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Slow sand filtration of secondary effluent for wastewater reuse: Evaluation of performance and modeling of bacteria removal

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“The future is here. It’s just not widely distributed yet.” (William Gibson)

TABLE OF CONTENTS

I ABSTRACT

1,800 million people will be living in regions with absolute water scarcity by 2025. Wastewater is a valuable resource that is reliably available wherever water is consumed and collected for treatment. Appropriate technologies are needed for disinfection of wastewater to allow safe reuse. Slow sand filtration (SSF) is a simple process used for pathogen and particle removal in drinking water purification. It may be adapted for wastewater disinfection but only a few studies have been conducted on tertiary treatment of wastewater using slow sand filters.

The purpose of this work was to evaluate the suitability and performance of slow sand filters at laboratory and pilot scale as a potential process for disinfection of secondary effluent. Fecal indicator bacteria removal during filtration was modeled and a filter-cascade tested. The performance of slow sand filters was evaluated in comparison to intermittent sand filters.

In the experiments the key process parameters hydraulic loading rate, sand grain size distribution and filter bed depth were systematically varied. The results showed that slow sand filters are promising for disinfection of wastewater, especially in arid developing countries, if reuse in agriculture is intended. They eliminated 1.9–2.6 $\log_{10}$-units of *E. coli* and 1.9–3.0 $\log_{10}$-units of intestinal Enterococci reaching effluent concentrations of 11–142 CFU per 100 ml of *E. coli* and 2–24 CFU per 100 ml of intestinal Enterococci at pilot scale. Bacteria removal was shown to be a function of sand surface area, dirt layer and supernatant water. The schmutzdecke and upper centimeters of the sand bed were a very efficient zone for bacteria removal whereas removal per filter length declined within deeper zones of the bed. Sand surface area per filter surface area should not be chosen below 2000 m²/m². Slow sand filters removed 70–84% of total suspended solids reaching effluent concentrations of 1.2–2.3 mg/l and turbidity levels of 0.5–0.8 NTU. Average runtime was between 59 and 148 days.

The comparison of filters at laboratory scale and pilot scale fed with secondary clarifier effluent of the same wastewater treatment plant and operated at the same hydraulic loading rate showed that scale up did not significantly affect bacteria removal.

A slow sand filter achieved comparable or higher bacteria removal than an intermittent sand filter (ISF) even under challenging conditions in terms of BOD_5 and ammonium. The SSF eliminated 2.7–3.4 log-units of *E. coli* and 3.0–3.2 log-units of intestinal Enterococci. The ISF removed less bacteria: 1.8–2.5 log-units of *E. coli* and 1.6–2.8 log-units of intestinal Enterococci. A major advantage of the SSF compared to the established technology of ISF is the elevated hydraulic loading rate that can be used.

Bacteria removal in a rotating cascade consisting of four slow sand filter stages amounted to 1.7 log-units of *E. coli* and 1.9 log-units of intestinal Enterococci. The rotating cascade did not fulfil the expectations of increased bacteria removal due to the existence of four schmutzdecke layers. A cascade consisting of two slow sand filters was was used to treat effluent of a vertical flow constructed wetland. It achieved *E. coli* removal of 1.78 log-units in the first and 0.33 log-units in the second filter. This was comparable to typical removal in constructed wetlands so that SSF may be attractive because of lower land requirements.

A quantitative description of the processes leading to bacteria removal in slow sand filters for tertiary treatment of wastewater was lacking. Therefore a model was developed for *E. coli* removal from secondary clarifier effluent in slow sand filters. Removal was successfully simulated for sands of variable grain size distribution and under a range of hydraulic loading rates compared to data obtained at pilot-scale filters. The most important process was retention of bacteria at the schmutzdecke and sand surface leading to an enrichment by a factor of up to 600 compared to the surrounding bulk phase. Bacteria elimination and inactivation both in the bulk phase and the schmutzdecke can be described by a first order kinetic.

II ABBREVIATIONS

0/2 building sand	Washed sand that has passed through a mesh width of 2 cm
2/4 gravel	Gravel passing a mesh width of 4 cm and held back by a mesh width of 2 cm
4/8 gravel	Gravel passing a mesh width of 8 cm and held back by a mesh width of 4 cm
BOD_5	Biochemical oxygen demand in 5 days
CASO	Tryptic soy agar
CFU	Colony forming unit
COD	Chemical oxygen demand
d_{10}	effective size of a grain size distribution; theoretical mesh width, through which 10 mass-percent of a sand sample would pass
d_{60}	theoretical mesh width, through which 60 mass-percent of a sand sample would pass
EN ISO	European and international standard
EPS	Extracellular polymeric substances
EU	European Union
HLR	Hydraulic loading rate
HRT	Hydraulic retention time
ISF	Intermittent sand filter / Intermittent sand filtration
p	Porosity
pH	Negative decimal logarithm of the hydrogen ion activity in an aqueous solution
Rpm	Rounds per minute
SSF	Slow sand filter / Slow sand filtration
T	Temperature
TBA	Tryptone bile agar
TOC	Total organic carbon
TSS	Total suspended solids
U	Uniformity coefficient: $U = d_{60} / d_{10}$
UV	Ultraviolet radiation
WHO	World Health Organization
WWTP	Wastewater treatment plant

III LIST OF FIGURES

IV LIST OF TABLES

1 INTRODUCTION

1.1 Water scarcity, wastewater reclamation and reuse

Currently, 450 million people in 29 countries suffer from water shortages. 1,800 million people will be living in countries or regions with absolute water scarcity by 2025 and two thirds of the world population could be under stress conditions (FAO-Water, 2009). Reasons for this dramatic projection are the world's growing population, increasing standards of living and the global climate change (UNESCO, 2006). There are several definitions of water stress and it may have physical or economic reasons. A widely-used definition is that water stress and scarcity occur, when annual renewable water resources per capita drop below 1,700 m^3 and 1,000 m^3 respectively. While this may seem a lot considering the daily drinking water requirement of 2 liters per person, it must be kept in mind that 2,000 to 4,000 litres of water are needed to produce one person's daily food (UNWater, 2008). The concept of virtual water has brought attention to the large quantities of water needed to produce certain goods and services.

With increasing pressure on water resources, efficient use of water and alternative water sources gain importance. Wastewater is a valuable resource that is reliably available wherever water is consumed and collected for wastewater treatment. Currently this potential is widely neglected: in developing countries 90% of the wastewater is discharged without any treatment harming both the environment and drinking water sources (UN, 2002). Worldwide reclamation and reuse of wastewater is called for by experts as in the Bellagio Principles in order to close both water and nutrient cycles (WSSCC, 2000). Since most conventional wastewater treatment releases high numbers of microorganisms, disinfection of secondary effluent is necessary to safeguard public health and the environment. Effluent quality should be fit for irrigation, because the agricultural sector accounts for 75% of freshwater consumption worldwide and up to 90% in developing countries (UNESCO, 2006). Sewage, often untreated, has already been used to irrigate 10% of crops grown worldwide by 2004 (Scott et al., 2004). Potable reuse would require more elaborate treatment and widely lacks public acceptance whereas reuse in agriculture allows recycling the fertilizers nitrogen and phosphorus. Disinfection of wastewater also permits the use of surface waters for recreation and impedes the spreading of antibiotic resistances. The World Health Organization has developed guidelines for safe reuse of wastewater and the European Union has defined quality requirements for bathing water (EU, 2006; WHO, 2006).

Safe, reliable and effective technologies for wastewater disinfection are needed. Above all, they should be appropriate for local conditions because water stress and scarcity affect many people in emerging and developing countries (Figure 1-1).

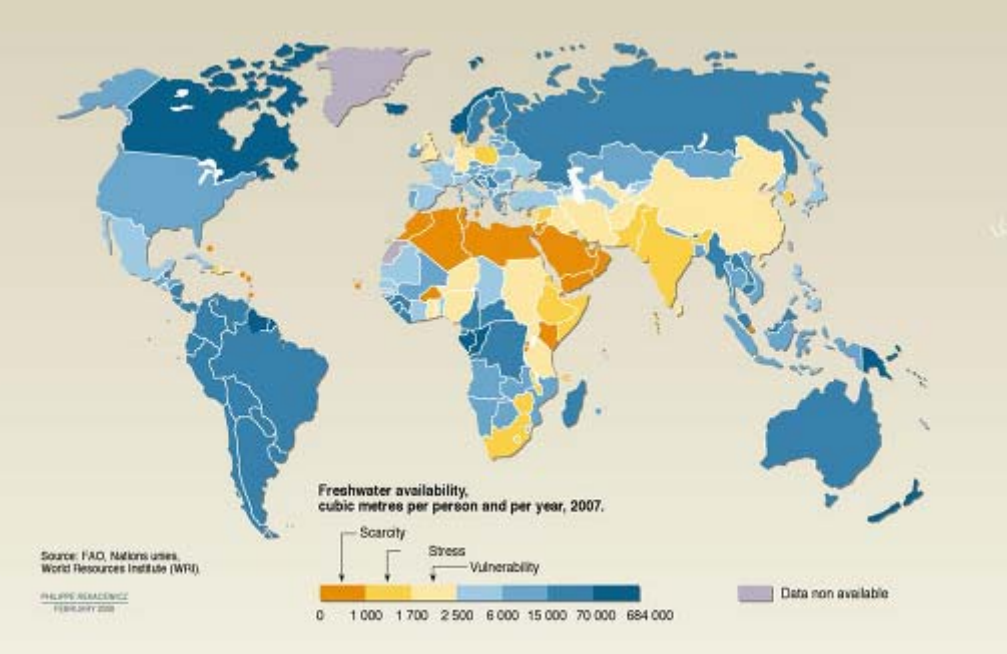


Figure 1-1: Global water stress and scarcity (UNEP, 2008)

Appropriate technologies must be both effective and deliver an excellent price-performance-ratio. To ensure sustainable operation, the relevant stakeholders should be involved in the decision-making and local conditions need to be taken into account such as availability of electricity, chemicals, spare-parts and skilled labor. A simple process such as slow sand filtration can fulfill these criteria (Visscher et al., 1987).

Other available technologies have several drawbacks: Chlorination has the disadvantage of by-product formation, a problem mainly associated with potable reuse. UV treatment, ozonation and membrane technologies effectively remove pathogens from secondary effluent but have considerable operating and investment costs and may require elaborate pre-treatment (Lazarova et al., 1999). In addition chlorination and UV-treatment may not be sufficiently effective against parasite eggs. Natural processes such as disinfection in constructed wetlands and wastewater stabilization ponds may be more appropriate solutions for emerging countries but require large surface areas and have the problem of evaporation losses (Gearheart, 1999).

1.2 Slow sand filtration

1.2.1 State of the art

Slow sand filtration (SSF) is a simple technology for purification of surface water in drinking water production. It is an effective particle and pathogen filter that combines biological, physical and chemical processes (Obst, 1990; Logsdon et al., 2002). The process is passive and the filter's effectiveness is dependent mostly upon the development of a biofilm attached to the sand grains and the schmutzdecke, a biologically active mat that develops on the filter surface. Slow sand filtration is the oldest engineered process of water purification. Plants of a design similar to modern units were developed by James Simpson as soon as 1829 for the London water supply. Water-borne epidemics encouraged a widespread use of this technology by the end of the nineteenth century. The most prominent example is the cholera outbreak that caused more than 7,500 deaths in Hamburg in 1892 and left neighboring Altona nearly unaffected. Located downstream of Hamburg, Altona's water supply was protected by slow sand filters and not only by sedimentation (Huisman and Wood, 1974). However, coagulation, rapid filtration and chlorination began to dominate surface water treatment in the first half of the twentieth century due to lower land requirements and less labor required. Still SSF is an essential element of drinking water purification in cities like London, Amsterdam, Zürich and Stockholm and is used for artificial recharge along the river Ruhr in Germany (Ellis, 1985). During the past 30 years slow sand filtration has experienced a renaissance in rural settings of northern America and in developing countries (Logsdon and Fox, 1988). In Latin America, Africa and Asia SSFs have been built for community water supply. Slow sand filters in the form of Bio-Sand Filters (BSF) serve an estimated number of 500,000 people in developing countries as a point-of-use treatment for surface water (Elliott et al., 2008; Manz, 2008; Blackburn&Associates, 2009; BushProof, 2009; CAWST, 2009).

Slow sand filters are gravity-driven and can be distinguished from rapid filters by their lower filter velocities of typically 0.05–0.4 m/h, finer filter material and the fact that they are not designed for backwashing (Rachwal et al., 1996). Due to the fine filter sand, suspended matter is mostly retained at the filter surface and does not penetrate into the depth of the bed. The microbial community in SSFs is not disturbed by backwashing so that biological purification mechanisms play an important role. Figure 1-2 shows a cross section of a typical slow sand filter (Logsdon, 1991). The supernatant water mainly serves to provide the driving pressure difference and protect the schmutzdecke from mechanical stress. The schmutzdecke itself, sometimes referred to as dirt layer, biological layer or filter skin, is a biologically very active

filter cake on top of the filter bed. The bed consists of an inert porous material, usually washed sand. The underdrainage system comprises gravel to support the filter medium and drainage pipes or channels to collect the treated water. A weir ensures that the bed stays permanently wet. The flow through the system is controlled by several valves. Typical recommendations for SSFs in drinking water purification are: 0.15 mm < d_{10} < 0.4 mm; U <5; filter bed depth >50 cm; hydraulic loading rate of 0.05–0.4 m/h. In the following paragraphs slow sand filter design, processes and performance in drinking water purification are summarized (Huisman and Wood, 1974; Visscher et al., 1987; Visscher, 1988; Hendricks, 1991; Logsdon, 1991; WeberShirk and Dick, 1997b, a; Huisman, 2004; Stevik et al., 2004; Sánchez et al., 2006). In-depth coverage can be found in these sources.

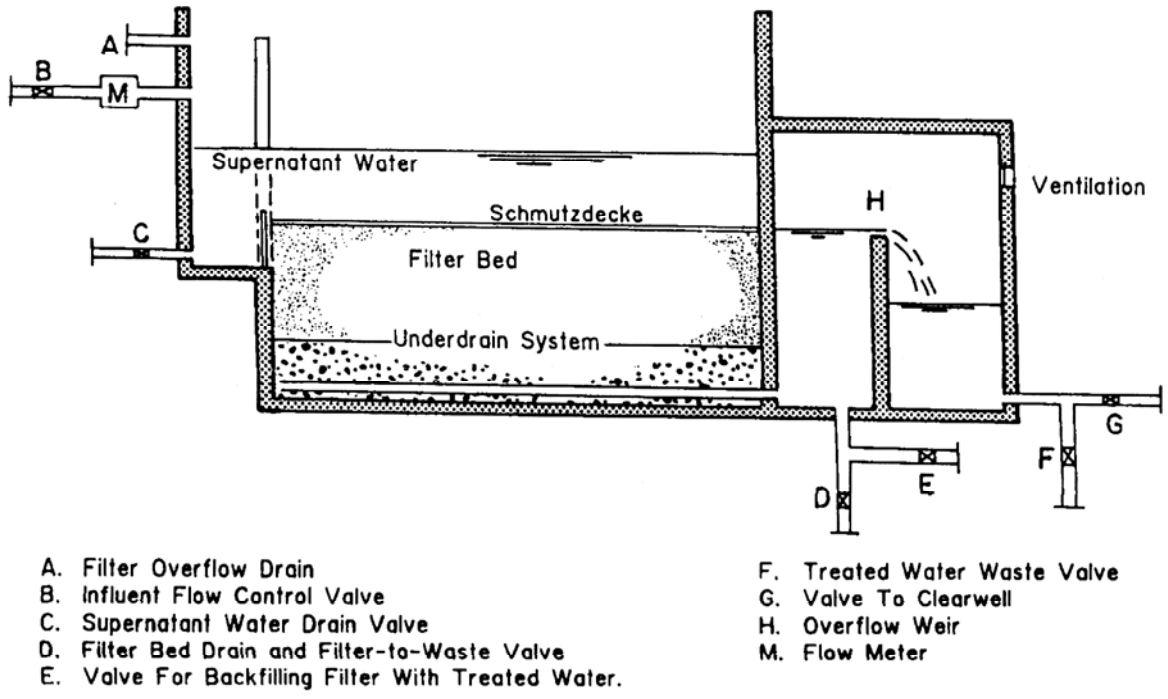


Figure 1-2: Basic components of a slow sand filter with influent control and rising water level (Logsdon, 1991)

The process of slow sand filtration starts in the supernatant, where the water is retained for several hours and some particles agglomerate and sediment. In uncovered filters, algal action is of importance, too. Before reaching the filter medium itself, the water has to pass through the schmutzdecke. This thin layer is of critical importance to the purification process (Hendel et al., 2001; Dizer et al., 2004; Unger and Collins, 2006). A considerable proportion of suspended solids is mechanically strained in the schmutzdecke independent of the filter velocity. The schmutzdecke consists of sedimented and entrapped particles and various

microorganisms. Bacteria and protozoa contribute to a high biological activity by breaking down the available organic matter. A ripening period of several weeks must be allowed before the schmutzdecke has fully developed after inauguration and the filter functions properly. Slow sand filters are operated continuously for about 2 months until the filter resistance has reached a critical value due to the development of the schmutzdecke. Then the schmutzdecke is removed by scraping or wet-harrowing and the cycle restarts (Eighmy et al., 1992). Filter runtime depends on the concentration of suspended solids, the hydraulic loading rate and biomass development (Mauclaire et al., 2004).

Below the schmutzdecke the water passes through the pores of the filter medium in a laminar flow. The processes taking place are comparable to those during groundwater recharge. Due to the retention in the schmutzdecke, it is unlikely that straining within the bed constitutes much to the purification process. Even in a fine sand with grains of 0.15 mm diameter, only particles larger than 23 μm in diameter would be completely held back by straining since pore channels in a sand of uniformly packed spheres have a maximum diameter of 15.5% of the grains. While mechanical straining is an important removal process for Helminth eggs, bacteria of 0.1–10 μm and even smaller viruses would pass through the filter unaffected (Tufenkji et al., 2004). They may be held back by straining when they are aggregated or attached to larger particles. Pore space is narrowed down by growth of biofilm but the most important mechanism for retention is adsorption. One cubic meter of filter sand easily provides a theoretical surface of 10,000 square meters. Microbial degradation ensures self-regeneration of the filter bed. Biological activity is highest in the schmutzdecke and upper 5–10 cm of the sand bed. It declines with filter depth since less substrate is available and may fade away below 40 cm bed depth (Ellis and Aydin, 1995; Campos et al., 2002).

The filtration mechanisms within a slow sand filter can be divided into retention caused by transport and attachment/adsorption as well as purification/elimination. Within the filter these processes constantly interact and are still not completely understood. Transport mechanisms include straining, interception, sedimentation, diffusion, inertial and centrifugal forces, mass attraction and Coulomb forces (Johnson et al., 1995; Harmand et al., 1996; Chen et al., 1998; Truesdail et al., 1998; Powelson and Mills, 2001; Auset and Keller, 2006; Foppen et al., 2006; Morales et al., 2007). Once particles and colloids have come into contact with the sand grain surface or biofilm on the sand or in the schmutzdecke, van-der-Waals forces, electrostatic attraction and adhesion to the biofilm can cause attachment/adsorption. In rapid filtration, particle removal due to transport and attachment mechanisms is described by Iwasaki's equation:

$$\frac{dc}{dz} = -\lambda c \qquad \text{Equation 1-1}$$

in which c is the concentration of particles (in number of particles/ml), z is the distance from the top of the filter bed (in meters) and λ is the filtration coefficient (in cm^{-1}). The coefficient λ is the product of the transport probability coefficient η and the attachment coefficient α (Hendricks, 1991).

Since both sand and bacteria have a negative surface charge the high extent of bacteria retention in slow sand filters is a matter of debate. Repulsion due to similar surface charges is the reason, why rapid sand filters are largely ineffective if not preceeded by coagulation. In slow sand filters, extracellular polymeric substances (EPS) in the biofilms that coat the sand surface are considered to play an important role in the process of adsorption (Chen et al., 1998; Truesdail et al., 1998).

Elimination processes can be attributed to biotic and abiotic factors. For the allochtonous intestinal microorganisms the conditions in the filter are unsuitable. Temperatures in the filter are too low and their nutritional needs are not satisfied. Intestinal bacteria must compete with well-adapted autochtonous microorganisms that sequentially assimilate and dissimilate particulate, colloidal and dissolved matter. The most important process for elimination of bacteria is considered to be predation by organisms such as protozoa as well as lysis by bdellovibrio and bacteriophages (Lloyd, 1996; Weber-Shirk and Dick, 1999).

These purification processes lead to the removal of bacteria, viruses and protozoa, typically 2–3 log-units or 99–99.9 % of fecal indicator *E. coli* (Fogel et al., 1993; Yahya et al., 1993; Timms et al., 1995; Dullemont et al., 2006). Suspended solids and turbidity are also removed. Nitrification and denitrification can occur simultaneously (Nakhla and Farooq, 2003). Due to the biological consumption of nutrients, regrowth of microorganism in the filtered water is minimized.

Limitations of slow sand filters are that color, persistent organic chemicals and heavy metals in the raw water may not be adequately removed. High levels of turbidity can be coped by pre-treatment with gravel filters as in multi-stage filtration. SSFs require more space and unskilled labor than higher engineered processes such as rapid filtration. High costs of land and labor can thus prohibit their use. Low temperatures in winter adversely affect purification and can only be overcome by structural precautions against freezing (Jabur, 2006).

The last two aspects usually do not present problems in rural areas of most developing countries whereas many advantages encourage the use of slow sand filters. Main aspects are

their simple and economical construction, operation and maintenance using local materials and skills. They do not require chemicals and if a sufficient slope exists, no energy is needed.

1.2.2 Open questions and challenges

Slow sand filters may be adapted for wastewater disinfection but only a few studies have been conducted on tertiary treatment of wastewater using slow sand filters (Adin, 2003). They showed total coliform bacteria removal of 0.3–3.5 log-units (Ellis, 1987; Farooq and Alyousef, 1993; Adin et al., 1998; Sadiq et al., 2003), fecal coliform removal of 2 log-units (Keraita et al., 2008), *E. coli* reduction of 2.3–3.7 log-units and Enterococci removal of 0.7–2.6 log-units (Mälzer, 2005) mainly depending on sand properties, filter design, hydraulic loading rate and raw water quality. However, information on relevant fecal indicator bacteria like *E. coli* and intestinal Enterococci is extremely limited. Total coliform bacteria do not appear to be a suitable indicator for monitoring disinfection in slow sand filters. Some coliforms are autochthonous to soil and water and may multiply in slow sand filters (Logsdon, 1991; Adin et al., 1998; Petry-Hansen, 2005; Petry-Hansen et al., 2006). In addition, the WHO guidelines for reuse of wastewater and EU Directive for bathing water quality only consider *E. coli* and intestinal Enterococci.

No systematic variation of the key process parameters filter bed depth, hydraulic loading rate and sand grain size distribution over the potential range of effective size (d_{10}) and uniformity with respect to the removal of relevant indicator organisms *E. coli* and intestinal Enterococci has been reported in the literature. Moreover, mechanistically plausible correlation of performance to the major process parameters, such as sand characteristics and hydraulic loading rate (HLR) deserves more attention. Sadiq et al. (2003) performed a multiple regression analysis of total coliform bacteria removal as a function of d_{10}, filtration rate and sand bed depth. They found that an increase in sand bed depth improved removal, whereas an increase in sand grain size did not diminish it significantly. Unfortunately this study only compared sands of two different $d_{10,}$ did not present their uniformity coefficients and used total coliform bacteria as an indicator.

1.2.3 Modeling of slow sand filtration

Up to date filter design and operation mostly rely on empirical experiences gained at laboratory and full scale over the last centuries. Variable ambient conditions as well as differences in key design and process parameters from one setting to another impede

comparison of the data as well as generalized insights into the filtration process and their influence on performance. In the field of drinking water purification models were developed to overcome these limitations (Ojha and Graham, 1994, 1996a, b; Rödelsperger, 2005; Campos et al., 2006a, b; Rödelsperger, 2006). However, the models of Campos (2006) and Rödelsperger (2006) were not used to simulate fecal indicator bacteria removal from secondary effluent. Another model is based on total coliform removal from secondary effluent (Sadiq et al., 2004). But the use of total coliforms as indicator organisms in slow sand filtration can be problematic because they have been shown to multiply in filters (see 1.2.2).

1.3 Intermittent sand filtration

Intermittent sand filtration or infiltration percolation is a simple technology widely applied for secondary and tertiary wastewater treatment. It is used on-site for treatment of septic tank effluent to oxidize organic matter and nitrogen and remove suspended solids and pathogens. Primary effluent of small rural communities is also treated. ISF is further used for polishing of secondary effluents when reclaimed wastewater is reused or disposed of in sensitive environments (Brissaud et al., 1999). A cross-section of an intermittent sand filter is shown in Figure 1-3. While the sand used in ISF is similar to slow sand filtration, the intermittent operation differs from continuously fed SSFs.

ISFs are operated in typically 1-12 cycles of feeding and draining. At the beginning of a cycle, the ISF is temporarily flooded with water that is dosed through a distribution network. The water infiltrates and percolates through the sand bed displacing the water from the previous cycle in the pores. A gravel layer and underdrainage system collect the water. Intermittent operation has the benefit of sucking air into the filter bed when the water of one feeding interval percolates down. This allows diffusion of oxygen into the film of water covering the sand grains thus improving oxidization. The introduction of the air-water-interface has also been reported to improve retention of bacetria (Auset et al., 2005).

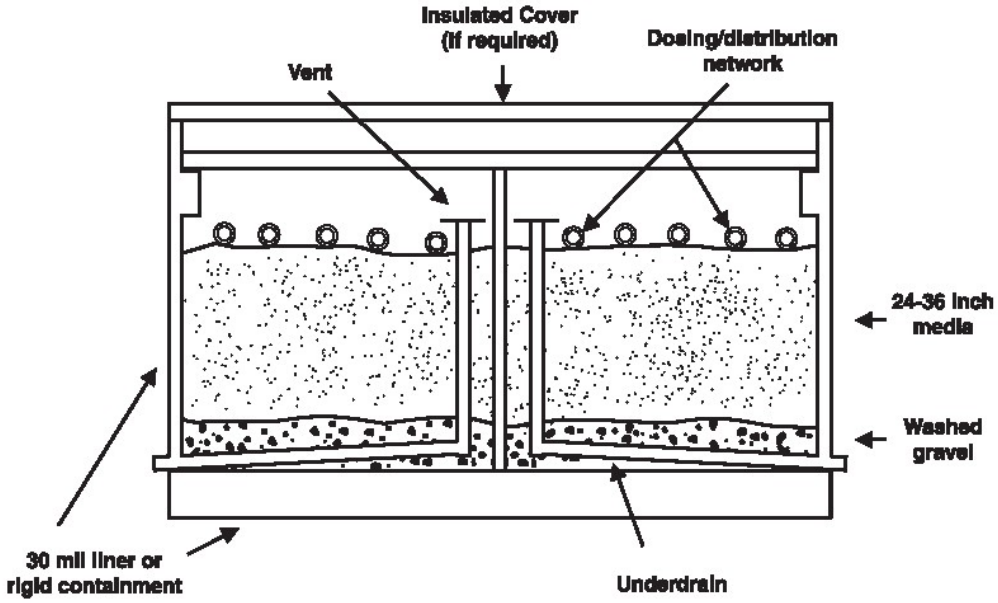


Figure 1-3: Cross-section of an intermittent sand filter (USEPA, 2007)

Table 1-1 summarizes major design and operational characteristics of intermittent and slow sand filters. The specifications for sand to be used in both filters are quite similar. ISFs tend to have deeper filter beds and are operated at lower hydraulic loading rates.

Table 1-1: Comparison of typical design and operational parameters of intermittent (ISF) and slow (SSF) sand filters (Huisman and Wood, 1974; Lienard et al., 2001; Makni, 2001; Bancole et al., 2003; Sánchez et al., 2006; Brissaud et al., 2007).

		ISF	SSF
Design	Material	Sand	Sand
	Bed depth [m]	1.5 – 2.0	0.5 – 1.5
	Sand size [mm]	d_{10}: 0.25 – 0.4	d_{10}: 0.15 – 0.4
		d_{50}: 0.2 – 0.8	
	U	< 10	< 5
	Supernatant water [m]	temporary after feeding	0.3 – 1.5
Operation	Loading	intermittent	continuous
	HLR [m/h]	< 0.03	0.05 – 0.4
	Schmutzdecke	no	yes

SSFs are permanently water-saturated and a level of supernatant water rests above the filter bed. Therefore, the surface area available for diffusion is much smaller and oxygen supply depends on the concentration of dissolved oxygen in the influent. SSF differs from ISF because no schmutzdecke develops on the latter. Apart from these differences, the processes of retention and removal of microorganisms in intermittent and slow sand filters are comparable (Stevik et al., 1999a; Selas et al., 2003).

Tertiary treatment with ISFs is capable of achieving effluents with a maximum of 1,000 CFU/100 ml of fecal coliforms (Salgot, 1996) and is dependent on the grain size (Stevik et al., 1999b; Ausland et al., 2002). Thus the performance of the available technology ISF is similar to but has not been directly compared with SSFs for disinfection of secondary effluent. The advantage of slow sand filtration may be the elevated hydraulic loading rate resulting in smaller plant design and thus less investment costs. But due to the better oxygen supply in ISFs, they should be superior to SSFs in treating secondary effluents with high oxygen demands. High oxygen demands might be detrimental to SSF performance, because anaerobic conditions have been reported to severly diminish bacteria removal (Ellis, 1985; Visscher et al., 1987).

1.4 Research objectives and hypotheses

The purpose of this work was to evaluate the suitability and performance of slow sand filters at laboratory and pilot scale as a potential process for disinfection of secondary effluent along a systematic variation of major design and operational parameters. Fecal indicator bacteria removal during filtration was modeled and an innovative concept of a filter-cascade tested. The performance of slow sand filters was evaluated in comparison to intermittent sand filters.

1.4.1 Slow sand filter performance

The objectives of the present work were to quantify the effect of sand grain size distribution, bed depth and hydraulic loading rate on the removal of *E. coli,* intestinal Enterococci and suspended solids as well as runtime between filter maintenance. A mechanistically sound correlation of filter performance to these key process parameters had to be developed. The scale-up from laboratory to pilot scale had to be evaluated. Bacteria removal and filter effluent concentrations had to be assessed with respect to the WHO guidelines for the safe reuse of wastewater and the EU bathing water Directive (EU, 2006; WHO, 2006). WHO guidelines are considerably less restrictive than water reuse standards in the U.S. and some

other developed countries. They were chosen because SSF may have the largest potential in developing countries. Bacteria removal had to be investigated within the filter compartments supernatant water, the schmutzdecke and different depths of the sandbed itself to identify the most active and efficient zones. Recommendations for filter design and operation based on generalized insights into the process and potential for optimization should be derived.

The corresponding hypotheses were tested:

1. The relevant processes for bacteria removal are straining and adsorption in the schmutzdecke, adsorption in the filter bed and elimination.

    a. Bacteria removal is mostly due to adsorption and thus a function of the available sand surface area. The finer and more homogenous a sand is (small d_{10} and U) and the deeper the filter bed, the higher should be the bacteria removal.

    b. The schmutzdecke plays an important role in bacteria removal.

    c. Bacteria removal per filter length may decline in deeper zones of the filter bed making deeper zones less efficient.

    d. An increase in hydraulic loading rate only leads to a minor decrease in bacteria removal, although hydraulic detention time is reduced. Elimination mostly affects irreversably adsorbed bacteria and their detention time does not depend on filter velocity.

2. The relevant process for particle removal is straining in the schmutzdecke.

    a. Sand grain size distribution and hydraulic loading rate do not have a significant influence on suspended solids removal, since the schmutzdecke accounts for the majority of removal. It develops independently of the underlying sand after the first ripening and the mechanical process of straining is not affected by filter velocity.

3. Clogging is mainly determined by the load of suspended solids in the filter influent.

    a. Filter runtime between two maintenance events is a function of the load of suspended solids. Therefore, an increase in HLR should lead to a shorter runtime.

b. The smaller the d_{10} of a sand and the higher the uniformity, the shorter should be the filter runtime. Fine sands have smaller pore channels and lower hydraulic conductivity than coarse sands, heterogenous sands have lower porosities.

1.4.2 Comparison of slow sand filtration and intermittent sand filtration

The performance of slow sand filters and intermittent sand filters for bacteria removal under low, medium and high oxygen demands was evaluated. ISFs are an established simple technology for bacteria and particle removal from secondary effluent. They are also used for secondary treatment, because the intermittent operation supplies sufficient oxygen for oxidization. Slow sand filters are usually used for treatment of surface water. Due to the limited oxygen diffusion, SSFs may not be suitable for disinfection of secondary effluent with high oxygen demand. In Germany, communal wastewater treatment plants may discharge up to 40 mg/l of BOD_5 and limits to ammonia concentrations regularly only exist for plants treating more than 200 kg/d BOD_5 (BMU, 2004).

The following hypotheses were tested:

1. Elevated levels of biochemical oxygen demand and ammonium cause depletion of oxygen and lead to anoxic and anaerobic conditions in zones of the slow sand filter. This harms the aerobic predators that are considered to be of central importance in the elimination of bacteria and leads to a significant decrease in bacteria removal.
2. The performance of the ISF should not be affected by elevated levels of biochemical oxygen demand and ammonium, because the intermittent operation ensures sufficient transport of oxygen into the filter.
3. The runtime of the SSF depends on the load of suspended solids. The ISF does not exhibit clogging even when it is fed with high concentrations of suspended solids.

1.4.3 Rotating cascade of slow sand filters

An innovative design of a filter cascade was evaluated. Four filters in series were rotated every week so that during one month a filter went through the first, fourth, third and second stages. The underlying idea was to move the biologically very active schmutzdecke and upper sand layer of the filter at first stage to stages of less biological activity. It is generally accepted

that in the depth of the bed of slow sand filters biological activity and bacteria removal per filter length decline. The cascade was therefore designed as one slow sand filter divided into four rotating units to improve activity and bacteria removal in the less efficient deeper zones of SSFs. It was also expected that runtime could be increased because the schmutzdecke was only challenged for one week with comparatively high TSS loads at first stage and it was allowed to recover afterwards. Hydraulic conductivity was anticipated to increase at stages four to two due to TSS removal almost exclusively taking place at first stage and allowing mineralization of the schmutzdecke during three weeks.

