

# Modelling of organic matter degradation in constructed wetlands for treatment of combined sewer overflow

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## Abstract

Subsurface vertical flow constructed wetlands (CWs) have been found to be a useful system to treat combined sewer overflow (CSO). The study presented uses numerical simulation to increase the understanding of the fundamental processes of COD degradation in CWs for CSO treatment. The multi-component reactive transport module CW2D was used for the simulation study. The simulation results showed that the measured behaviour of the system can only be modelled when COD adsorption is considered as additional process. A new parameter set for CW2D for modelling CSO treatment is presented. A range of values for COD adsorption parameters, COD fractionation and bacteria concentrations were estimated by an identifiability analysis. For the simulation a step wise approach was developed. On the one hand a lysimeter study was used for calibration and validation, and on the other hand field and lab-scale experiments were used for validation. Single-event simulations as well as long-term simulations were carried out. For the single-event simulations (lysimeter and field studies) a good match between measured and simulated data could be achieved. However, the long-term simulations showed that there is a need for further investigations mainly due to the uncertainties during long dry periods between the loadings.

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## 1. Introduction

Subsurface vertical flow constructed wetlands (CWs) have been found to be a useful system to treat combined sewer overflow (CSO) for decreasing the impact on receiving waters. Main objectives for CSO treatment are (1) detention and reduction of peak flows,

(2) reduction of suspended solids by filtration and (3) reduction of soluble and particulate pollutants by adsorption and subsequent biological degradation (e.g. Uhl and Dittmer, 2005).

Storm water tanks (STW) and CWs are cascaded treatment units. To prevent clogging of the filter surface due to excessive loads of suspended solids (SS), pre-treatment by sedimentation in a conventional STW is essential (DWA-M 178, 2005). Significant differences between CWs for CSO treatment and CWs for wastewater treatment exist because of the stochastic nature of rain and therefore CSO (Uhl and Dittmer,

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2005). The stochastic nature of rainfall leads on the one hand to very long inundation times and on the other hand to very long drought periods.

To increase the treatment efficiency CWs for CSO are throttled to a constant outflow rate of  $1\text{--}3 \cdot 10^{-5} \text{ m}^3 \text{ s}^{-1} \text{ m}^{-2}$ . The throttled outflow leads to a free water surface during a storm event. A huge difference of CWs for wastewater treatment is the functioning as a tank during a storm event. Here only adsorption and filtration of soluble and particulate pollutants take place. Only after inundation and re-aeration filtrated or adsorbed organic matter and ammonia are degraded.

CWs for CSO treatment are regarded as Black-Box-systems. Research on efficiency is often done by input–output analysis (LFU, 2002; Uhl and Jübner, 2004; Fuchs and Schnabel, 2001). It is generally agreed that soluble and particulate pollutants are adsorbed or filtrated during an event and degraded during dry periods. Recent work has carried out detailed investigations on internal processes in CWs for CSO treatment (Grobe et al., 2005; Schmitt et al., 2005; Wozniak et al., 2006).

The study presented has been carried out to increase the understanding of the fundamental processes of COD elimination by adsorption, filtration and degradation in CWs for CSO treatment. The CW2D module for the HYDRUS-2D software package (Langergraber and Šimunek, 2005) was used for the simulations.

## 2. Methods

### 2.1. Experiments

For the simulation studies experiments from 3 different scales were chosen: lysimeter, field and lab-scale experiments. The different dimensions have different uncertainties in the boundary conditions from defined lab-scale conditions with batch feedings to uncertain measurement influenced by the stochastic nature of rain in the field-scale.

The measured data used are taken from completed research projects. The lab-scale (Uhl and Jübner, 2004) and lysimeter experiments (Grobe et al., 2005) were done at the University of Applied Science Münster; the field measurements were realized by Fuchs and Schnabel (2001).

#### 2.1.1. Lysimeter experiments

The first part of the experiments has been carried out with lysimeters. The lysimeters have a diameter of 1.1 m and a 0.75 m sand layer (0–2 mm) above a drainage layer of 0.2 m fine and coarse gravel (2–8 mm and 8–16 mm).

