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## Task B, 2<sup>nd</sup> delivery: Report on pilot scale tests on technical feasibility

The three novel technologies for sorption of pollutants as well as the technologies for sand filtration applied in the demonstration project are tested on laboratory scale. The experimental results showed the applicability of the technologies to stormwater treatment and resulted in important design parameters.

The action leading to the monitoring program is in the contract with the EC described as:

The adsorption/absorption technologies to be applied in the demonstration facilities are tested in pilot scale experiments. The outcome of the experiments will – together with the scientific knowledge obtained in action B1 – constitute the basis for the detailed design of the full-scale adsorption/absorption technologies. As granular sorption media, a number of materials – e.g. crushed marble rock, crushed bivalve shells and limestone – are tested. For adsorption/absorption to flocks in the bulk water phase, different types and concentrations of aluminium salts are tested. For adsorption/absorption to the stormwater pond bottom, different types and concentrations of iron oxide enriched stormwater pond sediments are tested.

Contingency plans: In case the above described testing reveals conditions hindering or delaying the implementation in full scale, further testing with related technologies will be conducted and brought to a state allowing scaling up to full scale demonstration facilities.



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## 1 Introduction

Design of stormwater treatment facilities is subject to at least two major challenges. The first challenge relates to the fact that stormwater runoff events are rare. On an annual basis, a stormwater treatment facility only receives runoff a few percent of the time. Runoff events are often separated by long intermittent dry weather periods, and the facility must manage high inflow rates upon long dormant periods. The second issue is that rather large pollutant loads from urban surfaces are mixed into quite large water volumes, resulting in low concentration levels. Consequently, the task is to treat rather low pollutant concentrations to even lower levels.

A further complication arises from the need for stormwater detention, which is required to minimize the hydraulic load on downstream facilities and/or receiving waters. Furthermore, the facilities are often placed in areas with recreational value, and must therefore be designed with urban landscape architecture in mind. Consequently, stormwater treatment facilities must at the same time be capable of stormwater pollutant treatment, stormwater flow detention and be an asset of the urban landscape.

Due to the large volumes of stormwater to be treated and the inherent conveyance costs, stormwater is typically discharged to local receiving waters, corresponding to small catchments areas. The consequence hereof is a need for a large number of decentralized management facilities. It is an engineering objective to combine the demands for treatment, retention and recreational values into management facilities with low operational costs and low maintenance requirements.

During the last decades, several technologies for management of stormwater runoff have evolved. Some of the most reliable technologies, achieving good and consistent pollutant removal, are infiltration systems, wet detention ponds and artificial wetlands. Of these solutions, especially wet detention ponds are capable of meeting the above stated requirements.

## 2 The technologies implemented

A key factor for deciding which unit operations to incorporate in the design of a wet detention pond, is the size distribution of the pollutants, i.e. to which extent a pollutant is dissolved, associated with colloids or associated with different fractions of larger particles. As an example, Vaze et al. (2004) report for phosphorous in stormwater runoff from Melbourne, Australia, that around 25% of the total phosphorous load is “dissolved” (particle sizes below 0.45  $\mu\text{m}$ ), whereas the majority of total phosphorous is associated to particle size in the range from 0.01-0.3 mm. These results indicate that sedimentation cannot reduce total phosphorous concentration with more than around 75%; a conclusion which is in the same range as generally observed treatment efficiencies for wet detention ponds.<sup>1</sup>

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<sup>1</sup> Other pollutants have different particle size association patterns. E.g. for stormwater runoff from catchments in Maryland and Washington DC, USA, Hwang and Foster (2006) report that 87% of total PAH (polycyclic aromatic hydrocarbons) is bound to filterable particles. This high degree of association with particles is caused by the low water solubility of the relevant PAHs.

For heavy metals in residential and highway stormwater runoff, New Jersey, USA, Tuccillo (2006) found that Cu and Zn were predominately found in particle size fractions below 5  $\mu\text{m}$ , with a rather large fraction being dissolved (having

The sorption and filtration technologies that are planned for the demonstration facilities are tested in pilot scale experiments. Each demonstration facility combines filtration and sorption in novel ways in order to remove dissolved and colloidal pollutants from the stormwater.

The common denominator for the filtration technologies is that a porous and inorganic filter media is applied and that gravitational force is used to drive the water through the filter. Hereby a fraction of the fine grained and colloidal pollutants are filtered from the stormwater. Three different filter technologies are planned for the demonstration facilities, each of which is tested on laboratory scale.

The sorption technologies all apply media with active sites for binding of dissolved and colloidal pollutants. Three different approaches to sorption are applied: Enrichment of the bottom sediments with iron in order to increase the active sorption sites of the sediment surface (Figure 1); Creation of aluminum containing active flocs suspended in the water phase (Figure 2); Sorption to fixed media filters containing active granules of active sorption media (Figure 3).

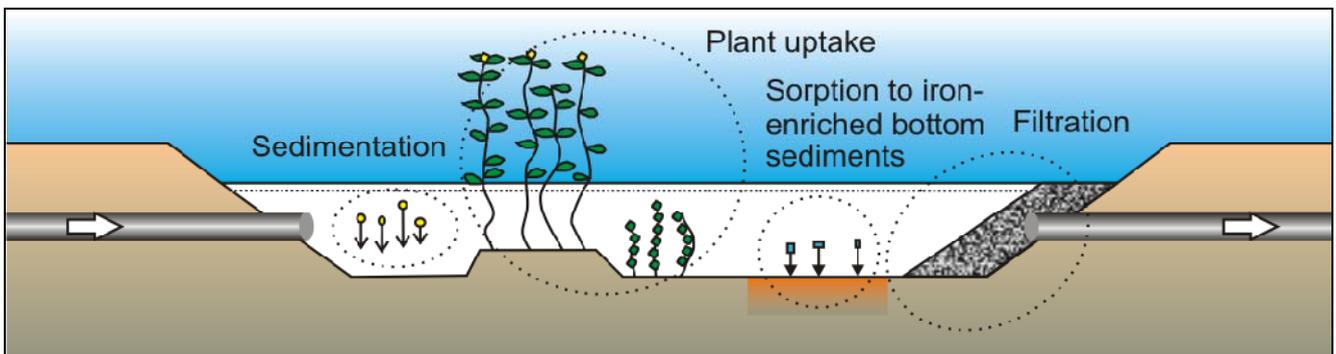


Figure 1 Sorption to iron enriched bottom sediments

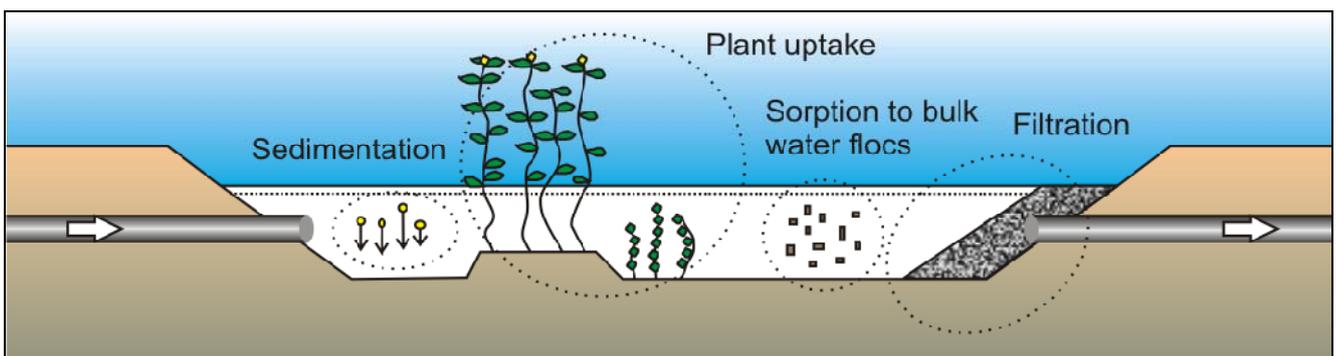


Figure 2 Sorption to bulk water aluminum flocs

masses below 10 kDa). Pb and Cr, on the other hand, were exclusively associated with particle size fractions above 5  $\mu\text{m}$ .

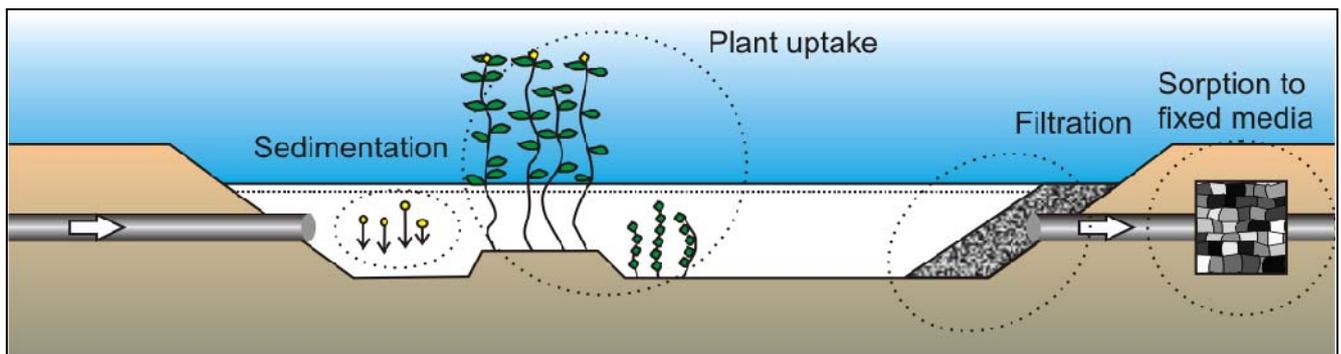


Figure 3 Sorption to filter media

### 3 Unit processes occurring in traditional stormwater ponds

By design, the traditional concept of wet detention ponds applies one major unit operation for treatment, namely sedimentation. It makes use of the fact that a significant part of the pollutants occurring in stormwater are associated with the particulate fractions of the stormwater runoff, and that these are settled out under non-turbulent conditions in the pond.

To an unknown extent, other unit operations are likely to occur in wet ponds, namely plant uptake of dissolved pollutants, sorption of dissolved matter and colloids to surfaces, as well as flocculation of fine particles and colloids. In contrast to sedimentation, these unit operations act on the dissolved and colloidal fraction of the stormwater pollutants – i.e. those fractions which are most mobile in the aquatic environment and consequently possess the highest risk of causing adverse effects.

#### 3.1 The sedimentation process

The concentration of particulate matter in stormwater runoff is low compared to many other environmental systems, e.g. activated sludge settling tanks. The distance between stormwater particles is therefore large, and settling occurs as free, gravitational settling – i.e. settling at laminar conditions can be described by Stokes' Law (Equation 1). Only in the immediate vicinity of the pond bottom, hindered settling can be expected.

$$v_s = \frac{g d^2 (\rho_p - \rho_w)}{18 \mu} \quad (1)$$

where

$\rho_p$	specific density of the particle ( $\text{kg m}^{-3}$ )
$\rho_w$	specific density of the water ( $\text{kg m}^{-3}$ )
$\mu$	dynamic viscosity of the water ( $\text{kg m}^{-1} \text{s}^{-1}$ )
$d$	diameter (spherical) of the particle (m)
$g$	gravitational acceleration ( $\text{m s}^{-2}$ )

From a purely theoretical point of view, the validity of Stokes' Law is limited to a certain particle size interval. E.g. for particles with a density of  $2,500 \text{ kg m}^{-3}$ , Stokes' Law is valid for diameters between approximately  $1 \mu\text{m}$  and  $65 \mu\text{m}$  (Jones and Berard, 1997). Outside this interval, Equation 1 overestimates the true settling velocity.



