AquaNES

Demonstrating Synergies in Combined Natural and Engineered Processes for Water Treatment Systems

Deliverable D3.3

Design recommendations for combining constructed wetlands with engineered pre- or post-treatments.
Including case studies of demonstration sites

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List of abbreviations

A	Anthracite
AC	Active Carbon
ACE	Acesulfame
AquaNES	Demonstrating synergies in combined natural and engineered processes for water treatment systems
ATS	Amidotrizoate
BAC	Biologically activated carbon
BEZ	Bezafibrate
BOD_5	Biochemical oxygen demand
BOF	Blast oxygen furnace
BTA	Benzotriazole/ 1H Benzotriazole
BV	Bed Volume(s)
BWD	Bathing water directive
CAN	Candesartan
CBZ	Carbamazepine
CFU	Colony forming unit
CLA	Clarithromycin
cNES	Combined natural and engineered treatment system
COD	Chemical oxygen demand
CSO	Combined sewer overflow
CW	Constructed wetland
D	Deliverable
DCF	Diclofenac
DOC	Dissolved organic carbon
DSS	Decision support system
DWA	German Association for Water, Wastewater and Waste
EBCT	Empty bed contact time
EU	European Union

FAA	Formylaminoantipyrine
FWS	Free Water Surface
GAB	Gabapentin
GAB-LA	Gabapentin lactam
GAC	Granular activated carbon
HHCB	Galaxolide
HLR	Hydraulic loading rate
HRT	Hydraulic retention time
HSSF	Horizontal subsurface flow
IOP	Iopamidol
Kf	Saturated water conductivity
LOD	Limit of detection
LOQ	Limit of quantification
MEC	Mecoprop
MEF	Metformin
MET	Metoprolol
MPN	Most probable number
N	Nitrogen
NDMA	N-Nitrosodimethylamine
NTU	Nephelometric Turbidity Units
OLM	Olmesartan
OM	Organic Matter
OMP	Organic Micropollutants
OXI	Oxipurinol
P	Phosphorus
PAC18	Poly-aluminum chloride 18 %
PE	Population Equivalents
PFOS	Perfluorooctanesulfonic acid
POC	Particulate organic carbon
PRI	Primidone
RSF	Retention soil filter
S	Sand
SF	Surface flow
SMX	Sulfomethoxazole
SSF	Subsurface flow

STL	Sotalol
tBV	Treated bed volume(s)
TCPP	Tris-(chloroisopropyl)-phosphate
TER	Tertbutryn
TKN	Total Kjedahl Nitrogen
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphor
TSS	Total suspended solids
UF	Ultrafiltration
UV	Ultraviolet
UVA	Ultraviolet absorption
UVA_{254}	Ultraviolet absorption at 254 nm
ΔUVA_{254}	Ultraviolet absorption at 245 nm elimination
VAL	Valsartan
VALA	Valsartan acid
VF	Vertical flow
WFD	Water framework directive
WHO	World Health Organisation
WP	Work package
WWTP	Wastewater treatment plant

Executive Summary

This design recommendation is a product of WP3 of the EU funded AquaNES project that demonstrates the combination of constructed wetlands (CW) and other natural treatment systems with different technical post- or pre-treatment options such as activated sludge systems, ozonation or disinfection in pilot and full scale sizes in different European climates (UK, Germany, Greece).

In this document, the design data and results from five AquaNES demo sites of WP3 are summarised and combined with existing guidelines for CWs.

CWs are implemented as primary, secondary or tertiary treatment of wastewater and combined sewage overflow. The first part discusses CWs as part of combined systems and outlines possible combinations with engineered pre- and post-treatments. The different implementations of CWs as primary, secondary or tertiary treatment as well as the dual CSO plus tertiary treatment use are highlighted. Important design parameters, in particular the German set of rules DWA A-262 (2017) for dimensioning, construction and operation of planted filters for municipal wastewater treatment, are summarised and combined with the results gained from the demo sites and experiences by the authors.

The second part presents the five AquaNES demo sites in Greece, Germany and the UK, their different fields of application, design and operational data as well as monitoring results. The two Greek sites, Antiparos and Thirasia wastewater treatment plants (WWTP), are both located on touristic influenced islands in the Aegean See. The main focus of the solution in Antiparos (AquaNES site 10b) was to create a robust, near-natural secondary treatment system that copes with the fluctuating hydraulic and pollution load and meets the Greek standard for reuse by restricted irrigation. During the AquaNES project this treatment plant was subject to further optimization.

On the Greek island Thirasia (AquaNES site 10a) a multiple treatment system has been built for demonstrating the combination of a horizontal subsurface flow (HSSF) CW with TiO_2-activated pretreatment and subsequent ultrafiltration and chlorination for water reuse. Other than planned the wetland was performing as surface/subsurface flow system with the main function of ammonium removal. Nevertheless, TN concentration in the WWTP effluent was the only parameter exceeding the Greek limits for reuse (45 mg/L). Testing the photo-catalysis unit with various TiO_2 dosages and without any chemical dosing but only aeration revealed that the TiO_2 had no additional treatment effect. Thus, the combination of activated sludge treatment (aeration plus sedimentation) and the CW would be the most economical solution for this treatment plant.

On the demo site 11 Rheinbach, Germany, the flexible treatment of combined sewer overflow (CSO) plus secondary effluent polishing during dry weather was demonstrated by three pilot scale retention soil filters (RSF). RSF has proven to be the most efficient and economic technology for CSO treatment because it integrates storage capacity at low additional cost. The flexible use of this natural treatment system for regular post treatment of secondary effluent during dry weather plus periodical flooding with untreated combined sewage overflow increases the economic value many fold. The long term study (> 3.5 years) on OMP removal showed varying performance depending on the nature of the OMPs. Particularly the addition of granular activated carbon (GAC) within the filter increased DOC and OMP reduction. In the effluent of the pilot-RSF with GAC only few OMPs were detected above limit of quantification (LOQ); whereas the removal of OMPs in the pilot-RSF without GAC varies from 0-80 %. Beside adsorption biological degradation was detected to be a major pathway for degradation of OMPs. Furthermore, pathogens removal (E. coli and Coliforms) was efficient

with 1-2 log-units. These positive results have led to the construction of a full scale combination for dual CSO plus tertiary treatment use for 27,000 PE called *RSF^plus^*.

At the WWTP Schönerlinde (AquaNES site 12), north of Berlin, Germany, two pilot vertical flow (VF) CWs combined with preceding ozonation were tested for removal of organic micropollutants (OMPs) and pathogens in municipal secondary effluent. Results from the trial at Schönerlinde, show that ozonation and CW treatment is a suitable combination to remove organic and microbial contamination. Synergy of the process combination could be clearly shown for removal of organic matter, comparing ozone and subsequent CW treatment (DOC removal of 21-22 %) with CW as a stand-alone solution (DOC removal of 4-9 %). OMPs were mainly reduced by the ozonation step; whereas the CWs fulfill a primary function of the post-treatment: the removal of biodegradable organic transformation products formed by ozonation. Also disinfection was improved by CW post-treatment. After ~2 log-units reduction of *E. coli* and Enterococci during ozonation they further decreased below LOQ in CW treatment. *C. perfringens* and somatic coliphages were insufficiently inactivated by ozone. CW post-treatment effectively retained both parameters and hence, compensated the short-comings of the ozone treatment. The process combination of ozone and CW works for a wider range of microorganisms and therefore provides higher disinfection safety. Both filter media (CW1: sand; CW2: lava gravel/biochar) were effective for this purpose.

The AquaNES site 13 in Packington (UK) demonstrated a steel slag reactive media CW following conventional biological treatment for the removal of phosphorus (P) from municipal wastewater. This long term demonstration scale trial of a reactive media CW has first confirmed the potential of the technology for the removal of P to low levels as a single step for small WWTPs. However, a number of limitations highlighted by the trial mean that the technology is not yet ready for full scale application. In praxis the high effluent pH and more importantly the release of vanadium and the relatively short life span (circa 1 year) of the media to maintain low P effluent concentrations (< 2 mg/L) are critical bottlenecks towards implementation in technical scale. The use of steel slag for water treatment in praxis is therefore not recommended.

This deliverable is directly linked to deliverable 3.1 - Combining constructed wetlands and engineered treatment for water reuse and deliverable 3.2 - Combining constructed wetlands and engineered treatment for surface water protection.

1 Introduction

At the beginning of the technical development of water purification in general and wastewater treatment in particular, there were natural treatment processes only. This means the so-called self-cleaning capabilities of nature have always been used. These include facultative ponds, soil infiltration with agricultural utilisation, bank filtration or slow sand filtration in river water treatment.

With industrialisation and increasing urbanisation, water purification became more complex due to hygienic and environmental quality targets. As a result, however, the technical development of wastewater treatment processes also made enormous progress. In the industrialised countries, natural processes were partly forgotten.

In the 1980s Germany and France began to develop vegetated soil filters for wastewater treatment. These are also known as reed bed filters, constructed wetlands (CW), or treatment wetlands and initially only had a niche existence in treatment technologies. In the meantime, natural treatment has developed and become mainstream in some areas and countries. This applies in particular to the so-called French system, which is used widely in France for the treatment of wastewater from combined and separated sewers for settlements under 2,000 inhabitants and often replaces larger lagoons. The largest such constructed wetland is located in Orhei, Moldova. The installed capacity is 20,000 PE.

With increasing experience and technical development of natural treatment systems these have also been optimised for special applications. That includes the treatment of industrial wastewater, combined sewer overflows, stormwater runoff, landfill leachate and agricultural drainage. Each technology has its own advantages but also its own limits. In wastewater treatment, it is common to combine processes in order to meet the overall treatment performance requirements.

The combination of natural and engineered systems (cNES) is not completely new, however, one of the tasks within the research and demo project AquaNES was to show its implications and advantages which may arise. Practice shows that misunderstandings about the functioning of wetlands lead to suboptimal integrated solutions without synergy effects.

Increasingly stringent environmental quality standards are driving the search for new solutions and adding tertiary treatment stages to conventional systems. Those are aimed at enhanced nutrient removal, the elimination of micropollutants or the provision of hygienically safe water, e.g. for reuse.

In general, technical wastewater treatment plants are mostly planned by civil and process engineers who have only limited experience in the application and planning of natural systems. There is a great uncertainty about the state of the art. Also failures have been reported in the past. As a consequence, combined natural and engineered systems are often not considered.

On the other hand, the potential benefits from cNES shall be pointed out for a wider use:

– Improvement of the robustness/ resilience/ reliability/ flexibility of the (combined) system even with (seasonal) fluctuations of hydraulic and organic loading due to the combination;
– Reduced operational costs;
– Reduced demand in consumables (less/ no use of chemicals, less use of fossil fuels/electricity);
– Providing habitat and other extra ecosystem services;
– This giving social benefit (no visual obstruction, but green infrastructure, urban landscaping and potential recreational use).

2 Purpose and use of this document

This document is intended to fill the gaps in knowledge by describing practical experience gained in the context of the AquaNES project or in practice and by using relevant literature. It is intended to help its users to utilize the respective advantages of natural systems safely and to combine them purposefully with conventional and innovative technical processes.

2.1 Who should use this document

Making use of this report is recommended for designers and operators of wastewater treatment plants with combined natural and engineered systems.

2.2 What must be taken into account when using these recommendations?

Adhering to the technical advice provided in this document does not relieve any professional from being responsible for their own actions or the correct application in a specific case. This applies in particular to the proper handling of the scope and margins identified in the guideline.

A fully reasonable decision and design can only be made if all technology specific assessment data and the local conditions (e.g. climate, availability of filter media, quality and amount of wastewater/ discharge hydrograph) are available.

It is recommended to use further information on treatment wetlands e.g.:

- Dotro, G., Langergraber, G., Molle, P., Nivala, J., Puigagut, J., Stein, O., Von Sperling, M. (2017), Treatment wetlands. Biological Wastewater Treatment Series, Vol. 7, IWA publishing.
- DWA-A 262E (2018), Principles for Dimensioning, Construction and Operation of Wastewater Treatment Plants with Planted and Unplanted Filters for Treatment of Domestic and Municipal Wastewater (November 2017), (English version). German Association for Water, Wastewater and Waste (DWA), Hennef, www.dwa.de.
- Hoffmann, H., Platzer, C., von Münch, E. and Winker, M. (2010), Overview of subsurface flow constructed wetlands for greywater and domestic wastewater treatment in developing countries. Sustainable Sanitation and Ecosan Program of Deutsche Gesellschaft Für Technische Zusammenarbeit (GTZ) GmbH, Germany.
- Kadlec, R. H., and Wallace, S. (2009), Treatment wetlands, 2nd ed. CRC/Taylor & Francis Group, Boca Raton, ISBN 978-1-56670-526-4, USA.
- Lombard Latune, R., and Molle, P. (2017), Les filtres plantés de végétaux pour le traitement des eaux usées domestiques en milieu tropical: Guide de dimensionnement de la filière tropicalisée. Agence Française pour la biodiversité, collection Guides et protocoles.
- Payne, E., Hatt, B. E., Deletic, A., Dobbie, M. F., McCarthy, D. T., Chandrasena, G. I. (2015), Adoption guidelines for stormwater biofiltration systems. Cooperative Research Centre for Water Sensitive Cities, Melbourne, Australia.
- UN-HABITAT. (2008), Constructed Wetlands Manual. UN-HABITAT Water for Asian Cities Programme Nepal, Kathmandu.

This document can give no detailed instructions other than basic information that has to be considered.

Utilities and operators should always rely on constructors who can show relevant and successful references in this field. In any case experienced consultants/designers should be involved if a new technology shall be implemented for the first time. However this is true for all technologies but natural systems are often mistaken as too simple.

2.3 What are these recommendations based on?

The information in this document is primarily based on conclusions drawn from the demo sites of AquaNES. Useful and possible additional knowledge has been added from other sources such as recent wetland guidelines, literature or other experience by the authors.

The technical scope in this document covers subsurface flow wetlands only.

Within AquaNES improved wastewater treatment systems combining wetlands with different technical post- or pre-treatment options such as activated sludge systems, ozonation or disinfection have been investigated. Demonstration sites are located in different European climates.

The following sites have been studied:

a. wetland as secondary treatment system with photo-catalytic pre-treatment (Thirasia, Greece),

b. wetland as secondary treatment plus chemical disinfection (Antiparos, Greece),

c. wetland used as flexible treatment for combined sewer overflow (CSO) and for tertiary treatment of Wastewater Treatment Plant (WWTP) effluent including removal of organic micropollutants (OMP) (Rheinbach, Germany),

d. wetland as tertiary treatment stage in combination with ozonation (Berlin, Germany),

e. wetland using reactive filter media for phosphorus removal as tertiary process (Packington, UK).

Technical results and discussion with operators as well as information from previously published data on wetland-based cNES (Combined natural and engineered treatment system) have been evaluated and result in these 'Design recommendations for combined systems'.

3 Constructed wetlands as part of combined systems

3.1 Treatment wetlands concept

Constructed wetlands are near-natural systems used for treating raw, primary or secondary treated sewage in a sustainable and environmentally friendly way.

Surface flow (SF) wetlands also named Free Water Surface (FWS) wetlands are aquatic systems, making use of macrophytes for enhanced biological activities and sedimentation processes. FWS CWs are densely vegetated units with areas of open water (see Figure 1). They resemble natural marshes and are commonly implemented for tertiary wastewater treatment. This wetland type is typically low loaded and therefore requires large areas. (Dotro et al., 2017; Kadlec and Wallace, 2009)

Subsurface flow wetlands (SSF) consist of basins filled with substrates for biofilm growth and filtration, usually sand or gravel, and are planted with wetland vegetation tolerant of saturated conditions (most often common reed, reed canary grass, and cattail, iris, reed sweet grass, papyrus). Their special design and the high biological activity in wetland ecosystems result in an increased treatment capacity compared to FWS. (Dotro et al., 2017; Kadlec and Wallace, 2009; Rozkošný et al., 2014; UN-HABITAT, 2008)

3.1.1 Hydraulic schemes

Most common treatment wetlands are divided into hydraulic functional groups. These are horizontal surface flow (= free water surface FWS), horizontal subsurface flow (= HSSF) and vertical flow (= VF, it is always subsurface). See figures Figure 1 and Figure 2 with flow direction indicated.

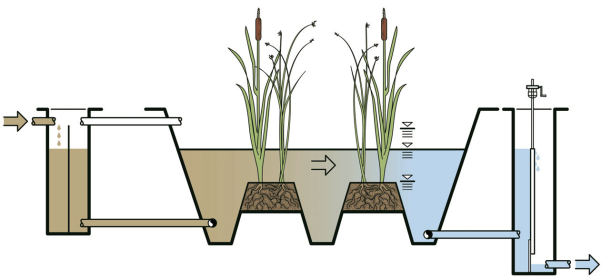


Figure 1 Overview schematic of FWS wetland (Dotro et al., 2017).

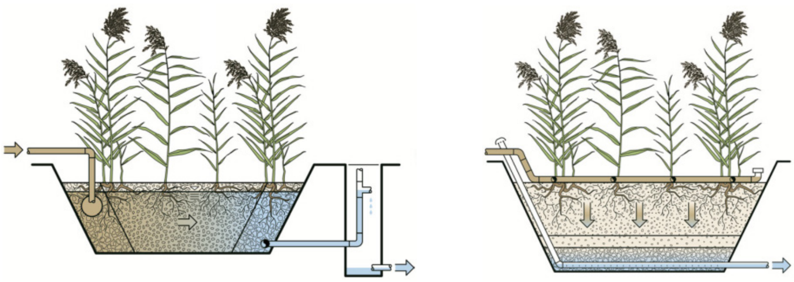


Figure 2 Overview schematics of HSSF (left) and VF (right) constructed wetland (Dotro et al., 2017).

The water level in HSSF wetlands is regularly kept below the surface of the filter medium. The water flows horizontally from the inlet to the collection zone. In VF systems there may be temporary surface flow for water distribution. During the treatment process the water percolates downstream through the plant root zone (see Figure 2).

HSSF systems can be operated fully or partly saturated with water and VF wetlands may be used with saturated flow or they are partly or fully drained.

Water saturation conditions are relevant for oxygen transfer. During horizontal flow mainly anaerobic processes occur if flowrate and water table are constantly high. VF wetlands are dominated by aerobic processes because of intermittent batch loading and draining. The fluctuating water table causes a natural ventilation of the filter pores (convection). Diffusive air transport and oxygen release by plant roots have comparative low effect in such systems.

Also the flow direction in subsurface systems plays a minor role for the treatment function. But it is very relevant for the calculation of head losses and water distribution. This fact is often neglected, then leading to unsuitable hydraulic designs, mainly with horizontal flow.

3.1.2 Function

The basic principle of SSF treatment is the flow of wastewater through the porous filter media which serves as substrate for biofilm. Pollutants are removed by several complex physical, chemical and biological processes. Aerobic degradation mechanisms play an essential role for BOD_5 reduction and nitrification. Aerobic degradation depends on the relation between oxygen demand (load) and oxygen supply (Hoffmann et al., 2010). In case of limited oxygen supply distinct oxidation and reduction zones will be built in the bio filter.

Vegetation may play a vital role in the treatment process by providing surfaces and a suitable environment for microbial growth and enhanced sedimentation in FWS wetlands. In subsurface flow wetlands the continuous growth of underground biomass such as roots, rhizomes and new sprouts maintain a permeable filtration layer especially in VF systems. (Dotro et al., 2017; Kadlec and Wallace, 2009; Rozkošný et al., 2014; UN-HABITAT, 2008) Plants may not be considered for nutrient removal in secondary wastewater treatment. The overall contribution to this may be around 5 % of the influent annual nutrient load as typical balances show.

In general a treatment wetland must fulfil any pre-defined requirements in terms of hydraulic conductivity and load of wastewater by pollution and flow rate or the capability to bind phosphorus and heavy metals. Filter media must stay permeable for the lifetime to avoid clogging and subsequent surface flow (Rozkošný et al., 2004; UN-HABITAT, 2008). In gravel based HSSF systems biomass growth and sedimentation within the pores may become a problem and reduce the hydraulic cross sectional area. Pretreatment and other combinations always have to be taken into account.

Table 1 Pollutant removal mechanisms in CWs (adapted from Dotro et al., 2017; UN-HABITAT, 2008)

Wastewater constituents	Removal mechanisms
Suspended solids	– Sedimentation – Filtration
Soluble organics	– Biological degradation (aerobic and/or anaerobic)
Phosphorous	– Adsorption-precipitation reactions – Plant uptake (usually of minor relevance)
Nitrogen	– Ammonification followed by microbial nitrification and denitrification – Plant uptake (usually of minor relevance) – Matrix adsorption (temporary) – Ammonia volatilization (mostly in FWS CWs)
Metals	– Adsorption and cation exchange – Complexation – Precipitation – Plant uptake – Microbial oxidation/reduction
Pathogens	– Physical retention – Natural die-off – Predation – Ultraviolet (UV) irradiation (in FWS CWs) – Excretion of antibiotics from roots of macrophytes
Micropollutants	– Biological degradation – Sorption – Photodegradation (in FWS CWs)

3.2 Constructed wetlands as primary treatment (TSS, COD, TKN)

SSF wetlands for primary treatment also known as “Raw Wastewater Filtration” provide an integrated sludge and wastewater treatment. They are widely used as first stage of the French system (see Figure 3 and chapter 3.2). The first stage relies on the development of a secondary organic filter layer on the filter surface. Filtration of suspended solids and biological processes in this deposit layer are intense. (Dotro et al., 2017; Hoffmann et al., 2010; Kadlec and Wallace, 2009)

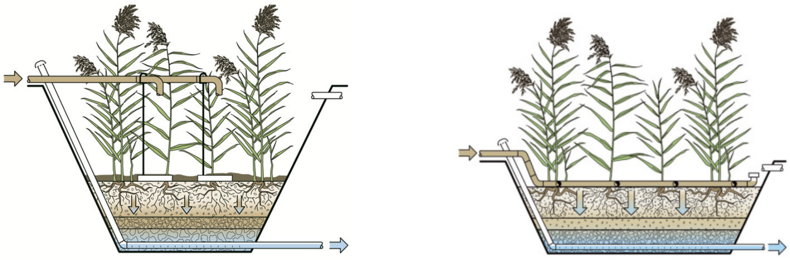


Figure 3 Two-stage French system (left: first stage, right: second stage) (Dotro et al., 2017).

In raw wastewater filtration sludge stabilization and a high level removal of organic matter (OM) and partly nitrification occurs (Dotro et al., 2017). In contrast to conventional pre-treatment (e.g. Imhoff tank with anaerobic sludge stabilization within 90 days) sludge volume is drastically reduced due to long term aerobic mineralization (composting) and drying on the relatively large filter surface. The treated sludge accumulates at a rate of approximately 2-3 cm per year when the system is operated at design load (Dotro et al., 2017; Lombard Latune and Molle, 2017). Only every 10 years the organic top layer has to be removed. In fact, this secondary organic filtration layer plays a vital role in the treatment process itself (see Figure 4 right). A combination of raw wastewater filtration with mechanical pre-treatment like screens, sand and grease traps is feasible and common practice (see Figure 4 left).

Figure 4 Compact screen and grit trap (*KUHN headworks unit KKA*, Aveleira, Portugal) (left). Raw sewage distribution and filtration at 1st stage of a “French system” (Aveleira, Portugal) (right).

Typically, the first stage filter is divided into three segments to ensure an alternated feeding of the cells with a resting period twice as long as the operation phase, e.g. 6 to 14 d resting and 3 to 7 d feeding. Feeding occurs batch wise with a hydraulic loading rate (HLR) of 20-50 L/m² per batch. Maximum HLR amounts to 250 L/(m²*d) considering the total filter surface. For water distribution, gravity flow or pressure pipes may be used with at least one feeding point per 50 m². (Dotro et al., 2017; DWA, 2018)

In order to always maintain aerobic conditions sufficient underground ventilation must be achieved. The choice of the filter media is one of the important details for correct functioning of the filter. To ensure aerobic conditions and to prevent clogging the main filter layer should consist of fine gravel as shown in Table 2. (Dotro et al., 2017; DWA, 2018).

Table 2 Filter media specifications for a raw wastewater filtration (DWA, 2018) and 1st stage of a French system (Dotro et al., 2017).

	Layer Depth	Media DWA, 2018	Media Dotro et al., 2017
Freeboard	≥ 0.3 m		
Sludge storage	≥ 0.2 m		–
Main layer	≥ 0.3 m	Fine gravel 2-8 mm	Fine gravel 2-6 mm
Transition layer	≥ 0.1 to 0.2 m	Gravel 16-32 mm	Gravel 5-15 mm
Drainage layer	≥ 0.2 m	Coarse Gravel 32-56 mm	Coarse Gravel 20-60 mm

The dimensioning is determined by maximum pollutant and hydraulic loads (see Table 3). In a temperate climate, for regions with separated sewer networks this requires a specific area of ≥1.2 m²/PE with 3 cells of ≥ 0.4 m²/PE each; for combined sewer networks ≥1.5 m²/PE are recommended (DWA, 2018). In the tropical zone, the specific filter area can be reduced to 0.8 m²/PE and the number of cells to two alternating filter beds (Lombard Latune and Molle, 2017). The specific area requirements per PE indicate a rule of thumb; the final dimensioning must be based on the hydraulic and pollutant load. In doing so, the parameter with the largest area requirement is relevant.

Table 3 Maximum design loads for raw wastewater filtration (DWA, 2018) and for the first stage of a French VF wetland (Lombard Latune and Molle, 2017) under dry weather conditions in temperate zones. Values refer to total filter surface of all parallel cells.

	HLR [L/(m²*d)]	HLR per batch [L/m²]	COD [g/(m²*d)]	BOD5 [g/(m²*d)]	TSS [g/(m²*d)]	TKN [g/(m²*d)]
DWA, 2018	250	20-50	100	-	-	-
Lombard Latune and Molle, 2017	250	20-50	117	50	50	10

By proper design and operation of the raw wastewater filter, high removal performance concerning organics and Total Suspended Solids (TSS) can be achieved. The performance capacity of the 1st stage of a French VF CW determined by Molle et al (2005) indicates a good removal for Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD_5) and TSS of 0.80*M_i, 0.90*M_i and 0.90*M_i, respectively (M_i: mass load in g/(m²*d)).

Lombard Latun and Molle (2017) found a higher organic removal rate and higher TKN removal rate with an increased filter layer depth of 80 cm. This was more effective than introducing recirculation on the 30 cm filtration layer (see Table 4).

In warm climates this first wetland stage with increased depth of layer may be sufficient if not denitrification or any other advanced treatment is necessary. In combination with disinfection the treated water may be ready for agricultural reuse.

Table 4 **Summary of expected removal rates and characteristics of different variants of the 1st stage French VF CW in tropical zones (Lombard Latune and Molle, 2017).**

Variant of 1st stage French VF CW	Removal rate [%]			
	COD	BOD_5	TKN	TN
30 cm filter layer	75	80	60	20
30 cm filter layer + recirculation	75	85	60	20
80 cm filter layer	90	90	80	20

3.2.1 cNES – Primary Treatment Wetlands Recommendation

Using CWs as primary treatment is a robust basic solution for domestic sewage and high strength wastewater treatment, which can be combined with almost any other process. Its advantages are especially very low operational costs, due to its substitution of costly sedimentation and sludge handling and further treatment. The raw wastewater filter is state of the art for primary treatment in remote areas. For enhanced treatment it may be combined with the following options.

The following combinations are common and have not been demonstrated in the AquaNES project.

Engineered pre-treatment step	Requirement
Pre-treatment	– Screens, sand and grease traps are possible (not necessary) pre-treatment options. Grease trap is useful in special cases (e.g. industrial influence). – Grease or oil, e.g. in Mediterranean countries originating from indirect discharge by olive-oil production, can lead to filter clogging and must be removed before entering the wetland[1]. – Suspended solids play an important role in sludge accumulation and build-up of a secondary filter layer and thus should not be removed beforehand.
Phosphorus precipitation	– Generally possible in front of a VF treatment wetland with sludge accumulation layer. Specific conditions such as increasing TSS load have to be explored depending on dosing chemical and taken into account when dimensioning. Effluent concentrations of 2 mg/L P are realistic. – Experiments by other authors clearly show that the aerobic conditions need to be maintained in the whole filter when ferric chloride is used (e.g. Boram et al., 2014). Otherwise reductive dissolution of P-bearing ferric species is induced and sulfides are formed. This means, the VF filter must always be completely drained. – There is no experience with use of aluminium salts for phosphorus precipitation in wetlands but it should work as well. No negative effects for plants were observed with *P. australis* in laboratory experiments and mining wastewater (e.g. Batty et al., 2002).
Engineered post-treatment step	**Requirement**
Enhanced oxidation and nitrification/ denitrification	– A second filtration system or any other biological treatment or mechanical clarifier may follow the raw wastewater treating wetland depending on COD and TSS limits. Standard is the use of a second stage VF wetland, known as two-stage VF French system. Batch wise aerated processes such as technical bio filters or aerated wetlands may be used for achieving a nearly complete nitrification and enhanced denitrification. – If denitrification is desired the primary treatment wetland should mainly be used for sludge removal and leave enough carbon for the following processes. This means the thickness of the gravel filter layer may be reduced.
UV disinfection	– Because of the very low turbidity of the filtrate UV can be applied for disinfection, nevertheless low transmittance of the primary treated effluent due to COD values > 50 mg/L may require high UV radiation. – In case of iron dosing to the VF the iron must be dosed carefully. Otherwise remaining soluble iron would be able to absorb UV radiation and reduce its disinfecting effect.
Chlorine disinfection	– Chlorine should only be used after a thorough (complete) BOD_5 and ammonia removal and is not recommended after a one stage raw wastewater treatment wetland.

[1] Experience by authors

3.3 Constructed wetlands as secondary treatment stage (BOD_5, COD, TKN)

CWs as main biological treatment step of domestic and municipal wastewater (secondary treatment) exist in a wide range of variations. Due to the development and efficiency of VF systems compared to horizontal subsurface flow wetlands HSSF have been widely substituted. The new German DWA guideline (2018) does not recommend use of HSSF for secondary treatment purposes any longer.

Three major VF wetland systems are recommended: A) sand filtration, B) two stages gravel bed systems, C) actively aerated gravel beds. These systems are well described in DWA-A 262E (2018).

Two possible combinations have been shown at the AquaNES demonstration site in Antiparos (see 4.1). A two-stage vertical flow wetland has been combined with Imhoff tank and disinfection. Additionally, a small pilot system with aerated VF filter was tested.

