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FILTER SYSTEMS FOR A SUSTAINABLE AGRICULTURE

FIELD CASE DESCRIPTION

Nitrate removal from drainage water by a Moving Bed Biofilm Reactor (MBBR)

KU LEUVEN

Location

Country: Belgium
City: Onze-Lieve-Vrouw-Waver
Coordinates: 51.052848, 4.546281

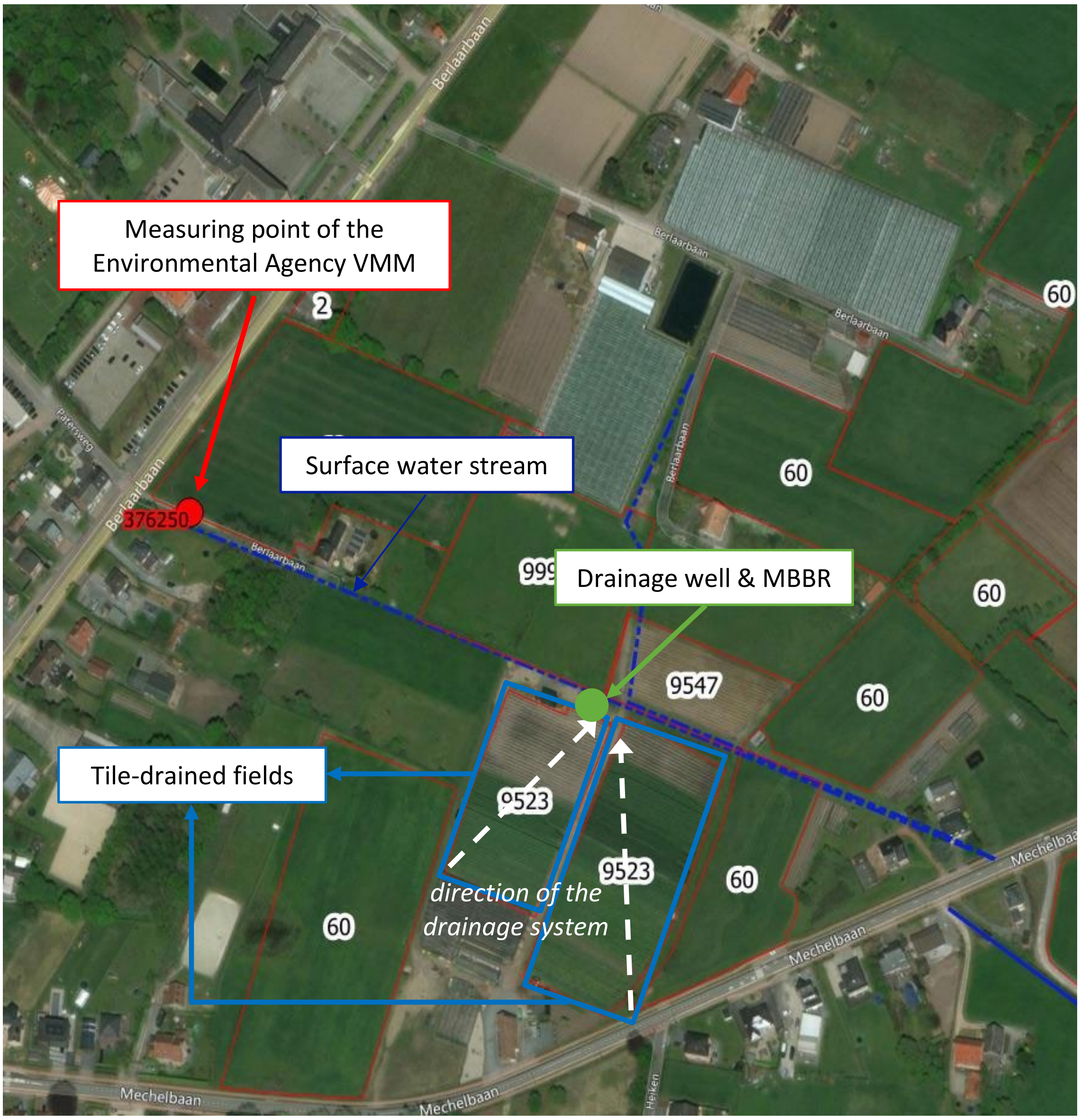

Figure 1 Location of the MBBR and catchment area

Problem description

In Flanders (the North part of Belgium), the most open agricultural fields are provided with a drainage system to collect the runoff water and discharge it into the local surface water streams. So, in a period with a lot of precipitation, the soil gets saturated and then the drainage water (excess runoff water) starts flowing. The so-called drainage season starts typical in October and can last until April or May. The drainage water is confronted with a high nitrate content because unused fertilizers leach out into the drainage water. In this field case, the drainage water of two runoff areas (1.43 ha, see Figure 1) was characterized with an average concentration of 30.7 mg NO_3-N/L, exceeding the EU standard of the receiving surface water, i.e., 10.3 mgNO_3-N/L. Specific issues determine the choice of technology used to treat the drainage water:

- Low water temperatures (between 5 – 15 °C)
- Variable flow rates and nitrate concentrations
- Simple and low budget system
- Limited footprint

Therefore, a Moving bed biofilm reactor (MBBR), using plastic carriers (AnoxK™5) on which a denitrifying biofilm will grow, has been selected in this study.

Filter description

The MBBR water treatment technology, which was developed in Norway in the late 1980s and early 1990s, is a biofilm process in which biomass grows on plastic carrier material (carriers) that is kept in motion in the reactor chamber using mechanical mixers and/or aerators. As a result, the MBBR does not experience clogging because of the constant movement and collision of the carriers. Unlike conventional activated sludge systems, the MBBR does not require a clarifier for the separation of the biomass from the treated water, and thus the performance of the system is independent of the settling efficiency. In addition, it is a cost-effective and highly efficient treatment system that requires little maintenance (McQuarrie and Boltz, 2011). An MBBR can be operated at very high loads and the process is insensitive to load variations (Al-Rekabi, 2015).

