

Treatment Systems for Agricultural Drainage Water and Farmyard Runoff in Denmark: Case Studies



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Contents

| | | |
|-----|--|----|
| 1 | Introduction | 46 |
| 1.1 | Agriculture Challenges in Northwestern Europe | 46 |
| 1.2 | Tile Drainage Water | 47 |
| 1.3 | Nutrient Losses in Tile Drains | 48 |
| 2 | State of the Art | 49 |
| 2.1 | Constructed Wetlands | 50 |
| 2.2 | Denitrifying Bioreactors | 51 |
| 2.3 | Filter Systems | 52 |
| 3 | Case Studies | 53 |
| 3.1 | Surface-Flow Constructed Wetland: Fillerup | 53 |
| 3.2 | Surface-Flow Constructed Wetland Paired with Woodchip Bioreactor: Ryaa | 57 |
| 3.3 | Filter System: Rodstenseje | 60 |
| 4 | Conclusions | 61 |
| 5 | Recommendations | 62 |
| | References | 62 |

Abstract Nutrient losses from agricultural lands are responsible for degrading water quality and accelerating eutrophication at local and regional scale. Subsurface tile drainage is an agricultural water management measure extensively used in some Northwestern European countries to improve poor internal field drainage and crop production. However, drains constitute artificial and direct conduits responsible for

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shortening the residence time and accelerating nutrient transport through the hydrological system. Thus, they put the aquatic ecosystems at risk since they promote rapid drainage water transport and reduce the possibility for natural nutrient removal.

Edge-of-field technologies have been developed and implemented in the last decades to reduce diffuse pollution caused by nitrogen (N) and phosphorus (P) losses. These technologies are primarily used in critical source areas, where measures at farm-scale appear insufficient to meet the removal targets. They are placed at the edge of agricultural fields and connected to the main drainage pipe. Widely recognized technologies include constructed wetlands, woodchips bioreactors, and filter systems. This chapter presents the results obtained in Denmark from three different and currently investigated edge-of-field technologies treating agricultural drainage water.

Keywords Agricultural drainage water, Constructed wetland, Edge-of-field technologies, Filter system, Nutrient losses, Woodchip bioreactor

1 Introduction

1.1 Agriculture Challenges in Northwestern Europe

Increasing nutrient load to coastal waters has been documented in the last decades on a global scale. More severe effects have been observed in regional seas having certain hydrographic characteristics and surrounded by high densely populated coasts. In Europe, the North and Baltic Seas are receiving increasing nutrient loads due to anthropogenic inputs of nitrogen (N) and phosphorus (P) being responsible for point source and diffuse pollution. The North Sea is a shelf sea covering an area of 570.000 km² and extending from Great Britain to Denmark. The water residence time ranges between 365 and 500 days [1]. The Baltic Sea is one of the largest blackish-water basin in the world with an area of 377.000 km². The semi-enclosed shape is responsible for water residence time of about 30–40 years [2].

Regional seas receive nutrients loads from inland waters such as lakes, rivers, and streams, resulting in adverse effects of structure and functionality for regional and local ecosystems. The overly nutrient enrichment of water bodies is responsible for eutrophication which causes biodiversity reduction by excessive blue-green algae growth.

The protection of water bodies has been on the EU agenda since 1980. Several directives were enacted and implemented over the last four decades to tackle the problem of eutrophication. Among the most important there are the Water Framework Directive 2000/60/EC, the Urban Wastewater Treatment Directive 91/271/EEC, and the Nitrates Directive 91/676/EEC. This resulted in a substantial reduction of point source pollution in the North and Baltic Sea area. However, diffuse

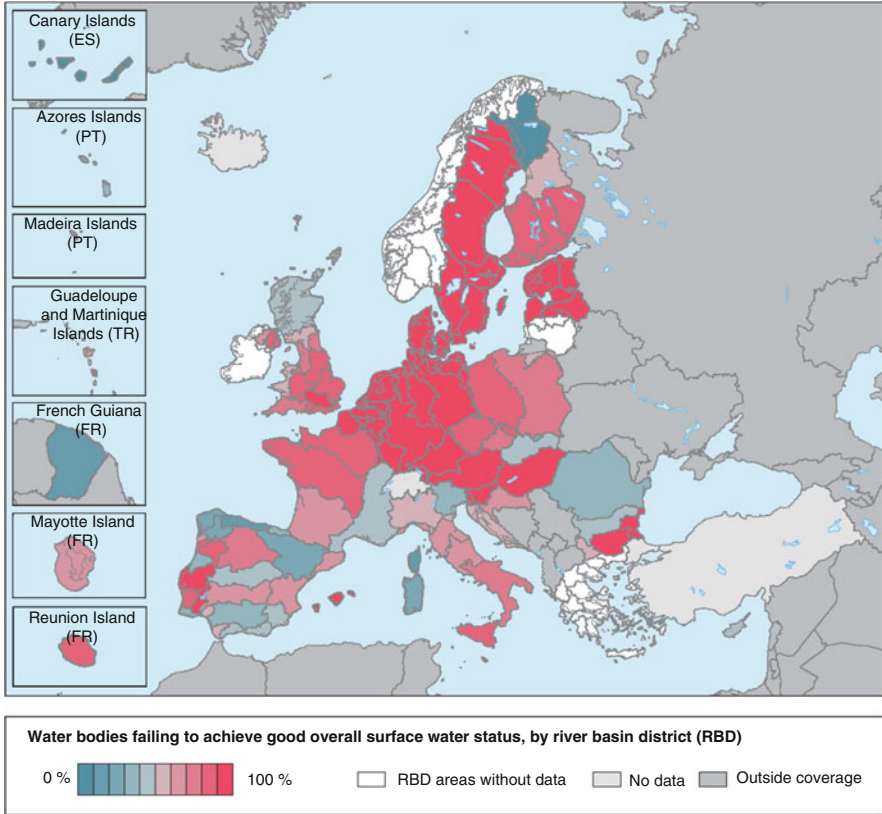


Fig. 1 Percentage of water bodies in Europe’s river basin districts that are not in good ecological status/potential: second river basin management plan [3]

pollution primarily caused by intensive agriculture remains a serious problem for meeting the EU’s minimum target of good status (Fig. 1).

