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Model-based design of horizontal subsurface flow constructed treatment wetlands: a review

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Abstract

The increasing application of constructed wetlands for wastewater treatment coupled with increasingly strict water quality standards is an ever growing incentive for the development of better process design tools. This paper reviews design models for horizontal subsurface flow constructed treatment wetlands, ranging from simple rules of thumb and regression equations, to the well-known first-order k-C^* models, Monod-type equations and more complex dynamic, compartmental models. Especially highlighted in this review are the model constraints and parameter uncertainty. A case study has been used to demonstrate the model output variability and to unravel whether or not more complex but also less manageable models offer a significant advantage to the designer.

Keywords: Constructed wetlands; Wastewater treatment; Regression equations; First-order models; Mechanistic models; Parameter uncertainty

1. Introduction

Treatment wetlands are either natural or constructed wetlands are almost completely covered with emerging macrophytes and they are being managed as water quality improving systems. Some commonly used helophytes are common reed (*Phragmites australis*), cattail (*Typha* spp.) and bulrush (*Scirpus* spp.), all characterised as water-tolerant macrophytes that are rooted in the soil but emerge above the water surface [1].

Although mainly applied for the purification of domestic wastewater, treatment wetlands are also used for purification of industrial wastewater [2,3], agricultural wastewater [4–6] and stormwaters [7,8]. They are furthermore applied to strip nutrients of eutrophied surface waters before these are discharged into vulnerable nature reserves [9–11].

It must however be stressed that treatment wetlands have several other functions. Next to water quality improvement, they can also function as a nature development area, a recreational area, a hydrological buffer or a reservoir [12].

Among the treatment wetlands, horizontal subsurface flow (SSF) constructed wetlands are a widely applied concept. Pretreated wastewater flows horizontally through the artificial filter bed, usually consisting of a matrix of sand or gravel and the helophyte roots and rhizomes. This matrix is colonised by a layer of attached microorganisms that forms a so-called biofilm. Purification is achieved by a wide variety of physical, chemical and (micro)biological processes, like sedimentation, filtration, precipitation, sorption, plant uptake, microbial decomposition and nitrogen transformations [13,14].

The increasing application of treatment wetlands coupled with increasingly strict water quality standards has been an incentive for the development of better design tools. This paper reviews some simple as well as some more elaborate design models and describes their

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Nomenclature

α	precipitation – evapotranspiration ($\mathrm{L\,T^{-1}}$)
ε	void fraction of wetland bed (1)
τ	hydraulic retention time (T)
θ	temperature factor (1)
A	bed surface ($\mathrm{L^2}$)
a	wetland cross-sectional area ($\mathrm{L^2}$)
b	time-based retardation coefficient ($\mathrm{T^{-1}}$)
C	concentration ($\mathrm{M\,L^{-3}}$)
C_{in}	influent concentration ($\mathrm{M\,L^{-3}}$)
C_{out}	effluent concentration ($\mathrm{M\,L^{-3}}$)
C^*	background concentration ($\mathrm{M\,L^{-3}}$)
d	bed depth (L)
K	half-saturation constant ($\mathrm{M\,L^{-3}}$)
K_0	initial first-order volumetric rate constant ($\mathrm{T^{-1}}$)
k_V	first-order volumetric rate constant ($\mathrm{T^{-1}}$)
k_A	first-order areal rate constant ($\mathrm{L\,T^{-1}}$)
$k_{A,T}$	first-order areal rate constant at temperature T (T in °C) ($\mathrm{L\,T^{-1}}$)
$k_{V,T}$	first-order volumetric rate constant at temperature T (T in °C) ($\mathrm{L\,T^{-1}}$)
$k_{A,20}$	first-order areal rate constant at temperature 20°C ($\mathrm{L\,T^{-1}}$)
$k_{V,20}$	first-order volumetric rate constant at temperature 20°C ($\mathrm{L\,T^{-1}}$)
$k_{0,V}$	zero-order volumetric rate constant ($\mathrm{M\,L^{-3}\,T^{-1}}$)
$k_{0,A}$	zero-order areal rate constant ($\mathrm{M\,L^{-2}\,T^{-1}}$)
L_{in}	influent load ($\mathrm{M\,L^{-2}\,T^{-1}}$)
L_{out}	effluent load ($\mathrm{M\,L^{-2}\,T^{-1}}$)
q	hydraulic loading rate HLR ($\mathrm{L\,T^{-1}}$)
Q	flow rate ($\mathrm{L^3\,T^{-1}}$)
r	removal rate ($\mathrm{M\,T^{-1}}$)
t	time (T)
T	influent temperature (°C)
v	water velocity ($\mathrm{L\,T^{-1}}$)
V	wetland holding volume ($\mathrm{L^3}$)
W	wetland width (L)
Z	wetland length (L)

