SHORT-TERM BEHAVIOUR OF CONSTRUCTED REED BEDS : PILOT PLANT EXPERIMENTS UNDER DIFFERENT TEMPERATURE CONDITIONS

Diederik Rousseau*[§], Dieter Geenens[&], Peter A. Vanrolleghem[#], Niels De Pauw*

* Dept. Applied Ecology & Environmental Biology, Ghent University, J. Plateaustraat 22, B-9000 Gent, Belgium
[&] Aquafin NV, Dijkstraat 8, B-2630 Aartselaar, Belgium
[#] BIOMATH, Ghent University, Coupure Links 653, B-9000 Gent, Belgium

[§]corresponding author – email: Diederik.Rousseau@rug.ac.be - fax: +32 9 264 41 99

ABSTRACT

Decades of research on constructed wetlands have revealed the need for better insight in internal processes and better design and management tools. Current research is therefore increasingly oriented towards modelling, especially dynamic modelling. These models require large and high-frequency datasets for calibration and validation purposes but at present little is known about the short-term behaviour of constructed wetlands. This study intensively examined a pilot-scale two-stage constructed wetland via both low-frequency and high-frequency sampling. Low-frequency sampling was conducted from Spring 1997 till Spring 2000. Two additional 10-day monitoring campaigns were conducted, one in winter conditions (January 2001) and one in summer conditions (August 2001), during which samples of influent and effluent were collected at intervals no longer than 8 hours. This paper describes the results from all three monitoring campaigns and compares the short-term datasets with the long-term one. Some modelling recommendations are deduced from the results.

KEYWORDS

constructed wetlands, sampling frequency, dynamic modelling, calibration, two-stage reed bed

INTRODUCTION

The increasing application of constructed wetlands for wastewater treatment coupled to increasingly strict water quality standards is an incentive for the development of better design tools. Originally working with simple regression equations, most researchers and designers evolved towards the use of the well-known first-order k-C* model [1]. However, this black-box model is based on only two parameters, the first-order decay rate k, and the background concentration C*, which is an obvious oversimplification of the complex wetland processes. Kadlec [2] also demonstrated that the so-called rate constant k was not constant at all but depends on factors such as loading rate, inlet concentrations, etc. The addition of an extra parameter did not improve the model output.

More recently, several dynamic, compartmental models have been presented in the literature , a.o. [3, 4], which explicitly take into account the different processes that occur in constructed wetlands. Simulation results of these models seemed very promising. These detailed models however have one major drawback : they contain several dozens of parameters that have to be estimated. A sensitivity analysis can reveal those insensitive parameters that do not require a very accurate estimation. Parameters on the other hand that have a major influence on the model output have to be determined precisely [5]. Since little has been published concerning the values of most of these parameters, calibration must be based on input-output data. Considering the fact that time constants of certain microbial and physical-chemical reactions range between seconds and hours, calibration probably requires large high-frequency datasets.

A limited literature survey revealed that little data of this high detail exist. In most studies, only wastewater flow rates and some physical-chemical characteristics like dissolved oxygen, pH and temperature were monitored (semi)continuously, whereas data on BOD, COD, suspended solids, nitrogen and phosphorus were only collected biweekly, e.g. [4, 6-9]. In some other studies, grab samples of influent and effluent were taken at monthly [10, 11] or three-monthly [12] intervals. Braskerud [13] on the contrary continuously collected flow-based composite samples that were however only analysed approximately every 10 days, which provided interesting information about the overall mass balances, but which also masked the dynamic behaviour of the system. Bolton and Greenway [14] took daily grab samples at the inlet, middle and outlet sampling stations with some time in between to take into account the hydraulic residence time.

This study therefore intends to investigate the influence of the data collection frequency on the manifestation of certain processes and thus on model building. An existing long-term dataset from a two-stage pilot-scale constructed wetland was first examined to pinpoint the major processes that should be included in a dynamic model [15, 16]. To differentiate between slow and fast processes, two additional monitoring campaigns have been conducted with the same pilot plant under different temperature conditions. During these 10-day campaigns, samples were collected at regular, small intervals. This paper describes the results from these additional monitoring campaigns and compares these short-term datasets with the long-term one.

MATERIALS AND METHODS

The 10 P.E. experimental pilot plant, constructed in 1997 by Aquafin NV, consists of two parallel vertical flow reed beds (VFRBs) followed by a single horizontal subsurface flow reed bed (HFRB). The system has an approximate footprint of 4.7 m²/P.E. For detailed design information, the reader is referred to Vandaele *et al.* [15]. The following paragraphs only summarise the main characteristics of the pilot plant.

In both VFRBs, a gravel (60 – 100 mm) layer of 30 cm was positioned on top of a HDPE foil. Two perforated PVC drainage tubes were buried in this layer to evacuate the effluent, at intervals of 50 cm. Above this drainage layer, a geotextile prevents the filter layer on top to penetrate in the drainage zone. The filter layer is a 60 cm thick 50/50 mixture of sand (d_{10} of 0.25 – 0.45 mm) and gravel (d_{10} of 2 – 4 mm). Seedlings of *Phragmites spp.* were planted in this substrate at a density of 12 plants/m².