Peristaltic pumps were used to control each outflow with constant rates of  $1 \cdot 10^{-5} \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{m}^{-2}$  (i.e.  $0.0095 \text{ L} \cdot \text{s}^{-1}$ ).

Five lysimeters have been operated in two phases over a period of 2.5 years. In phase 1 the lysimeters were loaded by artificial combined sewage following a fixed schedule of loadings representing different boundary conditions. In phase 2 the lysimeters were loaded by combined sewage overflow from a storm water tank following the natural occurrence of overflow events. The effluent concentrations were measured by taking 4-h-composite samples. Detailed investigations to analyse the elimination mechanism of COD were done by sampling the interstitial water by suction cups during one event in 4 different depths (5, 15, 25, 45 cm, respectively) as well as the effluent of the lysimeters. The measured concentrations showed a stepwise saturation of the adsorption depot from the surface of the sand layer to the surface of the drainage layer. It was not possible to use the measured outflow concentrations of the interstitial water for calibrating the model because of difficulties in sampling. The measurement must be seen as qualitative.

#### 2.1.2. Field experiments

A data set measured at the constructed wetland (retention soil filter) Langenalb in Baden-Württemberg was used for the simulation of fields experiments. The surface area of the filter is about  $439 \text{ m}^2$  for a channelized catchment of about  $496,000 \text{ m}^2$  with a paved area of about  $218,000 \text{ m}^2$ . The effluent was throttled to  $1 \cdot 10^{-5} \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{m}^{-2}$  (i.e.  $3 \text{ L} \cdot \text{s}^{-1}$ ). The height of the sandy main layer of the filter is 0.9 m and consists of sand with a grain size from 0–2 mm. The effluent is collected in a 0.25 m drainage layer of gravel 2–8 mm. The filter is planted with reed (*Phragmites communis*). Langergraber (2005) showed that the influence on nitrogen and phosphorous elimination by root water uptake is marginal, so it is not considered in this simulation study. The samples were taken as 2-h composite samples for a period of 24 h (Fuchs and Schnabel, 2001).

#### 2.1.3. Lab-scale experiments

The third type of experiments used in this study has been measured at lab-scale. A column of 2 m length and 0.2 m diameter was filled by a filter of 0.1 m coarse gravel (8–16 mm) at the bottom followed by 0.1 m fine gravel (2–8 mm) and a 0.8 m sand layer (0–2 mm) above. As for the other experiments the outflow was throttled to  $1 \cdot 10^{-5} \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{m}^{-2}$  (i.e.  $0.00028 \text{ L} \cdot \text{s}^{-1}$ ). Tracer experiments for similar columns showed plug flow with dispersion without preferential flow in a lab-scale column (Dittmer et al., 2005). The sieve curve is

homogenous so that a constant filter velocity and also a constant residence time in the filter can be assumed.

The experiments were carried out as batch feedings with a hydraulic load of  $1 \text{ m}^3 \cdot \text{m}^{-2}$  of artificial combined sewage. To reach CSO typical concentration 0.5 h settled wastewater was diluted with drinking water. After the first 6 loadings for inoculation the sand filters outflow of the next 10 loadings was observed. The total hydraulic load was  $16 \text{ m}^3 \cdot \text{m}^{-2}$  in 56 days. The samples were taken as one composite sample per feeding (Uhl and Jübner, 2004).