For naturally occurring grains, Fredse and Deigaard (1992) propose an empirically derived model for describing the relationship between particle characteristics and settling velocity (Equation 2). Figure 4 illustrates how Stokes law over-predicts the settling velocity of natural grains.

$$v_s = \frac{\sqrt{\left(\frac{36\nu}{d_n}\right)^2 + 7.5\left(\frac{\rho_p}{\rho_w} - 1\right)g d_n} - \frac{36\nu}{d_n}}{2.8} \quad (2)$$

where

- $\nu$  kinematic viscosity of water ( $\text{m}^2 \text{s}^{-1}$ )
- $d_n$  nominal diameter, i.e. the diameter of the sphere having the same volume and weight as the grain

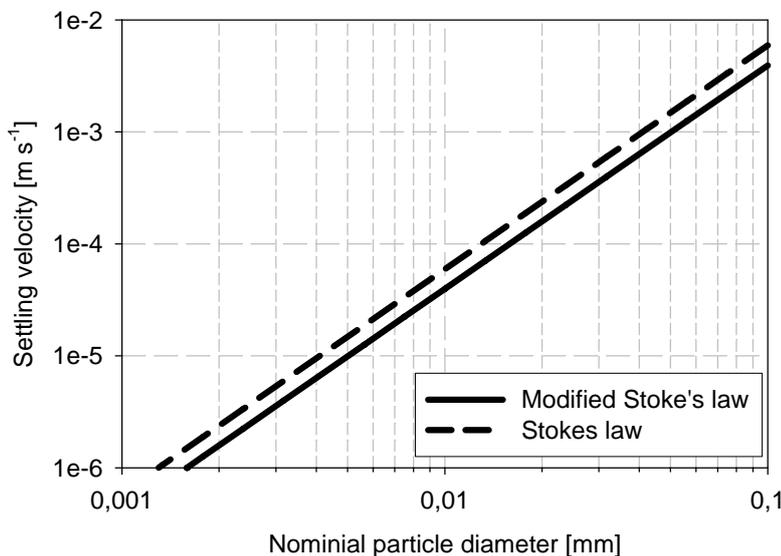


Figure 4: Comparison between Stokes law and the empirical equation proposed by Fredse and Deigaard (1992) (particle densities  $2650 \text{ kg m}^{-3}$ , temperature  $5 \text{ }^\circ\text{C}$ ).

Applying Stokes' Law or modifications hereof for determination of particle settling does furthermore require the water to be quiescent. In stormwater ponds, this assumption is not met, as wind and thermal variations induce mixing. Based on engineering experience, these laws should only be applied for particles in the size range of approximately  $50 - 200 \text{ }\mu\text{m}$ .

The actual settling velocity of a particle in a pond is determined by balancing the particle movement due to gravitational force, drag force, diffusion and mixing. If the random movement caused by diffusion and mixing is greater than the downwards movement due to the balance between the forces of gravitation and drag, the particle will not settle. In this case, particles are only removed from the water phase when diffusion or mixing causes it to impact with a surface to which it may adsorb.

Due to the inherent difficulties of relating settling velocities to particle sizes, settling velocities have been measured directly. However, also measured settling velocities must be applied with caution, as



the true settling velocity of a particle – as stated above – depends on turbulence and mixing conditions in the pond, and because the measurement of settling velocities of especial mixed solids is not well defined (Lucas-Aiguier et al., 1998).

Calculating the settling velocity for an inorganic particle of size 50  $\mu\text{m}$  at 5  $^{\circ}\text{C}$ , the equation of Fredse and Deigaard (1992) yields a settling velocity of  $10^{-3} \text{ m s}^{-1}$ . However, most studies on the settleability of stormwater solids indicate that residence times comparable to settling velocities as low as  $10^{-4}$  to  $10^{-5} \text{ m s}^{-1}$  are needed to remove the majority of suspended solids from stormwater runoff (Pisano, 1996).

The general conclusion is consequently that models like Stokes law or the model of Fredse and Deigaard (1992) are not adequate in assessing settling in a complex system like a wet detention pond. Instead the engineer has to fall back on simple empirical design rules to ensure adequate sedimentation. I.e. the pond must allow longitudinal flow (no short-circuiting of flow), water must be quiescent, residence times must be comparable to settling velocities of  $10^{-4}$  to  $10^{-5} \text{ m s}^{-1}$ .

### **3.2 The effects of aquatic macrophytes**

Aquatic macrophytes are often integrated in the design of wet detention ponds, but also where plants have not been actively included in the design, they tend to colonize the shallow parts of the ponds. Plants in wet detention ponds fulfil several purposes with respect to pollutant removal.

- Enhancement of sedimentation by reducing turbulence in the bulk water phase. The plant shoots constitute a mesh that introduces resistance to water movement and mixing, hereby providing quiescent zones where particles of low settling velocity can settle out of the water column (e.g. Pettecrew and Kalff, 1992).
- The shoots provide surfaces to which small particles and colloids can adsorb. The colloids are built into a matrix of other colloids, algae and bacteria. This material stabilizes over time, forming a coherent substance which stays stable also when flakes are torn off.
- The shoots submerged in the water provide surfaces for attached microbial growth (e.g. Gumbricht, 1993). This growth contains of photosynthetic algae as well as bacteria. In a similar way, the roots provide surfaces for attached bacterial growth in the pond soil and sediments (Hofmann, 1986). The microbial growth on the plant tissue is responsible for the major part of transformation processes in wetlands (Brix, 1997)
- Several wetland plants have been shown to accumulate significant amounts of heavy metals, with the highest concentrations found in the roots. Furthermore, plants take up nutrients (nitrogen and phosphorous) that become part of the plant biomass. The magnitude of plant uptake is seasonal, occurring only during the growing season (Cronk and Fennessy, 2001; Bragato et al., 2006). Even though macrophytes have a high capacity of taking up nitrogen and phosphorous, the net removal is only significant where wetland plants are harvested. If the plants are allowed to decay, nutrients will be returned to the bulk water (Brix, 1997).
- The dense root system of the macrophytes stabilizes the sediments on the pond bottom, hereby reducing the risk of sediment bed erosion.
- Aquatic macrophytes release oxygen from the roots and into the rhizosphere, hereby enhancing aerobic microbial breakdown of organic compounds in the pond sediments (Barko et al., 1991).



A drawback of macrophytes in ponds is a reduced reaeration rate due to the stems reducing the wind velocity over the water surface and hereby the turbulence in the water. Furthermore, as plant biomass is typically not harvested, a high biomass density will cause a correspondingly high formation of plant debris and thereby increase the formation of bottom sediments, reducing the lifetime of the pond. It is consequently not desirable that the entire pond is covered by vegetation, and a compromise between the beneficial effects and the negative effect must be accomplished.

The practical way of permanently controlling which parts of a pond are covered by emergent macrophytes is to control the water depth. Emergent macrophytes will colonize the shallow parts of the pond, but do not thrive at deeper water levels. Most of these wetland plants thrive at water depth above 0.3 m, however, e.g. the common reed can tolerate permanent inundation up to 1 m (Reed et al., 2006).

The extent of colonization by submerged species depends on the turbidity of the water, i.e. to what extent light can penetrate sufficiently to allow these plants to grow. Floating species, however, must be expected. E.g. in most wet detention ponds, duckweed is likely to occur, causing periodically reduced reaeration of the pond. The presence of duckweed cannot be prevented, and wet detention ponds should therefore have open-water zones large enough to allow wind to break up the vegetation cover. For the same reason, wind shelter due to e.g. trees and scrubs on the embankments should be avoided.

The overall effect of macrophytes on the pond performance and the pollutant removal is difficult to quantify, however, the effects of emergent macrophytes are predominantly beneficial, and should from a treatment efficiency point of view be actively included in the design of wet detention ponds.

In addition to enhancing pollutant removal, aquatic macrophytes provide habitats for birds, amphibians and aquatic insects. Furthermore, incorporated actively into the design of wet detention ponds, they contribute significantly to the aesthetic value of the system, strengthening the impression of a natural aquatic habitat.

### **3.3 Filtration**

Filtration of the effluent from a wet detention pond in a fixed media filter - such as a sand filter - is an efficient method for retaining particles. During the filtration process, particles are deposited on the surface of the filter, creating a colmation layer. This layer will typically have a much lower hydraulic conductivity than the filter medium and therefore controls the overall hydraulic capacity of the filter and the limiting factor for the filter capacity becomes the depth and hydraulic conductivity of the colmation layer. The conductivity of a colmation layer is governed by the degree and pattern of filter clogging, which again is governed by differences in the filter loading and the drying out between storm events. Different filter layouts are consequently expected to behave differently with respect to long-term conductivity of the colmation layer.

Assuming colmation layer to be homogenous with a well-defined depth and the conductivity of the layer much lower than the conductivity of the underlying filter material, the flow through the colmation layer occurs as saturated flow and can as a first estimation be described by Darcy's law, ignoring the underlying soil (e.g. Rauch and Stegner, 1994; Vollertsen and Hvitved-Jacobsen, 2003):



$$Q_{out} = A_{out} \Delta h \frac{k_f}{\Delta l} \quad (3)$$

where

$Q_{out}$	filter capacity [ $\text{m}^3 \text{s}^{-1}$ ]
$A_{out}$	area through which the water is filtered [ $\text{m}^2$ ]
$\Delta h$	water pressure on the filter [m]
$k_f$	hydraulic conductivity of the colmation layer [ $\text{m s}^{-1}$ ]
$\Delta l$	depth of the colmation layer [m]

In constructed stormwater filters, the hydraulic conductivity of the filter media (e.g. sand) will typically be much lower than that of the colmation layer, and the formation of the colmation layer, its composition, magnitude and physical properties hereby become the prime design parameter for such filtration. However, the knowledge reported on the colmation layer is limited and often stated as infiltration rates without reference to the magnitude and properties of the layer. In such a case it can be prudent to introduce a leakage factor:

$$L_{out} = \frac{k_f}{\Delta l} \quad (4)$$

and consequently

$$Q_{out} = A_{out} \Delta h L_{out} \quad (5)$$

The reported hydraulic conductivities of colmation layers from wastewater infiltration and river beds lead to the conclusion that the hydraulic conductivity of a colmation layer formed under a permanent water level is likely to have no less than  $10^{-7} \text{ m s}^{-1}$ , and that a likely value is 1-2 decades higher. For stormwater infiltration through a filter with intermittent loading, a design leakage factor of around  $1 \cdot 10^{-4} \text{ s}^{-1}$  seems to be a conservative choice.<sup>2</sup>

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<sup>2</sup> For the exfiltration of wastewater from sewer systems, Vollertsen and Hvitved-Jacobsen (2003) report values of the hydraulic conductivity,  $k_f$ , to be 1-2  $10^{-6} \text{ m s}^{-1}$ . For the infiltration of tertiary treated wastewater for ground water recharge, Houston et al. (1999) report the hydraulic conductivity of the colmation layer to depend on consolidation by seepage forces. They report the hydraulic conductivity of the colmation layer to be roughly around 1-5  $10^{-5} \text{ m s}^{-1}$ . Calver (2001) compared 41 studies on riverbed permeability. The hydraulic conductivity reported in the studies covered a range of  $1.0 \cdot 10^{-9}$  to  $1.0 \cdot 10^{-2} \text{ m s}^{-1}$ , with the majority of the reported values in the range of  $1.0 \cdot 10^{-7}$  to  $1.0 \cdot 10^{-3} \text{ m s}^{-1}$ .

Houston et al. (1999) report pilot scale investigations of infiltration of tertiary treated wastewater. The infiltration columns were operated over a period of 18 months, with regular cycles of flooding. The observed infiltration rates were typically no less than  $6 \cdot 10^{-6} \text{ m}^3 \text{ m}^{-2} \text{ s}^{-1}$  at a water depth of app. 0.2 m, i.e. corresponding to a leakage factor of  $3 \cdot 10^{-5} \text{ s}^{-1}$ .

Reed et al. (2006) state that hydraulic loading rates on an intermittent sand filter treating effluent from wastewater lagoons is typically  $4-6 \cdot 10^{-6} \text{ m}^3 \text{ m}^{-2} \text{ s}^{-1}$ , with somewhat lower values to be applied under cold climate conditions.

Dechesne et al (2005) report stormwater infiltration field measurements for 4 infiltration basins that had been in operation between 10 and 21 years. For the basin with the lowest infiltration capacity, they e.g. report an infiltration rate of around  $1.1 \cdot 10^{-4} \text{ m}^3 \text{ m}^{-2} \text{ s}^{-1}$  at 0.5 m of water depth, i.e. a leakage factor of  $2.2 \cdot 10^{-4} \text{ s}^{-1}$ .



The hydraulic conductivity will furthermore depend on the wetting cycle to which the filter is subjected. If the filter is permanently submerged, the hydraulic conductivity will be less compared to the situation where the filter is allowed to dry out between wettings.