In the inlet zone of the SSF bed, a stone layer of 60 cm depth and 125 cm width assures an equal distribution of wastewater over the entire bed width. Between the inlet and outlet zones and the filter layer, a water permeable geotextile prevents mixing of the matrix material of these zones. The 60 cm deep filter layer consists of coarse gravel with a d_{10} of 5-10 mm. Reed seedlings were again planted at a density of 12 plants/m².

Wastewater is pumped from the primary clarifier of the WWTP Aartselaar and fed alternately (daily interval) and intermittently (pumping interval of 100 minutes) to the vertical flow beds.

The experiments were run from 20 till 29 January and from 14 till 23 August 2001. From 20 till 25 January and from 14 till 17 August, the influent flow rate was set at 1.3 m³/day (1.0 DWF). During the second half of the measuring campaign, influent flow rates were raised to 1.9 m³/day (1.5 DWF) and 3.9 m³/day (3.0 DWF) for the winter and summer campaign respectively. Composite samples of the influent were taken at 2 or 3 hour intervals whereas composite samples of the effluents were taken at intervals between 4 and 8 hours. Common physical-chemical variables were measured *ex situ* in the lab according to Standard Methods [17].

The long-term dataset was collected from spring 1997 till spring 2000 and is based on more or less weekly grab samples from the influent, the effluent of the VFRB and the effluent of the

HFRB. The flow rate was set at 1.0 DWF. Detailed information about the low-frequent, long-term dataset can be found in Vandaele *et al.* [15].

RESULTS

Meteorological conditions

The meteorological conditions for the short-term measuring campaigns of January and August 2001 are given in Figure 1. During the long-term campaign, no detailed meteorological data were collected. Therefore, only average behaviour during summer and winter is being considered. Belgium has a temperate climate with an average winter and summer temperature of about 6 °C and 18 °C respectively.

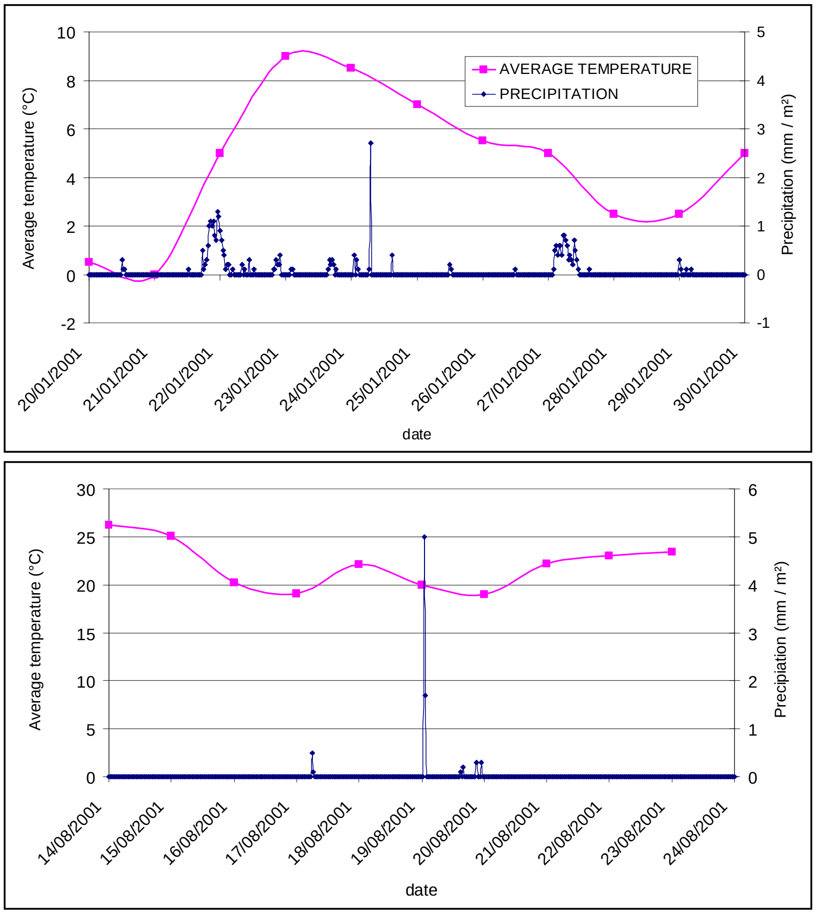


Fig. 1. Average air temperature (daily interval) and precipitation data (30 minute interval) of the measuring campaigns of January and August 2001 at Aartselaar.

Chemical Oxygen Demand (COD) and Suspended solids (SS)

Figure 2 shows a remarkably strong peek shaving exerted by the constructed wetland for both COD and SS and for winter and summer conditions. Indeed the effluent concentrations remain quite stable compared to the rapidly and strongly varying influent concentrations.