Sand filter media have at least a 10 times larger specific substrate surface than fine gravel. Fine media reduce infiltration speed but improve nitrification rates and carbon oxidation. At the same time the clogging risk increases because of fine pores. If these attached-growth biofilm wetlands get too high organic areal load the biofilm growth easily fills the pores. Alternated loading of parallel filters is recommended to avoid clogging of filter pores.

For that reason, pre-treatment reducing solids and soluble organic matter is absolutely required. VF treatment wetlands as main stage can be combined with primary treatment systems like raw wastewater filtration (see 3.2), settling ponds or Imhoff tanks.

Common VF systems are free drainage filters. They consist of a filtration layer of at least 50 cm depth, a drainage layer of a minimum of 20 cm and, if necessary, a transition layer. Feeding of the pre-treated wastewater is batch wise, so that air convection into the pores is possible during resting period. The percolate is collected by a drainage system.

An important design parameter is the filter media characteristics which influences directly the treatment efficiency: Finer media result in higher hydraulic retention time (HRT) and hence often in higher removal efficiencies; coarser filter media have less clogging potential and allow higher HLRs resulting in lower retention time and less removal efficiencies (Dotro et al., 2017). Increasing the depth of the main layer is one of the possible methods to tackle this shortcoming. In Table 5 filter media specifications for the two systems are summarized.

Table 5 Filter media specifications for a secondary A) VF CW with sand and B) a two-stage VF system with fine gravel and coarse sand (DWA, 2018).

	A) VF CW with Sand		B) Two-stage VF CW			
			First stage with Fine Gravel 2-8 mm		Second stage with Coarse Sand 0-4 mm	
	Depth	Material	Depth	Material	Depth	Material
Freeboard	≥ 0.3 m		≥ 0.3 m		≥ 0.3 m	
Filtration layer	≥ 0.5 m	Sand 0.2 -2 mm; (kf ≈ 10^{-4} m/s)	≥ 0.5 m	Fine Gravel 2-8 mm; (kf ≈ 10^{-1} m/s)	≥ 0.5 m	Coarse Sand 0.63-4 mm; (kf ≈ 10^{-3} m/s)
Transition layer	–	–	–	–	≥ 0.1 m	Fine Gravel 2-8 mm
Drainage layer	≥ 0.2 m	Fine Gravel 2-8 mm	≥ 0.3 m	Gravel 16-32 mm	≥ 0.2 m	Gravel 16-32 mm

The key design parameters for the sand filter and the two-stage filter system are the specific HLR and the COD or BOD_5 loading rates (see Table 6). The specific surface area requirements after primary treatment are ≥4 m^2/PE for the sand filter and ≥1 m^2/PE for each of the two-stage gravel filters.

The removal efficiency of the VF CW as secondary treatment for COD and NH4-N amounts to >90 %; TN is reduced by a rate of <20 % and coliforms removal accounts for 2-4 log-units (Dotro et al., 2017; Geller and Höner, 2003).

Combined treatment concepts for advanced treatment

There are a number of possibilities to improve the VF CW performance: recirculation of the treated wastewater, the implementation of a saturated drainage layer or the use of artificial aeration. A following polishing step could be another method for further denitrification.

Figure 5 Facultative pond as primary treatment with anoxic settling zone and recirculation of nitrates from VF sand filtration wetland for denitrification. Effluent from the opposite side is separated by two floating island baffles (Kappe, Germany).

Upstream denitrification has proven itself in many plants where the outlet of the vertical stage can be led into the inlet of the primary treatment. Then an increased formation of floating sludge blankets can be observed in the first chamber of settling tanks.

The recirculation of effluent from the secondary treatment wetland only makes sense if treatment conditions in the primary treatment are favorable for the desired process. E.g. denitrification needs anoxic condition and carbon supply. This does not happen in relevant order of magnitude by recirculation into a first stage vertical flow wetland due to the aerobic conditions.

In any case where recirculation is planned, the wetland stage must consider the additional hydraulic loading rate. For denitrification purposes a recirculation rate of more than 2 (= 200 % regular flow) has not shown to be useful in practice. A nitrogen reduction of 70 % can be expected by recirculation from secondary treatment wetlands to anaerobic primary treatment if nitrification in VF cells is almost complete (Rustige and Platzer, 2002).

It could be shown that volatile methane which is produced in anaerobic primary treatment systems may be collected from the gas phase in settling tanks and blown into the drainage layer of unsaturated VF secondary treatment systems. The methane was completely removed in the vertical flow filter. (Schalk et al., 2019).

Dimensions

The secondary wetland treatment stage will be designed according to the remaining organic load of the primary treatment. This load can be very low after a 1st stage French system (see 3.2). Also in combination with chemical precipitation/sedimentation methods as shown on demo sites Antiparos and Thirasia the expected remaining organic load is lower than from gravity sedimentation. Maximum organic and hydraulic design loads for VF CW with sand, two-stage VF CW and aerated VF CW according to DWA-A 262E are listed in Table 6.

Table 6 Maximum design loads for VF CW with sand, two-stage VF CW and aerated VF CW under dry weather conditions in temperate zones (DWA, 2018).
Values refer to total filter surface of all parallel cells.

		HLR [L/(m²*d)]	HLR per batch [L/m²]	COD [g/(m²*d)]
A) VF CW with Sand		≤ 80	≥ 20	≤ 20
B) Two-stage VF CW	1st stage with Fine Gravel		≥ 20	≤ 80
	2nd stage with Coarse Sand		≥ 20	
C) Aerated VF CW	With gravel (8-16 mm); 1 m depth			≤ 100

3.3.1 cNES – secondary treatment wetlands recommendation

Possible combinations of secondary treatment wetlands with post-treatment for further effluent polishing depend on the purpose and the effluent quality requirements. UV-disinfection or chlorination for enhanced removal of pathogens as well as HSSF CW for denitrification and removal of pathogens are possible post-treatment steps. Principally all existing physical or chemical treatment technologies could be combined with a secondary treatment wetland. The most relevant are listed here. Some of these combinations have been demonstrated in AquaNES demo sites. See chapters 4.1 and 4.2.

Engineered pre-treatment step	Requirement
Sedimentation as primary treatment	– Gravity sedimentation in tanks may not be used with combined sewer systems because sludge zone and sedimentation zone are not separated thoroughly. During high flow situations the sediments will be mixed and may be transported to the wetland stage. For small treatment systems a minimum volume of 300 L/PE is useful (DWA-A 262E). Max. 50 % is considered as sludge storage volume. – Settling ponds shall be designed by the specific surface area with a minimum of 1.5 m²/PE in order to ensure the necessary separation of TSS. Grits shall be removed beforehand and floating matter must be retained by means of a baffle. Open surface settling ponds may be the cause for bad odour development. These ponds may be covered by removable floating islands. – Imhoff tanks provide a separate sludge volume which allows higher flow rates in the settling zone. The minimum retention time for gravity settling is 2 hours. Sludge storage volume is designed for 90 days detention time. If chemical precipitation is planned the same rules as in 3.2.1 have to be considered. Recirculation from wetland to the primary Imhoff tank is not useful, since contact with sludge zone is not possible. – If sewage (and sludge) is delivered by trucks a flowmeter and a monitoring procedure is necessary. These deliveries can significantly increase the daily load. Since organic load to the wetland needs to be limited according to the allowable daily amount, the actual load to each cell should be monitored and used for adapting the feeding regime (pause/operation phases).
Anaerobic biological primary treatment	– Any primary treatment with long retention time will include biological decomposition to some degree. This applies for settling tanks and ponds where sludge compartments are not separated. – If not standard loads (e.g. DWA-A 262E) are used, measurements or other plausible data have to be used in each case. – Especially if recirculation from the wetland to the primary treatment is planned in order to make use of upstream denitrification, the additional hydraulic load has to be considered for each stage. For denitrification the recirculated water from the nitrifying wetland stage must be fed to the primary stage in a way that contact with sludge is achieved. Because of build-up of a sludge blanket on the primary tank, it has to be made sure that easy access to the water surface is possible for sludge removal. – If anaerobic primary treatment is combined with a VF wetland methane containing gas should be collected and treated in the wetland itself. When doing so a constant ventilation of the tank makes sure that no explosive atmosphere can arise. The wetland also serves as air filter and will reduce bad odours.
Aerobic biological primary treatment	– This combination sometimes is used for high strength organic wastewater treatment, such as runoff from organic waste handling grounds (biogas plants, compost works,…). In this case COD may have concentrations of more than 10 to 20 g/L. – For this purpose activated sludge systems are not designed for full treatment, reducing total volume and improving the energy balance in combination with a constructed wetland. – Only VF systems should be combined with this kind of primary treatment in order to prevent irreversible clogging in case of operational faults of the primary stage.
Photocatalysis, Coagulation and Sedimentation	– Enhanced sedimentation reduces the organic loading of the secondary wetland and may result in smaller wetland sites. – However, chemical treatment as primary is not a cost efficient solution for removing BOD_5 from municipal sewage as long as other standard solutions are available. In-

	stead intensified wetlands such as aerated systems effectively reduce the areal footprint. – Even in case of repeated use of chemicals (e.g. recirculation of TiO_2 as shown in Thirasia 4.2.1) chemicals will be lost to the sludge. – Only VF systems should be combined with this kind of primary treatment in order to prevent irreversible clogging in case of operational faults of the primary stage.
Engineered post-treatment step	**Requirement**
Polishing ponds	– For technical systems a natural pond is considered to improve the final effluent quality. Usually this is the case for denitrification and also for pathogen reduction if retention time is long. As could be shown at the demo site in Antiparos and many other places, secondary biomass/ algae growth is created. This is consuming chlorine and energy from any following disinfection method. In this case the last treatment stage should always be a subsurface flow wetland instead.
P-flocculation + granular media filtration	– When low P effluent concentrations are required combined P-flocculation and granular media filtration is a possible tertiary treatment step after a VF CW due to low DOC concentration of the filtrate.
Membrane filtration	– Membrane filtration can be applied for water reuse purposes. Important parameters to consider are TSS, scaling-causers iron, aluminium, calcium as well as fouling-causers fats, fibres and biomass (DWA, 2003). – Because of high removal efficiency concerning TSS and BOD_5 VF CW can be easily combined with membrane filtration. – Depending on filter media characteristics and pH, leaching of calcium and aluminium (at low pH) is possible. This should be taken into account when choosing the filter media. – In case of iron dosing to the VF the iron must be dosed carefully. Otherwise remaining soluble iron would lead to scaling of the membranes. – Chemicals are necessary to clean the membranes. Permeate disposal must be properly organised. – At the demo site in Thirasia an ultrafiltration unit was applied successfully (see D3.1).
Chlorination (Cl_2, NaOCl, ClO_2)	– Chlorination is a common post-treatment step for disinfection. – The secondary effluent should be transparent and should contain low amounts/no NH_4 and organic compounds which would consume the chlorine (DWA, 2003). This can be achieved by VF CW. – Application of chlorine gas requires high occupational safety for production, storage and dosing because of its high toxicity. NaOCl instead is easier to handle and comparatively harmless. (DWA, 2003) At the demo site in Antiparos disinfection with NaOCl showed to be an efficient technology. However, the supply with chemicals in remote locations can be difficult and inhibit continuous disinfection. In-situ electro-chlorination may overcome these shortcomings (see D1.2 – Design of bank filtration schemes and coupled engineered solutions).
UV disinfection	– Because of very low turbidity of the filtrate UV can be applied for disinfection.

3.4 Constructed wetlands as a tertiary treatment stage (TP, TN, micropollutants, pathogens)

CWs as downstream purification stage (tertiary treatment) are used to remove total phosphorus (TP), total nitrogen (TN), organic micropollutants (OMP) and pathogens. The German set of rules DWA-A 262E (2018) recommends VF sand filter (0-2 mm) and HSSF CW with coarse sand (0-4 mm) or gravel (2-8 mm) as polishing step. Within the scope of the AquaNES project the combination of ozonation and CWs as well as dual use retention soil filters (*RSF^{plus}*) have successfully demonstrated tertiary treatment OMP and pathogens removal. The AquaNES project also highlighted the potential and limitations of a (steel slag) reactive media CW for P removal.

There is no regulation for OMP in surface waters that allows for concrete evaluation of the treatment success. However, disinfection performance of the system can be assessed based on European Union Bathing Water Directive (EU BWD), where quality standards for indicator organisms *E. coli* and Enterococci are defined. Limits for phosphorus (P) and nitrogen (N) depend on the size of the WWTP and the sensitivity of the receiving water body and are set by national standards. If the treated wastewater is reused the compliance with national water reuse standards has to be assured (see D3.1).

CWs for tertiary treatment need efficient pre-treatment as they are implemented as polishing step. As for all wetlands the specific load is crucial for dimensioning and successful operation. According to the German set of rules DWA-A 262E (2018) the specific areal organic load of tertiary treatment VF CWs is limited (see Table 7). Due to the flow direction of HSSF CWs the cross-sectional area plays a major role for dimensioning of these types of wetlands. Depending on the filter media the specific COD loading on the horizontal cross-sectional area is therefore also limited (see Table 7).

Table 7 Maximum design loads for VF CW with sand, HSSF CW with coarse sand and HSSF CW with gravel under dry weather conditions in temperate zones (DWA, 2018).
Values given are per square meter of total filter surface/horizontal cross-sectional area.

		VF sand filter	HSSF CW with coarse sand	HSSF CW with gravel
HLR on the filter surface	L/(m²*d)	80 in winter (< 12°C); 120 in summer* (≥ 12°C)		
COD loading rate on the filter surface	g/(m²*d)	≤ 20	≤ 16	≤ 16
COD loading rate on the horizontal cross-sectional area	g/(m²*d)		≤ 40	≤ 200

*when monitoring the filter effluent for redox potential, an increased loading (up to the maximal loading rate) is possible, when oxic conditions are observed.

If not tested differently in each case these values should be respected in order to stay on the safe side and reduce the chance of clogging. Dimensioning and efficient operation of the polishing unit is directly related to the treatment performance of the main stage. In case of gravel beds (or crushed lava rocks) higher flow rates up to 1.000 mm/d have been tested successfully for post treatment in Schönerlinde.

It is plausible that TSS concentration also is an important factor for filtration wetlands (see TSS limits for retention soil filters in chapter 3.5.1). In case of tertiary treatment wetlands succeeding secondary clarifiers where a concentration of less than 10 mg/L TSS can be expected, remaining soluble organic load (BOD_5/COD/TOC) is more relevant though. For this reason, there is no specific regulation for TSS load of tertiary treatment wetlands.

According to DWA rules the media used in HSSF filter layer as well as in the distribution and collection sections should meet the requirements listed in Table 8.

Table 8 Characteristics of recommended filter material, distribution and collection layers of HSSF CWs as polishing step (DWA, 2018).

		HSSF coarse sand filter (0-4 mm)	HSSF gravel filter (2-8 mm)
Filter Layer	Depth of horizontal filter layer [m]	≥0.5	≥0.5
	kf-value [m/s]	~10^{-3}	~10^{-1}
	Effective grain size d_{10} [mm]	0.3 - 0.4	3
Distribution and collection layers	Grain size [mm]	2-8	8-16
	Thickness [m]	≥0.2	≥0.2

Different wetland types and operation regimes will be used depending on the polishing purpose. HSSF CWs or saturated flow sytems in general are particularly suitable for anaerobic denitrification. Reactive and adsorptive media are commonly used for P removal. Lately activated carbon or biochar have been introduced in order to increase OMP retention. Pathogens are efficiently removed in wetlands with fine-grained filter material (similar to slow sand filtration process).

Nitrogen

The combination of VF wetlands as biological main stage, where nitrification occurs, and HSSF wetlands for denitrification leads to 60-80 % removal of total nitrogen (TN) (Geller and Höner, 2003). Downstream denitrifcation is limited by the remaining biodegradable dissolved organic carbon (DOC) of the secondary treatment step. In this case especially large reed bed systems such as HSSF CWs contribute to the carbon cycle. Platzer found that downstream HSSF wetlands were eliminating 65 % of nitrogen (10% error level) if a specific nitrogen load of 1 g/m²/d was applied (Platzer, 1999). Bypassing a small part of carbon rich wastewater to the downstream HSSF wetland also was tested successfully (Rustige, unpublished).

Phosphorus

Phosphorus (P) removal in wetlands mainly occurs due to preciptation and adsorption processes. Plant uptake is of minor relevance and accounts only for ~5 % of the annual mass loading compared to the original wastewater load. This may be different in large area tertiary HSSF wetland systems where the main phosphorus load has been removed by technical means like simultanous P precipitation in activated sludge treatment.

Mainly particulate phosphorus is retained in regular treatment wetlands and surface flow systems are preferred. Long term retention rates through accretion by organic sediment of 40 % may be expected. Favourable are very long retention times in HSSF systems or very small hydraulic loading rates (Rustige et al., 2003). To design wetlands for phosphorus removal the main binding forms of phosphorus as well as pH and redox conditions have to be investigated.

As the conventional sand or gravel filter media usually has only few sorption capacity, adsorptive filter media may be implemented if an enhanched phosphorus removal is required. Filter media with high iron, aluminium, calcium or humus content have higher sorption capacity. Due to the finite sorption capacity of these media, a replaceable P-retention filter is an option (e.g. with FerroSorp® ferric hydroxide which has a specific P-sorption capacity of >10 g/kg). However, it has to be taken into account that the availability of these materials varies from place to place and should be selected according to the site-specific resources. The use of industrial byproducts such as steel slag and used iron gravel from waterworks has been tried in several pilot systems and was demonstrated on the AquaNES site in Packington, UK (see chapter 4.5) (Park et al., 2017; Hussain et al., 2015; Barca et al., 2013; Rustige and Platzer, 2002). Overall TP removal determined by several studies was in the range of 0.26 – 1.7 g P removed per kg of media over a period of 259 days to 2 years (Barca et al., 2013; Weber et al., 2007).

Organic micropollutants

Organic micropollutants (OMPs), like pharmaceuticals, industrial chemicals and pesticides, are present in the aquatic environment including sewage, surface water, groundwater and drinking water and may trigger unwanted ecological effects (Eggen et al., 2014; Luo et al., 2014; Heeb et al., 2012; Musolff et al., 2009; Osenbrück et al., 2007). OMPs originate from different point and diffuse sources and enter water bodies via different flow paths (Eggen et al., 2014; Osenbrück et al., 2007). One major source of OMPs is the effluents from conventional wastewater treatment plants (WWTPs) (Heeb et al., 2012; Osenbrück et al., 2007). Thus polishing steps are required to improve the water quality (Brunsch et al., 2018; Eggen et al., 2014).

Beside technical solutions, as ozonation, UV treatment and activated carbon filtration, CWs are suitable for OMPs reduction (Brunsch et al., 2018; Meyer et al., 2014; Verlicchi and Zambello, 2014). A review by Verlicchi and Zambello (2014) determined a broad range of pharmaceutical removal, ranging from 0 to 100 % removal, depending on the type of wetland and the investigated OMPs. Important design and operational parameters for OMP removal is the redox potential, as the degradation of different OMPs is better under aerobic, anoxic or anaerobic conditions; the HRT and the content of organic matter or an adsorbent, such as granular activated carbon (GAC) or biochar (Brunsch et al., 2018; Verlicchi and Zambello, 2014). (See chapter 4.3; D3.2)

Pathogens

Pathogens are efficiently removed in HSSF and VF CWs by 1.0-2.5 log-units (Geller and Höner 2003). This goes for each stage of primary, secondary or teritary wetlands. The main removal mechanisms are natural die-off, predation, excretion of antibiotics from roots of macrophytes and solar irradiation (in FWS CWs). The HRT and the grain size of the filter media are the principle design and operational criteria influencing removal rates. Furthermore, an enhanced bacteria and virus reduction can be achieved with sand rich in iron or aluminium oxides (WHO, 2006). (See also D3.1)

At the AquaNES pilot system in Rheinbach *E. coli* has been tested as relevant indicator organism for disinfection efficiency. Starting from rather low inlet concentrations of 10^3 MPN/100 mL after secondary treatment a median removal rate of 1 log-unit was reached during tertiary treatment by the RSF pilot systems with HRT of 3.25 h. This is an expected order of magnitude for this kind of treatment wetland.

At the AquaNES demonstration site in Schönerlinde, the process combination of ozonation and CWs works for a wider range of microorganisms (*E. coli*, Enterococci, *C. perfringens*, somatic coliphages) and therefore provides higher disinfection safety than the stand-alone solution ozone treatment (Table 9). During ozonation *C. perfringens* and somatic coliphages were not efficiently removed (<1 log-unit), but in the CWs they were additionally reduced to median concentrations in the range of MPN 1-10/100 mL (removal of 2-3 log-units). The comparison of a fine sand CW (0-2 mm) and a coarse lava gravel (4-8 mm)/biochar (8-20 mm) CW at the demonstration site Schönerlinde showed that the finer filter material led to better removal of pathogens at HRT of 10-48 h under aerobic saturated conditions. (See chapter 4.3; chapter 4.4; D3.2)

Table 9 **Median effluent concentration and pathogens removal (in brackets) of different tertiary treatments as stand-alone solution (O_3, CW, RSF) and cNES (O_3 + CW).**
O_3 – ozone treatment; CW – constructed wetlands (demonstration site 12); RSF – RSF pilot (demonstration site 11); n.d – not determined.

	E. coli	Enterococci	*C. perfringens*	Somatic coliphages
O_3	<10^2 MPN/100 mL (≥ 2 log-units)	<10^2 MPN/100 mL (≥ 2 log-units)	$6.3*10^3$/100 mL (< 1 log-unit)	$5.3*10^3$/100 mL (< 1 log-unit)
CW	<15-$3.6*10^2$ MPN/100 mL (1.5-3 log-units)	15-$1.2*10^2$ MPN/100 mL (1.5-2.5 log-units)	0-10^2/100 mL (2.5-4 log-units)	0-10^2/100 mL (2-3.5 log-units)
O_3 + CW	<15 MPN/100 mL (2-3.5 log-units)	<15 MPN/100 mL (1-3 log-units)	1-10/100 mL (2-4 log-units)	1-10/100 mL (2-4 log-units)
RSF	$5*10^1$ MPN/100 mL (≥ 1 log-unit)	n.d.	n.d.	n.d.

3.4.1 cNES – tertiary treatment wetlands recommendation

Engineered Treatment Step	Requirement
Secondary WWTP	Remaining TSS and organic load after secondary clarifier are important parameters for defining safe loading rates to the wetland system. – A maximum total annual TSS load of 7-10 kg/m² has been shown to be feasible for VF systems in RSF and should be applied here as well if relevant TSS concentrations in the effluent of the secondary treatment system can be expected. – The organic load (COD) to the wetland in case of VF sand filtration is also limited to 20 g/m²/d. – Intermittent VF operation depends on oxygen consumption and oxygen concentration after secondary WWTP. In case of oxygen deficit a fluctuating water level or unsaturated flow conditions in VF would be preferable.
Ozone treatment + CWs	– A synergetic effect of the cNES was observed for DOC removal (21-22 %) at the demonstration site in Schönerlinde. Biodegradable organic transformation products formed by ozonation are reduced in the CWs. – Due to oxygen saturation, the wetlands can easily be operated aerobically under water saturated conditions. Depending on hydraulics a horizontal flow wetland could also be an alternative option. – The process combination works also for a wider range of microorganisms (*E. coli*, Enterococci, *C. perfringens*, somatic coliphages) and therefore provides higher disinfection safety. – In case of ozonation break down the combined wetland serves as extra safety barrier.
Extra activated carbon (AC) layer	– If an activated carbon layer is used for adsorption of organic micropollutants the extra layer needs to be protected from other adsorptive COD residuals. This means the building of an organic surface layer should be supported in VF systems. – Also an extra carbon protection layer on the filter surface is useful. Such a layer could be regenerated, exchanged or filled up when saturation is monitored. – The grain size of the granular AC should not be less than the filter media itself. The lower density of the carbon has to be taken into account and the filter stability of chosen layers needs to be tested in laboratory before construction.

3.5 Constructed wetlands for dual use: CSO and tertiary treatment use

One of the most powerful wetland utilization is treating contaminated stormwater runoff or combined sewage overflow (CSO). No other treatment system combines such retention and treatment capacity for this kind of highly dynamic flow.

A retention soil filter (RSF) is a specially improved VF wetland with increased hydraulic retention capacity (Figure 6). The filter media is designed to react with soft rain water in order to increase pH buffer capacity. This will ensure near neutral pH-values along the nitrification process and it will prevent resolving heavy metals which may be part of stormwater runoff.

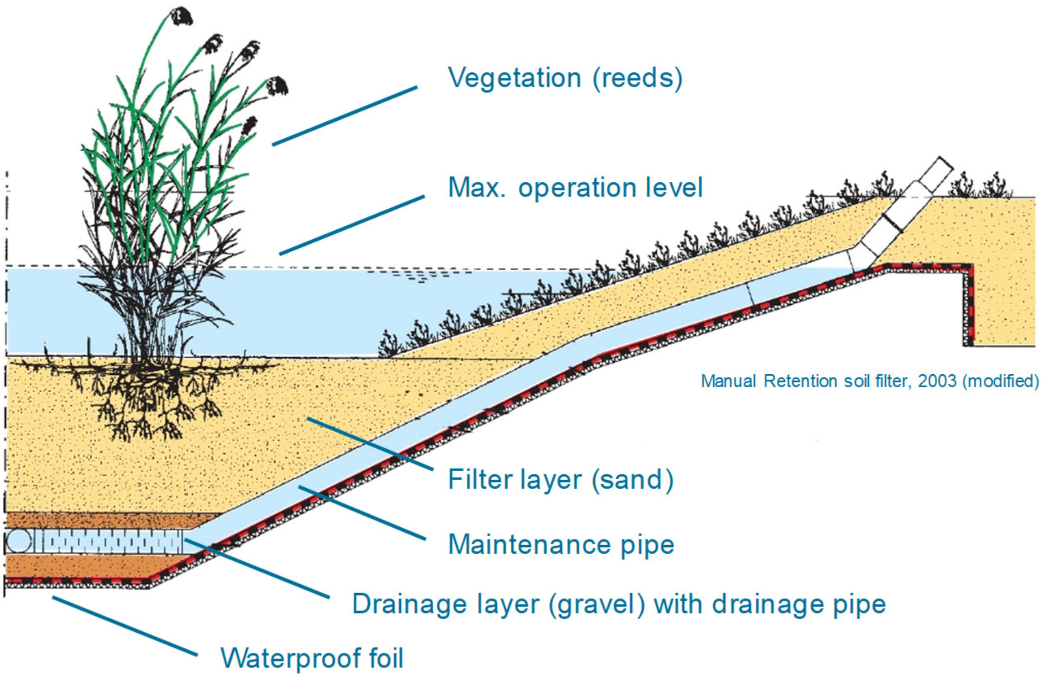


Figure 6 Principle structure of a retention soil filter (MKULNV [modified], 2015).

As in any other VF system a secondary layer is built on top of the filter. This layer contributes to the treatment process. However organic, hydraulic and particle loads are limited to the system specification. Organic load and particle load is strongly related to each other, so that TSS load may be used as critical design parameter. In principal the overall loading limits of secondary treatment wetlands should be respected. To control the hydraulic load the percolation rate is limited by a throttle in the outlet. The excess CSO volume is stored in an additional retention tank with an overflow into the receiving waters.

Phragmites australis is suggested as best operating plant. Its contribution is structuring the sediment layer on the filter surface and keeping a minimum hydraulic capacity. After some years of growth this plant is able to displace any other vegetation.

The retention area on top of the filter may also function as temporary horizontal surface flow wetland at very high loading. For this purpose an overflow weir has to be designed on the distant site of the inlet section or water distribution channels respectively. The dense vegetation increases settling effects compared to sedimentation tanks.

Wetlands receiving stormwater or CSO need to be designed by making use of modeling software in order to calculate dynamic loading rates considering wetland area, throttle flow and retention levels.

Table 10 Design parameters RSF
Based on DWA-A 178 (2017) and MKULNV (2015); EBCT = empty bed contact time

Design parameter	Sewerage system	
	Combined sewer	Separate sewer
Max. filter velocity Percolation rate V_F [L/m²/s]	0.03	0.04
Max. V_F [m/h]	0.11	0.14
EBCT [h]	6.4 – 9.1	5.0 – 7.1
Staple height [m/a]	30	40
Suspended solids load [kg/m²/a]	< 7	< 7

Empty bed contact time EBCT = h_{filter} sand/ V_F, this means min. height of the filtration layer: 0.7 – 1.0 m. Locally available material (0 - 2 mm) is preferable if the following parameters can be met:

Table 11 Particle size distribution of filter media for RSF
Based on DWA-A 178 (2017) and MKULNV (2015)

Grain fraction	Recommendation	
	mm	Weight by weight [%]
Clay / silt	<0.06	0
Fine sand	0.06 – 0.20	15
Middle sand	0.20 – 0.60	70
Coarse sand	0.60 – 2.00	15
Fine gravel	> 2.0	0

Of course any filter media for water treatment should be free of heavy metals (Pb, Zn, Ni, Cu, Cd, Co), metalloid (As) and nutrients (P). $CaCO_3$ is added by 20 % for pH-value stabilization.

An applet to assist the design, dimensioning and assessment of RSF has been developed in the project and is available in the AquaNES DSS: http://dss.aquanes.eu/Default.aspx?t=1749&RSF=0

"RSF*plus*" for flexible use

In order to achieve added value the Erftverband, a utility which is operating more than 20 RSF, has tested and demonstrated a new application at the AquaNES demo site in Rheinbach, Germany (see chapter 4.3 where detailed information is given). *Plus* - stands for additional use compared to single CSO treatment. During dry weather periods secondary treated wastewater from the standard treatment plant is polished. The main purpose then is OMP removal. According to piloting by Erftverband this is best achieved by integrating a GAC layer to the standard filter layout of a typical RSF.

The following specifications for the additional filter media have to be checked:

- Abrasion coefficient - proof that clogging is no problem in the long term,
- Hydraulic conductivity - percolation rate kf >10^{-3} m/s,
- Verification of particle size grading curve – 0.5 to 4.0 mm,
- Specific pore surface - min 950 m²/g,
- Loading value - min 900 mg/g,
- Hardness - min 95 %,
- Adsorption capacity (validated by lab analysis with GAC under consideration).