The test conducted on filtration attempt to investigate the capacity of different filter types in order to verify design parameters obtained from study of the relevant literature. Furthermore, the impact of continuous aluminum addition to the hydraulic conductivity is investigated as it is theorized that aluminum residuals could have a negative impact on the filter conductivity.

The filter types investigated are:

- A horizontal filter placed in the same level as the dry weather permanent water level of the pond. When the water level rises, all the filter surface becomes submerged simultaneously and becomes subject to an evenly distributed water pressure while the pond water level rises and falls. This filter is expected to build up the deepest colmation layer as all the surface is subject to stormwater, and consequently to develop the lowest leakage factor of the 3 filters tested.
- A sloping filter placed on the embankments. The active filter area starts in level with the dry weather permanent water level of the pond and goes up to the allowable maximum water depth of the storage volume. When the water level rises, the active filter area increases correspondingly. This filter is expected to build up the deepest colmation layer close to the water surface, with a decreasing colmation layer depth towards the maximum water level of the pond. The average leakage factor of the colmation layer is consequently expected to be higher than that for the horizontal filter.
- A vertical filter placed in the pond. The active filter area starts in level with the dry weather permanent water level of the pond and goes up to the allowable maximum water depth of the storage volume. When the water level rises, the active filter area increase correspondingly. The filter material is kept in place by a rigid structure and a geo textile. This filter is expected to build up only a thin colmation layer as the colmation layer is expected to crack off the geo textile during dry weather periods and fall back into the permanent water pool. The lowest leakage factor is expected close to the water surface. This filter is consequently expected to have the lowest average leakage factor of the 3 filter types tested.

### **3.4 Sorption to iron enriched bottom sediments**

Ferric iron binds phosphate under aerobic conditions, and the content of iron in aquatic sediments is recognized as a significant parameter in the phosphate binding capacity of such sediments. Furthermore, several heavy metals are strongly associated with iron and manganese oxides of lake sediments (Stead-Dexter and Ward, 2004),

Jensen et al. (1992) compared 116 Danish lakes and found a significant negative correlation ( $p < 0.01$ ) between the lake water total phosphorous content and the Fe:P ratio found in the pond sediments. They conclude that aerobic bottom sediments can control the bulk water phosphate if the Fe:P ratio is above 15 (by weight), and that an increasing ratio causes decreasing bulk water phosphorous concentrations. They furthermore studied the sorption kinetics of 15 selected lakes, and found that Langmuir isotherms described the phosphate adsorption properties of the surface sediments well.



Addition of iron can be applied as an effective method of controlling the release of phosphorous from shallow lakes (Hansen et al., 2003).<sup>3</sup> Iron binds bulk water phosphorous and interrupts the internal phosphorous loading of the pond. As numerous iron containing compounds also have excellent binding efficiency for other pollutants, it is believed that iron enriched bottom sediments in a similar way can bind bulk water control the release of e.g. heavy metals from the sediments.

Applying iron to immobilize pollutants, it is crucial that the redox potential of the sediments is sufficient to avoid ferric iron to be reduced to ferrous iron. If ferric iron is reduced, the bound phosphate, heavy metals and probably also other pollutants, are released back into the bulk water phase. It is therefore essential to ensure that the uppermost parts of the stormwater pond sediments do not become anaerobic, i.e. there must at any time be sufficient oxygen present in the bulk water to avoid oxygen depletion of the uppermost millimetres of the sediments.

### **3.5 Sorption to fixed media**

Certain materials have proven efficient in binding pollutants. E.g., materials containing calcite ( $\text{CaCO}_3$ ) or dolomite ( $\text{CaMg}(\text{CO}_3)_2$ ) like marble, limestone, dolomite rock and different types of shells from marine organisms are efficient in removing phosphorus (Westholm, 2006; Brix et al., 2001; Vohla et al., 2005). When it comes to removal of e.g. heavy metals, materials like alumina, bauxsol-coated sand or granulated ferric hydroxide are most effective (Genc-Fuhrman et al., in press).

When designing fixed filters, several parameters regarding the filter media must be considered. It is essential that the selected filter media has a high sorption capacity for the targeted pollutant(s)<sup>4</sup>. This allows for a long-term use of the filter without the need for changing the filter media. The kinetics of the intended sorption process should be fast, as this will have a significant impact on the minimum contact time required to achieve the desired pollutant removal.

In order to obtain good pollutant removal at the relatively low concentration levels characteristic of stormwater runoff it is also important that the adsorption equilibrium concentrations are low, i.e., the partitioning of the pollutant between the filter media and the liquid phase. This property is described by the adsorption isotherm, which relates the amount of pollutant adsorbed by the filter media to the liquid phase concentration.

In addition, the filter medium must meet specific requirements regarding the hydraulic capacity. These requirements are largely determined by the hydraulic conductivity and the porosity of the

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<sup>3</sup> Quaak et al. (1993) report the restoration of a shallow lake by injecting  $\text{FeCl}_3$  into the uppermost 15-20 cm of the lake sediments. They applied  $100 \text{ g Fe m}^{-2}$ , and achieved permanent reductions in phosphorous and chlorophyll-a concentrations. Jaeger (1994) reports the addition of  $5 \text{ g Fe m}^{-3}$  into the hypolimnetic layer of a stratified lake. The addition resulted in the sediments being covered by an iron hydroxide layer and the sediment pore water to contain  $4 \text{ g Fe m}^{-3}$ . Hereby the internal phosphorous loading from the sediments were effectively interrupted and a significant decrease in bulk water phosphorous concentrations and chlorophyll-a concentrations was achieved. Smolders et al. (2001) report that  $30 \text{ g Fe m}^{-2}$  would be needed to immobilize phosphate in the sediments of a studied phosphorous rich lake in The Netherlands.

<sup>4</sup> E.g. the sorption capacities for calcareous materials vary considerably with material characteristics. Typical phosphorus sorption capacities for calcareous filter media are expected to be in the order of  $0.2 - 5 \text{ mg P (gTS)}^{-1}$  (Westholm, 2006). Due to the low concentrations of phosphorous in stormwater, filter media capacities for stormwater treatment are in the lower end of this interval.



media. Prior to sorption in a fixed filter, the water must be pre-treated to remove fine particles. Such an initial filtration step is crucial in order to avoid clogging of the filter and thereby reducing its hydraulic capacity.

### **3.6 Sorption to flocs in the bulk water containing aluminum**

The addition of aluminum salts has been practiced for restoration of eutrophic lakes in terms of removal of phosphorous from the water column and immobilization of phosphorous in the lake sediments (Cooke et al., 2005). Also for small lakes and ponds, the addition of aluminum has been found to improve the water quality significantly (May and Baker, 1978). Another important application of aluminum salts for pollutant removal is potable water treatment. Aluminum has here been used for over 200 years, and aluminum salt is probably the most commonly used coagulation agent in this context (Ødegaard, 1990).

In the water phase, aluminum acts partly by precipitating orthophosphate and partly by enhancing the formation of flocs. The flocs entrap organic and inorganic colloids, which subsequently settle to the pond bottom. The entrapped colloids are the carrier of numerous pollutants in stormwater, however, they are too small and of too low mass to settle. The colloids stay in the suspension because of mutually repulsive forces between the particles. Adding coagulants like aluminum, the mutual repulsive forces are eliminated and the colloids bind together to form aggregates that subsequently flocculate to large particles. The later are readily affected by gravitation and settle out under quiescent conditions.

In the sediments, the aluminum hinders anaerobic release of phosphorus and bound colloids, as the aluminum complex is insensitive to variable redox conditions. Furthermore, the aluminum enriched bottom sediments act as an active surface to which phosphorous and colloids are adsorbed.

When aluminum sulfate – or another aluminum salt – is added to water, the salt dissociates and forms aluminum ions. These ions are immediately hydrated and aluminum hydroxides are rapidly formed. In the pH interval between app. 5.5 and 8.5,  $\text{Al}(\text{OH})_3$  will be the main product of this process.  $\text{Al}(\text{OH})_3$  is insoluble and forms visible flocs that settle in the water column. In case the pH rises above 8.5-9 – as can be the case during periods with intense photosynthesis –  $\text{Al}(\text{OH})_3$  is ionized to  $\text{Al}(\text{OH})_4^-$ , causing a risk of release of the sorbed phosphorous and colloids. Furthermore, if pH falls below app. 5, the aluminum ion,  $\text{Al}^{3+}$  is released, being toxic for the aquatic fauna.

The applied dosing ranges reported for efficient removal of orthophosphate and organic bound phosphorous from lakes are reported in the interval of 1.5 to 30 g Al m<sup>-3</sup>, with cost-effective dosing around 2 - 6 g Al m<sup>-3</sup> (Auvray et al., 2006; Cooke et al., 2005). The need of dosing aluminum to stormwater entering a wet detention pond is expected to be in the same range, as bulk water phosphorous concentrations are comparable in the two systems.

Aluminum sulfate has typically been the preferred coagulant. However, Hansen et al. (2003) suggest to instead using aluminum chloride, in order to reduce sulfide formation in the deeper parts of the sediment. Sulfide restricts the upward migration of Fe(II) and hereby the formation of iron oxides.



## 4 Laboratory setup

The laboratory scale setup was constructed as 4 rectangular tanks in combination with one sand filter unit each. The flow into each tank was controlled by a peristaltic pump together with a PLC control unit, allowing simulation of storm events with intermittent dry weather periods (Figure 8 and Figure 9). The water pumped into the tanks was stormwater sampled from a stormwater pond treating highway runoff (Figure 10). In order to increase the aging speed of the filters, sediments from the stormwater pond was collected and mixed into the sampled stormwater. Until the stormwater was pumped into the laboratory tanks, the stormwater was kept in suspension by continuous stirring (the blue tank on the right-hand side of Figure 8).

The sand filter types tested in the laboratory were a horizontal filter, a sloping filter and a vertical filter constructed as follows:

- The setup for the horizontal filter is shown in Figure 5. The filter was operated permanently submerged in order to increase the aging rate of the filter. The filter was placed in a 160 mm diameter reactor. The filter had a diameter of 110 mm and was filled with 10 cm homogenous filter sand with a d50 of 0.535 mm, overlying 5 cm of coarser material, the later acting as drainage for the overlying sand. This sand filter type was applied on the laboratory reactors 2 and 3.
- The setup for the sloping filter is shown in (Figure 6). The filter was operated with intermittent periods of wet and dry conditions. The filter was placed in a 160 mm diameter reactor. The filter had a diameter of 110 mm and was filled with 10 cm homogenous filter sand with a d50 of 0.535 mm, overlying 5 cm of coarser material, the later acting as drainage for the overlying sand. The filter had a free outlet, allowing the water to drain completely from the filter between events. This sand filter type was applied on the laboratory reactor 1.
- The setup for the vertical filter is shown in (Figure 7). The filter was operated with intermittent periods of wet and dry conditions. The filter was placed in a 160 mm diameter reactor. The filter had a diameter of 110 mm and was filled with homogenous filter sand with a d50 of 0.535 mm, overlying 5 cm of coarser material, the later acting as drainage for the overlying sand. The filter surface was made vertical by drilling a large number of holes in the sides of the 110 cm diameter reactor, and inserting a geo textile into the reactor to confine the filter sand. The filter had a free outlet, allowing the water to drain completely from the filter between events. This sand filter type was applied on the laboratory reactor 4.