1.4.4 Slow sand filtration for disinfection of constructed wetland effluent

The potential of slow sand filtration for disinfection was also evaluated with secondary effluent from a vertical flow constructed wetland. It was intended to establish a complete process for wastewater treatment including disinfection that only used simple, natural processes.

1.4.5 Model

Another purpose of this work was to establish a model of fecal indicator bacteria removal from secondary clarifier effluent in slow sand filters using the software AQUASIM developed at EAWAG (Reichert, 1994). The model was evaluated for appropriate description by comparing the simulation results to data of *E. coli* removal obtained from two of the pilot scale filters of different sand grain size operated over a range of hydraulic loading rates. The filters were also subject to changing ambient conditions like temperature and composition of the secondary effluent mostly due to seasonal variations.

The objectives of the present work were to quantitatively describe the most relevant processes leading to *E. coli* retention and removal. The influence of key process parameters on filter performance should be simulated and compared with experimental data. The model and its aggregated description of the filtration process should serve as a tool to predict performance under various design, operating and ambient conditions and also as a basis for systematic improvement of filter performance.

2 MATERIALS AND METHODS

2.1 Experimental setup

2.1.1 Laboratory scale slow sand filters of variegated grain size distribution

The experimental setup consisted of 4 filter columns with a diameter of 50 mm and a height of 210 cm containing a sand filter bed supported by gravel layers. The filter sands were mixed to vary in d_{10} while the uniformity coefficient (U) was kept constant. The sand fractions for mixing had been obtained by sieving washed and dried 0/2-building sand from the quarry operated by Freudlsperger (Sprotta, Germany). Each column was made of two tubes of acrylic glass that were joined 5 cm above the sand level and protected from light by an insulation foam. The minimum supernatant level was controlled by an outflow weir. The upper tube could be removed for maintenance of the filter when the supernatant water had reached the overflow at 150 cm above the sand level. Wet-harrowing was applied for maintenance: The supernatant water was drained to 5 cm above the sand surface, the schmutzdecke and the upper 2 cm of the sand stirred with a scoop and 80% of the dirty water drained with a syringe. The remaining matter was allowed to settle in order to form the next schmutzdecke. Filter design, material and operating conditions are listed in Table 2-1.

Table 2-1: Filter design, material and operating conditions of the 4 slow sand filters C1–C4 at laboratory scale

	C1	C2	C3	C4
d_{10} [mm]	0.25	0.4	0.63	0.8
U	1.6			
Porosity [%]	37.6	38.9	39.9	40
Target hydraulic loading rate [m/h]	0.05			
Sand bed depth [cm]	50			
Support layer	5 cm of 4/8 plus 5 cm of 2/4 gravel			
Supernatant [cm]	min. 30			

The filters were fed by a peristaltic pump (MCP Standard ISM 404, ISMATEC, Glattbrugg, Switzerland) with secondary clarifier effluent from the WWTP Langenreichenbach

(Germany). Feeding water was stored in a 120 liter barrel protected from light and insulated with foam, cooled to 4–8 °C (WKL 230, LAUDA, Lauda-Königshofen, Germany) and continuously stirred (MAXIMR1, IKA, Staufen, Germany) at 300 rpm for one week before renewal (Figure 2-1).

Figure 2-1: Laboratory installation of the 4 slow sand filter columns C1–C4 as well as the storage barrel, pumps and cooling unit

Samples for microbiological enumerations were taken from the filter influent, effluent and from sampling ports placed 2 cm above the sand bed surface as well as 5, 10 and 25 cm below. They were collected in autoclaved centrifuge tubes of 50 ml and analyzed directly. Additional samples of 50 ml were taken from filter influent and effluent and analyzed for temperature, dissolved oxygen, redox-potential, pH and conductivity. 2 sample bottels of 500 ml were allowed to fill and were analyzed for turbidity and suspended solids as well as BOD_5. A sample fraction of 10 ml was kept at –20 °C for monthly ion chromatography and determination of COD and TOC.

2.1.2 Pilot scale slow sand filters of variable grain size distribution

The experimental setup consisted of 6 PVC pipe filter columns (KG DN 200) containing a sand bed of 50 cm supported by 5 cm of gravel of 2–4 mm diameter on top of 5 cm of gravel of 4–8 mm diameter (Figure 2-2). The filters were completely protected from light to avoid

growth of algae but were not protected against temperature changes. Peristaltic pumps (MCP Standard ISM 404, ISMATEC, Glattbrugg, Switzerland) were used to continuously feed the columns with secondary clarifier effluent. The slow sand filters were located in Langenreichenbach (Saxonia, Germany) on the site of an activated sludge wastewater treatment plant with denitrification and biological P-elimination. The target hydraulic loading rate was increased from 5 cm/h in phase I to 10 cm/h in phase II and 20 cm/h in phase III of the experiments. Samples were taken from filter influent, effluent and from sampling ports placed 2 cm above as well as 5, 10 and 25 cm below the sand bed surface. The minimum supernatant level of 30 cm was controlled by an outflow weir.

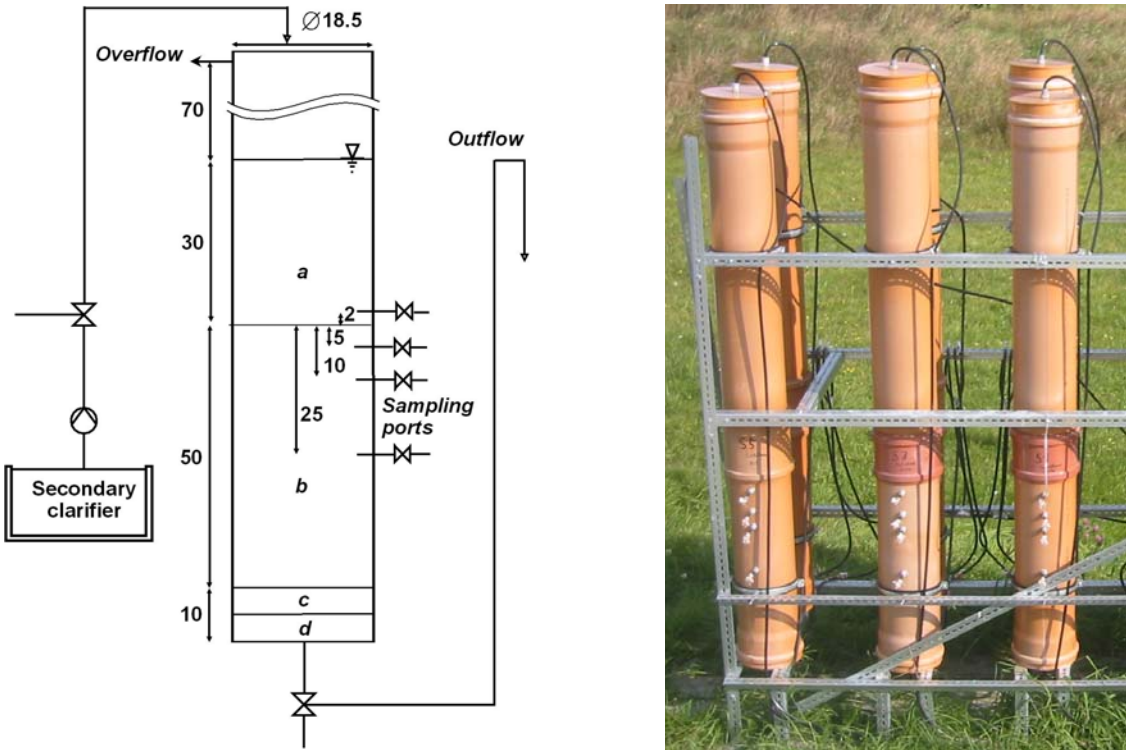


Figure 2-2: Schematic drawing and photo of the pilot scale filter columns S1–S6 consisting of the layers a) supernatant water, b) sand bed and c-d) gravel (dimensions in cm)

The supernatant water level could rise an additional 70 cm with increasing filter resistance. Each filter run was terminated when an overflow of water was noticed. Wet-harrowing was used for filter maintenance: supernatant water was drained to approximately 10 cm above the sand surface, the schmutzdecke and the upper 2 cm of sand were stirred with a shovel and then the muddy suspension drained through the sampling port 2 cm above the sand. The remaining matter was allowed to settle in order to form a new schmutzdecke.

Different grain size fractions of washed and dried sand were mixed to systematically vary effective size (d_{10}) and uniformity coefficient (U). They were supplied by Busch Quartz GmbH (Schnaittenbach, Germany) from a quarry esteemed for the spherical shape of the grains. The mixing yielded four sands of increasing d_{10} from fine to coarse (S1–S4) at a relatively constant U. The sands of S1, S5 and S6 were mixed to differ in uniformity while having approximately the same d_{10}. Material and operating conditions are listed in Table 2-2.

Table 2-2: Filter material characteristics and operating conditions of 6 slow sand filters at pilot scale (S1–S6)

		S1	S2	S3	S4	S5	S6
d_{10} [mm]		0.25	0.38	0.67	0.82	0.26	0.23
U		1.56	1.63	1.36	1.51	2.92	4.91
Porosity [%]		40.0	41.6	40.2	40.1	36.5	33.9
Hydraulic loading rate [cm/h]	Phase I	5.1	5.5	5.5	5.3	5.5	5.4
	Phase II	9.2	10.3	10.2	10.2	9.9	10.1
	Phase III	18.8	19.3	20.0	19.7	18.9	19.7
Specific sand surface area [m^2/m^3]		10388	6474	4226	3228	7590	7239
		Cumulated sand surface area [m^2]					
Filter bed depth [cm]	5	14	9	6	4	10	10
	10	28	17	11	9	20	19
	25	70	44	28	22	51	49
	50	140	87	57	43	102	97

A maturation phase of 4 weeks after start-up of the pilot filters was allowed to reach stable operating conditions. After raising the hydraulic loading rate from 10 to 20 cm/h, five days were allowed before sampling.

The hydraulic loading rate (HLR) was determined weekly for each filter by gathering the effluent for one minute in a graduated cylinder and calculating the volumetric flow rate ($\dot{V}$) per filter surface (A_F):

$$HLR = \frac{\dot{V}}{A_F} \qquad \text{Equation 2-1}$$

The actual HLR was compared to the target HLR and, if neccessary, adjustments were made to the pump speed. 50 ml samples were taken from all sampling ports twice per week in autoclaved centrifuge tubes for microbiological analysis. They were transported to the laboratory in a cool box and membrane filtration for determination of intestinal Enterococci started on the same day. The remaining samples were stored in the dark at 4–6 °C until the next morning for analysis of *E. coli.* 1,000 ml samples were collected once per week from filter influent and effluent. The samples were directly analyzed for temperature, dissolved oxygen, redox-potential, pH and electrical conductivity. Turbidity and suspended solids were measured in the laboratory and determination of BOD_5 started after samples had reached room temperature. A sample fraction of 10 ml was kept at –20 °C for monthly ion chromatography (ammonium, nitrate, nitrite, sulfate) and determination of COD and TOC.

2.1.3 Slow sand filter at larger pilot scale

Additionally one slow sand filter at larger pilot scale was installed adjacent to S1–S6. It consisted of a plastic barrel for rainwater catchment with a slightly conical shape (minimum diameter of 65 cm and maximum diameter of 70 cm). The barrel was filled with 25 cm of building sand in the range of 0–2 mm (obtained from Sprotta quarry; d_{10} = 0.45 mm; U = 2.3; porosity 37.9%) on top of 5 cm of 2/8 gravel and 5 cm of 8/16 gravel. The feeding water was the same as used for the other filters at pilot scale. The hydraulic loading rate was 5 cm/h or 400 liters per day. The secondary clarifier effluent was pumped through a tube fixed to a connector in the center of the lid. The lid was held on the barrel by three clamps. Another tube was fixed to the connector placed in the wall of the barrel 1 cm above the bottom. It led to a T-piece-connector serving as a weir that was fixed 65 cm above the bottom. This ensured that a minimum supernatant water level of 30 cm was maintained. At a height of 70 cm above the bottom another connector and tube were installed as an overflow. The filter was operated at the same time as S1–S6 in phase I. Sampling and control of the flow rate was performed on the same days and in a similar manner. Wet-harrowing was used as the maintenance technique.

2.1.4 Slow sand filter compared to intermittent sand filter at laboratory scale

An SSF and ISF were installed at laboratory scale (Figure 2-3). Each consisted of two glass tubes with an inner diameter of 5 cm and a total height of 210 cm that were wrapped in black foil to avoid growth of algae. The filters were equipped with the same washed 0/2-building-

sand that had been obtained from Freudlsperger at the Sprotta quarry. The sand layer of 50 cm height was supported by 5 cm of 2/4 gravel on top of 5 cm of 4/8 gravel and a perforated plate. The minimum supernatant water level of 30 cm in the SSF was controlled by an outflow weir whereas the water in the ISF was allowed to drain freely. With increasing filter resistance the water level could rise up to 140 cm above the sand bed surface and then exit through an overflow.

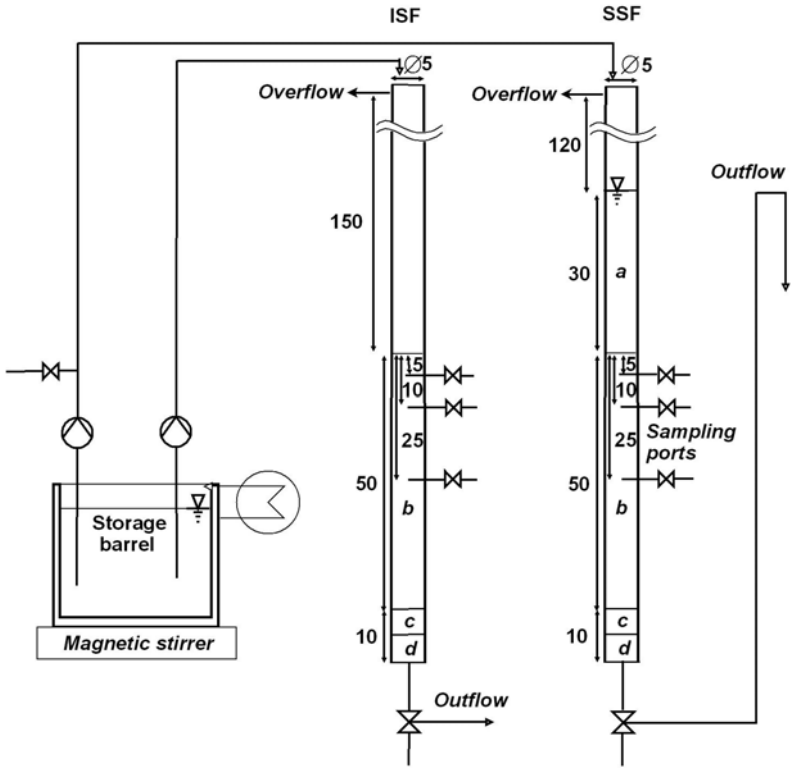


Figure 2-3: Schematic drawing (dimensions in cm) and photo of the experimental setup for comparison of slow sand filter (SSF) and intermittent sand filter (ISF) at laboratory scale

The filters were fed in three phases of increasing loading with biochemical oxygen demand, ammonium and particles. Filter influent in phase I was secondary clarifier effluent from the WWTP Langenreichenbach. Since the quality of the secondary clarifier effluent was very good, it was mixed in phase II and III with an increasing amount of combined primary effluent and recirculated sludge collected from the inflow to the aeration tank. The feeding water was stored as described in section 2.1.1. The SSF was continously fed by a peristaltic pump (Reglo, ISMATEC, Glattbrugg, Switzerland) at a target HLR of 65 cm/d or approximately 2.7 cm/h. The ISF was fed in 12 cycles per day (MCP Standard ISM 404, ISMATEC, Glattbrugg, Switzerland). To achieve the same HLR, 107 ml were pumped onto a diffuser plate above the filter sand every 2 hours for 40 seconds.

Composite samples of the effluent that had been collected in covered 500 ml bottles and grab samples from the storage barrel were analyzed for temperature, dissolved oxygen, redox-potential, pH, conductivity, indicator bacteria and BOD_5. Air in the bottle for collecting the samples had not been displaced by nitrogen. A sample fraction of 10 ml was kept at –20 °C for monthly ion chromatography and determination of COD and TOC. An additional composite sample of 500 ml was collected within the following 24 hours and analyzed for turbidity and suspended solids. Additional samples for determination of dissolved oxygen in the SSF were taken from sampling ports placed approximately 25 cm above the sand bed surface as well as 5, 10 and 25 cm below.

2.1.5 Slow sand filters in a rotating cascade at pilot scale

The filter cascade consisted of four filters that were operated in a series of four stages. The order of the filters was changed every week. Filters rotated from stage one to stage four, stage three and stage two so that every month a complete rotation was performed. The filters were similar in design to filters S1–S6 with the following exceptions. The sand bed only had a depth of 25 cm supported by 5 cm of 2/8 gravel and consisted of the same 0/2 building sand as the large pilot scale filter. The minimum supernatant water level was 10 cm and it could rise an additional 70 cm before reaching the overflow. The cascade was operated at a target HLR of 20 cm/h during the same days as phase II of filters S1–S6 and with the same secondary clarifier effluent. Samples were taken from the influent and effluent of the cascade and additional samples for determination of indicator bacteria were taken after stages one, two and three.

2.1.6 Slow sand filter cascade treating constructed wetland effluent at pilot scale

The cascade of two slow sand filters was fed with effluent of a vertical flow constructed wetland operated at the UbZ research facility at Langenreichenbach (Germany). Filter design and material were the same as for the large pilot scale filter. The effluent tube of the first stage filter was fixed to a connector in the side wall of the second filter barrel 20 cm above the sand surface. The SSF cascade was operated at a target HLR of 5 cm/h from May to July 2008. Samples were taken from the influent as well as the effluent of each of the two filters of the cascade.

2.2 Analytical procedures and calculations

2.2.1 Grain size distribution, porosity and hydraulic conductivity

In order to verify d_{10} and U of the sands used in this study, sand grain size distributions were analyzed (DIN18123, 1996). Sand samples of 500–1000 g were dried at 105 °C and analyzed in a vibratory sieve shaker (FRITSCH, Idar-Oberstein, Germany) with 13 sieves of mesh widths 0.063–8 mm. The weight of each empty sieve was subtracted from the weight of the sieve with retained sand and the mass calculated as percent of the total sand sample. The cumulated mass fractions were depicted over mesh width on a $\log_{10}$-scale. The d_{10} or effective size is the theoretical mesh width of a sieve through which 10 mass-percent of a sand sample can pass. The d_{60} is defined accordingly. The coefficient of uniformity (U) is calculated as the ratio of d_{60} to d_{10} and describes, how uniform in size the grains are. The result of a typical sieve analysis, showing how to graphically determine d_{10} and d_{60}, is shown in the annex (A-1).

Porosity was determined by filling a defined mass of dried sand (m_{sand}) into a graduated cylinder that had been weighed (m_{empty}) and then filled with water. The cylinder was shaken gently to achieve a similar porosity as in the filter columns. The supernatant water was removed, the volume of saturated sand was read (V_{total}) and the cylinder weighed again (m_{full}). Assuming a density of the water of 1 kg/l at room temperature, the porosity p in percent was calculated as follows:

$$p = \frac{V_{Pores}}{V_{total}} = \frac{m_{full} - m_{empty} - m_{sand}}{V_{total}} \times 100 \qquad \text{Equation 2-2}$$

Porosity depends on the uniformity of the grain size distribution, shape of the grains and the density of packing. It ranges from 25.9–47.6% for uniform spheres. The determined porosities were used to estimate hydraulic retention times (HRT) in the filters according to:

$$HRT = \frac{\sum l_i \times p_i}{HLR} \qquad \text{Equation 2-3}$$

with p_i being the porosity in the filter compartment (equal to 100% in the supernatant) and l_i the length of the compartment (measured from the regular level in the supernatant).

Hydraulic conductivity (k_f) of the different kinds of sand was determined experimentally. Darcy's law applies for laminar flow through a porous sample and can be given as follows:

$$k_f = \frac{V \times l}{A \times h \times t} \qquad \text{Equation 2-4}$$

A sand sample was filled into a cylindrical chamber of cross-sectional area A that was half-filled with water. The sand was fixed by several sheets of mesh wire and the length l of the sample determined. Water from a storage tank was allowed to flow through the sample. The level in the storage tank was held constant by a pump and overflow weir. The height difference h between the water level in the storage tank and the outflow of the cylinder was measured as well as the temperature of the water. Steady flow was allowed to establish for 5 minutes and then the water volume V passing through the sample during a certain time t was measured in a graduated cylinder.

The specific sand surface area (A_s) was approximated by:

$$A_s = \frac{6000}{d_s}(1-p) \qquad \text{Equation 2-5}$$

with d_s being the specific grain diameter:

$$d_s = d_{10}(1 + 2\log U) \qquad \text{Equation 2-6}$$

For any filter bed depth, the total sand surface area (A) that the water has passed through up to a sampling port can be calculated by:

$$A = A_s \frac{\pi d^2}{4} l \qquad \text{Equation 2-7}$$

with d being the inner diameter of the filter column and l the corresponding bed depth.

2.2.2 Microbiological analysis

All filters had been operated for a minimum of 4 weeks to allow for ripening of the schmutzdecke and stable operating conditions before sampling. At pilot scale, results were

obtained between September and November 2007 (Phase I, n_I = 16 samples), April and July 2008 (Phase II, n_{II} = 11 samples) and July and August 2008 (Phase III, n_{III} = 6 samples). At laboratory scale, 12 samples from the four filters C1–C4 were collected over a period of 10 weeks from January to March 2007. Phase I of the comparison of SSF and ISF lasted from July to October 2007 (n_I = 22 samples), phase II from November to December 2007 (n_{II} = 9 samples) and phase III from January to March 2008 (n_{III} = 13 samples).

Membrane filtration techniques were used for quantification of *E. coli* and intestinal Enterococci. An adquate sample volume and dilution were chosen to yield approximately 10–50 colony forming units. The sample was filtered through a sterile membrane filter with a pore size of 0.45 µm (GN-6 Metricell, PALL, East Hills, USA) in an autoclaved filtration apparatus. For *E. coli* determination according to EN ISO 9308-1 (DEV, 2007), the filters were then incubated at 36±0.5 °C on CASO-agar. After 4–5 hours the filters were transferred to a TBA-agar and incubated at 44±0.5 °C for 19–20 hours. An Indol-test was then performed by soaking a pad with Kovacz-reagent and placing the filter on top. Pink colonies were counted as *E. coli*. Selective cultivation for determination of intestinal Enterococci followed EN ISO 7899-2 (DEV, 2007). Filters were first incubated on Slanetz-Bartley-Agar at 36±0.5 °C for 40–48 hours and then on bile-esculin-agar at 44±0.5 °C for two hours. Colonies with a characteristic brownish-black colour that had appeared red on Slanetz-Bartley agar were counted as intestinal Enterococci.

Colony forming units per 100 ml were calculated from counted colonies, sample volume and dilution. All zero counts were replaced by the lowest possible count. Geometric means were calculated for each phase and sampling point. Bacteria concentrations were then $\log_{10}$-transformed and checked for normal distribution using the Kolmogoroff-Smirnov test in SPSS. The software SPSS was further used to calculate arithmetic mean (μ), standard deviation (σ), standard error, 95%-confidence intervals and generate box-plots. Bacteria removal in log-units was calculated from these mean values. 90^{th}-percentile values were calculated as described in the EU Directive (EU, 2006):

$$90^{th}\text{-percentile} = \text{Antilog}\,(\mu+1.282\sigma) \qquad \text{Equation 2-8}$$

SPSS was further used to compare groups of data using one-way ANOVA tests. For multiple comparisons, post-hoc tests according to Scheffe or Tamhane were further employed. The software SigmaPlot was used to plot log-removal, fit hyperbola functions and perform a regression analysis.

2.2.3 Analysis of physical, chemical and sum parameters

Total suspended solids (TSS) were determined according to Standard Method 2540 D (AWWA, 2005). Glass microfiber filters (935-PAH, WHATMAN SCHLEICHER & SCHUELL, Dassel, Germany) were weighed and then well-mixed samples of 500–1000 ml were filtered through in a vacuum filtration apparatus. Filters were dried at 105 °C for 2 hours, cooled in an exsiccator, weighed again and the weight difference per sample volume was calculated.

Turbidity was measured with a Hach Turbidimeter 2100 AN (HACH LANGE, Düsseldorf, Germany). Samples were well agitated to avoid settling of larger particles and the highest value was read. Turbidity was measured a minimum of two times and if results deviated by more than 10% a third time.

For electrochemical analysis (pH, T, electrical conductivity, redox potential, dissolved oxygen), a Multi 340i (WTW, Weilheim, Germany) was used that had been regularly calibrated and checked with standards. Samples were mixed by a magnetic stirrer. Dissolved oxygen in the filter horizons of the laboratory SSF was measured with an electrode in a 10 ml flow-through cell as well as with a fiber-optic oxygen meter (Fibox 3, PRESENS, Regensburg, Germany).

For the measurement of biochemical oxygen demand (BOD_5), an OxiTop (WTW, Weilheim, Germany) was used. An appropriate sample volume was filled into a bottle, allyl-thiocarbamide added to suppress nitrification and the bottle tightly closed with a lid containing a pressure sensor. The sample was mixed by a magnetic stirrer and kept at 20 °C. Carbon dioxide produced was removed from the gas phase by two NaOH-tablets placed in a tubular in the bottle neck. Oxygen consumption was measured from the pressure decline over 5 days and the BOD_5 calculated. Chemical oxygen demand (COD) was determined by cuvette tests (HACH LANGE, Düsseldorf, Germany). The test LCK 314 was used for a COD of 5–60 mg/l, the test LCK 414 for the range of 15–150 mg/l COD and the test LCK 514 for the range of 100–2,000 mg/l COD. A sample of 2 ml was filled into the cuvette containing the supplied reagents and heated to 148 °C for 2 hours. After cooling down to room temperature the cuvette was inserted into a photometer (Cadas 100 / Lpg 210, HACH LANGE, Düsseldorf, Germany), the appropriate program was chosen and the COD measured photometrically.

TOC was determined using a Shimadzu TOC-5050 analyzer. Ion chromatography was performed with the chromatograph DX-500 (DIONEX, Sunnyvale, USA) for anions (nitrate, nitrite, phosphate, sulphate). Columns used were AG4A-SC and AS4A-SC Analytical (4 × 250 mm, DIONEX, Sunnyvale, USA). Eluent was 1.8 mM Na_2CO_3 and 1.7 mM $NaHCO_3$

with a flow rate of 2 ml/min. Cations (ammonium) were determined with an ion chromatograph DX-120 (DIONEX, Sunnyvale, USA) with the columns CG12 and CS12A (4 x 250 mm, IONPAC, Sunnyvale, USA). Eluent was 0.2 M methanesulfonic acid with a flow rate of 1 ml/min.

For all these parameters, arithmetic mean values were calculated for influent and effluent for each phase of the experiments. Removals were then calculated from arithmetic mean values.

2.3 Model of slow sand filtration

The simulation tool AQUASIM (Reichert, 1994) provides models for several aquatic systems and/or reactor compartments. Compartments, links between compartments, processes and variables were specified in order to simulate slow sand filtration in the experimental filters (Figure 2-4).

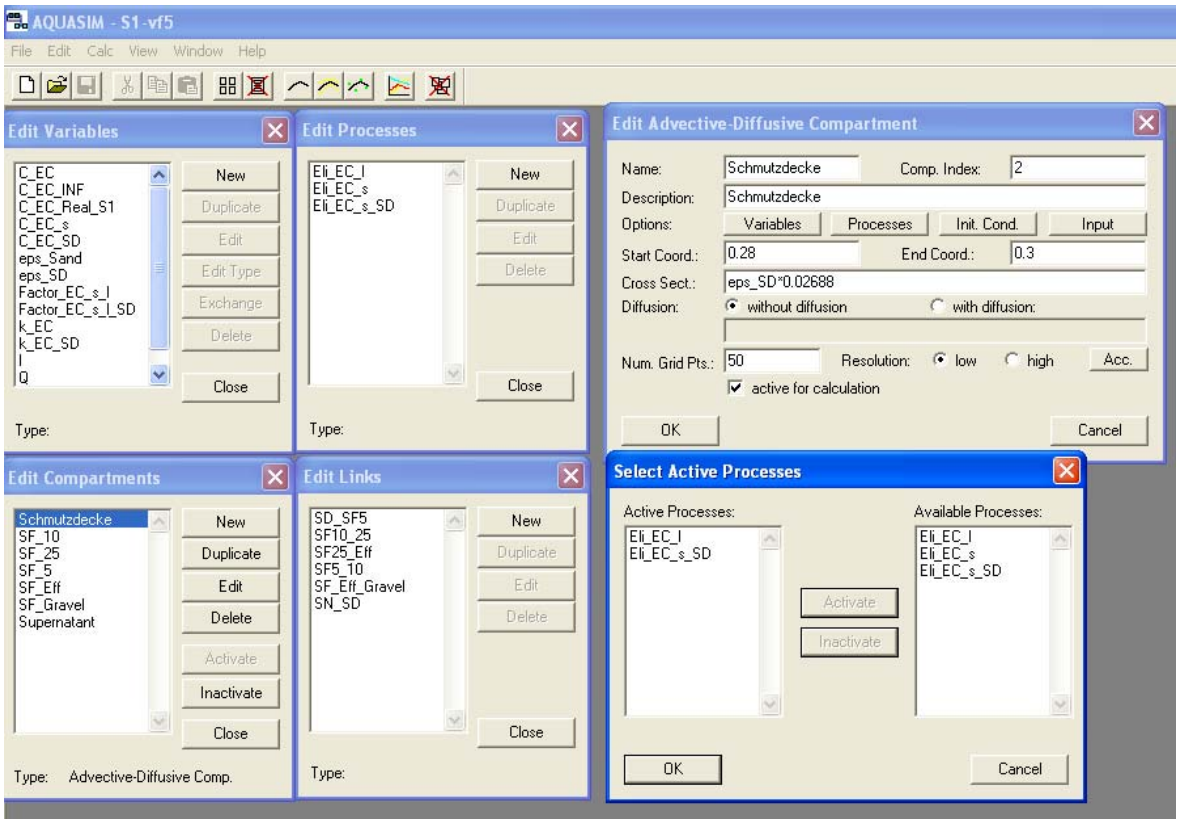


Figure 2-4: AQUASIM-window for editing variables, processes, compartments and links

The model was based on the following insights, simplifications and assumptions:

1. Previous experiments had shown that fecal indicator bacteria removal in slow sand filtration of secondary clarifier effluent mainly depended both on the sand surface area and the schmutzdecke.

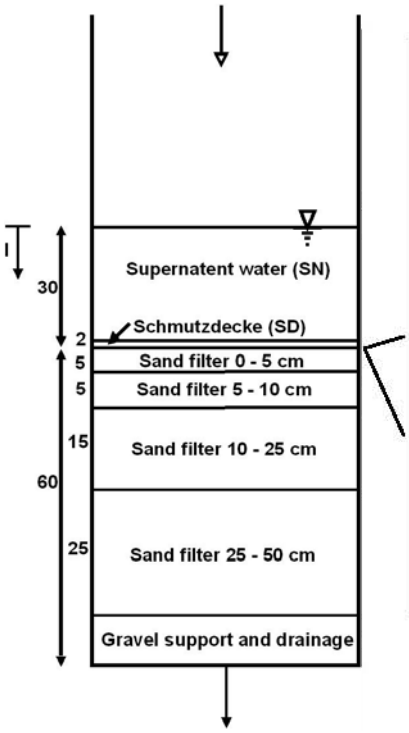
2. Within the slow sand filter transport mechanisms and active movement of bacteria lead to contact of bacteria with sand grains, biofilm growing on sand grains or the schmutzdecke. Retention is then due to adsorption and straining (Huisman and Wood, 1974).

3. The concentration of these immobilized bacteria is a function of their concentration in the surrounding bulk phase as well as the sand bed depth and the specific sand surface area. The relationship between retained and mobile bacteria in each filter horizon was calculated from the concentration measured in the bulk phase and in shake-off suspensions of extracted sand samples.

4. Elimination of both retained and mobile indicator bacteria follows a first-order reaction. The reaction rate constant is independent of filter length and encompasses biotic and abiotic processes such as predation, lysis and die-off due to a challenging environment. It was estimated from experimental determination within samples taken from the secondary clarifier.

The filter was divided into the compartments supernatant water, schmutzdecke, sand bed and gravel layer. The sand bed was further divided into four compartments, each confined by the location of the sampling ports (e.g. reaching from 5 to 10 cm filter bed depth). Because of the sampling process for determination of bacteria retained in the schmutzdecke, the schmutzdecke was defined in AQUASIM as a compartment of 2 cm height (see Factor_EC_SD later in this section). The compartment supernatant water was depicted with a height of 30 cm. Rising supernatant levels were neglected because of the relatively small impact of this compartment on bacteria removal. All compartments were defined as plug-flow reactors (advective diffusive reactor compartment) without dispersion and neighboring compartments were linked with each other (Figure 2-5).
The processes of bacteria inactivation and elimination were described as a first-order reaction. They were defined to apply to both the bacteria in the water or bulk phase and the bacteria immobilized after retention on the sand grains, the biofilm or the schmutzdecke.

In the supernatant water, only the process in the bulk phase was considered and in the gravel compartment no process was considered. The compartments, variables and processes are depicted in Figure 2-5 and explained below.



