ND	ND	1.70E+01	6.52E+01
PZ7.1		ND	ND	ND	ND	5.05E+01
PZ7.2		ND	ND	ND	ND	2.05E+01
Raw		ND	ND	ND	3.39E+04	5.54E+04
Pond		ND	ND	ND	ND	5.08E+01
Secondary		ND	ND	ND	ND	1.68E+02
Tertiary		ND	ND	ND	ND	4.73E+01

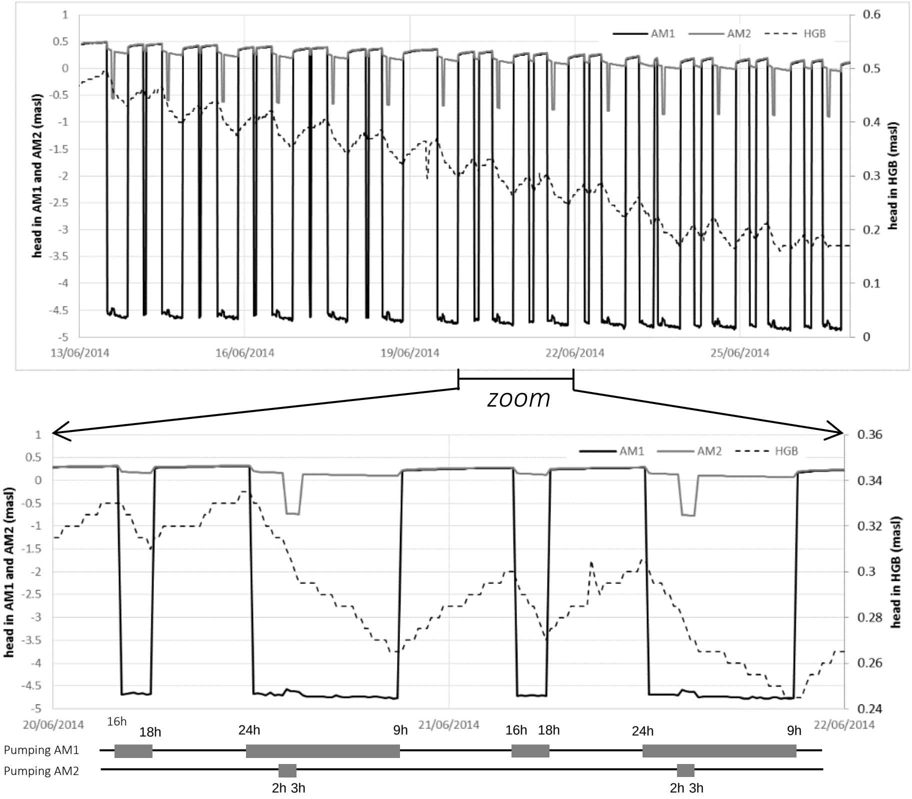

Figure 71 Evolution of hydraulic heads in AM1, AM2 and HGB and pumping scheme in wells AM1 and AM2 (note change in vertical scale for HGB).

APPENDIX 2 (Shafdan pilot plant)