Figure 5: The horizontal and permanently submerged filter applied on laboratory reactor 2 and 3

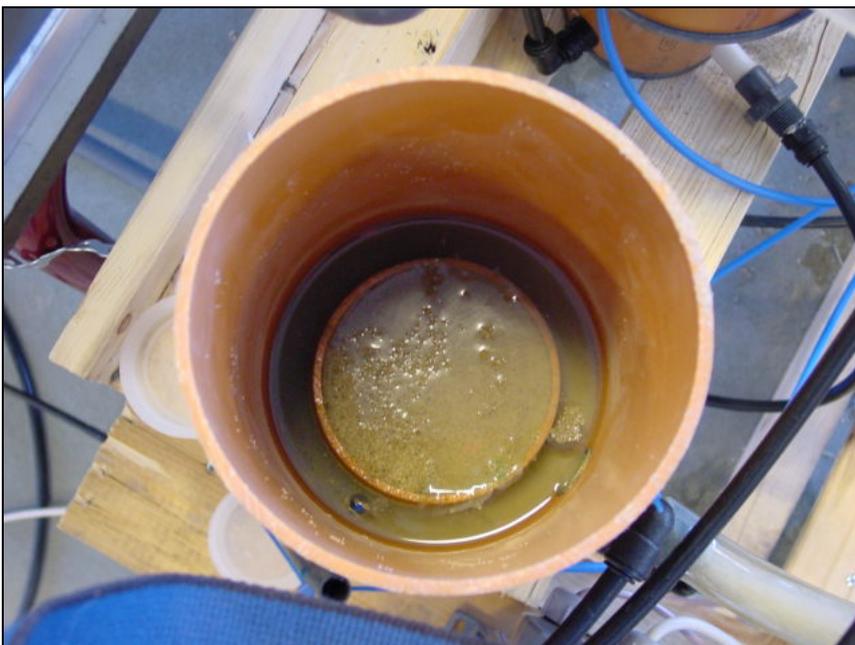


Figure 6: The sloping filter applied on laboratory reactor 1



Figure 7: The vertical filter applied on laboratory reactor 4



Figure 8 Overview over the laboratory scale reactors

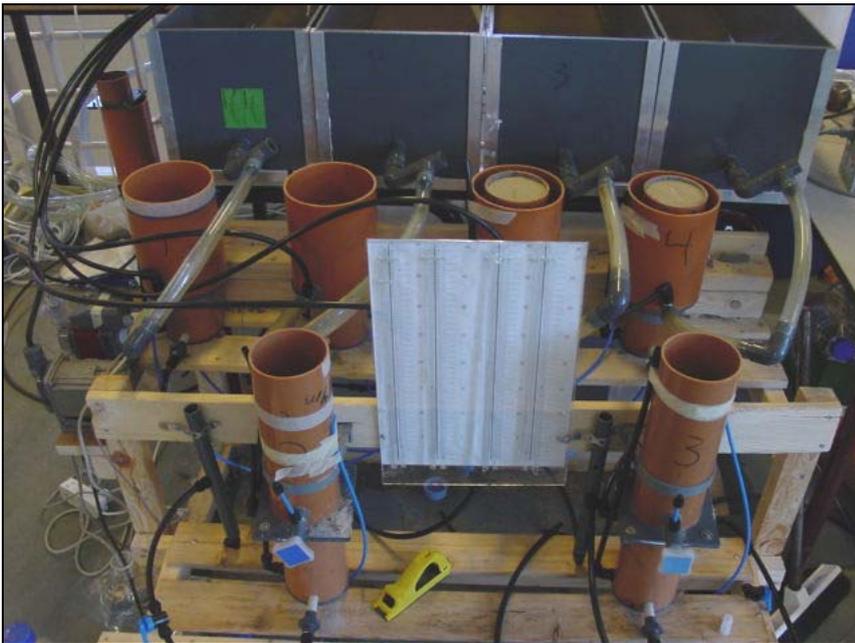


Figure 9 Filters connected to the laboratory scale reactors



Figure 10 Stormwater is collected from a stormwater pond treating highway runoff.



## 5 Results

### 5.1 Filtration

After some time of operation, the filters showed a clear tendency towards a linear increase of flow with increasing water pressure on the filter. This is in full agreement with the assumption, that a thin colmation layer was formed on the filter surface, causing the main resistance to the flow through the filters (Figure 11 to Figure 14). I.e. the assumption that the flow through the filters can be described by the simplified version of Darcy’s law postulated in Equations 4 and 5 is valid.

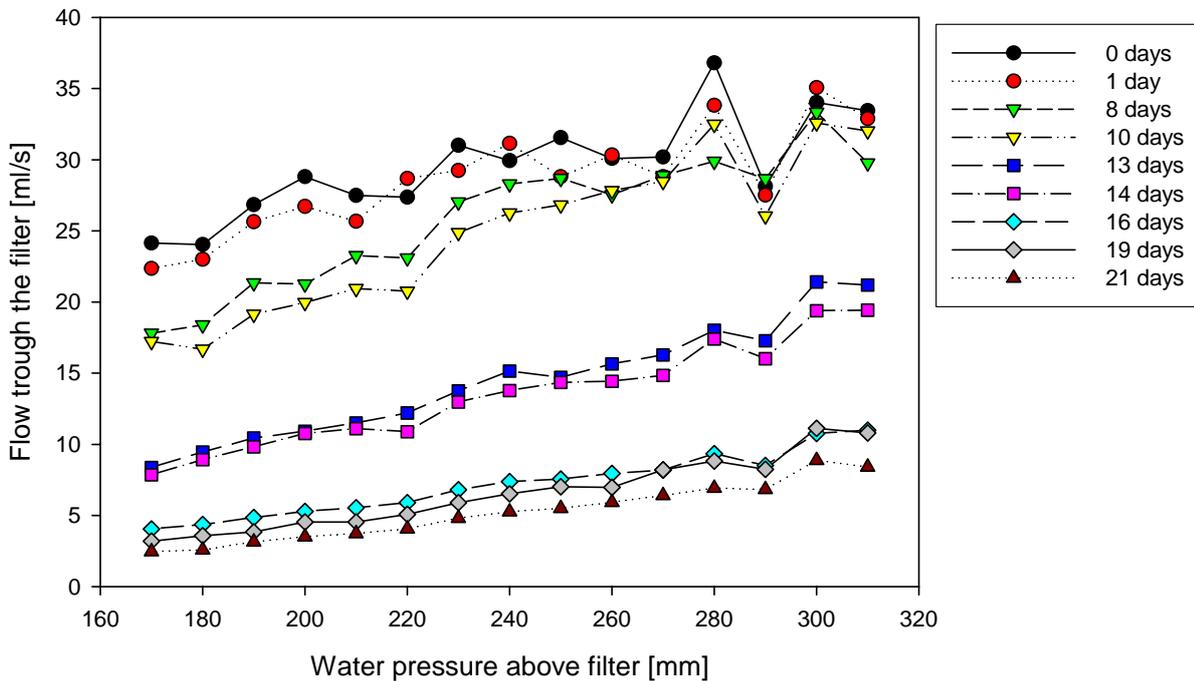


Figure 11: Laboratory reactor 1 with sloping filter. Relation between outflow and water pressure over 21 days of operation.

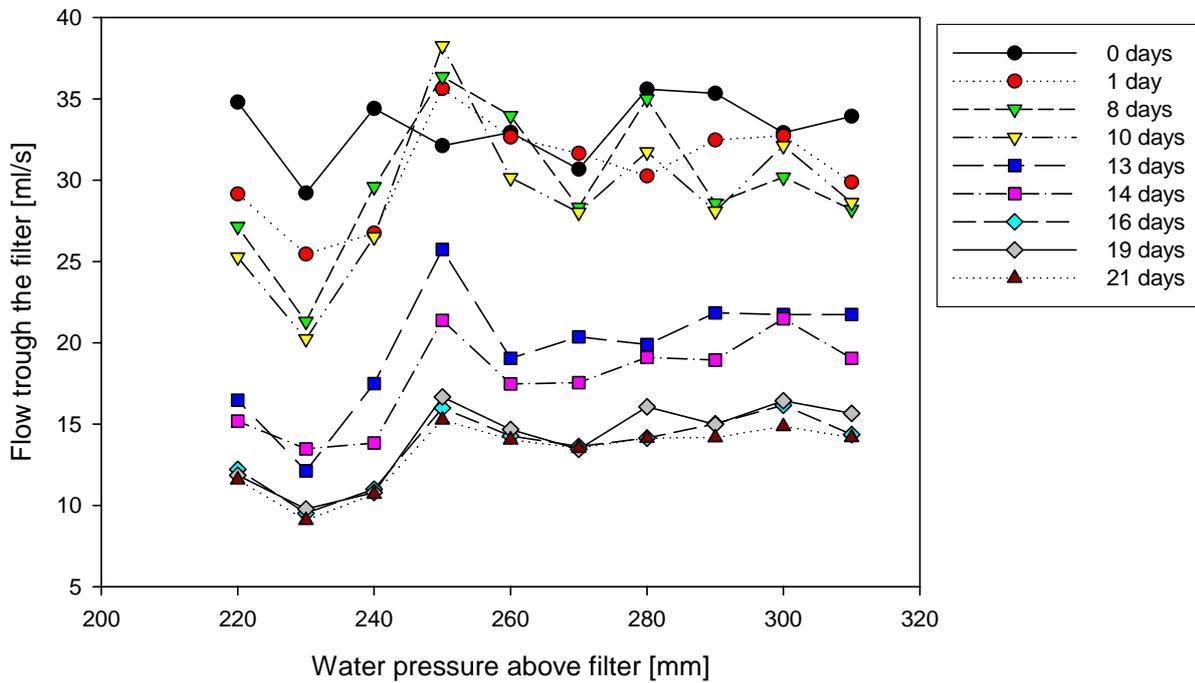


Figure 12 Filter 2, submerged filter. Relation between outflow and water pressure over 21 days of operation.

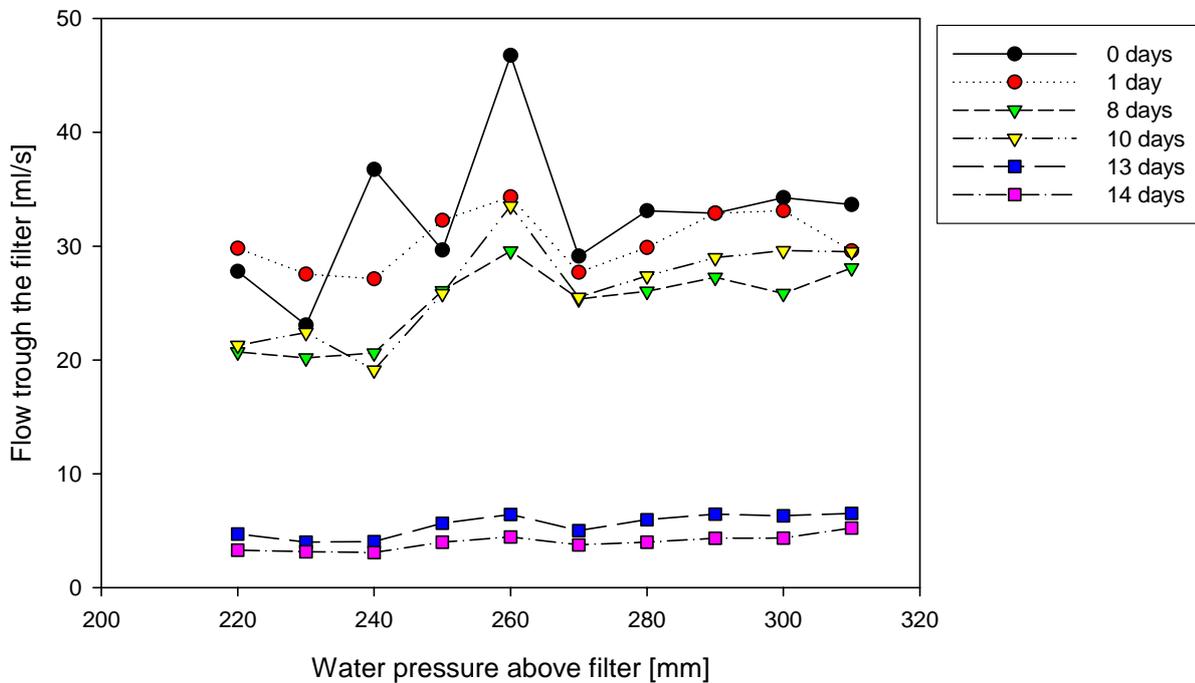


Figure 13: Laboratory reactor 3 with submerged filter. Relation between outflow and water pressure over 14 days of operation (due to technical problems, this filter could not be operated for 3 weeks like the other filters).