## 2.2. The multi-component reactive transport module CW2D

The multi-component reactive transport module CW2D (Langergraber, 2001) was applied to model the behaviour of the experiments. CW2D was developed to model transport and reactions of the main constituents of municipal wastewater in subsurface flow constructed wetlands and is able to describe the biochemical elimination and the transformation processes for organic matter, nitrogen and phosphorus. Organic matter is modelled to consist of three fractions: Readily degradable (CR), slowly degradable (CS) and inert (CI). CW2D is incorporated into the HYDRUS-2D variably-saturated water flow and solute transport program (Langergraber and Šimunek, 2005). The unsaturated hydraulic properties are modelled using the soil hydraulic functions of van Genuchten (1980):

$$\Theta(h) = \begin{cases} \Theta_r + \frac{\Theta_s - \Theta_r}{[1 + |\alpha \cdot h|^n]^m} & h < 0 \\ \Theta_s & h \geq 0 \end{cases} \quad (1)$$

$$S_e = \frac{\Theta - \Theta_r}{\Theta_s - \Theta_r} \quad (2)$$

$$K(h) = K_S \cdot S_e^{\ell} \cdot [1 - (1 - S_e^{1/m})^m]^2 \quad (3)$$

where  $\theta$  = volumetric water content [ $\text{m}_w^3 \cdot \text{m}_s^{-3}$ ];  $h$  = pressure potential [ $\text{m}$ ];  $\theta_r$  and  $\theta_s$  = residual and saturated water content, respectively [ $\text{m}_w^3 \cdot \text{m}_s^{-3}$ ];  $m = 1 - 1/n$ ;  $n > 1$ ;  $\alpha$  [ $\text{m}_s^{-1}$ ] and  $n$  [-] = empirical coefficients;  $S_e$  = effective water content [-];  $K(h)$  = unsaturated hydraulic conductivity [ $\text{m}_w^3 \cdot \text{m}_s^{-2} \cdot \text{s}^{-1}$ ];  $K_S$  = saturated hydraulic conductivity [ $\text{m}_w^3 \cdot \text{m}_s^{-2} \cdot \text{s}^{-1}$ ];  $\ell$  [-] = pore-connectivity parameter.

CW2D considers two types of bacteria: heterotrophic and autotrophic bacteria. Heterotrophic bacteria (XH)

are modelled as all-rounder bacteria responsible for hydrolysis, aerobic degradation of organic matter and denitrification. Autotrophic bacteria (2 species, XANs and XANb) are responsible for the two-step nitrification process (Langergraber, 2001).

CW2D had to be modified to model a throttled effluent (constant filter velocity) and impoundage for considering storage volume (Dittmer et al., 2005). For simulation of constructed wetlands for CSO treatment the standard CW2D parameters were used except of composition parameters for COD. In CW2D organic nitrogen and organic phosphorus are modelled as nitrogen and phosphorus content of the organic matter. For the simulations of CWs for CSO treatment a higher organic nitrogen content has been used (Dittmer et al., 2005). In addition adsorption for COD was introduced. The standard CW2D parameter set (Langergraber, 2001) considers adsorption only for ammonia nitrogen and inorganic phosphorus.

According to Simunek et al. (1999) adsorption in HYDRUS-2D is described by

$$s_i = \frac{k_{s,i} \cdot c_i^{\beta_i}}{1 + \eta_i \cdot c_i^{\beta_i}} \quad (4)$$

where  $i = 1..N$ ;  $N$  = total number of components;  $c_i$  = concentration in the aqueous phase [ $\text{mg}_i \cdot \text{dm}_w^{-3}$ ];  $s_i$  = concentration in the solid phase [ $\text{mg}_i \cdot \text{kg}_s^{-1}$ ]; and  $k_{s,i}$ ,  $\beta_i$ ,  $\eta_i$  = empirical coefficients. With  $\beta_i = 1$  Eq. (4) becomes the Langmuir equation, when  $\eta_i = 0$  it becomes the Freundlich equation, and when both  $\beta_i = 1$  and  $\eta_i = 0$  Eq. (4) is a linear adsorption isotherm. Adsorption of COD is modelled by a chemical non-equilibrium adsorption of CS (slowly biodegradable COD) with a linear adsorption isotherm (i.e. in Eq. (4) the parameter  $\beta = 1$  and  $\eta = 0$ ).