Table 74 Field measurements – 2014-2015

Date	Exp. name	Sampling point	Sampling point	pH (field)	Temp (field)	ORP (field)	Turb (field)	Conuctivity	DO (field)	DOLB (DO lab)	H2O2
					oC	mV	NTU	µmho/cm	mg/L	mg/L	mg/L
27/5/14	H2O2 (27)+Ozo (10)	Secondary Effluents	OZA500	7.28			2.45			4.1	
27/5/14	H2O2 (27)+Ozo (10)	After Biofilter	OZOASF	7.26			3.51			0.6	0.56
27/5/14	H2O2 (27)+Ozo (10)	After Ozonation	OZOAOZ	7.00			0.67			20.0	
9/7/14	H2O2 (40)+Ozo (10)	Secondary Effluents	OZA500	7.56	29.9	153.60	1.51			4.0	
9/7/14	H2O2 (40)+Ozo (10)	After Biofilter	OZOASF	7.19	30.7	239.30	0.33			10.0	0.34
9/7/14	H2O2 (40)+Ozo (10)	After Ozonation	OZOAOZ	7.23	30.1	553.00	0.20			20.0	
20/7/14	H2O2 (1)+Ozo (10)	Secondary Effluents	OZA500	7.53	28.8	143.20	1.57			5.4	1.26
20/7/14	H2O2 (1)+Ozo (10)	After Biofilter	OZOASF	7.30	28.9	204.30	0.29			9.2	0.61
20/7/14	H2O2 (1)+Ozo (10)	After Ozonation	OZOAOZ	7.20	28.8	765.00	0.37				
3/8/14	H2O2 (27)+Ozo (10)	Secondary Effluents	OZA500	7.44	30.7	117.50	1.64		4.50	3.5	
3/8/14	H2O2 (27)+Ozo (10)	After Biofilter	OZOASF	7.30	31.2	141.40	0.30		5.33	2.3	0.23
3/8/14	H2O2 (27)+Ozo (10)	After Ozonation	OZOAOZ	7.11	31.7	272.70	0.23			20.0	
9/9/14	H2O2 (~20)+Ozo (10)	Secondary Effluents	OZA500	7.48	29.8	164.30	2.02		2.78	2.7	
9/9/14	H2O2 (~20)+Ozo (10)	After Biofilter	OZOASF	7.29	29.8	149.30	0.49		3.21	2.4	0.27
9/9/14	H2O2 (~20)+Ozo (10)	After Ozonation	OZOAOZ	7.15	28.9	500.40	0.52			20.0	
28/9/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1								
28/9/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3								
30/9/14	H2O2 22 mg/L	Secondary Effluents	OZA500								
30/9/14	H2O2 22 mg/L	After Biofilter	OZOASF								
30/9/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	7.00	29.8	225.80	0.92		0.98	0.7	
30/9/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	7.15	29.2	133.90	1.45		0.47	1.4	
4/11/14	H2O2 ~27 mg/L	Secondary Effluents	OZA500	7.35	26.3	142.90	1.98		3.79	3.0	
4/11/14	H2O2 ~27 mg/L	After Biofilter	OZOASF	7.26	25.6	220.30	0.46		8.50	7.7	0.24
4/11/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	7.00	29.1	234.00	0.41		0.71	1.1	
4/11/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	7.20	29.2	235.20	0.27		0.73	1.3	
30/12/14	H2O2 ~27 mg/L	Secondary Effluents	OZA500	7.20	22.4	176.30	3.44	1190	6.30	4.8	
30/12/14	H2O2 ~27 mg/L	After Biofilter	OZOASF	6.96	22.7	155.50	0.77	1200	5.72	2.8	0.32
30/12/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	6.97	27.3	265.80	0.27	1230	3.03	3.5	
30/12/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	6.71	25.2	162.20	0.15	1330	0.65	0.6	
27/1/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500	7.24	19.8	196.50	3.37	1190	5.13	3.2	
27/1/15	H2O2 ~27 mg/L	After Biofilter	OZOASF	6.76	20.1	180.80	0.41	1200	3.20	0.7	0.34
27/1/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	7.21	25.7	257.20	0.19	1230	3.01	2.5	
27/1/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	7.20	24.5	261.20	0.24	1170	3.41	1.2	
29/1/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500				2.45				
29/1/15	H2O2 ~27 mg/L	After Biofilter	OZOASF				0.43				
3/3/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
9/3/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
23/3/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	7.12	24.1	220.70	0.23	1210	4.05	3.0	
23/3/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	7.10	27.5	166.20	0.18	1270	2.74	1.8	
28/4/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500	7.27	25.4	117.20	2.23	1320			
28/4/15	H2O2 ~27 mg/L	After Biofilter	OZOASF	6.80	25.9	153.10	0.63	1340			0.32
28/4/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	7.11	25.5	244.20	0.23	1370	5.57	2.5	
28/4/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	7.10	27.7	209.30	0.21	1500			
30/4/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
30/4/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1								
12/5/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
15/6/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500								
2123/06/2015	H2O2 ~27 mg/L	Ceramic cups	OZMAS1,2,3	7.64	31.4	108.00	0.58		7.15		
19/7/15	H2O2 ~27 mg/L	Ceramic cups	OZMAS1,2,3								
5/8/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500								
5/8/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
5/8/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1								