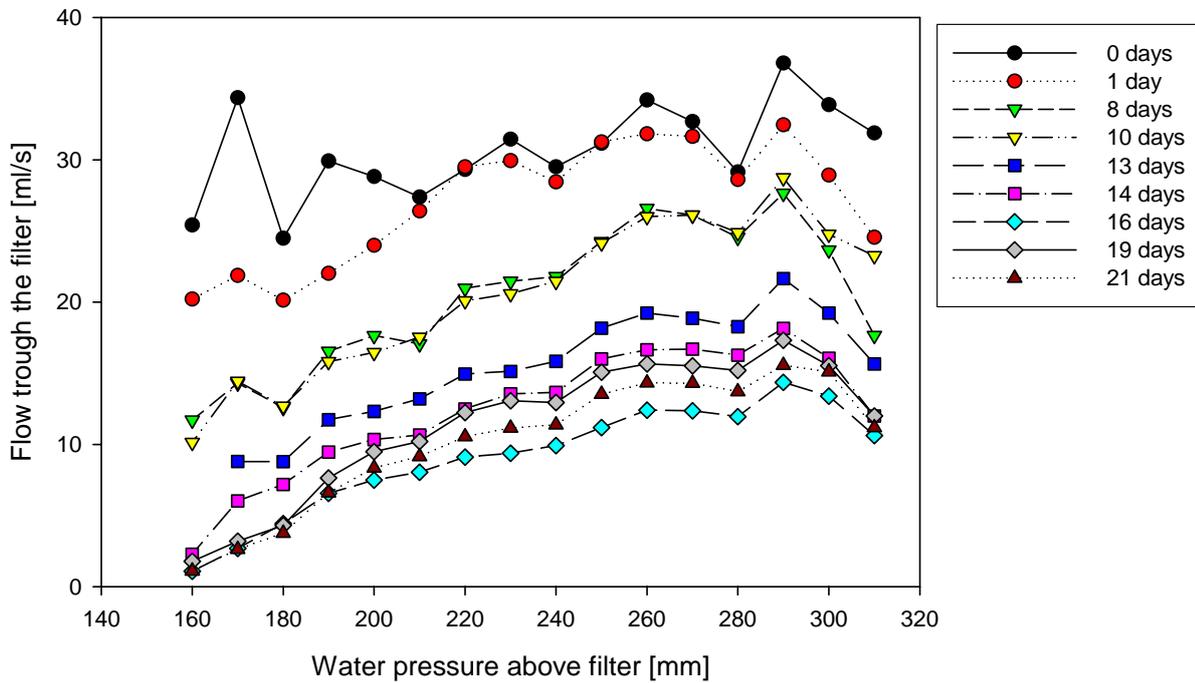


Figure 14: Laboratory reactor 4 with vertical filter. Relation between outflow and water pressure over 21 days of operation.

After 21 days of operation, filters 1, 2 and 4 were left without flow for 1 week, and the relation between outflow and water pressure was tested. This resulted in filter 1 and 4 to be dry, and filter 2 to be submerged but without flow for the rest period.

Immediately after the rest period, the exfiltration rate of filter 1 increased significantly, however, after 1 day of wet operation, the conductivity of the filter was back to the same level as before the dry rest (Figure 15). Filter 2, which was left wet, did not change hydraulic conductivity due to the rest period (Figure 16). Neither was Filter 4 influenced by the rest period (Figure 17).

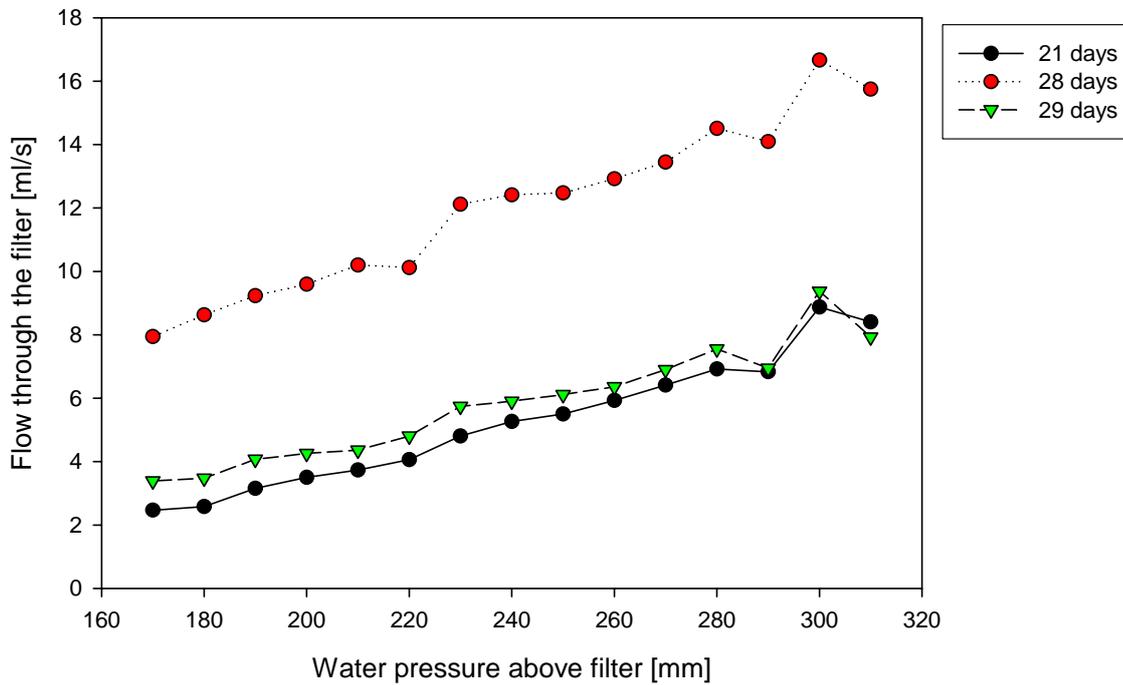


Figure 15: Laboratory reactor 1 with sloping filter. Relation between outflow and water pressure after 21 days of operation, followed by 1 week of rest (28 days) and subsequently by 1 day of operation (29 days).

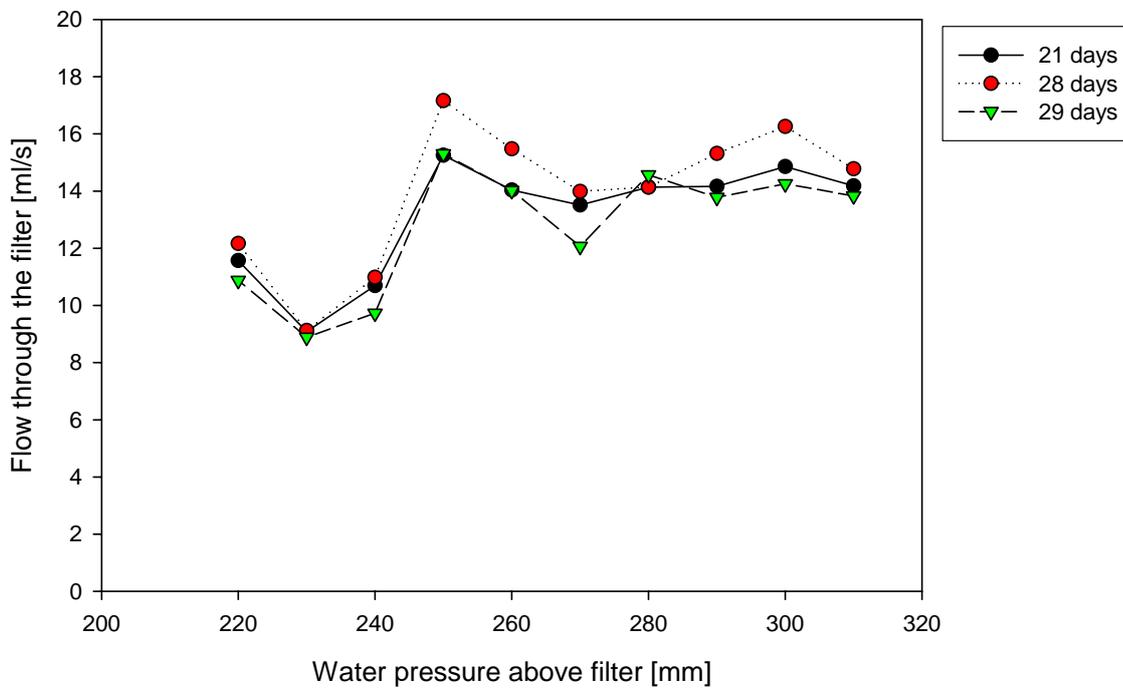


Figure 16: Laboratory reactor 2 with submerged filter. Relation between outflow and water pressure after 21 days of operation, followed by 1 week of rest (28 days) and subsequently by 1 day of operation (29 days).

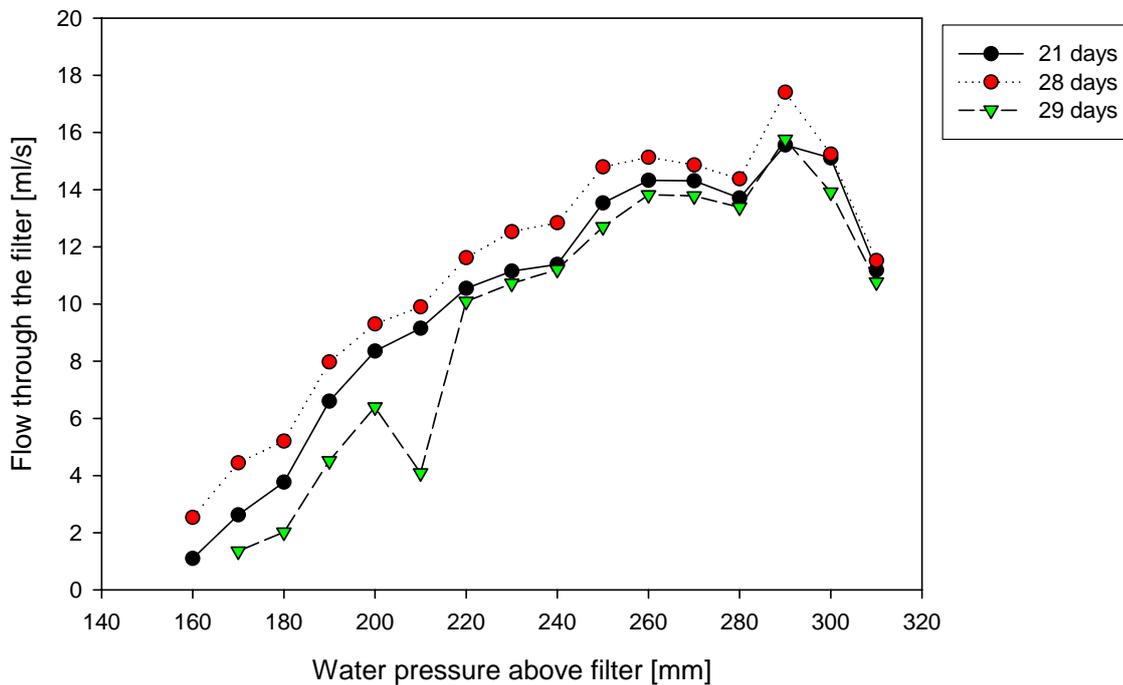


Figure 17: Laboratory reactor with vertical filter. Relation between outflow and water pressure after 21 days of operation, followed by 1 week of rest (28 days) and subsequently by 1 day of operation (29 days).

For all three filter types, a clear decrease in hydraulic conductivity was seen as the filter aged; measured as the total load of suspended solids on the filter. The vertical filter exhibited the highest leakage factor, followed by the sloping filter. Two tests were made with the horizontal filter, one of which showed a rapidly decreasing conductivity, the other showing a more moderate decrease.

The experimentally determined leakage factor (Equation 4) is around  $10^0$ - $10^{-1} \text{ s}^{-1}$  and no significant differences between the filter types could be observed (Figure 18). I.e. the permeability of the colmation layer seems to be independent on the operation conditions under which the layer was formed and only to depend on the total amount of deposited material. Comparing the experimentally determined leakage factors with the proposed long-term design leakage factor for the submerged, horizontal filter ( $10^{-4} \text{ s}^{-1}$ ), the experimentally determined value is 3-4 decades lower than the experimentally determined factors. The consequence hereof is that the allowable loading in  $\text{g TSS m}^{-2}$  of the filters is higher by a similar factor, as the leakage factor is proportional to the magnitude of the accumulated sediments (Equation 4).