Variable	Process
Q C_EC_Inf	
	$\frac{dC\_EC}{dt} = -k\_EC \cdot C\_EC$
eps_SD	$\frac{dC\_EC}{dt} = -k\_EC \cdot C\_EC \cdot (1 + Factor\_EC\_SD)$
eps_Sand sand_surf	$\frac{dC\_EC}{dt} = -k\_EC \cdot C\_EC \cdot (1 + Factor\_EC \cdot sand\_surf)$

Figure 2-5: Compartments, variables and processes in the AQUASIM-model of slow sand filtration (dimensions in cm)

- **C_EC**: Concentration of *E. coli* in the bulk phase (dynamic state variable; CFU/100 ml water)
- **C_EC_Inf**: Concentration of *E. coli* in the filter influent (constant variable; CFU/100 ml water)
- **C_EC_Real_Sx**: Mean values of measured *E. coli* concentrations in the filter horizons (real list variable; CFU/100 ml water)
- **C_EC_s** = C_EC × Factor_EC × sand_surf: Concentration of *E. coli* retained within a sand volume of 100 ml pore volume (formula variable; CFU/100 ml water)
- **C_EC_SD** = C_EC×Factor_EC_SD: Concentration of *E. coli* within the schmutzdecke (formula variable; CFU/100 ml water)
- **eps_Sand**: Porosity was rounded to 40% (constant variable)

- **eps_SD**: for the porosity of the schmutzdecke a value of 0.8±0.1 was used (constant variable; active in sensitivity and uncertainty analysis)

- **Factor_EC**: Retention factor that describes the equilibrium between retained and mobile bacteria related to the specific sand surface area and as a function of filter bed depth (real list variable; m^3/m^2; active in sensitivity and uncertainty analysis). All mechanisms of transport, straining and adsorption that lead to retention are summarized herein. The values were determined from bacteria concentrations in the bulk phase and those retained on sand and biofilm in the filter horizons of 5 cm, 10 cm and 25 cm depth. Shake-off suspensions had been prepared from approximately 2.5 g of sand sampled from the filter horizons of S1 and S4 in phase II and phase III. Samples were added to 20 ml of phosphate-buffer-solution in centrifuge tubes of 50 ml and vortexed for 2 minutes. Then the sand was allowed to settle for half a minute and the supernatant water was transferred for membrane filtration of appropriate dilutions. For phase I sand samples had been taken from laboratory columns with a d_{10} of 0.25 mm, 0.4 mm and 0.63 mm that had been operated with secondary clarifier effluent of the same WWTP and were well comparable to the pilot scale filters. All samples were analyzed in triplicates.

- **Factor_EC_SD**: Retention factor that describes the equilibrium between retained and mobile bacteria in the schmutzdecke (real list variable; active in sensitivity and uncertainty analysis). All mechanisms of transport, straining and adsorption that lead to retention in the schmutzdecke are summarized herein. The values were determined from bacteria concentrations in the bulk phase directly above the schmutzdecke and the schmutzdecke itself. For the latter, samples were taken after draining the supernatant water so that only a layer of 2 cm remained above the sand bed and then mixing this layer. In phases II and III samples from columns S1 and S4 were analyzed in triplicates. For phase I, the sample was taken from a laboratory column with a d_{10} of 0.8 mm operated with secondary effluent from the WWTP under closely comparable operating conditions. Samples were homogenized with an Ultraturrax (IKA, Staufen, Germany).

- **k_EC**: reaction rate coefficient of *E. coli* elimination/inactivation derived from experiments with secondary clarifier effluent from the WWTP in winter and summer (constant variable; active in sensitivity and uncertainty analysis; 1/d). 2 liter samples were kept at room temperature, protected from light and analyzed for decline of

indicator bacteria concentrations. Samples were taken daily during a period of three days and analyzed in triplicates. Logarithmic concentrations were plotted over time and a linear regression analysis was performed in Excel.

- **l**: filter length measured from the regular supernatant level (program variable; m)
- **Q**: arithmetic mean of the volume flow measured for each filter column (constant variable; m^3/d)
- **sand_surf**: specific sand surface area (constant variable; m^2/m^3)

Since parameters like the reaction rate coefficient were subject to variability as expressed by the standard deviation, AQUASIM was also used to perform an uncertainty analysis (Reichert, 1998).

3 RESULTS AND DISCUSSION

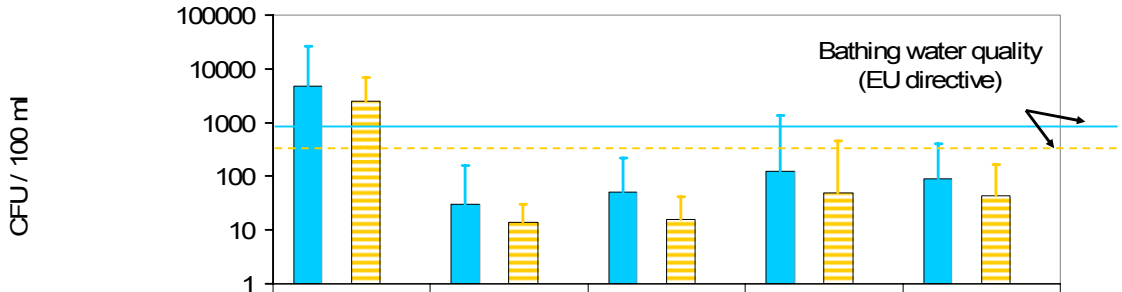
3.1 Experiments at laboratory scale

Laboratory scale experiments were performed and evaluated to prepare experiments at pilot scale. Different removals of fecal indicator bacteria *E. coli* and intestinal Enterococci in slow sand filters of variable sand grain size were observed. Table 3-1 shows fecal indicator bacteria concentrations in the influent and effluent of the 4 filters.

Table 3-1**:** Removal of indicator bacteria in laboratory scale slow sand filters of varying sand grain size at a target hydraulic loading rate of 5 cm/h

Filter column		C1	C2	C3	C4
d_{10} [mm]		0.25	0.4	0.63	0.8
E. coli					
90^{th}-percentile	Influent		26,700		
[CFU/100 ml]	Effluent	159	221	1,360	402
Mean	Influent		4,840		
[CFU/100 ml]	Effluent	30	52	126	91
Removal [log_{10}-units]		2.20	1.97	1.58	1.73
Intestinal Enterococci					
90^{th}-percentile	Influent		6,870		
[CFU/100 ml]	Effluent	30	41	447	166
Mean	Influent		2,550		
[CFU/100 ml]	Effluent	14	16	49	43
Removal [log_{10}-units]		2.26	2.21	1.71	1.78

E. coli removal amounted to 1.6–2.2 log-units or 97.4–99.4%. Intestinal Enterococci removal was comparable ranging from 1.7–2.3 log-units or 98.1–99.5%. The filter effluents reached bathing water quality (EU, 2006) and complied with monitoring levels of <1,000 colony forming units (CFU) *E. coli* per 100 ml (WHO, 2006) except for the filter column 3 with a d_{10} of 0.63 mm (Figure 3-1).



	Influent	C1 d10 = 0.25	C2 d10 = 0.4	C3 d10 = 0.63	C4 d10 = 0.8
E. coli (mean)	4837	30	52	126	91
E. coli (90th-percentile)	26682	159	221	1357	402
Enterococci (mean)	2550	14	16	49	43
Enterococci (90th-percentile)	6870	30	41	447	166

Figure 3-1: Geometric mean and 90th-percentile concentrations of *E. coli* and intestinal Enterococci in influent and effluents of laboratory scale slow sand filters of varying sand grain size d_{10} [mm] at a hydraulic loading rate of 5 cm/h

Highest bacteria removal corresponded to the finest sand (d_{10} = 0.25 mm). Both indicator bacteria showed the similar tendency that increasing sand grain size resulted in lower bacteria removal. Only column 3 with a d_{10} of 0.63 mm did not follow this trend. The profile of bacteria concentration over filter length (data not shown) revealed a lower bacteria removal in the schmutzdecke of column 3 compared with the other filters. Total coliform bacteria (TC) removal only ranged between 7–95%. The effectiveness decreased substantially during the ripening phase of the filters.

In order to identify the most efficient zones, bacteria removal within the filter compartments supernatant, schmutzdecke and filter bed was quantified. The resulting profiles of bacteria concentration as a function of filter length are shown in Figure 3-2 and Figure 3-3. Bacteria removal per filter length was greatest in the schmutzdecke and upper 5 cm of the sand bed. Removal rates were lower in the deeper zones of the sand bed dropping below the rates determined for the supernatant. The profiles for *E. coli* and intestinal Enterococci showed a high degree of similarity.

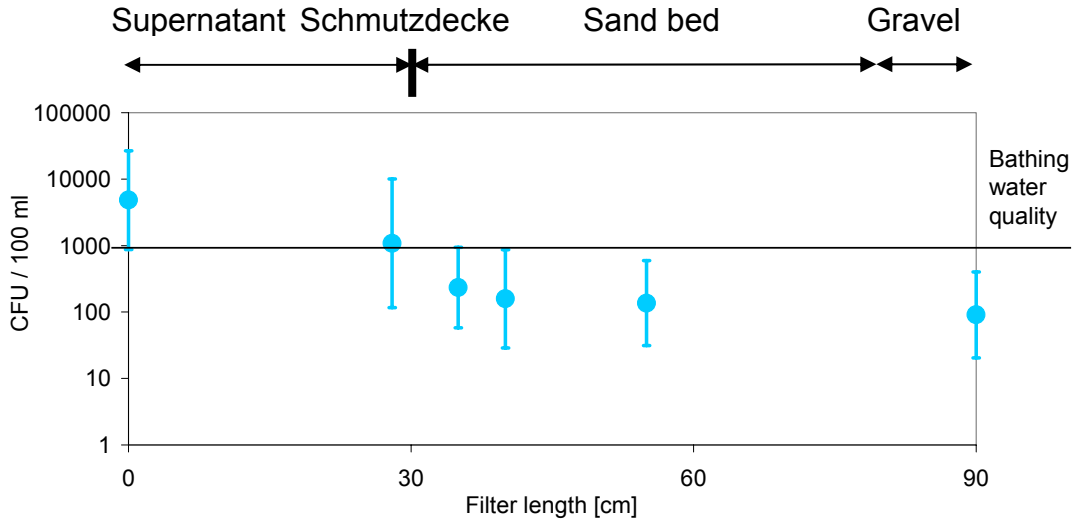


Figure 3-2**:** *E. coli* concentration as a function of filter length measured from regular supernatant water level with bars depicting the range between 10th- and 90th-percentile (laboratory scale slow sand filter C4, d_{10} = 0.8 mm, HLR = 5 cm/h)

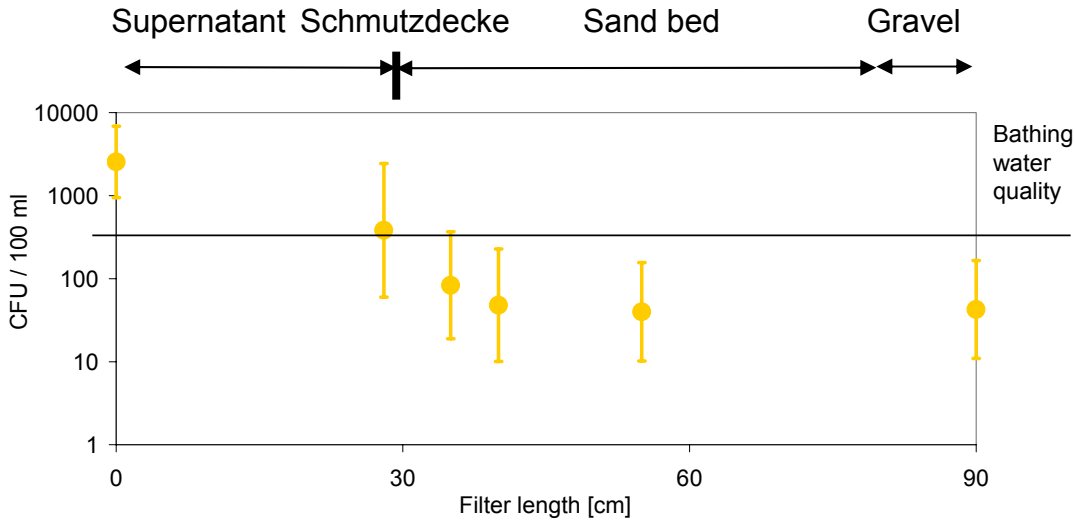


Figure 3-3**:** Intestinal Enterococci concentration as a function of filter length measured from regular supernatant water level with bars depicting the range between 10th- and 90th-percentile (laboratory scale slow sand filter C4, d_{10} = 0.8 mm, HLR = 5 cm/h)

Suspended solids removal amounted to 94–95%. The filter effluents reached average TSS concentrations of 0.5 mg/l. Sand grain size did not significantly affect solids removal. Turbidity removal was 59–77% with 0.9–1.5 NTU in the filter effluents. Physical, chemical

and sum parameters of the filter influent and effluent are listed in Table 3-2. Aerobic conditions prevailed throughout the bulk phase of the filter.

Table 3-2**:** Selected characteristics of laboratory scale slow sand filter influent and effluents

	Influent	Effluent			
Filter column		C1	C2	C3	C4
d_{10} [mm]		0.25	0.4	0.63	0.8
TSS [mg/l]	8.2 ± 2.7	0.4 ± 0.3	0.5 ± 0.2	0.4 ± 0.3	0.4 ± 0.1
TSS removal [%]		94	94	94	95
Turbidity [NTU]	3.6 ± 4.5	1.3 ± 0.8	0.9 ± 0.3	1.0 ± 0.4	1.5 ± 0.9
Turbidity removal [%]		64	77	73	59
Dissolved oxygen [mg/l]	10.8 ± 0.7	5.2 ± 2.0	5.0 ± 1.3	5.1 ± 1.2	5.2 ± 1.6
BOD_5 [mg O_2/l]	2.0 ± 3.2	1.9 ± 1.2	2.0 ± 1.4	1.8 ± 1.4	1.3 ± 1.6
COD [mg O_2/l]	35.8 ± 3.5	31.5 ± 2.7	30.7 ± 2.4	32.9 ± 4.3	33.7 ± 5.0
TOC [mg/l]	10.3 ± 2.1	9.3 ± 1.9	8.3 ± 1.5	9.5 ± 1.7	9.9 ± 2.8
pH	7.5 ± 0.2	7.1 ± 0.2	7.2 ± 0.1	7.2 ± 0.2	7.2 ± 0.1
Redox-potential [mV]	272 ± 23	258 ± 38	259 ± 27	259 ± 24	261 ± 26
Conductivity [mS/cm]	1,380 ± 67	1,360 ± 83	1,370 ± 87	1,350 ± 69	1,383 ± 79
Temperature [°C]	4 ± 1	21 ± 2	21 ± 2	21 ± 2	21 ± 2

The schmutzdecke developed differently on fine and coarse sand: for a d_{10} ≥0.4 mm it penetrated deeper into the upper sand layer. The maximum supernatant level of 1.5 m was only reached in the column with the finest sand, where the developing schmutzdecke had led to an increase in filter resistance. The regular supernatant level was restored after removing the schmutzdecke by wet-harrowing. The most effective removal was thus connected with the shortest runtime of close to 6 months.

The slow sand filters eliminated 1.6–2.2 log-units of *E. coli* and intestinal Enterococci from secondary clarifier effluent at a hydraulic loading rate of 1.2 m/d. For Enterococci this is similar to the 0.7–2.6 log-units removal observed by Mälzer (2005) and a little lower than the

E. coli removal of 2.3–3.7 log-units he reported. Lower removal was probably due to the fact that he used a deeper sand bed of 85 cm compared to 50 cm depth in our experiments.

Bacteria removal was within the range of 0.3–3.5 log-units reported for total coliform removal (Ellis, 1987; Farooq and Alyousef, 1993; Adin et al., 1998; Sadiq et al., 2003; Mälzer, 2005). Highest bacteria removal was observed for the finest sand. Farooq and Alyousef (1993) reported higher bacteria removal for fine sand compared to coarse sand whereas Ellis (1987) did not find substantial differences. Sadiq et al. (2003) did not observe a significant influence of d_{10} on TC removal but established a regression equation that inversly correlated removal with d_{10}. However, all authors only compared two types of sand: a fine type with a d_{10} of 0.3–0.31 mm with a coarse type in the range of 0.5–0.6 mm. Comparing the performance of filters filled with finer and coarser sand might have resulted in significant differences in bacteria removal. In addition, comparison is difficult since total coliform bacteria do not appear to be a suitable indicator for monitoring disinfection in slow sand filters. In our experiments, TC removal varied strongly and effluent concentrations were higher than in the influent on several occasions. Some coliforms are autochthonous to soil and water and may multiply in slow sand filters (Adin et al., 1998; Petry-Hansen, 2005).

Slow sand filters can provide an option for a tertiary treatment step for disinfection of wastewater. Depending on raw water quality, the filter effluent may be used in agriculture or may be discharged into water bodies used for recreation. In the experiment, all filter effluents fulfilled the requirements of the EU Directive for bathing water quality and complied with monitoring levels of <1,000 CFU *E. coli* per 100 ml with the exception of column 3.

In the schmutzdecke and the upper 5 cm of the sand bed, high bacteria removal was observed. For sand bed zones deeper than 25 cm, bacteria removal per filter length was even lower than in the supernatant. The schmutzdecke was the most efficient zone of bacteria removal but also led to clogging of one filter. While this clogging is typical for slow sand filters, long runtimes are desired in between maintenance. Wet-harrowing or scraping of the schmutzdecke restores hydraulic conductivity but removes the most efficient filter layer (Sánchez, 2006). The filter sand can to some extent be regarded as the support layer for the schmutzdecke. The fact, that filter performance seems to be strongly influenced by the schmutzdecke and upper 5 cm sand layer, was also observed by other authors (Adin, 2003). Filter bed depths below 25 cm had lower removal efficiencies than the supernatant. Therefore, simply increasing the filter depth does not seem to be the most efficient solution for optimizing the overall bacteria removal in such filters.

The slow sand filters achieved bacteria removal similar to the 1.5–2 log-units reported for vertical flow constructed wetlands (Hagendorf, 2002). However, slow sand filters can be operated at hydraulic loading rates higher by a factor of 10 and more and thus only require a fraction of the land area.

In later experiments bacteria removal and clogging behaviour at higher hydraulic loading rates, BOD_5, ammonia and suspended solids concentrations in the influent were evaluated to improve the knowledge on applicability of slow sand filtration for disinfection of secondary effluents. The sands studied at laboratory scale were chosen to be very uniform (U = 1.6). In order to increase the choice of material suitable for wastewater disinfection, sands of higher uniformity coefficients were studied at pilot scale. The larger column diameter and thus volume at pilot scale was expected to be beneficial, because samples from a certain filter horizon could be taken more precisely and with less disturbance of the flow patterns. In addition the treated wastewater was not stored for up to one week but applied directly.

3.2 Experiments at pilot scale

3.2.1 Removal of fecal indicator bacteria

Slow sand filters eliminated 1.9–2.6 log_{10}-units or 98.6–99.8% of *E. coli* and 1.9–3.0 log_{10}-units or 98.9–99.9% of intestinal Enterococci from secondary clarifier effluent (Table 3-3).

Table 3-3: Removal of indicator bacteria in pilot scale slow sand filters S1–S6 of varying sand grain size at three hydraulic loading rates

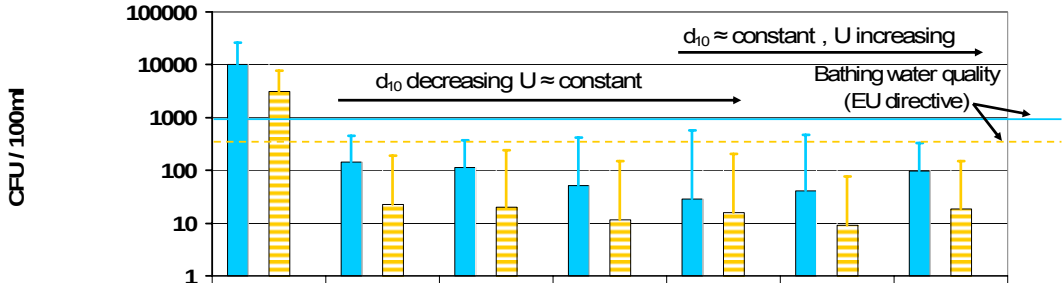
		Phase	Target HLR	S4	S3	S2	S1	S5	S6
Sand properties	d_{10}			0.82	0.67	0.38	0.25	0.26	0.23
	U			1.51	1.36	1.63	1.56	2.92	4.91
Log_{10}-removal	*E. coli*	I	5 cm/h	1.85	1.95	2.30	2.55	2.49	2.01
		II	10 cm/h	1.85	1.91	2.34	2.47	2.28	2.21
		III	20 cm/h	1.95	2.16	2.51	2.37	2.60	2.60
	Intestinal Enterococci	I	5 cm/h	2.14	2.21	2.44	2.29	2.54	2.24
		II	10 cm/h	1.94	2.29	2.60	2.59	2.35	2.35
		III	20 cm/h	2.07	2.27	2.22	2.39	2.41	3.01

Figure 3-4a depicts that for the average inflow concentration of 10,082 *E. coli* per 100 ml in phase I, all but two slow sand filter effluents complied with a monitoring level of <100 *E. coli* per 100 ml (based on a geometric mean). These two filters used sand of a d_{10} higher than the typical maximum recommendations of 0.4 mm in water purification. In phases II and III all filter effluents complied with a monitoring level of <100 *E. coli* per 100 ml (Figure 3-4b, c). Sufficient bathing water quality according to the EU Directive was achieved by all sand filter configurations under all hydraulic loading rates from 5 to 20 cm/h. Unlike the secondary effluent, the 90^{th}-percentile values of all filter effluents during all phases were below 900 CFU per 100 ml for *E. coli* and below 330 CFU per 100 ml for intestinal Enterococci.

Figure 3-4 also shows the tendency that a decrease in d_{10}, meaning a finer sand, led to lower *E. coli* concentrations in the effluent and thus improved removal. An increase in uniformity coefficient, corresponding to more heterogenous sand, resulted in a decline of *E. coli* removal or higher effluent concentrations with the exception of phase III. So a tendency was observed that the finer and more homogenous a filter sand, the higher was *E. coli* removal. The same tendency holds for intestinal Enterococci.

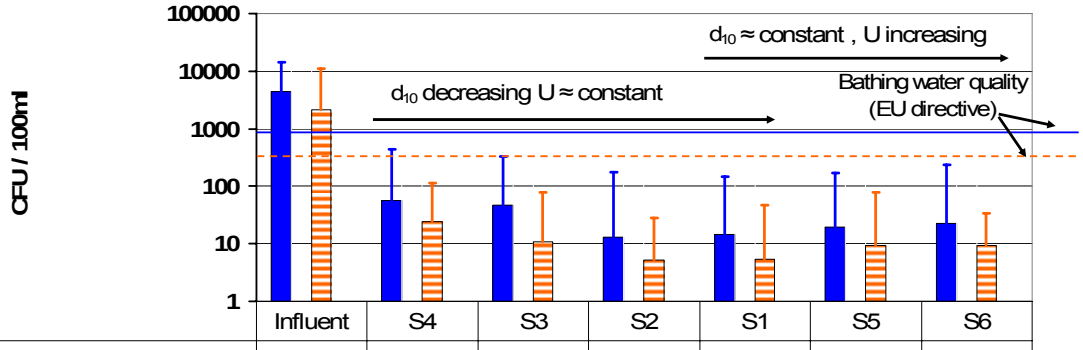
Significant differences were found between influent and effluent concentrations of each filter during phase I, II and III ($p < 0.01$) indicating that slow sand filtration significantly removed indicator bacteria. However, in between effluent concentrations of the 6 filters no significant differences ($p > 0.05$) were found during any phase (Figure 3-5 and Figure 3-6). The effluents of S2 and S5 during phase I as well as S4 during phase II were omitted from this comparison, because normal distribution had been rejected by the Kolmogoroff-Smirnov-test. Lack of significant differences was due to the fact that the differences between the filters were hidden by high variations in the influent concentration that were passed on through the columns. This is reflected by the wide range of the confidence intervals (Figure 3-5 and Figure 3-6). These variations are typical for secondary effluent and caused by dilution during rain among other factors. For intestinal Enterococci the observations were similar. No comparisons were made between different phases, because besides the HLR changed deliberately, variations in several ambient parameters were impossible to suppress.

a) Phase I: HLR = 5 cm/h



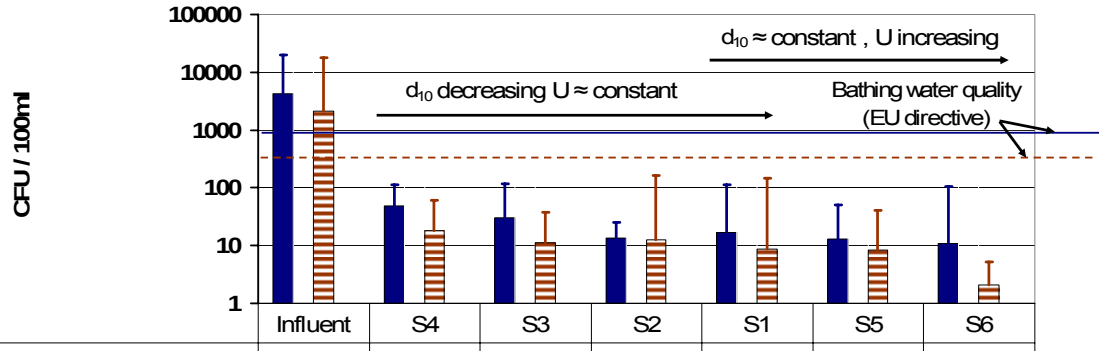
	Influent	S4 d10=0.82 / U=1.51	S3 d10=0.67 / U=1.36	S2 d10=0.38 / U=1.63	S1 d10=0.25 / U=1.56	S5 d10=0.26 / U=2.92	S6 d10=0.23 / U=4.91
E. coli Mean	10082	142	114	51	29	41	99
E. coli 90th-Percentile	26616	449	374	414	573	453	335
Enterococci Mean	3162	23	20	11	16	9	18
Enterococci 90th-Percentile	7806	186	236	148	206	76	152

b) Phase II: HLR = 10 cm/h



	Influent	S4	S3	S2	S1	S5	S6
E. coli Mean	4380	55	48	13	15	19	22
E. coli 90th-Percentile	14307	447	328	182	145	172	242
Enterococci Mean	2091	24	11	5	5	9	9
Enterococci 90th-Percentile	10990	115	78	28	47	79	34

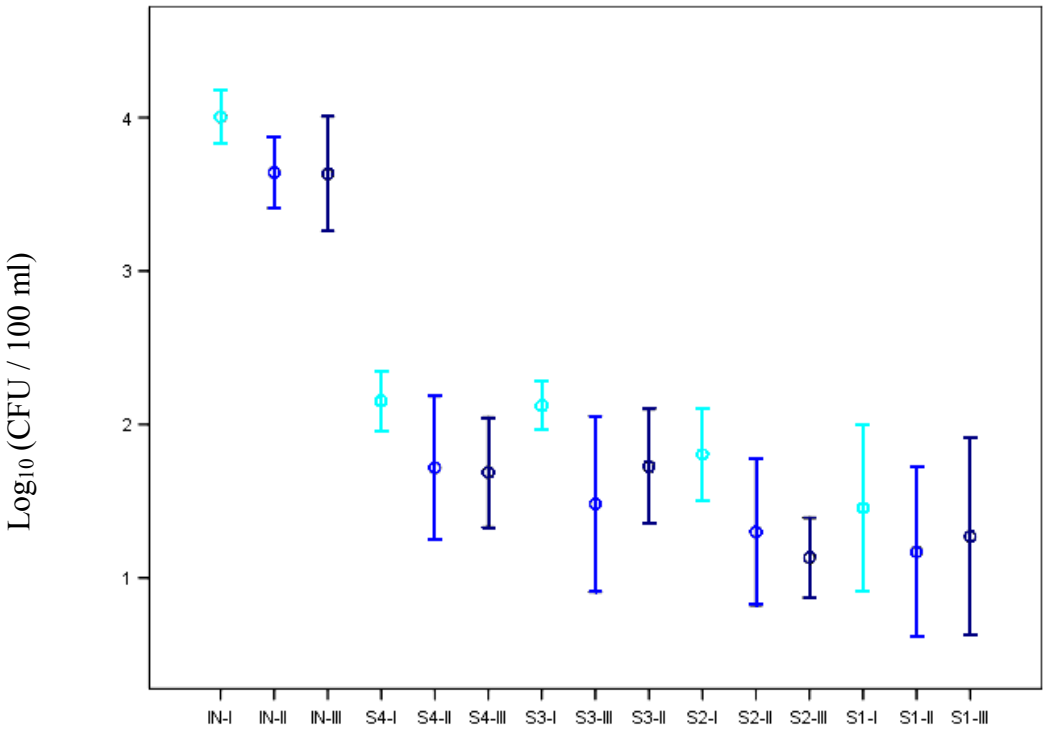
c) Phase III: HLR = 20 cm/h



	Influent	S4	S3	S2	S1	S5	S6
E. coli Mean	4323	49	30	14	17	13	11
E. coli 90th-Percentile	20133	114	118	25	113	49	104
Enterococci Mean	2114	18	11	13	9	8	2
Enterococci 90th-Percentile	17531	61	37	162	147	40	5

Figure 3-4: Geometric mean and 90th-percentile concentrations of *E. coli* and intestinal Enterococci in influent and effluents of pilot scale slow sand filters of varying sand grain size for phase I-III of increasing hydraulic loading rate

a)



b)

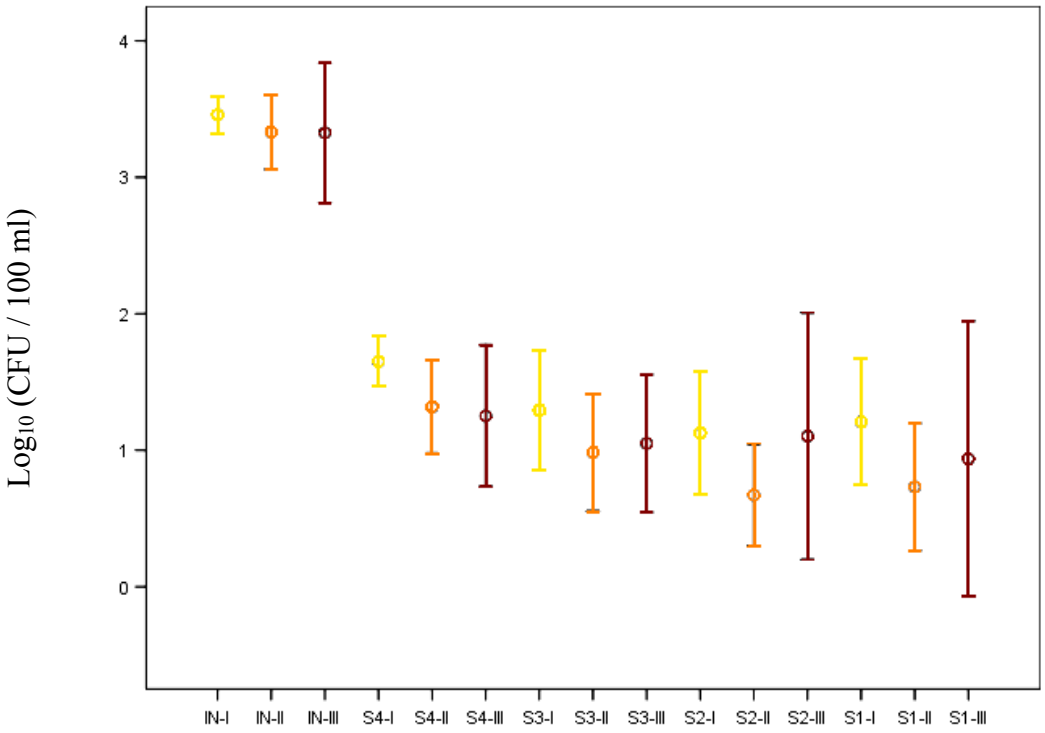
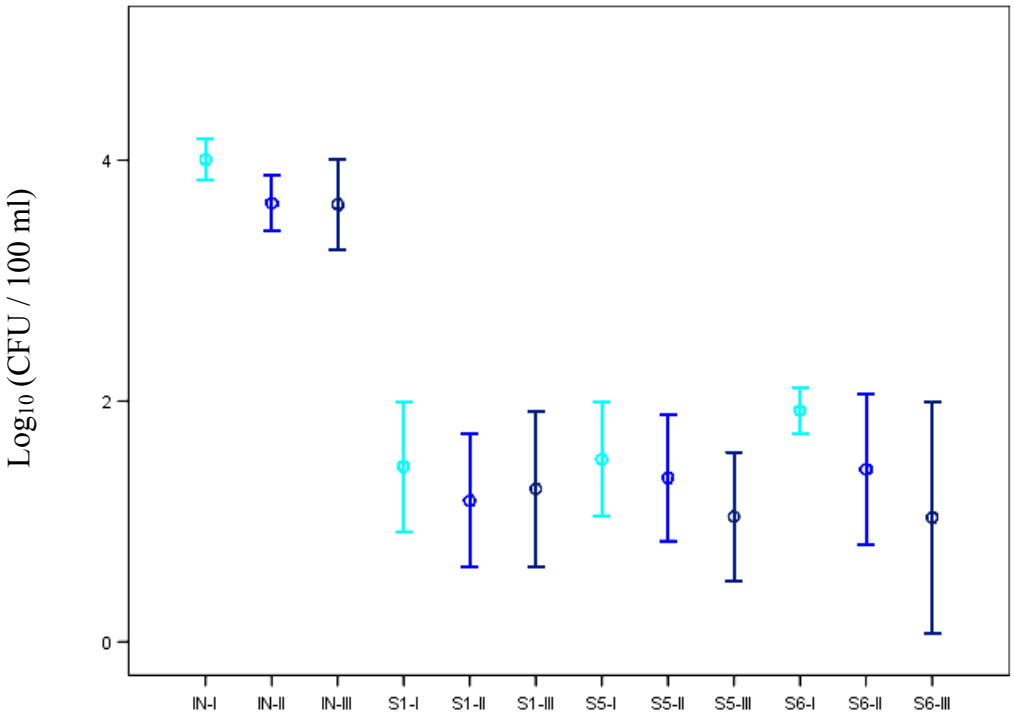


Figure 3-5: Arithmetic mean and 95%-confidence intervals of a) log_{10}-transformed concentrations of *E. coli* and b) intestinal Enterococci for influent (IN) and effluent of the pilot scale filters of decreasing d_{10} at constant U (S4–S1) during phases of increasing hydraulic loading rate (I: HLR = 5 cm/h; II = 10 cm/h; III = 20 cm/h)

a)



b)

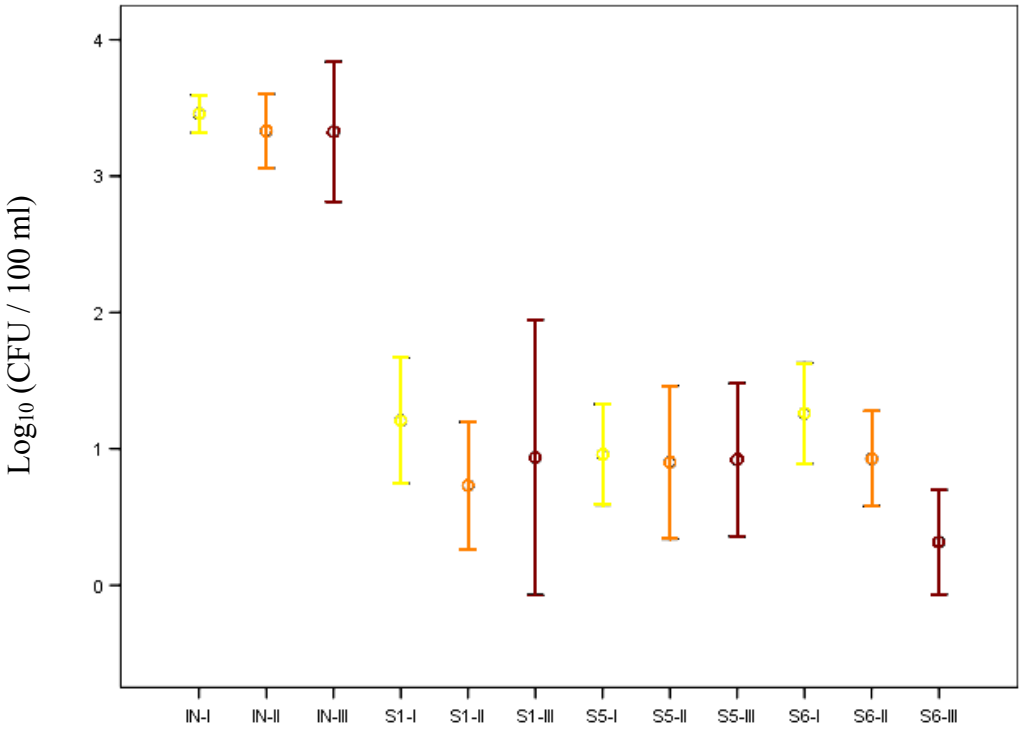


Figure 3-6: Arithmetic mean and 95%-confidence intervals of log_{10}-transformed concentrations of a) *E. coli* and b) intestinal Enterococci for influent (IN) and effluent of the pilot scale filters of increasing U at constant d_{10} (S1, S5, S6) during phases of increasing hydraulic loading rate (I: HLR = 5 cm/h; II = 10 cm/h; III = 20 cm/h)

The impact of grain size on bacteria removal can still be revealed considering mean concentrations. A strong correlation was found between indicator bacteria removal and the cumulated sand surface area that the water had percolated through. The cumulated surface area was calculated from the sand's specific surface area and the sand volume from the top of the filter bed to the sampling point (Table 3-4).