3.5.1 cNES – dual CSO and tertiary treatment wetlands – recommendation

Engineered Treatment Step	Requirement
Combined sewage overflow retention and pre-sedimentation tank	– A typical constellation is a retention tank with overflow after sedimentation. – In case of existing treatment sites the effluent quantity and quality shall be investigated and used for long term modelling of wetland loading rates (e.g. 30 years or longer). – Sedimentation of combined sewage will reduce the organic load and TSS load to the wetland. – The modelled annual suspended solid loading rate per filter area has to be limited to 7 kg/m²/a. This means either larger filter surface area or a larger settling tank if the loading rate is exceeded. – For CSO treatment the filter size must also be adapted to the maximum allowable water flow to the receiving water. This value (x L/s) influences the chosen filter size/ retention volume and throttle flow.
RSF for dry weather effluent	– A wetland for tertiary treatment can accept much higher daily hydraulic loads because of low particle and BOD_5 concentrations. This could be shown at AquaNES demo site in Schönerlinde where up to 1 m/d HLR was acceptable while suspended solids and BOD_5 was very low. – If activated carbon is added for additional OMP removal this filtration layer should be placed near the bottom in order to prevent early saturation because of binding other organics (COD). In addition, the top layer can be enriched with 10 % GAC. This relieves the load on the lower GAC layer. The upper layer is accessible so that GAC can be added if required. – In case of a flexible use with alternated loading of CSO the accumulation of sludge on the filter surface has to be taken into account though. This means that the design conditions (loading rates) of a regular retention soil filter have first priority. – As for all VF wetlands with sludge accumulation an operation cycle of 1 part operation and 2 parts of rest period has shown to be useful (e.g. 1 d / 2 d respectively, see French system). – By modelling dry weather conditions and overflow events for long periods the optimum wetland and CSO pre-treatment size can be found by iteration process.
Pumping station and water distribution	– Sizing of pumps and pipes is crucial for achieving the desired hydraulic treatment efficiency if stormwater and CSO is involved. This is part of modelling. – Practically water distribution on the wetland systems must be able to manage high loading rates during storm events and small amounts during daily feeding. – Separate feeding systems for both purposes can be useful.

4 Case studies of combined systems

The intention of this chapter is to demonstrate implemented solutions including operational results. Lessons learned in positive and negative aspects may improve the understanding and help to develop optimised and adapted combinations of engineered and natural treatment systems.

4.1 Treatment wetland with P-precipitation and chemical disinfection

The main focus of this solution was to create a robust, near-natural secondary treatment system that would result in a good reduction of the general parameters. Primary treatment comprised screening, settling and precipitation. At the end of the line a standard chlorination step was added to ensure hygienic quality for water re-use, meeting the Greek standards.

4.1.1 Treatment concept

The treatment site is located on the natural slope on the island's coast allowing gravity flow for the main treatment process (Figure 7). The treatment plant typically also accepts sewage trucks serving for sanitation of houses which are not connected to the central sewer system.

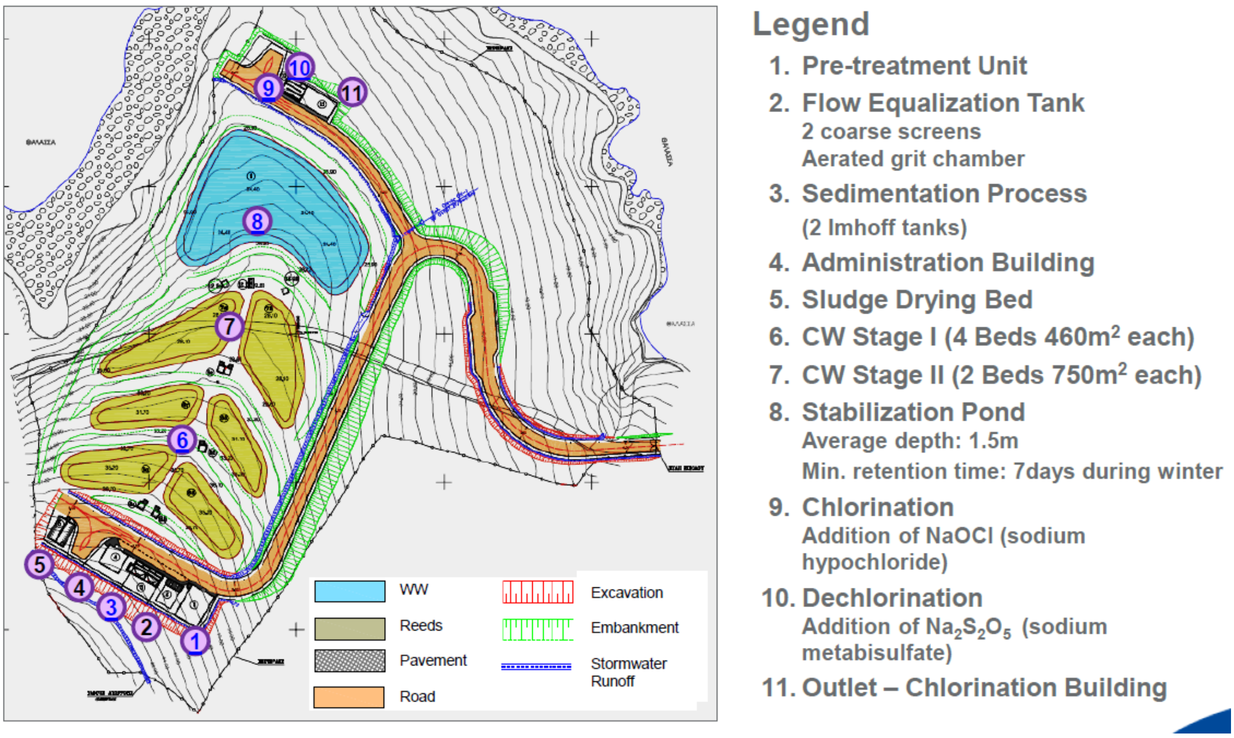


Figure 7 Overview of the WWTP in Antiparos.

The total design treatment capacity of the WWTP amounts to 480 m³/d, including wastewater from septic tanks delivered by trucks. The effluent is used for the irrigation of public spaces in the proximity of the WWTP (1.2 ha inside the WWTP and 10 ha nearby the WWTP).

As CWs adapt well to varying hydraulic and pollution loads, this natural technology is considered appropriate also for touristic islands. At Antiparos there are 1,211 permanent inhabitants (2011) plus

1,000 seasonal residents and tourists (2012). The number of tourists visiting the island has highly increased since then.

The mechanical pre-treatment consists of a compact 6 mm screening machine (HUBER Complete Plant ROTAMAT® Ro5; see Figure 8) including an aerated grit chamber followed by an equalization tank (160 m^3) and two parallel Imhoff tanks with a settling volume of 70 m^3 each.

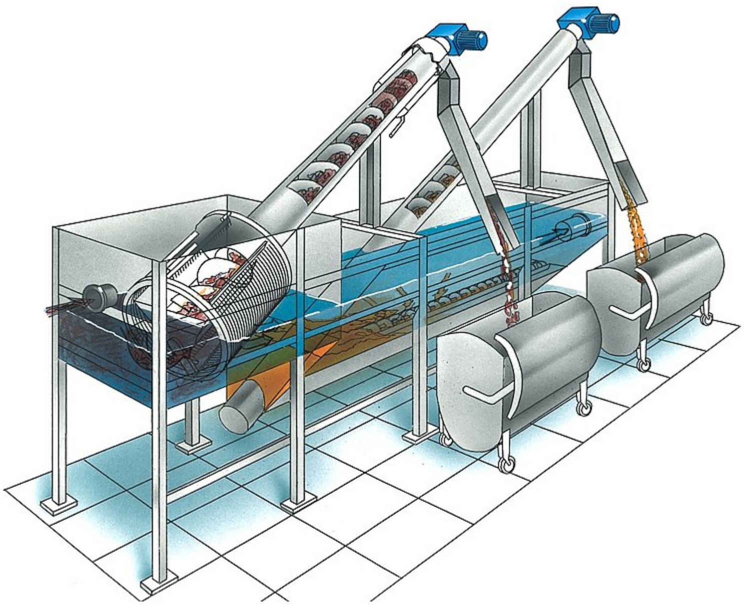


Figure 8 HUBER compact fine screen with grit removal and sand trap (Huber.de, 2019).

Phosphorus is removed in the pre-treatment by flocculation with poly-aluminum chloride 18 % (PAC18). In order to adapt to the touristic season, the pre-treatment capacity is increased by double dose of PAC18 from June to August (6 L/d) compared to the lower season from September to May (3 L/d).

The following two stage VF wetland system is the main biological treatment step. The first stage comprises four parallel treatment cells with a filter surface of 460 m^2 each. The second stage consists of two parallel cells with a size of 750 m^2 each. The wetlands discharge to a stabilization pond with a surface area of 2,100 m^2 and a water depth of 1.3 m, so the volume accounts for 2,730 m^3. Minimum HRT in the stabilization pond is 4 d, maximum HRT during low season is ~40 d.

Finally the effluent passes an ordinary chlorination/de-chlorination step. Chlorine (NaOCl) is added to the effluent which enters a meandering canal. An extended contact time is provided in this way.

Table 12 Estimated daily flow [m^3/d] of the WWTP Antiparos (May 2017-Nov 2018).

Month / Year	Estimated daily flow [m^3/d]
May – Sep 2017	350 - 550
Oct 2017 – Apr 2018	70 - 150
May – Jul 2018	350 - 550
Aug 2018	~ 600
Sep 2018	~ 380
Oct 2018	~ 150
Nov 2018	~ 70

The dosing rate varied according to the season: 80 L/d during summer (Jul-Sep); 65 L/d from Mar-Jun and Oct-Nov; 50 L/d from Dec-Feb. Afterwards de-chlorination occurs with $Na_2S_2O_5$ at the same dose as chlorine.

Table 12 presents estimates of the expected daily sewage derived from population equivalents (PE) living on the island for the monitoring periods May 2017 to November 2018.

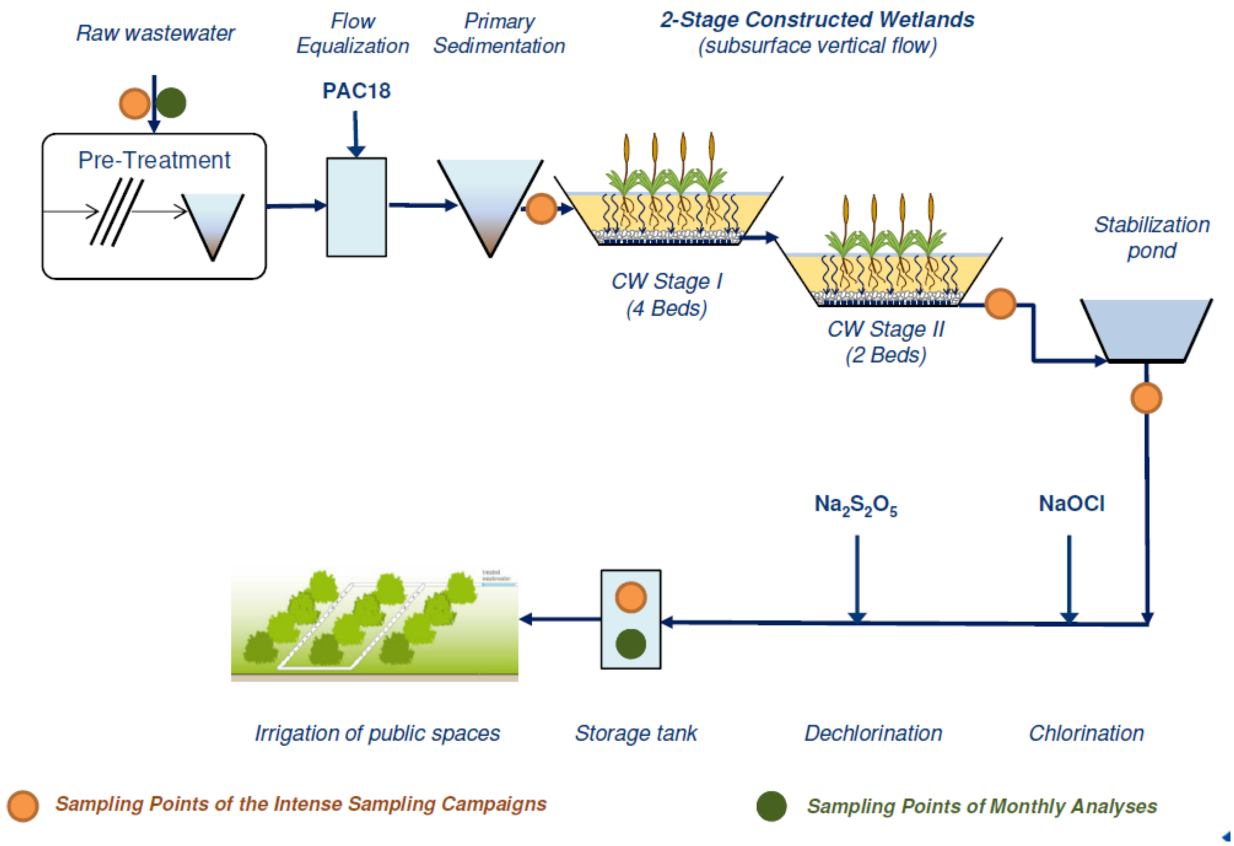


Figure 9 Flow scheme of WWTP in Antiparos with sampling points.

4.1.2 Operation results

Monthly water quality analyses of the in- and effluent were conducted. In summer (Aug 2017; Aug/Sep 2018) and low season (Nov 2018), intense sampling campaigns took place for 7-8 days. During these campaigns, samples were taken at each treatment step as shown in Figure 9. Both the samples of the monthly sampling and the one of the sampling campaigns were analysed in Athens. Additionally, in Oct/Nov 2018 samples were taken by AKUT and analysed on site. The data from the monthly sampling, from the intense sampling campaigns as well as from the on site analyses are presented in the following.

Actual flow rates during monitoring period were not available. The flow meter at the WWTP outlet was not in use. The control system had been partly damaged by electrical surge. For evaluation of the treatment performance estimations derived from number of inhabitants were used (see Table 12).

During winter 2017/18 the wetland was partly under reconstruction (see 4.1.2.1)

4.1.2.1 Filter material exchange, planting and system optimization

In 2017 clogging of the vertical flow wetlands was obvious. Wastewater infiltration stopped and filter overflow occurred. An investigation led to the conclusion that either the chosen filter media was too fine or the areal loading rate was too high for this kind of filter media. Due to ponding on the filter surface odour emission became an issue.

For these reasons it was decided to remove the top clogging layers. From February to April 2018 the accumulated sludge, plants and the fine layer (0-25 cm from top) were removed and the original vegetation (*Arunda donax*) was substituted by *Phragmites australis*.

Treatment performance was not affected negatively by removing the fine layer as the comparison of the two years 2017 and 2018 shows (see Table 13, Table 14).

Due to the filter material exchange hydraulic treatment capacity was increased. The total amount of sewage could be treated without ponding on the filters or without need for bypassing. Any odours from the wetlands which was noticed before were completely avoided by this measure. While the TSS concentration in WWTP effluent showed an exceedance of the Greek reuse limit for restricted irrigation in 2017, the limit was met in all samples after reconstruction (seeTable 14, Table 17).

In August 2017, the results of the intense sampling campaign revealed that the dosing amount of chlorine used for disinfection was insufficient with average effluent concentrations for *E. coli* >10^3 cfu/100 mL (see D3.1). After adjustmen in 2018, disinfection was efficient with mean effluent concentrations for *E. coli* of 0 cfu/100 mL.

Performance comparison 2017 vs. 2018 (before and after the filter modification)

Two data sets for comparison of the years were used A) intense sampling campaigns (Table 13) and B) monthly sample taking (Table 14).

Table 13 Average in- and effluent concentrations and removal rates of the CWs, Antiparos, 2017 and 2018 (intense sampling campaigns).
Aug 2017 (n=8); Aug/Sep 2018 (n=8); effluent Imhoff tank equals to inflow CW Stage I.

	Imhoff tank		CW Stage II		Removal rate [-]	
	Aug 2017	Aug/ Sep 2018	Aug 2017	Aug/ Sep 2018	Aug 2017	Aug/ Sep 2018
COD (mg/L)	558	391	64	54	0.89	0.86
BOD_5 (mg/L)	224	174	27	23	0.88	0.87
TSS (mg/L)	48	111	29	24	0.40	0.78
TN (mg/L)	43	32	15	10	0.65	0.69
Kjeldahl N (mg/L)	41	30	7	6	0.83	0.79
TP (mg/L)	6	5	4	3	0.40	0.39
E. coli (cfu/100mL) (median)	7.1E+05	1.2E+06	6.0E+03	2,9E+03	2.1 log-units	2.6 log-units
Coliforms (cfu/100mL) (median)	9.0E+05	1.1E+06	8.6E+03	3,9E+03	2.0 log-units	2.5 log-units

Table 14 Average in- and effluent concentrations and removal rates of the WWTP Antiparos, 2017 and 2018 (monthly samplings).
2017 (equals to Jan 2017 – Jan 2018; n=11) and 2018 (equals to May 2018 – Jan 2019; n=9).

	Inflow		Effluent		Removal rates [-]	
	2017	2018	2017	2018	2017	2018
COD (mg/L)	686	665	62	52	0.90	0.90
BOD_5 (mg/L)	285	280	19	16	0.92	0.93
TSS (mg/L)	224	237	37	14	0.81	0.90
TN (mg/L)	84	98	22	14	0.72	0.74
Kjeldahl N (mg/L)	83	97	5	2	0.94	0.98
TP (mg/L)	7.1	8.0	0.5	0.2	0.92	0.98
E. coli (cfu/100mL) (median)	3.5E+05	6.4E+06	< LOD	< LOD	4.9 log-units	6.1 log-units
Coliforms (cfu/100mL) (median)	8.0E+05	5.0E+06	< LOD	< LOD	5.0 log-units	7.0 log-units

Removal of organic bulk parameters and nutrients was not influenced by the removal of the top fine filter layer of stage I. In 2018, the limits set by the Greek Reuse Legislation for restricted irrigation were met (Greek limits for water reuse see Table 17 below).

4.1.2.2 Organic bulk parameters, TSS, nutrients, pathogens

In 2018, the performance of the various treatment steps was determined by the intense sampling campaigns in August/September and November. Reuse limits for restricted irrigation concerning COD, BOD_5, TSS and TN were reached after the filtration wetland. Furthermore, the effluent of the WWTP Antiparos complied with the limits for *E. coli* in 2018.

Organic parameters (COD and BOD_5) were mainly removed in the CWs by 85 % and 86 %, respectively. During pre-treatment organic compounds were reduced by 37 % on average (Figure 10). Mean WWTP effluent COD concentrations were 26 mg/L in Aug/Sep 2018 and 18 mg/L in Nov 2018. Likewise, TSS reduction was very good in the pre-treatment at an average of 56 % as well as in the CWs at an average of 78 % (Figure 11). Mean WWTP effluent TSS concentrations were 5 mg/L in Aug/Sep 2018 and 3 mg/L in Nov 2018. TN removal, took primarily place in the pre-treatment (~77 %) and in the CWs (~64 %) (Figure 12, Figure 13). Nitrification (Kjeldahl N removal) was highest in the vertical flow wetland. TP was removed to effluent concentrations of 0.2-1.1 mg/L with highest removal in the Imhoff tank due to chemical dosing for precipitation processes. As expected, best removal of pathogens was reached by the CWs (~2.5 log-units) and the chlorination/dechlorination step (~3 log-units).

The chemical analyses processed on-site by AKUT revealed another outcome concerning TN and TP removal. According to on-site analyses, TN and TP WWTP effluent concentrations were ~25 mg/L and ~3 mg/L, respectively in Oct/Nov 2018. This issue should be related to necessary transport of samples and the period of time until final analysis or even on the testing procedure itself. Also a different status of operation of the treatment plant during AKUT sampling moments could be one reason for different results. Resolving of phosphorus from decay of algae or sediment within the polishing pond could be another possibility for increased P-values.

Average removal rates for the various treatment steps are summarised in Table 15. COD, TSS, TN and TP concentrations at the different sampling points are depicted in Figure 10 to Figure 14.

Table 15 Average removal rates [-] of the various treatment steps, Antiparos, Aug/Sep/Nov 2018 (n=15).

	Imhoff tank	**CW Stage II**	**Stabilization Pond**	**Chlorination/ De-chlorination**
COD	0.37	0.85	0.46	0.25
BOD_5	0.37	0.86	0.44	0.25
TSS	0.55	0.80	0.49	0.61
TN	0.77	0.64	0.37	0.35
Kjeldahl N	0.78	0.79	0.46	0.40
TP	0.64	0.36	0.52	0.55
E. coli (median)	0.9 log-units	2.6 log-units	0.6 log-units	2.9 log-units
Coliforms (median)	0.9 log-units	2.5 log-units	0.4 log-units	3.3 log-units

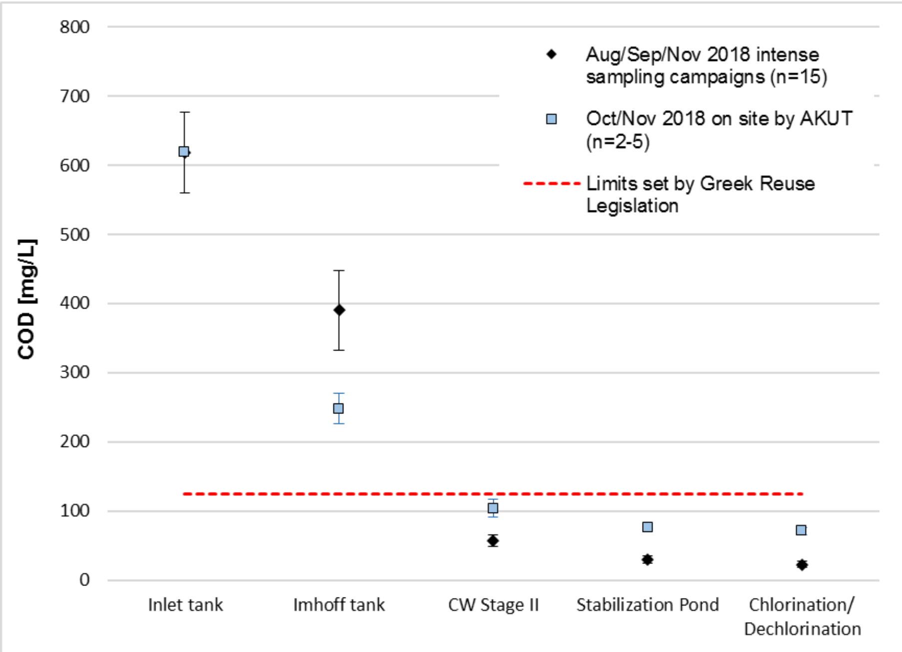


Figure 10 COD concentration at different sampling points, WWTP Antiparos (average; standard deviation). Intense sampling campaigns Aug/Sep/Nov: n=15 (black); Oct/Nov on site by AKUT: n=2-5 (blue), ”Inlet tank” = effluent equalization tank and “CW Stage II” = CW Stage I for samples taken by AKUT; red line: limit for restricted irrigation.

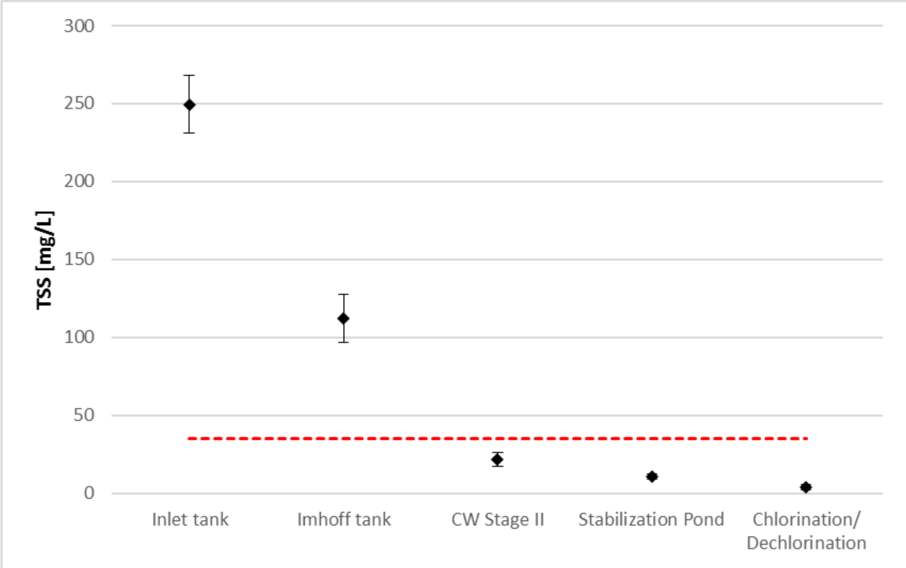


Figure 11 TSS concentration at different sampling points, WWTP Antiparos (average; standard deviation). Intense sampling campaigns Aug/Sep/Nov 2018: n=15; red line: limit for restricted irrigation.

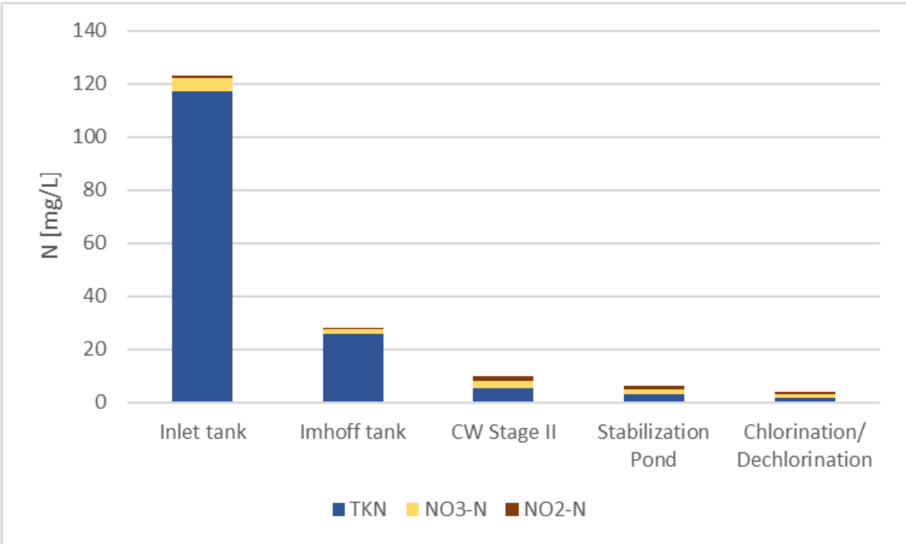


Figure 12 N fractions at different sampling points. Average values, n=15, Antiparos, Aug/Sep/Nov 2018.

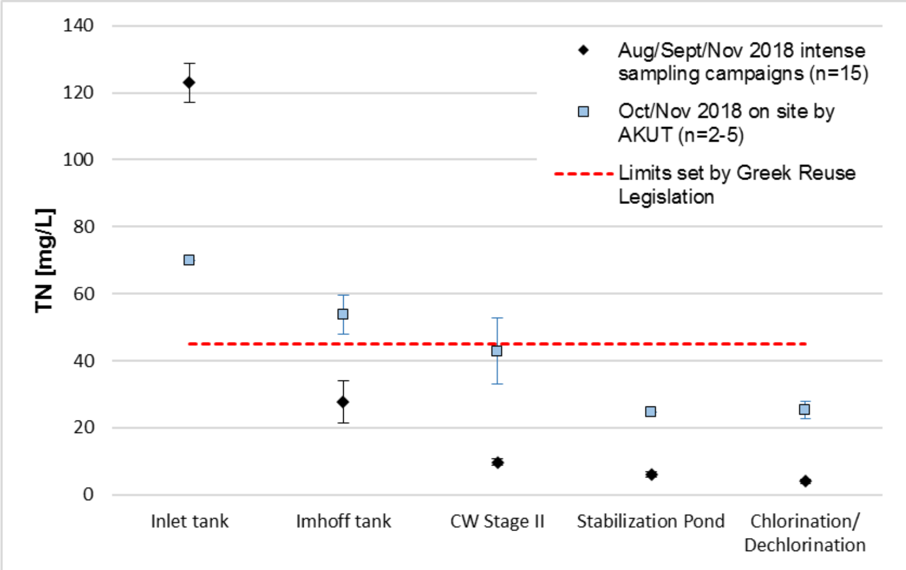


Figure 13 **TN concentration at different sampling points at WWTP Antiparos (average; standard deviation).** Intense sampling campaigns Aug/Sep/Nov: n=15 (black); Oct/Nov on site by AKUT: n=2-5 (blue), "Inlet tank" = effluent equalization tank and "CW Stage II" = CW Stage I for samples taken by AKUT; red line: limit for restricted irrigation.

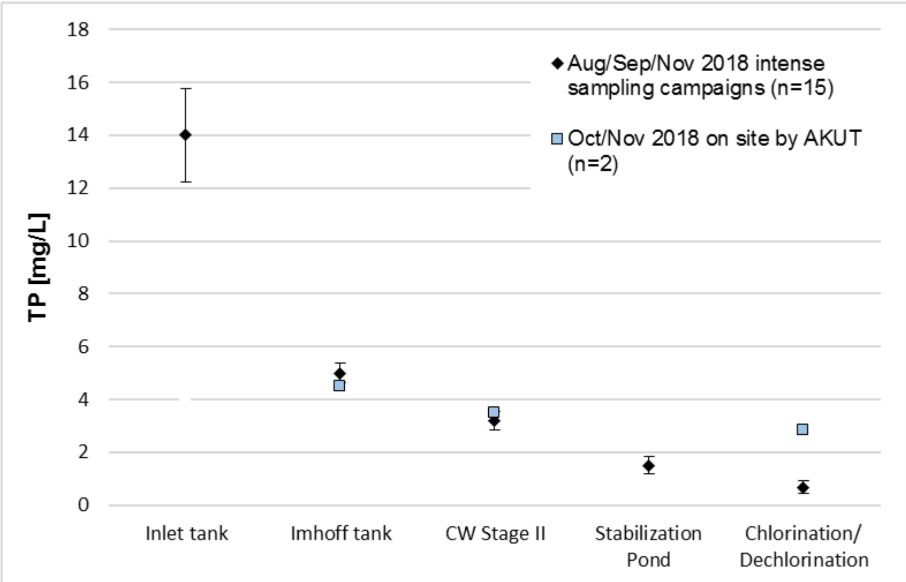


Figure 14 **TP concentration at different sampling points at WWTP Antiparos (average; standard deviation).** Intense sampling campaigns Aug/Sep/Nov: n=15 (black); Oct/Nov on site by AKUT: n=2-5 (blue), "Inlet tank" = effluent equalization tank and "CW Stage II" = CW Stage I for samples taken by AKUT.