Based on historical measurement data of the catchment area (flow rate and temperature), a plant was designed to handle 1000 L/h of drainage water with a nitrate concentration of maximum 45 mg NO_3-N/L (=200 mg NO_3/L). Considering a minimum water temperature in the winter months of 6 °C, a minimum MBBR volume of 12.4 m^3 should be provided, filled with 3.6 m^3 of carrier material (29% fill rate). Advised by Veolia Water Technologies Belgium, AnoxKaldnes™ Carriers type K5 (AnoxK™5) were chosen, and this mainly because this type of carriers possesses a very high specific surface area of 800 m^2/m^3 for biofilm attachment. The greater the specific surface area of the biocarriers, the greater the degradation capacity per m^3 of MBBR volume. To be able to meet the most extreme conditions, an underground concrete anoxic MBBR system of 15 m^3 (Figure 3 (a-c)) filled with 4.4 m^3 of AnoxK™5 (Figure 3 (d)) was installed on the pilot site in Onze-Lieve-Vrouw-Waver. In this way, a little reserve was built in to handle any capacity expansion in the future. Regarding the design calculations, reference is made to Appendix 1 of this document.

The drainage water is pumped from the existing pump well of the grower into the denitrifying MBBR pilot plant at an average flow rate of 1.66 m^3/h. The supply of drainage water is controlled by a low-high level controller which is mounted in the MBBR. In this way, the MBBR is fed via a semi-batch method where the water level in the MBBR is kept between the high- and low-level sensor. A volume of approximately 1 m^3 (= volume between the high and low level in the MBBR) is added after each pumping cycle to the reactor. This means that at maximum capacity, approximately once an hour the MBBR is replenished with fresh drainage water. During the filling of the MBBR to the high level, no carbon source is added (Phase 1 in Figure 2). After the MBBR is refilled with drainage water from the pump well, the influent pump will turn off, and the effluent pump will be activated (Phase 2 in Figure 2). This pump will transfer water from the bottom of the reactor to the ditch, whereby some of the water is recirculated in the MBBR itself to create turbulence in the reactor to keep the AnoxK™5 moving (=recycle indicated on Figure 2 phase 2 and 3). The effluent flow rate is adjustable via an effluent valve. In this way the hydraulic retention time, which must be at least 12 hours, can be maintained. Simultaneously with the discharge of the treated drainage water, a carbon source is also

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dosed to the MBBR (Phase 3 in Figure 2). The carbon source, which is necessary to perform the denitrification reaction, is added flow-proportionally to the MBBR. When no water is pumped to the MBBR, if for example the level in the sump is too low, no carbon source will be dosed either. The flow rate of the carbon source dosing pump is manually adjusted to maintain a proper COD/NO_3-N ratio. It is ensured that the carbon source is properly mixed with the water content in the MBBR by injecting the carbon source into the recycle stream. To summarize, the MBBR repeatedly goes through a filling phase (phase 1 and 4 without carbon dosing) and an emptying phase (phase 2 and 3 which involves the dosing of the carbon source). The flow rate of both drainage water and carbon source are manually aligned with each other to apply a proper COD/NO_3-N ratio.