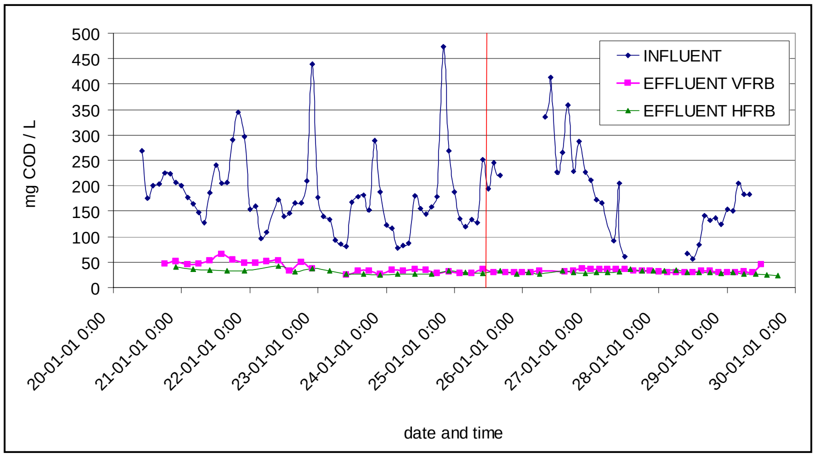
However, effluent disturbances do occur from time to time and can be clearly attributed to higher influent concentrations or higher influent flow rates.

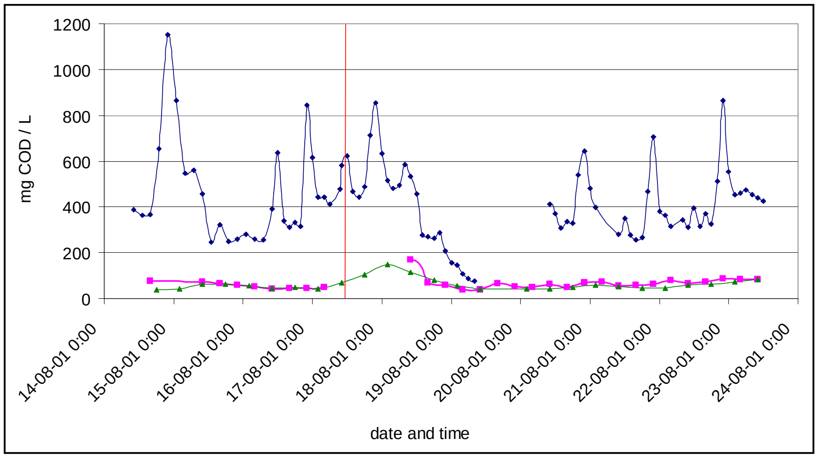
For instance, higher influent concentrations occurred during the winter of 1997/1998, due to a malfunctioning primary clarifier. This immediately caused increased COD and SS influent and effluent levels and resulted shortly afterwards in clogging problems [15].

The increase in flow rate from 1.0 to 1.5 DWF in January 2001 did not influence the effluent, but the increase to 3.0 DWF in August 2001 clearly did : on the 17th and 18th of August, effluent concentrations of COD as well as SS show an initial increase but level off after about 1.5 days on a slightly higher concentration as before. It was hypothesized that due to the higher flow velocity, some of the settled materials were resuspended and dragged out of the porous soil matrix. As soon as most of these loose materials were flushed, the effluent concentrations dropped again.

Effluent standards in Flanders for small-scale wastewater treatment plants (< 2.000 PE) are 250 mg COD/L and 60 mg SS/L. They were largely met during the short-term campaigns, although the data indicate that effluent SS levels could shortly exceed the standard when the flow rate increases considerably. However, it should be taken into account that the sewer network from which the influent is used, is a mixed one. Therefore, the influent concentrations for COD as well as SS are significantly lower during winter due to a higher dilution with urban runoff and the pretreated wastewater concentrations are therefore most of the time already lower than the effluent standards.

Data from the long-term campaign [15] only show effluent standard exceedances when the primary clarifier was out of order. Adequate primary treatment thus seems to be of utmost importance, because higher particulate influent loads cause higher effluent concentrations on a very short timescale, but also cause clogging after a couple of days, leaving the reed bed useless until the hydraulic conductivity has been restored.