merits as well as disadvantages with regard to the design of horizontal SSF constructed treatment wetlands. The focus is on the standard water quality variables such as chemical oxygen demand (COD), biochemical oxygen demand (BOD), total suspended solids (TSS), nitrogen (N) and phosphorus (P). Obviously, hydraulic models too can be very valuable for design purposes, but they merit a review on their own and are therefore not treated in this one. Special attention is also being paid to parameter uncertainty. All models have been tested in a case study with the aim to predict the required surface area. The case study was based on an existing dataset containing influent flows and concentrations, weather conditions and effluent requirements.

2. Model review

The model review starts with simple design models like rules of thumb and regression equations. Secondly, the well-known first-order $k - C^*$ model [13,15] and several of its extensions are treated. The review then continues with Monod-type equations and ends with a complex dynamic, compartmental model. Special attention is in each case paid to the model constraints and parameter uncertainty.

2.1. Rules of thumb

From an engineering point of view, rules of thumb are the fastest but also the roughest design methods. As an example, some of these rules for horizontal SSF constructed treatment wetlands described by Wood [16] and Kadlec and Knight [13] are summarised in Table 1. Since they are based on observations from a wide range of systems, climatic conditions and wastewater types, these rules of thumb show a large variation c.q. uncertainty and can thus better be used after more extensive calculations to check the design.

2.2. Regression equations

Considering the fact that the majority of the investigations on treatment wetlands have mainly been focused on input–output (I/O) data rather than on internal processes data, regression equations seem to be a useful tool in interpreting and applying these I/O data. However, these black box 'models' lump a complex system like a constructed treatment wetland into only two or three parameters, which is clearly an over-simplification. Important factors such as climate, bed material, bed design (length, width, depth), etc. are neglected, leading to a wide variety of regression equations and thus a large uncertainty in the design. A literature overview of regression equations for BOD, COD, TSS, TN and TP is presented in Table 2. The first two columns of Table 2 mention the reference and a short system description, the third column states the regression equation and the next three columns give the ranges of influent and effluent concentrations and hydraulic loading rates (HLRs) for which the equation

Table 1
Rule of thumb design criteria for horizontal SSF constructed treatment wetlands

Criterion	Value range	
	Wood [16]	Kadlec and Knight [13]
Hydraulic retention time (days)	2–7	2–4
Max. BOD loading rate ($kg\,BOD\,ha^{-1}\,day^{-1}$)	75	n.g.
Hydraulic loading rate ($cm\,day^{-1}$)	0.2–3.0	8–30
Areal requirement ($ha\,m^{-3}\,day$)	0.001–0.007	n.g.

n.g.: not given.

is valid. The last column indicates the coefficient of determination.

As shown in Table 2, most of these regression equations rely on wastewater concentrations. Looking for instance at the first table entry [17], this implies that for a constant BOD influent concentration, the same effluent concentration is predicted for a HLR of 0.8 as well as $22\,cm\,day^{-1}$, which suggests that the HLR is a non-limiting factor within certain boundaries. Only a limited number of regression equations rely on both influent concentration and HLR as inputs to predict the effluent concentration. Consequently, only those regression equations can be used to predict the maximum allowable HLR based on a given influent concentration and a given effluent standard.

2.3. First-order models

The state-of-the-art in constructed treatment wetlands' modelling consists of first-order equations [13,15] which in case of constant conditions (e.g. influent, flow and concentrations) and an ideal plug-flow behaviour predict an exponential profile between inlet and outlet (Eq. (1)):

$$\frac{dC}{dt} = -k_V C \overset{[1]}{\rightarrow} \left(\frac{C_{out} - C^*}{C_{in} - C^*}\right) = e^{(-k_V \tau)} \xrightarrow{[2],[3],[4]} \left(\frac{C_{out} - C^*}{C_{in} - C^*}\right) = e^{(-k_A/q)}, \quad (1)$$

transformation equations:

[1] $C_{in} = C(t=0)$
and $C_{out} = C(t=\tau)$, initial conditions;
[2] $k_A = k_V \varepsilon d$;
[3] $q = Q/A$;
[4] $V = Q\tau = Ad\varepsilon$.

The background concentration C^* in this model is explained by processes such as autochthonous production and/or sediment release.