4.1.2.3 Water distribution and metering system

An important operational factor is water distribution and wetland drainage. It was noticed that the system was not designed sufficiently and not operated optimally. E.g. it was not possible to alter the water table in each wetland cell. Batch feeding was lacking of volume (only 10 mm per batch) and the control system did not allow monitoring of hydraulic loading rates of each cell (e.g. number of daily batches).

The filter cells were not always operated in alternating mode. This may have caused organic overloads and did not allow resting periods for filter regeneration.

Since a flow meter at the entrance of the treatment plant was missing it was not possible to calculate actual hydraulic and organic loading rates. Flow information combined with knowledge of typical concentrations are necessary for controlling alternating filter operation. Especially if trucks deliver sewage or sludge from septic tanks this information has to be taken into account.

4.1.2.4 Stabilization pond

The stabilization pond following the two stage filtration wetlands had a small treatment effect during regular operation of the combined system. A significant but low effect was seen for further nitrogen reduction. The treatment capacity concerning COD, BOD_5 and TP was slightly better in November 2018 (57 %, 51 % and 62 %, respectively) compared to August/September 2018 (36 %, 37 % and 43 %) due to longer HRT. On the other hand, nutrients and sunlight lead to production of algae. Algae contribute to the COD of the effluent if there is no filtration afterwards.

4.1.2.5 Optional integration of aerated wetland systems

In July 2018, three test containers were installed at the WWTP Antiparos, one actively aerated filter and a two stage vertical flow system using artificial filter media (see Figure 16, Figure 17).

The actively aerated wetland is an intensified, space-saving system. The VF wetland is operated with permanent water saturation. Atmospheric oxygen (air) is introduced into the water at the bottom of the filter bed via an air compressor (Figure 15). According to the DWA A-262E (2018) only ≥1 m^2 per PE is needed; the average specific daily COD volumetric loading rate should not exceed 100 g/(m^3*d).

The pilot filter was constructed utilising an IBC-container of 1 m^3. Expanded clay was used as filter material; it was planted with *Phragmites australis*. The aerated CW was fed with effluent from the Imhoff tank with an HLR of ~150 L/d that correspond to an average specific daily COD volumetric loading rate of 65-90 g/(m^3*d). The aeration ran at 50 % time intervals; the bubble pattern was homogeneous.

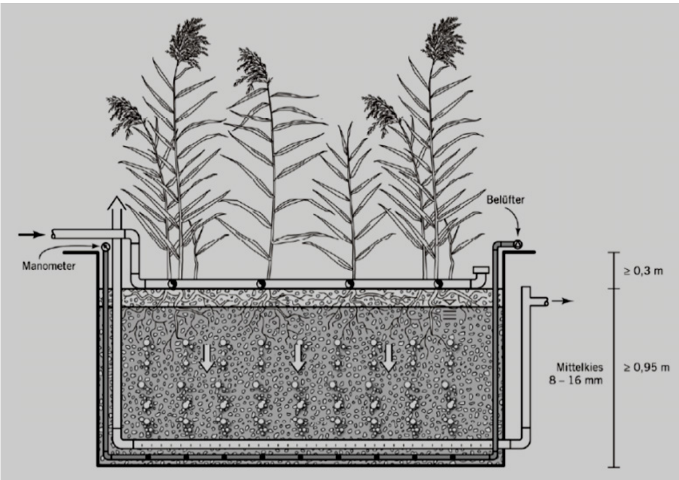


Figure 15 Actively aerated vertical filter with gravel 8-16 mm, schematic diagram (DWA A-262, 2017).

On site analyses taken end October/beginning November 2018 showed a very high treatment capacity concerning COD and nitrogen removal. Average removal rates amounted to 87 % for COD, 91 % for TN and 93 % for NH_4-N with mean effluent concentrations of 31 mg/L, <5 mg/L and 3 mg/L, respectively.

Taking these results into account, there is a huge potential for increasing the current capacity of the existing wetland site.

Figure 16 Test filter containers at WWTP Antiparos (Nov 2018).

Figure 17 Test filter containers at outlet of Imhoff tank at WWTP Antiparos (Nov 2018).

4.1.3 Conclusion

Synergy effects

At the touristic influenced island, the combination of CWs with chemical and mechanical pre- and post-treatments was able to reduce COD and TSS as well as TP, TN (N_{org}) and microorganism as desired. Depending on the seasonal load the chemical dosage should be adapted to save resources and costs. CWs are naturally appropriate for varying hydraulic and pollution loads.

The close cooperation with the mayor and the operators of the sewage treatment plant made it possible to quickly find and implement solutions for optimization during this AquaNES project.

Design and operation of the CW

Design, operation and maintenance are key elements for a high treatment capacity and durability of the WWTP. A CW requires the same knowledge, experience and adherence to the detailed design as any other technical WWTP. This showcase demonstrates that there are some important factors to be considered when designing a CW:

– The possibility to control the water level within the filter system should be given (by a shaft directly after each filter cell). This is necessary for start-up phase of vegetation.
– Homogeneous water distribution prevents local ponding and increases nitrification rates.
– Appropriate buffer capacity of feeding construction to ensure a feeding volume of > 20 mm per batch.
– Suitable filter material determines the treatment capacity and removal rates (typically sand or gravel).
– A control system is necessary to monitor and manage hydraulic loading rates of each cell.
– A flow meter for inflow control to adapt chemical dosing and hydraulic loading of CW (helps to prevent seasonal overload of CW).

Operation and maintenance of the WWTP:

– Alternated feeding of the CW cells allows resting periods.
– Regular checks of shafts, gates, distribution and drainage pipes (weekly).
– Regular sludge removal of Imhoff tanks (intervals depend on sludge amount).
– Limitation of delivering sewage trucks during high season to prevent overload.

Increasing wetland capacity for population growth

The increasing touristic attractiveness of the Greek island leads to an augmentation of wastewater inflow. A solution for increasing the capacity on the same treatment area could be intensification of the natural processes e.g. by substituting the existing stabilization pond with an aerated treatment wetlan.

An aerated VF filter can serve at least 1 PE per m². During touristic season an extra area of 1,000 m² (half of existing pond) could increase the capacity by 1,000 PE or 150 m³ per day, respectively.

The limiting factor would be the design of the primary treatment. According to the monitoring its capacity has not been reached yet. Especially the use of precipitates/flocculants provide sufficient margin in this engineered/natural treatment combination.

Table 16 Key Takeaways from CW + P-precipitation and chemical disinfection.

		CW + P-precipitation and chemical disinfection
Function (+ removal/ - increase)	+ + + + + +	COD TSS TKN TP *E. coli* Coliforms
Risks of malfunction		– Filter clogging by overload – Insufficient flocculation and disinfection due to lack of chemical supply
Operation and maintenance needs		– Regular inspection of water distribution (weekly) – Flow and quality monitoring – Daily supervision of dosing stations – Automated control of water distribution on wetland cells – Remote control and alarm system

4.2 Wetland with TiO_2-activated pre-treatment

On the Greek island Thirasia a multiple treatment system has been built for demonstrating the combination of several advanced wastewater treatment options for water reuse. This case describes the special combination of an activated pre-treatment with CW.

4.2.1 Treatment concept

At the WWTP of Thirasia various treatment processes have been implemented. Solar photo-catalysis with the catalyst TiO_2 is installed as oxidative pre-treatment system for CW. Two optional disinfection units serve as post-treatment to reach the Greek Environmental Protection Limits for reuse. The effluent is used for irrigation of the areas near the WWTP.

The mechanical pre-treatment unit consists of a screen and a sedimentation tank (Figure 18; Figure 19). A photo-catalysis unit with two basins of 77 m² each and a depth of 0.8 m as well as two parallel horizontal subsurface flow wetlands with a surface area of 208 m² each follow as denitrification treatment. The photo-catalysis reactor is operated similarly to an activated sludge system using fine-bubble aeration and sludge recirculation with clarifiers. The TiO_2 catalyst shall be separated from sludge by sedimentation and shall finally be recirculated several times. The main treatment steps, photo-catalysis and CW, are implemented in two parallel lines so that they can be operated according to the actual daily inflow. The optional use of chlorination and/or an ultrafiltration (Model: HYDRAcap 60) serves as disinfection.

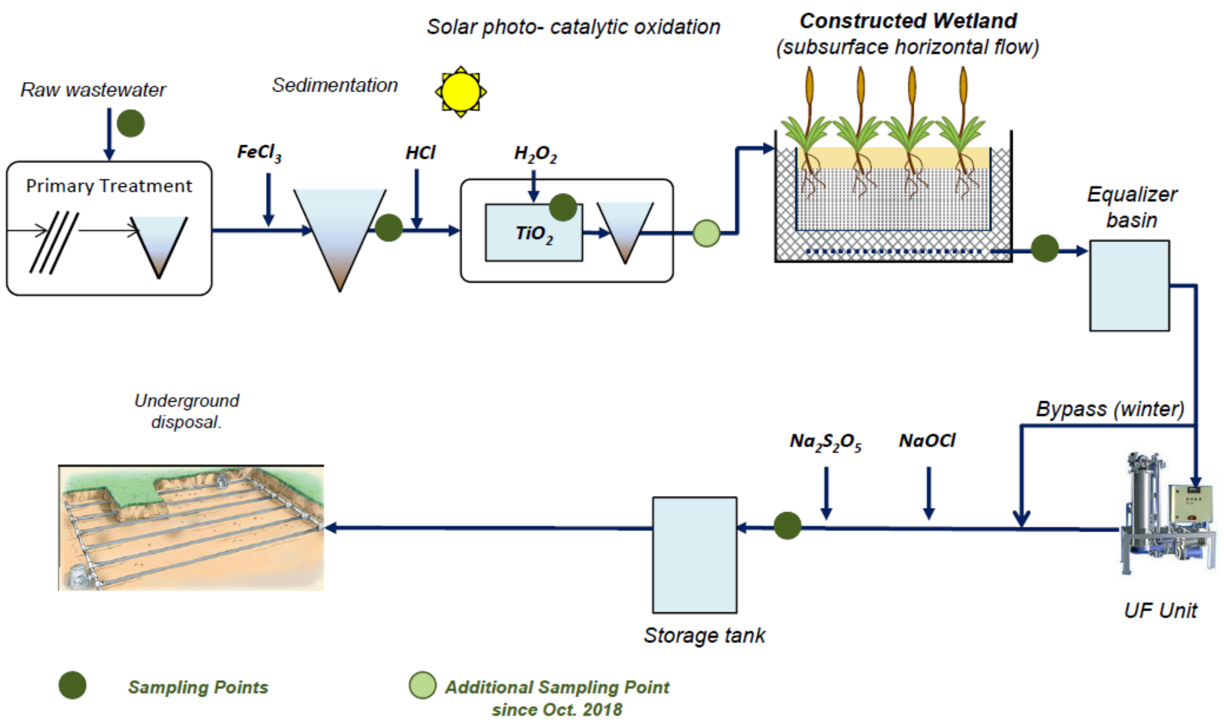


Figure 18 Flow scheme of WWTP in Thirasia with sampling points.

The design capacity of the WWTP amounts to 142 m³/d. From June 2017 to November 2018 inflow rates as well as physical, chemical and microbiological parameters were investigated.

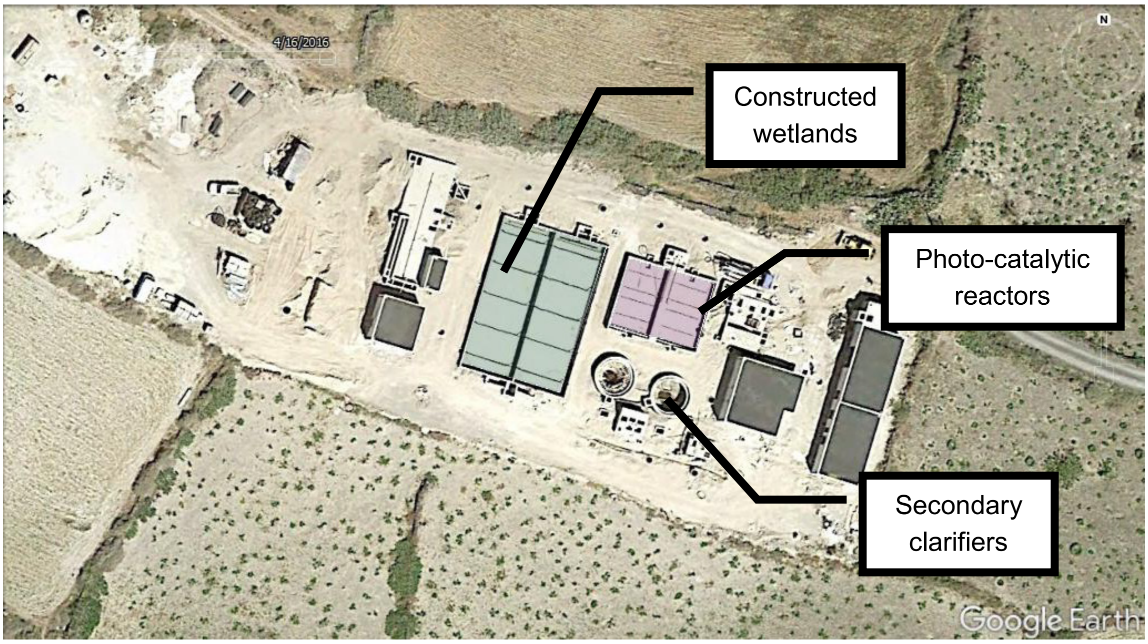


Figure 19 Top view of the WWTP under construction at Thirasia (Source: Google Earth, 2016).

4.2.2 Operation results

As it is the first central WWTP on the island of Thirasia, the sewer system and connections of houses needs to be developed successively. The operation of the two CW started in April 2017 and February 2018 respectively, whereas only one line of the photo-catalysis is operated since April 2017. By end of August 2018 a maximum daily inflow of ~60 m³/d was measured. From 2017 to 2018 the number of connected households in August was doubled. Seasonal fluctuations due to the impact of tourism are apparent as shown in Figure 20. Heavy rainfall at the end of November 2018 led to high inflow rates of 223 m³/d and 240 m³/d on two days. For evaluation only data from 2018 have been used here.

In October 2018, a new sampling point after the photo-catalytic reactor was implemented (equal to influent of CWs), so that the efficiency of the CWs could be assessed for this short period of time.

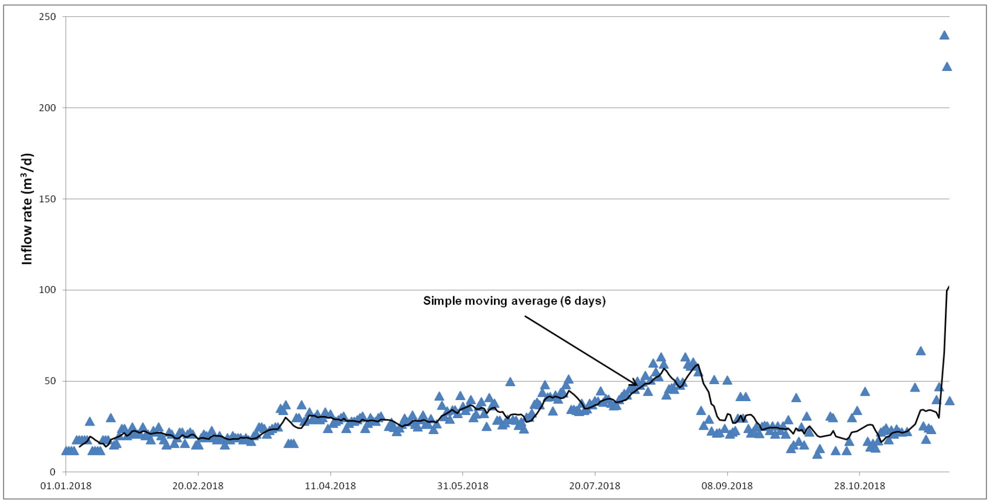


Figure 20 Inflow rate [m³/d] of the WWTP at Thirasia and a simple moving average (6 days) in 2018.

4.2.2.1 Bulk organic parameters, TSS and nutrients

The following standards have to be met by the effluent of the WWTP for water reuse (see Table 17).

Table 17 Limits set by the Greek Reuse legislation for restricted and unrestricted irrigation.

Pollutants	Limits set by the Greek Reuse Legislation (Unrestricted irrigation)	Limits set by the Greek Reuse Legislation* (Restricted irrigation)
pH	6.5 – 8.5	6.5 – 8.5
Dissolved oxygen (mg/L)	-	-
Turbidity (NTU)	≤ 2 median	-
BOD_5 (mg/L O2)	≤ 10 for 80 % of samples	≤ 25
COD (mg/L O2)	- (125)	- (125)
TSS (mg/L)	≤ 10 for 80 % of samples	≤ 35
TN (mg/L)	≤ 45	≤ 45
TP (mg/L)	-	-
Chloride ions (mg/L)	≤ 350	≤ 350
Conductivity (µS/cm)	≤ 3000	≤ 3000
Boron (mg/L)	≤ 2	-
Residual Chlorine (mg/L)	≥ 2	-
Sodium Absorption (%)	SAR ≤ 9	SAR ≤ 9
Escherichia coli (EC/100mL)	≤ 5 for 80 % of samples ≤ 50 for 95 % of samples	≤ 200 median

During this operation period end of 2018 with a flow rate of 25 m^3/d a very high COD reduction of 85 % was achieved by primary sedimentation. Another 64 % of the remaining COD was removed by the photo-catalysis unit (dosing TiO_2 plus aeration and secondary clarifiers). The CWs even reduced another 36 % compared to the preceding step. Final removal by ultrafiltration (UF) was another 39 % (UF refers to final effluent, after chlorination).

All in all, there was a 98 % COD reduction starting with a high average COD of 1,596 mg/L in the influent for the data from Sept. 20^{th} until Nov. 28^{th} 2018 (see Figure 21). A similar picture goes for BOD_5 (not shown) and TSS removal (see Figure 22). In this period with only 20 % of hydraulic capacity in operation, the treatment goals had been reached after the photo-catalysis unit.

There was a significant nitrogen removal by the horizontal flow wetland (see Figure 23). This behavior would be expected if nitrates were formed in the previous stage. But in fact, the average concentration of nitrate in the effluent of the photo-catalytic unit in this period was only about 7 mg/L and in the effluent of the wetland still around 5 mg/L. As depicted in Figure 24 the reduction of ammonium amounts to 75 % within the wetland (from 45 mg/L down to 11 mg/L on average). The effluent of photo-catalysis unit had almost no organic nitrogen (2 mg/L) which increased in the wetland up to 13 mg/L at the outlet.

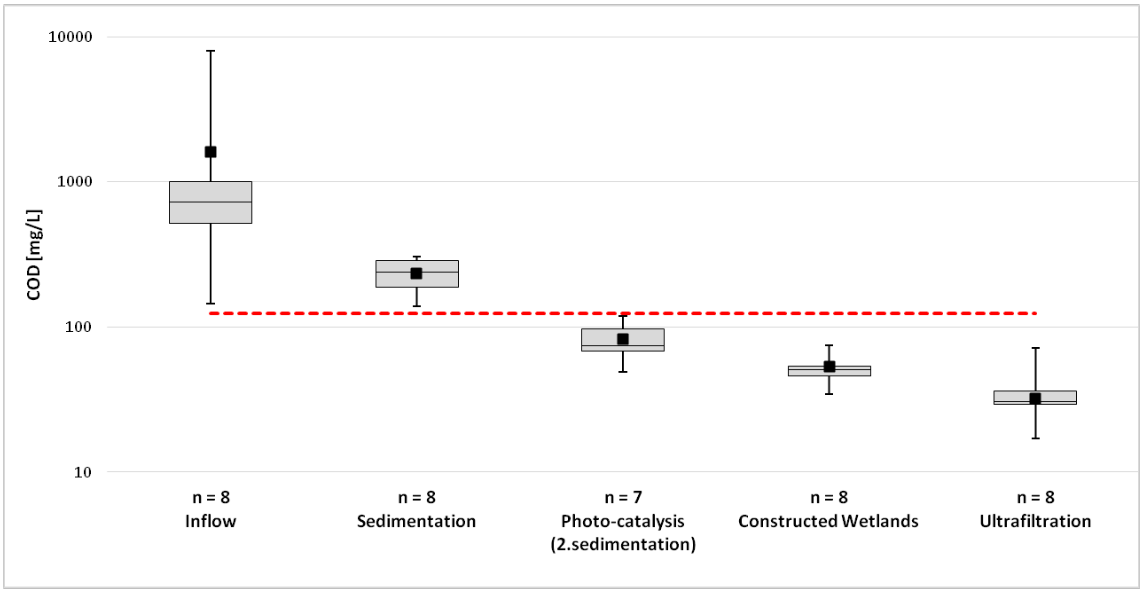


Figure 21 Boxplots of COD concentrations at different sampling points at WWTP Thirasia, Oct-Nov 2018. Boxplots with median, 25- and 75-percentil; black dots: average; whiskers: minimum and maximum; red line: limit for restricted and unrestricted irrigation.

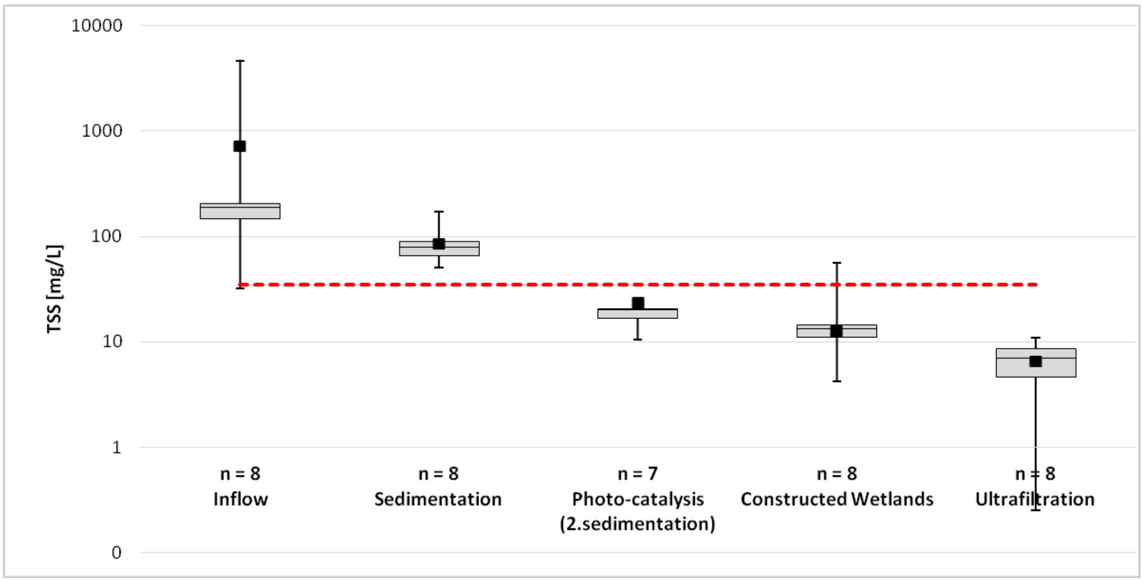


Figure 22 Boxplots of TSS concentrations at different sampling points at WWTP Thirasia, Oct-Nov 2018. Boxplots with median, 25- and 75-percentil; black dots: average; whiskers: minimum and maximum; red line: limit for restricted irrigation.

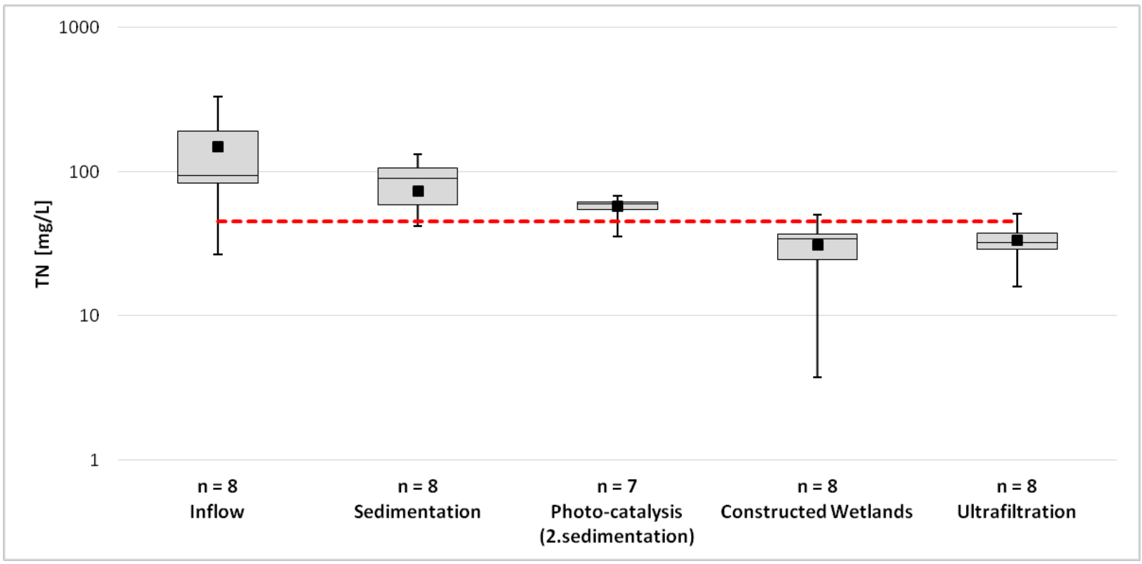


Figure 23 Boxplots of TN concentrations at different sampling points at WWTP Thirasia, Oct-Nov 2018. Boxplots with median, 25- and 75-percentil; black dots: average; whiskers: minimum and maximum; red line: limit for restricted and unrestricted irrigation.

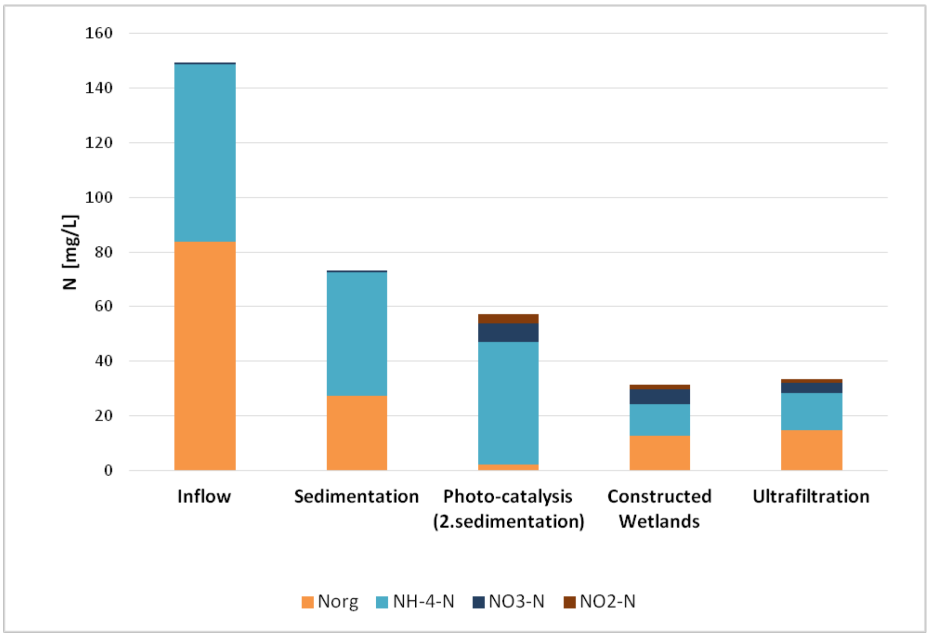


Figure 24 Nitrogen fractions at different sampling points. Average values, n = 8, Thirasia, Oct-Nov 2018.

4.2.2.2 Field observations

In summer 2018 ponding occurred on top of the horizontal subsurface flow wetland system. The water table could reach 10 cm above filter surface and there was an unhindered surface flow towards the outlet. It is not known what percentage of water went above or below ground. Algae growth was seen as well (see Figure 25).

Figure 25 **Ponding in horizontal subsurface flow wetland system (16-08-2018).**

Geotextiles had been used for separation of the different layers (Figure 26). It is well known that this kind of material clog immediately in treatment systems. The correct way of construction is the implementation of transition layers from coarse to fine. A transition layer acts as a bridging layer and prevents filter media from washing into the drainage layer (Payne et al., 2015; Water by Design, 2014).

Figure 26 **High density geotextile for separating rocks/ sand and visible intrusion of salt into concrete wall from sea sand used in the filter.**

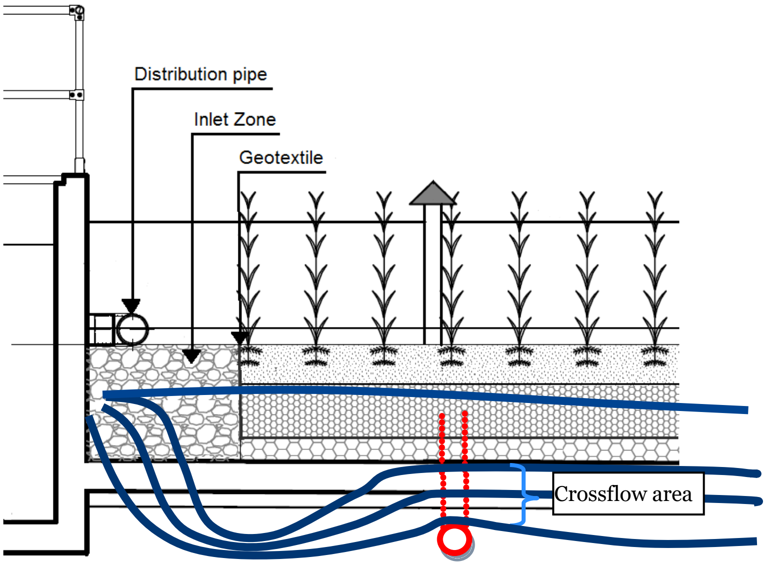


Figure 27 Drawing of inlet zone of the CWs (not to scale) with rocks and geotextile and possible flow paths.

The filter layout (Figure 27) demonstrates that not the whole transection may have been useable for horizontal flow due to infiltration resistance. Five ventilation pipes are crossing the flow direction within the water saturated zone and can have no positive effect there - unless air can be introduced by pressure. Gravity flow in horizontal subsurface flow systems relies on sufficient cross sectional area and sufficient water head. Possible accumulation of sludge within pores can even reduce the flow area.