Table 75 Field measurements – 2015-2016

Date	Exp. name	Sampling point	Sampling point	pH (field)	Temp (field)	ORP (field)	Turb (field)	Conuctivity	DO (field)	DOLB (DO lab)	H2O2
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	7.35	32.2	119.70	1.78	1240	5.50	3.7	
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.05	33.1	98.20	0.71	1240	2.54	0.8	0.31
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.10	33.0	193.20	0.72	1230		20.0	
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.18	27.4	141.20	0.29	1200	4.10	1.2	
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.20	24.1	146.50	0.52	1170	3.28	1.6	
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500								
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								0.32
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ								
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1								
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3								
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.35	27.6	145.30	0.26	1490	5.99	4.0	25.00
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	6.50	26.1	108.00	0.20	1240	1.78	1.5	
21/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	7.30	20.1	87.40	3.90	1150	6.15	4.7	
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	6.80	20.5	136.00	0.69	1160	4.19	2.9	
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	6.86	21.0	132.00	0.49	1140	13.28	15.3	
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3								
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.26	25.4	139.40	0.48	1240	1.82	1.8	
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.34	26.4	150.50	0.26	1270	4.39	2.2	
27/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
27/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ								
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500								
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.35	21.9	99.80	0.90	1370	5.12	3.2	
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.17	27.0	93.30	0.16	1400	0.11	0.1	
22/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
22/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ								
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	7.36	31.0	115.60	1.89	1060	5.67		
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	6.89	31.1	80.90	0.67	1060	0.35		
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	6.89	31.9	335.00	0.91	1080	19.93		
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3								
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.03	23.2	176.10	0.32	1210	6.14		
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.14	24.9	205.90	0.82	933	4.23		
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	7.48	29.9	182.40	2.67	1190	5.48		
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.12	30.4	185.70	0.67	1170	2.90		
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.04	30.8	210.40	0.69	1150	18.85		
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	7.50	30.3	219.30	1.36	1120	6.07		
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.15	24.7	226.90	1.31	1100	4.11	2.75	
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.25	24.8	186.20	0.75	1120	2.36	2.08	
14/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1				0.08				
14/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3				0.26				
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	7.38	31.5	115.60	2.78	1140	4.94		
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	6.95	31.8	126.20	0.56	1130	3.39		0.40
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.03	32.2	197.20	0.51	1170	17.10		
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3								
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.08	26.3	176.40	0.15	1150	2.53	3.20	
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.08	26.4	189.70	0.46	1196	1.76	1.60	
04/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1				0.40				
04/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3				0.40				
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	7.35	30.9	127.90	1.47	1120	5.53		
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	6.94	31.5	111.20	0.64	1100	2.93		0.38
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.08	32.4	183.00		1120	17.70		
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3								
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	7.02	26.4	223.70	0.24	1160	3.17		
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	7.08	24.5	220.80	1.70	1180	3.70		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500		30.2	94.80	1.30		5.63		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ		31.0	189.60	0.67		19.56		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB114m		28.20	222.00	8.07		3.20		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB118m		28.00	218.00	7.35		3.12		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB122m		27.00	235.00	7.79		2.80		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB126m		28.00	241.30	40.80		3.05		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1								
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB314m								
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB318m				2.70		3.30		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB322m		25.50	182.30	18.00		6.50		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB326m		24.40	206.00			3.50		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3		26.20	225.00	2.77		1.19		