The time dependency of adsorption is described by the concept of two-site sorption. Sorption is instantaneous on one part of the exchange sites whereas on the remaining sites it is considered to be time-dependent. The two-site sorption is described by (Simunek et al., 1999)

$$\frac{ds_i^k}{dt} = \omega_i \cdot (1-f) \cdot \left[ \frac{k_{s,i} \cdot c_i^{\beta_i}}{1 + \eta_i \cdot c_i^{\beta_i}} - s_i^k \right] \quad (5)$$

where  $s_i^k$  = adsorbed concentration of component  $i$  at the fraction of sites assumed to be time-dependent [ $\text{mg}_i \cdot \text{kg}_s^{-1}$ ];  $\omega_i$  = first-order exchange rate for component  $i$  [ $\text{h}^{-1}$ ]; and  $f$  = fraction of exchange sites assumed to be in equilibrium with the solute phase [-]. It is assumed that all adsorption sites are time-dependent, i.e.  $f = 0$ .

### 2.3. Simulation approach

A step wise approach was followed for calibration and validation:

1. calibration for a single-event measured from the lysimeter experiments
  - a. calibration of the flow and single-solute transport model using measured data from tracer experiments.
  - b. calibration of CW2D using measured effluent concentrations
2. validation of the calibrated parameter set for three other single-events from the lysimeter experiments.
3. validation of the parameter set for single-events gained from field experiments
4. long-term simulations using data from the lab-scale experiments

## 3. Results

### 3.1. Model set-up and boundary conditions

For the simulations a 2D finite element mesh with 99 rows and 3 columns was generated. The vertical discretization had to be finer because of mainly vertical adsorption and degradation processes. For the field study a section of 1 m width was modelled.

The supernatant water level (storage volume) was modelled by a virtual layer with a pore volume of 100% and a residual water content of 0%. On the top of the virtual layer an atmospheric boundary condition was applied. At the bottom of the drainage layer a new

throttled boundary condition that is based on a seepage face boundary with maximal outflow and minimal pressure head was used. For the simulations the maximal outflow was set to  $1.10^{-5} \text{ m.s}^{-1}$ , and the minimal pressure head was set to minus 10 dm.

### 3.2. Simulation results

The calibration of the transport model for simulations of the lysimeter studies was done by using data of a tracer experiment (Grobe et al., 2005). For the soil hydraulic parameters of the van Genuchten model (Eqs. (1)–(3)) the saturated hydraulic conductivity was set to  $10 \text{ dm.h}^{-1}$  and the empirical Parameters  $\alpha$  and  $n$  to 0.2 and 3, respectively. The residual water content varies between 0.05 (field and lab-scale study) and  $0.06 \text{ dm}^3 \cdot \text{dm}^{-3}$  (lysimeter study) and the saturated water content between 0.35 (field and lab-scale study) and  $0.45 \text{ dm}^3 \cdot \text{dm}^{-3}$  (lysimeter study).

#### 3.2.1. Simulation without COD adsorption

Fig. 1 shows measured and simulated effluent concentrations of COD, the COD fractions CR, CS and CI, dissolved oxygen  $\text{O}_2$  and the throttle outflow rate (qdr). The difference between measured and simulated COD effluent concentration is high. It was not possible to model COD degradation for CSO treatment with biochemical processes only. Therefore adsorption for COD was considered.

To model the coupled processes degradation, filtration and adsorption the fractionation of COD is used. For the readily degradable COD fraction (CR) biochemical degradation by respiration during percolation is