4.2.2.3 *Influence of TiO_2 dose on COD and BOD_5 removal in photo-catalysis*

During a trial period from July to end of November 2018 various TiO2 doses (0.1-0.5 kg/m³) as well as no addition of TiO2 (only aeration) were tested in the photo-catalysis basin. For COD and BOD_5 similar removal were observed independent of the TiO2 dosage. The results showed that only aeration of the pretreated wastewater is a valuable treatment step (Figure 28). In addition, the expenditure for operation on this site could be reduced from ~3.02 €/m³ when dosing 0.1 kg/ m³ TiO2 down to ~1.44 €/m³ without dosing TiO2.

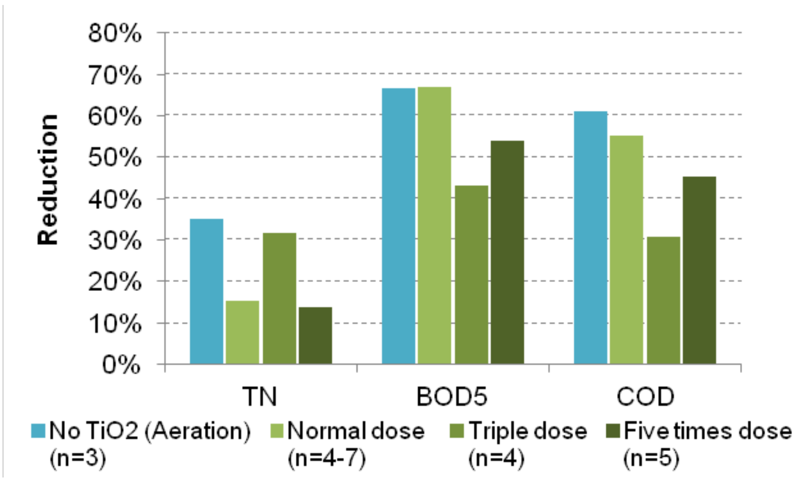


Figure 28 Mean removal of TN, BOD_5 and COD in the photocatalysis stage under different dosages of TiO_2. No TiO_2: Jul-Aug 2018; Normal dose (0.1 kg/m³ TiO_2): Mar-Jun 2018; Triple dose (0.3 kg/m³ TiO_2): Sep 2018; Five times dose (0.5 kg/m³ TiO_2): Oct-Nov 2018.

4.2.3 Conclusion

Other than planned the wetland in this case was functioning as surface/subsurface flow system. In this combination with an activated pre-treatment including aeration plus chemical precipitation (photo-catalysis) the function of the wetland mainly was ammonium removal. It is not clear in what way the ammonium reduction was achieved. This partly could occur due to volatilization of ammonia over the water surface (at high pH) or due to regular nitrification/denitrification processes. Oxygen level in the free water body was rather low (1 mg/L) and in the underground near to zero. Other processes such as anaerobic ammonium oxidation occur in combination with high nitrite concentrations. However, in this combination and with an HLR of 0.063 m/d (25 m^3 per 400 m^2) approx. 50 % TN was removed by the surface/subsurface flow wetland.

One obvious reason for filter clogging was the use of geotextile as filter media between sand and gravel layers/rocks in the inflow construction. A further reason was the layout of the horizontal filter with a limited cross-sectional area. Another critical point of a horizontal subsurface flow wetland could be an unfortunate discharge of sludge from the clarifiers in case of hydraulic overload. This would lead to fast buildup of sediment in the gravel pores.

If the hydraulic load of the treatment plant increases (e.g. factor 5) the water table in the surface flow wetland probably will have to be lifted in order to increase retention time. This will optimize nitrogen removal. Probably vegetation will also change at this water level. Planting common reed (*Phragmites australis*) would be a good alternative to the existing plant (*Arunda donax*).

As pointed out it may be a more economic way of operation to stop dosing TiO2 to the technical treatment reactor. In this case the combination of activated sludge treatment and surface flow wetland post treatment would be a good solution. This kind of pre-treatment reduces the organic loading rate to the filter and allows connecting more people in future.

The WWTP in Thirasia demonstrates very high COD removal (~95 %, on average in 2018). A variety of engineered solutions are combined with a relatively small wetland. In future the optimum balance between natural treatment (wetland size) and engineered solution (energy and chemical need) has to be found.

Table 18 Key Takeaways from CW + TiO_2-activated pre-treatment and post-treatment by UF.

	Pre-treatment + photo-catalysis/ aeration	HSSF CW	Ultrafiltration + Chlorination
Function (+ removal/ - increase)	+ BOD + COD + TSS + Turbidity + TP + N_{org} - NO_2-N	+ BOD + COD + NH_4-N + NO_3-N + NO_2-N - N_{org}	+ BOD + COD + TSS + *E. coli*
Risks of malfunction	– Sludge disposal due to overload	– Filter clogging due to geotextile in inlet zone – Surface flow due to filter clogging and limited cross-sectional area – Filter clogging by overload (sludge)	– Scaling due to iron, aluminium, calcium – Fouling due to fats, fibres and biomass – Lack of chemicals for cleaning
Operation and maintenance needs	– Only aeration is sufficient to achieve good COD, TSS and N removal (no TiO_2 necessary)	– Filter material analysis – Calibration of throttle valve – Regular inspection of water distribution (monthly) – Flow and quality monitoring – Plants care	– Membrane cleaning – Regular control of transmembrane-pressure (daily)

4.3 Retention and filtration wetland for dual use - "*RFS*[plus]"

The WWPT of Rheinbach in North-Rhine-Westphalia, Germany is connected to the river Wallbach, a tributary of the Swist river. In order to improve the water quality of the sensitive river system a pilot plant has been used to demonstrate the feasibility of a new combined filter system for the treatment of combined sewer overflow (CSO) *plus* advanced organic micropollutants (OMP) removal from secondary effluent. Its positive results have led to the construction of a full scale combination for dual CSO plus tertiary treatment use for 27,000 PE called "*RSF*[plus]".

4.3.1 Treatment concept

Retention soil filters (RSFs) are a specific form of vertical flow wetlands for the treatment of stormwater and/or wastewater (Brunsch et al., 2018). This is state of the art in Germany. As regular CWs for CSO often suffer from long dry periods without any water, the key innovation is the flexible use of the RSF. The new combination of systems for CSO treatment with WWTP effluent polishing reduces chemical and microbiological contamination of the receiving river used for recreational purposes and irrigation. Innovative substrate additions as granular activated carbon (GAC) improve the removal of targeted pollutants.

During dry seasons the RSF serves as tertiary treatment step to purify the WWTP effluent for 27,000 PE, while during heavy storm events the RSF is used to treat the CSO of the connected catchment area (Figure 29).

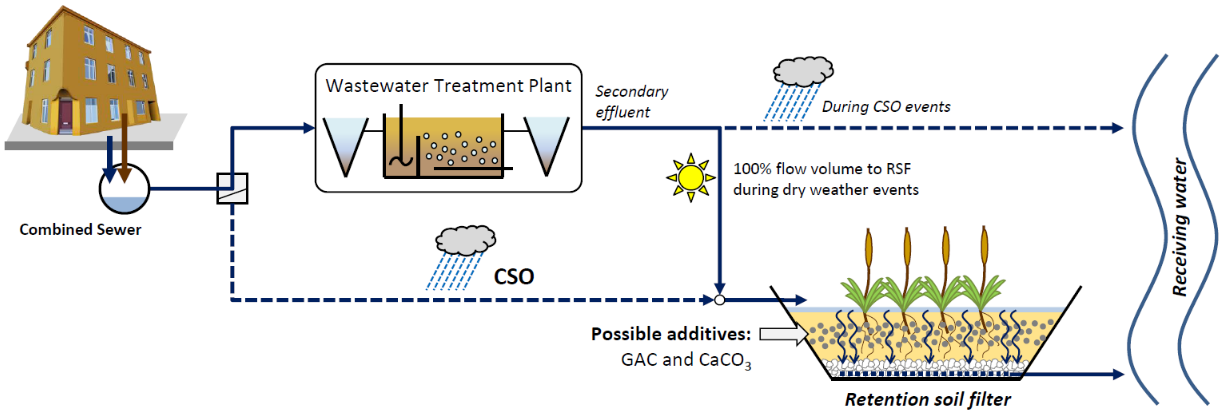


Figure 29 Flow scheme of full scale system at Rheinbach, Erftverband.

As the receiving stream „Wallbach" contains up to 100 % wastewater load during dry weather, very strict regulation on the WWTP effluent quality are foreseen. Especially nutrients and TSS shall be very low as depicted in Table 19. Until now there is no regulation for OMP in surface waters. The secondary WWTP is equipped with a nitrification/denitrification stage and a phosphor elimination stage downstream the second clarifier.

Table 19 Characteristics of the effluent of the Rheinbach WWTP (without RSF).

Parameter	DOC	COD	BOD_5	TSS	NH_4-N	TN	TP
Unit	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
Value	5.9	< 20	< 10	4.8	< 1	< 18	< 0.4

Figure 30 Aerial view of the WWTP Rheinbach and the full scale *RSF*^{plus} under construction.

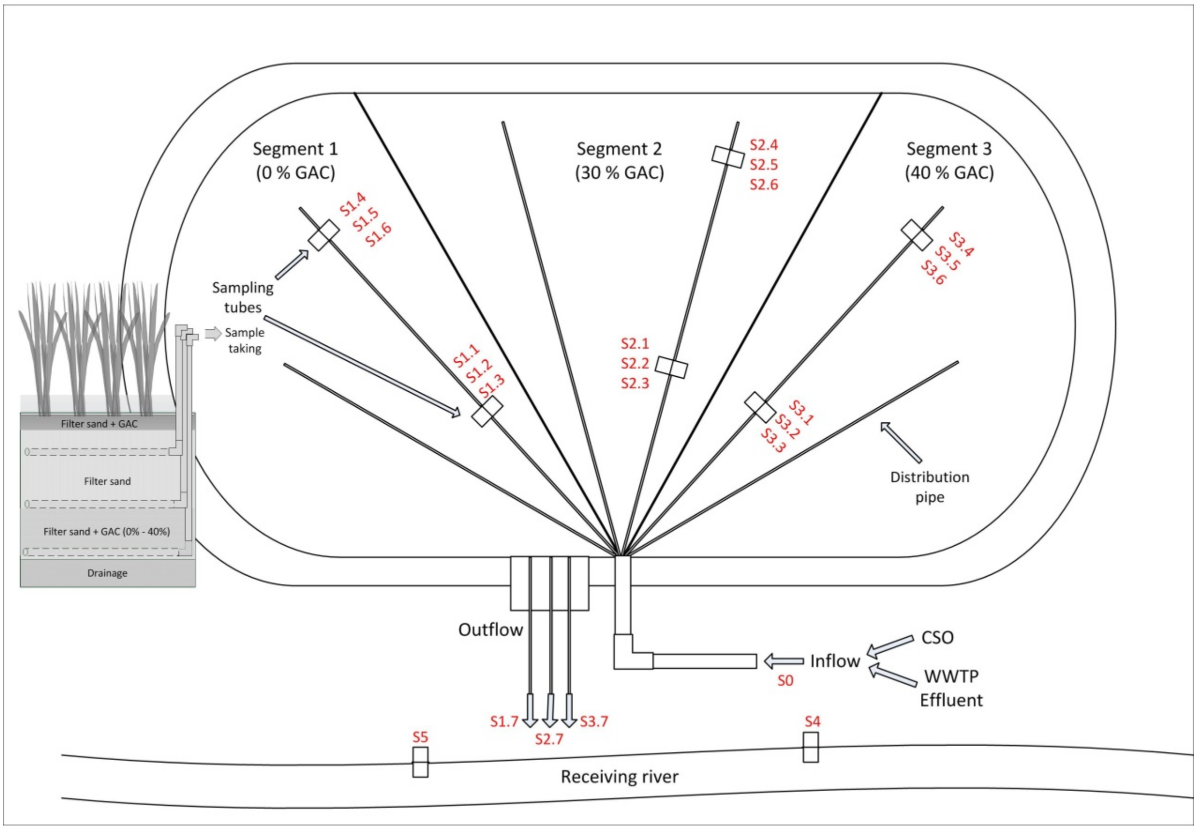


Figure 31 Schematic view of the setup of the full scale demonstration site in Rheinbach.

The filter has a total treatment area of 5,000 m² divided into three cells equally sized (Figure 30; Figure 31). These segments are fed intermittently so that each cell has a 24 h feeding period and a 48 h dry period to guarantee aerobic conditions in the filter. The water level is flexible from 0-50 cm above filter surface. During CSO events there is a constant loading of the entire filter area. The duration and the water level of each batch depend on the volume of each CSO event; a minimum of 18 h dry period after the CSO is recommended. The maximum retention volume accounts for 12,300 m³, the maximum water level is ~2 m above filter surface.

Sand 0-2 mm (kf-value = $1.8*10^{-4}$ m/s) enriched with $CaCO_3$ that is used to prevent remobilisation of accumulated heavy metals is applied as filter material. In two of the segments 20 vol-% GAC is added to the upper layer (0-10 cm) and 30 vol-% GAC (in segment II)/40 vol-% GAC (in segment III) is added to the lower layer (70-100 cm). There is a drainage layer of 25 cm filled with gravel 2-8 mm. Like conventional RSF, the total filter depth is 1 m and it is planted with *Phragmites australis*. The design infiltration rate is 0.03 L/s/m² corresponding to a hydraulic loading rate of ~2.6 m/d for the active cell and to ~0.865 m/d for the entire filter for dry weather periods.

4.3.2 Operation results

A pilot plant study with three pilot-RSF of 1.5 m² each was conducted to test the flexible use of RSFs. The filter depth corresponds to the full scale RSF, hence operation results are comparable. The operation of these pilot-RSFs started in 2014 and 2015, respectively. In the first 3.5 operational years it was tested if these RSF are suitable as post-treatment for WWTP effluent. In contrast to the full scale RSF, the pilots were fed during 28 h and allowed a dry phase of 56 h. Tracer tests at the pilot plant showed a retention time of 3.25 h at a filtration rate of 0.03 L/s/m².

Two pilot-filters contain conventional RSF material, taken from full scale RSFs that are in operation since 2005. The third filter is filled with sand with 22 % $CaCO_3$ and two additives: 13 vol-% biochar in the upper layer (0-10 cm) and 43 vol-% GAC in the lower layer (60-90 cm).

The results from 4 operational years showed that RSF with and without GAC are a suitable post-treatment step for wastewater. It was observed that the RSF with GAC performed even better in reducing OMP and DOC. In chapters 4.3.2.1 to 4.3.2.3 results from dry weather events are shown. Chapter 4.3.2.4 describes the capacity of the conventional pilot-RSF during artificial CSO events.

4.3.2.1 Organic parameters (tertiary operation during dry weather)

The comparison of the conventional pilot-RSF and the pilot-RSF with GAC showed that the RSF with GAC reduced DOC more efficiently than the conventional one. Median removal rates amount to 23 % and 76 %, with median effluent concentrations of 4.20 mg/L and 0.74 mg/L, respectively (Figure 32).

Constant removal of DOC in the conventional pilot-RSF was observed, while there was a decrease in reduction in the pilot-RSF with GAC. The lowest removal was ~50 % at 800 treated bed volume (tBV). Further the removal increased to rates >60 %.

Median COD removal was 16 % and 36 % in the pilot-RSFs without and with GAC, respectively. BOD_5 inlet concentration was already below limit of quantification (<LOQ; <3 mg/L).

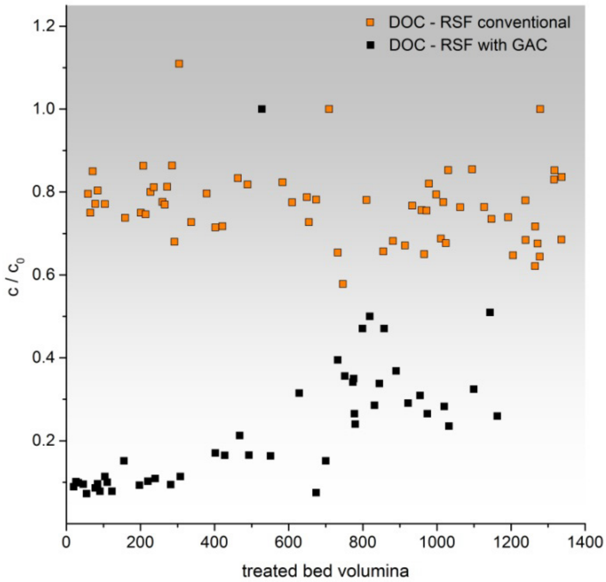


Figure 32 Relative effluent concentration and treated bed volumina (tBV) for DOC – conventional RSF vs. RSF with GAC.

4.3.2.2 Organic micropollutants (tertiary operation during dry weather)

During the test phase, it was observed that the pilot-RSF with GAC performed always better than the conventional one in reducing OMPs (Figure 33). The removal of OMPs in the pilot-RSF without GAC varies from 0-80 %. Metoprolol (MET), 4-Hydroxy-diclofenac and Valsartan (VAL) for example are removed very well; Carbamazepine (CBZ), Candesartan (CAN) and Diatrizoate (Amidotrizoic acid) were not or nearly not removed within the conventional pilot-RSFs. In the effluent of the pilot-RSF with GAC only few OMPs were detected above LOQ.

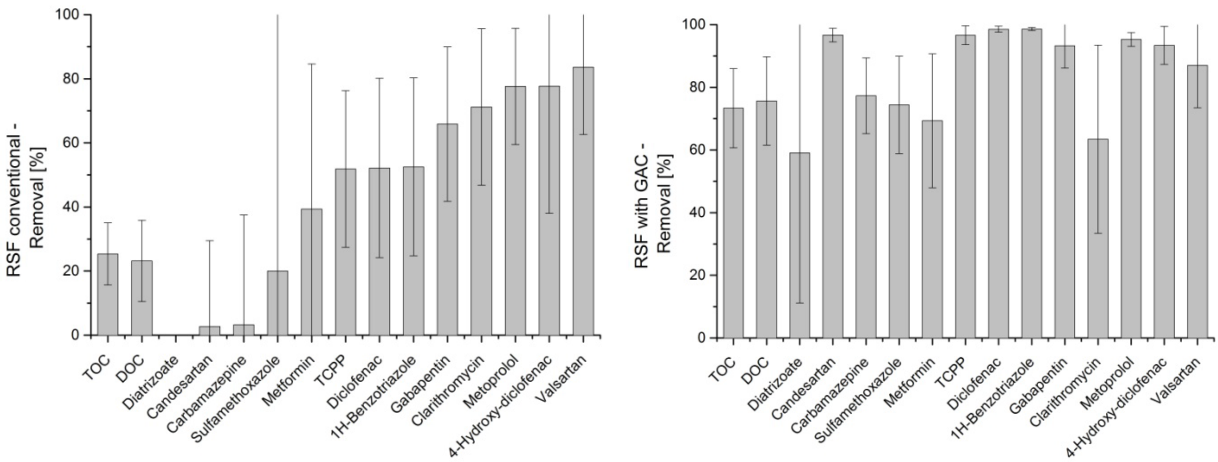


Figure 33 OMP removal (%) in conventional RSF (left) and RSF with GAC (right).
Columns: median; error bars: standard deviation; left: n=27-85, 09/2014 – 11/2018; right: n=14-56, 04/2015 – 11/2018; values < LOQ = LOQ.

Looking at the OMP removal of the pilot-RSF with GAC related to the treated bed volumina (referred to the GAC layer) a partial breakthrough of Metformin and Amidotrizoic acid is apparent. All other investigated micropollutants still show removal >80 % (data not shown).

The conventional pilot-RSF by contrast seems to need a start-up phase of at least 0.5 years, depending on the investigated OMP, to reach optimal removal for certain micropollutants, e.g. 1H-Benzotriazol (BTA), Sulfamethoxazole and MET. Figure 34 shows exemplarily the removal of MET (left) and Sulfamethoxazole (right) from 07/2014 to 12/2018.

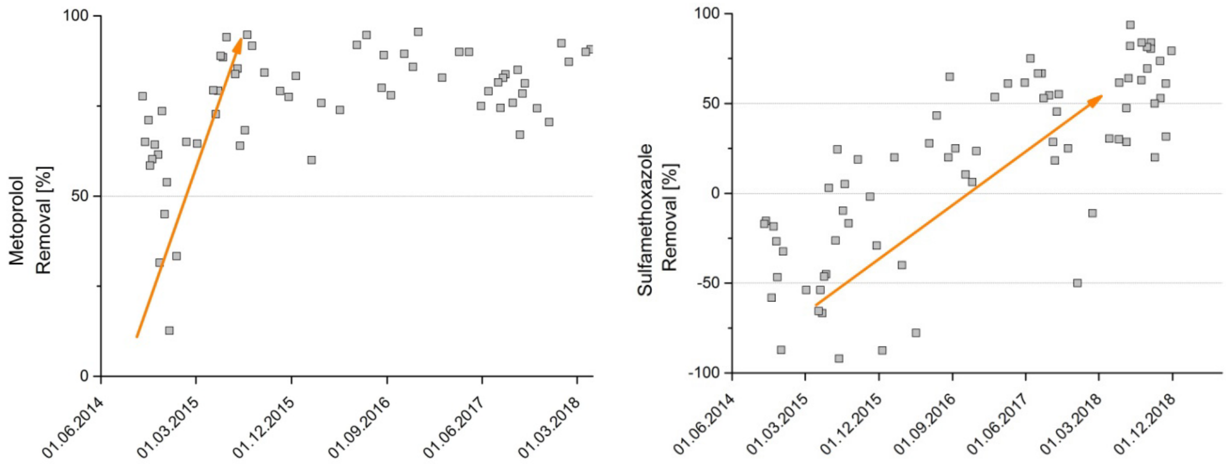


Figure 34 Removal (%) of Metoprolol (07/2014–03/2018) and Sulfamethoxazole (07/2014-12/2018) in the conventional RSF.

Furthermore, the removal capacity in the conventional pilot-RSF for some OMPs was observed along the feeding phase of 28 h with better removal in the beginning of the feeding phase (e.g. 1H Benzotriazole (BTA), Metformin (MEF), Diclofenac (DCF)) (data not shown). The question arises whether overall removal could be enhanced by shortening feeding periods. This could be a subject to further research.

Additionally to the in- and outflow analyses, a mass balance of BTA and MET was carried out. Therefore, filter material from the conventional pilot-RSF was analysed for OMPs. The theoretical accumulated concentrations in the filter material for 27 month, calculated from removal values, were compared to the actual concentrations measured in the filter material in Nov. 2016. The results showed no actual accumulation of measured micropollutants in the filter material, as the difference between theoretical and measured concentrations amounts to 2 log-units. Potentially biotransformation is one of the main reduction processes in RSF beside sorption and plant uptake (Brunsch et al., 2018).

4.3.2.3 Disinfection (tertiary operation during dry weather)

E. coli has been tested as relevant indicator organism for disinfection efficiency. Starting from rather low inlet concentrations of 10^3 MPN/100 ml after secondary treatment a median removal rate of 1 log-unit was reached during tertiary treatment by the pilot systems (Figure 35). This is an expected order of magnitude for this kind of treatment wetland. Apparently, there was a slight difference between the use of activated carbon or not with regard to the 95th percentile.

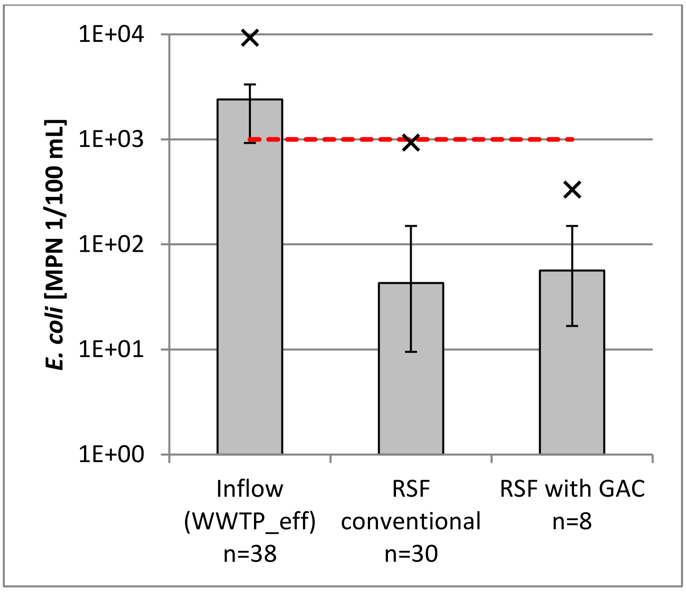


Figure 35 Concentrations of *E. coli* at different sampling points; columns: median, error bars: 25th / 75th percentiles, crosses: 95th percentile.
Red line: criteria for good quality according to EU BWD.

4.3.2.4 Simulated additional CSO events (alternating operation)

A trial phase with various artificial CSO events additionally to the existing feeding cycle was performed on one of the conventional RSF pilots; the second conventional filter served as control site. During the artificial CSO event, the test filter was impounded for 2.5 h, followed by a 3 h drainage period. The CSO feeding volume amounted to 4 L per trial. Samples were taken as composite samples. The aim was to assess efficiency of CSO treatment and its potential effects on tertiary treatment during dry weather situations. First results showed that the RSF is suitable for this flexible use. Additionally, to the regular OMP removal the filter can reduce nutrients, organic bulk parameters, *E. coli* and TSS during CSO. It was found that the resting phase between CSO treatment and WWTP effluent treatment seems to be an important operational parameter for removal of some OMP.

Nitrogen, Organic bulk parameters, TSS

The high NH4-N and org.-N input of the CSO was largely removed by the conventional RSF pilot (mean ammonium removal of 97 % and organic N removal of 87 %) (Figure 36, left).

During CSO events high removal rates of particulate organic carbon (POC), DOC and TSS were observed (average 99 %, 78 % and 95 %, respectively). So far no negative effect of CSO events on tertiary treatment efficiency concerning TOC and TSS is apparent after (Figure 36 , right; data for TSS not shown).

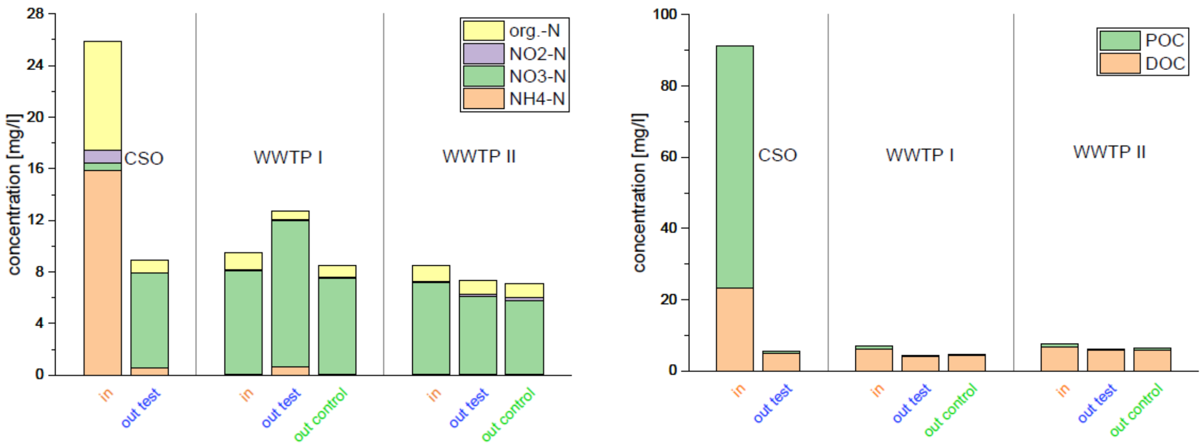


Figure 36 Mean nitrogen (left), DOC and POC (right) concentration in the in- and outflow of the CSO event, scenario I (WWTP I) and scenario II (WWTP II) on conventional pilot-RSF (without GAC; "out test"); "out control" equals to control-RSF (n=4).
CSO: feeding with artificial CSO; WWTP I: scenario I with 18 h dry period following the CSO feeding before WWTP effluent feeding; WWTP II: scenario II with no dry period.

Organic micropollutants

As OMP have very diverse characteristics, the removal mechanisms and rates differ during CSO events and the subsequent WWTP effluent feeding cycle. For some OMPs similar removal during CSO and subsequent WWTP effluent polishing was detected (e.g. BTA, DCF). The flexible use of RSF possibly might not have any disadvantages in the removal of micropollutans.

From comparing the two scenarios, (I) 18 h dry period after CSO treatment and (II) no dry period between CSO and tertiary treatment cycle, it seems likely that the removal rates for some OMP decrease if the dry resting phase is missing (data for selected OMPs shown in D3.2).

Disinfection

The test-RSF showed good removal efficiency of *E. coli* by ~3 log-units during CSO, but a significant negative influence on the removal capacity during subsequent WWTP effluent feeding was noted (Figure 37). The control filter reduced *E. coli* by 96 % to 93 % (~1 log-unit), whereas there was no removal of *E. coli* determined in the test filter after CSO events.

This is clearly a contamination by the high concentrated sewage overflow compared to the low concentrations of secondary treated sewage. The high removal rate measured at instant CSO event may be due to the previous low concentration and water exchange within the filter pores.

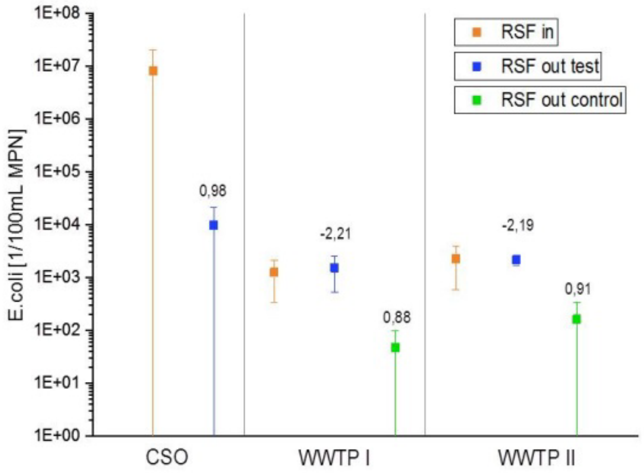


Figure 37 Mean *E. coli* concentration in the in- and outflow of the CSO event, scenario I (WWTP I) and scenario II (WWTP II) on conventional pilot-RSF (without GAC; "out test"); "out control" equals to control-RSF (n=4).
CSO: feeding with artificial CSO; WWTP I: scenario I with 18 h dry period following the CSO feeding before WWTP effluent feeding; WWTP II: scenario II with no dry period; numbers indicate average removal rate; error bar show standard deviation.