Some model enhancements have been proposed to incorporate the effect of precipitation and evapotranspiration on the wetlands' performance, yielding a power law profile (Eq. (2)) between inlet and outlet for steady-state conditions [15]:

$$\frac{C_{out} - C'}{C_{in} - C'} = (1 + [\alpha/q])^{-(1+k_A/\alpha)} \quad \text{with} \quad C' = C^*\left[\frac{k_A}{k_A + \alpha}\right]. \quad (2)$$

The influence of temperature is commonly modelled via an Arrhenius equation (Eq. (3)):

$$k_{A,T} = k_{A,20}\theta^{(T-20)} \quad \text{and} \quad k_{V,T} = k_{V,20}\theta^{(T-20)}. \quad (3)$$

According to Kadlec and Knight [13], removal of BOD, TSS and TP in treatment wetlands is generally found to be independent of temperature ($\theta = 1.00$) whereas nitrogen removal is negatively influenced by lower temperatures ($\theta = 1.05$).

Shepherd et al. [23] recently presented a time-dependent retardation model for COD removal that replaces the background concentration C^* by two other parameters K_0 and b. They assumed that removal rates decrease during the course of time, because easily biodegradable substances are removed first and fast, thus leaving a solution with less biodegradable constituents and hence with slower removal kinetics. This continuous change in solution composition can be represented by a continuously varying first-order rate constant k (Eq. (4)):

$$k_v = \frac{K_0}{(b\tau + 1)}. \quad (4)$$

This retardation model was considered to be more appropriate for constructed treatment wetland design because it allows a steady decrease in COD (or any other component) with increased treatment time rather than a constant residual COD (C^*) value. When applied on data from a pilot-scale horizontal SSF constructed wetland for winery wastewater treatment, model calibration yielded K_0 values from 9 to $12\,day^{-1}$ and b values from 2 to $5\,day^{-1}$. Compared to the $k - C^*$ model, the time-dependent retardation model had more consistent parameters for COD removal data across different depths and at different loadings.

Calibration of the parameters k, C^* and θ is mostly done on the basis of I/O concentrations, and not on the basis of transect data, although the latter are to be

Table 2
Regression equations for horizontal SSF constructed treatment wetlands according to different authors (q expressed as $\mathrm{cm\,day^{-1}}$)