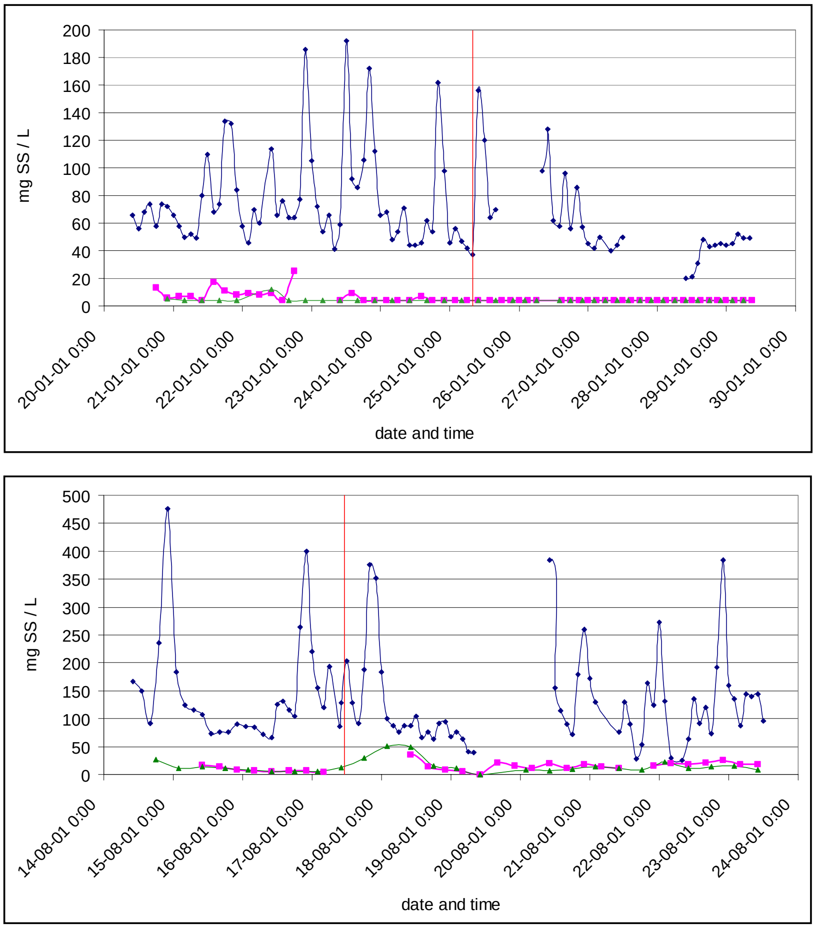


Fig. 2. Influent and effluent COD concentrations (two upper panels) and SS concentrations (two lower panels) of January and August 2001 sampled on the pilot-scale two-stage constructed wetland at Aartselaar.

The average removal efficiencies for COD and SS for the different reed beds, the different seasons and years and the different hydraulic loading rates can be found in Tables 1 and 2. The total two-stage system shows efficiencies not lower than 83 % for COD removal and 86 % for SS removal, which is within commonly reported ranges [18].

Table 1. Average removal efficiencies (expressed as %) during January and August 2001 for the different reed bed stages and for the different hydraulic loading rates (expressed as DWF).

	First stage (VFRBs)						**Second stage (HFRB)**						**Total two-stage system**					
	January 2001			**August 2001**			**January 2001**			**August 2001**			**January 2001**			**August 2001**		
DWF	1.0	1.5	Av.	1.0	3.0	Av.	1.0	1.5	Av.	1.0	3.0	Av.	1.0	1.5	Av.	1.0	3.0	Av.
COD	78	83	81	88	83	85	21	9	16	9	1	4	83	85	84	89	83	85
SS	91	93	92	94	87	90	37	0	23	neg	neg	neg	94	93	94	92	87	89
NH_4	64	86	75	93	70	77	23	18	25	34	6	13	72	88	81	89	72	80
NO_3	neg	neg	neg	neg	neg	neg	0	3	0	31	27	24	neg	neg	neg	neg	neg	neg
TN	12	neg	7	34	42	40	7	1	4	28	14	18	18	neg	11	53	50	51
orthoP	52	33	46	72	58	63	15	10	12	38	13	20	59	39	52	82	64	70
TP	71	72	72	78	54	59	27	10	19	23	42	41	79	75	78	83	73	76

Data from Table 1 suggest that COD removal efficiencies of the VFRB at 1.0 DWF are somewhat better during the summer period (88 vs. 78 % during winter). One possible explanation could be the temperature dependency of certain processes, although the temperature coefficient for COD removal is mostly said to be near to one [1]. Another explanation could be a better removal due to higher influent loadings, as a.o. suggested by Kadlec [2] and Mitchell and McNevin [19].

Suspended solids removal efficiencies of the VFRBs at 1.0 DWF show almost no seasonal influence (94 % during summer vs. 91 % during winter), as can be expected since most removal processes are physically based.

These seasonal trends are not confirmed by the data of the long-term dataset, summarised in Table 2, because of the clogging problems that occurred from time to time.

Table 2. Average removal efficiencies (expressed as %) based on the long term measuring campaign, for the different stages and for different seasons and years.

		COD	**SS**	**NH_4**	**NO_3**	**TN**	**orthoP**	**TP**
First stage (VFRBs)	Winter ('97, '98, '99)	85	98	62	neg	25	27	70
	Summer ('97, '98, '99)	88	88	87	neg	27	42	50
	1997	87	97	81	neg	32	52	70
	1998	79	76	75	neg	5	37	37
	1999	83	79	80	neg	12	13	30
Second stage (HFRB)	Winter ('97, '98, '99)	7	17	8	47	26	8	15
	Summer ('97, '98, '99)	17	29	41	59	42	50	43
	1997	13	23	7	43	33	37	38
	1998	34	40	31	56	46	23	40
	1999	22	41	40	52	37	29	22
Total two-stage system	Winter ('97, '98, '99)	86	98	65	neg	45	34	74
	Summer ('97, '98, '99)	90	92	93	neg	58	71	71
	1997	88	98	83	neg	55	70	82
	1998	86	86	83	neg	49	52	62
	1999	87	88	88	neg	45	38	45

The role of the HFRB in COD and SS removal is more or less limited to the one of a backup system: removal efficiencies are only considerable when the VFRBs are overloaded, e.g. when the primary clarifier malfunctions (data not shown). On the contrary, when the influent concentrations to the HFRB are very low, slightly higher effluent concentrations sometimes occur (e.g. August 2001 data in Table 1), thus suggesting in some way a production of SS and the existence of a background concentration.

