

Test of compact drainage filter system for phosphorus**Preface**

This report presents preliminary results of the project 'Videreudvikling og optimering af målrettede dræn- og lavbundsvirkemidler' for the year 2021 and 2022. The project is funded by the Promilleafgiftsfonden for Landbrug and conducted as collaboration between SEGES and the Department of Agroecology (AGRO), Aarhus University, under the *Rammeaftale om samfinansierede forskningsprojekter af indgået mellem Aarhus Universitet og SEGES*. AGRO provides annual status reports to SEGES. For this study, WaterCare Filtration A/S has provided a commercially available sediment filter unit including a feeder pump under a separate agreement with SEGES. AGRO owns a separate reactive filter unit and is responsible for monitoring and data analysis in the project.

The first part of the report describes the experimental setup and the monitoring data for the period October 2020 to December 2021 and has been sent previously. Subsequently, we have updated the report with monitoring data for the period December 2021 to March 2022 when the monitoring ended.

This report has been subject to internal peer-review by associate professor Bo Vangsø Iversen, Department of Agroecology.

Lorenzo Pugliese and Goswin Heckrath

Department of Agroecology, Aarhus University.

December 2022

1. Background

Phosphorus (P) losses from artificially drained agricultural land can locally contribute to surface water eutrophication. These losses are a function of hydrological processes and long-term P accumulation in soils due fertilization practices, which is why agronomic mitigation options tend to be ineffective. However, as subsurface drainage systems concentrate water flow spatially, drainage filter technologies represent a potentially cost-effective end-of-pipe mitigation practice for P losses. The aim of the current project is to test such a compact drainage filter system under field conditions. The filter system comprises a sediment filter provided by WaterCare Filtration A/S and a unit containing a reactive, permeable filter material for retaining dissolved P. This status report presents results for the monitoring period October 2020 to December 2021.

2. Description of filter system and measurements

A full-scale experimental drainage P-filter system was established in December 2019 at the Fensholt catchment near the town of Odder, Denmark (55.996127, 10.101370). The system is fed with tile drainage water from an arable field of ca. 25 ha with loamy soils that have developed on Weichselian glacial till. A fraction of the total discharge is pumped into the filter system. The sediment filter unit is dimensioned to handle an annual load of about 15000 m³, while the reactive filter only has a tenth of this hydraulic capacity.

The filter system consists of the following main units (Fig. 1):

- At the system inlet, a AP35 pump (Grundfos A/S) feeds drainage water at flow rates of typically 1-1.5 l/s from the adjacent drainage ditch into the inspection well situated before the sediment filter.
- Dual porosity filter (DPF) unit, WaterCare Filtration A/S, for retaining sediments in drainage water and thus particulate P.
- Krohne™ electromagnetic flowmeter OPTIFLUX 3070 measuring the actual amount of water transported through the WaterCare Filtration A/S sediment filter unit.
- Flow divider for directing ca. 10% of the incoming drainage water to the reactive filter for retaining dissolved P.
- Filter box, CGK group BV, containing iron-coated sand (ICS) as reactive filter for dissolved P.
- RBC flume at the outlet of the filter box measuring the actual amount of incoming drainage water.
- Three ISCO™ samplers for continuous automated water collection are positioned in the pumping well (ISCO1), between the sediment and the reactive filter (ISCO2) and at the system outlet (ISCO3) (Fig 1).

2.1. Sediment filter

The DPF unit (Fig. 2) is manufactured mainly in PE (polyethylene plastic) and has a working volume of roughly 2.3 m³ with outer dimensions of 3.2 x 1.4 x 0.7 m (L x W x H) and an approximate weight of 200 kg. Standard drainage pipes (φ 110 mm) are used for the inlet and outlet. Moreover, eight vertical pipes (φ 110 mm) along the center axis of the filter allow flushing and removal of retained sediment. The technology is scalable and flexible in size and placement either by adding extra units in extension of each other or by enlarging the units, making DPF capable of handling flows ranging from 0.1 to >500 l/s.

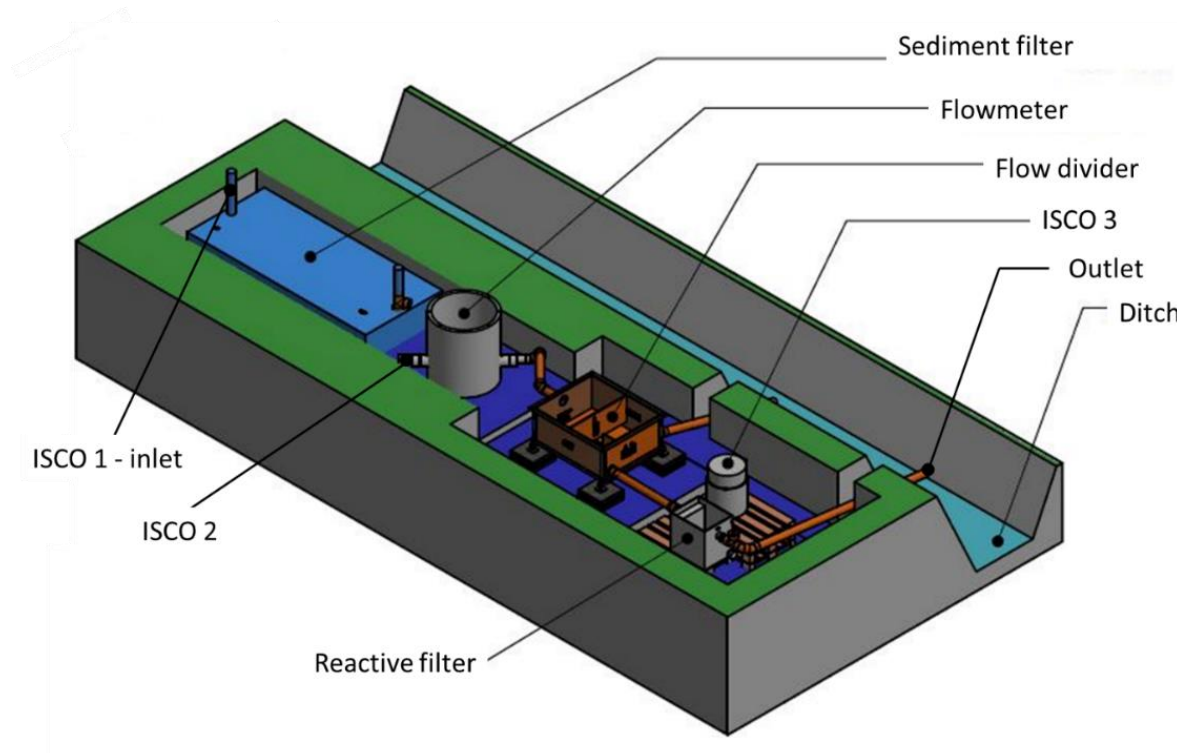


Figure 1. Schematic drawing of the drainage P filter system. The inlet is to the northwest. The pumping well feeding the system, the inspection well and the RBC flume are not shown.

Drainage water entering the DPF is uniformly distributed horizontally through a stack of sedimentation plates that are vertically spaced 2 to 4 mm from each other. Each sediment-plate measures 2.7 m² and has on its surface numerous low ridges perpendicular to flow direction creating shallow zones of no flow. As drainage water moves between the plates by laminar flow, particles settle out of the short water column into the no-flow zone on the plate surface where the sediment is trapped and contained. The water exiting the DPF filter passes through a flow meter before entering the flow divider.