Reference	System	Equation	Input range	Output range	q range	R^2
BOD[a]						
Brix [17]	Danish and UK soil-based SSF	$C_{out} = (0.11*C_{in}) + 1.87$	$1 < C_{in} < 330$	$1 < C_{out} < 50$	$0.8 < q < 22$	0.74
Knight et al. [18]	US gravel beds (NADB)	$C_{out} = (0.33*C_{in}) + 1.4$	$1 < C_{in} < 57$	$1 < C_{out} < 36$	$1.9 < q < 11.4$	0.48
Griffin et al. [19]	US unplanted rock-filter	$C_{out} = 502.20*\exp(-0.111*T)$	$10 < T < 30$	n.g.	n.g.	0.69
Vymazal [20]	SSF in Czech Republic	$C_{out} = (0.099*C_{in}) + 3.24$	$5.8 < C_{in} < 328$	$1.3 < C_{out} < 51$	$0.6 < q < 14.2$	0.33
Reed and Brown [21]	14 US SSF	$L_{removed} = (0.653*L_{in}) + 0.292$	$4 < L_{in} < 145$	$4 < L_{removed} < 88$	n.g.	0.97
Vymazal [22]	SSF in Czech Republic	$L_{out} = (0.145*L_{in}) - 0.06$	$6 < L_{in} < 76$	$0.3 < L_{out} < 11$	n.g.	0.85
Vymazal [20]	SSF in Czech Republic	$L_{out} = (0.13*L_{in}) + 0.27$	$2.6 < L_{in} < 99.6$	$0.32 < L_{out} < 21.7$	$0.6 < q < 14.2$	0.57
COD[b]						
Vymazal [22]	SSF in Czech Republic	$L_{out} = (0.17*L_{in}) + 5.78$	$15 < L_{in} < 180$	$3 < L_{out} < 41$	n.g.	0.73
TSS[c]						
Reed and Brown [21]	14 US SSF	$C_{out} = C_{in}*(0.1058 + 0.0011*q)$	$22 < C_{in} < 118$	$3 < C_{out} < 23$	n.g.	n.g.
Knight et al. [18]	Soil-based SSF (NADB)	$C_{out} = (0.09*C_{in}) + 4.7$	$0 < C_{in} < 330$	$0 < C_{out} < 60$	$0.8 < q < 22$	0.67
Knight et al. [18]	SSF (NADB)	$C_{out} = (0.063*C_{in}) + 7.8$	$0.1 < C_{in} < 253$	$0.1 < C_{out} < 160$	$1.9 < q < 44.2$	0.09
Vymazal [20]	SSF in Czech Republic	$C_{out} = (0.021*C_{in}) + 9.17$	$13 < C_{in} < 179$	$1.7 < C_{out} < 30$	$0.6 < q < 14.2$	0.02
Kadlec et al. [1]	NADB, Severn Trent	$C_{out} = 0.76*C_{in}^{0.706}$	$8 < C_{in} < 595$	$2 < C_{out} < 58$	n.g.	0.55
Brix [17]	Danish soil-based SSF	$C_{out} = (0.09*C_{in}) + 4.7$	$0 < C_{in} < 330$	$0 < C_{out} < 60$	n.g.	0.67
Vymazal [22]	SSF in Czech Republic	$L_{out} = (0.048*L_{in}) + 1.76$	$3 < L_{in} < 78$	$0.9 < L_{out} < 6.3$	n.g.	0.42
Vymazal [20]	SSF in Czech Republic	$L_{out} = (0.083*L_{in}) + 1.18$	$3.7 < L_{in} < 123$	$0.45 < L_{out} < 15.4$	$0.6 < q < 14.2$	0.64
TN[d]						
Kadlec and Knight [13]	NADB + others	$C_{out} = 2.6 + (0.46*C_{in}) + (0.124*q)$	$5.1 < C_{in} < 58.6$	$2.3 < C_{out} < 37.5$	$0.7 < q < 48.5$	0.45
Kadlec et al. [1]	Danish soil-based SSF	$C_{out} = (0.52*C_{in}) + 3.1$	$4 < C_{in} < 142$	$5 < C_{out} < 69$	$0.8 < q < 22$	0.63
Vymazal [20]	SSF in Czech Republic	$C_{out} = (0.42*C_{in}) + 7.68$	$16.4 < C_{in} < 93$	$10.7 < C_{out} < 49$	$1.7 < q < 14.2$	0.72
Vymazal [22]	SSF in Czech Republic	$L_{out} = (0.67*L_{in}) - 18.75$	$300 < L_{in} < 2400$	$200 < L_{out} < 1550$	n.g.	0.96
Vymazal [20]	SSF in Czech Republic	$L_{out} = (0.68*L_{in}) + 0.27$	$145 < L_{in} < 1894$	$134 < L_{out} < 1330$	$1.7 < q < 14.2$	0.96
TP[e]						
Kadlec and Knight [13]	US, European and Australian SSF	$C_{out} = 0.51*C_{in}^{1.1}$	$0.5 < C_{in} < 20$	$0.1 < C_{out} < 15$	n.g.	0.64
Kadlec and Knight [13]	US SSF	$C_{out} = 0.23*(q^{0.6}*C_{in}^{0.76})$	$2.3 < C_{in} < 7.3$	$0.1 < C_{out} < 6$	$2.2 < q < 44$	0.60
Brix [17]	Danish soil-based SSF	$C_{out} = (0.65*C_{in}) + 0.71$	$0.5 < C_{in} < 19$	$0.1 < Cout < 14$	$0.8 < q < 22$	0.75
Vymazal [20]	SSF in Czech Republic	$C_{out} = (0.26*C_{in}) + 1.52$	$0.77 < C_{in} < 14.3$	$0.4 < C_{out} < 8.4$	$1.7 < q < 14.2$	0.23
Vymazal [22]	SSF in Czech Republic	$L_{out} = (0.58*L_{in}) - 4.09$	$25 < L_{in} < 320$	$20 < L_{out} < 200$	n.g.	0.61
Vymazal [20]	SSF in Czech Republic	$L_{out} = (0.67*L_{in}) - 9.03$	$28 < L_{in} < 307$	$11.4 < L_{out} < 175$	$1.7 < q < 14.2$	0.58