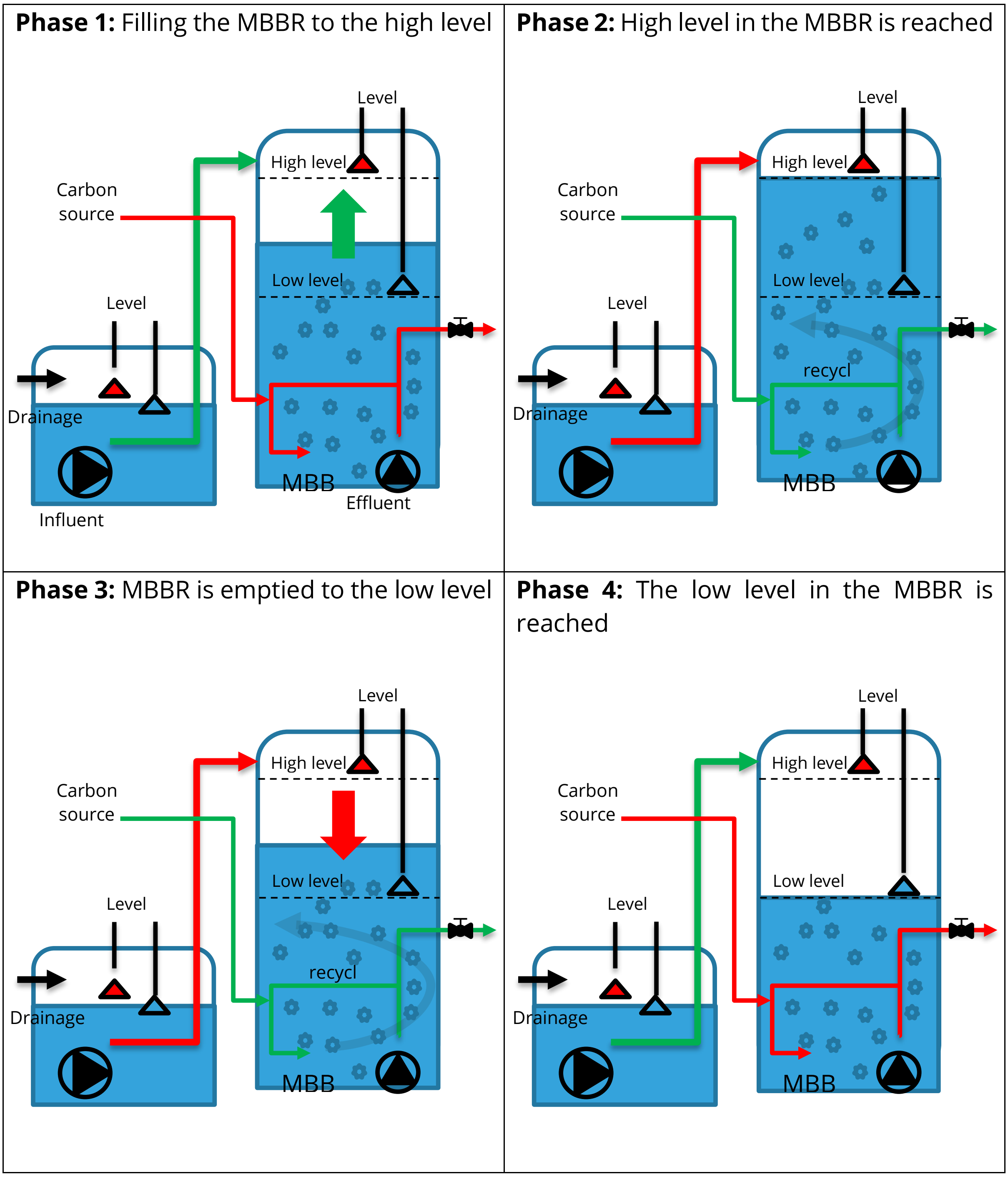

Figure 2 Control scheme for dosing the carbon source. Phase 1: Filling the MBBR; Phase 2: High level in the MBBR is reached; Phase 3: The MBBR is emptied (no water is pumped from the drainage well to the MBBR); Phase 4: The low level in the MBBR is reached and the influent pump in the drainage well is activated. The red arrows indicate that no water is flowing in the respective pipes, the green arrows indicate that water is being supplied or drained.

Carbon source

Four different carbon sources were considered: molasses, Bio-aid, Carbo ST and Carbo BWB-60. Molasses is a viscous by-product of the sugar refining industry; Bio-Aid is based on ethylene glycol butyl ether, naphthalene and vinyl acetate; Carbo ST HQ consists mainly of glycerol (>82%); and Carbo BWB-60 consists of methanol (<2.5%) and glycerol. After a prelaminar techno-economic evaluation Carbo ST was retained as the most suitable carbon source for the denitrification of drainage water. Main properties of Carbo ST: 1.177 €/kg with a density of 1.25 kg/L and a COD concentration of 1.3 kg COD/L. Carbo ST can be ordered at the following Dutch supplier: Melspring.

Based on the flow rate to be processed and the average NO_3 concentration in the drainage water, it can easily be calculated that the MBBR in Onze-Lieve-Vrouw-Waver needs to process 1.08 kg NO_3-N per day. If we assume an optimal COD/NO_3-N ratio of 8, this amounts to a demand of 8.68 kg COD/day. Considering the COD content of the carbon source Carbo ST of 1,3 kg COD/L, 6.68 L Carbo ST must be dosed per day, which corresponds with a cost of 9,82 € per day.

Mixing of the MBBR

Some turbulence must be provided in the MBBR to keep the carrier material in motion, to achieve homogeneity, to increase transport of the substrate (in this particular case, nitrate) to the biofilm, and to maintain an appropriate biofilm thickness. Extremely high turbulence is not recommended, as this can cause the biofilm to detach from the carriers. The increased friction and collisions between the biocarriers may then give rise to a greatly reduced biofilm thickness resulting in loss of degradation efficiency (Rusten et al., 2006). Typically, horizontal shaft-mounted (banana) mixers with two or three blades are used for heterotrophic denitrification in MBBR plants on an industrial scale. The maximum agitation speed of the mixers is 120 rpm (rotations per minute) to minimize damage to the bio-carriers (McQuarrie and Boltz, 2011). Since this type of slow speed mixer for application in small MBBR systems (smaller than 15 m^3) is not commercially available, low-budget alternatives that can be deployed were explored.

The most straightforward method of agitating the AnoxK™5 as an alternative to a stationary mixer is to periodically aerate the reactor volume. Because heterotrophic denitrification is a biological process that proceeds in the absence of oxygen, the aeration system must be operated so that an increase in dissolved oxygen concentration is limited. Therefore, it is recommended to operate with a coarse-bubble aeration characterized by poor oxygen transfer to the liquid and by aerating only periodically for very short time intervals. The major advantage of mixing by aeration is that it is very energy efficient (the mixing power can be limited to only 8 W/m^3) and the investment cost is very limited

(<500€*). In aerated MBBR systems, substrates (NO_3, NH_4 an COD) are transported into the biofilm via diffusion mechanisms, which enable the formation of aerobic, anoxic and anaerobic layers (Iannacone et al., 2019). If the oxygen concentration in the MBBR is not too high (<0.4 mg O_2/L) and the COD:N ratio is kept sufficiently high (>5 mg COD/mg N) then coarse-bubble aeration can be considered a good alternative to traditional stationary mixers. If these conditions are not met, this will result in a lower nitrogen removal efficiency and possibly also significant accumulation of nitrous oxide (N_2O), a powerful greenhouse gas (Iannacone et al., 2019).

Influence of phosphate concentration on denitrification efficiency

When the P:N ratio (PO_4-P-to-NO_X-N concentration ratio) is less than 0.000875 it is said to be phosphorus deficient, and the growth of the denitrifying biomass will be strongly inhibited with detrimental consequences for the denitrification efficiency (Boltz et al., 2012). Specifically for drainage water this means that the concentration of phosphorus in the drainage water should be around 0.2 à 0.03 mg PO_4-P/L. In many cases this means that extra phosphorus needs to be dosed for the growth of the denitrifying biomass. This is best done by dosing phosphoric acid (m%=75%, density H_3PO_4 = 1.36 kg/L) along with the carbon source. For the field case in Onze-Lieve-Vrouw-Waver, this amounts to 3 mL phosphoric acid per day or an addition of 44 mL to 100 liters of Carbo ST carbon source.