Finally, the data from the long-term dataset were used to investigate the importance of the reed bed age, as summarised in Table 2 for the years 1997, 1998 and 1999. COD removal efficiencies showed no significant decline or improvement during the first three years of operation whereas the SS removal efficiencies dropped after the first year. However, this is most likely attributable to clogging problems and not to maturation effects.

Total Nitrogen (TN)

The TN effluent concentrations shown in Figure 3 are less stable than those of COD and SS but do not seem to be correlated to the influent variations. Again, due to the mixed sewer network, summer influent concentrations are significantly higher than the winter ones. The different fractions of total nitrogen show these same trends (data not shown).

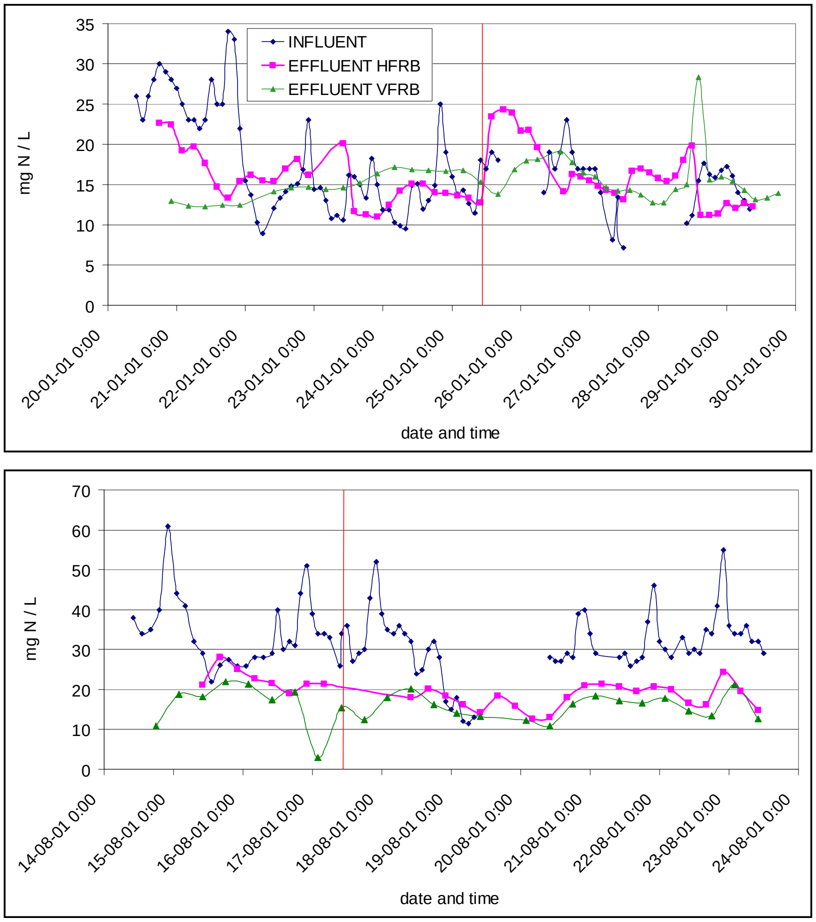


Fig. 3. Influent and effluent TN concentrations of January and August 2001 sampled on the pilot-scale two-stage constructed wetland at Aartselaar.

Investigation of the different nitrogen compounds showed that the majority of influent nitrogen occurs as the reduced form ammonium and that only a small fraction occurs as organic nitrogen. However, when the primary clarifier is out of order, the load of particulate organic nitrogen increases enormously. The presence of nitrite and nitrate in the influent was only evidenced after some rainfall events and always in minute quantities.

The VFRBs seem to play two important roles. First of all, they act as a physical filter by removing a great deal of the particulate nitrogen. Secondly, due to their capability to draw in oxygen, they acts as nitrification units. This oxygenation capacity is clearly demonstrated by data of the long-term dataset : when the primary clarifier malfunctioned and a thin sludge layer on the surface clogged the VFRBs, the nitrification rate dropped to near zero.