Figure 2. Dual porosity filter (DPF) unit, WaterCare Filtration A/S, with flushing pipes extending vertically and inspection well installed at the Fensholt site.

2.2. Filter box with iron-coated sand

The purpose-built filter box (0.61 x 0.51 x 0.60 m; L x W x H) is constructed from PVC plates and equipped with inlet, outlet, overflow and a chamber containing the reactive filter material (Fig. 3). The inlet and outlet are aligned and placed on opposing sites of the filter box. A 3-cm difference in height between inlet and outlet represents the minimum hydraulic head facilitating water movement through the filter box. The internal 48-l chamber for holding the filter material is completely sealed on the sides (Fig. 3). Large circular openings (ϕ 40 mm) are present on the top and bottom PCV plates of the internal chamber to permit water flow through the chamber. A stainless steel wire mesh (Streno, Denmark) with an opening of 1 mm is placed respectively on top and below the perforated bottom and top plate to hold the filter material in place. During the operation period, autumn 2019 to spring 2021, granular ICS was used as filter material consisting of sand grains covered by thick layers of iron oxide, a waste product from rapid sand filters used for the deferrization of drinking water. Air-dried ICS (2 - 5 mm in diameter) was placed in the internal chamber ensuring homogenous packing. Drainage water enters the filter box from the left, flows upwards through the ICS in the internal chamber and exits the filter box at the outlet on the left (Fig. 3). Physical properties of the material have been previously determined by Vandermoere et al. (2018). Due to the reduced hydraulic conductivity occurring in late winter, the first ICS batch (Pidpa, Belgium) was replaced by a new one at the end of February 2021. During the last drainage season, from October 2021 to March 2022, a new ICS batch provided by [De Watergroep, Belgium](#), was used for the reactive filter.

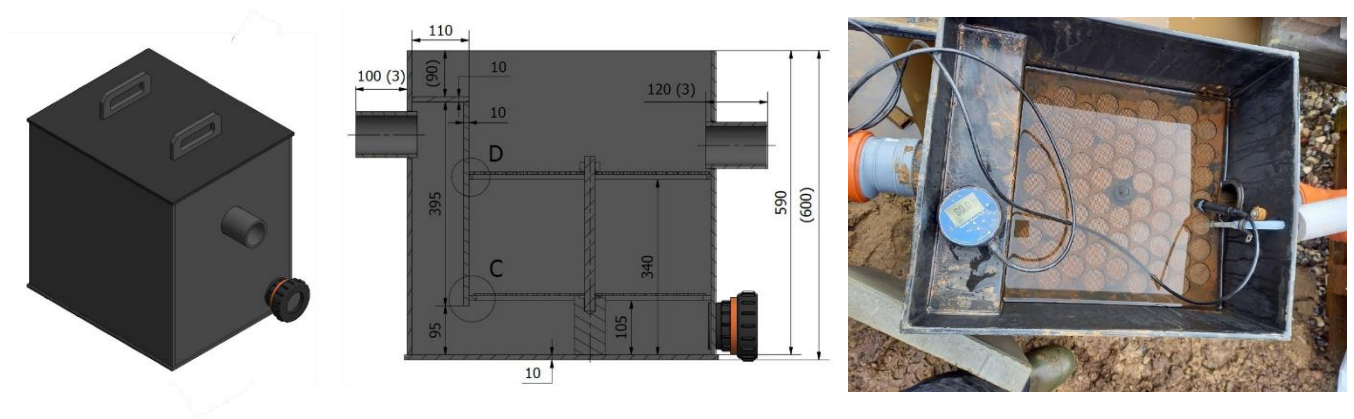


Figure 3. Schematic drawings of filter box including a cross-sectional view (left) and filter box in operation (right).

2.3. Flow measurements

Water discharge through the sediment filter was measured by an electro-magnetic flowmeter (Krohne A/S OPTIFLUX 3070), counting the number of 100-L pulses. The pulse data were logged every 15 min by a MadgeTech Pulse101A and subsequently converted into flow data. Additionally, tile drain discharge for the entire drainage catchment from which water was abstracted to the filter system was measured. Also in this case, a Krohne A/S OPTIFLUX 3070 flow meter was used. For measuring the hydraulic load of the reactive filter, an RBC steel flume (Clemmens et al., 1984) with a truncated V-shape was located at the outlet of the filter box. The flume water levels were logged every 15 minutes by a Campbell data logger. Discharge measurements were calibrated.

2.4. Drainage water analyses

Each ISCO sampler collected 30 ml drainage water once per hour pooling it on a daily basis in 1-litre polyethylene bottles. Every third week, the drainage water samples were transported to the laboratory and stored at 2 °C before analysis. During low flow periods, informed by the drainage discharge hydrograph, samples were pooled proportionally before analysis. Otherwise, samples were analyzed for each day.

All water samples were analyzed for pH with a pH meter (PHM220, MeterLab®), electrical conductivity with a conductivity meter (CDM210, MeterLab®), and turbidity (NTU) with a turbidity meter (Hach 2100AN, Hach). Total P (TP) was determined after persulfate digestion in an autoclave (European Standard, EN ISO 6878, 2004) as molybdate reactive P (MRP) according to Murphy and Riley (1962). For total dissolved P (TDP) samples were treated likewise, however, after filtration through 0.45 µm cellulose-acetate membrane filters. The difference between TP and TDP is considered to represent particulate P (PP). Suspended sediment (SS) in selected drainage water samples was measured as filter residue by weight difference after filtering 0.15 l drainage water through 0.45 µm cellulose-acetate membrane filters.

and subsequent drying at 105°C for three hours. Relationships between SS and NTU were then used for estimating SS concentrations in all remaining drainage water samples. Phosphorus and sediment loads were calculated by multiplying daily discharge with the corresponding concentrations.

2.5. System maintenance

The DPF unit was cleaned by spooling 5th November 2021 and 1st February 2022. The dirty water was pumped out.

2.6. Flocculation experiment

During February 2022, we tested the effect of aluminium sulfate additions as flocculant for fine particulate material in drainage water. For adding flocculant in the inspection well before the DPF unit, we vertically installed a 100-mm diameter tube that was perforated at the bottom. This end of the tube was submerged 15 mm below the water surface to enable contact between drainage water and flocculant and steady dissolution. Consolidated 2-kg blocks of $\text{Al}_2\text{SO}_4 \cdot x\text{H}_2\text{O}$ (Kemira Water Danmark A/S) containing 9% of Al were inserted in the tube on 01/02 (4 blocks), 02/02 (3 blocks), 04/02 (4 blocks), 06/02 (5 blocks), 09/02 (3 blocks), 11/02 (5 blocks), 15/02 (2 blocks), 17/02 (3 blocks), 22/02 (3 blocks).

During the flocculation experiments between 1st February and 1st March, variations of the electrical conductivity (EC) in water were recorded continuously at intervals of 1 minute in different filter compartments. To this end, EC probes (TETRACON 325, WTW, Weilheim, Germany) were connected to portable meters (Profiline Cond 3110, WTW, Weilheim, Germany) and located at the inlet of the filter system, at the outlets of the sediment and the reactive filter. Data were collected by USB cable at the end of the experiment.

3. Results and Discussion

3.1. Monitoring period 2020 to 2021

The hydraulic loading (Q) to the sediment filter during the 2020-2021 season reached 3.1 l/s (Fig. 4). Higher peaks were recorded between February 2020 and the beginning of June 2021. Peak reduction generally occurred in the 2-5 days following isolated rain events. The exceedance of the nominal flow rate of 1.5 l/s remains unexplained. During the summer the monitoring was put on stand-by and then resumed at the beginning of November 2021. The original pump fused in October 2021 and was replaced. The replacement pump maintained more constant flow rates through the filter varying little around 1.5 l/s.