Table 76 Chemical parameters – 2014-2015

Date	Exp. name	Sampling point	Sampling point	DOC	TOC	UVA	NO_3-N	NO_2-N	NH_4-N	DISS ORTHO PHOSPHATE AS P	BOD AS O_2	COD AS O_2	Fe (t)	Mn (t)	AL as AL	BROMIDE AS BR	BROMATE as BRO_3
				mg/L	mg/L	cm1X103	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	µg/L	µg/L	ug/L	mg/L	ug/L
27/5/14	H_2O_2 (27)+Ozo (10)	Secondary Effluents	OZA500	12.91					4.30								
27/5/14	H_2O_2 (27)+Ozo (10)	After Biofilter	OZOASF	10.08			4.34	0.08	ND								
27/5/14	H_2O_2 (27)+Ozo (10)	After Ozonation	OZOAOZ	9.00													
9/7/14	H_2O_2 (40)+Ozo (10)	Secondary Effluents	OZA500	9.50		224	0.39	0.60	2.69								
9/7/14	H_2O_2 (40)+Ozo (10)	After Biofilter	OZOASF	7.70		169	2.77	ND	0.06								
9/7/14	H_2O_2 (40)+Ozo (10)	After Ozonation	OZOAOZ	7.30		79	3.32	ND	0.09								
20/7/14	H_2O_2 (1)+Ozo (10)	Secondary Effluents	OZA500	8.50		201	0.45	2.94	2.62								
20/7/14	H_2O_2 (1)+Ozo (10)	After Biofilter	OZOASF	6.80		151	5.65	0.00	0.07								
20/7/14	H_2O_2 (1)+Ozo (10)	After Ozonation	OZOAOZ	6.70		56	5.65	0.00	0.09								
3/8/14	H_2O_2 (27)+Ozo (10)	Secondary Effluents	OZA500	8.50		207	0.23	0.86	6.12								
3/8/14	H_2O_2 (27)+Ozo (10)	After Biofilter	OZOASF	7.00		159	4.29	0.05	1.66								
3/8/14	H_2O_2 (27)+Ozo (10)	After Ozonation	OZOAOZ	6.80		65	5.20	0.01	1.08								
9/9/14	H_2O_2 (~20)+Ozo (10)	Secondary Effluents	OZA500	9.50		207	0.23	0.88	3.73	0.59	4.0	32.0	66.0	20		0.66	
9/9/14	H_2O_2 (~20)+Ozo (10)	After Biofilter	OZOASF	8.00		171	3.84	0.02	0.17	0.39	1.0	25.0		19			
9/9/14	H_2O_2 (~20)+Ozo (10)	After Ozonation	OZOAOZ	7.40		63	4.07	0.00	0.33	0.39	2.0	16.0		19			52
28/9/14	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1													0.58	ND
28/9/14	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3													0.64	ND
30/9/14	H_2O_2 22 mg/L	Secondary Effluents	OZA500			230											
30/9/14	H_2O_2 22 mg/L	After Biofilter	OZOASF			190											
30/9/14	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1	1.40		43	4.74	0.04	ND	ND	ND	20.0	17.0	35			
30/9/14	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3	1.10		34	4.04	0.13	ND	0.50	ND	20.0	209.0	450			
4/11/14	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500	10.10		209	0.56	1.90	1.02		4.0	35.0	54.0	17			
4/11/14	H_2O_2 ~27 mg/L	After Biofilter	OZOASF	7.30		159	3.54	ND	0.05		ND	21.0	23.0	5			
4/11/14	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1	1.30		42	5.59	0.06	ND		ND	3.8	ND	27			
4/11/14	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3	1.20		34	3.31	0.11	ND		ND	4.1	ND	188			
30/12/14	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500	12.60		243	0.51	0.59	2.84		6.0	45.0	90.0	20			
30/12/14	H_2O_2 ~27 mg/L	After Biofilter	OZOASF	9.60		196	3.61	ND	0.07		1.0	29.0	48.0	14			
30/12/14	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1	1.40		43	5.04	ND	ND		ND	20.0	ND	34			
30/12/14	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3	1.30		41	3.63	0.14	ND		ND	20.0	ND	375			
27/1/15	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500	12.30		223	0.40	0.73	6.03	0.90	6.0	44.0	64.0	23			
27/1/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF	9.30		171	4.13	0.03	1.72	0.60	1.0	27.0	26.0	21			
27/1/15	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1	1.30		42	4.76	ND	ND	ND	ND	4.0	ND	52			
27/1/15	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3	1.20		42	3.79	0.17	0.05	ND	ND	3.0	ND	529			
29/1/15	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500														
29/1/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF														
3/3/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF						3.50								
9/3/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF						ND								
23/3/15	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1	1.50		42	5.79	ND	ND				ND	45			
23/3/15	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3	1.40		38	4.65	0.16	ND				ND	572			
28/4/15	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500	12.00		215	0.23	0.35	3.35					27			
28/4/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF	9.40		177	3.16	0.11	0.05					22			
28/4/15	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1	1.50		47	4.52	0.02	ND					46			
28/4/15	H_2O_2 ~27 mg/L	Observation Well 2	OZOOB3	1.30		40	7.00	0.18	ND					532			
30/4/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF														
30/4/15	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1														
12/5/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF												185		
15/6/15	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500														
2123/06/2015	H_2O_2 ~27 mg/L	Ceramic cups	OZMAS1,2,3	5.20		113	4.56	0.04	ND					ND			
19/7/15	H_2O_2 ~27 mg/L	Ceramic cups	OZMAS1,2,3	3.80		118	3.61	0.01						6			
5/8/15	H_2O_2 ~27 mg/L	Secondary Effluents	OZA500														
5/8/15	H_2O_2 ~27 mg/L	After Biofilter	OZOASF														
5/8/15	H_2O_2 ~27 mg/L	Observation Well 1	OZOOB1														