Nitrification is often considered to be temperature dependent [1], and this is strongly confirmed by the data in Tables 1 and 2 : the NH_4 removal efficiency increases from 64 % (January 2001) to 93 % (August 2001) at 1.0 DWF and from 62 % (winters ’97, ’98, ’99) to 87 % (summers ’97, ’98, ’99).
Surprisingly enough, the higher flow rates do not really seem to influence the effluent TN levels as can be seen in Figure 3. They do however negatively influence the nitrification process : effluent NH_4 concentrations of the VFRBs are considerably higher and are more subject to the influent variations. This can be due to the shorter residence time but also to a lack of oxygen because of a higher oxygen demand for COD removal. Anoxic conditions can be evidenced by the

enhanced denitrification in the VFRBs since the TN removal efficiency rises from 34 % at 1.0 DWF to 42 % at 3.0 DWF. This mechanism guarantees stable effluent TN concentrations.
Finally the effect of bed maturity was investigated on the long-term dataset (Table 2) but no apparent change in nitrification capacity was found during the course of time.

In contrast to COD and SS removal, the HFRB no longer functions as a sort of backup or polishing system but actively contributes to nitrogen removal, mostly via denitrification processes. One should however notice that some nitrification also occurs in the HFRB. The electron acceptor oxygen could initially be supplied by the influent to the HFRB itself, but for these rather high removal rates, it can also be considered that root release plays an important role. Indeed, the ammonium removal capacity sharply rises after the first winter of operation (from 7 % in 1997 to 31 % in 1998 and 40 % in 1999), which could possibly be correlated to an expanding amount of roots and rhizomes and thus more oxygen leakage.

A lot of nitrate is being denitrified during warmer periods, but denitrification seems almost halted during colder periods (Tables 1 and 2) : the NO_3 removal efficiency of the HFRB increases from 0 % (January 2001) to 31 % (August 2001) at 1.0 DWF and from 47 % (winters ’97, ’98, ’99) to 59 % (summers ’97, ’98, ’99).
A last factor that seems to be of importance is the availability of a carbon source in the HFRB. After one year of operation of the reed beds, the denitrification capacity significantly increased (from 43 in 1997 to 56 % in 1998) and it was already hypothesized before that the first litter production could have provided an extra carbon source for denitrification [15, 20].

Phosphorus

The influent and effluent TP concentrations of January and August 2001 are shown in Figure 4. Incoming phosphorus mainly consists of orthophosphates, except for the January 2001 measuring campaign where organic phosphorus slightly predominates. However, when looking at absolute concentrations in stead of relative fractions, influent organic phosphorus concentrations remain relatively stable, but orthophosphate concentrations are under influence of the mixed sewer system (data not shown).

In Figure 4, it can be seen that the VFRB clearly plays the most important role in phosphorus removal. Two major processes can be assumed to occur : physical filtration of particulate phosphorus and sorption of dissolved phosphorus. Both processes are only of minor importance in the HFRB, due to its matrix material. Indeed, the gravel in the HFRB has less filtration and sorption capacity than the sand in the VFRBs. However, as for COD and SS, the HFRB serves as a backup unit and reduces P concentrations when the VFRBs fail.

From Tables 1 and 2, it can be concluded that orthoP removal in the VFRBs as well as the HFRB is influenced by several factors.
First of all, orthoP elimination decreases when the flow rate increases, although the concentration changes are minute in an absolute way.
Secondly, there is a seasonal dependence that is most likely linked to plant growth and subsequent phosphorus uptake. The OP removal efficiency of the total two-stage system increases from 59 % (January 2001) to 82 % (August 2001) at 1.0 DWF and from 34 % (winters of ’97, ’98, ’99) to 71 % (summers of ’97, ’98, ’99).
Finally, OP removal clearly decreases in time (from 70 % in 1997 to 52 % in 1998 and 38 % in 1999), due to saturation of the sorption sites, with a later breakthrough being noticeable for the HFRB.