1.2 *Tile Drainage Water*

Subsurface tile drainage is extensively used in order to improve water conditions in soils and thus soil cultivation and crop production. Artificial drainage is primarily used in low-lying areas characterized by (1) a relatively high groundwater table, (2) low permeable soil, and (3) wetlands with organic soils. Under these circumstances, artificial tile drainage effectively lowers the groundwater table, removes excess water, and improves soil porosity and structure. Dry soils warm up faster than wet soils, thus increasing soil temperatures in spring which favor earlier sowing and germination of seeds. Tile drainage also improves soil trafficability and reduces the

risk of soil compaction due to use of heavy machinery and tillage on wet soils. This is particularly important for climatic regions characterized by limited growing seasons. Additional positive effects include more water infiltration and less surface runoff, which decreases soil and nutrient losses. Finally tile drainage reduces peak flows and the water volumes lost from the watershed. The main drawbacks of this practice is the great impact on the water and nutrient balance of a landscape. Tail drains are direct pathways for the transport of nutrients, which are not filtered by contact with soil particles. Additionally, this management practice changes the hydrology of the landscape, reducing its overall water storage capacity.

Depending on soil/crop type and costs, drainage pipes are generally installed at a depth of 0.6–1.2 m and spaced 10–100 apart [4]. Relatively high presence of tile drains in agricultural areas is found in some Northwestern European countries, including the UK (30%), the Netherlands (34%), in the Scandinavian countries (40%) and Finland (91%) [5, 6]. In Denmark, tile drains cover 50% of the agricultural area primarily constituted of glacially derived clayey soils and located in the eastern part of the country [7]. Subsurface geology has proved to be a key factor governing drain flow dynamics, with higher drainage in stream valleys and wetlands and lower drainage in hilly catchment areas. Regarding soil type, heavy clay soils show an initial slow drainage response followed by a rapid increase, while other soil types have a more constant response. A statistically significant correlation was found between the sand percentage in the subsurface geology and the drain flow volume [8]. Moreover, results from a study on the clay till from the Norsminde catchment in Denmark revealed that the geology below 3 m has a large impact on drain flow. Deeper geology must thus be considered when modeling tile-drained agricultural fields to be able of capturing the transient dynamics, unless an impermeable layer is found at a shallow depth [9].

Tile drainage discharge is generally seasonally dependent and increases in response to precipitation/snowmelt while it decreases during drought periods. A study investigating the effect of tile drains on flow in a Danish Weichsel till area showed a piezometer head variation up to nearly 3 m between summer and winter [8]. The decrease in head occurred in spring–summer during 5–5.5 months, while a rapid increase was documented when autumn water surplus started. A relatively high average tile drain discharge contribution (83%) was measured in the Netherlands in response to some major rainfall events characterizing the winter and spring seasons [10]. Thus, a strong seasonal pattern can be identified for tile drainage with greater discharges occurring during the winter and spring seasons and very small or absent discharges across the summer season.

1.3 Nutrient Losses in Tile Drains

Natural flow routes for water and nutrient transport are partially dismissed by subsurface tile drains. These artificial and direct conduits shorten the residence time and cause a rapid nutrient transfer through the hydrologic system.

Tile drains facilitate N transport primarily in soluble forms. Factors facilitating the loss of N are local climate, soil and crop type, length of growing season, options of winter cover crops and catch crops, tillage and soil management practices. Certain factors are field specific and cannot be changed, while others may be adjusted to reduce N losses [11]. A study in the UK investigated the risk of nutrient transfer through tile drains for biosolids amended soils [12]. Significant concentration of nitrate varying between 3 and 34 mg/L was found before amendment. An experimental field study in the Netherlands estimated a 91% contribution from tile drains of the total nitrate load [10].

The major factors influencing P and sediment transport from tile drains are soil characteristics including preferential flow paths, P sorption capacity and redox conditions, drainage depth and spacing of the drains, surface inlets, management practices such as tillage, cropping systems, P application rate and timing, hydrology, and climate [13]. Total P (TP), dissolved P, and particulate P (PP) are the most commonly investigated P forms. In England, annual losses of TP and dissolved P through field tile drains were estimated to be 0.37–2.64 and 0.037–0.74 kg/ha/yr, respectively [14]. In the Netherlands, loads of TP from sandy soils to tile drains accounted for 0.14–0.31 kg/ha/year. [15]. The concentration of the different P forms in tile drains depends on the drain capacity and on the operating frequency, thus, varying with flow and season. Export of P is often characterized by an early pulse caused by preferential transport of surface water through macropores and tiles during precipitation, irrigation, and/or snowmelt events [16]. In Sweden, a number of multi-event, seasonal, and annual studies evaluated the TP load to tile drains to vary between 0.02 and 4.63 kg/ha/year [17, 18]. Heckrath et al. [19] found a larger fraction of dissolved P (66–68%) in UK tile drainage water in comparison to PP (8–35%). However, PP may represent a large fraction of TP under high-flow conditions. A six time greater P loss was measured in England during stormflow in comparison with baseflow from a grassland catchment. Dissolved P accounted for 70% of the total losses during baseflow conditions, while predominantly PP loss occurred during stormflow [20]. Dominant PP concentrations were also reported from studies in Sweden and Finland [21, 22]. The non-growing season (from December to March) was reported to account for 35–40% of annual P losses in Denmark [21]. A series of field experiments investigating subsurface P transport in Denmark reported a minimum and maximum TP concentration of 0.5 and 2 mg/L, respectively, with temporary maximum of 4.8 mg/L [23].