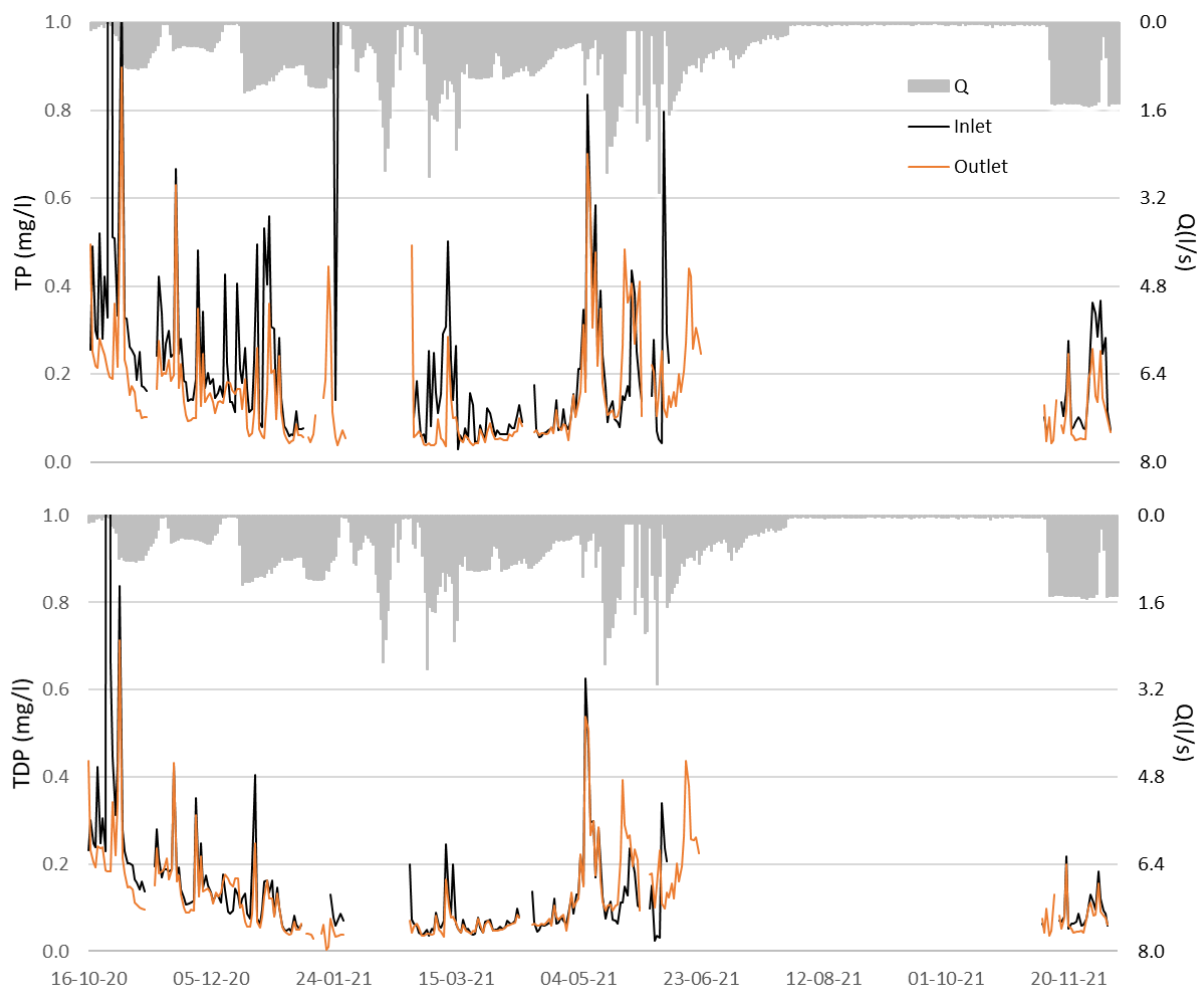


Figure 4. Daily values of total phosphorus (TP, top) and total dissolved phosphorus (TDP, bottom) for the sediment filter at the inlet and outlet. The hydraulic loading (Q) is given on the secondary axis.

Apart from technical problems with the pump, the filter system stopped working occasionally due to frozen pipes. The average hydraulic retention time (HRT) for the sediment filter was 92 minutes with a standard deviation of 60 minutes while minimum and maximum HRT were 32 and 177 minutes, respectively. The average Q to the filter box during the 2020-2021 season was 0.1 l/s, corresponding to a stable 10% of the Q treated by the sediment filter in the same time period (Fig. 5). During the runoff season 2020/21 the hydraulic loads to the sediment and the ICS filters were, respectively, 18000 m³ (Table 1) and ca. 1500 m³ (Table 2). Note that the number of operational days was slightly different for the sediment and the ICS filter (data not shown).