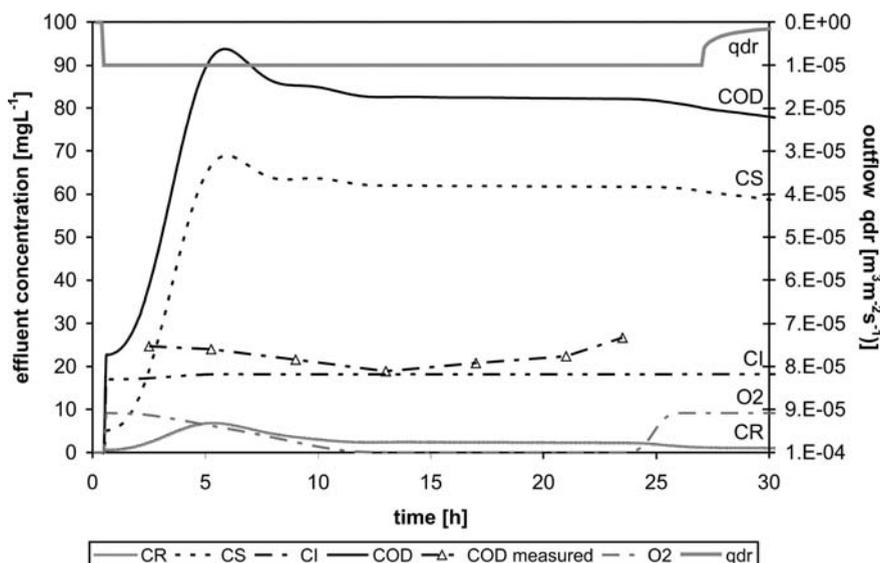


Fig. 1. Effluent concentration of COD and its fractions without CS adsorption and the throttle outflow rate (qdr).

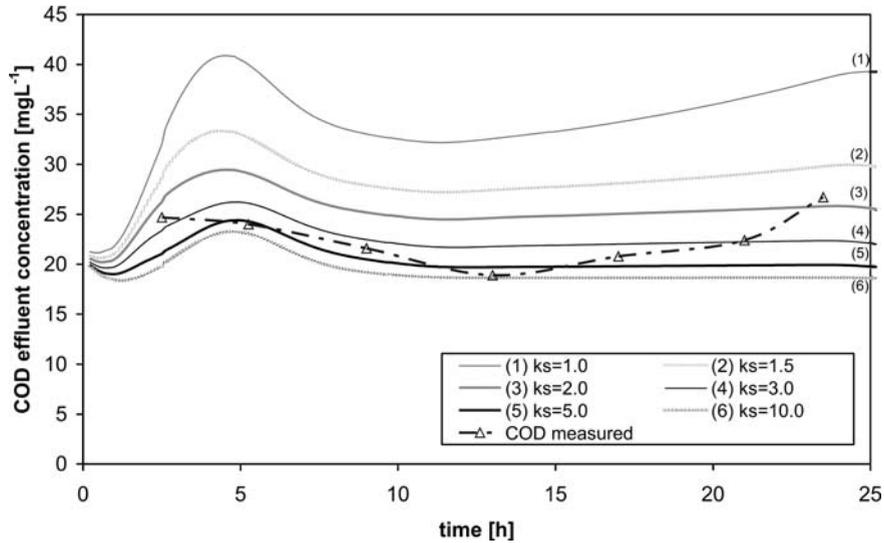


Fig. 2. Sensitivity analysis impact of CS adsorption on COD effluent concentration (different  $k_S$  values,  $\omega=0.15 \text{ h}^{-1}$ ).

assumed. CR represents the soluble degradable COD and CI the inert soluble COD. The slowly degradable soluble fraction, the particulate fraction and the biomass fraction are modelled as slowly degradable COD (CS). Slowly biodegradable COD is adsorbed during saturation of the filter and released during the dry periods and transformed into readily biodegradable COD by hydrolysis.

3.2.2. Identifiability analysis for the model parameters for CSO treatment

In literature no parameter values for COD adsorption could be found. Therefore the useful range of the values

for the parameters describing adsorption for CS has been identified by using a sensitivity analysis showing the influences of the single parameters on the simulation results.