Table 77 Chemical parameters – 2015-2016

Date	Exp. name	Sampling point	Sampling point	DOC	TOC	UVA	NO3-N	NO2-N	NH4-N	DISS ORTHO PHOSPHATE AS P	BOD AS O2	COD AS O2	Fe (t)	Mn (t)	AL as AL	BROMIDE AS BR	BROMATE as BRO3
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	10.30		208	0.35	0.73	5.83					30			
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	8.00		173	2.11	ND	2.95					28			
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.40		87	2.47	0.10	2.62					28			
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.30		38	4.01	0.05	ND					25			
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	1.20		44	4.57	0.10	ND					119			
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	9.40		202	0.45	1.29	9.47					16			
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.30		159	4.29	0.03	2.76					16			
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.00		86	4.74	0.05	1.35					16			
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.50		42	2.71	0.05	ND					14			
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	1.20		30	3.61	0.01	ND					39			
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.10		35	3.84	0.09	ND			3.0	ND	29			
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	1.20		32	7.91	0.04	ND			5.0	ND	306			
21/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF						2.60								
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	11.20		205	1.02	0.62	4.40	0.62		46.0	55.0	19	14		ND
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.90		154	5.55	ND	ND	0.33		28.0	27.0	18	164		ND
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.90		156	5.97	0.05	ND	0.36		27.0	32.0	19	213		ND
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3														
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.40		37	6.35	ND	ND	0.21			ND	11	ND		ND
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	1.20		34	3.09	0.11	ND	0.14			ND	482	ND		ND
27/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF			147											
27/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ			70											
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	9.70	10.60												
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.00	10.40												
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.20		32	8.76	0.03	0.02					44			
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	1.30		35	5.18	0.17	0.02					708			
22/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF			164											
22/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ			58											
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	9.50		203	0.68	0.97	2.82					18			
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.70		168	3.16	0.01	0.57					19			
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.30		62	4.07	0.01	1.55					21			
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	2.10		40								ND			
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.30		35	6.55	0.03	ND					43			
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	0.90		26	8.58	0.09	ND					167			
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	10.40		148	0.45	0.72	11.72					23			
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.30		149	2.94	0.01	6.91					23			
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	8.10		72	4.07	0.01	4.23					23			
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	2.00		43	10.84	0.01						ND			
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	1.00		26	6.33	0.04	ND					46			
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	1.00		26	6.55	0.09	ND					167			
14/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1														
14/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3														
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	9.60		193			5.07					22		0.49	
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.80		151			2.75					22			
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.40		67			2.46					24			15
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3														
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	0.80		24			ND					29			13
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	0.80		26			ND					138			ND
04/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1														
04/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3														
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	8.70		194			1.87					20		0.47	
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	7.30		168			0.33					24			
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	7.20		69			0.46					23			
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	1.60		38								3			
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	0.80		22			ND					35			
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	0.80		25			ND					170			
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500														
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ														ND
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB114m	0.90		21.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB118m	0.80		21.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB122m	0.90		22.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB126m	0.70		21.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	0.80		22.00											9
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB314m	0.90		23.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB318m	0.80		22.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB322m	0.70		21.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB326m	2.00		39.00											
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	0.80		26.00											ND