* ENVICON membrane plate aerator (6-8 Nm^3/h, 50€ excl. VAT) connected to an AquaForte type AP150 (11.4 m^3/h, 120 W, 250€ excl. VAT) aeration pump. The aeration pump is activated and deactivated with a timer (150€ excl. VAT) (e.g. 1 minute activated per hour or per 2 hours).

Photos filter

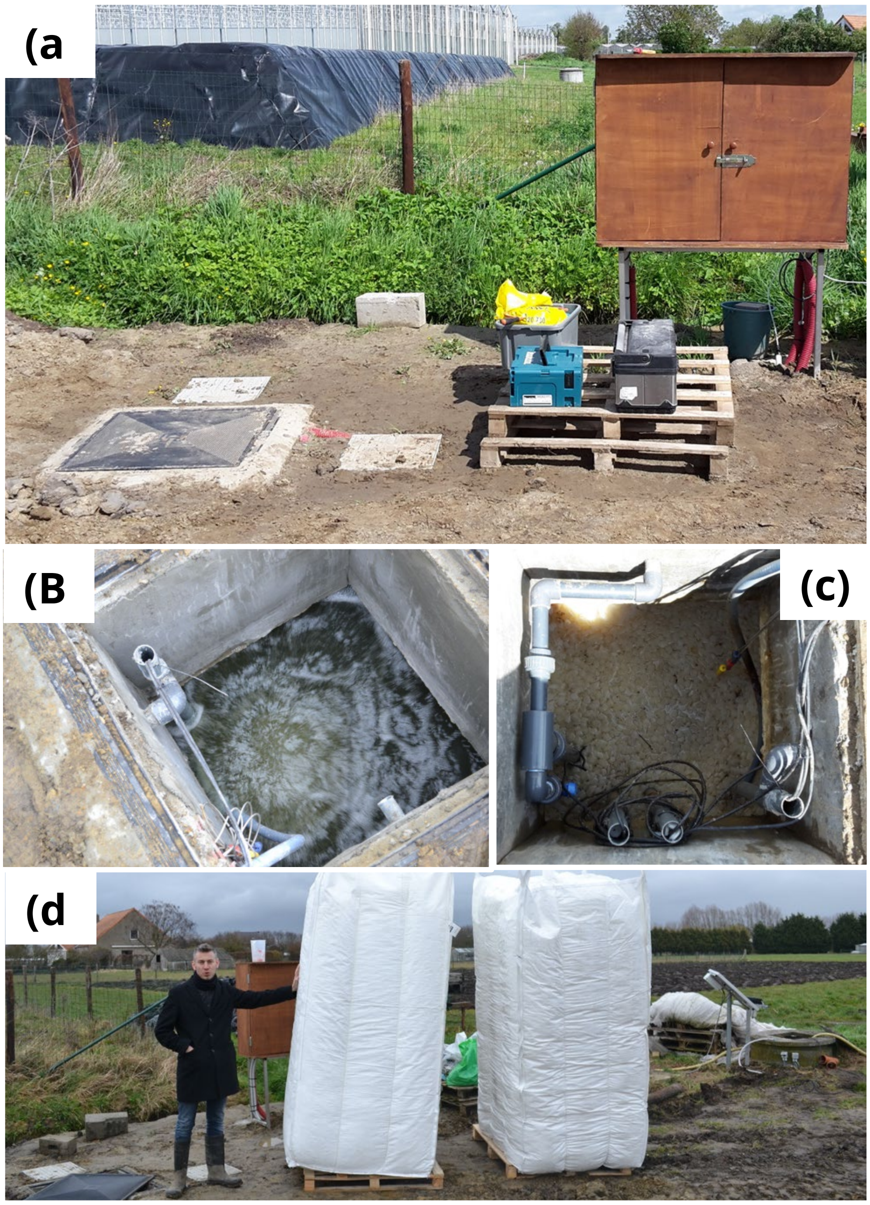

Figure 3: Photos of the Moving bed biofilm reactor (a: manhole of the underground concrete reactor, together with the electrical cabinet; b & c: Inside view of the MBBR during mixing by aeration and without mixing, respectively; d: Big bags with AnoxK^{TM}5 carriers, each 2.25 m^3)

Results (Drainage season 2020-2021)

Characteristics of the Drainage water

The drainage season 2020-2021 started at the end of October 2020 (22 October 2020) and stopped at the end of May 2021 (31 May 2021). During this period a total amount of 2910 m^3 drainage water was treated by the MBBR installation. An overview of the most important characteristics (flow rate, temperature and pH) and the composition of the drainage water is given in Table 1. The flow rate differs between 1.2 m^3/day and 41.2 m^3/day, whereas the low flow rates are measured in the beginning and at the end of the drainage season and the high flow rates during December and March. The pH remains stable around a value of 6.5 and the temperature fluctuates between 6.0°C and 14.9°C. Figure 4 shows the evolution of the temperature of the drainage water throughout the whole drainage season. Also the evolution of the average day temperature is represented. The highest temperatures are observed in the beginning and at the end of the drainage season, reaching values of almost 15°C. The coldest temperature, i.e., 6°C, is measured on 19 February 2021 (Day 120), situated in the middle of the drainage season. The average day temperatures are in most cases lower than the temperature of the drainage water. Certainly, when the average day temperature drops below 0°C, only a slight decrease of the temperature of the drainage water is observed. The underground concept prevents that the temperature of the drainage water will decrease below 5°C, which is necessary to guarantee activity of the denitrifying biofilm in the MBBR.
Only in October (at the start of the drainage season), low nitrate concentrations were measured, i.e., 1.4 $mgNO_3$-N/L on 22 October 2020 and 2.8 $mgNO_3$-N/L on 30 October 2020. In the beginning of November, the nitrate concentration suddenly increases towards values above 30 $mgNO_3$-N/L. From this point on until the end of March, the nitrate content was stable, resulting in an average concentration of 34.6 $mgNO_3$-N/L. In April and May, this average concentration decreases with 30.6% towards 24 $mgNO_3$-N/L.