Table 3-4: Surface area of the different filter sands used in pilot scale slow sand filters S1–S6

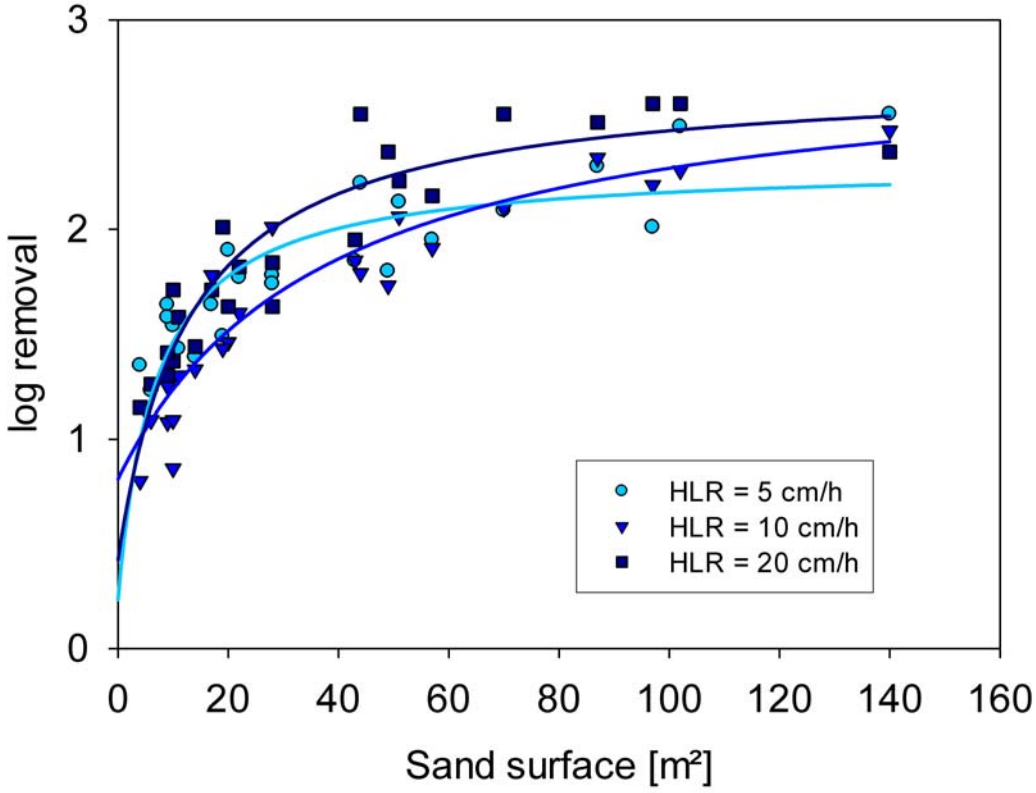
	Sand bed depth [cm]	S1	S2	S3	S4	S5	S6
d_{10} [mm]		0.25	0.38	0.67	0.82	0.26	0.23
U		1.56	1.63	1.36	1.51	2.92	4.91
Porosity		0.4	0.42	0.4	0.4	0.37	0.34
d_s [mm]		0.35	0.54	0.85	1.11	0.50	0.55
Specific sand surface area [m^2/m^3]		10,390	6,470	4,230	3,230	7,590	7,240
Cumulated sand surface [m^2]	5	14	9	6	4	10	10
	10	28	17	11	9	20	19
	25	70	44	28	22	51	49
	50	140	87	57	43	102	97

Figure 3-7 shows mean log-removal data for all sampling points. Removal of *E. coli* and Enterococci was similar. The data were fit using the hyperbola function:

$$y = y_o + \frac{ax}{b + x} \qquad \text{Equation 3-1}$$

Fecal indicator bacteria removal (y) was described as a function of sand surface area (x). High coefficients of correlation as given in Table 3-5 show that fecal indicator bacteria removal (y) can be described as a function of sand surface area (x) as well as schmutzdecke and supernatant water (y_0) for every hydraulic loading rate. They also indicate that bacteria removal in the pores was less important. This becomes especially evident when comparing S1 and S4. The sands nearly had the same porosity but bacteria removal was not identical. The steep slope of the hyperbola for small x-values reflects the high removal of bacteria in the relatively short zone of the upper centimeters of the sand bed. The hyperbola approaches an asymptote with increasing x-values: Removal of fecal indicator bacteria in SSFs is limited because biological activity declines with filter bed depth.

a)



b)

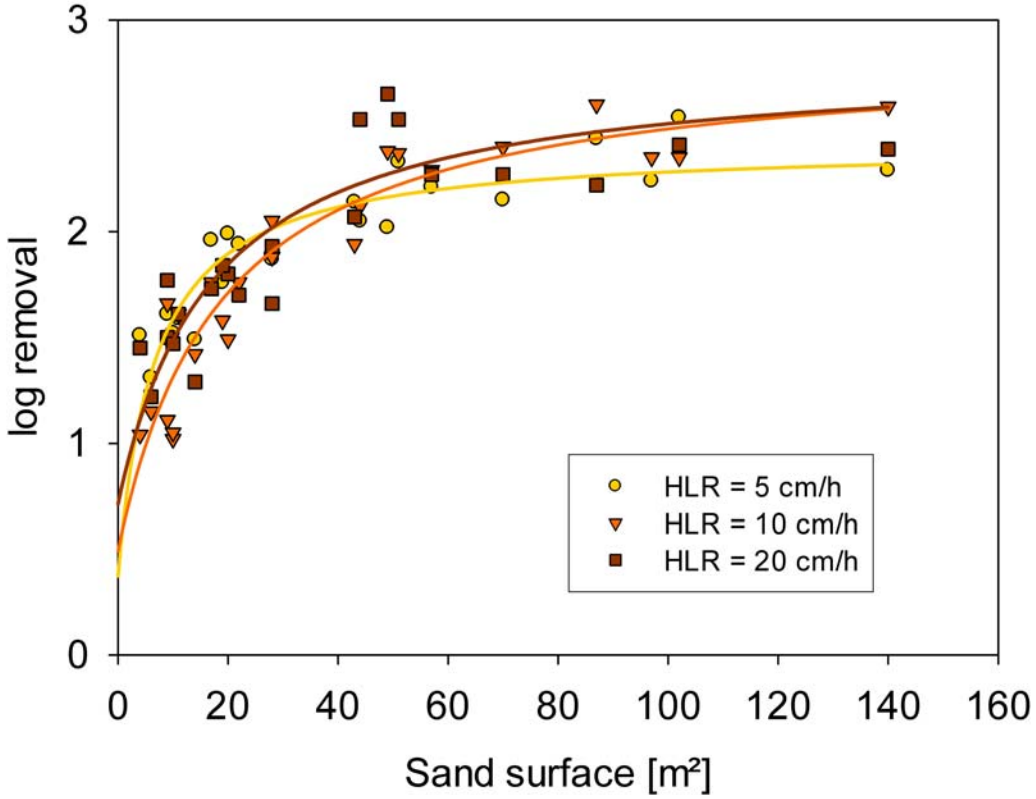


Figure 3-7: Log_{10}-removal of a) *E. coli* and b) intestinal Enterococci as a function of cumulated sand surface at three hydraulic loading rates in pilot scale slow sand filters S1–S6

Table 3-5. Parameters and coefficients of correlation (R^2) for fitted hyperbola functions of indicator bacteria removal in pilot scale slow sand filters operated at three hydraulic loading rates (see Equation 3-1)

	HLR		
	5 cm/h	10 cm/h	20 cm/h
E. coli	$R^2 = 0.93$	$R^2 = 0.83$	$R^2 = 0.95$
	$y_0 = 0.23$	$y_0 = 0.81$	$y_0 = 0.42$
	$a = 2.08$	$a = 2.04$	$a = 2.31$
Intestinal Enterococci	$R^2 = 0.96$	$R^2 = 0.94$	$R^2 = 0.90$
	$y_0 = 0.38$	$y_0 = 0.49$	$y_0 = 0.71$
	$a = 2.04$	$a = 2.37$	$a = 2.10$

With increasing sand surface area bacteria removal first increased sharply but then slowly approached a maximum average removal ($y_0 + a$) of 2.3–2.9 log-units for *E. coli* and 2.4–2.9 log-units for Enterococci. So an increase in bed depth or a finer sand is not expected to have an appreciable impact on filter performance beyond a certain point. The y_0-values indicate that the supernatant and the schmutzdecke are responsible for 10–28% of the maximum average removal of *E. coli* and 16–25% of Enterococci removal. Mälzer (2005) found 0.5 log-units or 22% of total *E. coli* removal in the schmutzdecke and the upper 5 cm of the SSF. Despite being a sand filter, a significant contribution to the total removal in the filter is independent of the sand. Because of its low height of less than 2 cm the schmutzdecke seems to be a highly efficient zone of bacteria removal. Sanchez (2006) reported a drop in fecal coliform removal of up to 3 log-units after scraping of the schmutzdecke. Within the sand bed, the upper centimeters are the zone of most substantial bacteria removal. This has been reported for many sand filters (Ellis, 1985; Stevik et al., 2004) and is evident from the steep slope of each hyperbola for small cumulated surface areas.

The impact of schmutzdecke and sand surface on bacteria removal can clearly be seen in the profiles of *E. coli* concentration over the length of filters S1 and S4 (Figure 3-8). These profiles were chosen, because S1 has the maximum and S4 the minimum sand surface area used in the experiments. It is evident that bacteria removal in the supernatant water, schmutzdecke and upper 5 cm of sand was very similar. This can consistently be explained by the fact that supernatant and schmutzdecke are independent of the underlying sand. After an initial phase, the schmutzdecke develops on top of the existing schmutzdecke and sand properties have no further influence. While only 0.22 and 0.11 log-units were removed in the supernatant, the schmutzdecke and upper 5 cm of sand were responsible for 1.17 and 1.24 log-units, corresponding to 52% and 67% of the total removal in the filters. This is

remarkable because this zone of 7 cm only accounts for 8% of total filter length. It clearly shows the importance of the schmutzdecke and upper centimeters of the sand bed and thereby indicates the importance of the biomass in the filter.

In the sand bed below 5 cm the influence of sand surface on bacteria removal can be seen. The water in filter S1 passes through a sand surface approximately three times the surface of the sand in S4. Therefore a removal of 1.16 log-units in the lower 45 cm of the sand bed in S1 compared to 0.5 log-units in S4 seems reasonable. It can also be noted that bacteria removal per filter length declined in deeper zones of the sand bed. This decline in efficiency was especially evident for the filter S4 with a comparatively small specific sand surface area. Over the lower 35 cm of the sand bed only 0.08 log-units were removed. Compared to 0.11 log-units removed in the supernatant water of 30 cm filter length, the efficiency of *E. coli* removal was lower in the depth of the sand bed than in the supernatant. This supports the findings in treatment of surface water that led to the recommendations of minimum sand bed depths of 50 cm, since no major biological activity was to be expected below 40 cm (see 1.2.1).

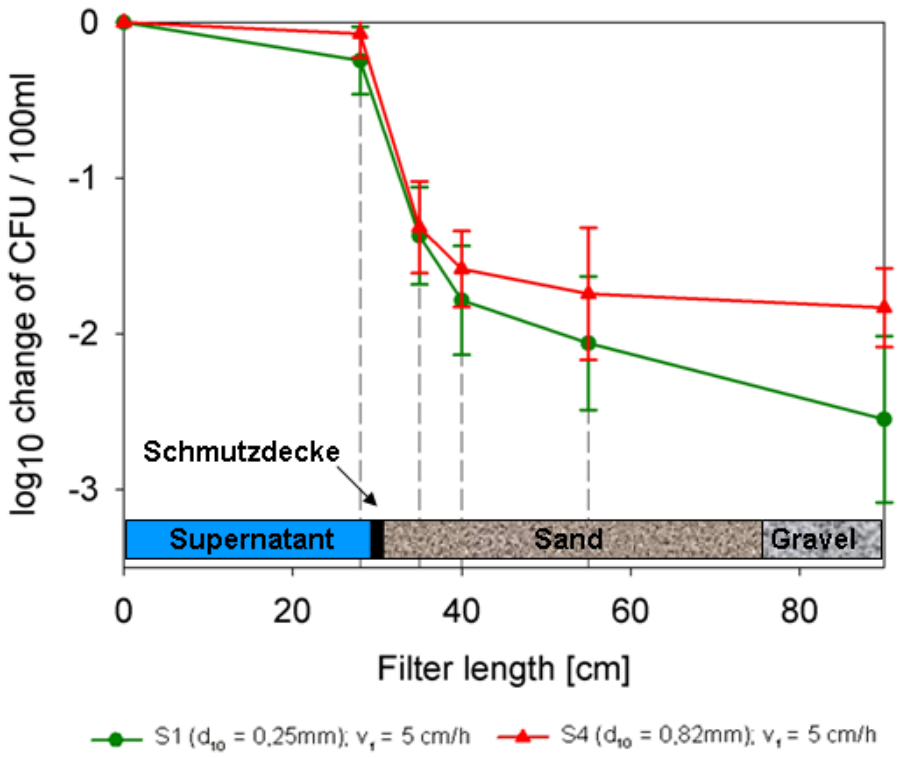


Figure 3-8: Profiles of *E. coli* concentration over the length of pilot scale slow sand filters S1 and S4 at a hydraulic loading rate of 5 cm/h

In the range studied, HLR only had a minor influence on bacteria removal. This is in accordance with the fact, that porosity did not have a major impact on bacteria removal, because they both determine hydraulic retention time in the sand bed. Phase II (HLR = 10 cm/h) and III (HLR = 20 cm/h) are comparable because of similar conditions, but the experimental setup did not allow simultaneous operation at several HLRs. Due to these reasons, HLR was not included in the regression. The profiles of *E. coli* removal over filter length in Figure 3-9 show that the increase in HLR did not lead to significant differences. This may be surprising because elimination of fecal indicator bacteria in natural systems for wastewater treatment like ponds can usually be described by first-order kinetics. Doubling the HLR and thus decreasing the hydraulic retention time by half might be expected to have a more pronounced impact. Figure 3-10 shows that the small impact of HLR on bacteria removal was due to the fact that removal per hydraulic retention time increased after doubling the HLR. This may have been caused by an increase in the reaction rate coefficient or higher concentrations of retained bacteria. An elevated HLR is expected to lead to a faster development of the schmutzdecke and also a better development of biofilm in the bed depth leading to an increase in retained bacteria. This will be discussed further in chapter 3.7.

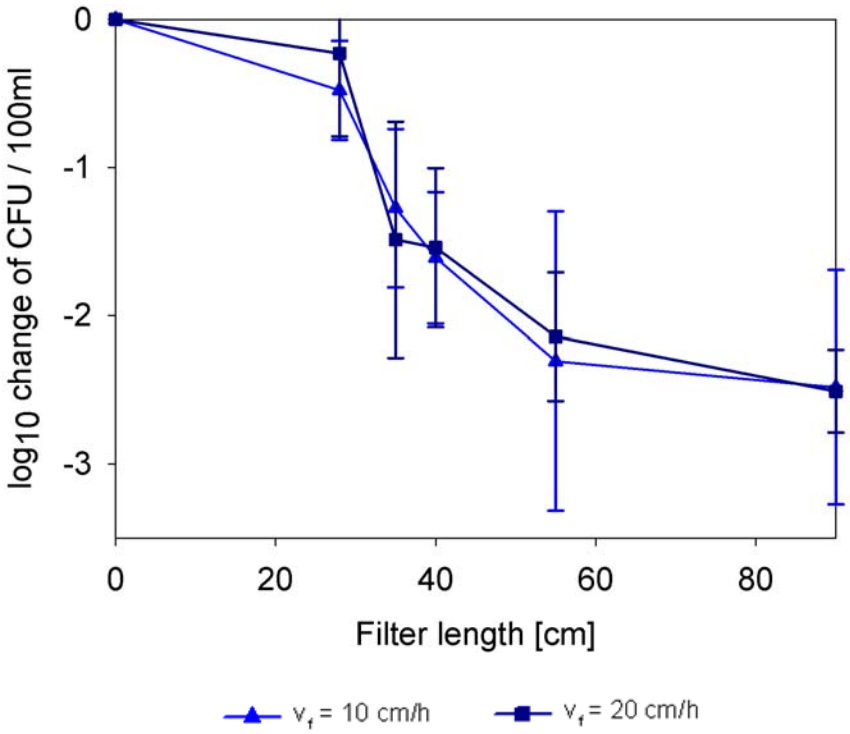


Figure 3-9: Profiles of *E. coli* concentration over filter length of pilot scale slow sand filter S5 at hydraulic loading rates of 10 cm/h and 20 cm/h

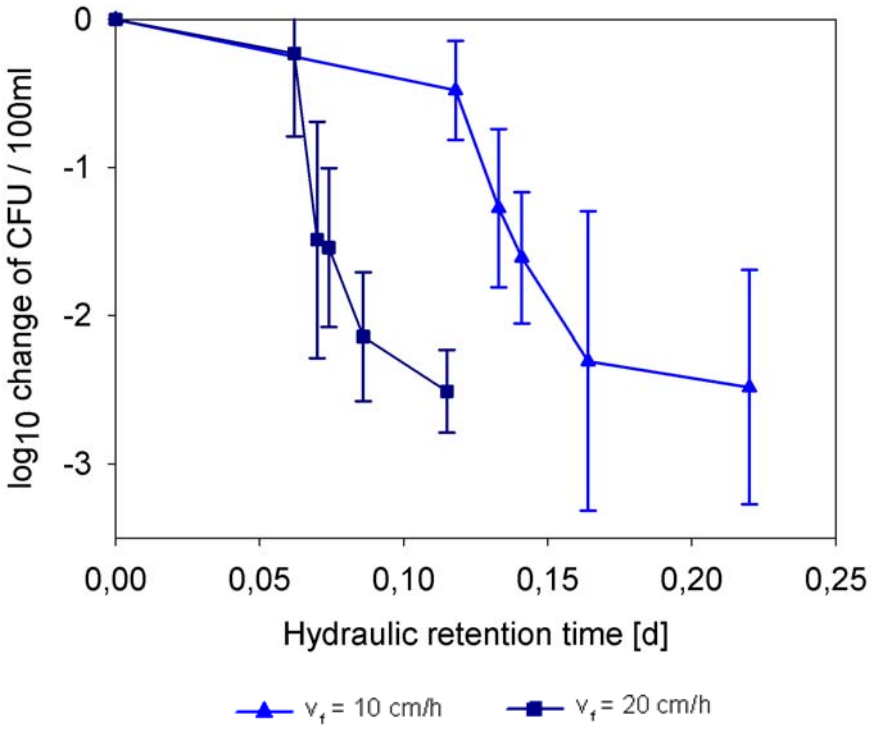


Figure 3-10: Profiles of *E. coli* concentration over hydraulic retention time of pilot scale slow sand filter S5 at hydraulic loading rates of 10 cm/h and 20 cm/h

The limited impact of HLR on bacteria removal has also been observed in drinking water purification (Huisman, 2004) and may consistently be explained by the fact that retained bacteria are not affected by reduced hydraulic retention times due to an increase in HLR. Also removal in deeper zones of the filter is expected to increase when higher filter velocities carry more substrate deeper into the sand bed encouraging biofilm growth and thus improving retention (Ellis, 1985). Mälzer (2005) even found an increase in *E. coli* removal from 2.3 to 3.7 log-units after increasing the HLR from 1 to 5 m/d. On the other hand, Sadiq (2003) reported that bacteria removal declined with increasing HLR. It seems possible that these contradictions are due to the fact that experiments at different HLRs were influenced by changing ambient conditions. Also the removal may have been measured without having allowed sufficient time for biomass to accumulate at the higher HLR. Therefore changes in bacteria removal cannot be exclusively attributed to a change in hydraulic loading rate.

The correlation of bacteria removal with sand surface area for a certain hydraulic loading rate is mechanistically sound. Bacteria with an average size of 1–10 μm cannot be removed by straining, because pore channels are too wide (0.155 times the sand grain diameter corresponding to 36 μm for the smallest d_{10} in these experiments) even considering that they

have been narrowed down by biofilm growing on the sand grains (Huisman, 2004). Huisman (2004) concluded that adsorption is the most important purification process. An increase in sand surface area leads to an increase in possible adsorption spots on sand and biofilm attached to the sand grains. For SSF of surface water it has been reported that an increase in d_{10} of filter sand from 0.25 mm to 0.63 mm resulted in a linear decrease in total coliform bacteria removal from 98.6% to 96% (Bellamy et al., 1985; Logsdon, 1991). While this trend is similar to the one observed in our experiments, bacteria removal seems to be linked to the d_{10} only indirectly, because it is essential in determining the specific sand surface area. Sadiq (2003) found that an increase in sand bed depth improved total coliform removal, whereas an increase in sand grain size did not significantly diminish it. Unfortunately their study only compared sands of two different d_{10} and did not present their uniformity coefficients.

A major advantage of the correlation of bacteria removal with sand surface area is that all three major design parameters are encompassed: bed depth, effective size (d_{10}) and uniformity. To make filters of different dimensions comparable, log removal was plotted against cumulated sand surface area (that the water had percolated through) per filter surface area in Figure 3-11. It is remarkable that the design guidelines for SSF in drinking water purification prohibit filters of less than 1,877 m²/m² (derived from d_{10} = max. 0.4 mm; U = max. 5; porosity = 40%; sand bed height = min. 50 cm). Regarding our data the hyperbolas increase nearly linearly in the acceptable zone. A drop below 1,877 would result in a rapid decline of bacteria removal whereas a further increase leads to a noticeable but minor improvement of filter performance.

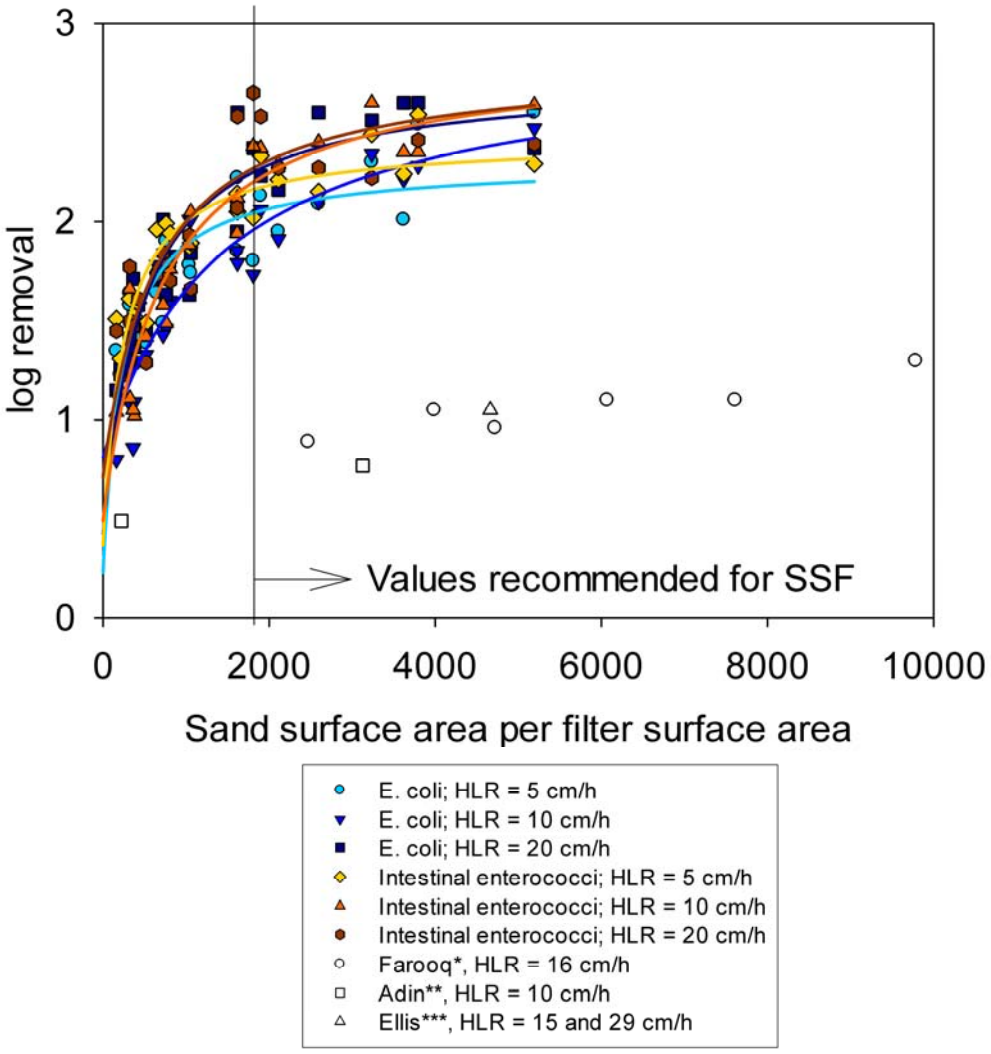


Figure 3-11: Comparison of indicator bacteria removal in SSFs of different design (*(Farooq and Alyousef, 1993), **(Adin et al., 1998), ***(Ellis, 1987)) and at different hydraulic loading rates as well as range of sand surface area per filter surface area suggested by authoritative sources for SSF of surface water

Total coliform removal by SSF of secondary effluent has been reported by several authors: 0.3–1.2 log-units (Adin et al., 1998), 2.4–3.5 log-units (Mälzer, 2005), 79.1–98.9% (Sadiq et al., 2003), 91–99% (Ellis, 1987) and 93–99% (Farooq and Alyousef, 1993). Mälzer (2005) also found *E. coli* removal of 2.3–3.7 log-units and Enterococci removal of 0.7–2.6 log-units. The reduction of *E. coli* and Enterococci measured is comparable to the data of Mälzer (2005), who used similar configurations with a filter sand in the range of 0.2–2 mm and hydraulic loading rates of 1 and 5 m/d. In general, comparisons are difficult because of differing process parameters such as sand bed depth and filter velocity.

Total coliforms only have limited importance as a fecal indicator organism, since only *E. coli* is used by the WHO guidelines on wastewater reuse and *E. coli* as well as Enterococci by the

EU bathing water Directive. It also seems that total coliforms may multiply in slow sand filters (Logsdon, 1991; Adin et al., 1998; Petry-Hansen, 2005). The bacteria removal reported in this study is within the typical range of 2–3 log-units of *E. coli* reported for treatment of surface water (Huisman and Wood, 1974; Hijnen et al., 2004). The concentration of *E. coli* in the product water is likely to be higher when SSFs are used for treatment of secondary effluent, since higher concentrations can be expected in secondary effluent compared to surface water. Higher removal of up to 4 log-units has also been reported. It may be reached by using a bed depth of more than the minimum recommended value of 50 cm and sands with a d_{10} <0.23 mm.

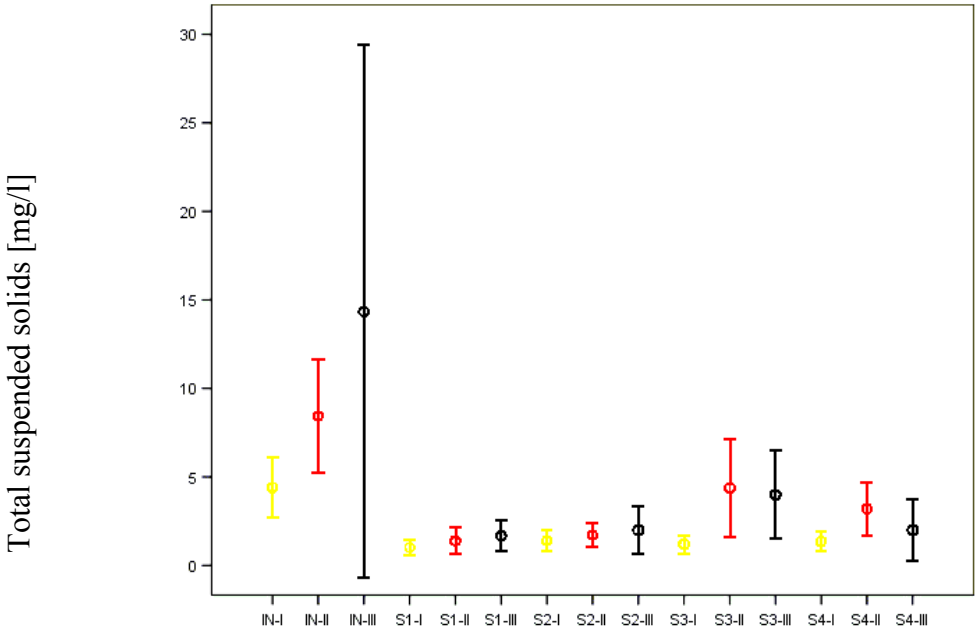
The finding that substantial bacteria removal is achieved in the schmutzdecke is supported by a study performed in Israel (Adin et al., 1998). Another study showed that an increase in sand surface area, either by smaller $d_{10,}$ U or deeper sand beds only results in little extra bacteria removal beyond 2,000 m²/m² (Farooq and Alyousef, 1993). Ellis (1987) found the same removal after doubling the hydraulic loading rate. This is similar to the limited impact of HLR on bacteria removal found in our experiments. A plausible explanation for the comparatively low bacteria removal found by all these authors may be that they used total coliforms as indicator organisms that are more likely to regrow in the filter than *E. coli*. Our conclusion is also supported by the finding that sand bed depths of 0.5–1m did not have a significant influence on fecal coliform removal (Keraita et al., 2008).

Sand surface area per filter surface area seems to largely determine bacteria removal. It can be useful for designing slow sand filters, especially since individually optimal combinations of grain size distribution and bed depth can be used. However, a prerequisite for this design process is a broad basis of experience and data obtained under a variety of hydraulic loading rates and ambient conditions. Individual variation of all relevant process parameters would be neccessary for exact correlation of filter performance to the parameters. Modeling as discussed in section 3.7 may be a more efficient way to predict filter performance.