For the sensitivity analysis the adsorption parameters  $k_S$  (Eq. (4)) and  $\varpi$  (Eq. (5)) have been varied. For each simulation only one parameter was altered while the other was kept constant. In Fig. 2 the results for varying  $k_S$  and in Fig. 3 the results of varying  $\varpi$  are shown. For the simulations the values for  $k_S$  have been varied from 1–10  $\text{L}\cdot\text{kg}^{-1}$  and for  $\varpi$  from 0.01–1.0  $\text{h}^{-1}$ . The average COD effluent concentration varies between

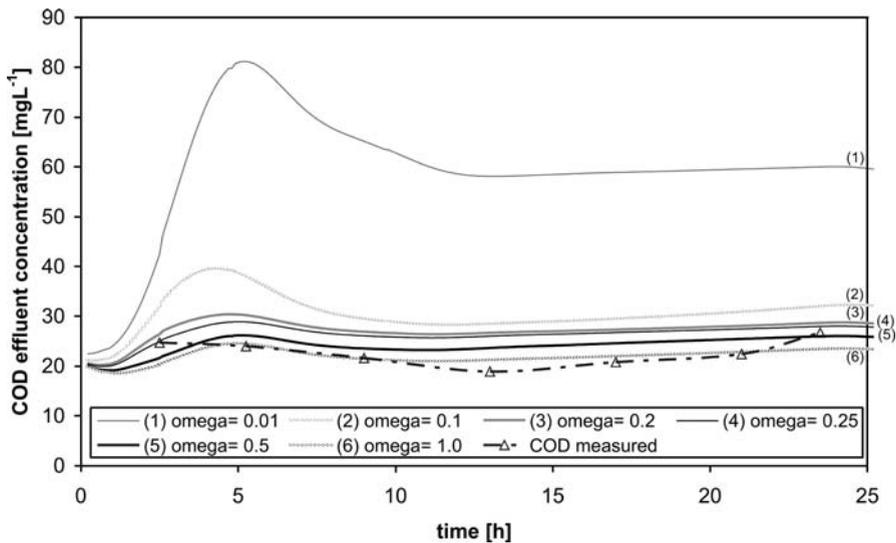


Fig. 3. Sensitivity analysis impact of CS adsorption on COD effluent concentration (different  $\omega$  values,  $k_S=1.5 \text{ L}\cdot\text{kg}^{-1}$ ).

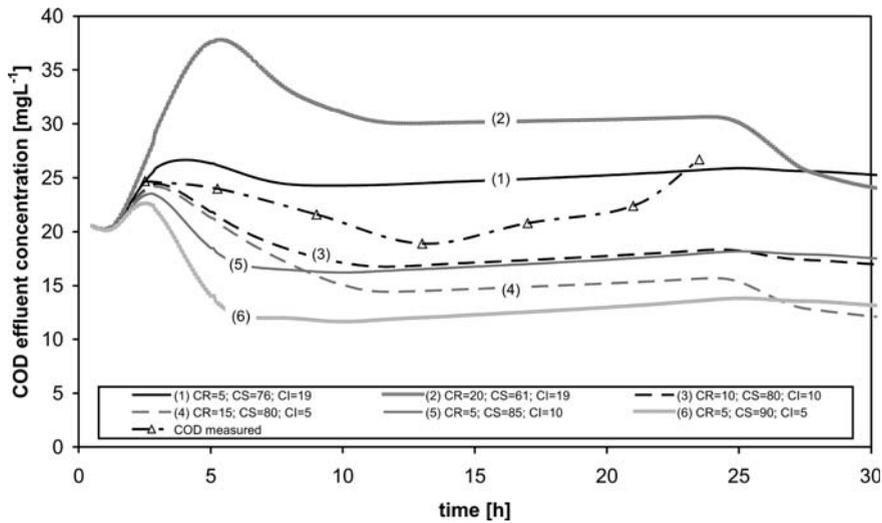


Fig. 4. Influence of the COD influent fractionation on the COD effluent concentration (total COD influent  $99 \text{ mg}\cdot\text{L}^{-1}$ ;  $k_S=1.5 \text{ L}\cdot\text{kg}^{-1}$ ;  $\omega=0.15 \text{ h}^{-1}$ ).

20–35  $\text{mg}\cdot\text{L}^{-1}$  ( $k_S$ ), and between 20–60  $\text{mg}\cdot\text{L}^{-1}$  ( $\omega$ ). For small values of  $k_S=1.0 \text{ L}\cdot\text{kg}^{-1}$  and  $\omega=0.01 \text{ h}^{-1}$  a peak in the COD effluent concentration can be observed (Figs. 2 and 3).