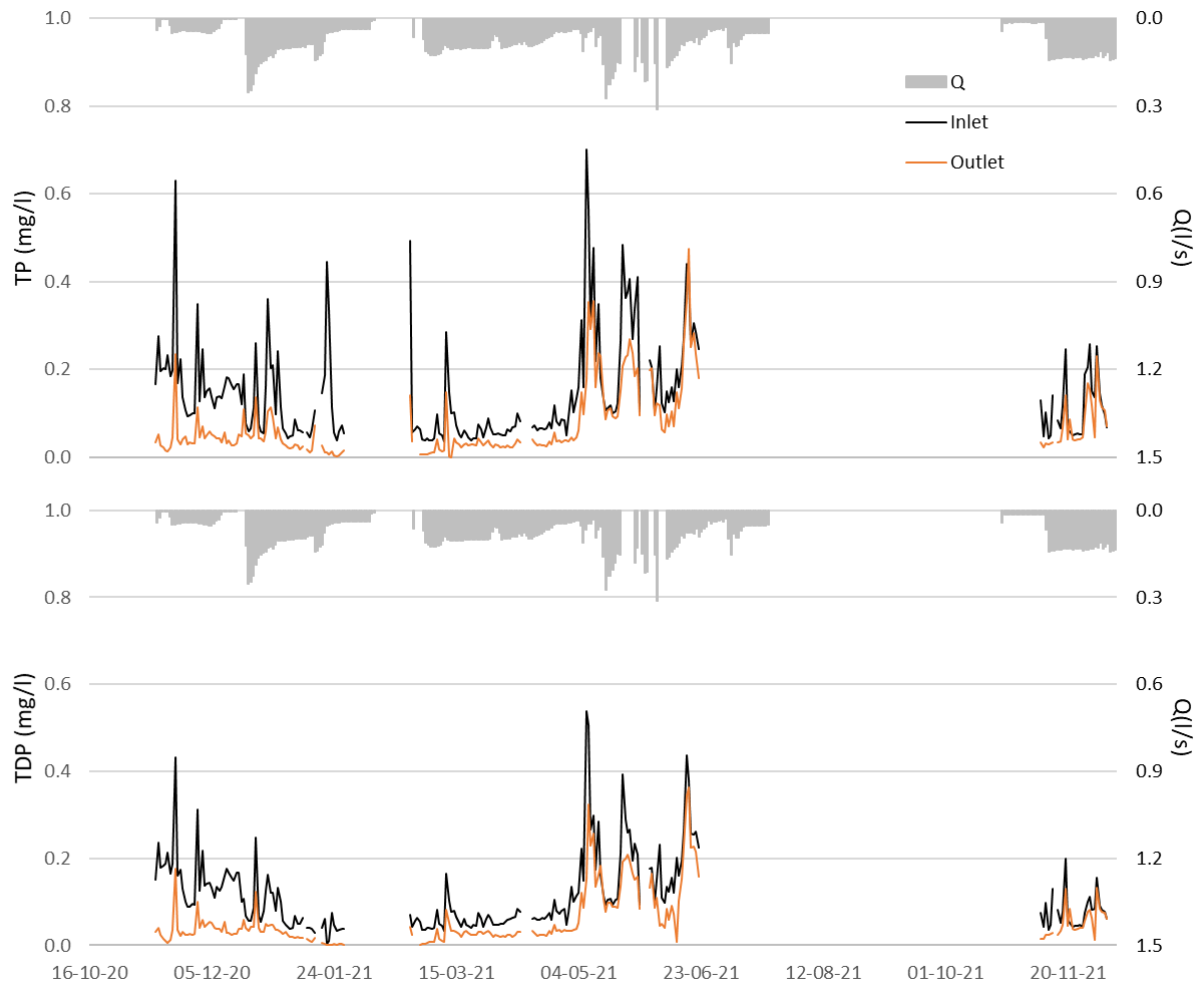


Figure 5. Daily values of total phosphorus (TP, top) and total dissolved phosphorus (TDP, bottom) at the inlet and outlet of the ICS filter in the filter box. The hydraulic loading (Q) is given on the secondary axis. Note the different scaling of the flow axis compared to Figure 4 as approximately 10% of Q from the sediment filter was directed to the ICS filter.

Total P concentrations in drainage water at the sediment filter inlet varied substantially between 0.03 and 2.47 mg TP/l, while TDP varied between 0.04 and 0.84 mg/l (Fig. 4). Values of TP at the filter box outlet ranged from below detection limit to 0.36 mg/l, while TDP ranged from below detection limit to 0.32 mg/l (Fig. 5). The three highest peaks in October and January were due to partial remobilization of P during cleaning operations of the filter system. In general, TP and TDP concentrations at the inlet of the filter system were highly correlated ($r=0.89$). However, during the monitoring period the proportion of TDP to TP varied between 30 and 80% on a monthly basis and averaged 60% (Fig. 3; Table 1). The missing data in February 2021 are due to very low temperatures and consequent system freezing. Other missing data relate to pump failure. The pH in drainage water was relatively high and varied between 7.3 and 8.5 at the system inlet.

Turbidity was measured as a surrogate for suspended sediment in drainage water as the analysis of the latter is much more expensive. Like P concentrations, turbidity in drainage water at the system inlet varied greatly (Fig. 6) and was characterized by periodic peaks (Borah et al., 2003). Often turbidity was lower at the outlet of the sediment filter indicating sediment retention. Turbidity in drainage water is often positively correlated with hydraulic loading as we observed during the drainage season 2019/20 (Fig. 7; $r=0.68$). However, this correlation was much weaker during the season 2020/21 (Fig. 7; 0.28).

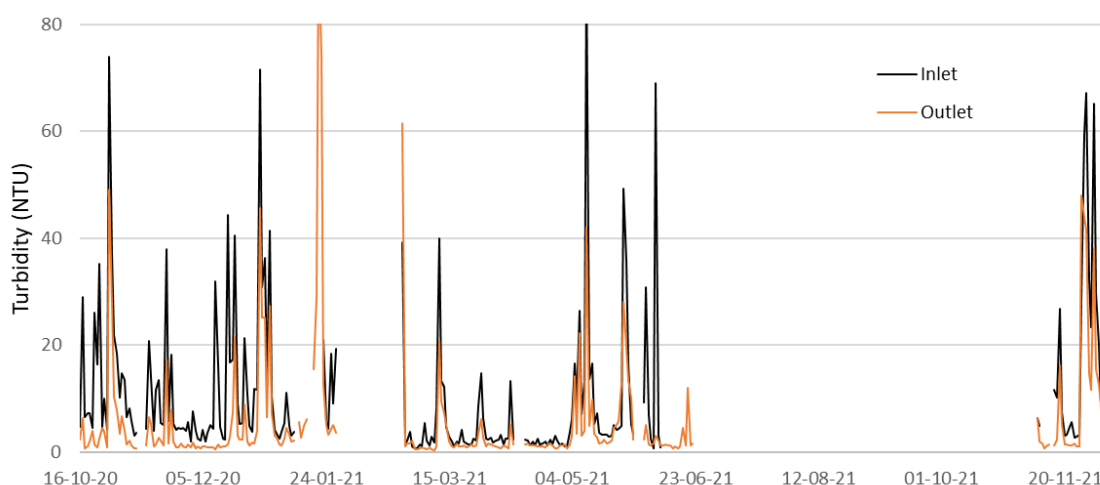


Figure 6. Turbidity (TDP) in NTU at the inlet and outlet of the sediment filter.

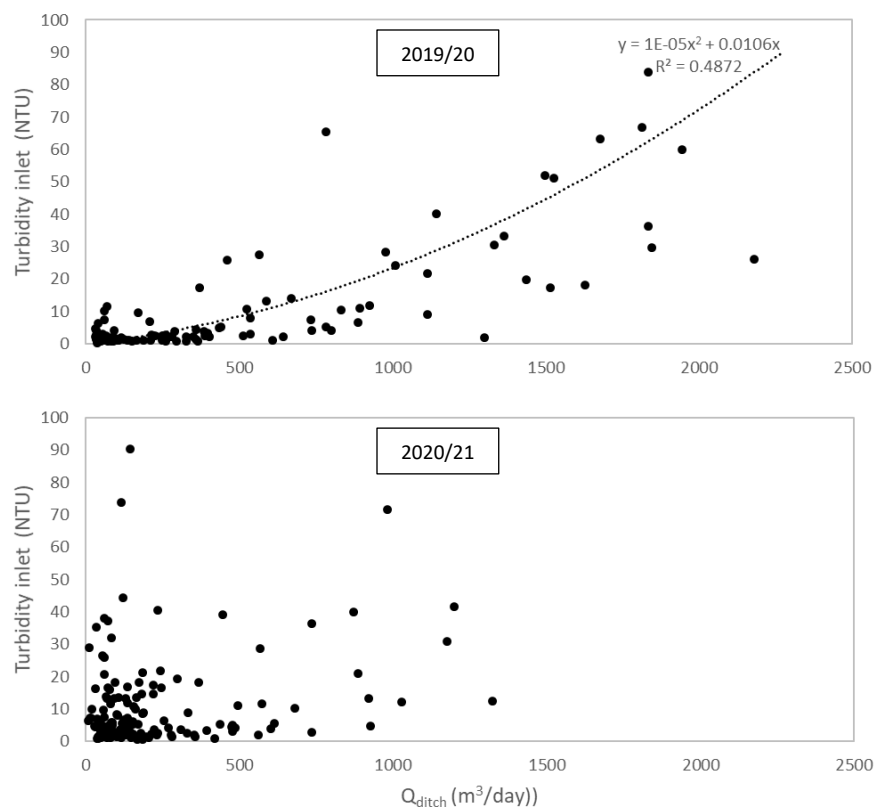


Figure 7. Relationship between turbidity (NTU) and hydraulic loading in the ditch (m³/day) for the season 2019/20 (top) and 2020/21 (bottom). This hydraulic loading exclusively represents the drainage discharge from the whole drainage catchment.