Applying the data to estimate design leakage factors, the long term accumulation of sediments on the horizontal filter is expected not to exceed 20-30 cm, on the sloping filter the maximum depth is expected less than 5-10 cm, and on the vertical filter the depth is expected to reach no more than 0.5-1 cm. Assuming the colmation layer to be largely formed by a mix of fine organic and inorganic material, a sediment density of  $1,200$ - $1,600 \text{ kg m}^{-3}$  is an appropriate estimate, and the corresponding amount of deposited sediments for the design situation for the 3 filters are around  $180 \text{ kg TSS m}^{-2}$ ,  $50 \text{ kg TSS m}^{-2}$ , and  $5 \text{ kg TSS m}^{-2}$ , respectively. Comparing with the loads achieved in the



laboratory experiments, the leakage factor for the aged filter becomes  $2.5 \cdot 10^{-4}$ ,  $10^{-3}$ , and  $10^{-2} \text{ s}^{-1}$ , for respectively the horizontal filter, the sloping filter and the vertical filter.

These design leakage factors are applied in the design of the full scale demonstration facilities in Silkeborg, Odense and Aarhus. As the horizontal filter is expected to be most prone to clogging, a design leakage factor of  $10^{-4} \text{ s}^{-1}$  is chosen for the full-scale demonstration facility. For the sloping filters and the vertical filters, design leakage factors of  $10^{-3}$ , and  $10^{-2} \text{ s}^{-1}$ , respectively, are chosen.

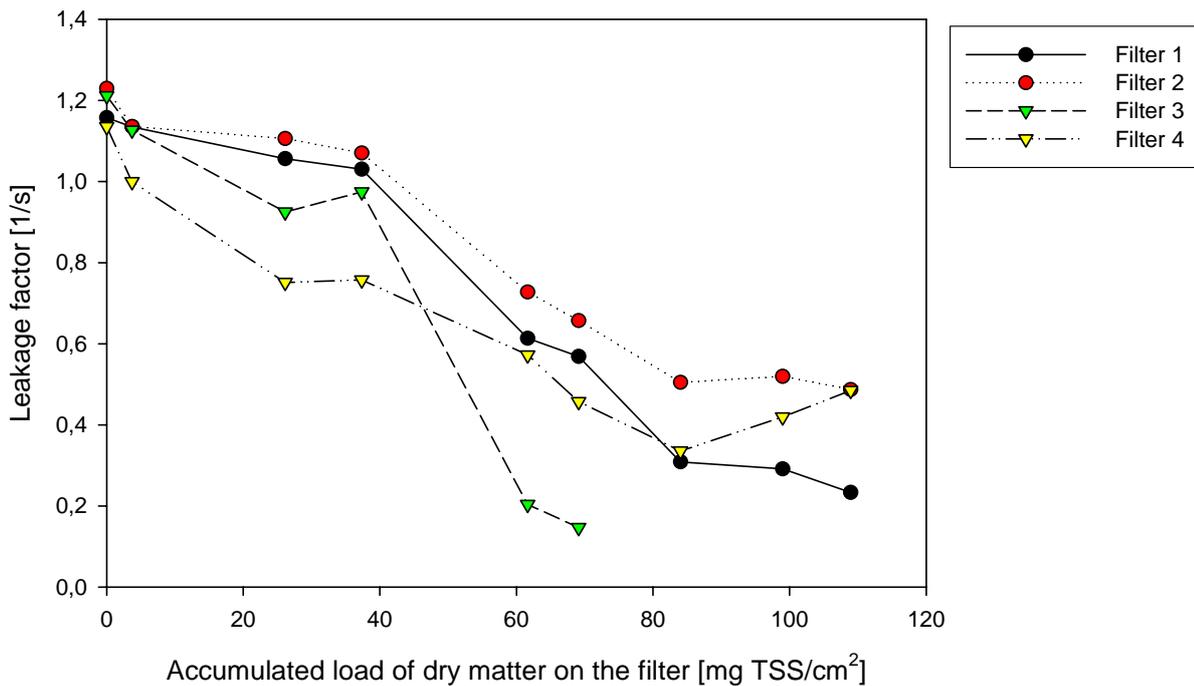


Figure 18: Development of the leakage factor with accumulated dry matter loads on the filters.

## 5.2 Pollutant removal

The removal of pollutions by means of the different unit operations implemented in the demonstration facilities was tested by phosphorous as a model substance. Phosphorous is one of the important pollutants to manage in stormwater treatment, and exhibits a number of characteristics which makes it suited as model substance.

### 5.2.1 Sedimentation tank and sand filters

To test the phosphate removal efficiency of the basin and the sand filters, 4 experiments were conducted in 2 of the setups. In the two first experiments, the stormwater was taken as it was, in the two later experiments; roughly 1 mg/L of phosphate was added (Figure 19 and Figure 21).

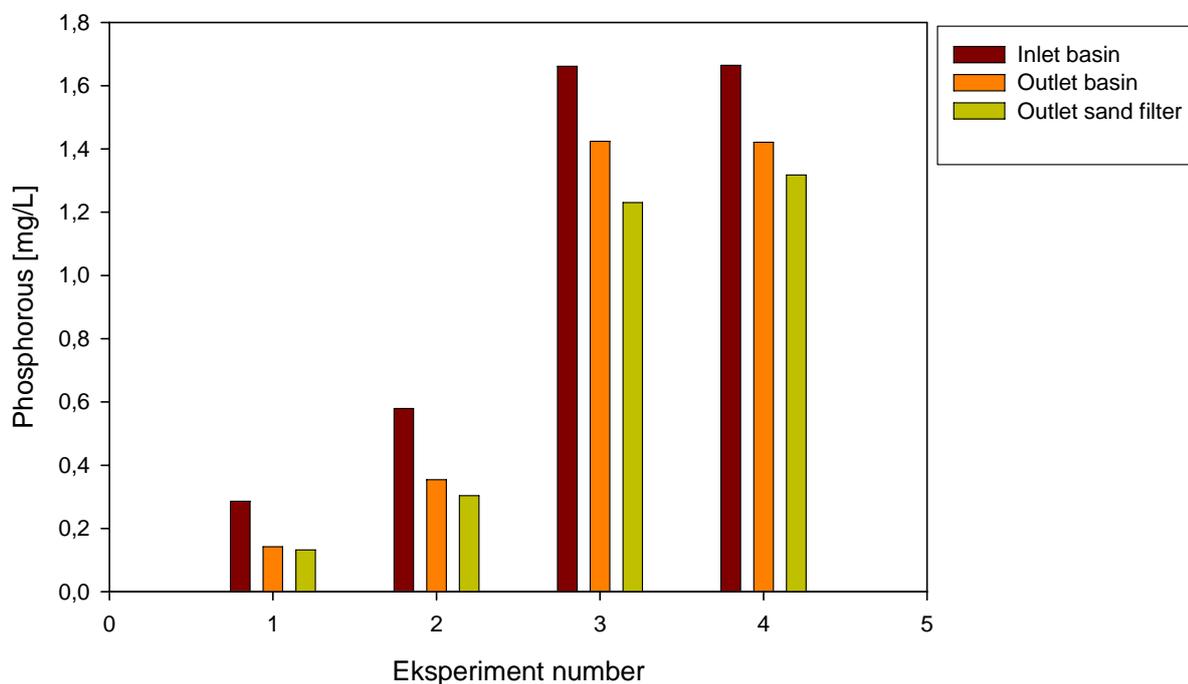


Figure 19: Phosphate removal in laboratory reactor 1, cf. text.

The results clearly show that phosphate is removed in the basin, but that the sand filters do not remove significant amounts of phosphate. Total phosphorous was also measured and showed similar behavior, however with higher variability due to larger errors in measurement and sampling.

### 5.2.2 Fixed media sorption filters

Prior to the tests, two different filter media were tested towards sorption characteristics: Skelsand – a Norwegian product of crushed, marine shellfish shells, and crushed Oyster shells (Figure 20).



Figure 20: Sorption test filters

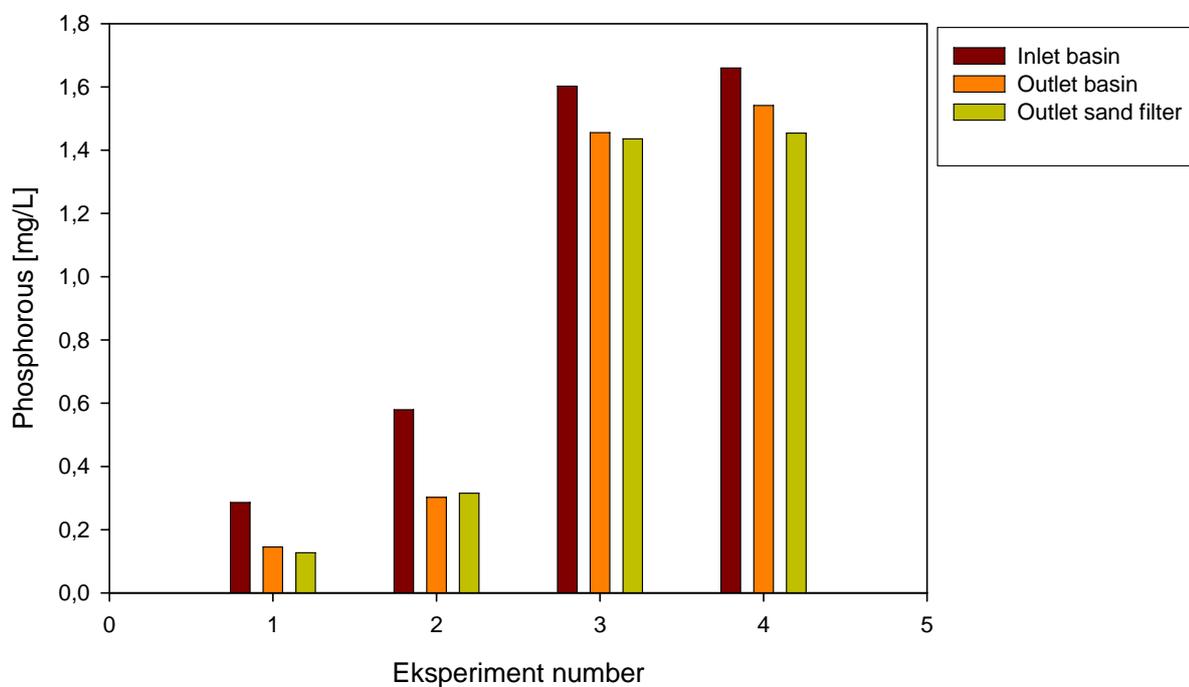


Figure 21: Phosphate removal in laboratory reactor 3, cf. text.



Figure 22 and Figure 23 show the results of a series batch experiments in which the two calcareous shell sands were treated with artificial phosphate solutions. In the experiments, 5 g filter media was mechanically agitated in 100 mL phosphate solutions of varying initial concentration at pH 8 and 20 °C. The residual phosphate concentration in solution was measured after 0, 0.5, 2 and 24 hours by the ascorbic acid method. The results clearly demonstrate that the sorption process is not instant. It is also evident that sand (Skelsand) had much better sorption properties than crushed oyster shells. Indeed, crushed oyster shells releases phosphorus into solution when the initial concentration was less than 150 mgP m<sup>-3</sup>.

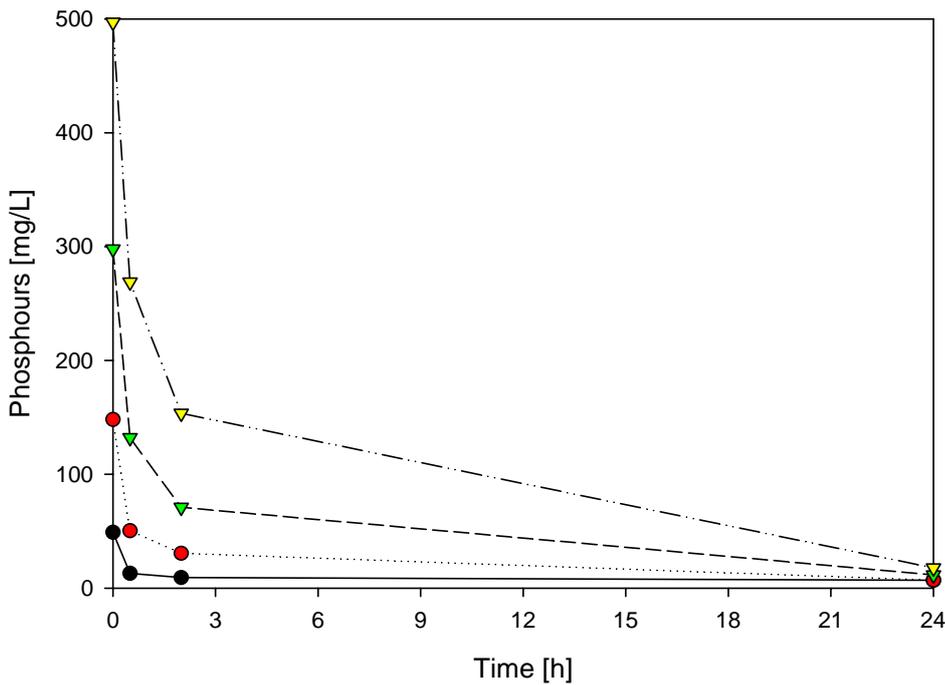


Figure 22 Kinetics of phosphate-sorption by calcareous shell sand (Skelsand).