Table 1: Characteristics of the drainage water during 2020-2021

Parameter	Average value	Minimum value	Maximum value
Flow rate (m^3/day)	13.6	1.2	41.2
Temperature (°C)	9.4	6.0	14.9
pH	6.5	6.2	7.1
Nitrate concentration ($mgNO_3$-N/L)	30.7	1.4	45.2
Nitrite concentration ($mgNO_2$-N/L)	0.2	0	2.0

Ammonia concentration ($mgNH_4$-N/L)	0.2	0	0.8
Nitrogen concentration (mgN/L)	34.3	17.8	52.9
Phosphate concentration ($mgPO_4$-P/L)	0.2	0	2.4
Total Carbon concentration (mgC/L)	44.6	29.6	83.6

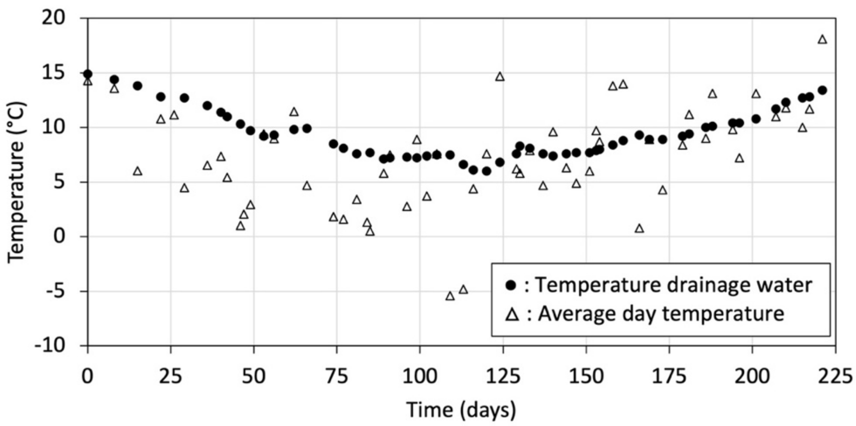

Figure 4: The average day temperature and the temperature of the drainage water throughout the drainage season 2020-2021

Nitrogen reduction

The effect of the denitrifying MBBR on the nitrogen concentrations in the drainage water are shown in Figure 5. Based on these results, the removal efficiency of nitrate and total nitrogen (TN) was calculated (see Figure 6). At last, Figure 7 represents the impact of discharging the treated drainage water into the local surface water stream.

In the first 90 days, varying nitrate concentrations in the effluent of the MBBR were measured. High nitrate and nitrogen removal efficiencies alternate with lower values. The nitrate removal efficiency differs from 51% to 100%, achieving an average value of 80%. A similar trend can be noticed for the nitrogen removal efficiency, although the difference between minimum and maximum value, respectively 5% and 95%, is much larger. This can be explained by the high nitrite concentrations measured when the nitrogen removal efficiency is low. Nitrite concentrations up to 18.4 $mgNO_2$-N/L were observed. This indicates clearly that the denitrification reaction is not complete and that nitrite is building up in the MBBR. High nitrite concentrations should be prevented at all cause because their presence is recognized to inhibit the growth of denitrifying bacteria (Rake and Eagon, 1980). Moreover, increased N_2O emissions can be expected when nitrite is accumulating (Kampschreur et al., 2009). Knowing that the potential for global warming is roughly 300 times higher than CO_2, the production of this greenhouse gas should certainly not be underestimated. Low COD/N ratios, oxygen concentrations greater than 0.4 mg O_2/L, nitrite accumulation, low pH values (< 7.5) and rapid fluctuations of these parameters can give rise to increased N_2O production during the denitrification process (Hanaki et al., 1992; Kamschreur et al., 2008; Tallec et al., 2006, Tallec et al., 2008). This means it is important to include NO_2-N measurements while evaluating the MBBR denitrifying performance since they can be used as an indicator of long-term variations in N_2O emissions and reveal abnormal process conditions (Kuokkanen et al., 2021). Nevertheless, Conthe et al. (2009) concluded that denitrification can act as a potential N_2O sink since the N_2O reducing capacity of a denitrifying community is in general higher than the producing capacity (factor of 2-10).

After day 90, the nitrate effluent concentrations show a clear increasing trend, resulting in a decreasing nitrate and nitrogen removal efficiency. At day 124 (23 February 2021), only 21% nitrate removal efficiency and 23% nitrogen removal efficiency were observed. During this period also the coldest temperatures were measured inside the MBBR. However, the first 90 days already indicated that nitrate was not completely converted into nitrogen gas and an intervention was necessary. A good mixing is required to uniformly distribute the plastic carriers throughout the MBBR and to achieve satisfactory results. Since this mixing was minimized during the first 124 days (to limit oxygen uptake), it was chosen to intensify the mixing, i.e. 5 min aeration/hour instead of 3 min aeration/3hours. The grey vertical line in Figure 5, Figure 6 and Figure 7 indicates when the intervention occurred. After a short adaption period, both low nitrate and nitrite concentrations are observed, indicating the complete denitrification. This statement is also supported by the low TN concentrations. An average TN concentration of 6.5 mgN/L between day 158 (29 March 2021) and day 221 (31 May 2021) is determined, resulting in an increased nitrate and nitrogen removal efficiency of 87% and 78%.

Finally, the treated effluent of the MBBR is discharged into the local surface water stream. To illustrate the effect of the effluent on the quality of the receiving surface water, the nitrate concentration directly before and after the MBBR is measured and showed in Figure 7. Approximately 500 meters downstream from the discharge point of the MBBR, the Environmental Agency VMM has a measuring point (see Figure 1) to check the water quality of the catchment area. The results showed clearly that the nitrate concentration of the surface water increases when low removal efficiencies were achieved, while high removal efficiencies resulted in similar or slightly lower concentrations. So, the surface water quality (on nitrate basis) is not significantly improved, although an adverse effect can be avoided. The EU standard limit was exceeded only once at the VMM measuring point.