2 State of the Art

Nutrient losses through drainage discharge can be mitigated at farm-scale. Measures include better control in the application of mineral fertilizers and the use of agricultural management practices. Among the most common practices there are cover/

catch crops, controlled trafficking, subsoiling, vegetated buffer strips. Cover/catch crops protect soils from erosion, reduce the amount of runoff, and ensure that nutrients stay in the root zone. Controlled trafficking preserves soil structure and reduces runoff and erosion by restricting compaction to specific traffic lines. Indirect benefits of this measure include reduced waterlogging and enhanced soil biological activity due to improved levels of organic matter. Vegetated buffer strips are strips of land located between agricultural fields and watercourses, aiming at trapping sediments and immobilizing soluble nutrients through plant uptake or microbial degradation. However, measures at farm-scale may be insufficient, especially in critical source areas of nutrient losses.

Edge-of-field technologies are recognized as non-point and economically feasible nutrient mitigation solutions. Despite the higher cost in comparison to farm-scale measures, these technologies provide high nutrient removal, promote biodiversity, and increase flood control. They are generally used for critical source areas and installed at the edge of tile-drained agricultural fields. Constructed wetlands, woodchips bioreactors, and filter systems are among the edge-of-field technologies tested in Denmark to tackle nutrient losses from drainage discharge.

2.1 Constructed Wetlands

Constructed wetlands (CWs) are ecologically engineered systems ameliorating water quality through natural processes involving wetland vegetation, soil, and their associated microbial assemblages [24]. The primary treatments include sedimentation, filtration, oxidation, reduction, adsorption, and precipitation. The growing popularity in the last decades is due to low operational costs and energy consumption.

The classification of CWs generally depends on three main factors: water level in the system, water flow movement, type of macrophytes (emergent, submerged, floating-leaved, free-floating) [25, 26]. Surface flow (SF)/free water surface CWs and subsurface flow (SSF)-CWs are considered the main two types. The former (SF-CW) is similar in appearance to natural marshes and it is constituted of basins naturally or artificially waterproofed. In this system, the water surface is kept above the substrate at a depth which typically ranges between 0.3 and 1 m. Aerobic conditions characterize the near-surface layer of water, while anaerobic conditions are found in the deeper layers. Shallow areas favor emergent plant growth and homogenize the water flow through the formation of small channels simulating a plug-flow reactor behavior. Organic removal occurs via settling and filtration of suspended particles and microbial degradation and settling of colloidal particles. An artificially waterproofed basin entirely filled with inert material of appropriate particle size characterizes the SSF-CWs. According to the water flow movement in the system, SSF-CW can have a horizontal (HSSF-CW) or vertical (VSSF-CW) flow configuration. Anoxic conditions are predominant for both configurations as the water flow level is continuously kept below the inert material. However, continuous

flow from the inlet is generally found in HSSF-CWs, whereas discontinuous flow from the inlet occurs in VSSF-CWs, which resemble more batch reactors [27]. The construction cost of SF-CWs is lower in comparison to SSF-CWs. Additionally, the former offers greater flow control and more diverse wildlife habitat. However, nutrient removal per unit of land is lower. Thus, larger agricultural areas are needed to achieve comparable results. The SSF-CWs perform better under cold climate due to the insulation effect of the unsaturated surface layer [28].

The removal of N in SF-CWs is highly variable and primarily controlled by the inflow load and the ratio between the CW and the drained catchment (W/C) surface area [29]. Results from agricultural wetlands in Finland, Sweden, and the USA demonstrated that for W/C of at least 2%, a N removal of 20% can be obtained, while for W/C > 7% a N removal of 50% can be reached [30]. Removal of N equal to 40% was reported for a W/C ratio of 5% by Tanner et al. [31]. A more recent review [32] on SF-CWs revealed that W/C > 1% does not result in any substantial increase of N removal. However, the W/C ratio does not account for the system volume defined by the water depth and may lead to inaccurate evaluation of the CW effectiveness. Several studies indicated that SF-CWs work as P sinks. Nevertheless, other studies demonstrated that these systems can release P when receiving event-driven drainage discharge peaks. Retention of P occurs through (i) deposition of PP on the basin soil surface, (ii) sorption of dissolved reactive P (DRP) onto the reactive soil sites, (iii) DRP precipitation with Fe and Al oxides and Ca in the water column to the top sediments, and (iv) biological uptake by plants, phytoplankton, and bacteria of bioavailable forms [33]. The contribution by deposition generally constitutes the main retention mechanism. The DRP retention by sorption sites is variable and depends on geochemical characteristics, pH, redox conditions, and P concentrations at the soil–water interface. Calcium is the major P sorbent for mineral alkaline soils, while Fe and Al oxides are the most common sorbents for mineral acidic soils. Retention by plant uptake increases during the growing season and it is normally low.