Suspended sediment in drainage water comprises the fine particle size fractions especially below 5 μm at our site (data not shown). Phosphorus tends to bind strongly in this size fraction, which is why sediment is an important carrier for P. Accordingly, we observed a strong correlation between turbidity and particulate P (Fig. 8; $r=0.92$).

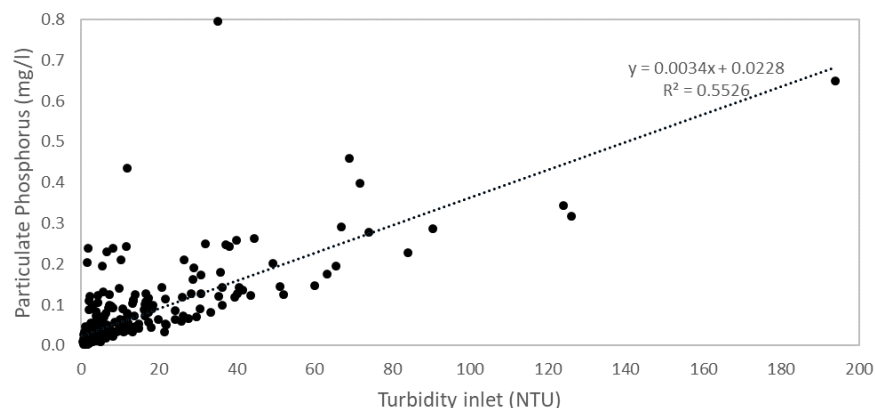


Figure 8. Relationship between particulate phosphorus (PP, mg/l) and turbidity (NTU) at the system inlet for the whole monitoring period.

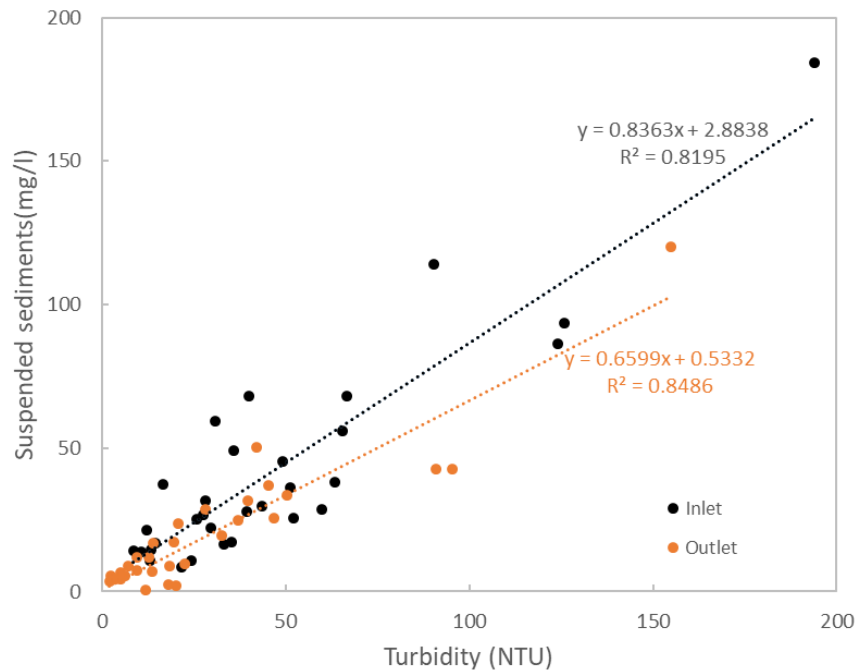


Figure 9. Relationships between suspended sediments (mg/l) and turbidity (NTU) at the inlet and outlet of the sediment filter for selected samples. The linear regression was used for estimating suspended sediment concentrations.

To estimate sediments loads and sediment retention in the filter system we regressed measured suspended sediment on turbidity (NTU) for selected drainage waters samples throughout the monitoring period. For both the inlet and the outlet of the sediment filter, linear relationships between suspended sediment and NTU explained a large degree of the variation (Fig. 9). On average, suspended sediment concentrations were lower at given turbidity at the outlet compared to the inlet. This is consistent with the sedimentation and removal from the water column of relatively heavier mineral particles in the sediment filter.

Table 1. Monthly hydraulic loading (Q), load and removal of total phosphorus (TP), total dissolved phosphorus (TDP), particulate phosphorus (PP) and suspended sediments (SS) for the sediment filter. Suspended sediment was estimated from a calibration relationship with turbidity. Months with incomplete data are shown in bold.

Month	Q (m ³)	TP		TDP		PP		SS	
		Load (g)	Removal (%)	Load (g)	Removal (%)	Load (g)	Removal (%)	Load (kg)	Removal (%)
oct-20	613	243	30	190	23	53	56	11	73
nov-20	1299	276	31	207	16	69	75	11	79
dec-20	1798	448	28	250	2	197	75	30	76
jan-21	2133	253	48	74	20	180	75	13	66
feb-21	1825	13	35	17	16	5	67	2	49
mar-21	3245	463	31	228	10	235	69	23	72
apr-21	1904	133	13	108	2	25	60	8	72
maj-21	2842	581	6	403	-7	178	42	33	63
jun-21	2398	287	-73	164	-155	123	45	20	75
nov-21	2646	270	27	162	14	108	45	29	64
dec-21	697	155	34	75	12	80	55	17	56

Loads and removal rates are reported separately for the sediment and the ICS filter (Table 1 and 2). With the exception of February 2021, the monthly TP loads to the sediment filter varied between 130 g and 581 g, while the corresponding PP loads varied between 25 and 235 g. The February TP load was exceptionally low in part because the system was not operational during a prolonged period of freezing. Our results illustrate the extent of sediment loading to drainage filter systems. For the drainage season 2020/21, we estimated a total sediment load of 150 kg. If this sediment is not retained, it can easily clog porous reactive P filters such as the ICS filter in our study. On average for the drainage season 2020/21, the sediment filter retained 71% of the estimated sediment load. Similarly, 64% of PP were retained during the same period. The small discrepancy between sediment and PP retention can be explained by the temporal variability of the sediment composition (Fig. 8) as well as uncertainties in the sediment prediction (Fig. 9). While sediment and PP retention were effective, 85% of the TP load passed through the sediment filter entering the ICS filter. A challenge with the sediment filter is potential remobilization of P. During the monitoring period, this was observed especially for TDP in late spring. We have no data that could explain the mechanism. However, both chemical and biological processes could have led to a release of P from the solid into the liquid phase in the sediment filter. Such release processes are important to investigate in the future. Likewise, the effective storage capacity of the sediment filter has to be better understood including the required frequency of filter cleaning.

Table 2. Monthly hydraulic loading (Q), load and removal of total phosphorus (TP), total dissolved phosphorus (TDP) and particulate phosphorus (PP) for the ICS filter. Months with incomplete data are shown in bold.