4.3.2.5 Influence of operational parameters

The influence of different operational parameters was tested and analysed during the 4 trial years, among others seasonality and contact time. It became clear that there's no significant relation between seasonality and reduction for most OMPs (Brunsch et al., 2018). While an extended contact time had a great influence on the removal capacity. A decrease of the HLR down to 0.01 L/s/m² led to better removal in the conventional RSF. The removal rates for Galaxolide (HHCB), Metoprolol (MET), Diclofenac (DCF), Tris(2-chlorisopropyl)phosphate (TCPP), 1H Benzotriazole (BTA) and Sotalol (STL) were 1.2 - 2.9 times higher at low HLR of 0.01 L/s/m² compared to 0.03 L/s/m². Removal of DCF and MET could even reach levels close to the removal in the RSF with GAC as shown in Table 20.

Table 20 Average removal of TCPP, HHCB, DCF, STL, BTA, MET with different filtration rates (0.01 L/s/m² and 0.03 L/s/m²) in the conventional RSF and the RSF with GAC (n=4).

Filter	Type	HLR [L/s/m²]		TCPP	HHCB	DCF	STL	BTA	MET
RSF 3	RSF with GAC	0.01	Av. removal [%]	95.2	93.5	98.9	95.2	99.1	96.1
RSF 2	RSF conventional	0.01	Av. removal [%]	63.7	84.9	92.8	73.2	79.3	94.1
RSF 1	RSF conventional	0.03	Av. removal [%]	39.2	69.2	63.1	25.3	46.6	80.6
			Ratio RSF2/RSF1 [-]	1.6	1.2	1.5	2.9	1.7	1.2

Comparing RSF1 and RSF2 reveals the relevance of retention time or wetland size for these parameters. RSF3 shows the potential of reactive filter media such as technical GAC for intensification of the process. It is expected that the same removal rates for RSF3 will be reached at 0.03 L/s/m² because the reaction speed of the activated carbon is decisive in this combined process.

4.3.2.6 *Influence of filter depth and filter media*

Reduction rates for OMPs in various filter depths were compared. The results of the different filter depths showed a clear dependence of removal on the specific characteristics of the filter material. Greatest positive influence on the cleaning performance of the conventional RSF had the amount of organic matter within the uppermost filter layer.

The reduction efficiency in the optimised RSF with GAC was positively influenced by the addition of biochar in the uppermost and of GAC in the lowermost filter layer. Both media increase the sorption capacity of the filter material. The middle parts of the RSFs (ca. 10 – 60 cm) with no substrate additive have only little influences on the OMP reduction rates. Figure 38 shows exemplarily the concentration of BTA and DCF for the pilot RSFs in the different filter depths.

Biochar turned out to increase removal capacities for OMPs and DOC in the first operational years. In the following seasons, the gradual decrease of sorption capacities and related removal pointed out the limited adsorption of this material (Brunsch et al., 2018). GAC is therefore used instead of biochar in the upper layer of the large-scale filter in Rheinbach. The use of biochar also entails the risk of unwanted contaminations entering the retention soil filter.

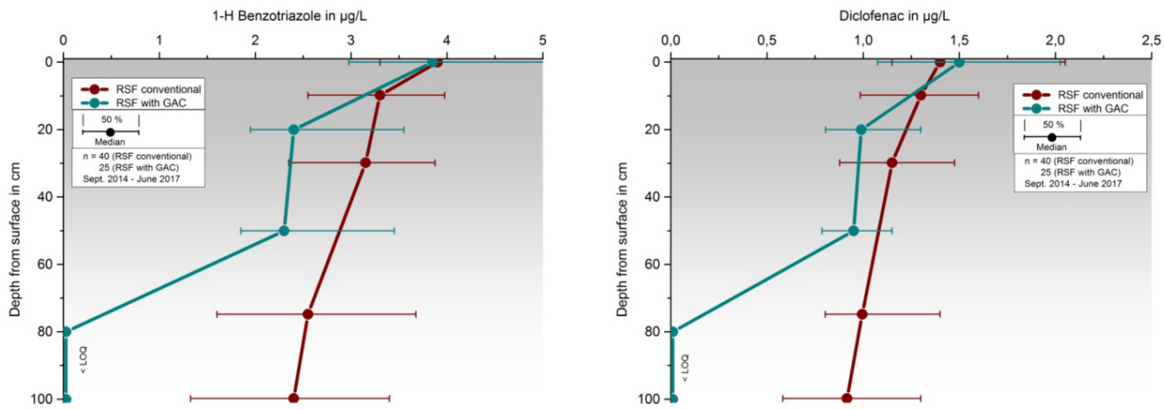


Figure 38 Behavior of micropollutant concentration in RSFs effluent shown on the example of 1-H Benzotriazole (left) and Diclofenac (right).

4.3.3 Conclusion

Synergy effects

The flexible use of a natural treatment system for regular post treatment of secondary effluent plus periodical flooding with untreated combined sewage overflow increases the economic value many fold. RSF has proven to be the most efficient and economic technology for CSO treatment because it integrates storage capacity at low additional cost. At the demonstration site Rheinbach, calculations indicated that the area required for CSO treatment and WWTP effluent polishing was the same, so that no extra space was needed. The flexible use had no or only a slight effect on treatment capacity of the conventional RSF for OMPs, organic bulk parameters and nitrogen.

Adding biochar or GAC to the filter material increases OMP and DOC removal and supports plant growth, as reported in other studies. An organic top layer, which develops in any vertical flow treatment system, additionally improves removal after a certain initial period. This secondary filtration layer is a natural protection for the activated carbon in the lowermost filter layer, so that the lifespan of the GAC's sorption capacity is naturally increased.

The combination of inert mineral natural substrates with activated carbon enables the immediate good removal of OMPs and DOC in the RSF by using the adsorption capacity, even before a biofilm and an organic layer has been formed in the filter system. So a good performance of the RSF is to be expected from the beginning.

Design of WWTP

However due to finite sorption capacity of these organic filter media and limited organic loading rates to soil filters the performance of the secondary WWTP still has a great influence on the tertiary RSFs and its period of use. Thus a flocculation filtration as used in Rheinbach or other measures reducing TSS concentrations are helpful for optimum sizing of the tertiary treatment wetland. According to the German set of rules DWA-A 178 (2017, draft) the annual load of solids to the RSF must be limited to < 7 kg/m²/a (see also MKULNV, 2015).

Design of retention and filtration wetland

Some results indicated that resting periods after CSO treatment with periodic flooding of the wetland are valuable. This is also known from intermittently loaded vertical flow secondary treatment wetlands. The use of three parallel filter cells allows alternating operation with one day operation and two following days rest. The sizing of the tertiary treatment wetland has to evaluate (model) the frequency and level of filter flooding by CSO events as well as the necessary availability for secondary effluent treatment (hydraulic efficiency). With a filtration rate of 0.03 mm/s and a 1/3 operation interval an average daily loading rate of 0.864 m/d of the wetland cell in operation mode would be a safe design.

Table 21 Key Takeaways from dual use *RSF*^{plus}.

	Dual usage – *RSF*^{plus}	Single usage RSF *conventional*
Function (+ removal/ - increase)	CSO treatment phase + OMPs + TOC, COD + TSS + NH_4-N, TN + TP + *E. coli* + Heavy metals Tertiary treatment phase + OMPs + DOC, COD + *E. coli*	CSO treatment + TOC, COD + TSS + NH_4-N + TP + *E. coli* + OMPs + Heavy metals
Risks of malfunction	– Filter clogging by overload – Saturation of AC	– Filter clogging by overload
Operation and maintenance needs	– Calibration of throttle valve – Regular inspection of water distribution (weekly) – Flow and quality monitoring – Automated control system (loading and resting periods) – Plants care	– Calibration of throttle valve – Regular inspection of water distribution (monthly) – Flow and quality monitoring – Plants care

4.4 Treatment Wetland with Ozonation

The 30 years old WWTP Schönerlinde, north of Berlin (Germany), needs to be upgraded for advanced organic micropollutant (OMP) removal and disinfection of municipal secondary effluent. The combination of an ozone reactor plus a treatment wetland shall demonstrate safe and cost efficient treatment. A pilot plant was operated from May 2017 until December 2018 as AquaNES demonstration site No. 12.

There is no regulation for OMP in surface waters but OMP shall not be found in recirculated potable water e.g. from riverbank filtration. Disinfection performance can be assessed based on EU BWD, where quality standards for indicator organisms *E. coli* and Enterococci are defined.

4.4.1 Treatment concept

This pilot plant combined ozonation with two types of vertical flow CW for removal of OMP and microbial indicators. Technical deep-bed filter systems parallel to the CWs were used for performance comparison (Figure 39).

The ozonation unit was operated with a target value for the applied ozone dose of 0.7 mg O_3/mg DOC. During the first months a constant ozone dose of 7.7 mg O_3/L was applied assuming a constant DOC of 11 mg/L in the WWTP effluent. Later a closed-loop control for ozone dosing was implemented based on the online monitoring of the ultraviolet absorption at 245 nm (UVA254) elimination (ΔUVA254). The ΔUVA254 target value corresponding to the desired ozone dose of 0.7 mg O_3/mg DOC was determined to 47 %.

Both CW have a surface area of 11 m² each and were planted with *Phragmites australis* and *Carex acutiformis* in equal parts. In CW1, technical sand is used as filter material (bed depth = 0.55 m, d = 0.2-2 mm). In CW2, coarser filter material (bed depth = 0.8 m) consisting of a homogeneous mix of lava gravel (d = 4-8 mm) and 30 vol-% biochar (d = 8-20 mm) was tested. Both were operated under saturated conditions with filtration rates of approximately 200 mm/d, 400 mm/d and 1000 mm/d in different phases.

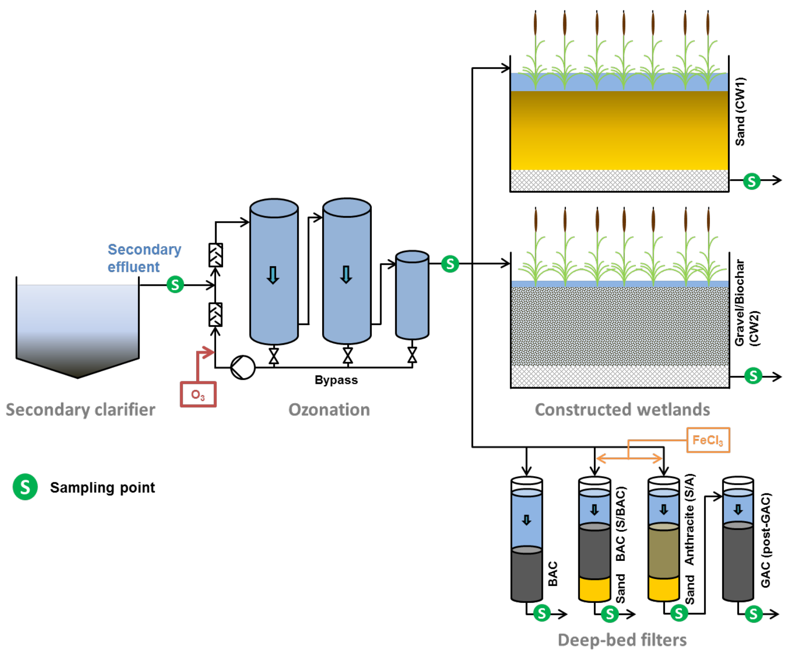


Figure 39 Pilot-plant flow scheme at WWTP Schönerlinde.

4.4.2 Operation results

4.4.2.1 Organic bulk parameters

As shown in Figure 40 almost no removal for DOC was observed during ozonation whereas COD was reduced by 14 % on average. The oxidation products usually have an increased biodegradability which could be demonstrated with additional BOD_5 analyses. In a total of 9 grab samplings the influent BOD_5 of <3-5 mg/L was raised by ~50 % on average during the ozonation process. All post-treatment steps showed substantial reduction for the organic bulk parameters. Average removal for DOC and COD was very similar in CW1 (21.9 % and 32.4 %, respectively) and CW2 (21.4 % and 32.6 %, respectively).

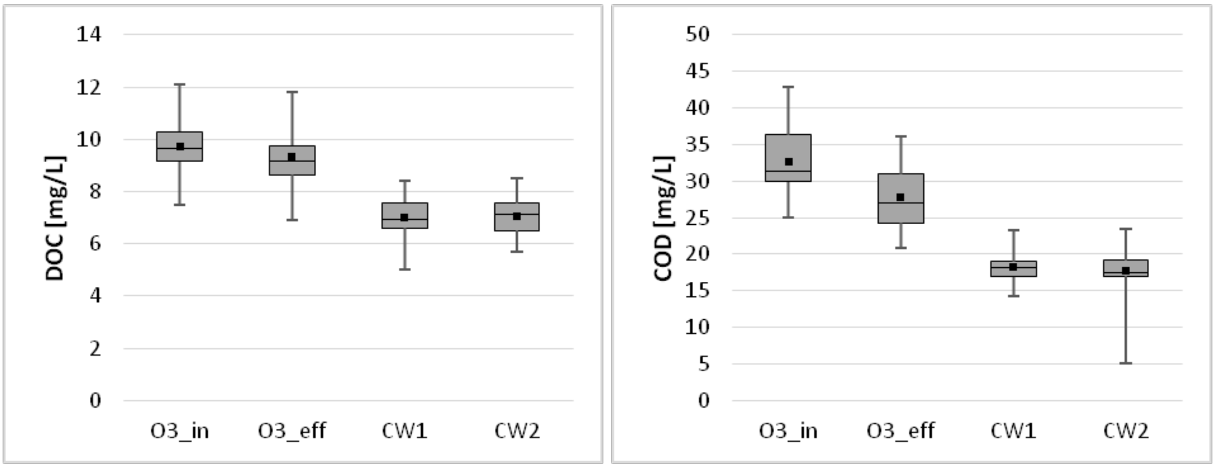


Figure 40 Boxplots of DOC (left, n=22-34) and COD (right, n=22-24) concentrations at different sampling points.
Boxplots with median, 25- and 75-percentil; black dots: average; whiskers: minimum and maximum.

Both CW were operated at different HLR and hence different HRT. As shown in Figure 41 average DOC removal in CW stayed constant during all three operational phases (HLR ≈ 200, 400 and 1000 mm/d; HRT ≈ 48, 24, 10 h). A reduction of HRT did not result in decreased DOC removal.

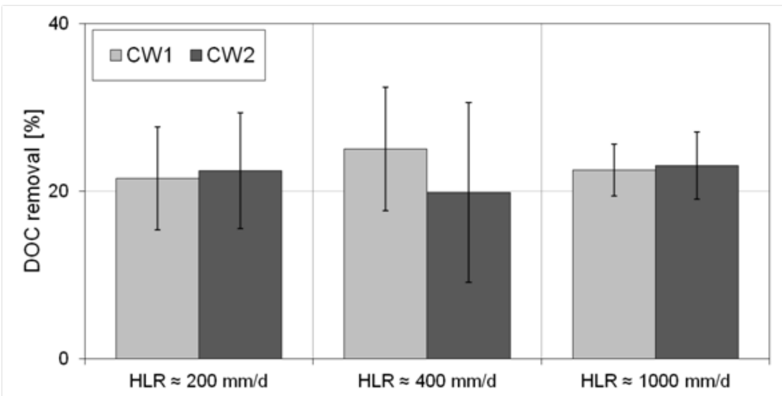


Figure 41 Mean DOC removal with standard deviation (n=5-10) at different HLR in CW.

Additional potential impacting factors on DOC removal were investigated. No correlation was found for water temperature. In contrast, influent DOC concentration could have an impact on the removal (data not shown).

4.4.2.2 Nutrients and TSS

P removal by both wetlands decreased with time (Figure 42). As expected, by the end of the project no reduction or desorption was observed. The Sand filter CW1 had a higher efficiency in the beginning. None of the filters had been designed for P removal.

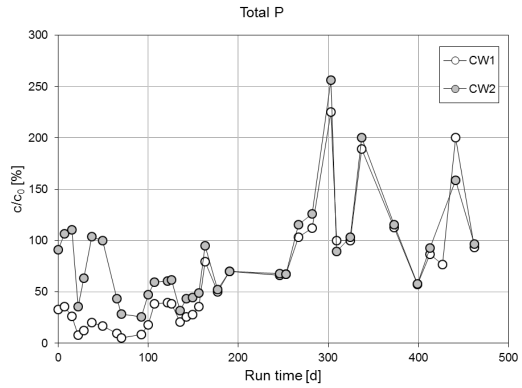


Figure 42 Total P breakthrough curve over run time in CW1 and CW2.

The ammonium nitrogen concentration in the secondary effluent of the WWTP was already at a low level of <0.03-1.5 mg/L NH4-N. But due to the ozonation step the NH4-N concentration at first increased by ~50 %. Still the mean reduction rate in the combined systems was ~26 %. The outlet concentrations of CW1 was <0.03-0.67 NH4-N and of CW2 <0.03-0.53 NH4-N.

In the CWs a slight nitrate removal could be observed (mean removal: CW1 = ~7 %; CW2 = 10 %) (Figure 43, left). Since denitrification is temperature sensitive an increase with water temperature was noticed in both CWs (Figure 43, right). However oxygen level was high in both wetlands due to previous ozonation and denitrification was not expected. Especially in the phase of growth, plant uptake should be in this order of magnitude. Nonetheless removal rate in CW2 was slightly higher, which may be related to biochar effects (carbon source).

The CWs showed a very good performance concerning the reduction of TSS with average removal rates of 93 % (CW1) and 94 % (CW2). TSS concentrations were reduced to 0.15-0.53 mg/L in CW1 and <0.1-1.1 mg/L in CW2. Both CWs performed slightly better than deep-bed filters (Figure 44).

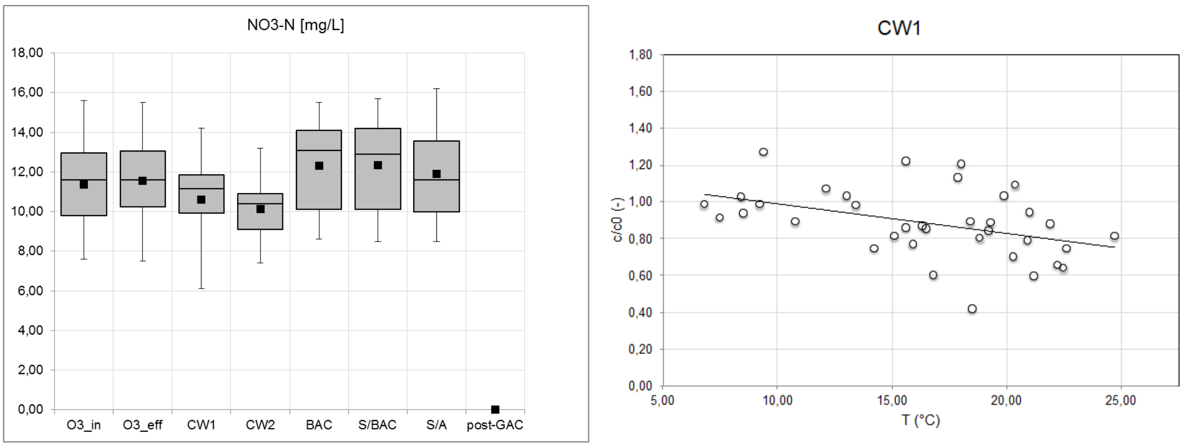


Figure 43 Boxplots of NO_3-N concentrations (n=7-24) at different sampling points (left); correlation of water temperature and NO_3-N reduction in CW1 (representative for both CW) (right).
Boxplots with median, 25- and 75-percentil; black dots: average; whiskers: minimum and maximum. No data for post-GAC.

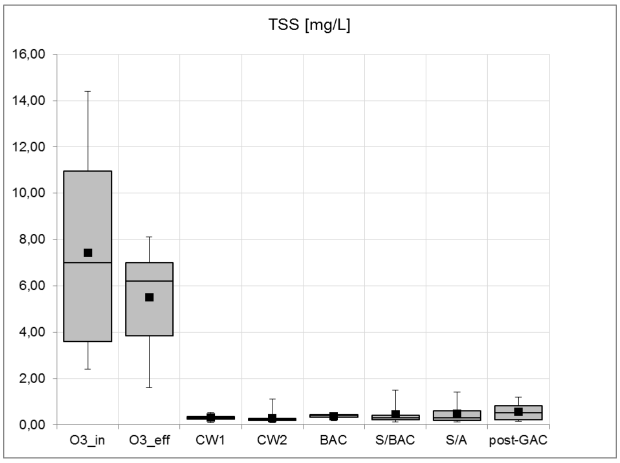


Figure 44 Boxplots of TSS concentrations (n=8-20) at different sampling points.
Boxplots with median, 25- and 75-percentil; black dots: average; whiskers: minimum and maximum.

4.4.2.3 Organic micropollutants

A set of 25 OMP was monitored during the study period. The behaviour of the investigated compounds during ozone treatment varied a lot. As shown in Figure 45 average removal ranged from 99 % for Formylaminoantipyrine (FAA) to ~0 % for MEF and Amidotrizoate (ATS). Removals depicted in Figure 45 were determined by a conservative approach (if value < LOQ it was replaced by LOQ) and it has to be considered that actual removal efficiency might be higher for certain compounds.

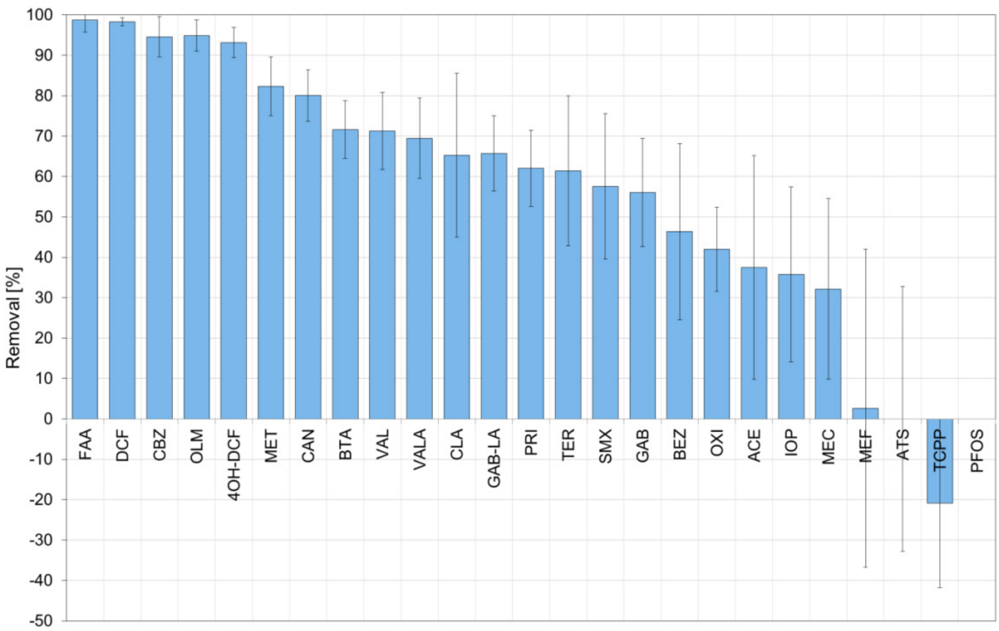


Figure 45 Mean removals with standard deviations (n=8-41) of monitored OMP during ozonation calculated with conservative approach regarding LOQ.
When effluent concentration was <LOQ the value of LOQ was used for calculation.

Due to either low concentrations or recalcitrance only 11 suitable substances remain for the assessment of post-treatment. Figure 46 displays the average removal of these compounds in the CW.

In CW1 significant reductions were only observed for TCPP (66 %) and VAL (61 %). With reductions of 50 % for both compounds the reduction rates in CW2 were even slightly lower.

Additional removal in CW2 that was not observed in CW1 did occur for BTA (90 %), MET (44 %) and Oxipurinol (OXI) (20 %). Since all three compounds are known to be well adsorbable, removal can be explained by adsorption onto biochar.

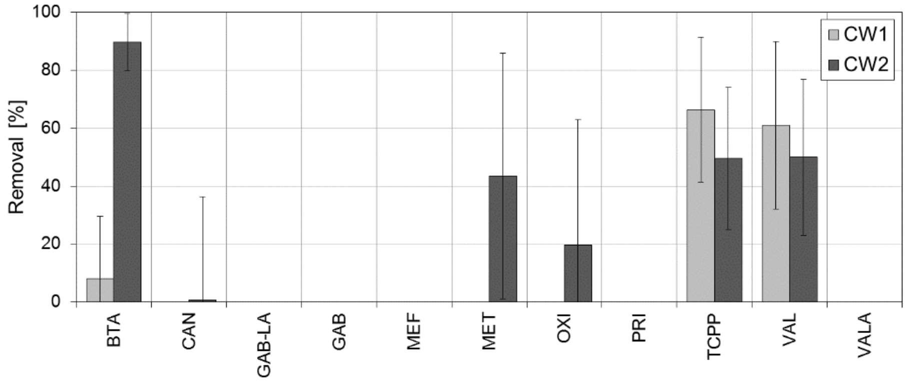


Figure 46 Mean removals with standard deviations (n=8-25) of OMP during post-treatment in CW calculated with conservative approach regarding LOQ.

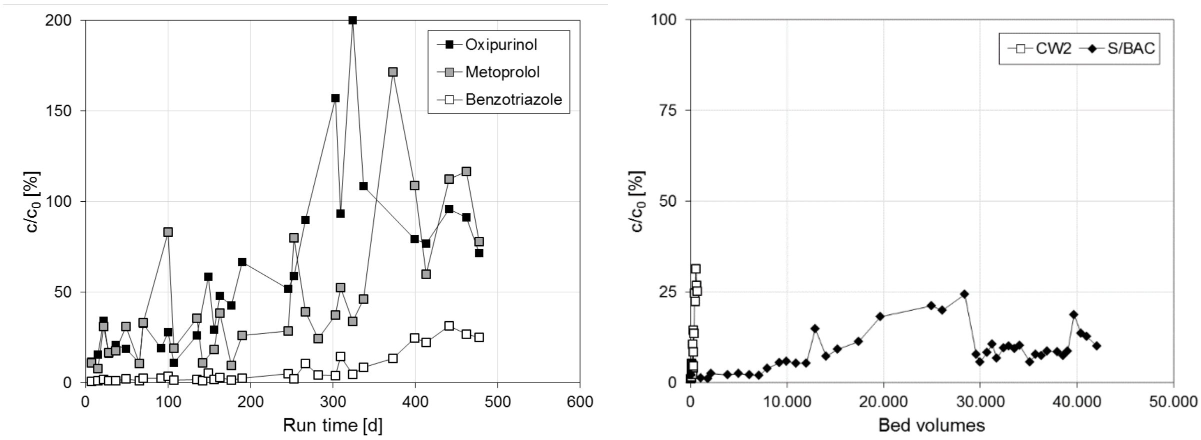


Figure 47 Breakthrough curve over run time for OXI, MET and BTA in CW2 (left) and BTA breakthrough curve over bed volumes (BV) for CW2 and S/BAC filter (right).

It has to be considered that removal efficiency for these substances will decrease with advancing breakthrough, as depicted in Figure 47 (left). Full breakthrough for OXI was observed after ~300 d of run time, while for MET it was reached after ~400 d, for BTA ~30 % of breakthrough was reached by the end of the study period. Results demonstrate that biochar addition to the filter material can be beneficial for removal of well adsorbing compounds. However, when comparing the breakthrough of BTA (plotted over normalised throughput [carbon BV]) in CW2 with the sand/biological activated carbon filter (S/BAC) adsorption capacity of biochar was revealed to be very limited (Figure 47 (right)).

4.4.2.4 Transformation products

Ozonation not only helps to crack persistent organic trace elements but it is also a potential source for transformation products. It was investigated if these can be removed by the additional natural treatment system (CW).

Due to its potential carcinogenic and mutagenic effect of bromate the World Health Organisation (WHO) defined a threshold of 10 µg/L in their Drinking Water Guideline (WHO, 2011). Four samplings with different ozone doses were carried out for ozonation influent and effluent and bromate formation was quantified. As shown in Figure 48 with the target ozone dose of 7-8 mg/L O_3 (approximately as in sampling 2 and 3) low bromate concentrations of around 2 µg/L are expected and are not considered relevant.

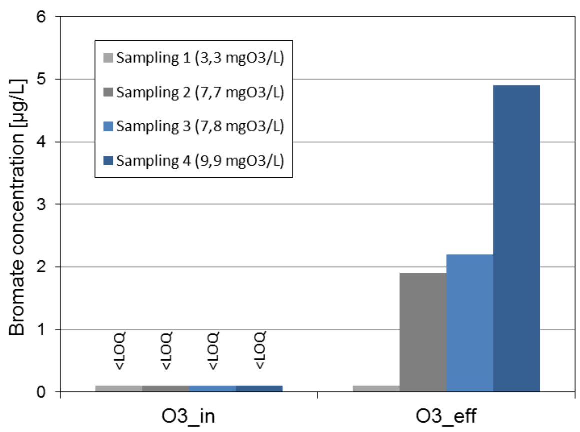


Figure 48 Bromate formation at different O_3 doses.

The most important representative of nitrosames is N-Nitrosodimethylamine (NDMA). Due to its toxicological relevance, WHO Drinking Water Guideline suggests a concentration of 100 ng/L (WHO, 2011). NDMA formation during ozonation was relatively stable between 22 and 33 ng/L (Figure 49, left). During post-treatment NDMA was removed efficiently below LOQ (<5 ng/L) (Figure 49, right). Effluent of CW2 was not analysed.

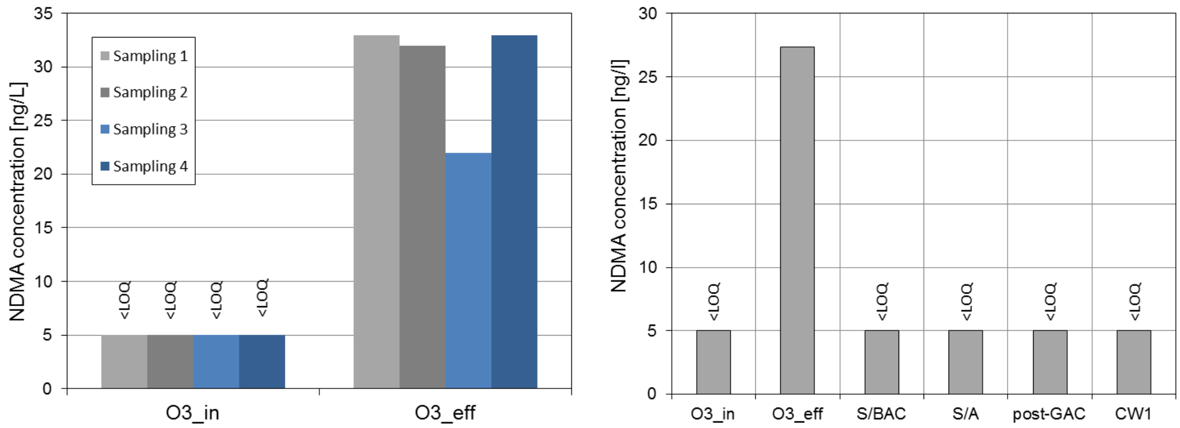


Figure 49 NDMA formation at 4 different days during 1 week (left) and average NDMA concentrations (n=3) at different sampling points (right).