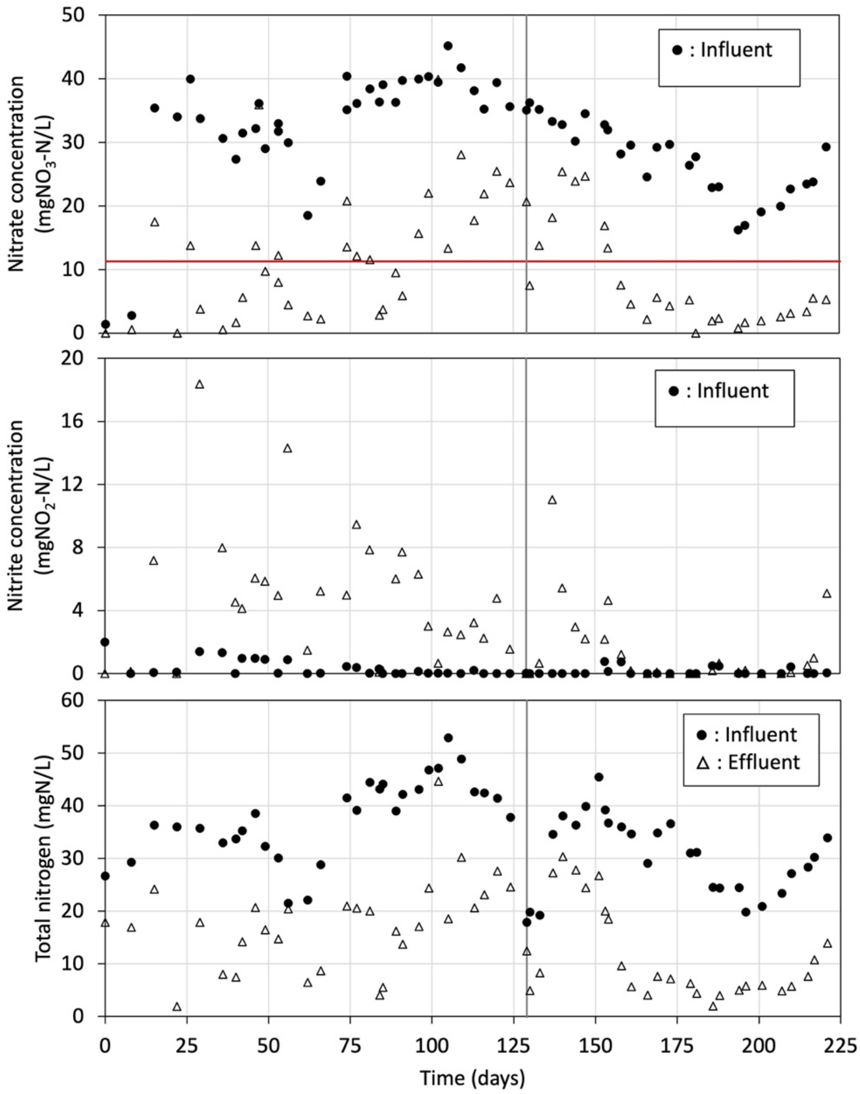

Figure 5 Evolution of nitrate, nitrite and total nitrogen concentration throughout the drainage season 2020-2021 (the red horizontal line indicates the nitrate discharge limit, the grey vertical line shows when the mixing is intensified)

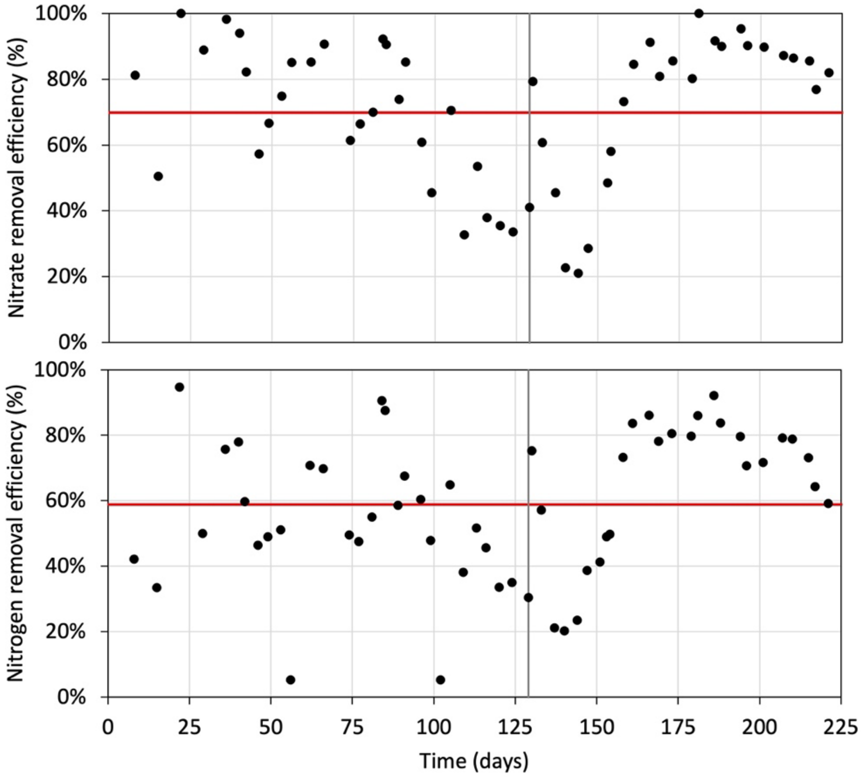

Figure 6 The removal efficiency (•) of the MBBR throughout the drainage season 2020-2021 (the red line represents the average removal efficiency considering the whole drainage season, the grey vertical line shows when the mixing is intensified)