2.2 *Denitrifying Bioreactors*

Carbon-based SSF-CWs are optimized engineered systems, targeting primarily N losses via subsurface tile drainage. The failure or success of these systems largely depends on the hydraulic retention time (HRT), temperature, and microbiology [34]. The HRT is defined by the system design, the flow rate, and the filter media porosity. Different design methods have been proposed in the last decades to cope with variable flow rates. Denitrification processes are affected by temperature and thus follow seasonal variations. Better performances are generally achieved at higher temperature. The carbon (C) media represents the source supporting denitrification under anaerobic conditions. The choice of the bioreactor carbon media depends upon cost, porosity, C: N ratio, and longevity. The most commonly used media include cobs, corn stalks, wheat and barley straw, pine and almond shells, and woodchips

media [35]. All these media are characterized by dual porosity structure which promotes diffuse mass exchange into immobile domains. However, woodchips media are generally preferred due to the relatively low cost, hydraulic conductivity, longevity, and high C:N ratio.

Bruun et al. [36] investigated the N removal efficiency in different hydraulic designed woodchip-based SSF-CW. The aspect ratio (width-to-height ratio) appeared to be an important parameter defining the cross sectional area and the flow pathway length. Initial results showed that the horizontal flow design had better N removal efficiency compared with vertical upward and downward flow designs. However, a later tracer test and modeling investigation revealed that the vertical downward design had the largest N removal rate (67% under low flow conditions), which was primarily attributed to the longer residence time. In contrary, the vertical upward design showed the most pronounced non-equilibrium and lowest N removal [37]. Field studies reported N reductions varying from 2 to 22 g/m³/d in woodchips bioreactor (WB) having a size of <1 to >1,000 m³ with lower rates often associated with N limitations [34]. A practice-oriented review carried out by Christianson et al. [35] showed that the removal rate for SSF-CWs varied between 12 and 76%. Chun et al. [38] demonstrated the ability of these systems to effectively reduce N pulses at high concentrations.

Studies on P retention by woodchip-based bioreactors treating agricultural drainage water are limited. However, comparable studies can be found in the literature. Robertson et al. [39] investigated WB treating septic effluent and found a <30% reduction of soluble P concentration. On the contrary, no significant change was found by Schipper et al. [40] in P concentrations for dairy shed and treated domestic effluent after treatment in large-scale (80–300 m³) WBs. Gottschall et al. [41] demonstrated that WBs can also retain P (30%). Finally, Choudhury et al. [42] investigated a treatment system for vegetable wash water constituted of a sedimentation tank and woodchip filter installed in a subsurface trench. Results showed that P removal was mostly associated with PP (91%), while DRP in the outflow was only affected to a minor extent.

2.3 Filter Systems

Filter systems or simply removal structures have been widely tested in the last decades to tackle primarily DRP losses to surface waters. Effective filters have a sufficient size and hydraulic conductivity and an effective P sorption capacity. The system is placed in hot spots and hydrologically active areas receiving DRP concentration preferably larger than 0.2 mg/L [43]. The unconsolidated P sorbing material (PSM) is the core of the system. Besides, having a strong sorption capacity, the PSM should be physically stable, operate under natural pH, and must not produce pollution by releasing toxic contaminants. A natural, synthetic, or industrial by-product PSM characterized by a strong P sorption capacity is generally used. These materials can be grouped in iron (Fe)/aluminum (Al) based and calcium

(Ca)/magnesium (Mg) based PSM [44]. The DRP retention occurs by chemisorption or ligand exchange onto the sorption sites surface for the Fe/Al based PMs, while it occurs through precipitation for Ca/Mg based PSM [45].

Results from laboratory experiments revealed that Fe-oxide based PSM presents higher sorption capacity, reactivity, and stability in comparison to Ca-based PSM [46]. High P removal efficiency (>95%) in solution up to 100 mg/L was also found by Allred and Racharaks [47] who tested Fe-based PSM including zero-valent Fe, porous Fe composite, sulfur modified Fe and Fe oxide/hydroxide. Further experiments confirmed good P retention and hydraulic conductivity in treating agricultural drainage waters with porous Fe composite [48]. Studies investigating P retention from agricultural drainage discharge at field scale are limited. Performances are expected to be lower due to hydrological factors, which are not taken into account at laboratory scale. The P retention of two on-site filters located in southwestern Finland was monitored in a long-term field-scale experiment [49]. The sand filters were enhanced with a layer of Fosfilt-s, a side product of titanium dioxide production. Results showed that the system was able to remove 37% of TP and 45% of DRP. Small scale field filters containing iron coated sand were successfully installed and tested by Vandermoere et al. [50]. During the 10 week field trials, the authors reported a P removal efficiency between 70 and 90%.

3 Case Studies

In Denmark, a number of edge-of-field technologies have been developed and investigated following the national regulation in order to reduce nutrient losses from agricultural land to surface waters. Despite the positive results, a continuous effort is put in place in order to find more suitable and cost-effective solutions contributing to long-term and stable nutrient removal from drainage waters. Below are presented three case studies and their corresponding results obtained during long-term monitoring campaigns.