4.4.2.5 Disinfection

Pathogen indicator organisms *E. coli* and Enterococci were found in the secondary effluent at median concentrations of 2.0*10^4 MPN/100 mL and 5.7*10^3 MPN/100 mL, respectively. As depicted in Figure 50 both parameters were reduced to low levels during ozonation, mostly below 10^2 MPN/100 mL. These results demonstrate that ozone reaches *E. coli* and Enterococci removals ≥ 2 log-units at applied target ozone doses of 0.7 mg O_3/ 1 mg DOC.

All post-treatment steps further improved microbial effluent quality. 95th percentiles also decreased after post-treatments. The chosen filter media of CWs showed to have a slight impact on pathogen removal. CW1 with sand (0.2-2 mm) removed both *E. coli* and Enterococci more robust (lower 95th percentiles) than CW2 with the coarser mix of gravel (4-8 mm) and biochar (8-20 mm).

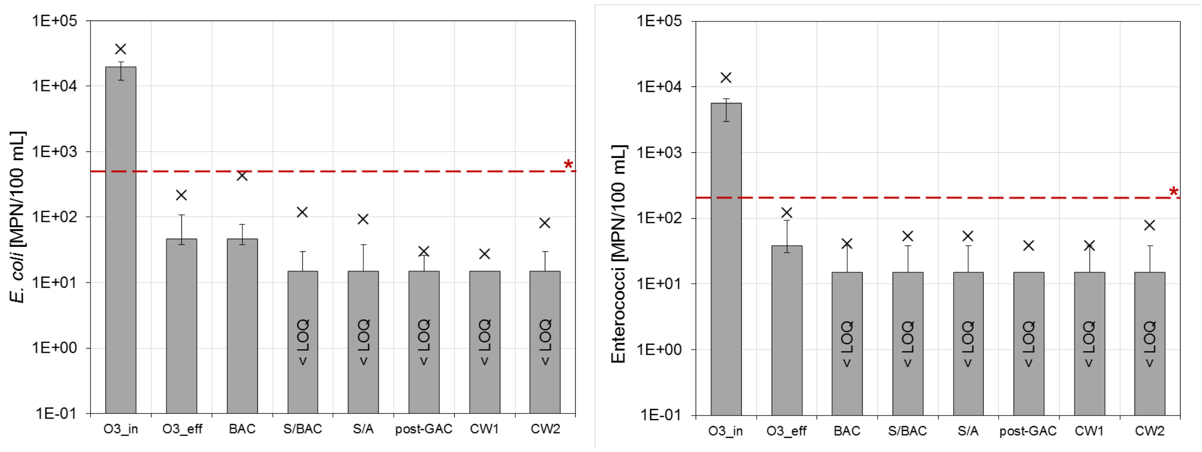


Figure 50 **Concentrations of E. coli (left, n=10-13) and Enterococci (right, n=10-13) at different sampling points.**
Columns: median, error bars: 25th / 75th percentile, crosses: 95th percentile. (*criteria for excellent quality according to EU BWD).

In addition to EU BWD parameters microbiological analysis was also carried out for *Clostridium perfringens* and somatic coliphages which are utilised as indicator parameters for spore-forming bacteria and viruses, respectively. Both are known to be more resistant to disinfection processes. *C. perfringens* and somatic coliphages were present in the WWTP effluent at similar median concentrations of 6.3*10³/100 mL and 5.3*10³/100 mL, respectively. As expected, removal by ozonation was less than 1 log-unit for both parameters (Figure 51).

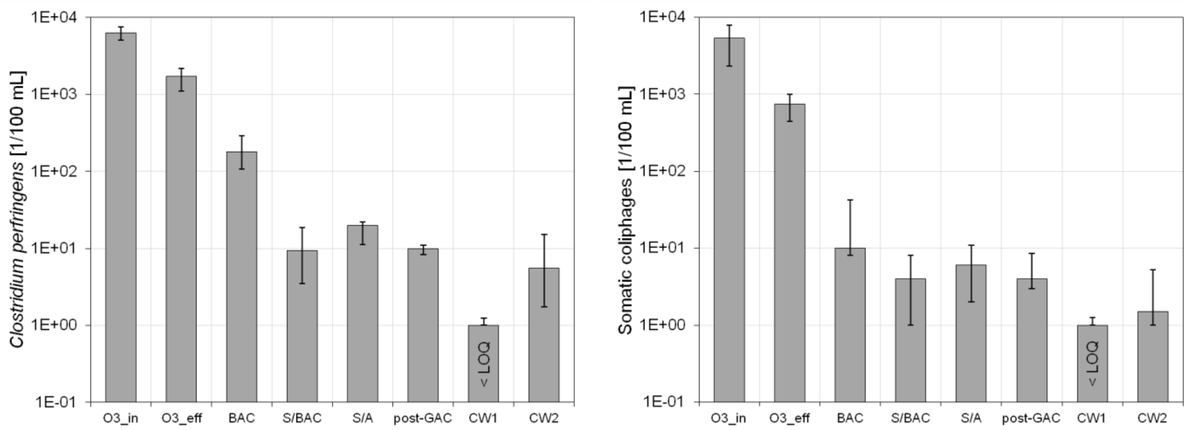


Figure 51 **Concentrations of *Clostridium perfringens* (left, n=9-11) and somatic coliphages (right, n=11-13) at different sampling points.**
Columns: median, error bars: 25th / 75th percentile.

However *C. perfringens* and coliphages after ozonation were efficiently reduced in post-treatments down to median concentrations mostly in the range of 1-10/100 mL (removal rate 2-3 log-units). Both CW performed slightly better than deep-bed filters.

Sand filter CW1 showed the best removal performance among all post-treatments with median effluent concentrations for *C. perfringens* and coliphages below LOQ (1/100 mL).

4.4.2.6 Stand alone CW operation (without ozone dosing)

A combined treatment is supposed to be safer than a single step treatment (multi barrier concept). For this reason additional measuring campaigns were done when ozonation was off.

The comparison of average DOC concentrations in the influents and effluents of CW with and without ozone dosing is shown in Figure 52. DOC removals of 4 % in CW1 and 9 % in CW2 without ozone confirmed decreased efficiency compared to the combined treatment (both CW: 21-22 %).

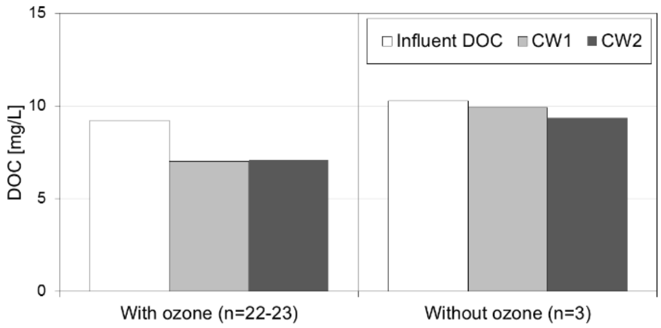


Figure 52 Mean DOC concentrations in CW influent and effluents with (left) and without (right) ozone dosing.

Disinfection was also studied without previous ozone dosing. As displayed in Figure 53 EU BWD indicators *E. coli* and Enterococci were well removed by CW. However, median concentrations of both parameters were slightly higher than after the combined treatment (both < LOQ).

CW1 reduced both *C. perfringens* and somatic coliphages below LOQ like the combined treatment also did. Again CW2 did not perform as efficiently as CW1 for *C. perfringens* and somatic coliphages with median effluent concentrations of 10-10²/100 mL. With previous ozone dosing median effluent concentrations in CW2 stayed below 10/100 mL.

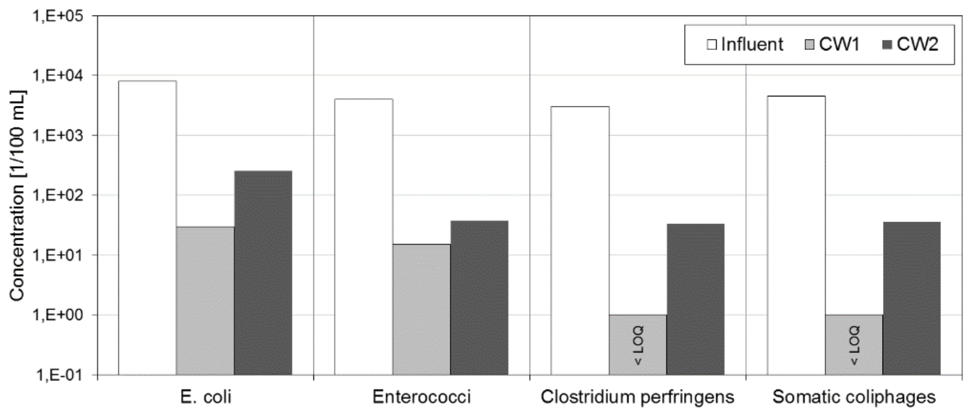


Figure 53 Median concentrations (n=3) of E. coli, Enterococci, Clostridium perfringens and somatic coliphages in CW influent and effluents without ozone dosing.

4.4.2.7 Vegetation

Both CW were planted with *Phragmites australis* and *Carex acutiformis* in equal parts. After 18 month an uneven distribution of the plants was observed (see Figure 54). While in CW1 (Sand) the reed expanded over the whole filter area and replaced the sedge, in CW2 (Lavagravel/Biochar) there was nearly no expansion of the reed. The sharp-edged lava gravel seems to restrict the growth of roots and shoots of the reed in the underground. Also the height and density of the vegetation differed widely. At the end of September 2018, in CW2 the reed was 3 times denser than in CW1, the sedge 5 times. *Phragmites australis* was at an equal height in both CWs, while the *Carex acutiformis* was ~ 12 % higher in CW2.

The vitality and growth of the plants in CW2 appeared to be greater than in CW1. The dark green colours of the leaves in CW2 indicated a better supply of nutrients or minerals (see picture). The reed in CW1 showed typical yellowish necrosis, which often can be seen in sand type post-treatment wetlands. It is not clear, weather lava or biochar is contributing to a better supply with trace elements. Biochar generally has a positive effect on plant growth as observed by Dobner et al., (2016) too.

Figure 54 CW 2 (left container) with reed (left part) and sedge (right part), CW 1 (right container), Sept. 2018.

4.4.3 Conclusion

Synergy effects

Results showed that ozonation and CW treatment is a suitable combination to remove organic and microbial contamination. Comparison of average DOC concentrations in the influents and effluents of CW with and without ozone dosing: DOC removals of 4 % in CW1 and 9 % in CW2 without ozone confirmed decreased efficiency compared to the combined treatment (both CW: 21-22 %). This outcome highlights the synergy of the process combination since neither ozone nor CW treatment alone removed considerable amounts of DOC, but combined they do.

The process combination improves disinfection capacity and results in an excellent water quality according to EU BWD. Disinfection was improved by CW post-treatment. After ~2 log-units reduction of *E. coli* and Enterococci during ozonation they further decreased below LOQ in CW treatment. *C. perfringens* and somatic coliphages were insufficiently inactivated by ozone. CW post-treatment effectively retained both organisms and hence, compensated the short-comings of the ozone treatment. Thus the process combination of ozone and CW works for a wider range of microorganisms and therefore provides higher disinfection safety.

NDMA formation during ozonation was relatively stable between 22 and 33 ng/L. During post-treatment in CW1 NDMA was removed efficiently below LOQ. CW fulfil a main function of the post-treatment: the removal of biodegradable organic transformation products formed by ozonation.

OMP were mainly reduced by the ozonation step. However, for selected OMP with insufficient reaction rates during ozonation due to substance specific removal by ozone, removal could be complemented by CW. Potential bio-transformation of VAL and TCPP in both CW and limited adsorption of OXI, MET and BTA in CW2 were observed.

Hydraulic loading rate of CW

The maximum tested HLR within the scope of the AquaNES project was 1.0 m/d (Table 22), with a minimum HRT of ~10 h. No negative impact regarding reduction rates of investigated parameters or filtration capacity was observed yet. The maximum loading rates of 1 m/d may be used as limitation for evaluating treatment performance. For sand filtration this high hydraulic load is critical.

Resulting filter surface area for a large-scale plant in Schönerlinde with a dry weather flow of 105,000 m^3/d would be 10.5 ha. In our former studies in Hobrechtsfelde near Schönerlinde (1.500 m^2 VF wetland without ozonation) with less coarse natural sand at a d_{10} of less than 0.2 mm a max. hydraulic loading rate of only 0.2 m/d had been approved for the effluent of this WWTP. In general long-term investigations are necessary for finding critical loading rates of natural filtration systems. The operation cycles of these systems have been too short for confirmation. Loading rate limits have to be evaluated with respect to the grain size of the used filter media.

Table 22 Daily specific COD and TSS loading rates of the CWs in Schönerlinde at different daily HLR.

Daily HLR	COD inflow concentration (average)	Daily spec. COD loading rate (average)		TSS inflow concentration (average)	Daily spec. TSS loading rate (average)	
[L/m²/d]	[mg/L]	[g/m²/d]		[mg/L]	[g/m²/d]	
200	28	6	n=8	6	1	n=8
400	29	11	n=10	5	2	n=8
1000	27	27	n=8	5	5	n=5

Filter media

Both filter media were effective for this treatment purpose. CW1 showed the best removal performance among all post-treatments installed at Site No. 12 with median effluent concentrations for *C. perfringens* and coliphages below LOQ (1/100 mL) and a robust removal of *E. coli* and Enterococci with effluent concentration at 95^{th} percentiles of ~ 30/100 mL and ~ 20/100 mL, respectively.

The implementation of biochar in CW2 had a limited impact on increasing the removal efficiency: Only certain OMPs (BTA, MET, OXI) were additionally removed. An advancing breakthrough was observed which indicates the low adsorption capacity of biochar compared to activated carbon.

On a long run biochar is a good substrate for biofilm development and it supports plant growth.

Table 23 Key Takeaways from Ozonation + CW combination show case.

	Ozonation	CW
Function (+ removal/ - increase)	+ OMPs + Pathogens (E. coli, Enteroccoci) + COD - Transformation products	+ DOC, COD + Pathogens (*E. coli*, Enteroccoci, *C. perfringens*, coliphages) + Transformation products + some OMPs
Risks of malfunction	– UVA probes (explanation of function see D3.2)	– Clogging of sand filter if overloaded (organics/ TSS) – Gravel size media show less tendency for clogging
Operation and maintenance needs	– Daily online check of operational parameters – Daily inspection of function control of important parts – UVA-probes (weekly check) – DOC and NO2 measurements are essential for specific ozone dose calculation	– Check water distribution systems regularly – Plants care – Filter material analysis – Flow and quality monitoring

4.5 Post-Treatment wetland with P-reactive steel slag

The effluent from the WWTP Packington in the UK is discharged in the Gilwiskaw Brook, a tributary of the River Mease. The River Mease catchment is designated as Special Area of Conservation under the European Commission Habitats Directive (92/43/EEC) and a Site of Special Scientific Interest. To meet the objective set by the Water Framework Directive (WFD), the consent for P discharge will be reduced to ≤ 0.3 mg P/L. For this reason a trial on reactive media CWs as a sustainable alternative to coagulant dosing for P removal at small WWTPs was conducted.

4.5.1 Treatment concept

The Packington WWTP treats the domestic sewage from residents as well as effluents from industries in the area with a dry weather flow of 57 L/s (~17,000 PE). The current treatment scheme of the full scale site comprises screening and grit removal in the inlet works, biological treatment in 2 oxidation ditches in parallel followed by clarification and tertiary filtration in a deep bed sand filter.

The pilot reactive media CW was fed with secondary treated effluent (Figure 55). The effluent from the full scale oxidation ditches was taken after clarification but before the sand filter and fed to a flash mixing tank, in which the P concentration (dosing of acid phosphoric) can be adjusted for the purpose of the trial, before being fed to the demonstration CW. The demonstration CW consists of a conventional subsurface horizontal flow wetland with a surface area of 100 m² and a depth of 0.6 m. It was operated for a total of 3.3 years. The reactive media used is blast oxygen furnace (BOF) steel slag with particle size ranging between 8 and 14 mm. Steel slag, a waste product from the steel industry, is mainly made of calcium oxide and other metal oxides such as iron, magnesium and aluminium (Table 24), all known to react well with P to form precipitates or act as adsorbents. The bed was planted with *Phragmites australis* at 4 plants/m². The CW was fed at a flow rate of 0.35 L/s, corresponding to an experimental HRT of 18 h (according to tracer tests carried out annually) or to a daily hydraulic load of 300 mm/d.

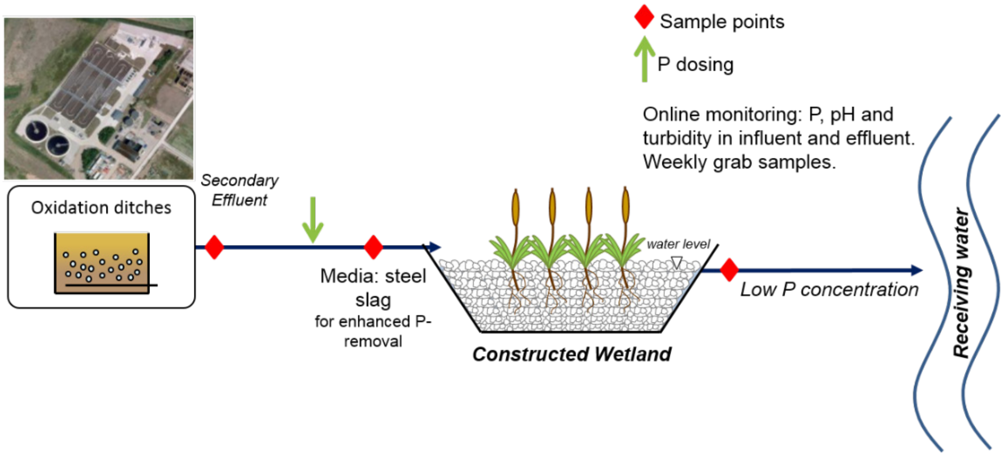


Figure 55 Process scheme of the pilot reactive media HSSF CW at WWTP Packington, UK.

Table 24 Composition of the BOF steel slag media.

Elements	CaO	Fe2O3	SiO2	MgO	Al2O3	MnO	P2O5	V2O5
Fractions [%]	42	23	13	7.5	3.1	2.4	1.2	0.71

4.5.2 Operation results

The data reported below mainly focuses on the initial two years of operation during which significant variations were observed. A more comprehensive report about the demonstration site and treatment performance throughout the demonstration trial can be found in the deliverable D3.2 on combining constructed wetlands and engineered treatment for surface water protection.

4.5.2.1 Phosphorus removal

The reactive media CW was operated mostly in continuous steady-state conditions with an empty bed contact time (EBCT) of 48 hours and with a phosphorus influent concentration of about 7.6 mg/L. Over the initial 2 years of operation for phases could be identified in terms of the variation in phosphorus removal (Figure 56). Indeed, the CW's effluent P concentration remained constantly below 0.1 mg/L over the initial 210 days of operation (Phase 1). The effluent P concentration then gradually increased to reach a maximum of 2.8 mg/L after 430 days of operation (Phase 2), at which point, it started decreasing to reach a minimum of 1.2 mg/L after 580 days (Phase 3). Finally, the effluent P concentration was found to increase rapidly again to reach a maximum of 7.3 mg/L at 720 days of operation followed by another decrease down to 4.5 mg/L at 760 days (Phase 4). The results obtained in Phase 1 confirmed the potential of steel slag as a reactive media to remove P from wastewater to low levels in a single step with over 98 % removal. However, if we consider that small WWTPs are unlikely to be assigned very strict consents due to the probable high dilution of the effluent discharged into a receiving water course and that achieving P targets of 1-2 mg/L is more realistic for small sites, the trials highlight that the system was essentially overachieving in this first phase and eventually reached a breakthrough of above 1 mg/L after 320 days of operation. The increase in effluent P concentration initially in Phase 2 and more significantly in Phase 4 suggested that the media may have reached removal capacity. However, these phases of increase were always followed by a phase of decrease (Phase 3 and end of Phase 4) demonstrating that the capacity of the bed for P removal was not reached but changes occurred in the process. In fact, with a total of 1.39 g of P removed per kg of media over 2 years of operation, this system demonstrated greater P removal capacity overall than most other steel slag media beds reported in the literature with real wastewater (Park et al., 2017; Hussain et al., 2015; Barca et al., 2013). This can partly be explained by the fact that previous studies were generally conducted at smaller scale and/or over shorter periods of time, as this trial is, to our knowledge, the first long term (> 3 years) demonstration scale evaluation of the steel slag media. Having said that, the implementation of reactive media CWs would only be sustainable if the system could sustain the initial treatment performance for several years. It then becomes essential to better understand the variations observed in this system and the possible changes in mechanism in order to improve the design and operation of such systems and ultimately deliver long term sustainable treatment performance.

The steel slag media is mainly composed of calcium oxide and ferric oxides but also contains other compounds such as aluminium, magnesium and vanadium oxides among others. Previous studies have reported that the main P removal mechanisms with steel slag as a reactive media include precipitation as calcium phosphate through reaction with calcium dissolving from the solid media, adsorption of the precipitates or P directly on the media itself, the precipitates or through ligands exchange on the aluminium and iron compounds. Further analysis of the effluent P demonstrated that it was mainly soluble (Figure 57). This suggests that, as the concentration increased in the effluent of the CW in the later stages of the trial, the phosphorus did not react with either the calcium in solution, the already formed precipitates or even the media, and remained in solution as opposed to reacting and the precipitates being wash-out in the effluent. This then highlights a change in removal mechanisms during the trial.

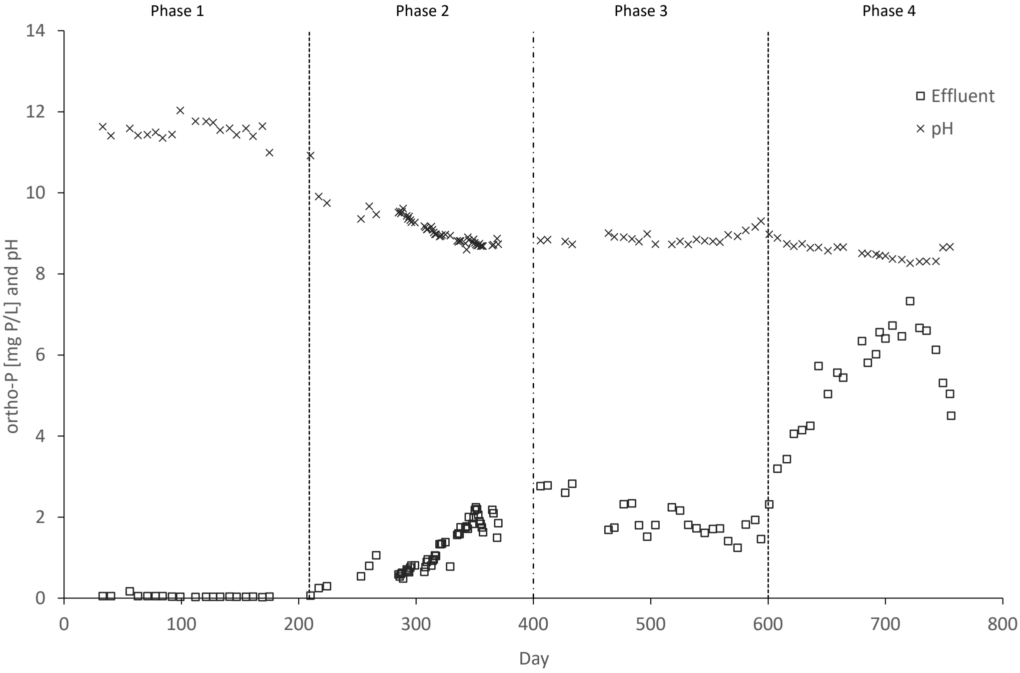


Figure 56 **Evolution of the effluent phosphorus concentration and pH over time.**

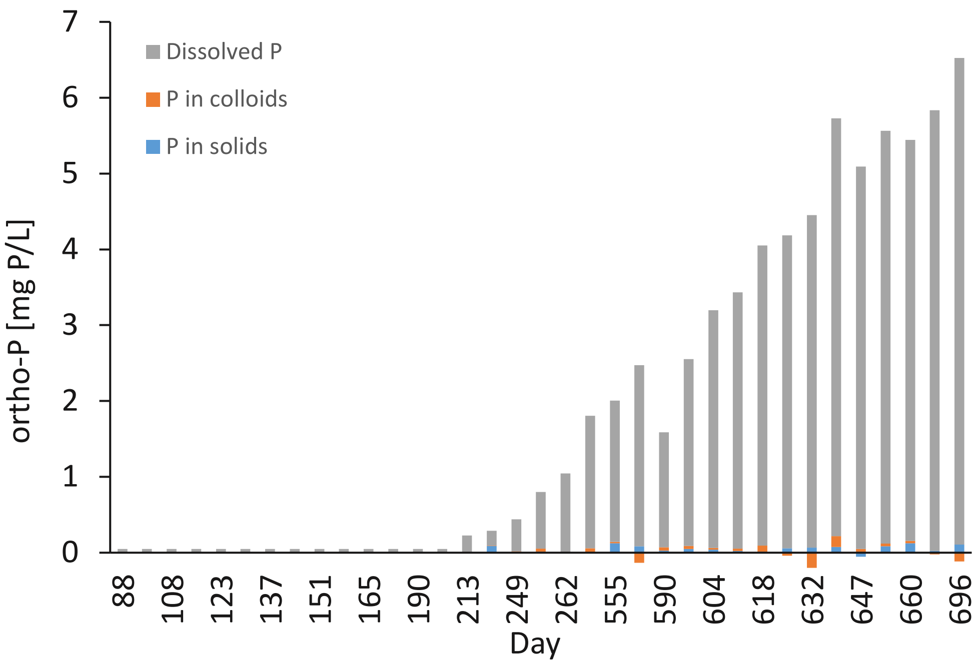


Figure 57 **Fractionation of the phosphorus present in the effluent of the CW for selected days during the trial.**

4.5.2.2 pH variations

The pH in the effluent was found to be initially very high with values between 11 and 12 in Phase 1, gradually decreased to about 8.7 over Phase 2 and then remained relatively stable thereafter with values between 9.3 and 8.2 (Figure 56). The increased pH measured in the effluent throughout the trial can be explained by dissolution of the media as reported by previous scientific studies (Barca et al., 2012). When the media is in contact with water, it will partially dissolve and release calcium and hydroxide ions (and other compounds) in the effluent. As mentioned before, the calcium is likely to react with compounds such as phosphate and carbonate to form precipitates. In parallel, the hydroxide ions will contribute to the pH increase. It is important to note that the pH in the effluent remained higher than the pH in the influent (7.4) demonstrating that some hydroxide ions release occurred throughout the trial. However, the fact that the pH decreased over time hence the release of hydroxide ions slowed suggests that media became gradually coated by precipitates limiting direct contact of the media with the water and consequently limiting the leaching of ions in the water.

The analysis of the results obtained shows a correlation between the pH and phosphorus concentration in the effluent (Figure 58). Indeed, the P concentration was found to increase as the pH decreased. To illustrate, the effluent P concentration remained below 1 mg/L with pH above 9 and then rapidly increased up to 7 mg/L for pHs between 9 and 8. It is known that pH above 9-10 are favourable for the precipitation process of calcium phosphate to occur which supports the findings with better treatment performance obtained at the higher pH and highlights possible changes in treatment mechanisms over time as the pH changed. Overall, these results show a direct impact of the pH on the removal mechanism when high but the effluent P concentration variations over Phase 3 and 4 when the pH was mostly stable demonstrate that lower pH (<9) has no impact on the removal and other parameters impacted the P removal process.

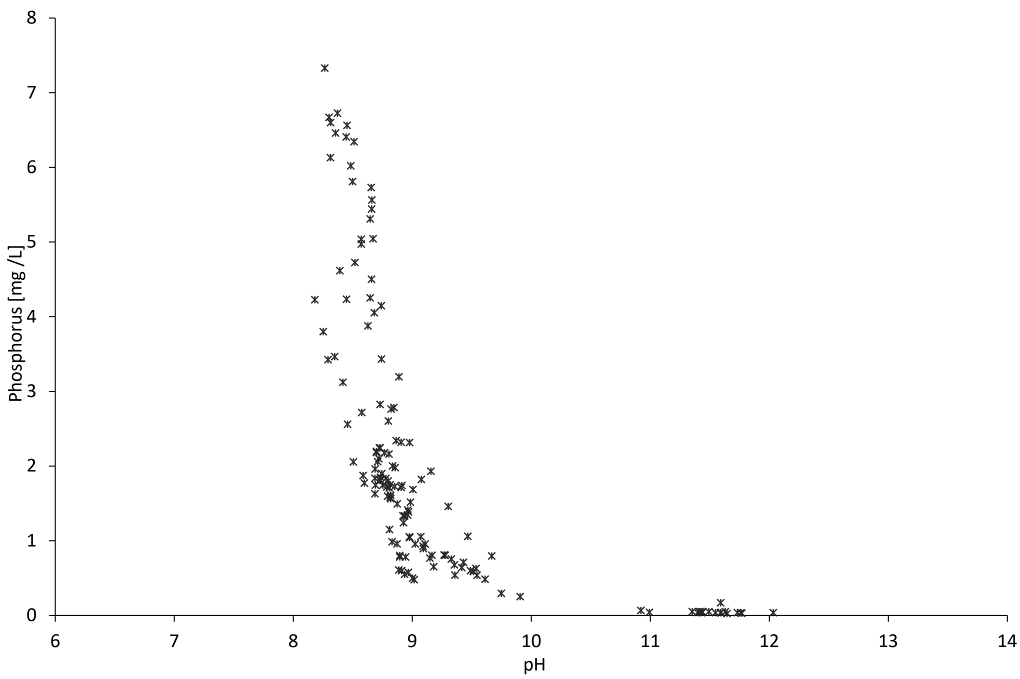


Figure 58 Impact of pH on the effluent phosphorus concentration.