n.g.: not given.
NADB: North American Treatment Database [18].
[a] C_{in} and C_{out}: influent and effluent concentrations ($\mathrm{mg\,BOD\,L^{-1}}$); L_{in} and L_{out}: influent and effluent loads ($\mathrm{kg\,BOD\,ha^{-1}\,day^{-1}}$); $L_{removed}$: load removed ($\mathrm{kg\,BOD\,ha^{-1}\,day^{-1}}$).
[b] L_{in} and L_{out}: influent and effluent loads ($\mathrm{kg\,COD\,ha^{-1}\,day^{-1}}$).
[c] C_{in} and C_{out}: influent and effluent concentrations ($\mathrm{mg\,TSS\,L^{-1}}$); L_{in} and L_{out}: influent and effluent loads ($\mathrm{kg\,TSS\,ha^{-1}\,day^{-1}}$).
[d] C_{in} and C_{out}: influent and effluent concentrations ($\mathrm{mg\,TN\,L^{-1}}$); L_{in} and L_{out}: influent and effluent loads ($\mathrm{g\,N\,m^{-2}\,year^{-1}}$).
[e] C_{in} and C_{out}: influent and effluent concentrations ($\mathrm{mg\,TP\,L^{-1}}$); L_{in} and L_{out}: influent and effluent loads ($\mathrm{g\,P\,m^{-2}\,year^{-1}}$).

preferred for calibration purposes [24]. Because these parameters lump a large number of other characteristics representing the complex web of interactions in a constructed treatment wetland as well as external influences like weather conditions, a large variability can be observed in reported k_A, k_V, C^* and θ values. Table 3 presents an overview of first-order rate constants for horizontal SSF constructed treatment wetlands. Looking for instance at BOD removal, the reported k_A values vary between 0.06 and 1.00 m day^{-1} whereas k_v values range from 0.17 to 6.11 day^{-1}. For a given BOD influent concentration and effluent limit, the predicted maximum loading rate based on k_A values thus varies by a factor of 36. Kadlec and Knight [13] therefore recommend using 'global' average rate constants between these extremes.

Next to this variability, some other major drawbacks of the first-order models need to be mentioned. First of all, the equations are based on the assumptions of plug-flow and steady-state conditions. However, small scale wastewater treatment plants under which most treatment wetlands can be ranged are subject to large influent variations [33] whereas the larger ones are subject to hydrological influences [8,15], thus causing in both cases

Table 3
First-order rate constants for horizontal SSF constructed treatment wetlands according to different authors

Reference	No. of beds	k_A (m day^{-1})	k_v (day^{-1})	Remarks
BOD				
Crites [25]			0.8–1.1	0.8 = sand; 1.1 = gravel (T°C)
Reed and Brown [21]			1.104	K_{20} with $\theta = 1.06$
Tanner et al. [4]	8		0.17	k_T—gravel beds
Tanner et al. [4]	8		0.22	K_{20} with $\theta = 1.06$—gravel beds
Wood [16]			1.84	$\varepsilon = 0.42$—medium sand (20°C)
Wood [16]			1.35	$\varepsilon = 0.39$—course sand (20°C)
Wood [16]			0.86	$\varepsilon = 0.35$—medium sand (20°C)
Kadlec and Knight [13]		0.085–1	0.3–6.11	
Kadlec [15]		0.49		$C^* > 3$ mg L^{-1} and $\theta = 1.00$ (20°C)
Vymazal et al. [26]		0.19		Proposed by Kickuth
Brix [27]		0.118 ± 0.022		Mean ± 95% limits—depends on load
Schierup et al. [28]	49	0.083		Danish systems
Cooper [29]		0.067–0.1		UK systems
Brix [17]	70	0.16		$C^* = 3.0$ mg L^{-1}—soil based
Brix [17]	70	0.068		$C^* = 0$ mg L^{-1}—soil based
Kadlec et al. [1]		0.133		Czech republic wetlands
Kadlec et al. [1]	1	0.07–0.097–0.13–0.18–0.31–0.17		6 consecutive years, Czech republic wetlands
Cooper et al. [30]		0.06		$C^* = 0$ mg L^{-1}—secondary wetlands
Cooper et al. [30]		0.31		$C^* = 0$ mg L^{-1}—tertiary wetlands
Kadlec et al. [1]	14	0.17		$C^* = 0$ mg L^{-1}—tertiary wetlands USA
Liu et al. [31]			0.86	Gravel beds—soluble cBOD, 20°C
TSS				
Kadlec and Knight [13]		2.74		k_{20} with $\theta = 1$ and $C^* > 7$ mg L^{-1}
Kadlec [15]		8.22		k_{20} with $\theta = 1$ and $C^* > 7$ mg L^{-1}
Kadlec et al. [1]		23.1		Laboratory columns
Kadlec et al. [1]		31.6		Large scale pilot wetland
Kadlec et al. [1]	33	0.119		Data from Czech republic
TN				
Tanner et al. [5]			0.16	k_T—gravel bed
Kadlec and Knight [13]		0.074		k_{20} with $\theta = 1.05$ and $C^* = 1.5$ mg L^{-1}
Kadlec and Knight [13]		0.007–0.1		k_T with $C^* = 1.5$ mg L^{-1}
Wittgren and Maehlum [32]	73		0.06	k_T—Norway
Kadlec et al. [1]		0.028		Czech systems
TP				
Tanner et al. [5]			0.14	k_T—gravel bed
Kadlec and Knight [13]		0.033		k_{20} with $\theta = 1.00$ and $C^* = 0.02$ mg L^{-1}
Wittgren and Maehlum [32]	71		0.28	k_T—Norway