Month	Q (m ³)	TP		TDP		PP	
		Load (g)	Removal (%)	Load (g)	Removal (%)	Load (g)	Removal (%)
nov-20	59	12	76	10	79	2	14
dec-20	225	30	58	23	61	7	18
jan-21	214	17	72	9	72	8	79
feb-21	27	3	82	1	68	2	72
mar-21	272	18	58	15	58	3	68
apr-21	190	11	55	10	57	1	28
maj-21	284	53	30	41	29	12	15
jun-21	209	39	23	36	33	3	-140
nov-21	254	25	35	18	33	8	35
dec-21	68	9	13	6	19	3	-3

During winter, the TDP retention in the ICS filter varied between 57 and 79% before it dropped late in the season to about 30% (Table 2). The main purpose of the reactive filter is to bind dissolved inorganic P by reaction with the iron oxides of the ICS (Vandermoere et al., 2018). For the drainage season TDP, retention averaged 51% and the filter has thus performed satisfactorily, although to a lesser degree than in a comparable setup during trials in Belgium (Vandermoere et al. 2018). One explanation could be the remaining sediment load to the ICS filter, which was substantially higher than in the Belgian trials. During the course of the runoff season, our ICS filter suffered from increasing clogging probably lowering the accessibility of P binding sites.

The monthly retention of TP for the overall system varied between -33% and 88% during the 2020/21 season averaging 61%. June was an exceptional month. At this time the DPF was beginning to fill up with sediment and turned from sink to source for TDP (Table 1) which explains the negative TP retention. We consider the overall performance promising especially since upscaling the sediment filter and better filter maintenance may yield ready improvements. Other end-of-pipe solutions such as constructed wetlands tend to have lower TP retention efficiencies (Mendes et al. 2020). Under Danish conditions, previous studies at different sites have estimated a TP overall retention for surface-flow constructed wetlands varying between 41 and 51% (Mendes et al. 2018).

During the drainage season 2021/22, the addition of flocculants to drainage water at the inspection well and a different type of ICS will be tested.

3.2. Monitoring period December 2021 to March 2022

As the previous reporting period did not include data from all of December 2021, we present here an update for the whole month. After replacing the pump, the hydraulic loading to the sediment filter was more stable around 1.5 l/s compared to the first monitoring period (Fig. 10). With the start of the flocculation experiment in February, we attempted to reduce the pump rate by half. However, at this low pumping rate it proved technically difficult to maintain stable flow. Inlet concentrations of TP and TDP tended to be somewhat lower during winter 2021/22 than in the previous winter (Fig. 10).

Concentration peaks above 0.5 mg TP/l occurred in February. Especially in February, the proportion of TDP in TP was comparatively low. While we observed moderate reductions of TP concentrations during passage of the sediment filter, TDP concentrations at the outlet of the sediment filter were distinctively low during February (Fig. 10).

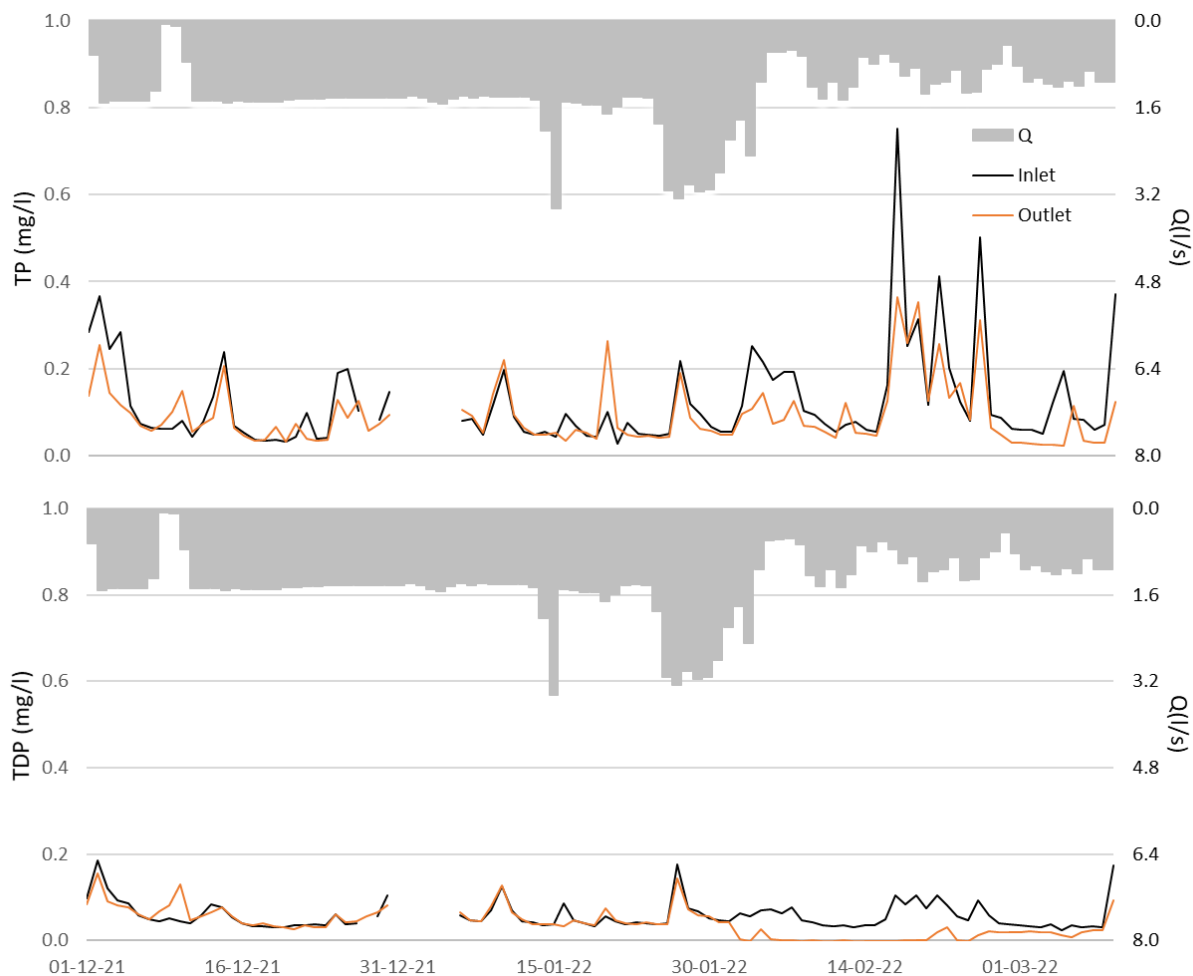


Figure 10. Daily values of total phosphorus (TP, top) and total dissolved phosphorus (TDP, bottom) for the sediment filter at the inlet and outlet, from December 2021 to March 2022. The hydraulic loading (Q) is given on the secondary axis.

During the monitoring period 2021/22, the different ICS filter material provided by De Watergroep tended to yield lower P retention than the ICS from Pidpa. In December 2021 and January 2022, TP concentrations at the inlet and outlet of the ICS filter were very similar (Fig. 11). However, during February 2022 outlet TP concentrations dropped typically below inlet concentrations. During the same period, the ICS filter had no effect on TDP concentrations, which remained very low at the filter outlet.