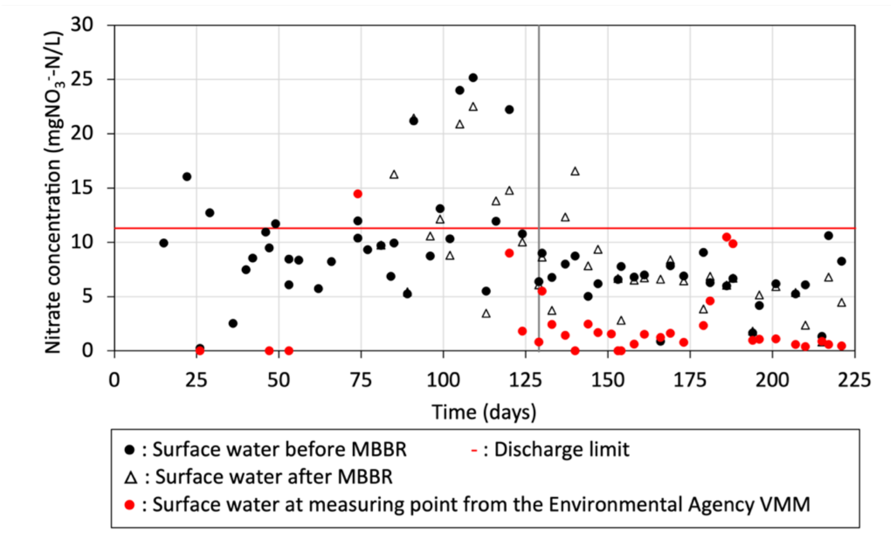

Figure 7 The effect of the MBBR on the nitrate concentration of the receiving surface water (the red line represents the average removal efficiency considering the whole drainage season, the grey vertical line shows when the mixing is intensified)

Effect of carbon source

The carbon source is necessary to maintain the denitrification reaction. Without the carbon, nitrate cannot be converted into nitrogen gas. Furthermore, to enable the total nitrate conversion and to prevent the formation of intermediate nitrogen species, excess quantities of carbon source are necessary, resulting in the COD/NO_3-N ratio of 8. Consequently, this will influence the effluent quality of the MBBR, which is shown in Figure 8. The concentration of Total Carbon (TC) in the effluent of the MBBR is significantly higher than the influent concentration. In the effluent is an average value of 172 mgC/L observed, which is almost 3 times higher than the average TC concentration of the untreated drainage water. Clearly, this affects the receiving surface water directly after discharge and at the measuring point of the environmental agency. A solution to reduce this effect is necessary.

Together with the carbon source, a small amount of phosphate is added to the MBBR to allow the biofilm to grow. It is important that the phosphate is used for biofilm growth and it will not result in an increased phosphate concentration of the effluent. The average effluent concentration was 0.09 mgPO_4-P/L, even lower than the average influent concentration, i.e. 0.16 mgPO_4-P/L. This indicates that the desired effect was achieved.

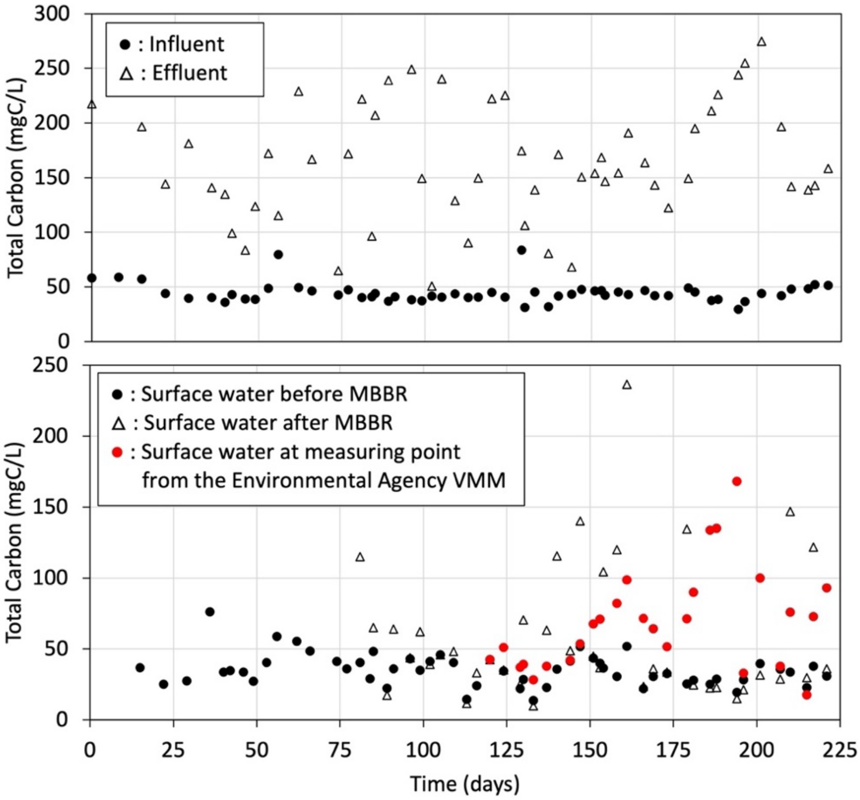

Figure 8 Evolution of the Total Carbon concentration throughout the drainage season 2020-2021

Financial aspect

The economic analysis of the MBBR in Onze-Lieve-Vrouw-Waver is summarized in Table 2. The investment cost includes the equipment, installation and AnoxK^{TM}5 carriers. The operational cost is calculated based on the consumables of the drainage season 2020-2021. The most important consumable cost is the carbon source. The energy cost, on the other hand, is rather limited. Finally, the operational cost also considers spare parts and maintenance. Based on these costs, the annual cost is calculated, considering a depreciation of the investment cost of 15 years and an interest of 4%. Finally, a total cost efficiency of 103.4 €/kg NO_3-N is determined.