non steady-state conditions. Short-circuiting and dead zones are common phenomena in constructed treatment wetlands causing non-ideal plug-flow conditions, thus jeopardising the use of the first-order model [24]. Secondly, the so-called rate 'constants' do not seem to be constant at all but dependent on the influent concentrations, the HLR and the water depth [15,24]. Table 3 also shows some influence of the void fraction, the maturity of the bed and the chosen background concentration on the rate constants.

2.4. Variable-order or Monod-type models

Mitchell and McNevin [34] identified another physical impossibility of the first-order model, namely the fact that the removal rates continue to increase with increasing loading rates (Eq. (5)):

$$r = Q(C_{\text{in}} - C_{\text{out}}) \Rightarrow r = QC_{\text{in}}(1 - \exp(-k_V\tau)). \tag{5}$$

However, in most cases, a maximum allowable loading rate has been demonstrated. Therefore, Mitchell and McNevin [34] advocate the use of a Monod-type design model, which represents first-order rate reactions for relatively low concentrations but zero-order rate reactions for high concentrations. Still with the assumption of plug flow, the model presents itself as (Eq. (6))

$$r = k_{0,\text{V}}V\frac{C}{K+C} \quad \text{and} \quad \frac{\mathrm{d}C}{\mathrm{d}t} = \frac{-r}{V}$$
$$\xrightarrow{[1],[2],[3],[4]} \frac{\mathrm{d}C}{\mathrm{d}z} = -\frac{k_{0,\text{V}}\varepsilon a}{Q}\frac{C}{K+C} = -\frac{k_{0,\text{A}}}{qZ}\frac{C}{K+C}, \tag{6}$$

transformation equations:

[1] $k_{0,\text{A}} = k_{0,\text{v}}\varepsilon d$;

[2] $q = Q/A = Q/(W*Z)$;

[3] $z = v*t$;

[4] $v = Q/(\varepsilon a)$.

One other interesting feature of this model is an alternative explanation of background concentrations (C^*). Indeed, if concentrations drop to near zero, the Monod equation predicts a very low reaction rate, which may prevent total decomposition of the pollutant within the given hydraulic retention time.

The authors did not try to assess parameter values, but used a graphical representation of loading and removal rates from the North-American treatment database [18] to extract some design parameters. They found a maximum allowable loading rate for horizontal SSF constructed treatment wetlands of 80 kg BOD $\text{ha}^{-1}\,\text{day}^{-1}$ and 130 kg TSS $\text{ha}^{-1}\,\text{day}^{-1}$. Data from a.o. several Danish [35] and UK systems [36] show most actual loading rates well below these maximum recommended levels. Several exceptions are however mentioned where, despite significantly higher loading rates, effluent concentrations are still of acceptable quality.

Kemp and George [37] used a comparable model to represent ammonia removal in a pilot-scale horizontal SSF constructed wetland treating domestic wastewater. They found a $k_{0,\text{V}}$ of 7.8 mg N $\text{L}^{-1}\,\text{day}^{-1}$ and a K of 5.5 mg N L^{-1}. The coefficient of determination R^2 indicated that the Monod-type model better described the variability of the data than a first-order model.

2.5. Mechanistic, compartmental models

Only recently, a mechanistic, compartmental simulation model of a horizontal SSF constructed treatment wetland has been presented by Wynn and Liehr [38]. The model consists of six interlinked submodels, representing the carbon and nitrogen cycles, the water and oxygen balances, and the growth, decay and metabolism of heterotrophic and autotrophic bacteria. Removal of phosphorus and suspended solids is not modelled since they mainly depend on physical and not on biological processes. Hydraulic behaviour is modelled via a tanks-in-series approach to mimic the mixing regime, and via the Darcy equation to imitate flow in a porous medium.

The required model inputs are air temperature, day length, precipitation, flow rate and the concentrations of BOD, NH_4-N, NO_3-N, organic N and dissolved oxygen. The model output consists of flow rate and the same five concentrations as for the input. The dynamics of the 15 state variables are modelled via 15 ordinary differential equations that contain a total of 42 parameters related to physical, microbiological and biological processes. On the one hand, this complexity of the model enables to better summarise the processes that occur within constructed wetlands as well as to demonstrate interactions between certain components. On the other hand, it requires estimation of 15 initial conditions for the state variables and knowledge about or estimation of 42 parameters, which is not an obvious task.