3.2.2 Removal of solids

The pilot scale slow sand filters removed 70–84% of total suspended solids from secondary clarifier effluent. Mean filter effluent concentrations of 1.2–2.3 mg/l were significantly lower than influent concentrations ranging from 4.4–14.3 mg/l. No significant differences were found between the effluent TSS concentrations of the filters during phase I to III of the experiments (Figure 3-12).

a)



b)

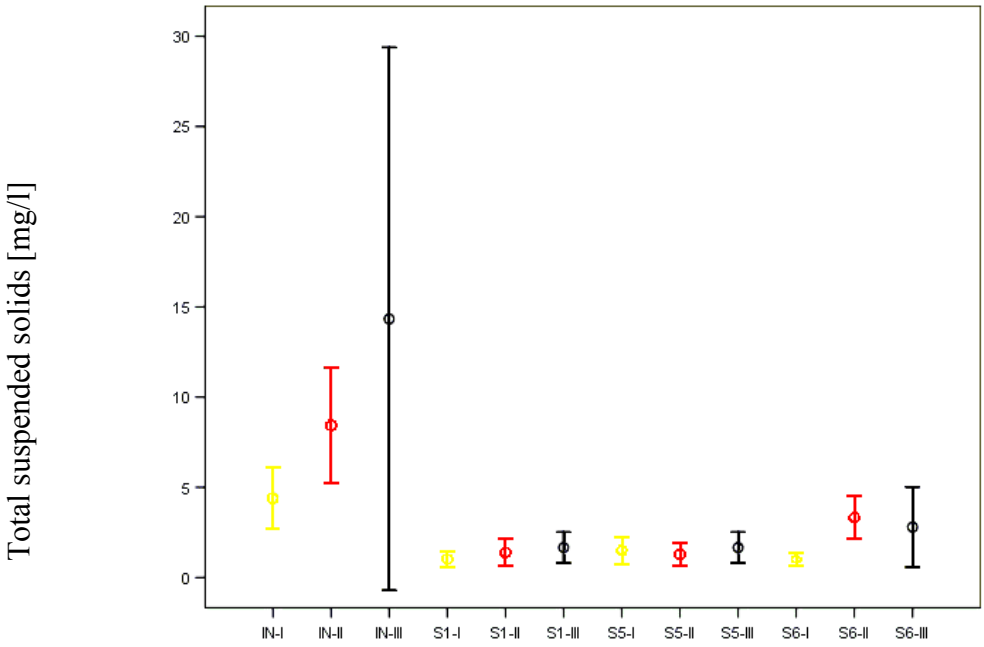


Figure 3-12: Arithmetic mean and 95%-confidence intervals of total suspended solids concentrations in influent (IN) and effluent of slow sand filters of a) decreasing d_{10} at constant uniformity (S4–S1) and b) increasing uniformity at constant d_{10} (S1, S5, S6) during phases of increasing hydraulic loading rate (I: HLR = 5 cm/h; II = 10 cm/h; III = 20 cm/h)

Mean TSS removals were within the range reported by Ellis (1987) from 58–93% and a little higher than the 60–75% removal that Adin et al. (1998) measured. Variations are probably due to different process parameters and influent quality. Turbidity removal amounted to 73–

89% reaching 0.5–0.8 NTU in the filter effluent. It was a little below the range reported by Farooq and Alyousef (1993) and a little higher than Ari (2006) reported. Neither sand grain size distribution nor hydraulic loading rate had a considerable impact on TSS removal. This seems plausible, since solids removal is mainly due to mechanical straining that is not affected by filter velocity and almost exclusively takes place in the schmutzdecke which after the short period of ripening is no more influenced by sand grain properties. Apart from straining in the schmutzdecke, the TSS removal processes are not similar to those causing the bacteria removals. Ellis (1987) made similar observations.

3.2.3 Clogging and maintenance

Slow sand filters require regular maintenance to remove the schmutzdecke when it has developed to an extent that the desired HLR can no longer be achieved. In the experiments the supernatant levels rose exponentially but clogging events did not follow a plausible pattern, probably because of the limited amount of incidents during the total runtime of 295 days (Annex A-3). During this time the frequency of clogging for column S5 was five times, for columns S1, S3 and S6 four times and twice for columns S2 and S4.

The runtimes can be considered typical in SSF of potable water (Adin et al., 1998). Adin et al. (1998), Farooq and Alyousef (1993) and Ellis (1987) concluded that coarse sands have the advantage of a longer runtime between maintenance events. The hypothesis that the finest and most heterogenous sand (S6) with smallest pore channel diameters and porosity would lead to quickest clogging could not be validated within the duration of our experiments. In drinking water treatment, an increase of runtime had been observed for sands of higher uniformity coefficients or higher d_{10} (Di Bernardo and Escobar Rivera, 1996; van der Hoek et al., 1996). The relationship between suspended solids loading rate, sand grain size and runtime needs to be studied in further detail.

The exponential increase of filter resistance and thus supernatant water level may consistently be explained by the self-amplifying process of the increasing water level which further compresses the schmutzdecke. Due to the exponential increase filters may be designed for only a minor increase in supernatant water of 10–30 cm, especially when operated at higher hydraulic loading rates. This will reduce the investment costs for the filter structure whereas generally recommended heights of 1.5 m of supernatant water do not seem to efficiently gain extra runtime. Wet-harrowing was used as the maintenance technique. Thereby hydraulic conductivity could be restored for all filter sands including the coarsest ones. This points out that clogging only occurred at the filter surface or schmutzdecke and that filter sands were

chosen fine enough to prevent excessive transport of particles into the filter bed. No significant deterioration of effluent quality was observed one day after maintenance.

3.2.4 Physical, chemical and sum parameters

An overview of further parameters in the filter influent and effluent is given in Table 3-6.

Table 3-6: Arithmetic mean and standard deviation of physical, chemical and sum parameters in the influent and effluent of pilot scale slow sand filters S1–S6 during phases of increasing hydraulic loading rate

		Secondary clarifier	Filter column effluent					
	Phase	effluent	S1	S2	S3	S4	S5	S6
O_2 [mg/l]	I	6.75 ± 0.40	8.08 ± 0.76	7.92 ± 2.21	8.54 ± 1.09	8.33 ± 1.10	8.03 ± 1.01	8.56 ± 0.91
	II	6.74 ± 1.37	5.26 ± 2.19	6.08 ± 1.75	6.44 ± 1.67	6.03 ± 1.65	6.31 ± 1.42	5.79 ± 1.61
	III	5.47 ± 0.56	4.00 ± 1.67	4.28 ± 2.04	4.21 ± 2.48	3.65 ± 2.17	3.82 ± 2.06	3.94 ± 2.31
T [°C]	I	12.1 ± 5.2	9.5 ± 5.6	9.1 ± 5.3	9.5 ± 5.7	9.1 ± 5.1	9.7 ± 5.6	8.0 ± 3.8
	II	19.1 ± 5.4	17.6 ± 6.2	17.5 ± 6.0	17.5 ± 5.6	17.9 ± 5.8	18.1 ± 6.1	17.6 ± 6.1
	III	20.8 ± 2.6	20.3 ± 3.2	20.9 ± 3.4	20.3 ± 3.7	20.5 ± 3.4	20.1 ± 3.3	20.5 ± 3.9
pH	I	6.9 ± 0.1	7.0 ± 0.2	7.0 ± 0.2	7.0 ± 0.2	7.0 ± 0.2	7.0 ± 0.2	7.0 ± 0.2
	II	6.7 ± 0.2	6.8 ± 0.2	6.8 ± 0.2	6.8 ± 0.2	6.6 ± 0.7	6.8 ± 0.2	6.7 ± 0.2
	III	6.9 ± 0.1	6.8 ± 0.1	6.9 ± 0.1	6.8 ± 0.2	6.8 ± 0.2	6.8 ± 0.2	6.9 ± 0.2
Redox potential	I	235 ± 30	228 ± 17	218 ± 46	207 ± 49	216 ± 44	206 ± 46	220 ± 39
[mV]	II	179 ± 62	176 ± 75	189 ± 63	191 ± 62	182 ± 53	199 ± 46	177 ± 56
	III	148 ± 32	170 ± 36	156 ± 37	170 ± 44	171 ± 43	184 ± 28	152 ± 46
Conductivity	I	1317 ± 154	1320 ± 148	1315 ± 144	1315 ± 148	1318 ± 148	1370 ± 214	1322 ± 149
[µS/cm]	II	1216 ± 144	1205 ± 173	1204 ± 160	1204 ± 159	1209 ± 153	1205 ± 159	1205 ± 160
	III	1305 ± 54	1315 ± 71	1313 ± 67	1320 ± 79	1318 ± 77	1315 ± 69	1295 ± 68
TOC [mg/l]	I	7.7 ± 1.9	7.4 ± 1.0	6.4 ± 1.8	6.2 ± 1.9	7.5 ± 0.7	7.5 ± 0.8	7.8 ± 1.1
	II	9.9 ± 1.3	9.2 ± 1.7	9.0 ± 1.3	9.3 ± 1.3	9.1 ± 1.4	8.9 ± 1.3	8.5 ±1.5
	III	17.1 ± 15.1	13.3 ± 8.0	16.6 ± 14.7	7.6 ± 4.1	9.2 ± 2.1	16.5 ± 14.7	16.5 ± 12.1
COD [mg/l]	I	25.8 ± 5.8	22.6 ± 3.4	22.1 ± 3.0	21.7 ± 3.9	21.7 ± 4.0	21.9 ± 2.5	23.8 ± 0.7
	II	31.3 ± 6.7	22.3 ± 2.8	23.6 ± 3.2	22.8 ± 4.5	24.7 ± 3.1	23.9 ± 3.2	24.2 ± 3.6
	III	37.3 ± 11.3	20.1 ± 2.2	20.6 ± 0.8	22.8 ± 2.2	20.8 ± 3.1	21.3 ± 2.5	21.5 ± 2.8
BOD_5 [mg/l]	I	2.9 ± 2.3	0.5 ± 0.6	1.7 ± 0.8	0.8 ± 0.3	1.4 ± 1.6	0.5 ± 0.5	1.1 ± 0.3
	II	4.9 ± 3.0	2.8 ± 2.1	3.1 ± 2.7	3.4 ± 3.0	3.5 ± 2.9	3.2 ± 2.8	3.3 ± 2.8
	III	7.9 ± 4.7	3.3 ± 1.9	3.6 ± 1.8	3.2 ± 2.2	3.5 ± 2.2	3.5 ± 2.1	3.3 ± 2.1
NO_3^- [mg/l]	I	21.7 ± 7.0	19.7 ± 2.2	20.2 ± 5.2	19.4 ± 5.3	20.2 ± 5.9	19.2 ± 5.6	18.7 ± 3.4
	II	27.3 ± 6.8	26.1 ± 4.9	27.3 ± 7.0	28.1 ± 5.9	28.9 ± 9.1	27.8 ± 6.6	25.9 ± 6.7
	III	19.1 ± 5.4	17.1 ± 7.1	18.7 ± 4.9	19.1 ± 4.6	16.3 ± 8.1	19.0 ± 4.5	17.4 ± 5.1
PO_4^{3-} [mg/l]	I	2.7 ± 1.5	2.1 ± 1.8	3.0 ± 1.5	2.6 ± 1.8	2.6 ± 1.3	3.1 ± 2.3	2.6 ± 1.5
	II	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
	III	6.6 ± 7.0	0.9 ± 0.9	1.0 ± 1.0	1.4 ± 1.2	1.0 ± 1.1	0.6 ± 0.6	1.2 ± 1.6

I = 5 cm/h; II = 10 cm/h; III = 20 cm/h; NH_4^+ always below detection limit

Aerobic conditions prevailed in the water phase of all filters while denitrification may have occured at micro-gradients of redox-potential within the biofilm. BOD removal ranged from an average 34–66% and COD removal from 14–43% during the three phases with no significant and systematic differences detected between filters.

3.3 Evaluation of scale-up

The comparison of filters at laboratory and pilot scale fed with secondary clarifier effluent of the same wastewater treatment plant and operated at the same hydraulic loading rate shows that scale up did not significantly affect bacteria removal (Figure 3-13 and Figure 3-14). The sampling points chosen for comparison were the effluent of the large pilot scale filter with a sand bed depth of 25 cm (described in section 2.1.3) and the filter horizons at 25 cm bed depth of laboratory and small pilot scale filters that were well comparable in terms of cumulated sand surface area. These were C2 and C3 (laboratory scale) as well as S2 and S3 (small pilot scale).

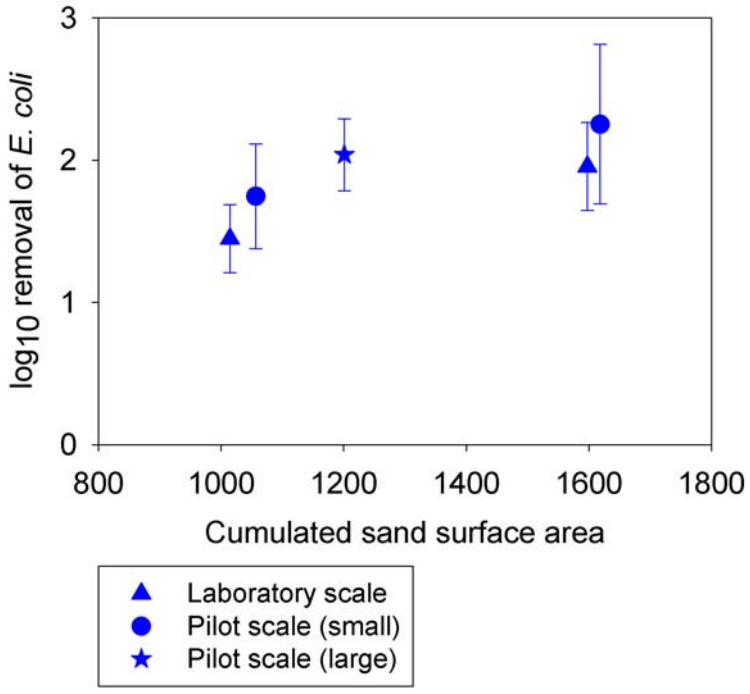


Figure 3-13: Mean values and 95%-confidence intervals of *E. coli* removal at a hydraulic loading rate of 5 cm/h in slow sand filters at different scale as compared by their cumulated sand surface area [m²]

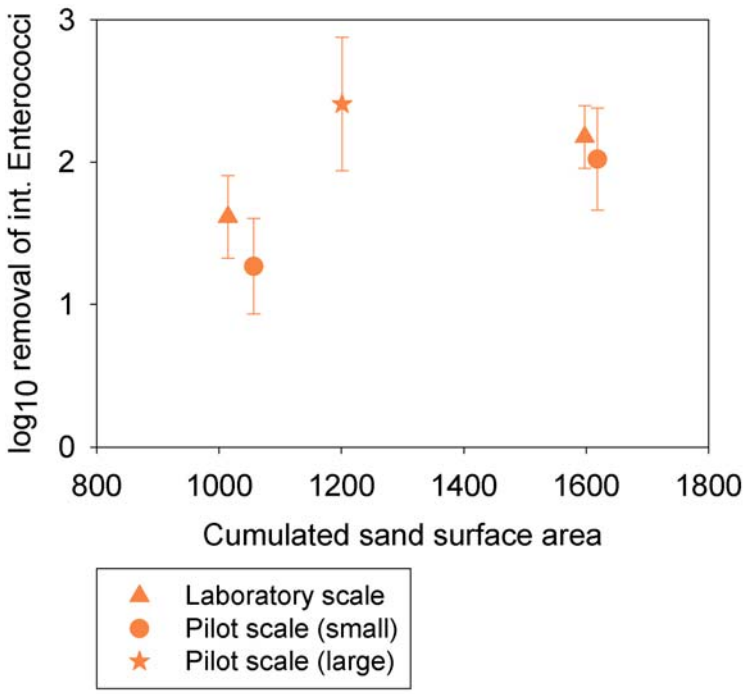


Figure 3-14: Mean values and 95%-confidence intervals of intestinal Enterococci removal at a hydraulic loading rate of 5 cm/h in slow sand filters at different scale as compared by their cumulated sand surface area [m²]

The overlapping confidence intervals indicate that the scale of the experiments did not significantly affect indicator bacteria removal. The fact that only in the large pilot scale filter the water had passed a gravel bed of 10 cm depth before sampling, may be the reason, why removal in this filter is a little higher than expected from the removal in the other filters. Further scale-up is promising.

Physical, chemical and sum parameters characterizing the effluent of the large pilot scale filter were closely comparable to the values of the other filters (Table 3-7). This also holds for particle removal.

During a period of operation of 170 days the filter had to be maintained once. The fecal indicator bacteria removal of about 2 log-units was sufficient to reach bathing water quality in the effluent (383 CFU/100 ml of *E. coli* and 244 CFU/100 ml of intestinal Enterococci). A monitoring level of 100 CFU/100 ml of *E. coli* was just not reached (geometric mean of 107 CFU/100 ml in the filter effluent).

Table 3-7: Selected characteristics of the effluent of the large pilot scale slow sand filter

O_2 [mg/l]	5.9 ± 0.7
T [°C]	10.9 ± 5.2
pH	7.0 ± 0.3
Redox potential [mV]	218 ± 34
Conductivity [µS/cm]	1,312 ± 165
TOC [mg/l]	7.1 ± 1.3
COD [mg/l]	22.2 ± 4.9
BOD_5 [mg/l]	1.2 ± 0.5
NH_4^+ [mg/l]	below detection limit
NO_3^- [mg/l]	18.7 ± 9.8
PO_4^{3-} [mg/l]	3.8 ± 0.9
Turbidity [NTU]	0.5 ± 0.4
TSS [mg/l]	1.0 ± 0.8

The material cost of about 40 € for the whole filter including barrel, sand, tubes and all other elements was extremely low considering that the filter may easily disinfect 4 people equivalents or roughly 500 liters per day. This slight increase compared to the HLR used is expected to only result in a little decrease of runtime. The unit may be beneficially applied for any pilot testing undertaken to evaluate the potential of SSF for disinfection of secondary effluent at a specific setting. It may also serve as the basis of a modular, effective and cheap system to disinfect effluent of decentralized wastewater treatment plants.

3.4 Comparison of intermittent sand filter and slow sand filter

3.4.1 Oxygen conditions

The objective of these experiments was to evaluate the performance of a slow sand filter and an intermittent sand filter. Unlike the filters C1-C4 at laboratory scale they were also exposed to challenging conditions (see section 2.1.4). Table 3-8 shows the biochemical oxygen demand, ammonium and suspended solids concentration during the three phases of increasingly challenging influent water. The presence of oxygen is considered to be important for bacteria elimination since predation is a major mechanism for removal and predators rely on aerobic conditions. The established technology of intermittent sand filtration has a better oxygen supply since it is replenished by the air sucked in after each feeding.

Table 3-8: Mean concentrations of selected parameters in the influent of the intermittent and slow sand filter during the three experimental phases of increasing oxygen demand caused by BOD_5 and ammonium at a HLR of 0.65 m/d

	Phase		
	I	II	III
BOD_5 [mg/l]	2.7	14.5	55
NH_4^+ [mg/l]	below detection limit	4.1	8.9
TSS [mg/l]	12.5	94.1	250

Figure 3-15 shows arithmetic mean values and standard errors of the oxygen concentrations in influent and effluent of the filters. The concentration of oxygen in the influent during phase I was close to saturation at the temperature of around 5 °C in the storage barrel. Despite cooling of the barrel, oxygen consumption led to lower concentrations during phases II and III in the influent. During phase I the concentration of dissolved oxygen in effluent of the ISF was still close to saturation at the lab temperature of 22 °C. During phases II and III more oxygen was consumed than replenished due to higher oxygen demands in the feeding water. This led to concentrations well below saturation but still indicating aerobic conditions in the water phase of the whole filter. In the SSF oxygen consumption caused effluent concentrations below saturation in phase I already. Nevertheless aerobic conditions prevailed in the water phase of the filter during this phase. With increasing oxygen demand, effluent concentrations dropped to 0.9 and 0.7 mg/l (the electrode used reported 0.2 mg/l in water free of oxygen). This

indicated that during phases II and III deeper zones of the SSF were close to anoxic in the water phase.

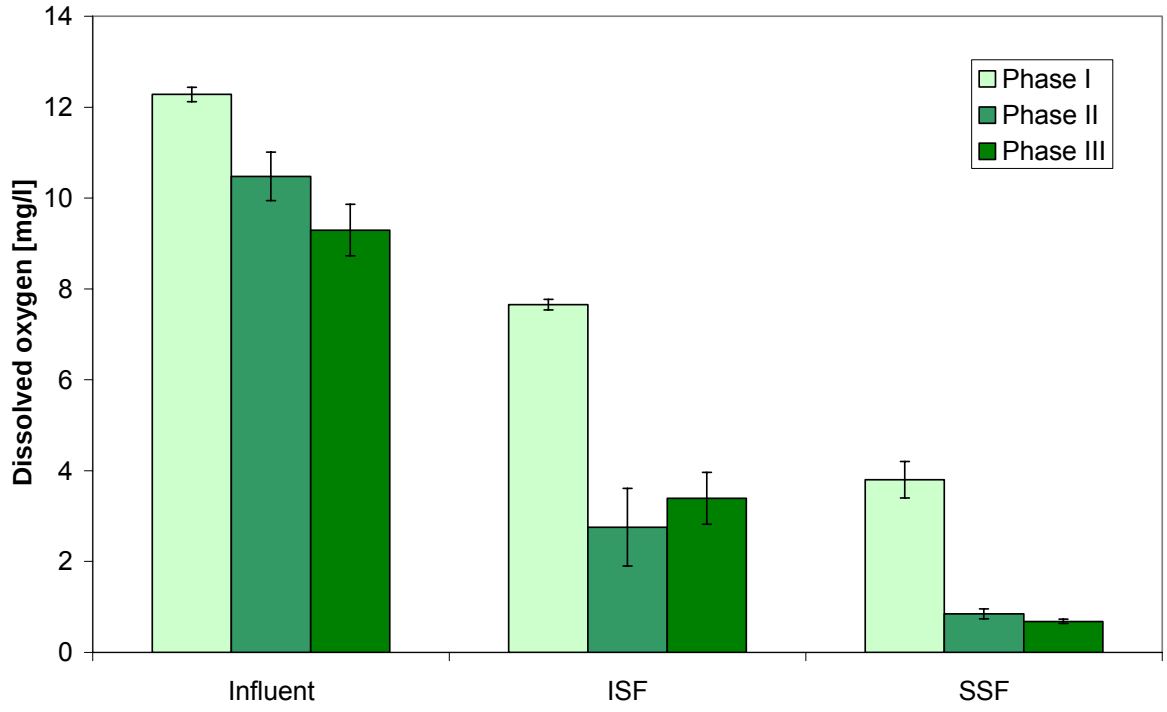


Figure 3-15: Arithmetic mean and standard error of the concentration of dissolved oxygen in the influent and effluent of the ISF and SSF during phase I–III of increasing oxygen demand

Higher oxygen consumption in the SSF is due to two factors. The hydraulic retention time of about 19 hours in the SSF is much longer than in the ISF, where water is approximately retained for one feeding interval of 2 hours. This allows for longer action of microbial processes and thus consumption of oyxgen. Also the supply of oxygen is very limited in the SSF. Both filters are supplied with dissolved oxygen in the feeding water. However, diffusion of oxygen into the SSF is only possible through the small surface area of the supernatant water. In the case of the ISF, the pore water is surrounded by air sucked into the filter bed after feeding allowing for much better diffusion through a large boundary layer.

Zones of low dissolved oxygen concentrations in the SSF were identified. Figure 3-16 shows how an increase in oxygen demand led to lower concentrations of dissolved oxygen over the length of the slow sand filter. Fastest consumption and thus highest biological activity was in the filter zones supernatant, schmutzdecke and upper 5 cm of the sand bed. During phase II the zone of high consumption stretched further down to 10 cm bed depth. As less oxygen was available in the bed depth, consumption per filter length declined. It should be noted that even

during phase III the dissolved oxygen was not completely consumed in the SSF. This result was cross-checked by measuring both with an electrode (WTW) in a flow through cell and a fluorescence flow through cell (PreSens) connected to the sampling ports.

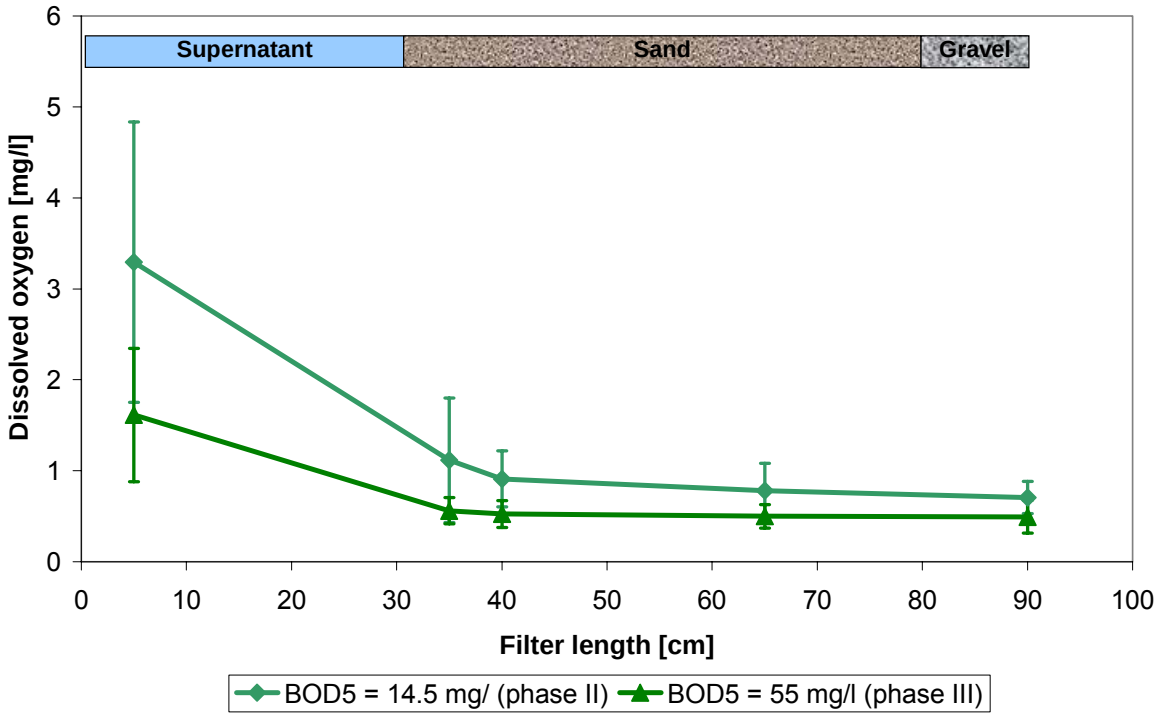


Figure 3-16: Mean concentrations and standard errors of dissolved oxygen over the length of the slow sand filter affected by the increase in oxygen demand of the influent water from phase II to phase III

Even under aerobic conditions in the water phase, anoxic or anaerobic zones may exist in the biofilm due to limited diffusion into the biofilm, where microbial processes deplete oxygen. Within such a micro-gradient, microorganisms may switch to nitrate and sulphate as an electron acceptor. Denitrification will be discussed in section 3.4.4. An indication for anaerobic zones within the filter was the black colour of the sand bed that was probably caused by iron sulfide (Figure 3-17).

a)  b) 

Figure 3-17: Regular color of sand and schmutzdecke in phase I (a) compared to a black color of the same slow sand filter caused by a high oxygen demand in the influent in phase III (b)

Redox-potentials of filter influent and effluents are given in Table 3-9. They support the finding that conditions in the ISF were more oxidative than in the SSF.

Table 3-9: Arithmetic mean and standard error of the redox-potential in influent and effluent of the intermittent and slow sand filters during phases I-III of increasing oxygen demand

	Phase	Influent	ISF effluent	SSF effluent
Redox-	I	225 ± 16	213 ± 11	179 ±10
potential [mV]	II	236 ± 9	226 ± 3	185 ± 21
	III	184 ± 16	193 ± 14	154 ± 9

In order to improve the availability of oxygen in the slow sand filter, the hydraulic retention time was decreased by increasing the hydraulic loading rate. Unlike intermittent sand filters, SSFs are typically operated at a higher HLR than 0.65 m/d (0.03 m/h). An increase to 0.16 m/h resulted in aerobic conditions throughout the water phase of the filter (Figure 3-18). From the results described in section 3.2.1, it may be expected that an increase in HLR does not significantly diminish bacteria removal but reduce runtime. This encourages the choice of elevated HLRs for waters of high oxygen demands. The loading rate could be raised to 0.4 m/h or even more. Also the filter bed depth and height of supernatant water should be minimized to achieve short detention times.

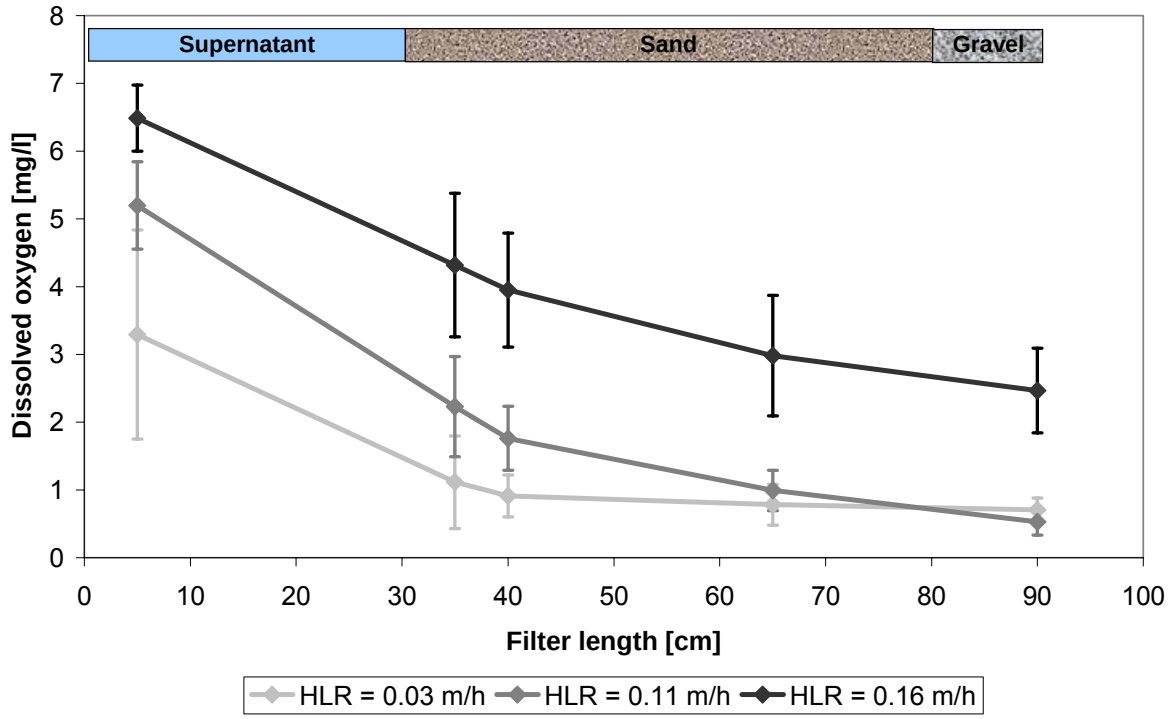


Figure 3-18: Mean concentrations and standard errors of dissolved oxygen over the length of a slow sand filter affected by the hydraulic loading rate at a BOD_5 of 14.5 mg/l in the influent

3.4.2 Bacteria removal

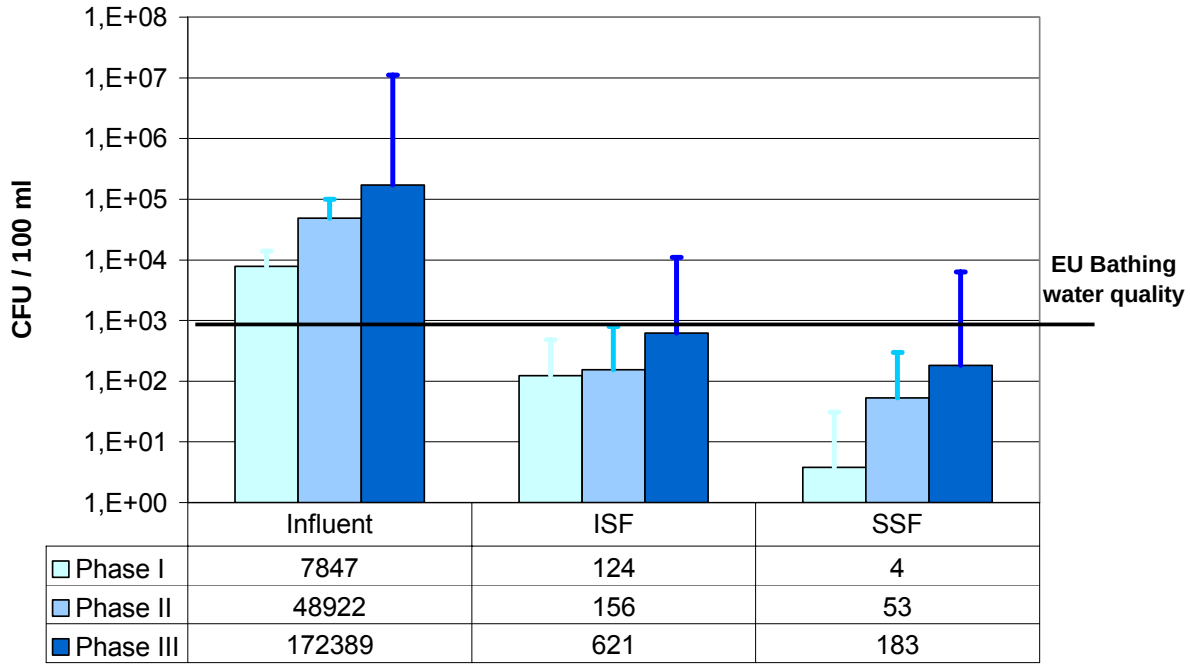
The hypothesis that fecal indicator bacteria removal in the SSF would collapse due to high oxygen demand of the feeding water was not confirmed (Table 3-10).