Table 78 TrOCs – 2014-2015

Date	Exp. name	Sampling point	Sampling point	IPDL	IHX	IPRM	BZF	VLX	CBZ	DCF	SMX
				ng/L	ng/L	ng/L	ng/L	ng/L	ng/L	ng/L	ng/L
27/5/14	H2O2 (27)+Ozo (10)	Secondary Effluents	OZA500	ND	27710	9548	71	2254	6240	12916	5104
27/5/14	H2O2 (27)+Ozo (10)	After Biofilter	OZOASF	3771	27710	9548	ND	2721	6240	14053	5104
27/5/14	H2O2 (27)+Ozo (10)	After Ozonation	OZOAOZ	1691	9810	2331	ND	ND	16	3544	7
9/7/14	H2O2 (40)+Ozo (10)	Secondary Effluents	OZA500	13333	21398	20202	ND	321	35	ND	ND
9/7/14	H2O2 (40)+Ozo (10)	After Biofilter	OZOASF	8389	16936	14980	ND	ND	1539	ND	ND
9/7/14	H2O2 (40)+Ozo (10)	After Ozonation	OZOAOZ	1040	4930	1476	ND	ND	ND	ND	ND
20/7/14	H2O2 (1)+Ozo (10)	Secondary Effluents	OZA500	ND	15138	ND	ND	1076527	1385	ND	ND
20/7/14	H2O2 (1)+Ozo (10)	After Biofilter	OZOASF	ND	9934	4361	ND	1946	1322	ND	ND
20/7/14	H2O2 (1)+Ozo (10)	After Ozonation	OZOAOZ	830	3399	1074	ND	1	ND	ND	ND
3/8/14	H2O2 (27)+Ozo (10)	Secondary Effluents	OZA500	ND	11235	ND	ND	743	1427	ND	ND
3/8/14	H2O2 (27)+Ozo (10)	After Biofilter	OZOASF	ND	9934	ND	ND	674	987	ND	ND
3/8/14	H2O2 (27)+Ozo (10)	After Ozonation	OZOAOZ	ND	ND	777	78	48	72	ND	ND
9/9/14	H2O2 (~20)+Ozo (10)	Secondary Effluents	OZA500	934	19389	5632	ND	916	1093	1559	3604
9/9/14	H2O2 (~20)+Ozo (10)	After Biofilter	OZOASF	934	13882	5099	ND	570	1102	1328	2961
9/9/14	H2O2 (~20)+Ozo (10)	After Ozonation	OZOAOZ	465	4663	1522	37	4	127	9	43
28/9/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1								
28/9/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3								
30/9/14	H2O2 22 mg/L	Secondary Effluents	OZA500								
30/9/14	H2O2 22 mg/L	After Biofilter	OZOASF								
30/9/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	806	ND	16	6	14	449	ND	16
30/9/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	788	ND	ND	ND	2	794	ND	ND
4/11/14	H2O2 ~27 mg/L	Secondary Effluents	OZA500	1909	22694	7463	12867	1958	856	2457	721
4/11/14	H2O2 ~27 mg/L	After Biofilter	OZOASF	423	13252	4656	10768	859	637	479	380
4/11/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	654	ND	ND	ND	24	634	ND	6
4/11/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	1067	ND	5	ND	ND	622	ND	2
30/12/14	H2O2 ~27 mg/L	Secondary Effluents	OZA500	1790	14609	5051	330	631	ND	574	137
30/12/14	H2O2 ~27 mg/L	After Biofilter	OZOASF	1079	8334	3959	330	602	417	574	166
30/12/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	481	ND	ND	ND	17	501	9	27
30/12/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	471	ND	ND	ND	ND	494	14	6
27/1/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500	5996	31936	8032	1176	584	ND	ND	67
27/1/15	H2O2 ~27 mg/L	After Biofilter	OZOASF	5996	22872	5244	1275	375	ND	ND	109
27/1/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	926	ND	ND	35	49	886	ND	20
27/1/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	792	ND	ND	28	64	722	ND	17
29/1/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500								
29/1/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
3/3/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
9/3/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
23/3/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	226	ND	ND	ND	ND	163	ND	ND
23/3/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	317	ND	ND	ND	ND	205	ND	ND
28/4/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500	5072	31831	22354	15922	ND	4849	ND	5139
28/4/15	H2O2 ~27 mg/L	After Biofilter	OZOASF	3657	24191	20391	15794	ND	9265	ND	5502
28/4/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	6952	ND	ND	ND	ND	2229	ND	983
28/4/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	1152	ND	ND	ND	ND	542	ND	165
30/4/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
30/4/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1								
12/5/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
15/6/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500								
2123/06/2015	H2O2 ~27 mg/L	Ceramic cups	OZMAS1,2,3	229	128	41	ND	203	123	ND	100
19/7/15	H2O2 ~27 mg/L	Ceramic cups	OZMAS1,2,3	157	ND	5	ND	50	141	ND	134
5/8/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500								
5/8/15	H2O2 ~27 mg/L	After Biofilter	OZOASF								
5/8/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1								