3.1 *Surface-Flow Constructed Wetland: Fillerup*

The SF-CW is located in Fillerup, within the Norsminde Fjord catchment – Denmark (55°57'34"N 10°05'30"E) [50]. The system was constructed in 2010 and placed at the outlet of the main discharge pipe receiving tile drainage water from a 45 ha sandy clay loam morainic agricultural field. The system occupied a total surface area of 0.28 ha (0.6% of the drainage area). Three main components could be identified: (1) a sedimentation basin, (2) deep zones, and (3) shallow zones (Fig. 2). The 1-m deep sedimentation basin allowed removal of coarse particles and facilitated even flow across the entire system. Deep zones (1-m deep) increased the HRT and provided flow redistribution. Shallow zones (0.3-m deep) promoted vegetation

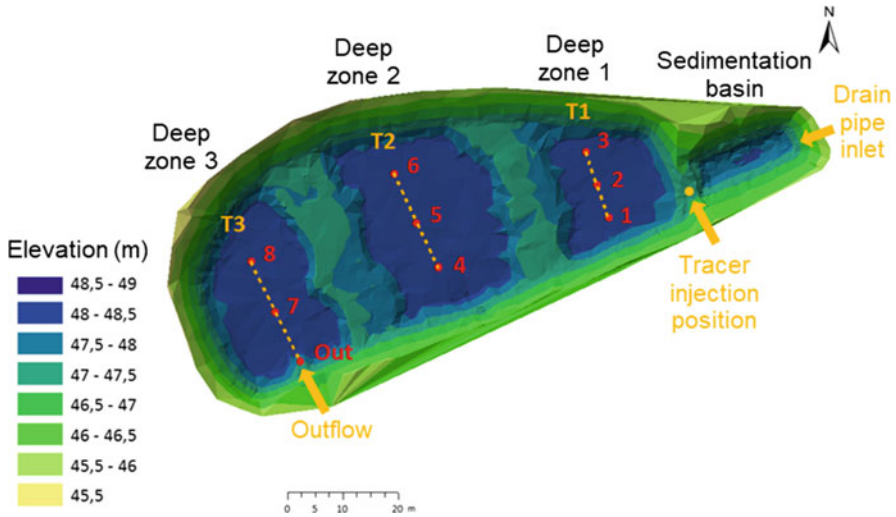


Fig. 2 Schematic of the surface-flow constructed wetland (SF-CW) in Fillerup. The main system components are also indicated (modified from [51])

growth and limited flow dispersion. *Typha* sp., *Phragmites australis*, *Epilobium* sp. were the dominant emerging plants, while Characeae was the dominant submerged species.

The monitoring program ran from January 2013 to May 2017 under the iDRAIN project (<https://idraen.dk/>). Values of tile drainage discharge (Q) were measured by an electromagnetic flow meter and continuously logged by a pulse data logger having a 10 s resolution. Daily water samples were collected automatically at the inlet and outlet of the system for the determination of the nutrient concentration. Chemical analysis of the water samples was carried out either individually or as a composite sample according to the Q hydrograph. In particular, individual analysis was carried out during periods characterized by highly variable drainage discharge, while composite samples were analyzed during periods with steady and low drainage discharge. Analysis for TN was carried out on unfiltered samples, which were initially autoclaved for 30 min at 121 °C in an alkaline peroxydisulfate solution. Subsequently, the digested samples were acidified and moved to an autoanalyzer for measuring their adsorbance. Analysis for TP was carried out using a spectrophotometer at 890 nm with ascorbic acid (5%) after 30 min acid persulfate (5%) digestion in autoclave at 121 °C [52, 53].

Values of Q across two entire hydrological years (from August to end of July) were highly variable with distinct peaks caused by heavy precipitation events (Fig. 3a). A number of two periods, corresponding to the autumn–winter season characterized by highly variable Q values and peaks up to 24 L/s, can be identified. On the contrary, lower Q values were generally recorded during the spring–summer seasons. Daily TN inlet concentrations varied between 0.0 and 18.6 mg/L, while

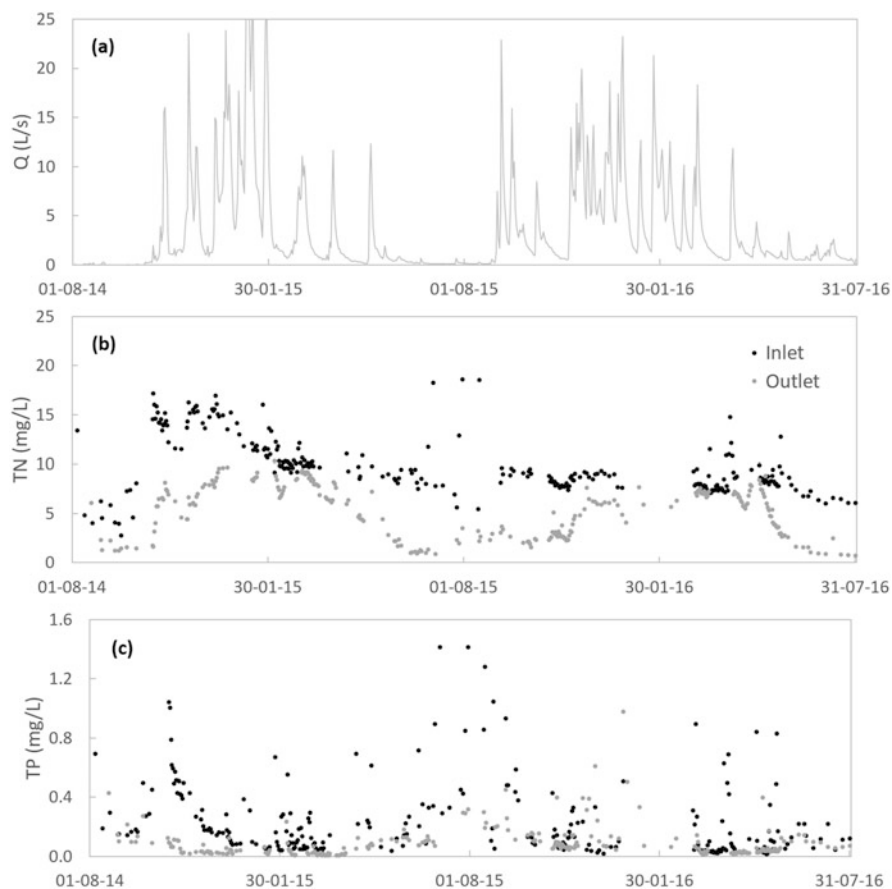


Fig. 3 Daily (a) drainage discharge (Q), (b) inlet and outlet concentration of total nitrogen (TN), and (c) total phosphorus (TP) for the SF CW in Fillerup