Table 3-10: Bacteria removal in the intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand

	Phase	Bacteria removal			
		Log_{10}-units		Percent	
		ISF	SSF	ISF	SSF
E. coli	I	1.8	3.4	98.42	99.95
	II	2.5	3.0	99.68	99.89
	III	2.4	2.7	99.64	99.89
Intestinal	I	1.6	3.0	97.70	99.88
Enterococci	II	2.7	3.0	99.81	99.90
	III	2.8	3.2	99.71	99.96

The SSF eliminated 2.7–3.4 log-units or 99.89–99.95% of *E. coli* and 3.0–3.2 log-units or 99.88–99.96% of intestinal Enterococci. The ISF removed less bacteria: 1.8–2.5 log-units or 98.42–99.68% of *E. coli* and 1.6–2.8 log-units or 97.70–99.81% of intestinal Enterococci.

This bacteria removal was high enough to reach bathing water quality according to EU Directive in the effluent of the SSF during phases I and II and the ISF during phase I (Figure 3-19, Figure 3-20 and Table 3-11). A monitoring level of <10 CFU/100 ml of *E. coli* was achieved by the SSF during phase I and <100 CFU/100 ml during phase II while all filter effluents during all phases reached a level of <1000 CFU/100 ml of *E. coli*.



	Influent	ISF	SSF
Phase I	7847	124	4
Phase II	48922	156	53
Phase III	172389	621	183

Figure 3-19: Geometric mean and 90th-percentile of *E. coli* concentration in influent and effluent of intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand

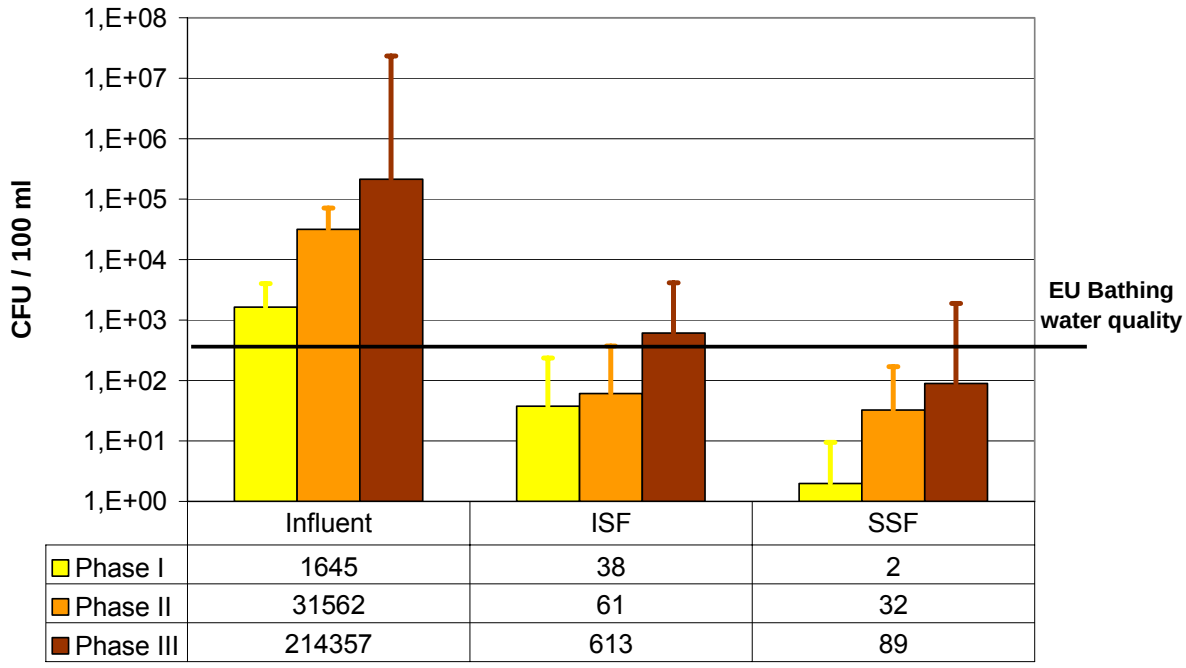


Figure 3-20: Geometric mean and 90th-percentile of intestinal Enterococci concentration in influent and effluent of intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand

Table 3-11: 90th-percentile of *E. coli* and intestinal Enterococci concentration in influent and effluent of intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand

		Concentration [CFU/100 ml]		
	Phase	Influent	ISF	SSF
E. coli	I	14,024	486	31
	II	100,368	800	300
	III	11,066,306	10,973	6,401
Intestinal	I	4,026	237	10
Enterococci	II	71,417	374	200
	III	23,236,635	3,812	1,927

A graphic comparison of fecal indicator bacteria removal in the ISF and SSF is given in Figure 3-21 and Figure 3-22.

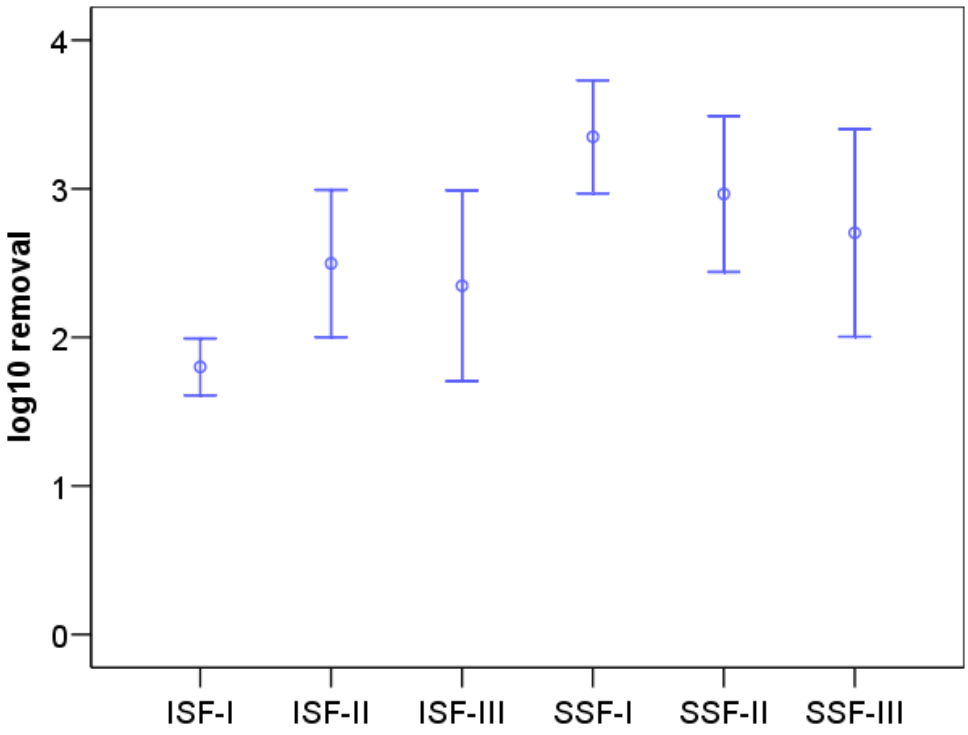


Figure 3-21: Mean and 95%-confidence intervals of *E. coli* removal in intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand

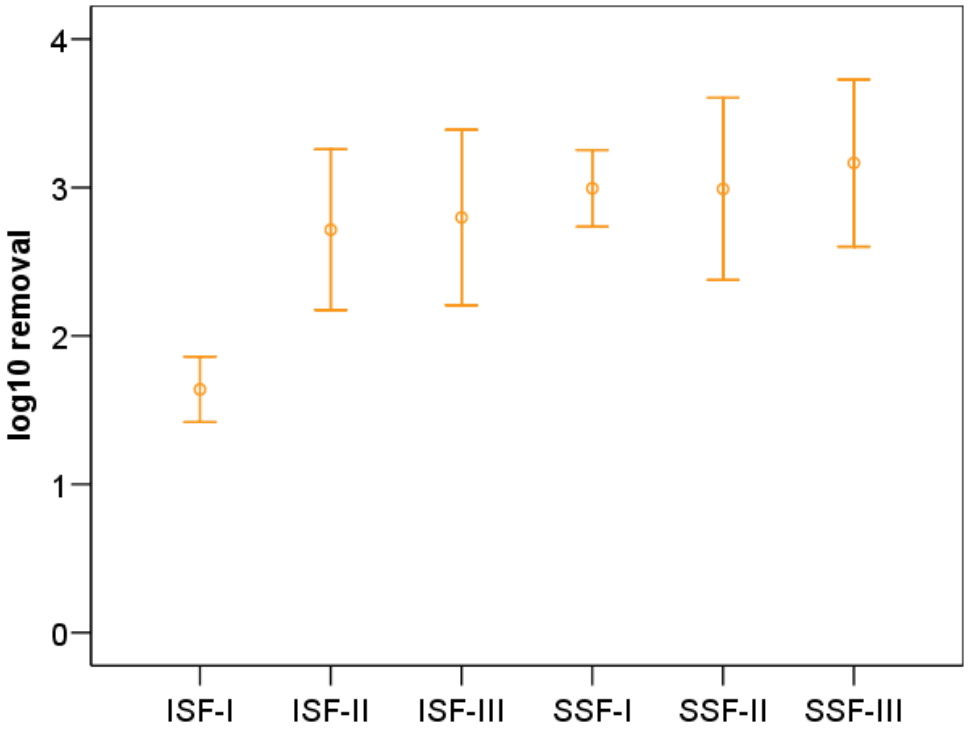


Figure 3-22: Mean and 95%-confidence intervals of intestinal Enterococci removal in intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand

The slow sand filter removed significantly more indicator bacteria than the intermittent sand filter during phase I ($p < 0.001$, Mann-Whitney-U-test). During phase II and III no significant differences between bacteria removal of ISF and SSF were found ($p > 0.05$). So a slow sand filter of the same sand and bed depth operated at the same hydraulic loading rate as an ISF performed much better when challenged with a low oxygen demand. The SSF removed an additional 1.6 log-units of *E. coli* and 1.4 log-units of intestinal Enterococci. A schmutzdecke was only present on the SSF but not the ISF. This zone is of great importance in bacteria removal (see section 3.2.1). With increasing oxygen demand and concentration of suspended solids during phase II and III bacteria removal in the SSF declined but improved in the ISF so that no significant differences were found any more.

Removal of *E. coli* and intestinal Enterococci in the ISF was significantly higher in phase II compared to phase I ($p < 0.001$, Mann-Whitney-U-test). This can be explained by the development of a filter cake on top of the ISF during phase II due to the increased load of suspended solids. This layer provided additional straining and adsorption spots. Also the elevated concentrations of BOD_5 and other substrate favored development of biofilm in the filter. It is assumed that biofilm facilitates adsorption in comparison to the sand surface and also narrows down pore channels which improves straining.

Contrary to the development in the ISF, *E. coli* removal in the SSF declined with increasing oxygen demand. The difference between phase I and II was nearly significant ($p = 0.057$, Kruskal-Wallis-test). Removal of intestinal Enterococci did not significantly differ in between the three phases. This may be explained by two opposing effects. Predation is considered to be an essential mechanism of bacteria removal and the predators need aerobic conditions. In the zones of low dissolved oxygen concentration their activity may be impeded. On the other hand, the increased load of suspended solids led to growth of the schmutzdecke, that plays an important role in bacteria removal. Also higher concentrations of substrate favored biofilm development resulting in improved straining and adsorption. In further experiments TSS concentration and oxygen demand should be varied independently to distinguish their influence on bacteria removal.

In the conducted experiments the SSF achieved comparable or higher bacteria removal than the ISF. A major advantage of the SSF compared to the established technology of ISF is the elevated hydraulic loading rate. Surface area requirements of SSFs can be up to 15 times less, even when the highest rates recommended for ISFs are considered.

3.4.3 Removal of suspended solids

Due to the experimental procedures the increase in oxygen demand was accompanied by an increase in the concentration of suspended solids in the filter influent. Figure 3-23 and Table 3-12 show the impact on effluent concentrations and removal. While removal during phase II and III was high as expected with values ranging from 98.6–99.9% and effluent concentrations below 2 mg/l, it was low during phase I. This may have been due to pieces of biofilm washed out of the effluent tube. During phases II and II the tube was washed weekly to avoid this possible source of error.

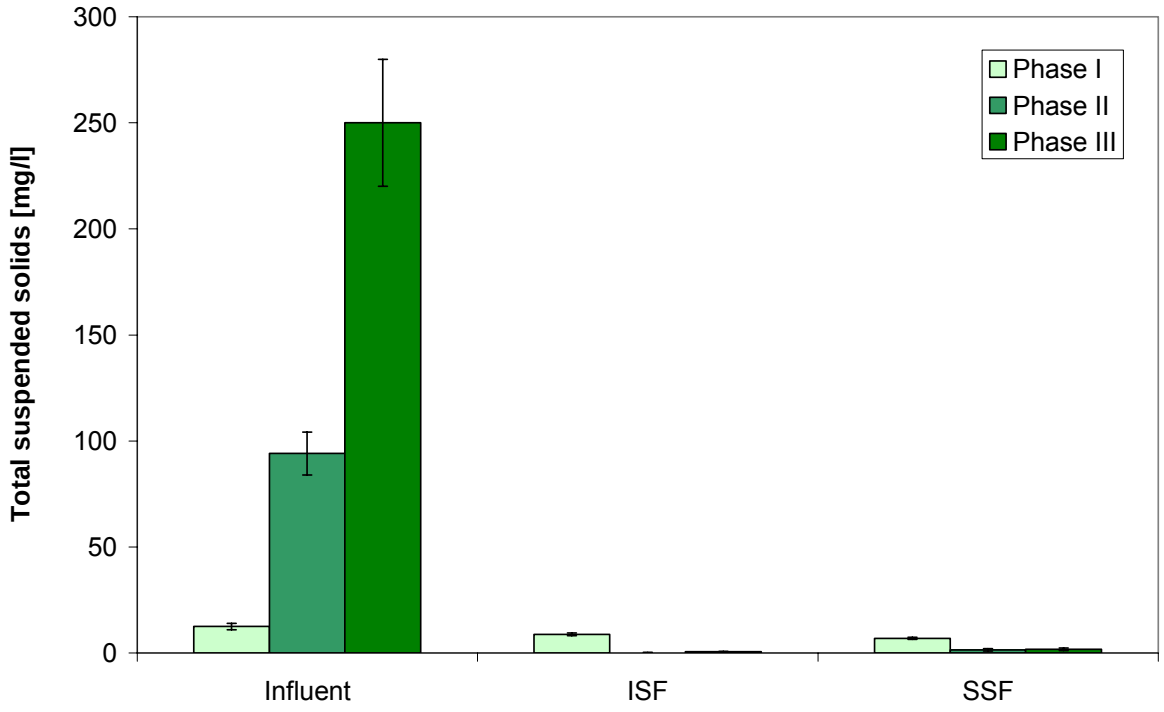


Figure 3-23: Arithmetic mean and standard error of the concentration of total suspended solids in the influent and effluent of the intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand and TSS concentration

Table 3-12: Removal of suspended solids in the intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand and TSS concentration

	Phase	Removal [%]	
		ISF	SSF
Suspended	I	29.6	44.8
solids	II	99.9	98.6
	III	99.8	99.3

3.4.4 Removal of biochemical oxygen demand and nitrogen

The effluent of the WWTP was low in oxygen demand (low BOD_5 and concentration of NH_4^+). The demand was increased during phase II and III by addition of recirculating sludge and primary effluent. Table 3-13 and Figure 3-24 summarize influent and effluent concentrations of the sand filters during phases I–III.

Table 3-13: Arithmetic mean and standard error of oxygen demand and nitrogenous compounds in the influent and effluent of the intermittent (ISF) and slow sand filter (SSF) during phase I–III

	Phase	Influent	ISF	SSF
BOD_5 [mg/l]	I	2.7 ± 0.6	3.1 ± 0.5	4.7 ± 0.7
	II	14.5 ± 3.0	2.5 ± 1.3	3.9 ± 1.2
	III	55.0 ± 8.6	3.7 ± 1.2	7.1 ± 1.1
COD [mg/l]	I	28.4 ± 1.3	25.2 ± 1.1	30.9 ± 1.7
	II	96.0 ± 18.1	32.6 ± 0.5	29.0 ± 1.0
	III	344.1 ± 31.3	26.5 ± 1.6	28.9 ± 2.2
NH_4^+ [mg/l]	I	b. d.	b. d.	b. d.
	II	5.3 ± 2.1	6.7 ± 2.3	2.1 ± 1.4
	III	11.5 ± 4.1	12.9 ± 4.1	8.8 ± 2.8
NO_2^- [mg / l]	I	b. d.	b. d.	b. d.
	II	b. d.	0.22 ± 0.16	0.82 ± 0.82
	III	b. d.	0.23 ± 0.23	0.94 ± 0.51
NO_3^- [mg/l]	I	21.2 ± 2.2	17.5 ± 1.0	1.4 ± 0.8
	II	20.8 ± 2.6	11.3 ± 2.7	8.3 ± 1.6
	III	23.0 ± 2.2	10.4 ± 2.9	16.9 ± 3.5

b.d. = below detection limit

The increase in oxygen demand during phase I is attributed to the same reason as the increase in TSS concentration. The BOD_5 concentration in the effluent of the ISF was a little lower than in the SSF while COD concentrations in the effluents did not differ significantly.

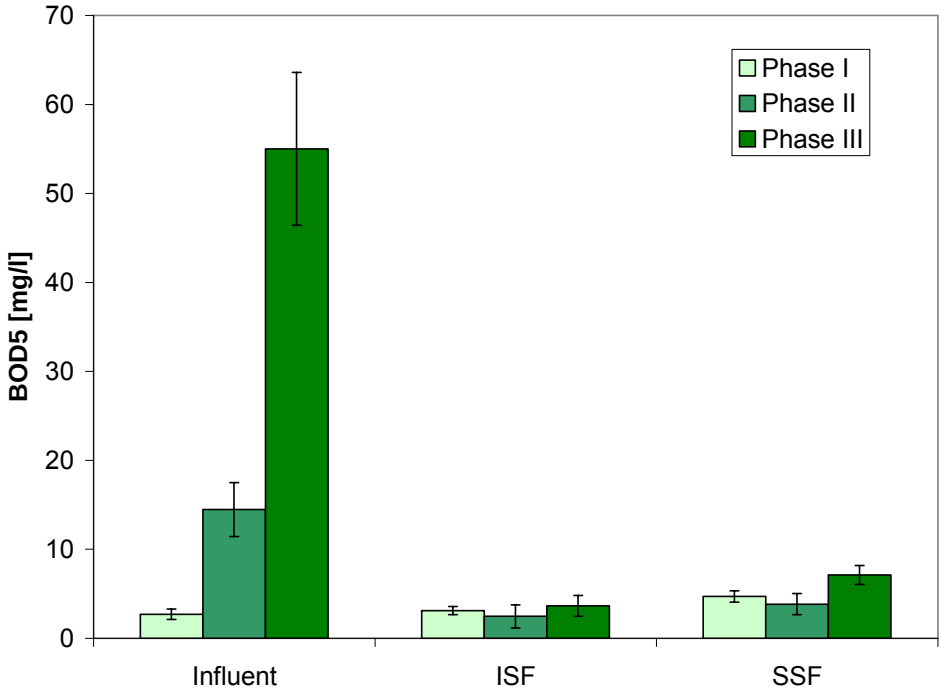


Figure 3-24: Arithmetic mean and standard error of the biochemical oxygen demand in the influent and effluent of the intermittent (ISF) and slow sand filter (SSF) during phase I–III of increasing oxygen demand and TSS concentration

Contrary to the hypothesis, easily biodegradable matter was only removed slightly better in the ISF. This can be explained by the fact that removal was primarily dependent on mechanical retention and not degradation. Filtration through a glass-fibre filter, as used for TSS measurement, proved that the dominant fraction of the BOD_5 and COD was constituted by particulate matter of which more than 98% was retained in the filters (Table 3-14). Finally a massive build-up of the schmutzdecke on the bed of the SSF and development of a filter cake on the bed of the ISF was noted during phases II and III. This supports the explanation that straining exceeded degradation. More oxygen demand was removed than oxygen was available not even considering the oxygen demand for nitrifying ammonium: At the average room temperature, saturated water may have contained a maximum of 8.7 mg/l dissolved oxygen whereas removal of BOD_5 in phases II and III amounted to 10.6 and 47.9 mg/l.

Table 3-14: Dissolved biochemical and chemical oxygen demand in mg/l and in percent of total BOD_5 and COD in the influent of the sand filters

	Phase	[mg/l]	[%]
Dissolved	I	0.7	25.9
BOD_5 [mg/l]	II	2.4	10.7
	III	7.1	10.5
Dissolved	I	29.7	84.0
COD [mg/l]	II	30.8	20.0
	III	30.2	6.9

Nitrate was increasingly used as an electron acceptor in the ISF when the availability of dissolved oxygen decreased from phase I to II. Nitrate concentrations in the effluent were comparable between phase II and III as were concentrations of dissolved oxygen. More nitrate was removed in the SSF during phase I and II. The fact that nitrate removal decreased with decreasing concentrations of dissolved oxygen in phases II and III may be caused by increased ammonification and nitrification. The latter is indicated by the presence of nitrite.

No ammonium was detected in the effluent of ISF and SSF during phase I. Concentrations of dissolved oxygen were sufficiently high to guarantee complete nitrification of ammonium released during microbial dissimilation of organic matter. The slight increase of ammonium concentration in the effluent of the ISF compared to the influent in phases II and III was attributed to ammonification of the retained and dissolved organic matter that was not completely followed by nitrification. Lower concentrations of ammonium in the effluent of the SSF compared to the ISF were unexpected because of the lower availability of dissolved oxygen. They were probably caused by the longer retention times of the water in the filter leaving more time for nitrification. Table 3-15 lists further relevant parameters of the influent and effluent of the intermittent and slow sand filter.

Table 3-15: Arithmetic mean and standard error of selected parameters in the influent and effluent of the intermittent (ISF) and slow sand filter (SSF) during phase I–III

	Phase	Influent	ISF	SSF
Temperature [°C]	I	4.9 ± 0.1	22.2 ± 0.4	22.2 ± 0.4
	II	6.3 ± 0.1	17.2 ± 0.1	17.2 ± 0.1
	III	7.4 ±0.1	17.3 ± 0.1	17.2 ± 0.2
pH	I	7.54 ± 0.05	7.55 ± 0.04	7.37 ± 0.09
	II	7.33 ± 0.06	7.08 ± 0.03	6.76 ± 0.02
	III	7.11 ± 0.05	7.02 ± 0.05	6.68 ± 0.05
Conductivity [µS/cm]	I	1256 ± 41	1247 ± 39	1233 ± 37
	II	1309 ± 33	1298 ± 33	1288 ± 31
	III	1228 ± 52	1249 ± 49	1209 ± 46
TOC [mg/l]	I	8.7 ± 0.3	7.2 ± 0.4	9.1 ± 0.4
	II	9.5 ± 0.3	7.5 ± 0.4	7.9 ± 0.3
	III	11.5 ± 0.6	8.3 ± 0.5	8.7 ± 0.3
$P0_4^{3-}$ [mg/l]	I	5.3 ± 2.6	3.0 ± 0.9	3.0 ± 1.0
	II	2.4 ± 0.8	3.6 ± 0.6	3.0 ± 0.7
	III	1.2 ± 0.5	5.3 ± 1.6	8.5 ± 0.8

3.4.5 Clogging and runtime

For comparison of ISF and SSF both filters were operated at a HLR of 0.65 m/d (about 0.03 m/h). While this is recommended as the upper limit for ISF, SSFs are usually operated at higher hydraulic loading rates of 0.05–0.4 m/h. Slow sand filters require maintenance when the schmutzdecke has developed to such an extent that filter resistance exceeds a value defined by the available increase in supernatant water level. However, the SSF did not require any maintenance during phase I to III. Although the schmutzdecke built up increasingly, filter resistance and supernatant water level did not increase. This was attributed to the low hydraulic loading rate that did not compress the schmutzdecke but allowed for a voluminous development leaving it permeable. Also rising gas bubbles, probably nitrogen developed during denitrification, lifted the schmutzdecke during phases II and III on several occasions.
On the other hand the ISF that typically does not exhibit clogging had to be maintained after 5 weeks of operation in phase III. It was not sufficient to scrape off the filter cake and wash the upper 2 cm of sand. The whole sand bed had to be removed and washed to restore hydraulic

conductivity. This clogging in the depth of the sand bed was probably caused by excessive development of biofilm due to elevated concentrations of substrate in the influent and favored by the intermittent operation. It is assumed that under the aerobic conditions in the ISF 50% of the substrate was assimilated by microorganism leading to biomass clogging the pore channels. Limited oxygen supply in the SSF resulted in less biofilm development and did not affect filter resistance, because under anoxic conditions only 10% of the substrate is assimilated and 90% dissimilated. The advantage of the SSF lies in the fact that if clogging occurs it is always located at the surface of the sand bed and it is therefore easy to restore hydraulic conductivity. Clogging of an ISF at technical scale would be very costly: either the filter needs to be taken off line for months to regenerate or the sand has to be taken out and washed completely.

The experiments performed to study the impact of elevated hydraulic loading rates on dissolved oxygen in the SSF were also evaluated with respect to runtime. Figure 3-25 shows that with increasing TSS surface loading runtime diminished. This increase was either due to higher TSS concentrations or hydraulic loading rates. No increase in supernatant water level was found for TSS surface loadings of 0.04–0.75 g/cm²h (HLR = 3 cm/h, TSS = 13–250 mg/l). The supernatant water level increased exponentially for surface loadings of 1.13 g/cm²h (HLR = 12 cm/h, TSS = 94 mg/l), 1.69 g/cm²h (HLR = 18 cm/h, TSS = 94 mg/l) and 3 g/cm²h (HLR = 12 cm/h, TSS = 250 mg/l).

Runtime increases with decreasing TSS surface loading (Figure 3-26). At low values such as 0.075 g/cm²h no clogging was observed during the experiments. Organic matter was mineralized faster than deposited. Separate evaluation of the influence of hydraulic loading rate and TSS concentration is recommended.

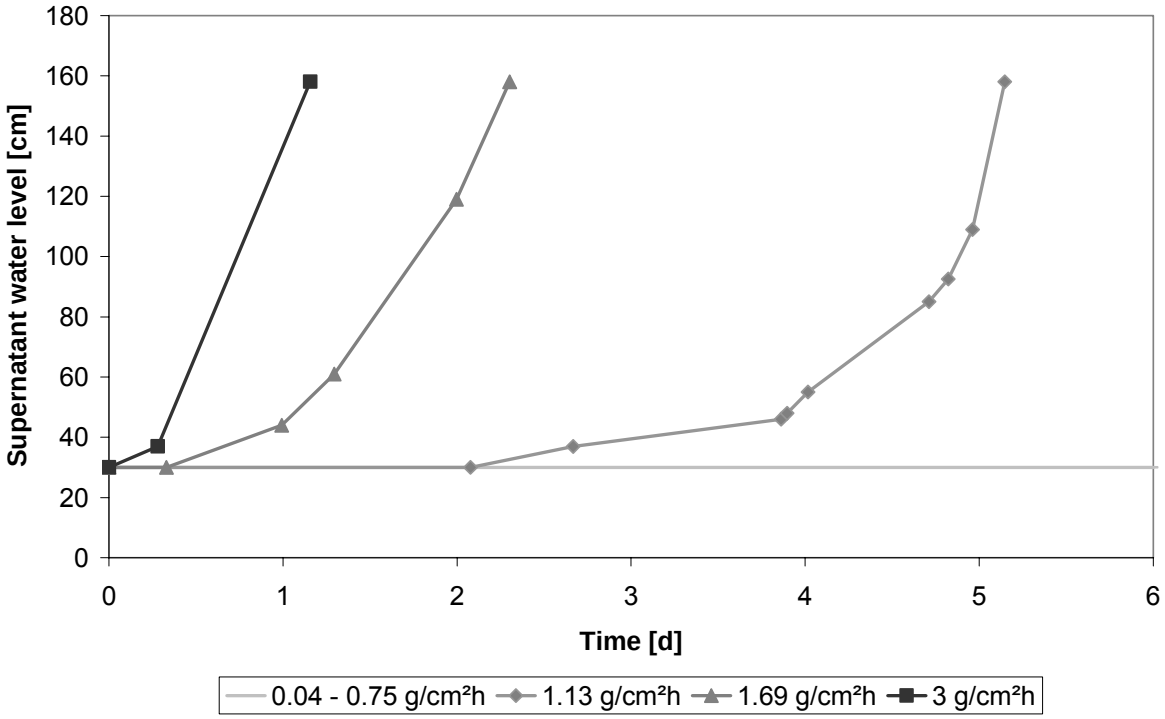


Figure 3-25: Increase in supernatant water level over runtime in a slow sand filter for TSS surface loadings of 0.04-3 g/cm²h

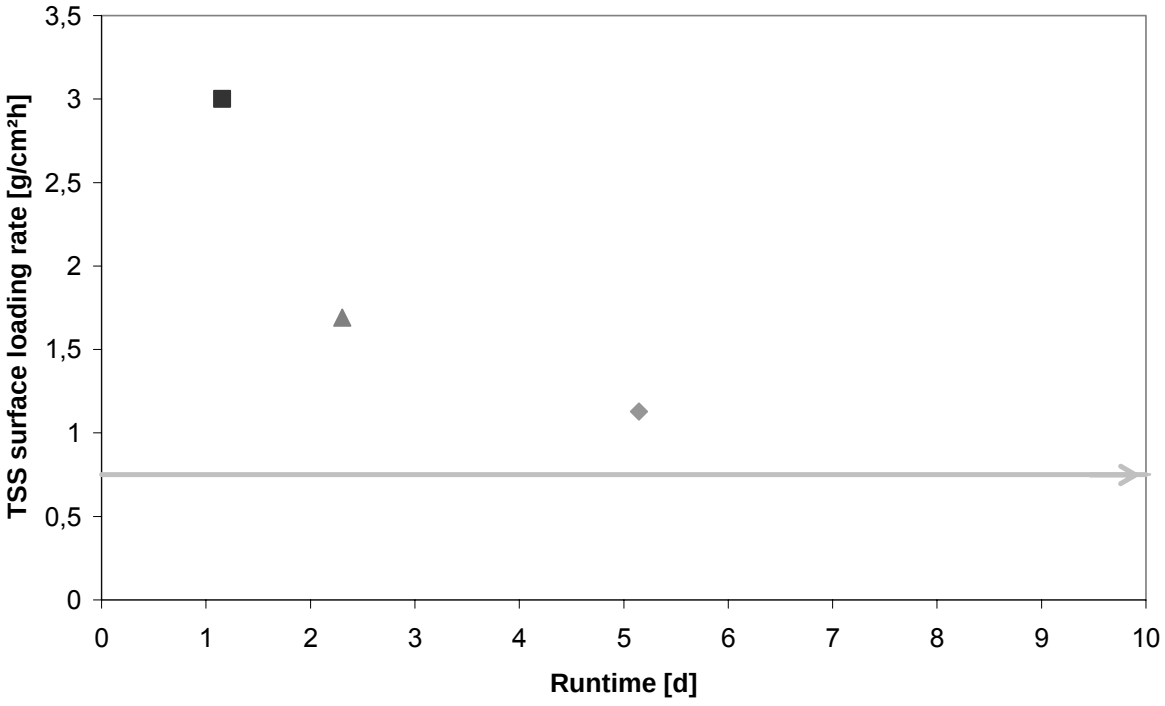


Figure 3-26: Runtime of a slow sand filter at different TSS surface loading rates

3.5 Rotating cascade of slow sand filters

Bacteria removal in a rotating cascade SSF consisting of four filter stages as described in section 2.1.5 was evaluated. It amounted to 1.71 log-units of *E. coli* and 1.86 log-units of intestinal Enterococci (Table 3-16).

Table 3-16: Cumulated log_{10}-removal of fecal indicator bacteria in the rotating cascade SSF at a HLR of 20 cm/h

	Cumulated log_{10}-removal	
	E. coli	Int. Enterococci
Stage 1	1.78	2.01
Stage 2	1.48	1.73
Stage 3	1.44	1.82
Stage 4	1.71	1.86

Unexpectedly bacteria removal slightly decreased after the first stage. Also, 90th-percentile concentrations increased after the first stage (Figure 3-27).