Table 79 TrOCs – 2015-2016

Date	Exp. name	Sampling point	Sampling point	IPDL	IHX	IPRM	BZF	VLX	CBZ	DCF	SMX
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	1517	21241	6646	3565.64	473.36	1884.94	ND	3263.68
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	1018	22069	4032	ND	493.02	770.23	ND	1966.21
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	1031.903904	7074.673632	2083.061469	181.9	25.2	ND	72.1	105.7
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	483	ND	ND	ND	27	561	ND	63
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	628	ND	ND	ND	ND	1088	ND	69
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	ND	6874	2353	ND	313	problem	ND	805
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	ND	5012	ND	ND	156	problem	ND	640
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	355.4	2400.8	1047.2	284	ND	ND	ND	58.7
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	487	ND	ND	ND	4	856	ND	73
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	616	ND	ND	ND	ND	1022	ND	69
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	889	ND	ND	ND	63	292	ND	11
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	172	ND	ND	ND	8	307	ND	ND
21/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	1525	39371	5483	3744	377	726	ND	2747
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	ND	32292	4794	2125	377	327	ND	3411
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	2333	21689	2103	561	224	253	1012	100
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	588	535	128	16	200	311	1845	54
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	791	ND	ND	ND	44	498	ND	ND
05/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	387	ND	ND	ND	10	376	ND	ND
27/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
27/01/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ								
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500								
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	820	ND	ND	26	61	977	ND	ND
23/02/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	916	ND	ND	ND	ND	486	ND	ND
22/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF								
22/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ								
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	1449	21952	4066	ND	246	878	337	211
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	1662	20732	3313	ND	392	1218	630	450
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	573	5499	992	ND	ND	36	ND	ND
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	199	143	83	ND	ND	44	ND	ND
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	315	ND	ND	ND	69	602	ND	139
27/06/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	440	ND	ND	ND	ND	456	ND	123
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	1163	37074	8274	156	286	1037	517	380
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	1269	28609	6383	ND	337	1112	708	577
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	587	8309	1582	ND	ND	43	ND	ND
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3								
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	406	ND	ND	ND	ND	994	ND	ND
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	115	ND	ND	ND	92	174	ND	124
14/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1						ND		
14/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3						630		
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	778	37049	9366	99	275	870	995	306
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	837	36859	7385	119	313	901	1023	375
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	155	7178	1801	ND	ND	ND	ND	ND
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	77	488	121	ND	4	ND	7	ND
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	68	ND	ND	ND	53	ND	ND	13
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	419	ND	ND	ND	ND	934	ND	ND
04/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1						ND		
04/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3						710		
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	1268	42026	9559	ND	241	901	557	403
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	1404	28697	6039	ND	254	1021	664	400
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	827	10139	1976	ND	ND	ND	ND	ND
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Ceramic cups	OZMAS1,2,3	ND	ND	ND	ND	ND	ND	ND	ND
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	ND	ND	ND	ND	41	ND	ND	ND
13/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	404	ND	ND	ND	ND	808	ND	ND
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500								
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ								
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB114m						ND		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB118m						ND		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB122m						ND		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB126m						ND		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1						ND		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB314m						635		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB318m						688		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB322m						687		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB326m						666		
19/09/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3						800		