Table 2: Economic analysis of the MBBR for the drainage season 2020-2021

Economic analysis	Cost	remarks
Investment cost	**€ 30 000**	*Installation + carriers; depreciation: 15 years*
Operational cost (for 1 drainage season)	**€ 3 185**	
- Carbon source (1 237 L)	€ 1 820	*Carbo ST HQ: € 1.47/L*
- Energy cost (1 118 kWh)	€ 90	*Electricity: € 0.08/kWh*
- Spare parts + maintenance	€ 1 275	
Total annual cost	**€ 5 956**	*Interest: 4%*
Total cost efficiency for 2020-2021	**€ 103.4/kg NO_3-N**	

Conclusion

During the drainage season of 2020-2021, a total amount of 2910,1 m^3 drainage water has been treated by the Moving bed biofilm reactor. In total 59 kg NO_3-N is removed, resulting in an average nitrate and nitrogen removal efficiency of 70% and 59%, respectively. Despite these good results, which are comparable with alternative technologies such as constructed wetlands and wood chips bioreactors (Bell et al., 2015; Lavrnic et al., 2018; Weerakoon et al., 2018), the monitoring of the MBBR has showed that improvements are possible if the following aspects are considered:

1. **<u>Mixing:</u>** the mixing of the carriers is very important to increase the transport of nitrate and carbon source to the biofilm. By increasing the mixing intensity, the removal efficiencies of both nitrate and nitrogen removal were improved significantly and showed a stable trend. Above all, the formation of nitrite was no longer observed. Nitrite accumulation should be prevented since it can trigger the production of N_2O. Together with nitrite accumulation, low COD/N ratios, low pH values, high oxygen concentrations (> 0.4 mg O_2/L) and rapid fluctuations of these parameters needs to be avoided to prevent or limit the N_2O production.
2. **<u>Carbon source:</u>** a high COD/N ratio is necessary to maintain the denitrification and to prevent the nitrite accumulation. If the supply of carbon source is hindered, a direct negative effect is noticed on the removal efficiency. Maintenance on a regular basis is necessary to ensure the proper operation of the MBBR installation. Unfortunately, a part of the carbon source remains unused and will end up in the effluent of the MBBR and eventually in the receiving surface water. This effect was clearly shown because of the higher TC values at the discharge point of the MBBR, but also more downstream at the measurement point of the environmental agency. This aspect needs more research and an action point in the next steps.
3. **<u>Phosphate source:</u>** a small amount of phosphate is dosed to improve the biofilm growth. Because phosphate itself has an important impact on eutrophication and cannot end up in the effluent of the MBBR, it is important that it is sufficiently removed. With an average phosphate concentration of 0.09 mgPO_4-P/L, an even lower phosphate level was observed compared with the original drainage water. So, no effect of the phosphate addition on the effluent quality was noticed.

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Appendix 1:

The minimum necessary data for calculating the dimensions of an MBBR plant are:

(i) The influent nitrate concentration $\left(C^{IN}_{NO_3-N}\right)$ and the predetermined effluent nitrate concentration $\left(C^{EFF}_{NO_3-N}\right)$ expressed in mg NO3-N/L

(ii) The drainage flow rate to be processed by the MBBR (Q) expressed in m^3/day,

(iii) The degree of filling with AnoxK™5 carriers (%K5) expressed in % and

(iv) The minimum water temperature (T_{min}).

First, the mass flow rate of nitrate Q_{mass}, expressed in (kg NO_3-N/day) can be calculated.

$$Q_{mass}\left(\frac{kg\ NO_3 - N}{day}\right) = \frac{\left(C^{IN}_{NO_3-N} - C^{EFF}_{NO_3-N}\right) * Q}{1000} = 1.08\ \frac{kg\ NO_3 - N}{day}$$

Based on the mass flow rate of nitrate Q_{mass} and the minimum denitrification rate $\left(k_{T,MIN}\right)$ expressed in (g NO_3-N/m^3.day) that can be guaranteed at the predetermined minimum water temperature (T_{min}), the MBBR volume in m^3 (V_{MBBR}) can then be easily calculated.

$$V_{MBBR}\ (m^3) = \frac{Q_{mass}}{k_{T,MIN} *\ \%K5} * 1000 = 12.4\ m^3$$

Note that the minimum denitrification rate depends on the temperature. For 6°C for example, it is 302 g NO_3-N/m^3.day or 0.38 g NO_3-N/m^2.day. To express the denitrification rate of g NO_3-N/m^2.day in g NO_3-N/m^3.day, multiply by the specific surface area of the AnoxK™5 carriers (800 m^2/m^3). $k_{T,MIN}$ is calculated as follows based on a minimum denitrification rate of 1 g NO_3-N/m^3.day at a water temperature of 20°C:

$$\begin{aligned} \mathrm{k_{T,MIN}} &= \mathrm{k_{20}} * \theta^{(\mathrm{T}-20)} = \ 1\ \mathrm{g\ NO_3 - N}/(m^2.day) * 1.072^{(T-20)} = 0.38\ \mathrm{g\ NO_3 - N}/(m^2.day) \\ &= \ 800\ \mathrm{g\ NO_3 - N}/(m^3.day) * 1.072^{(T-20)} = 302\ \mathrm{g\ NO_3 - N}/(m^3.day) \end{aligned}$$

From this MBBR volume, the following parameters can then be easily calculated:

(i) Total volume of AnoxK™5 (m^3): $V_{K5} = \frac{Q_{mass}}{k_{T,MIN}} * 1000 = \ 4.35\ m^3$

(ii) Total area of AnoxK™5 (m^2): $A_{K5} = \ V_{K5} * 800\ m^2/m^3 = 3480\ m^2$

(iii) The hydraulic retention time (h): $HRT = \frac{V_{MBBR}}{Q} = 12\ h$