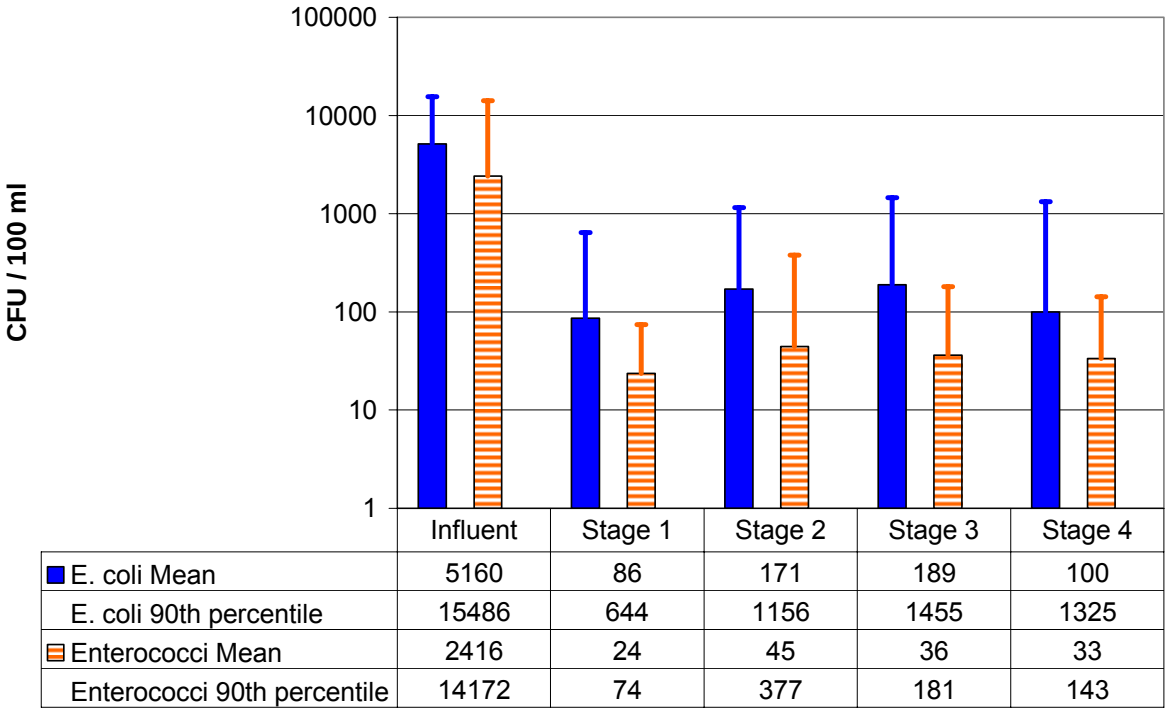


Figure 3-27: Geometric mean and 90th-percentile concentrations of fecal indicator bacteria in the influent and effluent of the stages of the rotating slow sand filter cascade

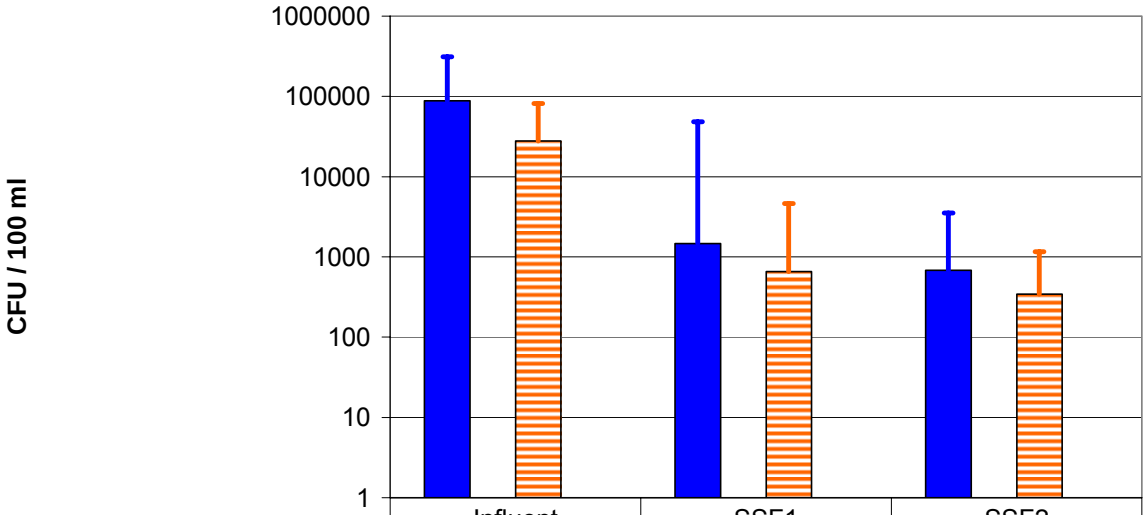
Bacteria removal in the first stage of the cascade was comparable to the removal in pilot filters S2 and S3. At a bed depth of 25 cm and operated at a HLR of 20 cm/h the filters removed 1.84 and 2.55 log-units of *E. coli* as well as 1.66 and 2.53 log-units of intestinal Enterococci. Bacteria removal had been shown to depend on the sand surface area. The specific surface area of the sand used in the cascade (4,804 m^2/m^3) was between the specific surface areas of S2 and S3. The rotating cascade did not fulfil the expectations of increased bacteria removal due to the existence of four schmutzdecke layers. Cumulated removal even decreased after the first stage. The rotation must have resulted in re-mobilisation and possibly regrowth of fecal bacteria that had been moved from the first to the last stage. Influent and effluent water of the filter cascade is further characterized by Table 3-17.

Table 3-17: Arithmetic mean and standard deviation of selected parameters in the influent and effluent water of the filter cascade

	Influent	Effluent Stage 4
TSS [mg/l]	4.2 ± 2.7	2.9 ± 2.0
TSS removal [%]		31
Turbidity [NTU]	3.8 ± 2.1	1.9 ± 1.2
Turbidity removal [%]		50
Dissolved oxygen [mg/l]	6.7 ± 1.4	2.6 ± 1.3
Temperature [°C]	19.1 ± 5.4	21.5 ± 4.6
pH	6.7 ± 0.2	6.9 ± 0.1
Redox-potential [mV]	179 ± 62	149 ± 40
Conductivity [µS/cm]	1,216 ± 144	1,302 ± 49
BOD_5 [mg O_2/l]	4.9 ± 3.0	4.1 ± 3.8
COD [mg O_2/l]	31.3 ± 6.7	24.1 ± 4.1
TOC [mg/l]	9.9 ± 1.3	9.2 ± 1.3
NH_4^+ [mg/l]	Below detection limit	
NO_3^- [mg/l]	27.3 ± 6.8	23.5 ± 4.9
PO_4^{3-} [mg/l]	2.2 ± 2.9	2.3 ± 1.2

3.6 Slow sand filtration for disinfection of constructed wetland effluent

The cascade achieved *E. coli* removal of 1.78 log-units in the first and 0.33 log-units in the second filter as well as intestinal Enterococci removal of 1.62 log-units in the first and 0.28 log-units in the second filter. This is comparable to typical removal in constructed wetlands (Hagendorf, 2002). A monitoring level of <1,000 CFU/100 ml of *E. coli* was achieved after the second stage (Figure 3-28). Average removal in the second stage was comparatively low. This may consistently be explained by the fact that bacteria removal per filter length declines with bed depth and that no schmutzdecke developed on the surface of the second stage, because it was nearly unchallenged by suspended solids. However, the second stage was effective in minimizing the 90th-percentile concentrations. Still, the effluent of the cascade did not comply with bathing water quality requirements according to EU-Directive.



	Influent	SSF1	SSF2
E. coli Mean	87640	1468	680
E. coli 90th percentile	311838	48499	3532
Enterococci Mean	27766	660	344
Enterococci 90th percentile	81945	4640	1165

Figure 3-28: Geometric mean and 90th-percentile concentrations of fecal indicator bacteria in the influent and effluent of the stages of the cascade of slow sand filters SSF1 and SSF2 treating effluent of a constructed wetland

No maintenance was required during the whole operation of the cascade. This was attributed to the low load of suspended solids. Characteristics of the influent and effluent water of the filter cascade are given in Table 3-18. Disinfection of the wastewater was achieved without loosing the valuable plant fertilizer nitrate. This can be seen as a major advantage of the SSF

compared to a horizontal flow constructed wetland operated in series after a vertical flow constructed wetland. Also, evaporation losses will be lower. In addition, the hydraulic loading rate of 5 cm/h (120 cm/d) is substantially higher than typical values of 120 mm/d applied for constructed wetlands. This would result in much smaller land requirements, especially considering that an increase in HLR to 20 cm/h can be expected to influence bacteria removal only slightly.

Table 3-18: Arithmetic mean and standard deviation of selected parameters in the influent and effluent water of the slow sand filter cascade treating effluent of a constructed wetland

	Influent	SSF 1	SSF 2
TSS [mg/l]	4.5 ± 4.0	1.5 ± 0.7	1.1 ± 0.7
Cumulated TSS removal [%]		67	76
Turbidity [NTU]	4.7 ± 4.0	1.4 ± 0.4	1.3 ± 0.4
Cumulated turbidity removal [%]		70	72
Dissolved oxygen [mg/l]	3.9 ± 0.7	2.6 ± 1.2	2.3 ± 0.7
Temperature [°C]	20.9 ± 3.8	21.1 ± 4.4	20.7 ± 3.8
pH	6.2 ± 0.1	6.3 ± 0.1	6.3 ± 0.1
Redox-potential [mV]	187 ± 32	174 ± 35	175 ± 39
Conductivity [µS/cm]	1,650 ± 232	1,653 ± 230	1,638 ± 212
BOD_5 [mg O_2/l]	12.6 ± 7.3	7.9 ± 3.5	6.6 ± 2.9
COD [mg O_2/l]	53.1 ± 10.4	40.9 ± 3.6	39.0 ± 3.7
TOC [mg/l]	19.2 ± 4.1	15.1 ± 1.5	14.8 ± 1.9
NH_4^+ [mg/l]	1.0 ± 2.5	Below detection limit	
NO_3^- [mg/l]	173.4 ± 71.0	179.6 ± 43.0	180.3 ± 41.1
PO_4^{3-} [mg/l]	16.1 ± 4.1	13.1 ± 2.3	13.8 ± 3.0

3.7 Modeling *E. coli* removal in slow sand filters

The purpose of this part of the work was to develop a model of *E. coli* elimination yielding simulation results comparable to the concentrations measured over the length of slow sand filters of different sand grain size distribution under three hydraulic loading rates.

In ponds, the elimination or inactivation of *E. coli* is usually modeled assuming first-order kinetics (Von Sperling, 2005). The reaction rate coefficient (k) can be calculated as the difference of the log-normal of concentrations (c_0 and c) per time difference ($t - t_0$) or can be read as the slope of log-normal concentrations plotted over time.

$$\frac{dc}{dt} = -k \cdot c \Rightarrow \frac{\ln c_0 - \ln c}{t - t_0} = k \qquad \text{Equation 3-2}$$

For the secondary clarifier effluent used in this study, reaction rate and coefficients of *E. coli* elimination can be seen in Figure 3-29. The decline of indicator bacteria concentrations was measured in four samples of secondary clarifier effluent from the WWTP Langenreichenbach (see section 2.3). In three cases elimination closely followed first-order kinetics (coefficient of determination >0.89) and rate coefficients ranged from 0.44 to 1.5. The arithmetic mean of 1.09 and standard deviation of 0.4 were used in the model.

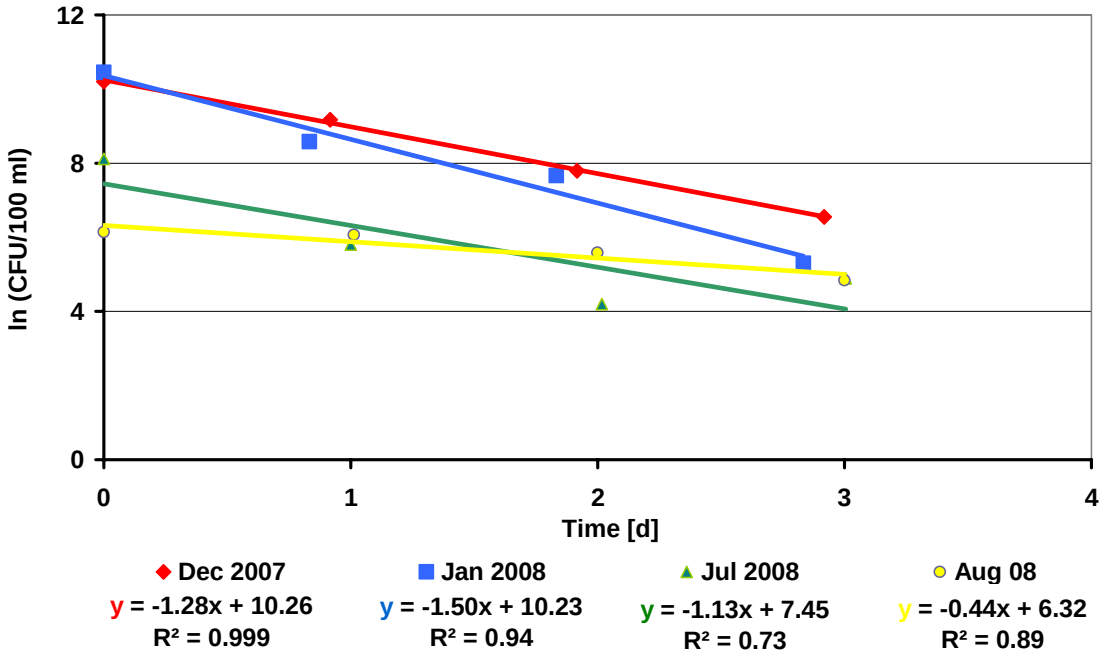


Figure 3-29: Determination of the reaction order and rate constants for *E. coli* elimination / inactivation by plotting ln-transformed concentrations measured over time in samples of secondary clarifier effluent

The model and parameters described were applied to simulate *E. coli* concentrations as a function of filter length in filters S1 and S4 for hydraulic loading rates of 5, 10 and 20 cm/h. The values for all variables used in the model are summarized in Table 3-19.

Table 3-19: Overview of the values of all variables used in the model of slow sand filtration

		Phase I (HLR = 5 cm/h)		Phase II (HLR = 10 cm/h)		Phase III (HLR = 20 cm/h)	
Filter		S1	S4	S1	S4	S1	S4
C_EC_INF [CFU/100 ml]		10,082 (10^{th}-Perc.: 3,819) (90^{th}-Perc.: 26,616)		4,380 (10^{th}-Perc.: 1,341) (90^{th}-Perc.: 14,307)		4,323 (10^{th}-Perc.: 928) (90^{th}-Perc.: 20,133)	
k_EC [1/d]		1.09 ± 0.4					
eps_Sand		0.4					
eps_SD		0.8 ± 0.1					
Q [m³/d]		0.033		0.059	0.066	0.129	0.127
sand_surf [m²/m³]		10,388	3,228	10,388	3,228	10,388	3,228
Factor_EC_SD		126 ± 68		113 ± 64		465 ± 191	
Factor_EC [m³/m²] at bed depth	5 cm	0.006 ± 0.0034		0.0123± 0.0033		0.01 ± 0.0058	
	10 cm	0.0063 ± 0.0013		0.0177 ± 0.0123		0.0041 ± 0.0004	
	25 cm	0.0038 ± 0.0039		0.006 ± 0.0056		0.01 ± 0.0033	

Table 3-20 and Table 3-21 show in detail how the retention factors were determined for phase I of the experiments. The Factor_EC was calculated as the arithmetic mean of retained *E. coli* per mobile *E. coli* and sand surface area in the three horizons of filter C1–C3 for example.

Table 3-20: Determination of the retention factors in the sand bed of slow sand filters C1–C3

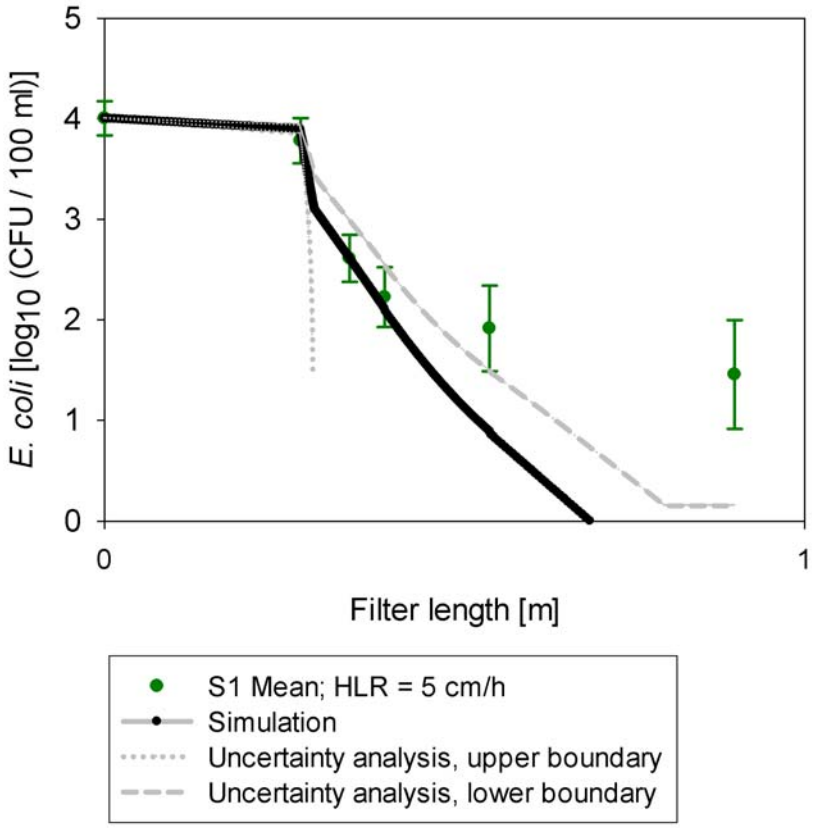
Filter	Bed depth [cm]	Retained *E. coli* [CFU/ 100 ml]	*E. coli* in mobile Phase [CFU/ 100 ml]	Retained *E. coli* per mobile *E. coli*	Spec. sand surface area [m^2/m^3]	Retained *E. coli* per mobile *E. coli* and sand surface area
C1	5	5,385	185	29	10,388	0.0028
	10	4,740	77	61	10,388	0.0059
	25	2,917	36	82	10,388	0.0079
C2	5	7,011	189	37	6,474	0.0057
	10	2,804	55	51	6,474	0.0078
	25	1,246	54	23	6,474	0.0036
C3	5	16,584	412	40	4,226	0.0095
	10	5,853	261	22	4,226	0.0053
	25	0	172	0	4,226	0.0000

Table 3-21: Determination of the retention factors in the schmutzdecke of slow sand filter C4

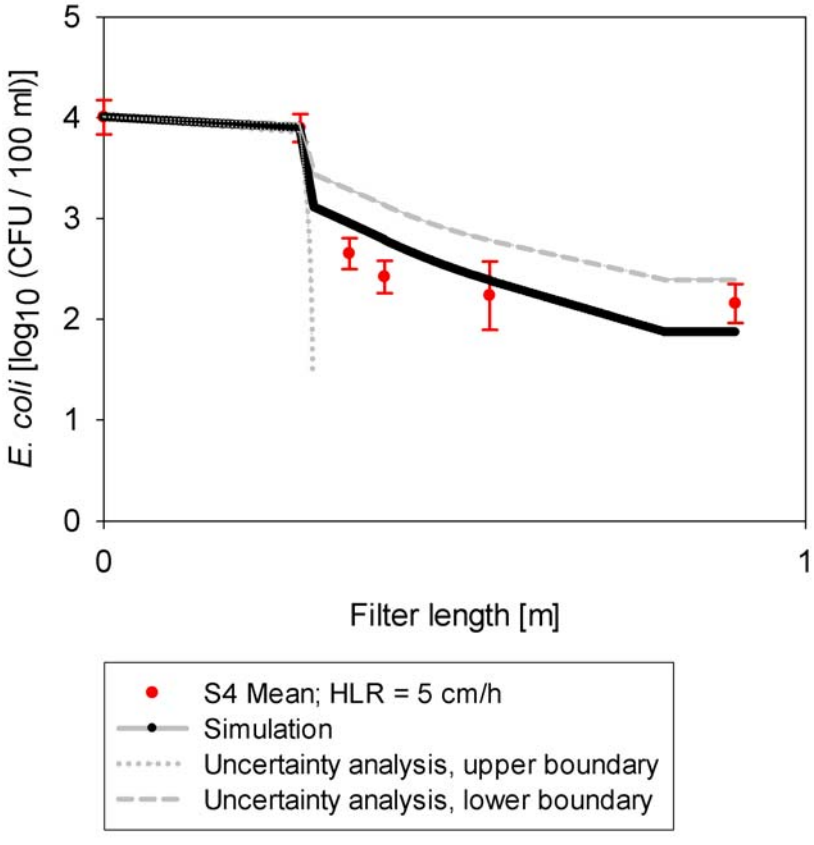
Filter horizon	Triplicate measurement	*E. coli* [CFU/ 100 ml]	*E. coli* in the schmutzdecke per *E. coli* in the adjacent supernatant water
Schmutzdecke	Mean	136,667	126
	Standard deviation	73,711	68
Supernatant water 2 cm above the sand bed	Mean	1,084	

Figure 3-30 shows calculated mean values with 95%-confidence intervals of measured concentrations of *E. coli* compared to simulation results with boundaries limiting the range of results obtained from mean values of the parameters plus and minus one standard deviation (uncertainty analysis) as a function of filter length at a HLR of 5 cm/h (a, b), 10 cm/h (c, d), 20 cm/h (e, f) as well as simulated and calculated 90^{th}-percentile and 10^{th}-percentile concentrations at a HLR of 5 cm/h (g, h).

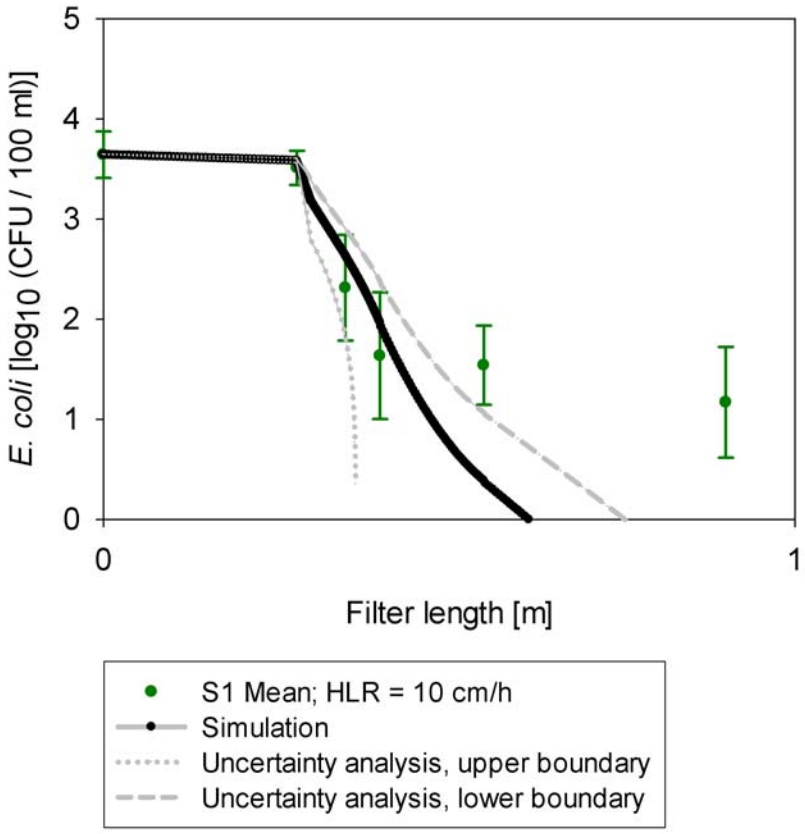
a)


5
4
3
2
1
0
0
1
E. coli [log10 (CFU / 100 ml)]
Filter length [m]
S1 Mean; HLR = 5 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

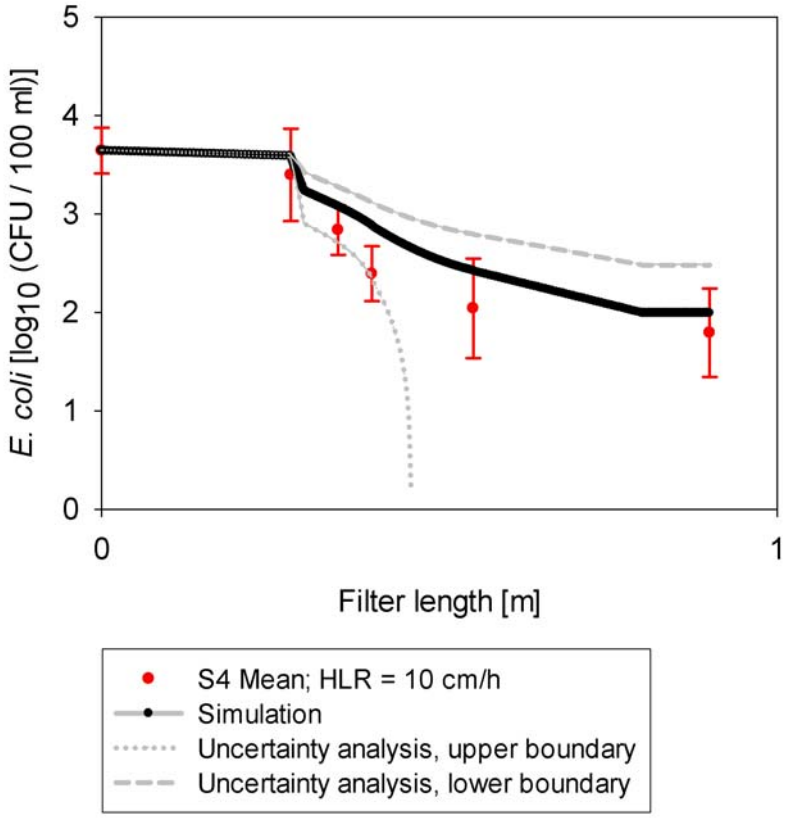
b)


5
4
3
2
1
0
0
1
E. coli [log10 (CFU / 100 ml)]
Filter length [m]
S4 Mean; HLR = 5 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

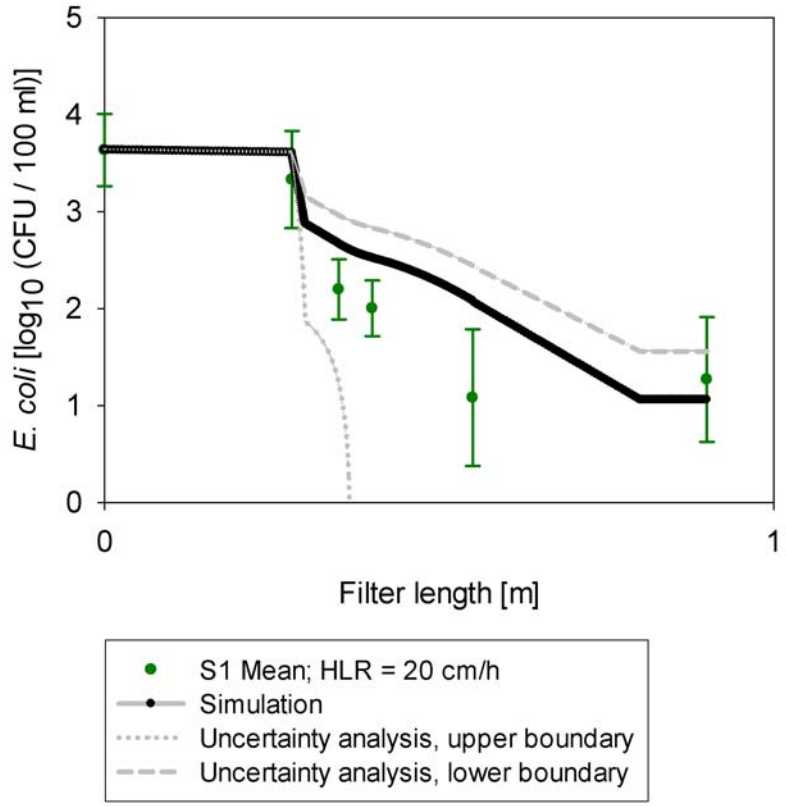
c)


E. coli [log10 (CFU / 100 ml)]
5
4
3
2
1
0
0
1
Filter length [m]
S1 Mean; HLR = 10 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

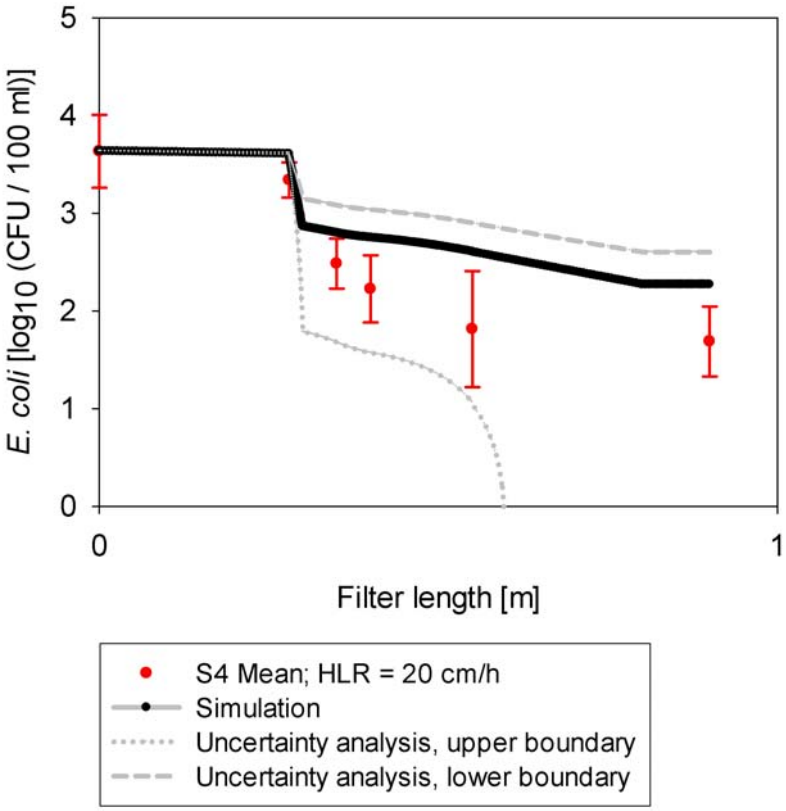
d)


E. coli [log10 (CFU / 100 ml)]
5
4
3
2
1
0
0
1
Filter length [m]
S4 Mean; HLR = 10 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

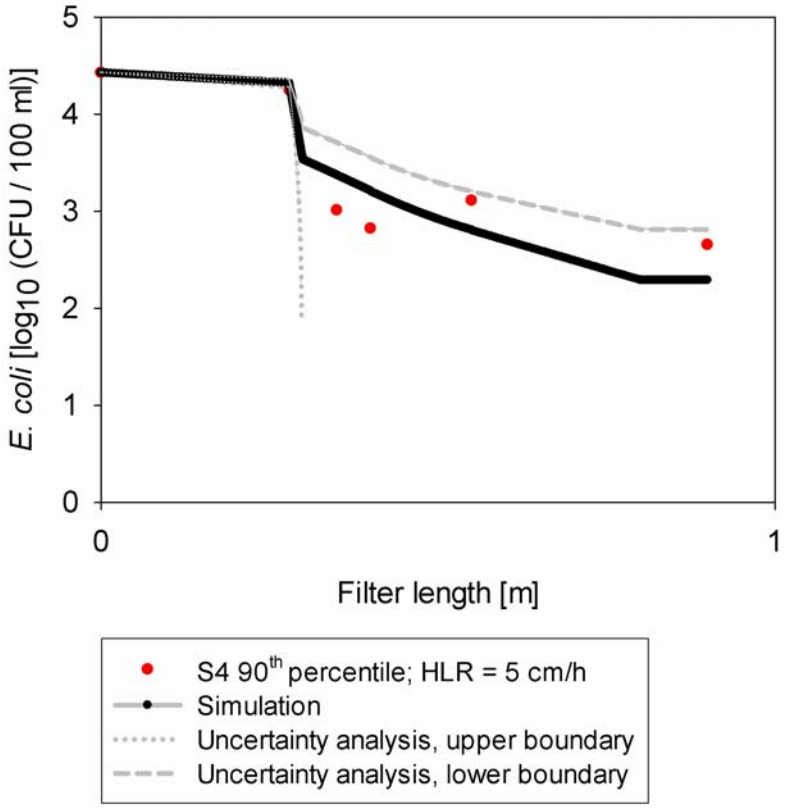
e)


E. coli [log10 (CFU / 100 ml)]
5
4
3
2
1
0
0
1
Filter length [m]
S1 Mean; HLR = 20 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

f)


E. coli [log10 (CFU / 100 ml)]
5
4
3
2
1
0
0
1
Filter length [m]
S4 Mean; HLR = 20 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

g)


E. coli [log10 (CFU / 100 ml)]
5
4
3
2
1
0
0
1
Filter length [m]
S4 90th percentile; HLR = 5 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

h)

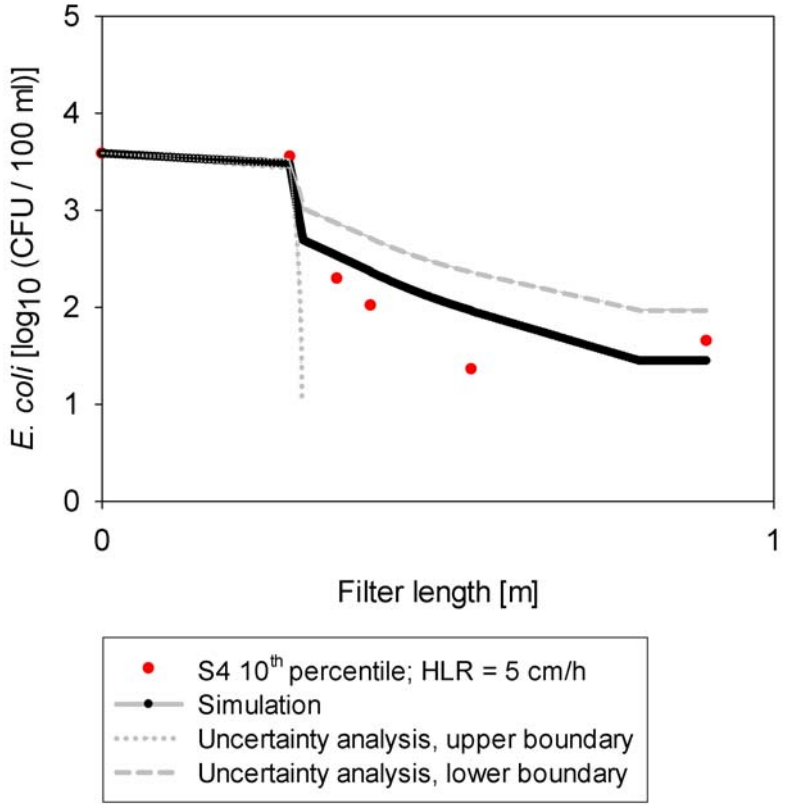

E. coli [log10 (CFU / 100 ml)]
5
4
3
2
1
0
0
1
Filter length [m]
S4 10th percentile; HLR = 5 cm/h
Simulation
Uncertainty analysis, upper boundary
Uncertainty analysis, lower boundary

Figure 3-30: Calculated mean values with 95%-confidence intervals of measured concentrations of *E. coli* compared to simulation results with boundaries limiting the range of results obtained from mean values of the parameters plus and minus one standard deviation (uncertainty analysis) as a function of filter length in slow sand filters S1 and S4 at a HLR of 5 cm/h (a, b), 10 cm/h (c, d), 20 cm/h (e, f); simulated and calculated 90^{th}-percentile and 10^{th}-percentile concentrations at a HLR of 5 cm/h (g, h)

Agreement between simulated concentrations and those calculated from experimental data was found to be satisfactory if confidence intervals overlapped with the corridor of the uncertainty analysis. For S1 this was the case for 12 out of 15 confidence intervals, considering all hydraulic loading rates and excluding the starting points at 0 m filter length. Furthermore, the simulation lay within 8 of 15 confidence intervals. In the case of S4, all confidence intervals overlapped with the corridor generated by the uncertainty analysis and the simulation lay within 7 out of 15 confidence intervals. So the model, its assumptions and simplifications were generally acceptable. It can be concluded that bacteria elimination followed a first order reaction depending on bacteria concentration in the mobile bulk phase as well as the concentration of immobilized bacteria retained in the schmutzdecke and within the biofilm attached to the sand surface.