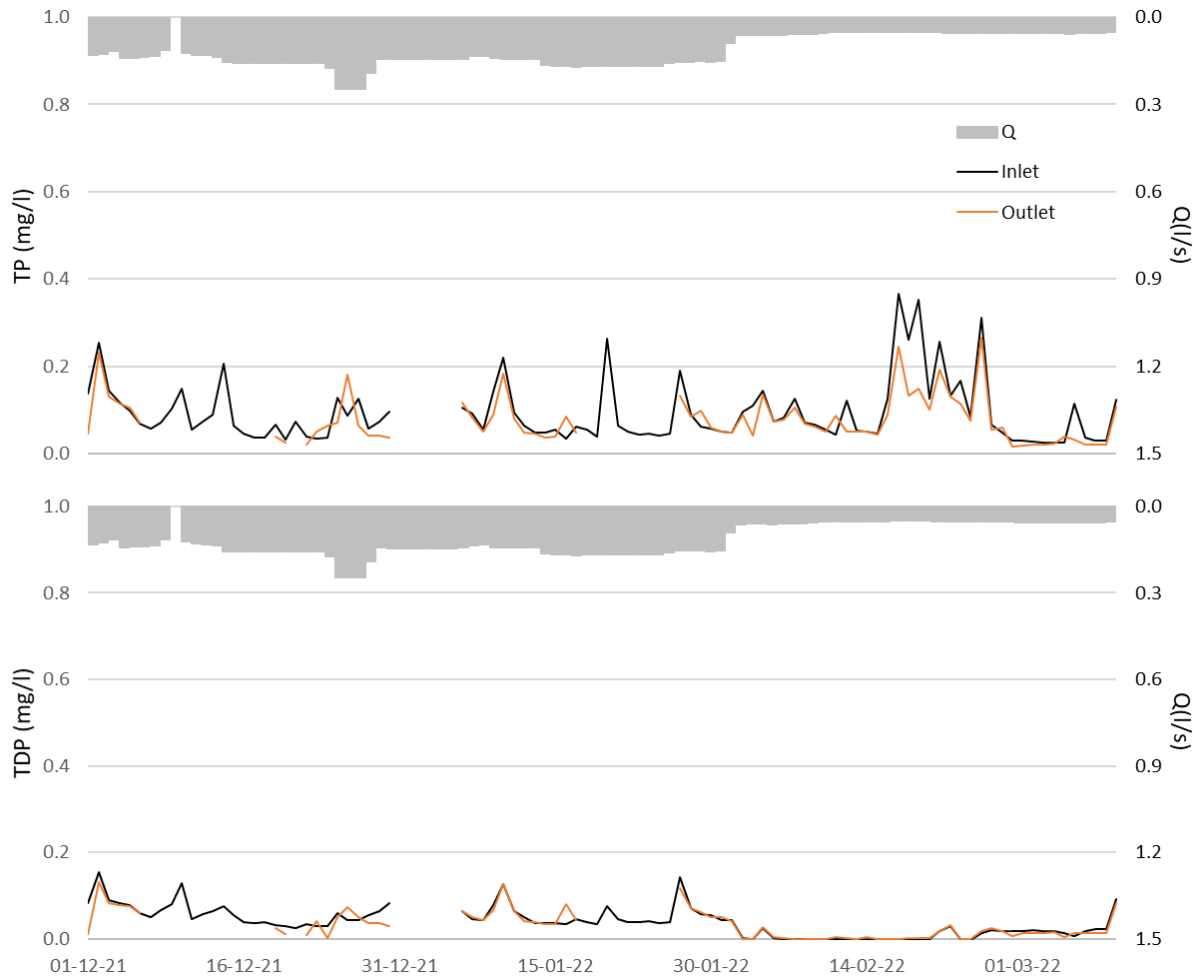


Figure 11. Daily values of total phosphorus (TP, top) and total dissolved phosphorus (TDP, bottom) at the inlet and outlet of the ICS filter in the filter box, from December 2021 to March 2022. The hydraulic loading (Q) is given on the secondary axis. Note the different scaling of the flow axis compared to Figure 4 as approximately 10% of Q from the sediment filter was directed to the ICS filter.

As for the first monitoring period, loads and removal rates are reported separately for the sediment and the ISC filters (Table 3 and 4). During December 2021 and January 2022, TP retention was low in the DPF unit compared with the previous year. This could have been due to declining storage capacity for sediment and the filter was therefore cleaned by spooling on 1st February. However, apparent suspended sediment removal by the DPF was much higher than the low PP retention indicated (Table 3).

The reason for this is not clear. During February and the flocculation trial, TP removal efficiency in the DPF increased. This was largely due to very high TDP removal, the highest observed during the whole filter operation (Table 3). At the same time, PP and suspended sediment retention became clearly negative. A likely explanation is the formation of colloidal amorphous Al hydroxide ($\text{Al}(\text{OH})_3$) in the DPF after addition of Al sulfate. These colloids have a high affinity for P and a rather low settling velocity. The data indicate that only some $\text{Al}(\text{OH})_3$ was retained in the DPF where it acted as effective sorbent for TDP (Fig. 10). The $\text{Al}(\text{OH})_3$ that passed the DPF led to negative PP and probably also suspended sediment retention. Whether $\text{Al}(\text{OH})_3$ acted on the original particulate matter in drainage water to increase the risk of remobilization in the DPF is not known.

Table 3. Monthly hydraulic loading (Q), load and removal of total phosphorus (TP), total dissolved phosphorus (TDP), particulate phosphorus (PP) and suspended sediments (SS) for the sediment filter, from December 2021 to March 2022. Suspended sediment was estimated from a calibration relationship with turbidity. Months with incomplete data are shown in bold.

Month	Q (m ³)	TP		TDP		PP		SS	
		Load (g)	Removal (%)	Load (g)	Removal (%)	Load (g)	Removal (%)	Load (kg)	Removal (%)
dec-21	3529	384	5	199	-7	185	2	37	53
jan-22	5013	356	-6	263	1	93	-7	26	58
feb-22	2632	456	22	154	89	302	-37	81	-51
mar-22	942	112	52	44	43	69	59	6	67

Until February, the De Watergroep ISC showed generally lower TDP retention compared to the Pidpa ISC of the first monitoring period (Table 2 and 4). This could be explained by the sorption properties of the materials. In December 2021 and January 2022, there was a marked release of PP from the ISC filter, which could have been caused by remobilization of fine colloidal material from the filter. In February, TDP removal efficiency was up again albeit at a low TDP load. Some of the $\text{Al}(\text{OH})_3$ that passed the DPF unit was retained in the ICS filter and visible on the surface (Fig. 12). During this period there was no evidence that $\text{Al}(\text{OH})_3$ colloids were lost from the ICS filter.

Table 4. Monthly hydraulic loading (Q), load and removal of total phosphorus (TP), total dissolved phosphorus (TDP) and particulate phosphorus (PP) for the ICS filter, from December 2021 to March 2022. Months with incomplete data are shown in bold.

Month	Q (m ³)	TP		TDP		PP	
		Load (g)	Removal (%)	Load (g)	Removal (%)	Load (g)	Removal (%)
dec-21	404	34	12	22	23	12	-57
jan-22	413	28	0	19	-6	9	-35
feb-22	135	16	14	1	64	15	14
mar-22	47	2	22	1	22	1	1



Figure 12. Insoluble $\text{Al}(\text{OH})_3$ flocs at the surface of the reactive filter observed on the 15-02-22.

Measurements of EC at the inlet (black line) varied between 220 and 650 $\mu\text{S}/\text{cm}$ in February 22, with significantly reduced values towards the end of the month (Fig. 13). This reflected the change in precipitation values recorded at the nearby meteorological station (DMI 10km grid, 10315) in the second half of February (83 mm) in comparison to the first half of the month (41 mm). Increased salinity and thus higher EC values were generally recorded in response to the Al sulfate addition by the meters located at the outlet of the sediment and reactive filter, except for the period 17-22 February (green area).