The authors calibrated the model only approximately by adjusting the parameters to a certain extent to make the model output corresponding with the available site data. This procedure however yielded values for several microbial parameters that were one or more orders of magnitude lower than those typically mentioned in the literature. Due to the complexity of the model, it is very well possible that certain parameters compensate for each other, thus causing insensitivity to parameter changes (see e.g. [39]). However, it would be more reasonable to assume that certain important phenomena were not included in the model, although they are influencing microbial reactions. As an example, diffusion limitations in the biofilm can be mentioned.

3. Case study

To demonstrate the use of the above models and to illustrate the variability and uncertainty of the predictions, a case study was performed. The different design models were used to calculate the required surface area of a horizontal subsurface constructed treatment wetland, able to produce an effluent in compliance with the legal standards. Real influent data were used, collected at a pilot-scale constructed reed bed (10 PE) belonging to Aquafin NV and located in Aartselaar, Belgium. For a detailed description, one is referred to Vandaele et al. [40] and Rousseau et al. [41,42]. Table 4 gives an overview of the influent characteristics and the applied effluent standards, based on the Flemish Environmental Legislation [43]. The low influent concentrations are due to the combined effect of a mixed sewer system and a primary treatment phase.

Whenever possible, the minimum and maximum values of reported parameter values (Tables 1–3) were applied to show the maximal variability of the areal prediction. Regression equations and area-based first-order models allow to calculate the HLR, q, from which the required area $A(A = Q/q)$ can be derived. Volume-based first-order models allow to calculate the hydraulic retention time τ and consequently the required volume $V(V = Q*\tau)$. An assumed water depth of 0.6 m and a pore volume of 40% was used to transform water volume into surface area ($A = V/(d*\varepsilon)$). For the purpose of this case study, the simplest first-order model was used, i.e. without background concentrations and temperature coefficients, since many researchers do not mention values for those parameters (Table 3). The Monod-type model of Mitchell and McNevin [34] could not be tested because of a lack of parameter data.

Table 4
Influent characteristics and effluent standards used in the case study

Variable	Average influent characteristics	Effluent standards
Flow rate ($m^3\,day^{-1}$)	1.9	
BOD ($mg\,BOD\,L^{-1}$)	48.0	25.0
COD ($mg\,COD\,L^{-1}$)	184.7	125.0
TSS ($mg\,TSS\,L^{-1}$)	71.0	35.0
TN ($mg\,N\,L^{-1}$)	17.2	15.0
TP ($mg\,P\,L^{-1}$)	2.8	2.0

Results of the rules of thumb, the regression equations, the first-order model and the time-dependent retardation model are presented in Fig. 1. These different, simple design methods predict required surface areas ranging from as low as 0.1 m^2 up to 950 m^2 for the given influent data and effluent standards. Generally speaking, the rules of thumb seem to be the more conservative ones as they consistently predict larger surface areas. This observation raises the question whether or not a scientifically more sound but also more complex, compartmental model like the one of Wynn and Liehr [38] could be useful for design purposes and if the uncertainty could be reduced.

The model of Wynn and Liehr [38] was first implemented in the wastewater treatment plant simulator WEST™ (Hemmis NV, Kortrijk, Belgium [44]. One major adjustment was made to the water balance: the effluent flow rate was allowed to drop to zero if water loss by evapotranspiration exceeded the water supply as influent and precipitation. The aerobic heterotrophic growth equation was also corrected as follows: the anaerobic fraction of heterotrophs was changed to its complement (1—anaerobic fraction of heterotrophs) in the differential equation of aerobic heterotrophic growth.

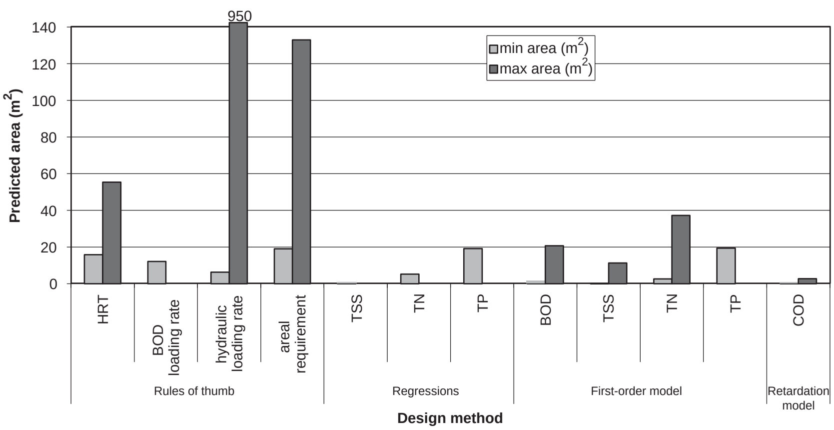

Fig. 1. Required area predictions according to the different design methods used in the case study. Minimum and maximum areas indicate the output variability due to parameter uncertainty.