outlet concentrations ranged from 0.7 to 15.7 mg/L (Fig. 3b). Major differences between inlet and outlet TN concentrations can be identified especially across spring–summer seasons. The yearly TN removal efficiency of the system varied between 37 and 42% (Table 1). Discharge-weighted TP inlet concentrations varied between 0.01 and 1.42 mg/L , while outlet concentrations ranged from 0.0 to 1.0 mg/L (Fig. 3c). In contrast to TN, differences between inlet and outlet TP concentrations do not follow a clear seasonal trend. The yearly TP removal efficiency was higher in comparison with TN and ranged between 43 and 73% (Table 1).

Table 1 Yearly drainage discharge (Q), total nitrogen (TN), and total phosphorus (TP) at the inlet, outlet, and removed (%) in Fillerup

| Case study | Hydrological (year) | Q (m ³) | TNin (kg/ha) | TNout (kg/ha) | TNremoval (%) | TPin (kg/ha) | TPout (kg/ha) | TPremoval (%) |
|-----------------------|---------------------|---------------------|--------------|---------------|---------------|--------------|---------------|---------------|
| Fillerup ^a | 2014/2015 | 103,291 | 35 | 22 | 37 | 0.565 | 0.151 | 73 |
| | 2015/2016 | 120,197 | 27 | 16 | 42 | 0.875 | 0.497 | 43 |

^aKjærgaard et al. in preparation

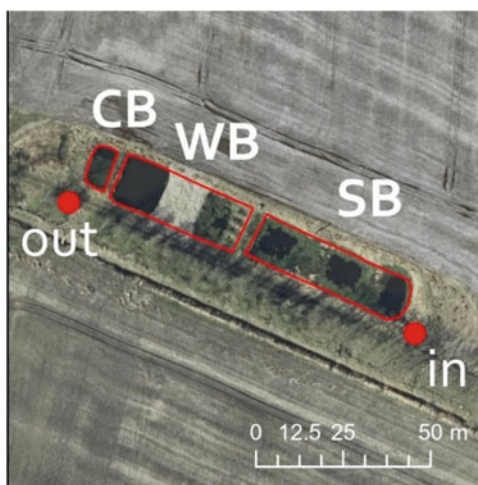
3.2 Surface-Flow Constructed Wetland Paired with Woodchip Bioreactor: Ryaa

The SF-CW + WB was located in Ryaa, within the Limfjorden catchment – Denmark ($57^{\circ}13'05.8''\text{N}$, $9^{\circ}44'19.7''\text{E}$) [54]. This design concept pairs a SF-CW and a WB in one system to ensure significant reduction of both P and N. Limited retention of P was in fact documented in WBs, while a similar effect for N was reported in CWs [41, 55]. The additional advantage is the smaller agricultural area needed for construction (0.1% to 0.2% of the catchment area) compared with standalone and currently used SF-CWs (~1%) [54]. The SF-CW + WB was established in 2011 and received through a pumping system tile drainage water from 85 ha of freshwater peat agricultural fields. The system occupied a total surface area of 0.19 ha (0.1% of the drainage area). The main system components were (i) a sedimentation basin, (ii) the WB, and (iii) a clarification basin, all being 1-m deep (Fig. 4). The sedimentation basin promoted sedimentation of suspended particles and PP. Denitrification was activated in the WB, which also allowed P immobilization and deposition. The clarification basin finalized the process by oxygenating the anoxic effluent leaving the system. *Phragmites australis* was the dominant emerging plants.

The monitoring program ran from August 2016 to June 2019 under the Future Cropping project (<https://futurecropping.dk/>). The procedure for Q sampling and TN and TP analysis is described in details for the previous field case.

During three entire hydrological years (from August to end of July), a number of three high Q periods can be identified with peak flows >28 L/s (Fig. 5a). These periods correspond to the autumn–winter season. On the contrary, lower Q values were generally recorded during the spring–summer seasons. However, this trend in 2017/18 was less pronounced. Daily TN inlet concentrations varied between 1.3 and 17.5 mg/L, while outlet concentrations ranged from 0.6 to 14.1 mg/L (Fig. 5b). Inlet

Fig. 4 Orthophoto of the surface-flow constructed wetland (SF-CW) paired with the woodchips bioreactor (WB) in Ryaa. The main system components are also indicated as sedimentation basin (SB), WB and clarification basin (CB) together with inlet (in) and outlet (out) (modified from [54])