4.5.2.3 Calcium and vanadium release

Comparison of the calcium concentrations in the influent and effluent of the CW shows an initial net release of calcium from the media into the effluent (Figure 59) which, combined with the high pH, will lead to rapid precipitation with the phosphate and most likely carbonates present in the water to form calcium phosphate and calcium carbonate. The net release of calcium was found to decrease over Phase 1. This confirms the assumption that the precipitates formed coated the surface of the media and reduced the release of all ions into the water. Towards the end of Phase 1 and throughout Phase 2, calcium was found to be removed from the water. This does not show that calcium was no longer released from the media but demonstrates that any calcium released from the media as well as some of the media from the influent reacted and was removed within the bed, again confirming rapid precipitation in this initial phases. A net release of about 10 mg/L of calcium was then observed in the subsequent phase (Figure 59) again confirming a change in mechanism in the process.

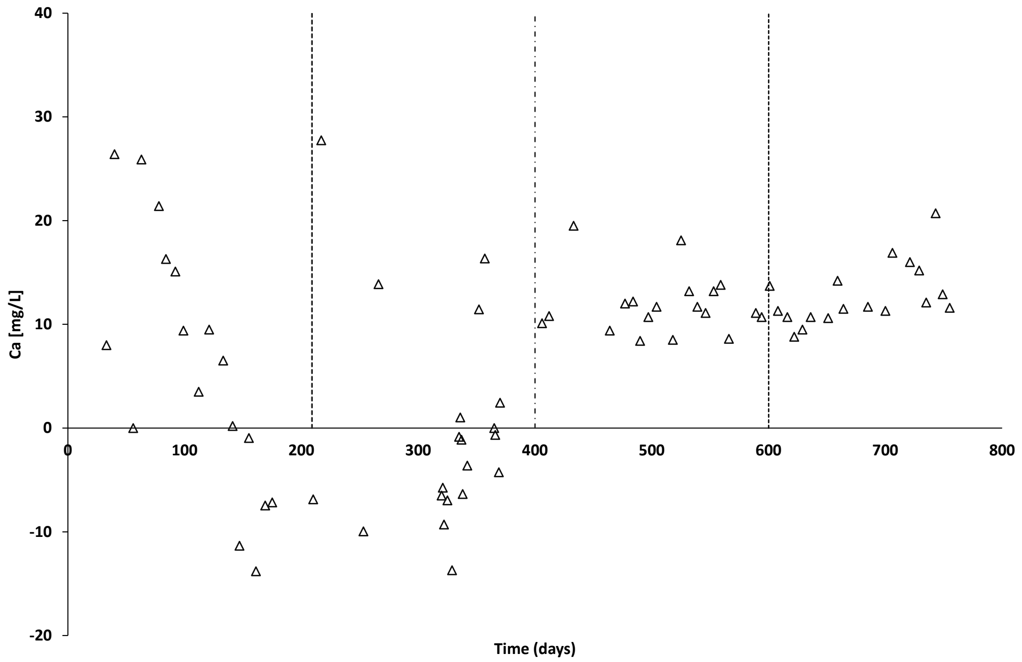


Figure 59 Calcium ions release (Effluent concentration – Influent concentration) in the effluent of the CW.

As mentioned above, the steel slag media used in the CW is mainly composed of calcium oxide but it also contains other compounds such as iron, aluminium and vanadium. Vanadium is of particular interest because it is toxic (possibly carcinogenic) in most its forms and a release of the compounds into the environment would not be acceptable as part of a full scale system. As observed for the calcium and hydroxide ions, a release was observed over Phases 1 and 2 with an initial concentration of about 734 µg/L which then gradually decreased to about 80 µg/L after 300 days of operation and then remained stable with values mostly between 50 and 90 µg/L (Figure 60). These results confirm all previous assumptions that as precipitates are formed in the bed, they gradually coat the media surface and reduce the release of any compounds from the media. Importantly, the released of vanadium observed at the start of the trial would not be acceptable for a full scale application and consequently this would have to be solved before the technology can be fully implemented.

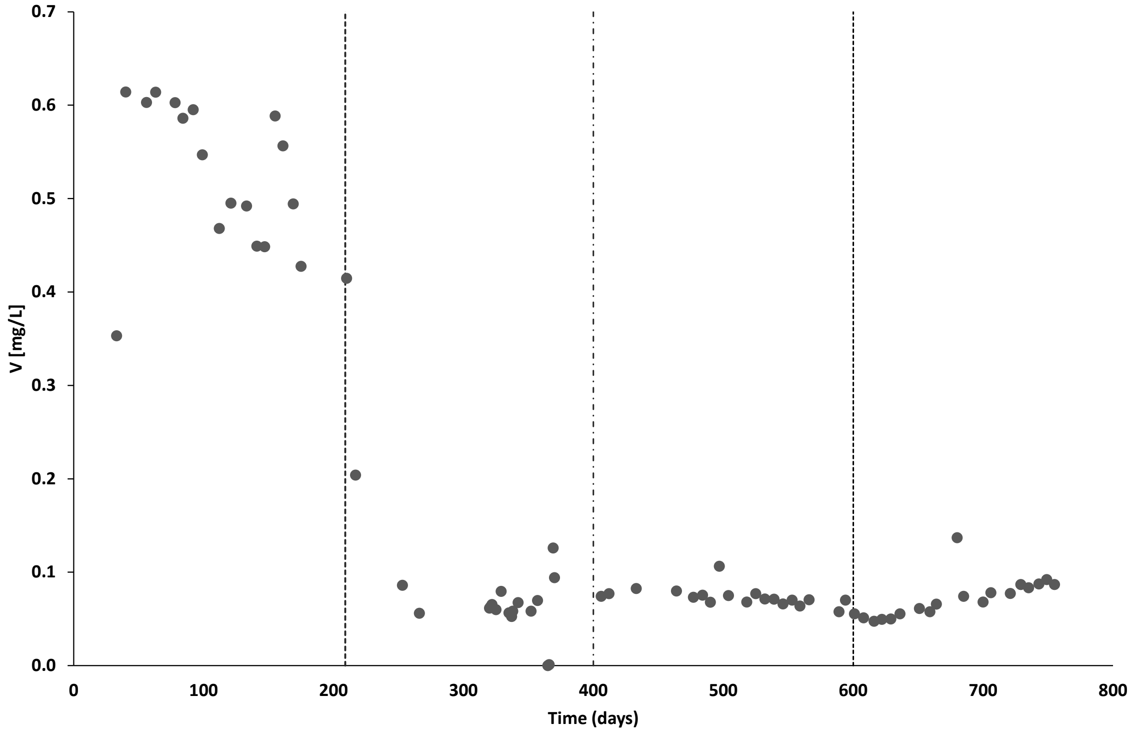
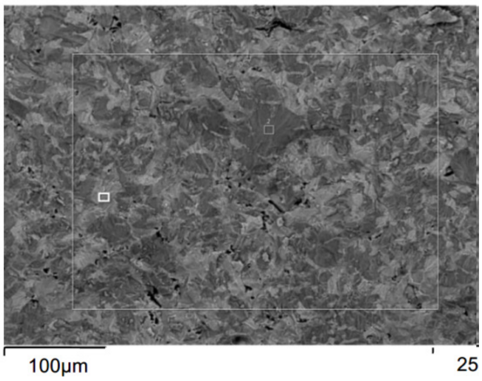


Figure 60 Vanadium release from the steel slag media over time.

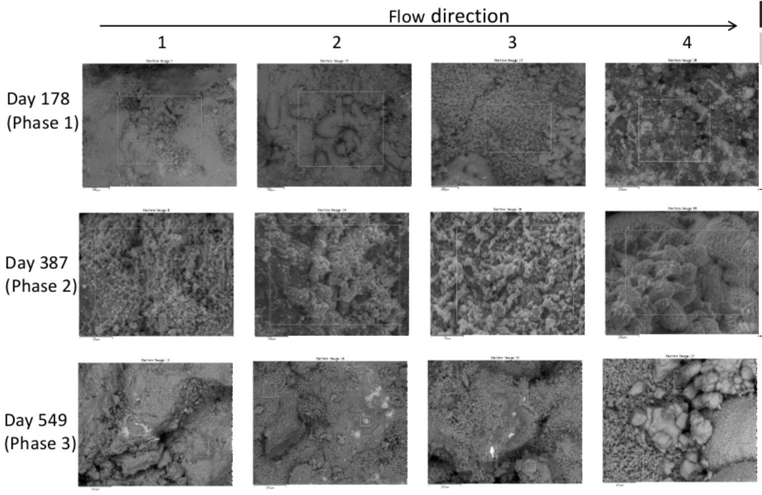
As part of the monitoring programme, a range of other metals (i.e. Fe, Al, Si, Ti, Cr, Ni, Cu, Zn, Se, Ag, Cd, Pd, Hg, Mn) were also measured and all of them were found either in very low concentrations or not detected in influent and effluent of the CW demonstrating these did not pose a risk to the environment. Interestingly, some elements including Fe, Ni and Zn were actually removed by the CW.

4.5.2.4 Media analysis

Specific characterisation of the media has helped clarifying some of the hypotheses mentioned above. Indeed, microscope imaging of fresh steel slag media and used media from the bed at different times through the trial (Figure 61), first has shown a clear difference between the surface of the fresh and used media confirming formation of a coating layer then corroborating previous suggestions. However, the elemental analysis of the materials present on the surface of the used media does not show an increase in P over time suggesting that the P removed from the water did not actually accumulate on the media itself but is more likely captured within the bed as precipitate and/or adsorbed on precipitates. Further analysis of the media after 2 years of operation through a sequential extraction of the P present shows that the phosphorus already present on the fresh media (washed and unwashed) is mainly calcium bound (Figure 62). For all used media samples, the biggest fraction of phosphorus remains calcium bound (except for E2) but more significant proportions of phosphorus were then iron bound and in the calcium bound stable pool. These results suggest that the phosphate attached to the media preferentially binds with the iron from the media to form iron phosphate which is known to be very insoluble at pH above 8, as observed in this study, or with calcium to make more stable forms of calcium phosphate. Accumulation of large amounts of precipitates over time in the void space between the media grain could lead to clogging of the bed and short circuiting of the flow which ultimately could partially explain the poorer performance observed in the later stages of the trial. In order to evaluate any possible changes in the hydraulics in the CW, tracer and hydraulic conductivity test were carried out at regular intervals during the trial (the detailed results can be found in deliverable D3.2). The tests generally showed that there was no change in hydraulic retention time over duration of the trial hence confirming that no clogging occurred in bed so this cannot explain the variation in performance.



(a)



	w% (n= 4)			
	C	O	Ca	P
Day 178 (Phase 1)	10.1 ± 3.8	48.7 ± 1.1	32.1 ± 3.5	2.2 ± 1.9
Day 387 (Phase 2)	8.2 ± 0.4	49.6 ± 1.9	31.4 ± 1.2	3.0 ± 1.7
Day 549 (Phase 3)	9.4 ± 1.1	45.8 ± 1.1	33.8 ± 2.8	2.2 ± 0.7

(b)

Figure 61 Scanning electron imaging of (a) the fresh media and (b) the used media including energy dispersive X-ray spectroscopy analysis of the surface material.

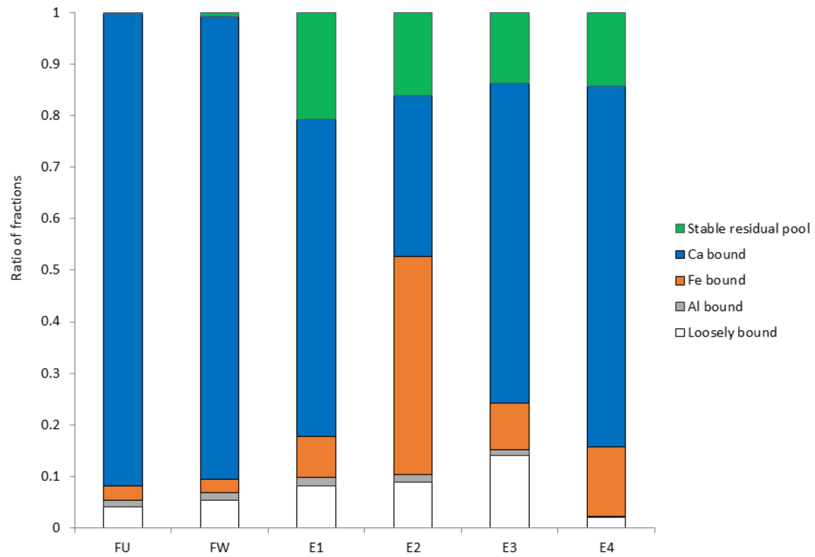


Figure 62 Phosphorus sequential extraction from fresh media unwashed (FU) and washed (FW – rinsed with clean water to remove any loose deposits), and used media from four locations in the direction of the flow in the middle of the bed, with E1 closer to the inlet and E4 closer to the outlet.

4.5.2.5 Seasonal variations

The variations in P removal in the systems were found to be directly linked to seasonal changes and in particular temperature (Figure 63). The peaks and troughs in effluent P concentration were found to match with the highest and lowest seasonal temperatures, respectively. This clearly demonstrates a direct influence of temperature on the P removal mechanisms. This phenomenon was previously observed by other researchers working on a similar concept with other types of media and the variations were attributed to enhanced calcium precipitation at higher temperatures (Herrmann et al., 2014; Barca et al., 2013; Shilton et al., 2006).

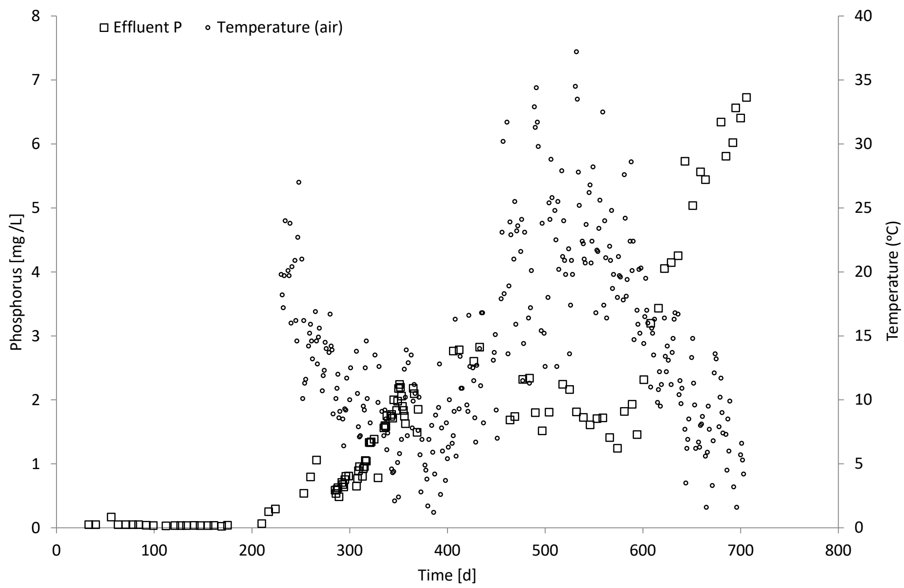


Figure 63 Evolution of the effluent P concentration and air temperature over time for the initial 700 days of operation.

4.5.3 Conclusion

This long term demonstration scale trial of a steel slag reactive media CW has first confirmed the potential of the technology for the removal of P to low levels as a single step for small WWTPs. However, a number of limitations highlighted by the trial mean that the technology is not yet ready for full scale application. Indeed, the high effluent pH and more importantly the release of vanadium and the relatively short life span (circa 1 year) of the media to maintain low P effluent concentrations (< 2 mg/L) are critical bottlenecks towards wide spread implementation. The fact is that there is still no sustainable solution, other than coagulant dosing, available for P removal on small WWTPs and the interest in a reactive media CW by water utilities remains very high.

The present work has highlighted the complexity of the mechanisms involved and the impact some parameters (pH, temperature ...) have on the removal process and provides a basis for further developments. For example, future studies could evaluate the following:

- The potential for pre-conditioning of the media before operation.
- The development of a controlled coating layer to limit pH increase and vanadium leaching.
- The implementation of a regeneration process of the media to extend the life of the bed and to recover the P captured within the bed.
- Solutions for sustainable pH control to maintain highest P removal capacity while discharging an effluent meeting all regulatory requirements.

Table 25 Key Takeaways from pilot reactive media CW for P-removal with BOF steel slag.

		Reactive media CW with BOF steel slag
Function (+ removal/ - increase)	+	COD
	+	TSS
	+	TP
	+	Fe, Ni, Zn
	+	Ni
	+	Zn
	-	Vanadium
	-	pH
Risks of malfunction		– Early saturation of filter media and break-through of P – pH level in phase 1 exceeding 9 – Fast release of vanadium in the order of magnitude = 50 g/m³ of slag filter volume – Slag as filter media is not recommended.

Literature

Guidelines and technical reports

Dotro, G., Langergraber, G., Molle, P., Nivala, J., Puigagut, J., Stein, O., Von Sperling, M. (2017), Treatment wetlands. Biological Wastewater Treatment Series, Vol. 7, IWA publishing. https://www.iwapublishing.com/books/book-series/biological-wastewater-treatment-series.

DWA-M 205 (2003), Desinfektion von biologisch gereinigtem Abwasser (Fact sheet DWA-M 205: Disinfection of biologically treated wastewater), (in German). German Association for Water, Wastewater and Waste (DWA), Hennef, www.dwa.de.

DWA-A 178 (2017, draft), Empfehlungen für Planung, Bau und Betrieb von Retentionsbodenfilteranlagen zur weitergehenden Regenwasserbehandlung im Misch- und Trennsystem (Standard DWA-A 178: Recommendations for planning, construction and operation of retention soil filters for advanced stormwater treatment in combined and separate sewer systems) (in German; draft). German Association for Water Management, Wastewater and Waste (DWA), Hennef, www.dwa.de.

DWA-A 262 (2017), Grundsätze für Bemessung, Bau und Betrieb von Kläranlagen mit bepflanzten und unbepflanzten Filtern zur Reinigung häuslichen und kommunalen Abwassers (Principles for Dimensioning, Construction and Operation of Wastewater Treatment Plants with Planted and Unplanted Filters for Treatment of Domestic and Municipal Wastewater) (November 2017), (German version). German Association for Water, Wastewater and Waste (DWA), Hennef, www.dwa.de.

DWA-A 262E (2018), Principles for Dimensioning, Construction and Operation of Wastewater Treatment Plants with Planted and Unplanted Filters for Treatment of Domestic and Municipal Wastewater (November 2017), (English version). German Association for Water, Wastewater and Waste (DWA), Hennef, www.dwa.de.

Geller, G., Höner, G. (2003), Anwenderhandbuch Pflanzenkläranlagen (Users manual for constructed wetlands). ISBN-13: 978-3540401353, Springer, Berlin, Germany (in German).

Hoffmann, H., Platzer, C., von Münch, E., Winker, M. (2010), Overview of subsurface flow constructed wetlands for greywater and domestic wastewater treatment in developing countries. Sustainable Sanitation and Ecosan Program of Deutsche Gesellschaft Für Technische Zusammenarbeit (GTZ) GmbH, Germany.

Kadlec, R. H., Wallace, S. (2009), Treatment wetlands, 2nd ed. CRC/Taylor & Francis Group, Boca Raton, ISBN 978-1-56670-526-4, USA.

Lombard Latune, R., Molle, P. (2017), Les filtres plantés de végétaux pour le traitement des eaux usées domestiques en milieu tropical: Guide de dimensionnement de la filière tropicalisée. Agence Française pour la biodiversité, collection Guides et protocoles.

MKULNV (Ministry of Climate Protection, Environment, Agriculture, Nature and Consumer Protection of the State of North Rhine-Westphalia, Germany) (2015), Retentionsbodenfilter. Handbuch für Planung, Bau und Betrieb (Retention Soil Filter. Manual for Planning, Building and Operation), Düsseldorf, Germany. https://www.umwelt.nrw.de/fileadmin/redaktion/Broschueren/retentionbodenfilter_handbuch.pdf

1.

Payne, E., Hatt, B. E., Deletic, A., Dobbie, M. F., McCarthy, D. T., Chandrasena, G. I. (2015), Adoption guidelines for stormwater biofiltration systems. Cooperative Research Centre for Water Sensitive Cities, Melbourne, Australia.

UN-HABITAT. (2008), Constructed Wetlands Manual. UN-HABITAT Water for Asian Cities Programme Nepal, Kathmandu.

World Health Organization (WHO) (2006), WHO guidelines for the safe use of wastewater, excreta and greywater. Volume 4: Excreta and greywater use in agriculture. World Health Organisation, Geneva, Switzerland. https://www.susana.org/en/knowledge-hub/resources-and-publications/library/details/1004.

Other references

Barca, C., Gérente, C., Meyer, D., Chazarenc, F., Andrès, Y. (2012), Phosphate removal from synthetic and real wastewater using steel slags produced in Europe. Water Research 46.7, pp. 2376–2384.

Barca, C., Troesch, S., Meyer, D., Drissen, P., Andres, Y., Chazarenc, F. (2013), Steel slag filters to upgrade phosphorus removal in constructed wetlands: Two years of field experiments. Environmental Science and Technology 47.1, pp. 549–556.

Batty, L., Baker, A., Wheeler, B. (2002), Aluminum and Phosphate Uptake by Phragmites australis: the Role of Fe, Mn and Al Root Plaques. Annals of Botany 89, pp. 443-449.

Boram, K., Gautiera, M., Olvera Palma, G., Molle, P., Michel, P., Gourdana R. (2014), Pilot-scale study of vertical flow constructed wetland combined with trickling filter and ferric chloride coagulation. Proceedings of the 14th IWA International Conference on Wetland System for Pollution Control (ICWS2014: p. 548-558), Gdansk, October, 12 - 16, 2014.

Brunsch, A. F., ter Laak, T. L., Christoffels, E., Rijnaarts, H. M., Langenhoff, A. M. (2018), Retention soil filter as post treatment step to remove micropollutants from sewage treatment plant effluent. Science of the Total Environment, 637-638 p. 1098-1107.

Dobner, I., Siol, A., Filser, J., Warrelmann, J. (2016), Entwicklung eines innovativen Pflanzenfilters zur Eliminierung von Arzneimittelrückständen im Ablauf kleiner Kläranlagen und dezentraler Kleinkläranlagen (Phasen 2 u. 3). Deutsche Bundesstiftung Umwelt. Bremen.

Eggen, R. I., Hollender, J., Joss, A., Schärer, M., Stamm, C. (2014), Reducing the discharge of micropollutants in the aquatic environment: the benefits of upgrading wastewater treatment plants. Environmental Science & Technology 2014 48 (14), 7683-7689.

Heeb, F., Singer, H., Pernet-Coudrier, B., Qi, W., Liu, H., Longrée, P., Müller, B., Berg, M. (2012), Organic micropollutants in rivers downstream of the megacity Beijing: sources and mass fluxes in a large-scale wastewater irrigation system. Environmental science & technology, 46(16), 8680-8688.

Herrmann, I., Nordqvist, K., Hedstrom, A., Viklander, M. (2014), Effect of temperature on the performance of laboratory-scale phosphorus-removing filter beds in on-site wastewater treatment. Chemosphere 117.1, pp. 360-366.

Huber.de (2019), https://www.huber.de/fileadmin/_processed_/c/b/csm_skizze_r05_01_24428f14e7.jpg, 20.02.2019.

Hussain, S. I., Blowes, D. W., Ptacek, C. J., Jamieson-Hanes, J. H., Wootton, B., Balch, G., Higgins, J. (2015), Mechanisms of Phosphorus Removal in a Pilot-Scale Constructed Wetland/BOF Slag Wastewater Treatment System. Environmental Engineering Science 32.4, pp. 340–352.

Luo, Y., Guo, W., Ngo, H. H., Nghiem, L. D., Hai, F. I., Zhang, J., Liang, S., Wang, X. C. (2014), A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. Science of the total environment, 473, 619-641.

Meyer, D., Molle, P., Esser, D., Troesch, S., Masi, F., Tondera, K., Pinnekamp, J., Dittmer, U. (2014), Constructed Wetlands for Combined Sewer Overflow Treatment. Sustainable Sanitation Practice, 18/2014. http://www.ecosan.at/ssp/issue-18-outcomes-from-the-ufz-wetland-workshop/SSP-18_Jan2014_21-24.pdf/view.

Molle P., Liénard A., Boutin C., Merlin G., Iwema A. (2005), How to treat raw sewage with constructed wetlands: an overview of the French systems. Water Science and Technology 51(9):11-21.

Musolff, A., Leschik, S., Möder, M., Strauch, G., Reinstorf, F., Schirmer, M. (2009), Temporal and spatial patterns of micropollutants in urban receiving waters. Environmental pollution, 157(11), 3069-3077.

NTUA (National Technical University of Athens) (2019), Decision support system (DSS) Roadmap – Design a RSF^{plus} System, http://dss.aquanes.eu/Default.aspx?t=1746&Roadmap=Design%20a%20RSF%20Plus%20system&DiagramHeight=800&Step=1, 11.03.2019.

Osenbrück, K., Gläser, H. R., Knöller, K., Weise, S. M., Möder, M., Wennrich, R., Schirmer, M., Reinstorf, F., Busch, W., Strauch, G. (2007), Sources and transport of selected organic micropollutants in urban groundwater underlying the city of Halle (Saale), Germany. Water research, 41(15), 3259-3270.

Park, J., Wang, J. J., Kim, S., Cho, J., Kang, S., Delaune, R., Seo, D. (2017), Phosphate removal in constructed wetland with rapid cooled basic oxygen furnace slag. Chemical Engineering Journal 327, pp. 713–724.

Platzer, C. (1999), Design recommendations for subsurface flow constructed wetlands for nitrification and denitrification, Wat. Sci. Tech. Vol. 40 (3) p. 257-263.

Rozkošný, M., Kriška, M., Šálek, J., Bodík, I., Istenič, D. (2014), Natural Technologies of Wastewater Treatment. Global Water Partnership Central and Eastern Europe.

Rustige, H., Tomac, I., Höner, G. (2003), Investigations on Phosphorus Retention in Subsurface Flow Constructed Wetlands, Wat. Sci. Tech., Vol. 48, No 5 pp. 67-74.

Rustige, H., Platzer, C. (2002), Constructed wetlands for use within sensitive water bodies (in German language), report - AKUT Umweltschutz Ingenieurgesellschaft, Teilprojekt im Rahmen des Verbundprojektes „Bewachsene Bodenfilter als Verfahren der Biotechnologie“, AZ 14178-01, gefördert durch die Deutsche Bundesstiftung Umwelt, Osnabrück, 1998.

Schalk, T., Effenberger, J. , Jehmlich, A., Nowak, J., Rustige, H., Krebs P., Kühn V. (2019), Methane oxidation in vertical flow constructed wetlands and its effect on denitrification and COD removal. Ecological Engineering: 128 (2019) 77–88.

Shilton, A. N., Elmetri, I., Drizo, A., Pratt, S., Haverkamp, R. G., Bilby, S. C. (2006), Phosphorus removal by an ’active’ slag filter-a decade of full scale experience. Water Research 40.1, pp. 113–118.

Verlicchi, P., Zambello, E. (2014), How efficient are constructed wetlands in removing pharmaceuticals from untreated and treated urban wastewaters? A review. Science of the Total Environment, 470, 1281-1306.

Water by Design (2014), Bioretention Technical Design Guidelines (Version 1.1). Healthy Waterways Ltd, Brisbane. https://waterbydesign.com.au/download/bioretention-technical-design-guidelines.

Weber, D., Drizo, A., Twohig, E., Bird, S., Ross, D. (2007), Upgrading constructed wetlands phosphorus reduction from a dairy effluent using electric arc furnace steel slag filters. Water Science and Technology, 56(3), 135-143.

World Health Organization (WHO) (2011), Guidelines for Drinking-water Quality, 4th ed. ISBN 978 92 4 154815 1.

Annex

Poster “Constructed wetlands in cNES flowsheets – what for and how o design”

AquaNES
Demonstrating Synergies in Combined Natural and Engineered Processes for Water Treatment Systems

Constructed wetlands in cNES flowsheets – what for and how to design

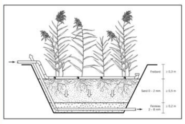
Figure 1: Raw Wastewater Filtration.

Primary treatment

Pre-treatment

- Screens, sand and grease traps are possible (not necessary) pre-treatment options.
- Phosphorus precipitation.

Secondary treatment



- Bioreactor,
- 2nd stage vertical flow wetland,
- UV disinfection.

Challenge/ Purpose

Integrated sludge treatment (TSS, COD NH_4).

Dimensioning of the wetland according to DWA–A 262 (2017).
- Hydraulic and COD areal loading rates,
- Water distribuition.

Advantage

- Very low operational costs, substitution of further sludge treatment.
- State of the art for primary treatment in remote areas.

Challenge/ Purpose

Biological treatment step of domestic and municipal wastewater (BOD, COD, NO_3).

Dimensioning of the wetland according to DWA-A 262 (2017).
- Hydraulic and COD areal loading rates,
- Water distribution.

Advantage

- Very good adaption to varying hydraulic and pollutant loads.
- Ecological habitat.

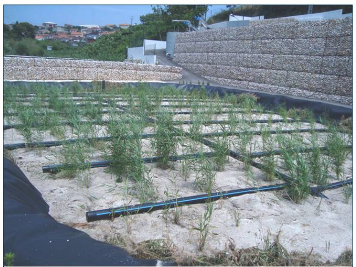
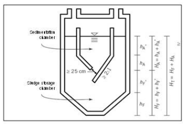


Figure 2: Vertical flow (VF) wetland.

Secondary treatment

Primary treatment



- Imhoff tank,
- Settling pond,
- Anaerobic or aerobic biological primary treatment.

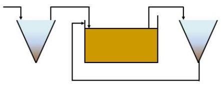
Tertiary treatment

UV disinfection
- Because of the very low turbidity of the filtrate UV can be applied for disinfection.

Horizontal subsurface flow (HSSF) wetland
- For further denitrification and removal of pathogens.

Secondary treatment

Secondary WWTP



Reactive filter (media)

Granulated activated carbon (GAC) layer
- For further removal of micropollutants.

Industrial by-products
- Ferric hydroxides, ferric sulfates, calcium carbonate.

Challenge/ Purpose

Downstream purification stage (TP, TN, micropollutants, pathogens).

Dimensioning of the wetland according to DWA-A 262 (2017); DWA-A 178 (2018, draft).
- Hydraulic, COD and TSS areal loading rates,
- Water distribution.

Advantage

Ozone treatment + Constructed wetlands
- Reduction of biodegradable transformation products,
- Higher disinfection safety.

Water reuse.

Figure 3: Tertiary treatment for water reuse. 20,000 m² vertical flow wetland.

Tertiary treatment

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