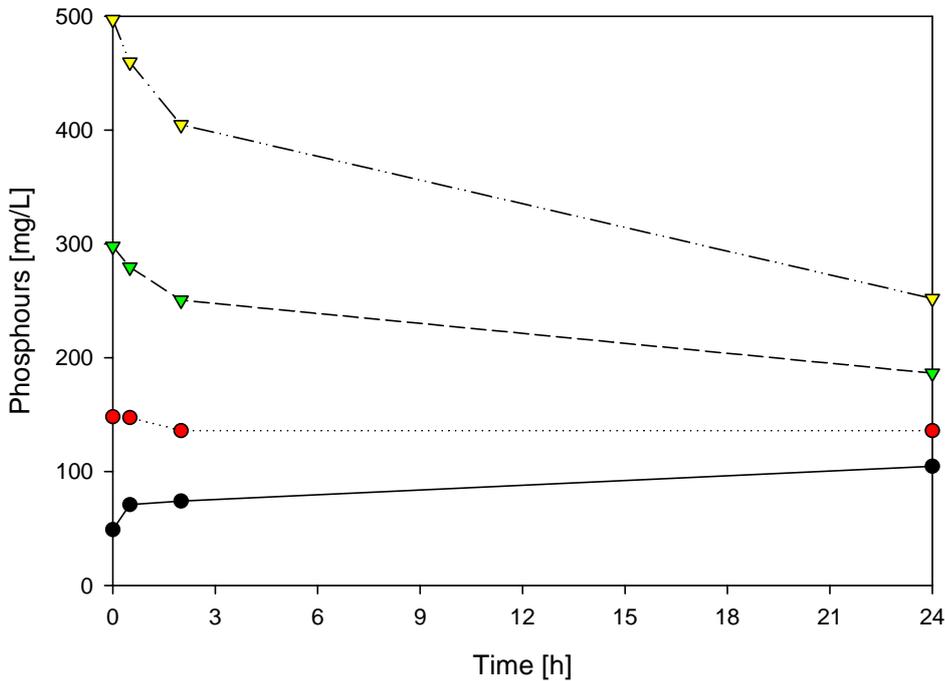


Figure 23 Kinetics of phosphate-sorption by calcareous shell sand (crushed oyster shells).

In addition to the time dependencies, the absorption capacity of the two materials was tested (Figure 24 and Figure 25). Also in this respect, Skelsand was the superior sorption material, and Skelsand was chosen as the fixed sorption media for further tests.

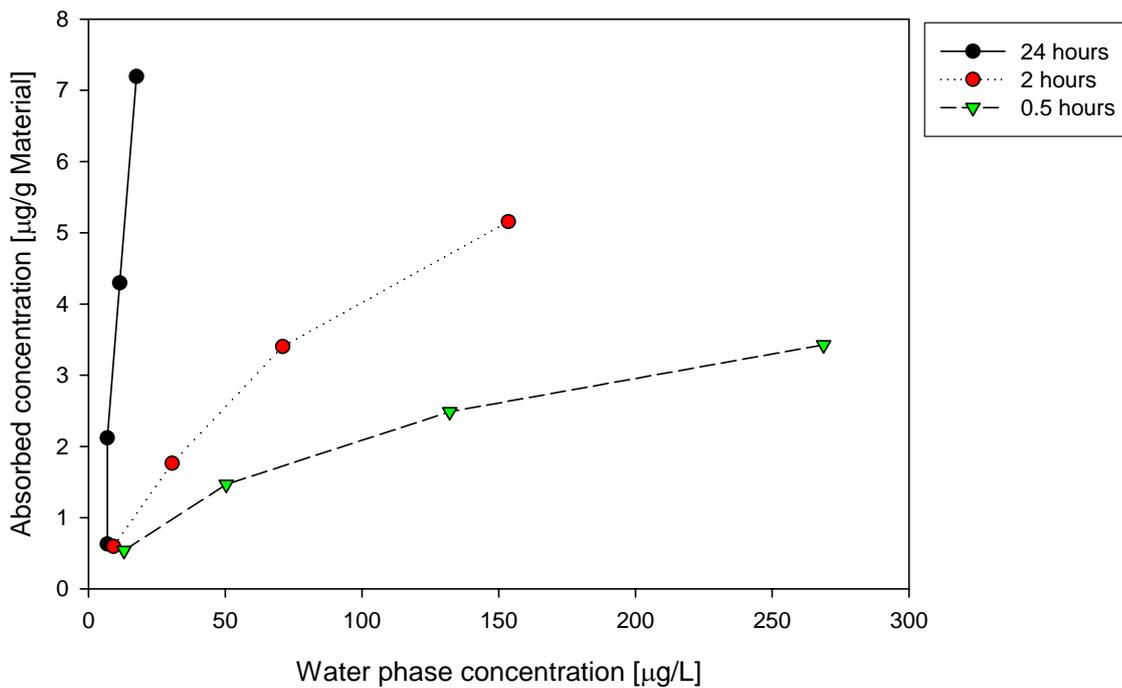


Figure 24 Phosphate-sorption isotherms by calcareous shell sand (Skelsand).

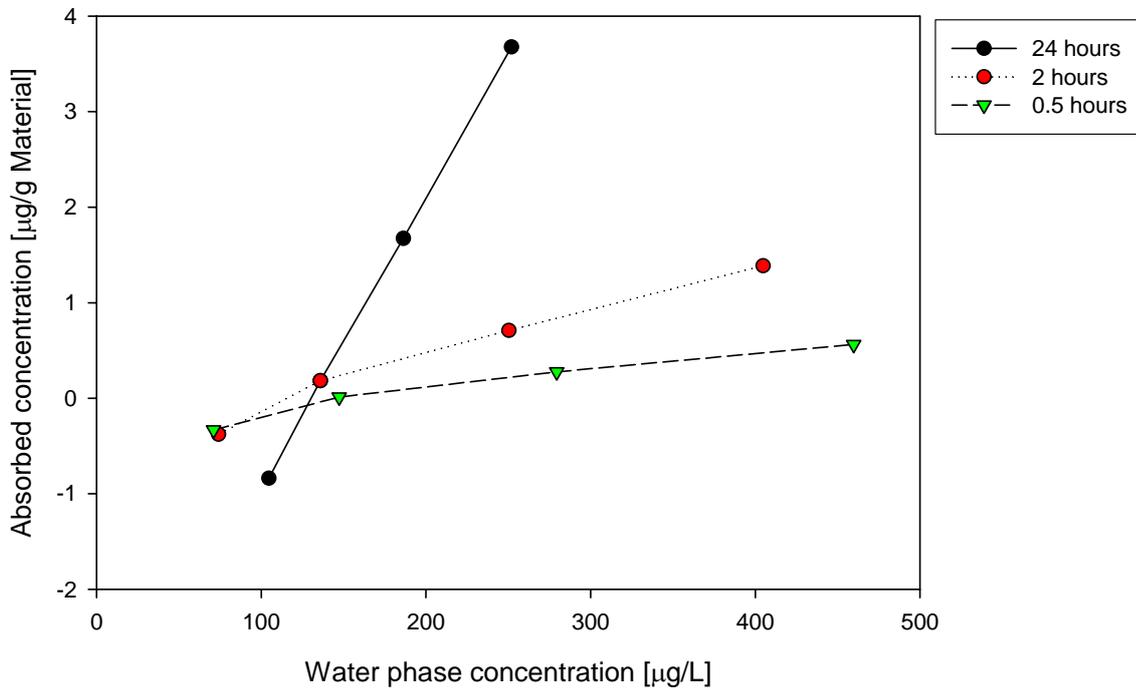


Figure 25 Phosphate-sorption isotherms by calcareous shell sand (crushed oyster shells).

Fixed media filters containing Skelsand was tested in two of the four laboratory reactors (reactor 2 and 4). The reactors were operated with sorption filters after the sand filters (Figure 26 and Figure 27). The sorption filter showed a significant reduction in the phosphorus outlet concentrations.

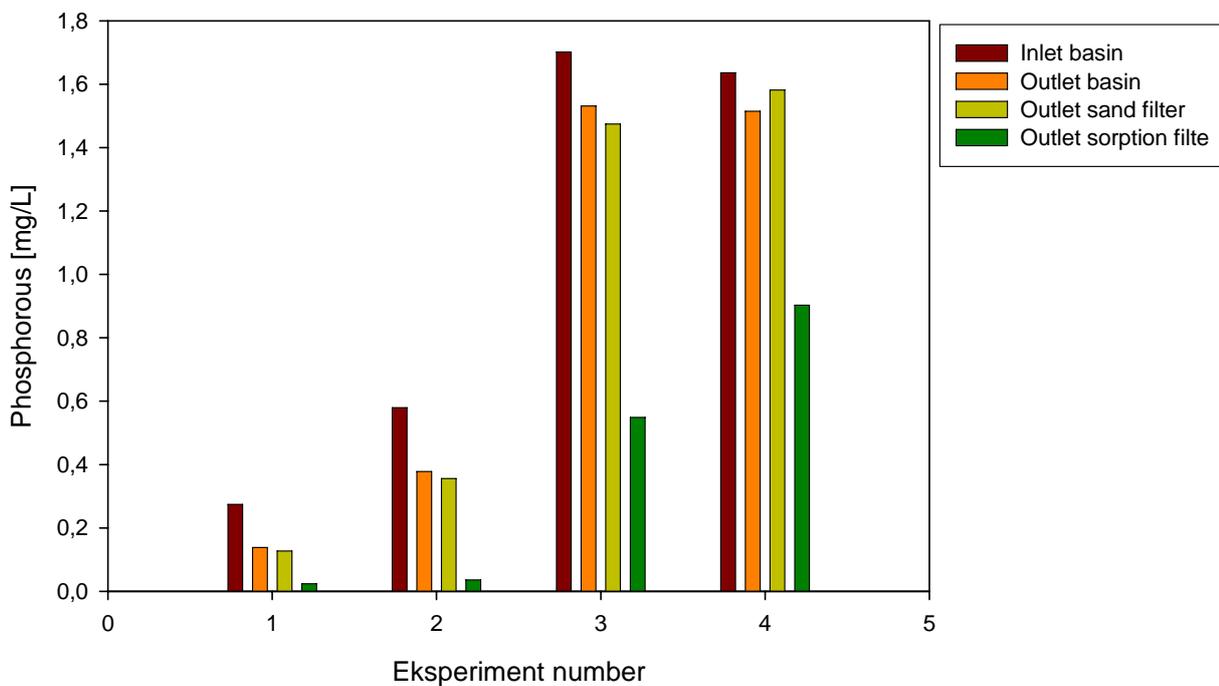


Figure 26: Phosphate removal in laboratory reactor 2, cf. text.

The results from the fixed media filter tests were applied in the design of the demonstration facility in Odense.