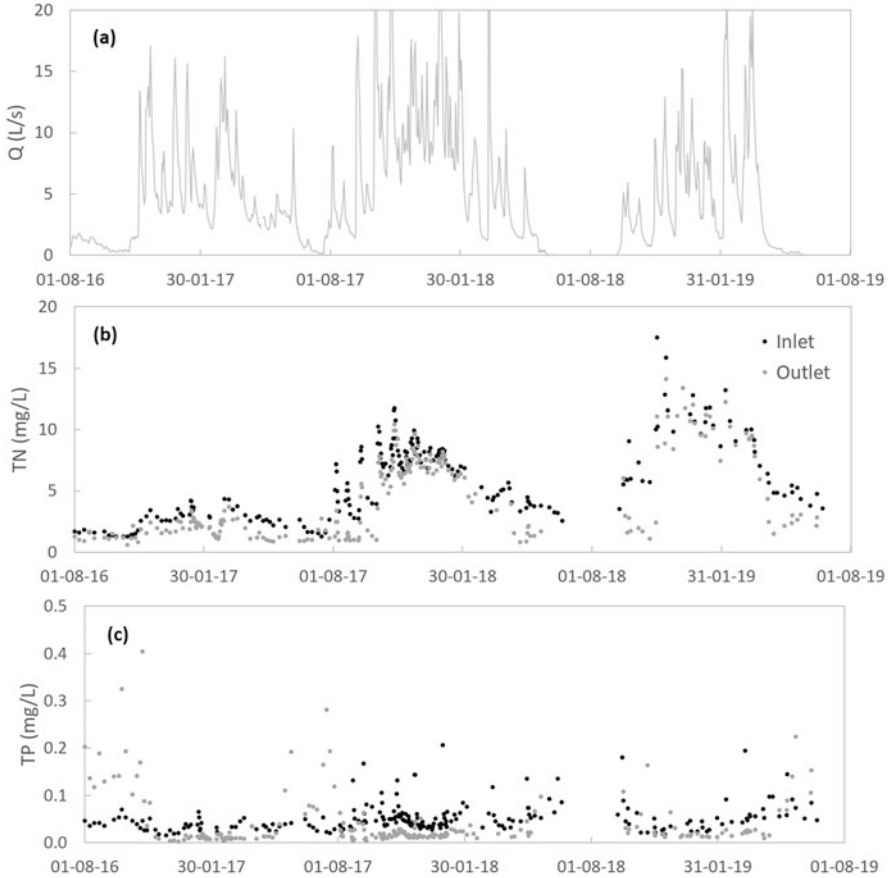


Fig. 5 Daily (a) drainage discharge (Q), (b) inlet and outlet concentration of total nitrogen (TN), and (c) total phosphorus (TP) for the SF CW + WB in Ryaa

and outlet TN concentration increased for increasing Q and their difference remained stable across the entire monitored period. The yearly TN removal efficiency of the system varied between 8 and 32% (Table 2). Discharge-weighted TP inlet concentrations varied between 0.0 and 0.4 mg/L, while outlet concentrations ranged from 0.0 to 0.2 mg/L (Fig. 5c). Differences between inlet and outlet TP concentrations did not follow a clear seasonal trend. Data scattering was observed during the first hydrological year with outlet TP concentrations bigger than inlet values. These variations must be due to the release of sediment-bound P, which is caused by biogeochemical factors affecting P cycling into the system [33]. The yearly TP removal efficiency varied markedly and ranged from 0 to 67% (Table 2).

Table 2 Yearly discharge (Q), total nitrogen (TN), and total phosphorus (TP) at the inlet, outlet, and removed (%) in Ryaa

| Case study | Hydrological (year) | Q (m ³) | TN _{in} (kg/ha) | TN _{out} (kg/ha) | TN _{removal} (%) | TP _{in} (kg/ha) | TP _{out} (kg/ha) | TP _{removal} (%) |
|----------------------|---------------------|---------------------|--------------------------|---------------------------|---------------------------|--------------------------|---------------------------|---------------------------|
| Ryaa GT ^a | 2016/17 | 124,670 | 4 | 3 | 32 | 0.055 | 0.055 | 0 |
| | 2017/18 | 194,155 | 16 | 13 | 18 | 0.138 | 0.045 | 67 |
| | 2018/19 | 108,647 | 13 | 12 | 8 | 0.064 | 0.036 | 44 |

^aPugliese et al. [54]

3.3 Filter System: Rodstenseje

The filter system was located in Rodstenseje, within the Norsminde Fjord catchment – Denmark ($55^{\circ}57'22.0''\text{N}$, $10^{\circ}09'34.3''\text{E}$). The system was constructed in 2011 to treat a fraction of the tile drainage water collected from a 100 ha agricultural fields with loamy, morainic soils [56]. Drainage water entered first a sedimentation pond and was subsequently abstracted by a submerged siphon pipe. The main components of the system filter were the following: (1) a distribution well, (2) a well for discharge measurements, (3) a sediment well, (4) a monitoring well, (5) a reactive P-filter, and (6) a monitoring well at the outlet of the system (Fig. 6). Cylindrical concrete wells housed the different components which were connected by 200 mm diameter PVC pipes. A hydraulic head of 0.45 ± 0.05 m existed between the distribution well and the outlet led water through the system over a distance of 7.2 m.

The focus of the filter system study was to evaluate the capability of the reactive P-filter to remove soluble P. The reactive P-filter was constituted of a purpose-build metallic cage defined by two concentric cylinders having an inner and outer diameter of 0.8 and 1.5 m, respectively, and a height of 1.8 m. This specific design was chosen to provide a high cross sectional area, thereby ensuring longer contact between flowing water and the sorption sites in the filter matrix. Commercially available crushed seashells (2–5 mm) were chosen as a filter material based on previous lab results evaluating its physicochemical properties [57]. Seashells were retained into the filter cage by a polypropylene net having a mesh opening of 1.5 mm.