It is evident from the experimental data in Figure 3-30 that filter S1 removed more bacteria from secondary clarifier effluent than S4. Since both filters approximately had the same volume of the bulk phase due to similar porosity and there was no evidence of substantial differences in their schmutzdecke, the main reason causing this observation must have been the sand surface in the filter bed. The model accounted for this hypothesis by describing the concentration of immobilized bacteria in the filter bed as a function of specific surface area. The model also respected declining bacteria elimination per filter length in deeper zones of the sand bed by making the retention factor a function of bed depth (see Table 2).
The simulation results for S1 were not satisfactory at a hydraulic loading rate of 5 cm/h and a sand bed depth of 50 cm (corresponding to 90 cm filter length) as well as at a HLR of 10 cm/h and the sand bed depths of 25 cm and 50 cm. But for the same hydraulic loading rates, the simulation results for supernatant water, schmutzdecke and upper 10 cm of the sand bed are nearly identical to the calculated confidence intervals. So we can state, that the model does not exhibit a systematic weakness. Rather, the Factor_EC needs to be measured repeatedly over a longer time. The value had been determined by triplicate measurements of bacteria in

shake-off suspensions of samples extracted from three sand bed depths of 3 (phase I) or 4 (phase II) filter columns. The calculated relative standard deviations for Factor_EC were quite high ranging from 21% to 103% and the Factor_EC did not always decrease with increasing bed depth as expected (see Table 3-19). Highest standard deviations of 85% and 103% were measured at bed depths of 25 cm at hydraulic loading rates of 5 and 10 cm/h. In addition, the model used linear interpolation to determine the retention factors for the whole depth of the filter bed of up to 50 cm relying upon measurements at 5, 10 and 25 cm depth. This overestimated the retention factors and no samples could be taken from 50 cm depth. These facts may explain the unsatisfactory simulation at deep bed depths for S1. The impreciseness is multiplied by a specific surface area three times higher in the case of S1 compared to S4.

Bacteria removal did not substantially decrease with increasing HLR. This is in accordance with the findings of Huisman and Wood (1974) for treatment of surface water. The missing decrease can be explained by the much higher concentration of bacteria in the immobile phase (biofilm) in comparison to the bulk phase. A reduced theoretical hydraulic retention time as a result of an elevated HLR did not affect elimination of immobilized bacteria. In addition reduced retention time between HLRs of 5 and 10 cm/h was compensated by an increase in the retention factor (Table 3-19). In the sand bed, the retention factor (Factor_EC multiplied by specific surface area) was 61 on average ranging between 0 and 187. Comparable enrichment of fecal coliforms by factors of 51 to 220 between bulk phase and biofilm was found in an artificial stream system (Schultz-Fademrecht et al., 2008). An increase in Factor_EC with increasing HLR seems plausible, because more substrate can be transported deeper into the filter bed. This favors development of biofilm that in turn may lead to improved straining and adsorption of bacteria. However, the factor decreased between the HLR of 10 and 20 cm/h. This may be the reason, why the simulated *E. coli* concentrations in the sand bed were higher than the mean values measured at 20 cm/h.

The substantial contribution of the schmutzdecke and the upper 5 cm of sand towards bacteria removal can clearly be demonstrated by the data in Figure 3-30. It is reflected by the high retention factors in the schmutzdecke (Table 3-19). Compared to the surrounding bulk phase, concentrations of immobilized bacteria were higher by a factor of 113–465 on average. Higher retention factors in the schmutzdecke than in the sand bed are plausible, because less mechanical straining is expected to occur in the pore channels of the sand in comparison to the schmutzdecke. In addition, it can be considered that *E. coli* adsorbs much better to the schmutzdecke composed of 90% organic material than to the inorganic sand grain surface. A doubling of HLR was expected to result in a higher retention factor, because doubling the

particle load leads to an increase in the thickness of the schmutzdecke. This will be reflected by a higher retention factor due to the experimental procedure for determining the concentration of immobilized bacteria in the layer of the schmutzdecke. So the approximate quadruplication in Factor_EC_SD from 126 for a HLR of 5 cm/h to 465 at 20 cm/h is plausible. The value of 113 at 10 cm/h falls short of the expectations. More samples from several filters are needed for a further discussion.

A first order reaction for modeling the elimination of bacteria in the SSF was successfully applied. The reaction rate constant determined for *E. coli* removal kinetics in the secondary clarifier effluent was used throughout all zones of the filter. So biotic and abiotic factors in the filter do not seem to cause faster elimination or inactivation of *E. coli* than in the secondary clarifier. This leads to the conclusion that the function of the SSF is primarily retention and not the creation of environmental conditions that are more hostile to fecal bacteria than in the secondary clarifier. Meanwhile the applicability of the same reaction rate coefficient for the secondary clarifier and the whole filter suggests that the concentration of predators in the SSF, especially in the schmutzdecke, must be much higher than in the secondary effluent. Bacteria removal in the supernatant water was not faster than in the secondary clarifier. Therefore, the main role of the supernatant water seems to be the protection of the schmutzdecke from shear forces caused by inflowing water. Faster elimination from the schmutzdecke seems possible since many predators or lytic microorganisms such as Bdellovibrio require minimum concentrations of prey which are only present in the schmutzdecke (Wand et al., 2007). Lower rate coefficients in the sand bed are also conceivable, since bigger predators might not be able to enter the pore channels. More detailed investigations of bacteria elimination in samples from the schmutzdecke and other horizons of the filter are needed to determine reaction rate coefficients more closely.

The simulation results for the 10^{th}- and 90^{th}-percentile in Figure 3-30 g-h show that the model is suited for variable concentrations of *E. coli* in the filter influent. All values lie within the corridor of the uncertainty analysis and are scattered closely above and below the plot of the simulation. This justifies the decision to correlate the concentration of retained bacteria with their concentration in the surrounding bulk phase.

The corridors of up to 2.5 log-units bacteria removal determined by uncertainty analysis depict how heavily the variability of some parameters defined in the model affected the simulation result. Sensitivity analysis in AQUASIM showed that the retention factors and reaction rate coefficient most strongly influenced bacteria removal. Slow sand filtration is a process that does rely on biological mechanisms and is thus less determinable than a physical

process such as membrane filtration. High variations are also reflected by the results on slow sand filtration of surface water in the literature that commonly cites bacteria removals between 2 and 4 log-units (Hendricks, 1991; Huisman, 2004). It has been strongly recommended to conduct pilot-studies prior to establishing a slow sand filter for drinking water purification if in-depth experience in the region is lacking (Ellis, 1985). Some variability may be reduced by determining the retention factors repeatedly. Other parameters like the reaction rate coefficient are likely to vary seasonally depending on composition of the secondary effluent and are expected to decrease with decreasing temperature. To narrow down the corridor of uncertainty and improve the simulation, the dependency could be incorporated into the model after extensive determination of the reaction rate in the schmutzdecke and the bulk phase at different temperatures.

It has to be admitted that the model described has limited potential as a tool to predict filter performance. Its main contribution is a quantitative description of the most relevant processes leading to bacteria removal in slow sand filters. It may allow comparison of experimental data from SSF of secondary clarifier effluent obtained under various ambient, design and operating conditions. This would further ensure understanding of the filtration process and could lead to a database of retention factors and reaction rate coefficients to be used in predictive modeling of filter performance. The model could also be applied to bacteria removal from surface water in drinking water purification with SSFs .

4 CONCLUSIONS

The presented work investigated slow sand filtration of secondary clarifier effluent. This simple process is usually used in drinking water purification and published data on its application to wastewater disinfection are rare. Results showed fecal indicator bacteria removal of 1.9–3 log-units. Pilot scale slow sand filter effluents reached bathing water quality according to the EU Directive and met a monitoring level of <100 *E. coli* / 100 ml in all but two configurations. This is the first study that analyzed removal of fecal indicator bacteria *E. coli* and intestinal Enterococci relevant for water reuse and systematically varied the key process parameters sand grain size distribution, filter bed depth and hydraulic loading rate. Although a simple technology, the processes in slow sand filters and their interactions are highly complex. Removal was shown to primarily depend on sand surface area as determined by sand grain size distribution and filter length, the schmutzdecke and supernatant water. The hydraulic loading rate as the fourth major parameter of design and operation only had a limited impact. Experimental data that have been obtained under different process parameters by other authors become better comparable, because the impact of sand grain size distribution and bed depth is summarized in the sand surface area. From the experience in drinking water purification and the conducted experiments it is recommended that sand surface area per filter surface should not be chosen below 2,000 m²/m², if bacteria removal is intended. Most efficient zones of bacteria removal were the schmutzdecke and upper 5 cm of the sand bed.

Elevated hydraulic loading rates of 20 cm/h adversely affected bacteria removal only slightly compared to lower HLRs of 5 cm/h. Clogging might occur more frequently, but more long-term data are needed to discuss this aspect in depth. Elevated hydraulic loading rates are economically promising: Depending on local cost of labor, smaller systems with elevated HLRs may compensate more frequent clogging by lower investment costs, making them more appropriate and easier to implement.

The sands examined in this study represent a wide range and include a typical 0/2-building sand that can be cheaply obtained almost anywhere. The disadvantage of small grain sizes can lie in the fact that clogging occurs more often while grains too large may lead to transport of particles and clogging in the depth of the filter. In the experiments clogging events did not follow a plausible pattern, probably because of the limited amount of incidents during the total runtime of 295 days. During this time the frequency of clogging was between two and five times. The choice of sand is then mainly an economic consideration. A wide range of sands can be used so that sieving, availability and price do not become limiting factors.

Nevertheless, pilot plant studies are recommended before designing a SSF for disinfection of secondary effluent because this biological system, especially the schmutzdecke, may vary in performance.
The pilot scale slow sand filters removed 70–84% of total suspended solids from secondary clarifier effluent. Mean filter effluent concentrations of 1.2–2.3 mg/l were significantly lower than influent concentrations ranging from 4.4–14.3 mg/l. Neither sand grain size distribution nor hydraulic loading rate had a considerable impact on TSS removal. Therefore the filter sand should be chosen according to the above recommendations.

A model was developed for bacteria removal from secondary clarifier effluent in slow sand filters. It is the first model of bacteria removal in SSF treating secondary effluent that was successfully used to simulate *E. coli* removal in filters of variable sand grain size and under a range of hydraulic loading rates. Simulation results were evaluated with respect to data on *E. coli* removal obtained from pilot-scale filters. The most important process was retention of bacteria at the schmutzdecke and sand surface. Immobilization in the schmutzdecke and sand bed was defined as a function of the concentration of bacteria in the surrounding bulk phase as well as the specific surface area and depth of the sand bed. In the sand bed, bacteria enrichment by a factor of 61 on average was found by comparing immobile and surrounding bulk phase. In the schmutzdecke, straining and adsorption led to a concentration of immobile *E. coli* by a factor of 113–465 higher than in the surrounding bulk phase. Overall, bacteria elimination and inactivation followed a first order kinetic. The reaction rate in the slow sand filter was not faster than in the secondary clarifier. So the creation of a hostile environment for fecal bacteria does not seem to be the main role of the SSF.
An increase in hydraulic loading rate did not lead to a substantial decrease in bacteria removal. This effect can be explained by the much higher concentration of bacteria in the immobile phase in comparison to the bulk phase. A reduced theoretical hydraulic retention time as a result of an elevated HLR did not affect elimination of immobilized bacteria.
The model allows to better compare fecal indicator bacteria removal from secondary effluent in slow sand filters operated under a variety of process parameters. It has enhanced understanding of the processes of retention and elimination leading to bacteria removal. The model has a potential as a tool for prediction of filter performance.

In this study the performance of a slow sand filter was compared to an intermittent sand filter under low and challenging influent concentrations in terms of BOD_5 (55 mg/l) and

ammonium (8.9 mg/l). The hypothesis that fecal indicator bacteria removal in the SSF would collapse due to high oxygen demand of the feeding water was not validated. The SSF eliminated 2.7–3.4 log-units or 99.89–99.95% of *E. coli* and 3.0–3.2 log-units or 99.88–99.96% of intestinal Enterococci. The ISF removed less bacteria: 1.8–2.5 log-units or 98.42–99.68% of *E. coli* and 1.6–2.8 log-units or 97.70–99.81 % of intestinal Enterococci. In the conducted experiments the SSF achieved comparable or higher bacteria removal than the ISF. A major advantage of the SSF compared to the established technology of ISF is the elevated hydraulic loading rate that can be used. Another advantage of the SSF lies in the fact that if clogging occurs it is always located at the surface of the sand bed and it is therefore easy to restore hydraulic conductivity.
In order to improve the availability of oxygen in the slow sand filter, the hydraulic retention time should be decreased by increasing the hydraulic loading rate. Also the filter bed depth and height of supernatant water should be minimized to achieve short detention times.

The comparison of filters at laboratory and pilot scale fed with secondary clarifier effluent of the same wastewater treatment plant and operated at the same hydraulic loading rate showed that scale up did not significantly affect bacteria removal. The fecal indicator bacteria removal of about 2 log-units in the larger pilot-scale filter was sufficent to reach bathing water quality in the effluent. A monitoring level of 100 CFU/100 ml of *E. coli* was just not reached (geometric mean of 107 CFU/100 ml in the filter effluent). During a period of operation of 170 days the filter had to be maintained only once.
The material cost of about 40 € for the whole filter including barrel, sand, tubes and all other elements was extremely low considering that the filter may disinfect 4 people equivalents or 500 liters per day. The unit may be beneficially applied for any pilot testing undertaken to evaluate the potential of SSF for disinfection of secondary effluent at a specific setting. It may also serve as the basis of a modular, effective and cheap system to disinfect effluent of decentralized wastewater treatment plants.

Bacteria removal from secondary clarifier effluent in a rotating cascade of SSFs only amounted to 1.71 log-units of *E. coli* and 1.86 log-units of intestinal Enterococci. The performance of this system in comparison to the regular filters did not justify the more sophisticated setup and operation. Although the setup consisted of a series of four filters, each with a schmutzdecke, it was mainly the primary section of the cascade and its schmutzdecke that achieved removal.

Two slow sand filters in series treating effluent of a vertical flow constructed wetland achieved *E. coli* removal of 1.78 log-units in the first and 0.33 log-units in the second filter as well as intestinal Enterococci removal of 1.62 log-units in the first and 0.28 log-units in the second filter. This is comparable to typical removal in constructed wetlands. A monitoring level of <1000 CFU/100 ml of *E. coli* was achieved after the second stage. Average removal in the second stage was comparatively low. However, the second stage was effective in minimizing the 90^{th}-percentile concentrations. Still, the effluent of the cascade did not comply with bathing water quality requirements according to EU-Directive.

The results suggest that slow sand filters may serve as a tertiary treatment step to achieve the requested microbiological quality for wastewater reuse and bathing water. Slow sand filtration has a vast potential as a simple technology for disinfection of wastewater. Compared to other natural reclamation technologies, land requirements are low because of elevated hydraulic loading rates of 4.8 m/d as in the experiments. A further increase of HLR seems promising. SSFs can achieve comparable or better *E. coli* removal than constructed wetlands. Their by far lower surface area requirements make SSFs an interesting alternative to replace the second stage of constructed wetlands operated in series to achieve hygienic standards of wastewater reuse. If reuse in agriculture is intended, SSF will also prove beneficial because nitrate removal is expected to be very low compared to horizontal flow constructed wetlands. Also maturation ponds or intermittent sand filters when used for disinfection might be replaced by slow sand filters. Evaporation losses in SSFs will be significantly lower than in ponds or the loss by evapotranspiration by the plants in constructed wetlands. Their advantages of lower evaporation loss as well as lower investment costs due to less land requirement need to be outweighed against the expenses of regular maintenance they require. Slow sand filters are expected to be higly effective in removing helminth eggs from wastewater due to straining.

Research on the disinfection performance of SSF using the effluent of secondary treatment systems like waste stabilization ponds, trickling or anaerobic filters is encouraged to develop a complete process for reclamation of wastewater using simple technology. Filter performance should also be measured using pathogens as well as indicator organisms and include virus and protozoa removal.

Slow sand filtration is a simple, low cost, appropriate and robust technology. It uses local material, skill and labor and does not require chemicals, sophisticated spare parts and energy,

provided there is a slope. SSFs are especially suited for warm climates. They may significantly contribute to safely reuse wastewater, especially in arid developing countries in order to mitigate water stress and scarcity.

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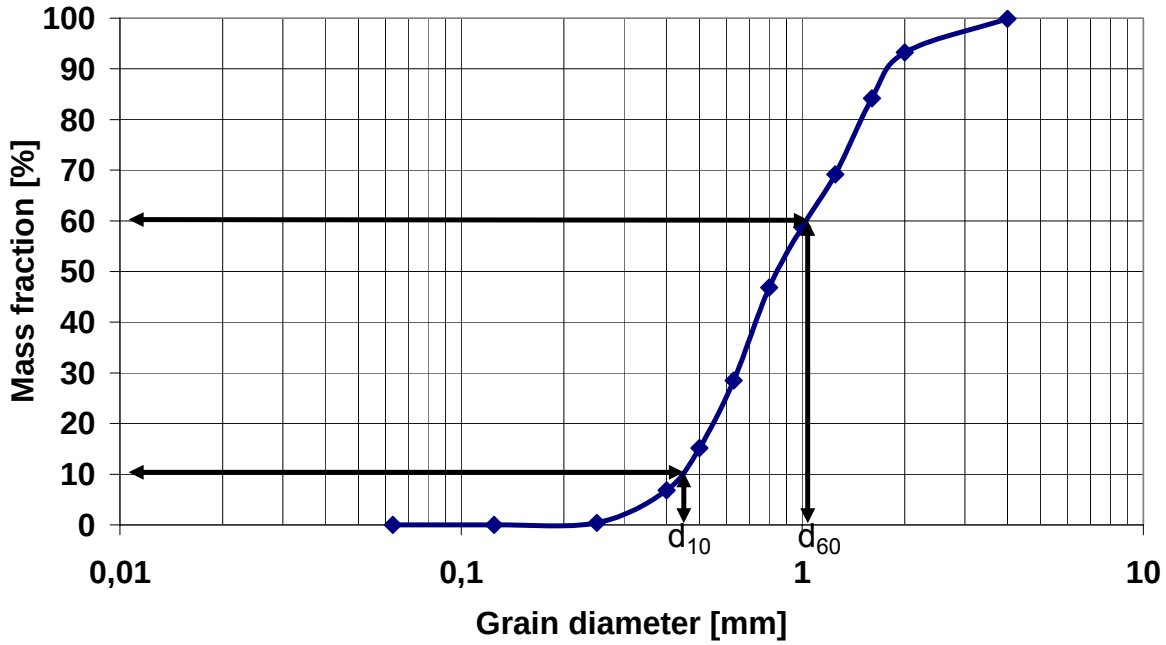
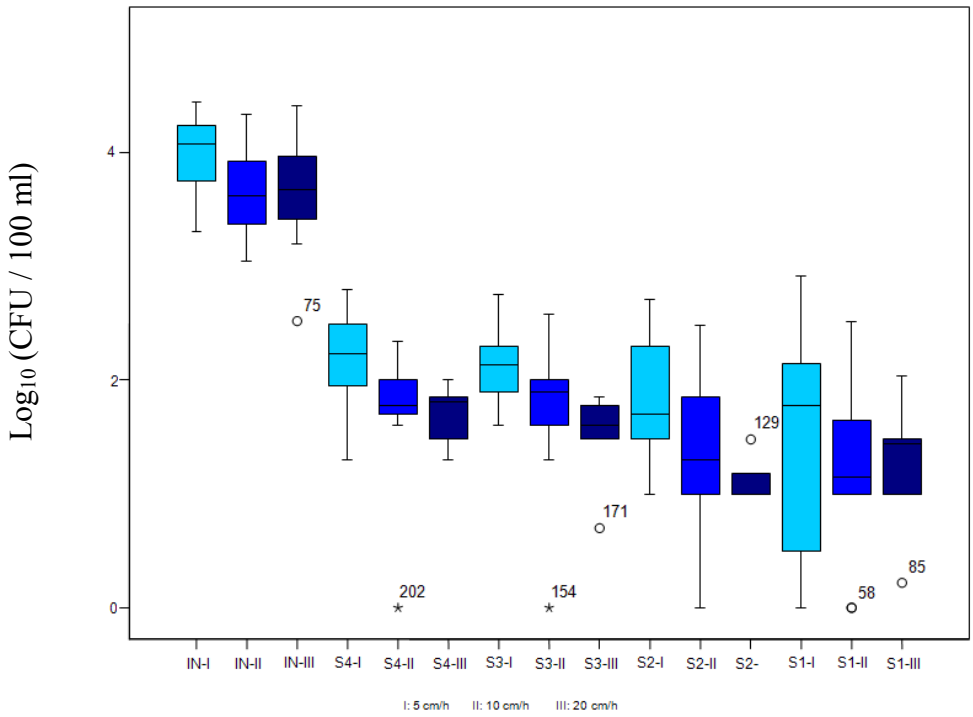
A. Annex

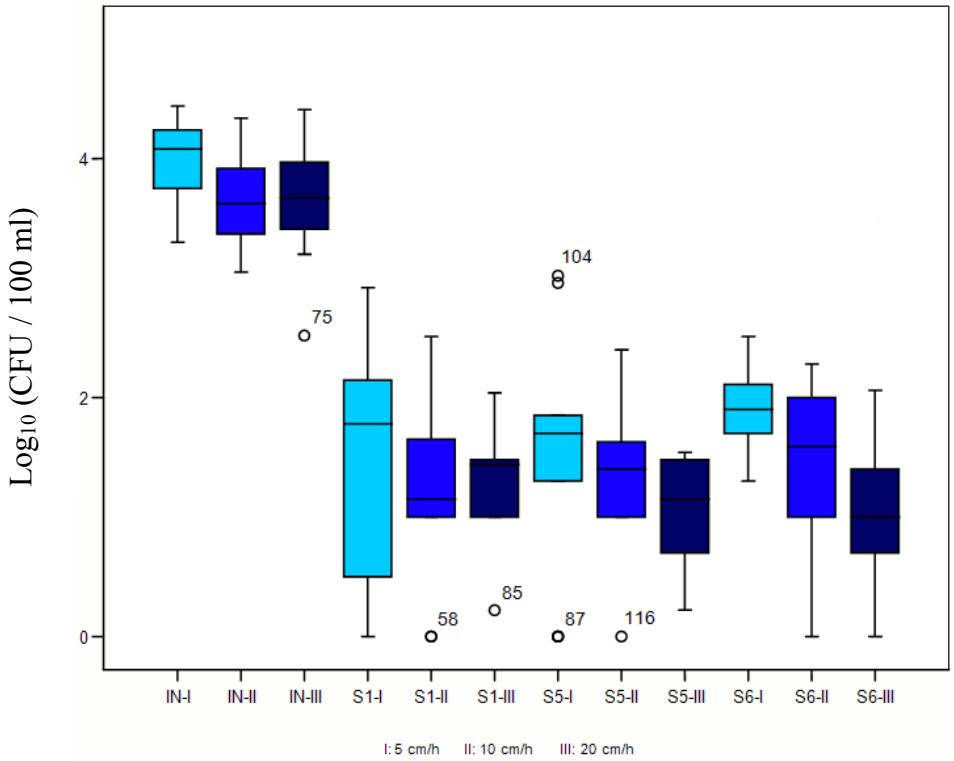


A-1: Grain size distribution of the filter sand used with arrows showing how to graphically determine d_{10} and d_{60}

a)

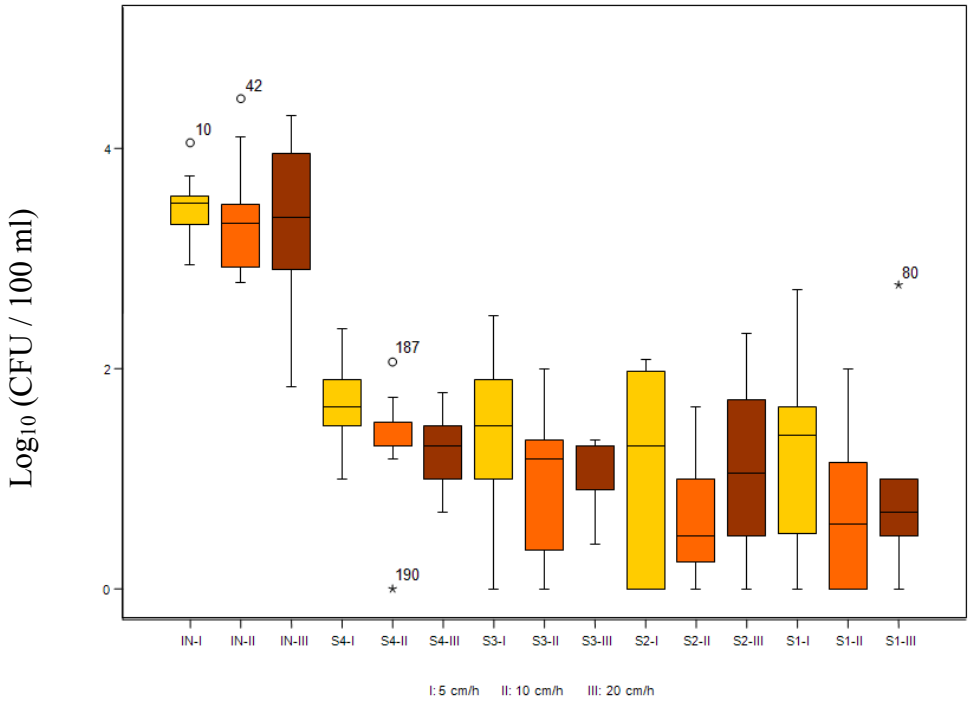


b)

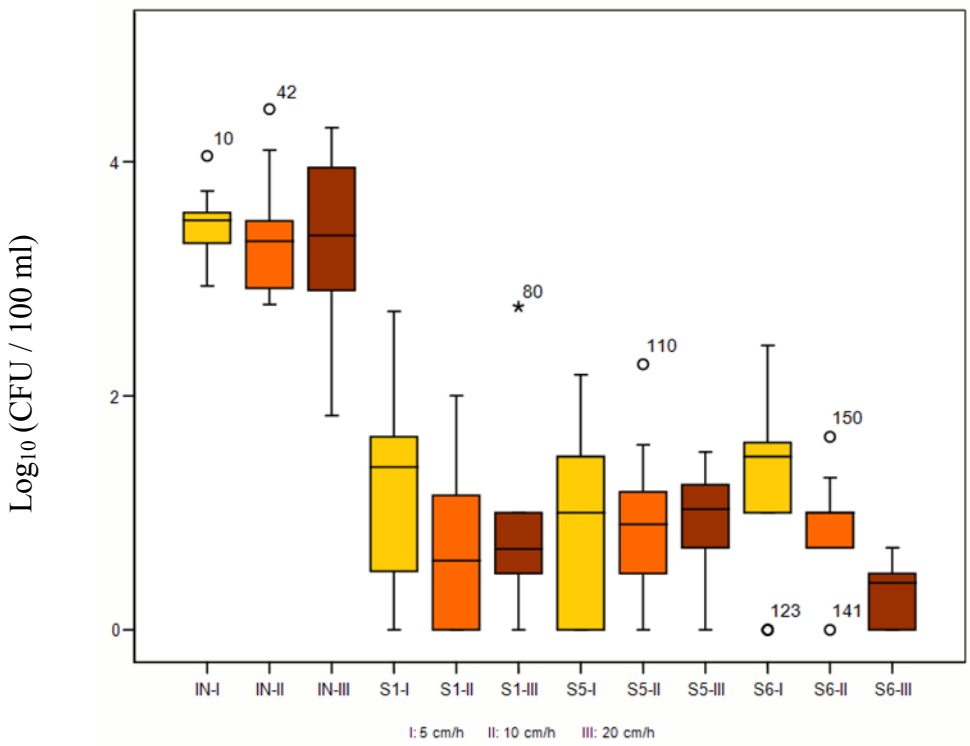


A-2: Boxplots depicting concentrations of *E. coli* in the influent (IN) and effluent of slow sand filters varying in a) d_{10} (S1–S4) or b) uniformity (S1, S5, S6) for phase I–III of increasing hydraulic loading rate

a)

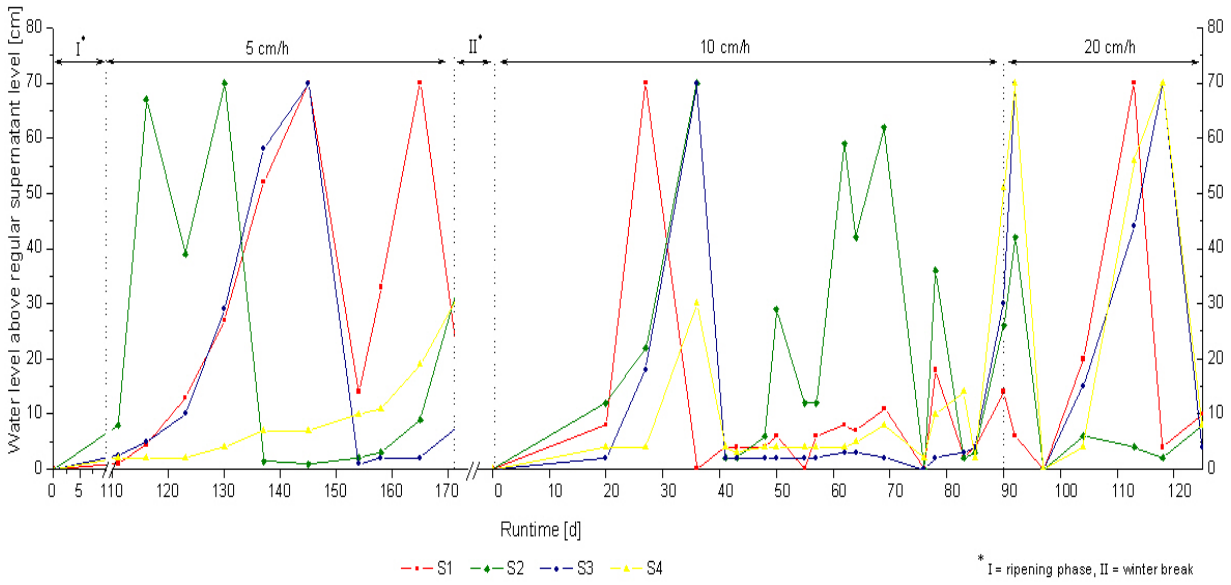


b)

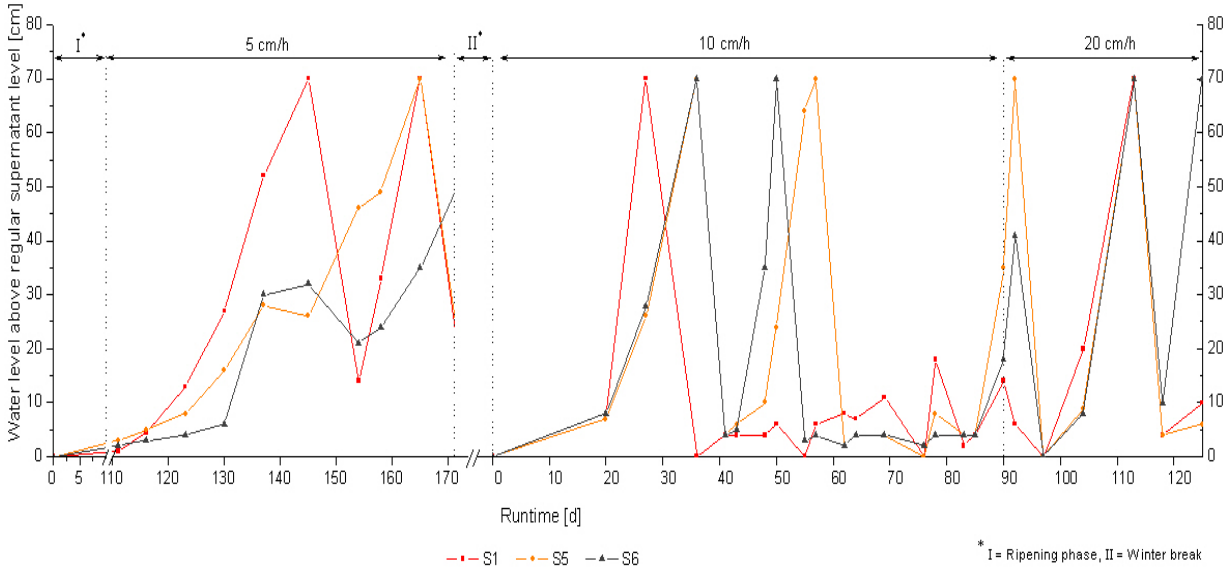


A-3: Boxplots depicting concentrations of intestinal Enterococci in the influent (IN) and effluent of slow sand filters varying in a) d_{10} (S1–S4) or b) uniformity (S1, S5, S6) for phase I–III of increasing hydraulic loading rate

a)



b)



A-4: Increase in supernatant water level in slow sand filters of a) variegated d_{10} (S1–S4) and b) uniformity (S1, S5, S6) during phases of increasing hydraulic loading rate from 5 to 20 cm/h; hydraulic conductivity of the filters was restored by wet-harrowing when the level had reached 70 cm

Epilogue

Wasn't that the alchemists' dream: Making gold out of worthless material? We can do it. We can make blue gold out of wastewater. Water for irrigating agriculture. Water for recreation. How can we achieve this?

If you had 3.5 billion years, would you think that you could come up with a good solution to any given problem? Probably. Life on earth has had this time. We must copy the sustainable management of resources: Close the loops!