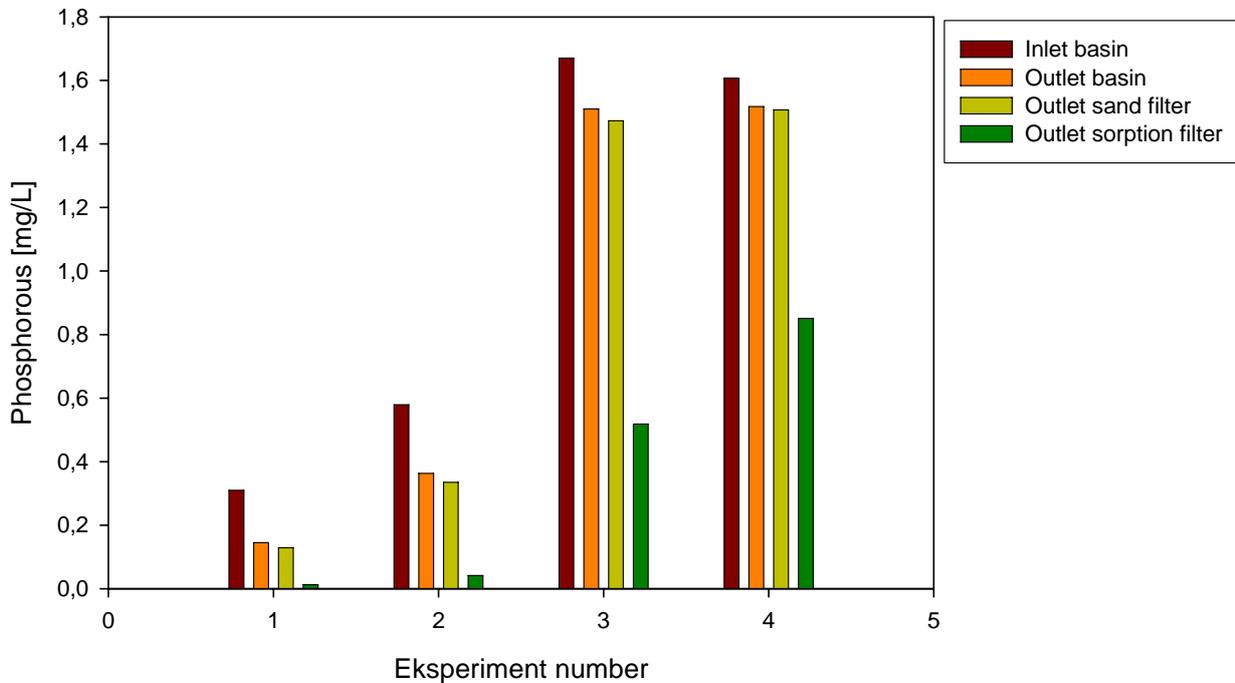


Figure 27: Phosphorous removal in laboratory reactor 4, cf. text.

### 5.3 Aluminum addition

The stormwater treated in the experimental setups was spiked with 1 mg/L of phosphate. Aluminum was added flow proportionally to the stormwater pumped into the basin. From day 0 to day 5, app. 1 mole of aluminum was added per mole of phosphorous. After day 5, 2 moles of aluminum was added per mole of phosphorous (Figure 28 and Figure 29). The addition of aluminum showed a significant effect compared to the experiments without aluminum addition (Figure 19 and Figure 21). However, doubling the aluminum addition caused only slightly improved removal rates. Based on the experiments, the demonstration facility in Silkeborg is designed for an average aluminum addition of a 1-2 mg/L.

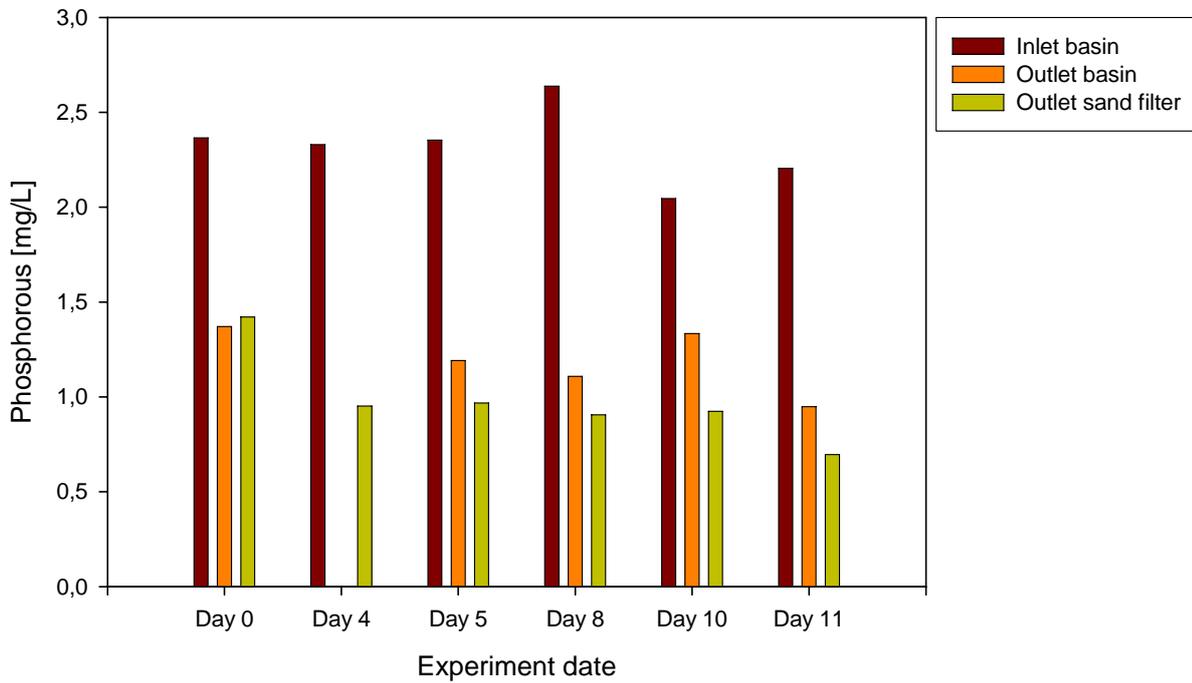


Figure 28: Phosphorous removal in laboratory reactor 1 with aluminum addition

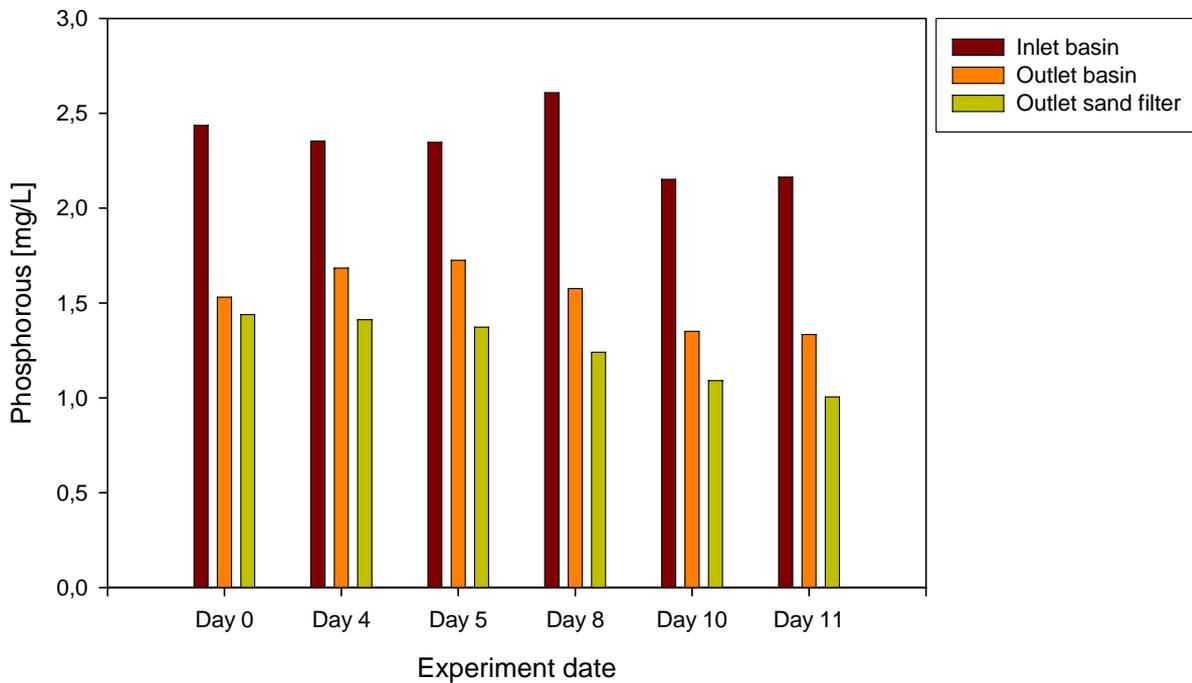


Figure 29: Phosphate removal in laboratory reactor 3 with aluminum addition

The results from the aluminum addition tests were applied in the design of the demonstration facility in Silkeborg.

## 5.4 Iron addition

Iron was added to laboratory reactor 2 on day 8 of the experiment shown in Figure 30. Throughout the experiment, the basin was operated with both sand filter and sorption filter. 3 g of Fe(III) was added on day 6, and the flow through the reactor stopped. On day 8, the flow was resumed. The experiment shows a clear reduction in outlet concentration after the basin as well as after the sorption filter, showing the effect of the iron enrichment of the bottom sediments in the basin.

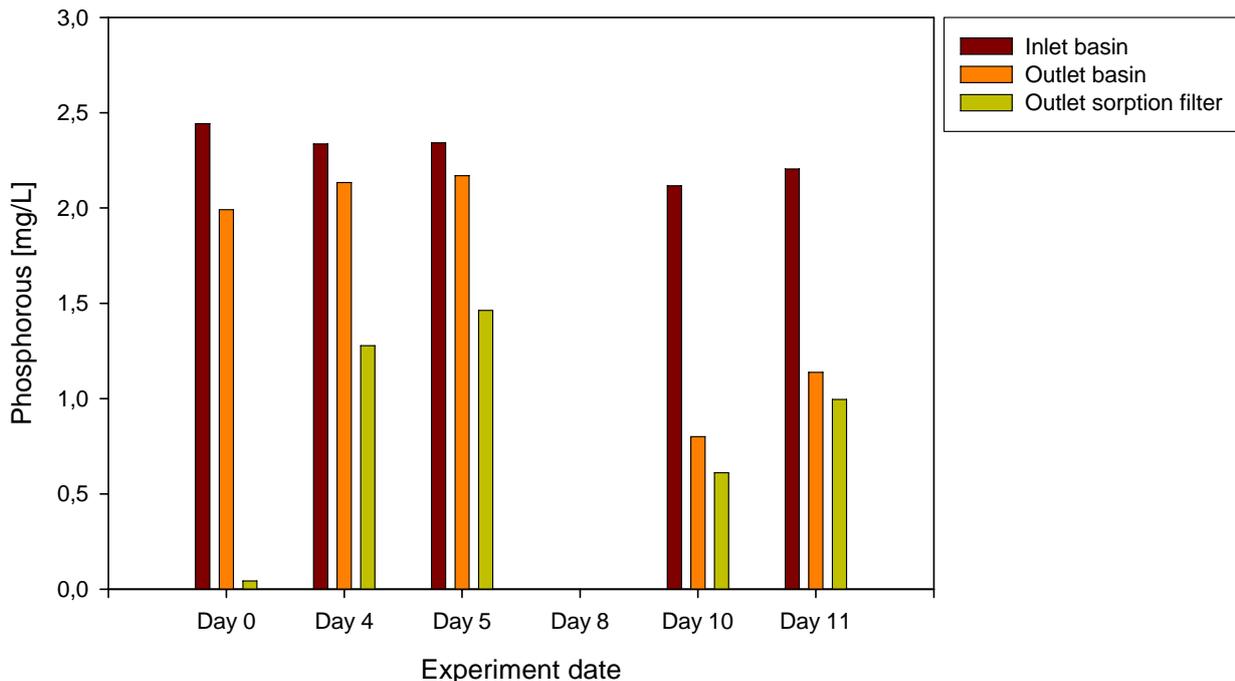


Figure 30: Phosphate removal in laboratory reactor 2 with sorption filter. Iron was added day 8, cf. text.

The results from the iron enrichment tests were applied in the design of the demonstration facility in Århus.

## 6 Concluding remarks

The experiments conducted yielded design parameters on sand filtration which – together with values obtained from literature studies – allow the sizing of the full scale filters in the demonstration facilities. All three sorption technologies were shown to be efficient with respect to pollutant removal in stormwater runoff. The laboratory testing yielded parameters necessary for design of the full scale technologies. Together with results obtained from literature, the tests allow the technologies to be implanted in the three full scale demonstration facilities.



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