With increasing  $k_S$  or  $\omega$  CS and also COD effluent concentration decrease. The average CS effluent concentration is between 0–15  $\text{mg}\cdot\text{L}^{-1}$  ( $k_S$ ) and 2–35  $\text{mg}\cdot\text{L}^{-1}$  ( $\omega$ ). For values higher than  $5 \text{ L}\cdot\text{kg}^{-1}$  ( $k_S$ ) and higher than  $1 \text{ h}^{-1}$  ( $\omega$ ) the CS effluent concentration becomes very small. For the further simulations  $k_S$  values between 1 and  $3 \text{ L}\cdot\text{kg}^{-1}$  and  $\omega$  values between 0.1 and  $0.3 \text{ h}^{-1}$

were used. In Fig. 2 a saturation effect for COD adsorption (increasing COD effluent concentrations) can be observed if  $k_S$  values smaller than  $2.0 \text{ L}\cdot\text{kg}^{-1}$  are chosen.

The fractionation of the influent COD especially influences the processes respiration and hydrolysis. Respiration is inhibited by a lack of oxygen. Whereas the CI concentration is not degraded in the filter. In dry periods CI is produced by lysis processes of heterotrophic and autotrophic bacteria. At the beginning of a new loading CI is leached out of the soil column causing a peak in the COD effluent concentration.

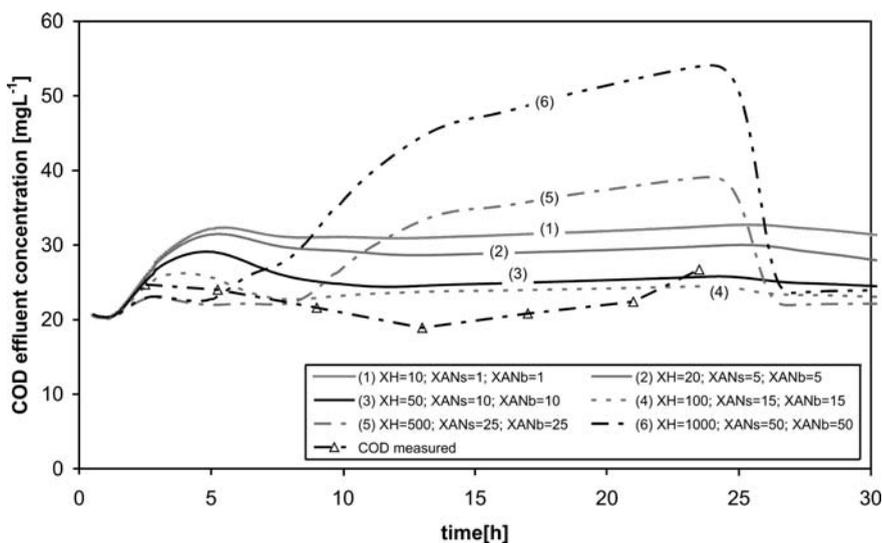


Fig. 5. Influence of the bacteria concentration on the COD effluent concentration (total COD influent  $99 \text{ mg}\cdot\text{L}^{-1}$ ; fractionation CR=10%, CS=71%, CI=19%;  $k_S=1.5 \text{ L}\cdot\text{kg}^{-1}$ ;  $\omega=0.15 \text{ h}^{-1}$ ).

For the sensitivity analysis of the COD fractionation 6 simulation runs have been carried out (Fig. 4). The percentages of the influent COD used for CR were between 5–20%, for CS between 60–90%, and for CI between 5–19%, respectively. The CI influent concentration was considered to be between 80 and 90% of the measured COD effluent concentration. The COD effluent concentration of the composite samples varies between 13–30 mg·L<sup>-1</sup> for the 6 simulation runs. Increasing CI as well as increasing CR causes higher COD effluent concentrations. Higher CR fractions also cause higher NH<sub>4</sub>-N effluent concentrations because the CR and NH<sub>4</sub>-N degradation processes compete for the oxygen concentration.