The monitoring program ran from May 2015 to May 2017 under the SupremeTech project (<https://supremetech.dk/>). Values of Q entering the system were automatically measured at the second well through an electromagnetic flow meter and continuously logged every 15 min. Automatic water sample was collected at the distribution well (ISCO_{in}) and in the monitoring well after the reactive P-filter (ISCO_{out}). The majority of the water samples were analyzed individually and only 2% were analyzed as composite samples. Analysis for molybdate reactive P (MRP)

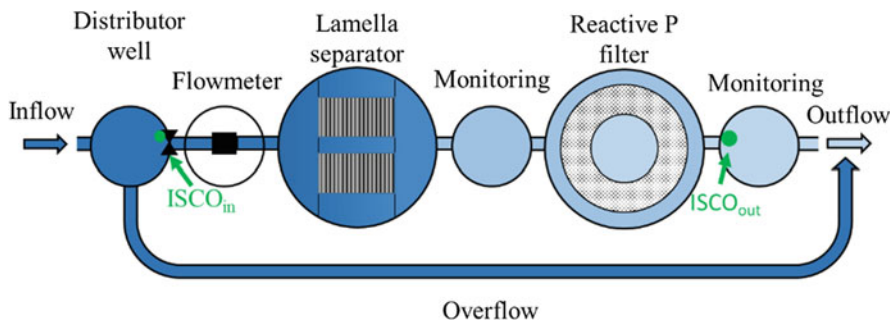


Fig. 6 Schematic of the filter system in Rodstenseje. The main system components are also indicated [56]

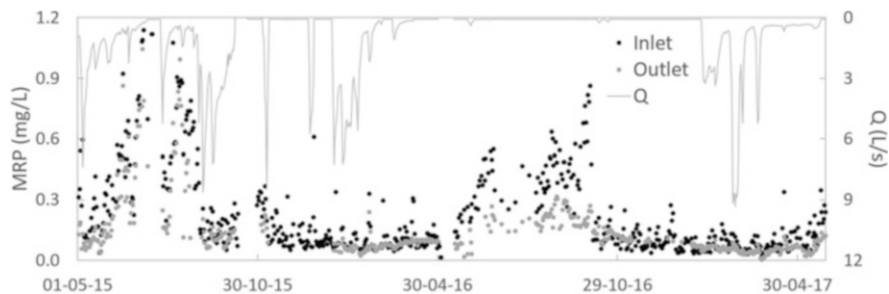


Fig. 7 Molybdate reactive phosphorus (MRP) for the filter system in Rodstenseje. Daily values of tile drainage discharge (Q) are given on the secondary axis (modified from [56])

was performed according to the method of Murphy and Riley [52] after filtration through a 0.45 μm cellulose-acetate membrane.

Higher Q characterized the first monitoring year with peaks of 8.9 L/s (May 2015–May 2016) (Fig. 7). Remarkable increase of Q during the second monitoring year was only registered between January and March 2017. Inlet values of MRP varied between 0.0 and 6.3 mg/L, while outlet values range from 0.0 to 1.3 mg/L. The highest values were generally measured during the spring–summer season with more pronounced differences between inlet and outlet during the second monitoring year. More scattering was recorded during the first spring–summer season.

4 Conclusions

This chapter presents the results from long-term monitoring programs of three edge-of-field technologies tackling nutrient losses from agricultural fields. The drainage water (Q) entering these systems has relatively low nutrient concentration and highly variable loads. Values of N and P removal performance varied between hydrological years and systems. Results showed that the system performance depends on the hydrological, hydrogeochemical, and biogeochemical factors.

The removal of N was generally lower than that of P for both Fillerup and Ryaa. Results from the monitoring campaign showed a recurrent trend between HRT, loading rate, seasonality, and removal rates. Higher removal efficiencies were in fact obtained for higher HRT and lower loading rates which generally characterize the spring–summer season. Removal of P was generally higher than removal of N. This is most likely due to the high fraction of PP from the total P, which is removed through sedimentation representing the primary and generally long-term retention mechanisms in these systems. Results from the statistical analysis demonstrated for Ryaa that the hydrological parameters (HRT and loading rates) have a large effect on P removal efficiency.

Results from the filter system in Rodstenseje showed that during a 2-year monitoring campaign a removal efficiency of dissolved reactive P equal to 62%

was reached. This result was obtained by using a numerical model for simulating the retention capacity of the filter material. In particular, the model showed high frequency of Q values being responsible for the highest retention capacity. Under optimal conditions Q peaks must be reduced.

5 Recommendations

Overall, the long-term monitoring carried out at the Department of Agroecology, Aarhus University (DK) showed a large nutrient removal variability across different edge-of-field technologies and hydrological years. The catchment characteristics and the system design regulate the hydrological, hydrogeochemical, and biogeochemical factors, which define the nutrient performance and its variability. Removal insufficiencies appeared primarily during autumn–winter seasons, which are generally characterized by the highest loading rate, shortest HRT and lowest temperature. Thus, there is a need to develop more effective technologies exhibiting a more stable removal throughout the entire hydrological year. A potential solution may be achieved by adding additional storage basins or reactive P-filters in parallel (Rodstenseje) activated during the autumn–winter seasons. However, a detailed cost-efficiency assessment must be carried out beforehand.

Numerical models represent valuable tools for cost-efficiency assessment of already operating and potential optimal edge-of-field technologies. The numerical simulation provides in fact insights of the systems and highlights their strengths and weaknesses taking into account different hydrological, hydrogeochemical, and biogeochemical factors. Thus, numerical models must be used where possible to find the optimal technology allowing to meet the nutrient removal targets.

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