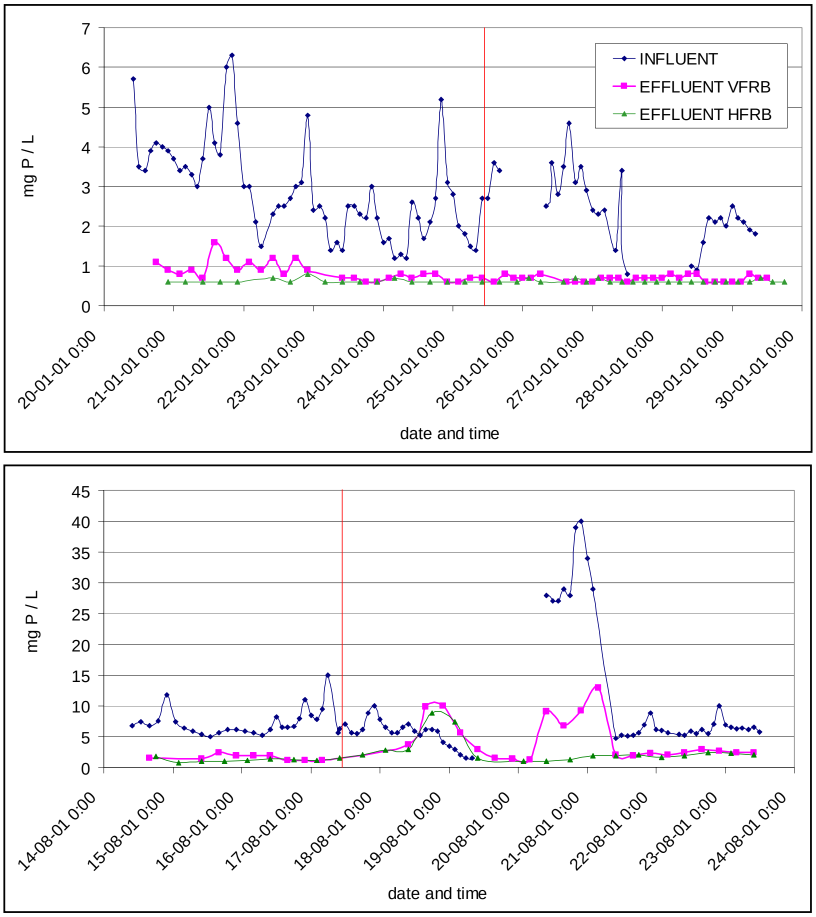


Fig. 4. Influent and effluent TP concentrations of January and August 2001 sampled on the pilot-scale two-stage constructed wetland at Aartselaar.

Concerning TP removal, similar conclusions can be drawn since TP mainly consists of orthoP. However, the data are sometimes blurred by a sudden failure of primary treatment with higher particulate phosphorus loads as a consequence. Figure 4 again reveals the same trend after the increase in flow rate from 1.0 tot 3.0 DWF : an immediate but short-lasting wash out of particulate phosphorus.

DISCUSSION

When looking at both high-frequency datasets, it immediately becomes clear that a lot of dynamics would be lost when low-frequent monitoring would have been applied. This is especially the case for the highly variable influent, but to a certain extent also for nitrogen and phosphorus effluent concentrations.

Important consequences can be deduced for the monitoring strategy. A low sampling frequency could in most cases still adequately describe the effluent behaviour, but could give a totally wrong impression of the influent and could thus lead to false interpretations of the system dynamics and of the removal efficiencies.

It is therefore recommended to tune the sampling frequency to the processes under study. If the goal of the study for example is nutrient uptake by plants, low-frequent flow-averaged sampling will suffice since plant growth is a slow process. If on the contrary the goal would be to investigate the behaviour under different hydraulic conditions, high frequent sampling will be needed.

Considering the removal efficiencies of this two-stage constructed wetland, one could question the ecological and economical relevance of the HFRB since most removal efficiencies appear to be quite low. This reed bed is however an essential part of the concept since it contributes significantly to nitrogen removal, and since it serves as a backup and polishing unit in case the VFRBs get overloaded. Indeed, the VFRBs are responsible for the major part of pollution reduction, but they are, unfortunately enough, extremely sensitive to clogging in case the primary treatment fails. Similar experiences were also obtained by Aquafin NV with full-scale reed beds of the same concept.

Bearing in mind the temperate climatic conditions of Belgium, this pilot-scale two-stage constructed wetland does a good job in removing oxygen demand and suspended solids during all seasons, but shows a reduced phosphorus removal and an almost non-existing nitrogen removal during winter. Fortunately, the watercourses in which the effluent is discharged are less subject to eutrophication during colder periods, so the ecological impact remains rather limited.

Another important performance factor was the influent flow rate or rather the change in flow rate. It was demonstrated that when the flow rate suddenly increased from 1.0 to 3.0 DWF, a wash-out of particulate substances occurred. In most cases, effluent concentrations almost decreased again to their original concentrations after a short period of time.

Finally, the bed age seemed to be positively correlated to denitrification, supposedly due to a higher availability of carbon supplied by the breakdown of plant litter. A negative correlation was on the contrary detected between bed age and phosphorus removal, most likely due to a saturation of sorption sites.

MODELLING RECOMMENDATIONS

Applying the first-order k-C* model implies using averaged conditions. Averaging over at least 3 times the hydraulic residence time is recommended by Kadlec & Knight [1]. This method therefore does not allow predicting the effluent variability and consequently the number of exceedances of a certain effluent standard. If that is of interest, one should switch to dynamic models like the ones from McBride and Tanner [3] or Wynn and Liehr [4].

Based on the data from the three measuring campaigns, the following conclusions and recommendations can be formulated concerning dynamic modelling of COD, SS, N and P removal processes in constructed wetlands :

- Knowledge of the hydraulic behaviour is of utmost importance : reed beds showing preferential channeling and dead zones will have a faster breakthrough and will exert less buffering influence on the effluent concentrations. For existing constructed wetlands, this can be assessed by means of high-frequency sampling campaigns or by means of a tracer test. However, during the design stage of a constructed wetland, these data are not available. Therefore, assessment of the hydraulic conductivity of the bed material and knowledge of its decrease during the course of time are important issues.
- A good knowledge of the bed material is also of importance to predict the availability of sorption sites for phosphorus removal.
- An adequate influent characterisation is needed : working with daily averaged influent concentrations and flow rates is common practice but is strongly discouraged for dynamic modelling purposes. For nitrogen and phosphorus removal, knowledge about the different fractions is advisable.