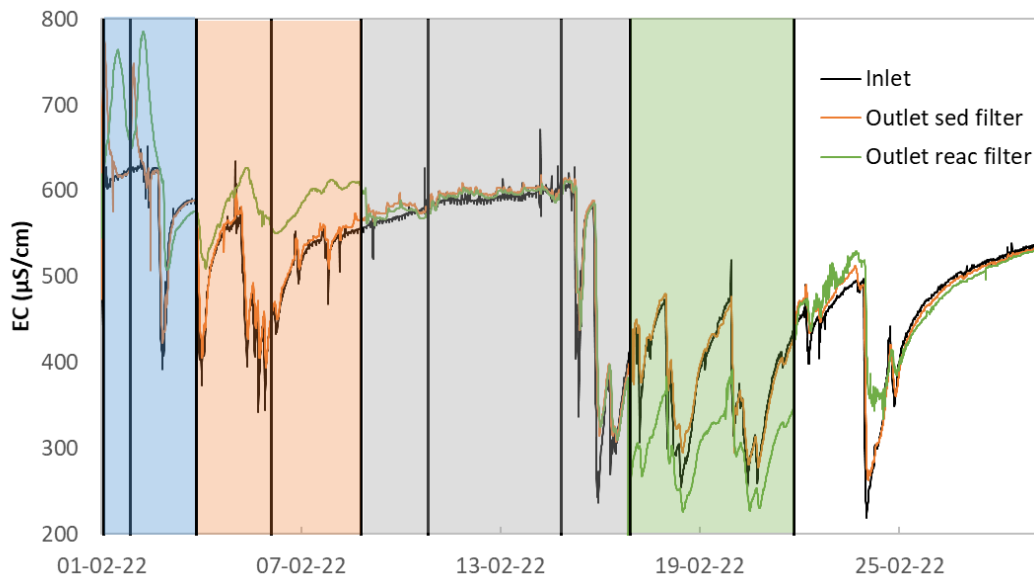


Figure 13. Daily values of electrical conductivity (EC) at the inlet, the outlet of the sediment and reactive filter during the flocculation experiment carried out in February 2022. The vertical lines indicate the

addition of aluminum sulfate blocks while the colored areas indicate different experimental phases characterized by similar EC trends.

In particular, the first two salt additions (vertical black lines) produced pronounced breakthrough curves (BTCs) reflecting the large salt addition in a relatively short time interval (one day) and the hydraulic retention time of the different units of the filter system (blue area). The third and fourth addition produced a moderate rise of EC at the outlet of the sediment filter, while the effect at the outlet of the sediment filter was modest. The peak arrival time of the BTCs was less staggered indicating a reduction of the hydraulic retention time within some units of the filter system. The EC variation following the salt addition in the period 9-17 February was at the minimum (grey area). An opposite trend was recorded between the 17 and the 22 February when EC measurements at the outlet of the reactive filter were markedly lower than EC values at the outlet of the sediment filter or the system inlet indicating $\text{Al}(\text{OH})_3$ retention in the reactive filter (green area). Following the last salt addition, a small EC increase was only recorded at the outlet of the reactive filter.

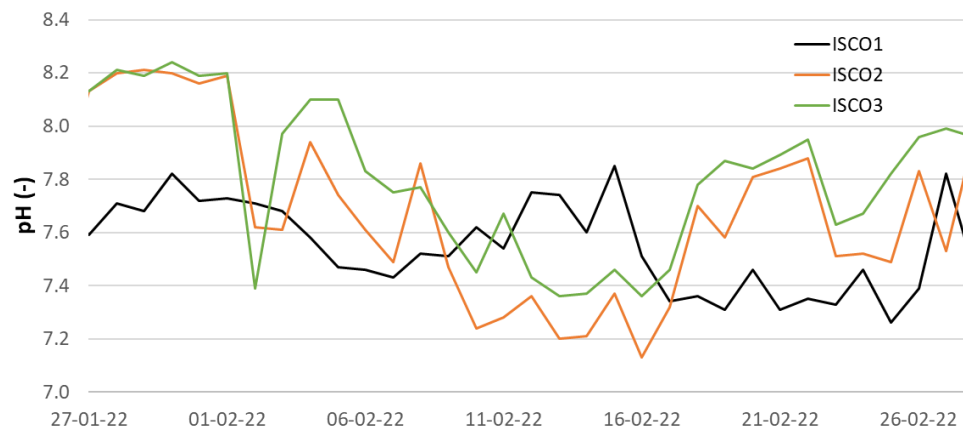


Figure 14. Daily values of pH at the inlet (ISCO1), the outlet of the sediment (ISCO2) and reactive filter (ISCO3) during the flocculation experiment done in February 2022.

Averaged water pH at the inlet (black line) was 7.5 in February 22 and varied between 7.3 and 7.9 with the lowest values recorded at the end of the month (Fig. 14). Typically, pH tends to increase as water passes through the filter system, as observed for the last days of January (Fig. 14). The first salt addition on 1st February produced a pronounced decrease of pH for two days in water passing the filter system. This is consistent with the formation of $\text{Al}(\text{OH})_3$ after the addition of a relatively large amount of Al sulfate (Kennedy and Cook 1982). Thereafter pH quickly increased at the outlets, however, remained lower than in January. Until 16th February, pH in water at the outlets of the DPF and the ICS filter tended to decline again and dropped below inlet pH for a period which can be explained by repeated additions of smaller amounts of Al sulfate. A marked pH increase was recorded at both outlets from 16th February, which coincides with the lower EC values measured at the outlet of the reactive filter (Fig. 13, green area). We currently lack data to fully explain pH dynamics, which may also relate to $\text{Al}(\text{OH})_3$ redistribution in the filter system.

In summary, our initial results indicate that Al sulfate additions to drainage waters do not increase the sedimentation of particulate matter in a simple manner. The formation of colloidal $\text{Al}(\text{OH})_3$ creates a phase that readily sorbs P but that is not easily retained in compact filter systems with hydraulic retention times in the order of one hour. In our system, we could not control the dosage of Al sulfate precisely. This may have resulted in over application and an increased risk of P-enriched colloidal $\text{Al}(\text{OH})_3$ loss from the system.

References

- Borah, D. K., Bera, M., Shaw, S. 2003. Water, sediment, nutrient, and pesticide measurements in an agricultural watershed in Illinois during storm events. *Transactions of the ASAE* 46: 657-674.
- Clemmens, A. J., Bos, M. G., and Replage, J. A. 1984. Portable RBC Flumes for Furrows and Earthen Channels. *Transactions of the ASAE*, 27:1016-1021.
- European Standard EN ISO 6878. 2004. Water quality – Determination of phosphorus – Ammonium molybdate spectrometric method. European Committee for Standardization.
- Kennedy, R.H. and Cook, G.D. 1982. Control of lake phosphorus with aluminum sulfate: dose determination and application techniques. *JAWRA Journal of the American Water Resources Association*, 18, 389-395.
- Mendes, L.R.D.; Tonderski, K.; Iversen, B.V.; Kjaergaard, C. 2018. Phosphorus retention in surface-flow constructed wetlands targeting agricultural drainage water. *Ecol. Eng.*, 120, 94–103
- Mendes, L.R.D 2020. Edge-of-Field Technologies for Phosphorus Retention from Agricultural Drainage Discharge. *Applied sciences* 10:634
- Murphy, J., Riley, J.P. 1962. A modified single solution method for the determination of phosphate in natural waters. *Anal. Chim. Acta* 27:31-36.
- Vandermoere, S., Ralaizafisoliarivony, N.A., Van Ranst, E., De Neve, S. 2018. Reducing phosphorus (P) losses from drained agricultural fields with iron coated sand (- glauconite) filters. *Water Research* 141:329-339.