Table 80 Bacteriology – 2014-2016

Date	Exp. name	Sampling point	Sampling point	TOT.BACT.ST.PLATE COUNT (TOTB)	COLIF. CONFIRMED (CMPN)	FECAL COLIF. MPN (FMPN)	STREP. F. CONFIRMED (EMPN)
				cfu/1 mL	MPN/100mL	n/100mL	MPN/100mL
9/9/14	H2O2 (~20)+Ozo (10)	Secondary Effluents	OZA500	7.E+04	1.E+03		
9/9/14	H2O2 (~20)+Ozo (10)	After Biofilter	OZOASF	7.E+04	1.E+03		
9/9/14	H2O2 (~20)+Ozo (10)	After Ozonation	OZOAOZ	1.E+00	1.E+00		
9/9/14	H2O2 (~20)+Ozo (10)	Before Infiltration	OZDWIN	3.E+04	2.E+01		
27/1/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500	2.E+05	2.E+04	7.E+03	2.E+03
27/1/15	H2O2 ~27 mg/L	After Biofilter	OZOASF	1.E+05	2.E+03	1.E+02	7.E+00
27/1/15	H2O2 ~27 mg/L	Before Infiltration	OZDWIN	1.E+04	5.E+03	8.E+02	3.E+01
27/1/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	2.E+04	2.E+00	ND	ND
27/1/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	4.E+04	2.E+00	ND	ND
23/3/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	3.E+03	4.E+00	ND	2.E+00
23/3/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	4.E+03	4.E+00	ND	ND
28/4/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500	6.E+04	5.E+04	4.E+04	5.E+03
28/4/15	H2O2 ~27 mg/L	After Biofilter	OZOASF	3.E+04	9.E+03	9.E+03	2.E+03
28/4/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	1.E+04	5.E+00	2.E+00	
28/4/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	7.E+03	ND	ND	ND
2123/06/2015	H2O2 ~27 mg/L	Ceramic cups no. 2+3	OZMAS1,2,3	5.E+05	2.E+01	2.E+01	ND
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	2.E+05	2.E+05	2.E+05	9.E+03
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	6.E+04	2.E+04	4.E+04	2.E+03
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	3.E+04	5.E+00	2.E+00	ND
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	2.E+02	ND	ND	ND
18/08/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	8.E+03	ND	ND	ND
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	8.E+04	2.E+05	1.E+03	2.E+04
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	4.E+04	2.E+03	5.E+02	2.E+03
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	5.E+02	2.E+01	8.E+00	2.E+00
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	8.E+02	ND	ND	ND
06/10/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	3.E+03	ND	ND	ND
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	8.E+02	ND	ND	ND
15/12/2015	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	3.E+03	ND	ND	ND
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	8.E+03	2.E+04	2.E+04	5.E+02
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	1.E+05	2.E+04	2.E+04	2.E+04
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	8.E+02	5.E+01	8.E+00	ND
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	3.E+03	ND	ND	ND
26/07/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	4.E+03	ND	ND	ND
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Secondary Effluents	OZA500	1.E+05	2.E+05	1.E+05	5.E+04
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After Biofilter	OZOASF	3.E+04	2.E+04	2.E+04	2.E+04
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	After ozonation	OZOAOZ	9.E+02	2.E+01	4.E+00	ND
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 1	OZOOB1	2.E+04	ND	ND	ND
16/08/2016	H2O2 ~27 mg/L+ozone 10 ppm	Observation Well 2	OZOOB3	6.E+03	ND	ND	ND

Table 81 TSS, DSS, Alkalinity, Cl^-, Hardness, Ca, Mg – 2014-2015

Date	Exp. name	Sampling point	Sampling point	SUSPE0ED MATTER AT 105°C (SS10)	SUSPE0ED MATTER AT 550°C (SS55)	DISSOLVED MATTER AT 105° C (DS10)	DISS.MATTER AT 550°C (DS55)	ALKALINITY M AS CACO3	Chloride as Cl	HARDNESS AS CACO3	MAGNESIUM AS MG	CALCIUM AS CA
				mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
9/9/14	H2O2 (~20)+Ozo (10)	Secondary Effluents	OZA500	2.9	0.5	0.7	0.7	224		228.916	16.6	64.3
9/9/14	H2O2 (~20)+Ozo (10)	After Biofilter	OZOASF	0.7	0.3							
9/9/14	H2O2 (~20)+Ozo (10)	After Ozonation	OZOAOZ	0.9	0.4							
9/9/14	H2O2 (~20)+Ozo (10)	Before Infiltration	OZDWIN	0.7	0.3	1	0.9	198		226.831	16.7	63.3
30/9/14	H2O2 ~27 mg/L	Observation Well 1	OZOOB1	1.7				223		265		79
30/9/14	H2O2 ~27 mg/L	Observation Well 2	OZOOB3	1.5				234		348		89
28/4/15	H2O2 ~27 mg/L	Secondary Effluents	OZA500					229	216			
28/4/15	H2O2 ~27 mg/L	After Biofilter	OZOASF					207	216			
28/4/15	H2O2 ~27 mg/L	Before Infiltration	OZDWIN									
28/4/15	H2O2 ~27 mg/L	Observation Well 1	OZOOB1					243	180			
28/4/15	H2O2 ~27 mg/L	Observation Well 2	OZOOB3					226	206			