- A submodel for particulate matter behaviour should be included, with the filtration efficiency being dependent on flow velocity. This model should also be able to predict clogging phenomena.
- Some production processes should be modelled as well, as was demonstrated by the background concentrations for SS or by the extra carbon source that became available from plant debris after the first year of operation. A litter compartment is therefore recommendable.
- Measured phosphorus removal rates suggested that plant uptake is not negligible and thus should be modelled.
- Temperature dependencies should be included, especially for reactions in the nitrogen cycle.

ACKNOWLEDGEMENTS

The authors would especially like to thank MSc students An De Moor and Wouter De Wilde for their valuable help during sampling and data treatment.

REFERENCES

[1] Kadlec, R.H. and Knight, R.L. (1996). *Treatment wetlands*. CRC Press, Boca Raton FL, 893 pp.

[2] Kadlec, R. H. (2000). “The inadequacy of first-order treatment wetland models”. Ecological Engineering (15): 105-119.

[3] McBride, G. B. and Tanner, C. C. (2000). “Modelling biofilm nitrogen transformations in constructed wetland mesocosms with fluctuating water levels”. Ecological Engineering (14): 93-106.

[4] Wynn, T. M. and Liehr, S. K. (2001). “Development of a constructed subsurface-flow wetland simulation model”. Ecological Engineering (16): 519-536.

[5] Dochain, D. and Vanrolleghem, P.A. (2001). *Dynamical Modelling and Estimation in Wastewater Treatment Processes*. IWA Publishing, London, UK. ISBN 1-900222-50-7. pp. 342.

[6] Tanner, C. C., Clayton, J. S. and Upsdell, M. P. (1995a). “Effect of loading rate and planting on treatment of dairy farm wastewaters in constructed wetlands - I. Removal of oxygen demand, suspended solids and faecal coliforms”. Water Research 29(1): 17-26.

[7] Tanner, C. C., Clayton, J. S. and Upsdell, M. P. (1995b). “Effect of loading rate and planting on treatment of dairy farm wastewater in constructed wetlands - II. Removal of nitrogen and phosphorus”. Water Research 29(1): 27-34.

[8] Kozub, D. D. and Liehr, S. K. (1999). “Assessing denitrification rate limiting factors in a constructed wetland receiving landfill leachate”. Water Science & Technology 40(3): 75-82.

[9] Sakadevan, K. and Bavor, H. J. (1999). “Nutrient removal mechanisms in constructed wetlands and sustainable water management”. Water Science & Technology 40(2): 121-128.

[10] Bulc, T., Vrhovsek, D. and Kukanja, V.(1997). “The use of constructed wetland for landfill leachate treatment”. Water Science & Technology 35(5): 301-306.

[11] Cerezo, R. G., Suarez, M. L. and Vidal-Abarca, M. R. (2001). “The performance of a multi-stage system of constructed wetlands for urban wastewater treatment in a semiarid region of SE Spain”. Ecological Engineering (16): 501-517.

[12] Kern, J. and Idler, C. (1999). “Treatment of domestic and agricultural wastewater by reed bed systems”. Ecological Engineering (12): 13-25.

[13] Braskerud, B. C. (2001). “Factors affecting nitrogen retention in small constructed wetlands treating agricultural non-point source pollution”. Ecological Engineering 2001(18): 351-370.

[14] Bolton, K. G. E. and Greenway, M. (1999). “Pollutant removal capacity of a constructed Melaleuca wetland receiving primary settled sewage”. Water Science & Technology 39(6): 199-206.

[15] Vandaele, S., Thoeye, C., Van Eygen, B. and De Gueldre G. (2000). “SWWTP's in Flanders (Belgium): standard approach and experiences with constructed reed beds”. Water Science & Technology 44(1):57-63.

[16] Rousseau, D., De Wilde, W., Jonkers, C., Geenens, D., Vanrolleghem, P.A. and De Pauw, N. (2001). “Short-term behaviour of an experimental constructed wetland at low temperatures”. MFLBER, 66(4), 233-237.

[17] *Standard Methods for the Examination of Water and Wastewater* (1992). 18th Edn. American Public Health Association / American Waterworks Association / Water Environment Federation, Washington DC, USA.

[18] Kadlec, R.H., Knight R.L., Vymazal J., Brix H., Cooper P. and Haberl R. (2000). Constructed wetlands for pollution control: processes, performance, design and operation. IWA specialist group on use of macrophytes in water pollution control. Scientific and technical Report No. 8, IWA Publishing, London, UK, 156 pp.

[19] Mitchell, C. and McNevin, D. (2001). “Alternative analysis of BOD removal in subsurface flow constructed wetlands employing Monod kinetics”. Water Research 35(5): 1295-1303.

[20] Gersberg, R. M., Elkins, B. V. and Goldman, C. R. (1983). “Nitrogen removal in artificial wetlands”. Water Research 17(9): 1009-1014.