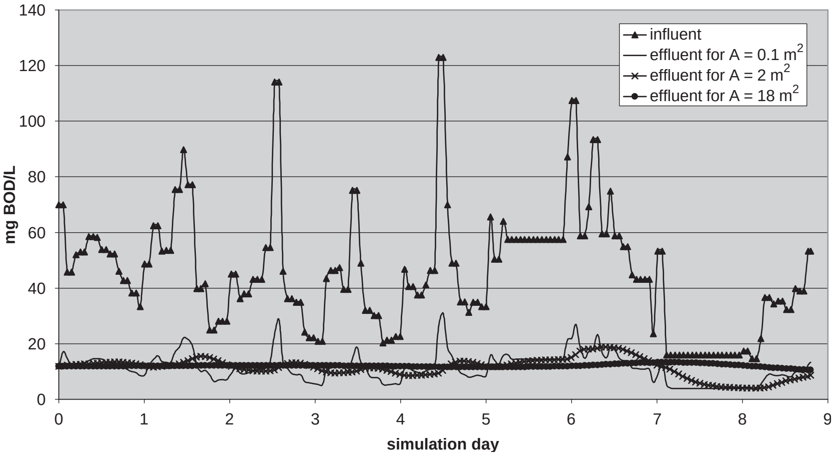

Fig. 2. Predicted BOD removal with the model of Wynn and Liehr [38] for different wetland bed areas.

To imitate an intermediate flow regime between completely mixed and plug flow, two continuously stirred tank reactors (CSTRs) were placed in series. In accordance with the previous calculations, the bed depth was set at 60 cm and the porosity at 40%. Parameter values were copied from the original paper of Wynn and Liehr [38] and missing values were estimated based on literature data and some preliminary simulations [45]. Initial conditions were set to the end values of a steady-state simulation with average influent flow, concentrations (o.a. Table 4) and meteo conditions (average air temperature 5.2°C). Finally, dynamic simulations were carried out for a bed area of 0.1, 2 and 18 m^2.

As an example, results for BOD removal are shown in Fig. 2. Quite remarkably, even with a bed area of only 0.1 m^2, the predicted effluent concentrations stay below 25 mg BOD L^{-1} for most of the time. This is due to the fact that the model assumes that the suspended solids removal is 100%, thus causing immediate removal of all particulate BOD, even in a wetland bed of this very small size. The results for a larger bed area of 18 m^2 show very little variations in the predicted BOD effluent concentrations. This is not surprising, since a hydraulic residence time of 2.57 days is more than sufficient to cause a buffering influence on the remaining dissolved BOD.

4. Conclusions and recommendations

Confronted with different models of horizontal SSF constructed treatment wetlands and the numerous different parameter values, the obvious question is which ones should be used and which one is the most reliable one? The case study clearly demonstrated that the predicted required surface areas are highly variable and that this variability does not only exist among the different models, but due to parameter uncertainty also within the same model category.

The rules of thumb seemed to be the more conservative design models. Since these are easily applicable, designers could be tempted to stick to those models. However, they may be guaranteeing good quality effluent, but they will likely be counteracted by economic constraints: conservative designs tend to increase the investment costs.

The mechanistic model of Wynn and Liehr [38] on the other hand did not offer real help for design purposes due to some assumptions and empirical relations that are not physically based and thus corrupt model output. The immediate and complete removal of all particulate substances is obviously the most important one. However, this model is a useful tool to gain understanding of certain processes and it is well able to demonstrate several interactions within the wetland system. The model should be considered as an important framework for future model development.

At present, the state-of-the-art $k-C^*$ model seems to be the best available design tool if the designer makes sure that all the assumptions are fulfilled and if he is aware of the pitfalls in the model. Concerning the issue of parameter uncertainty, it is advisable to implicitly take this into account during the design. If possible, parameter values should be used from constructed treatment wetlands that operate under similar conditions as the one to be constructed: climatic conditions, wastewater composition, bed material and macrophyte species.

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