Measured data for bacteria concentrations in retention soil filters are very rare (Schwarz, 2004; Grobe et al., 2005). It is generally agreed that bacteria concentrations decline with the depth of the filter. For identifiable analysis bacteria concentration are varied in a wide range.

Fig. 5 shows the impacts of the varying bacteria concentrations on the COD effluent concentration. For the heterotrophic bacteria (XH) a range of values between 10–1000 mg COD·kg<sup>-1</sup> are chosen. Reasonable results are achieved in a range between 50–100 mg COD·kg<sup>-1</sup>. For the initial condition a linear decrease of bacteria concentration with filter depths was assumed. The values given represent bacteria concentration at the surface of the filters. For the bottom of the filter layer half of the bacteria concentration of the surface was assumed (cp. Schwarz, 2004).

The effluent concentration of CI and CS are not influenced by varying bacteria concentrations. This is because CI is not degraded and CS is retarded by adsorption. With heterotrophic concentrations of 500 or 1000 mg COD·kg<sup>-1</sup> it is not possible to get reliable simulation results. On the one hand the lack of oxygen in deeper regions is responsible for the higher COD effluent concentrations because CS is hydrolysed and not adsorbed near the surface area but the oxygen is consumed by respiration and nitrification processes. Because of the anoxic conditions in the soil column denitrification of NO<sub>3</sub>-N takes place.

### 3.2.3. Simulation with COD adsorption

During the calibration process for the adsorption parameters values of 2.0 kg·L<sup>-1</sup> ( $k_s$ ) and 0.2 h<sup>-1</sup> ( $\omega$ ) were obtained. The CI concentration was calculated as 80% of the COD effluent concentration, and the CS concentration as the difference between the total COD influent concentration and the sum of CI and CR. The total COD is fractionated in approximately 15% CI, 10% CR and approximately 75% CS. For the bacteria concentration values of 50 mg·kg<sup>-1</sup> (surface of sand layer) to 25 mg·kg<sup>-1</sup> (bottom of sand layer) with a linear distribution were assumed. These parameters were used for the following simulations of the lysimeter study.

Fig. 6 shows the simulated effluent concentrations of COD and its fractions after using the described parameter set. The deviation between the composite samples of the simulated (22.5 mg·L<sup>-1</sup>) and measured

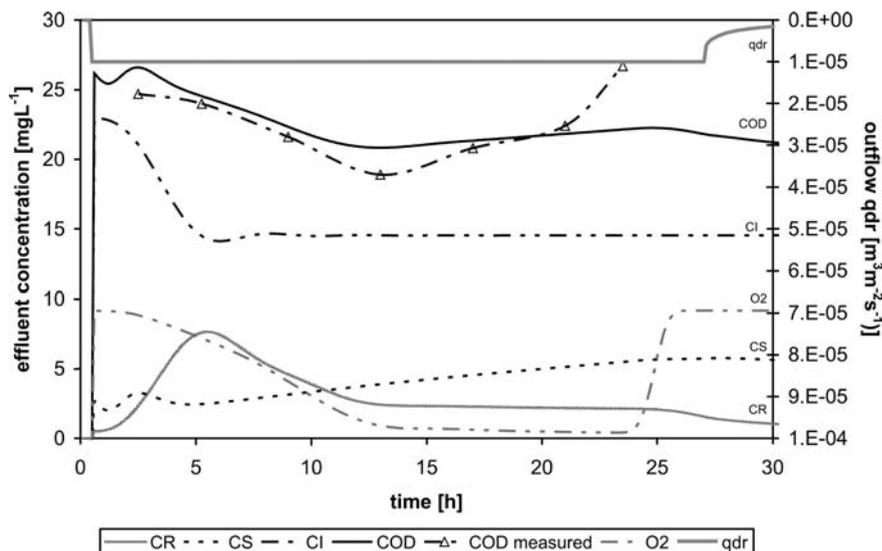


Fig. 6. Effluent concentrations of COD and its fractions with COD adsorption and the throttle outflow rate (qdr).

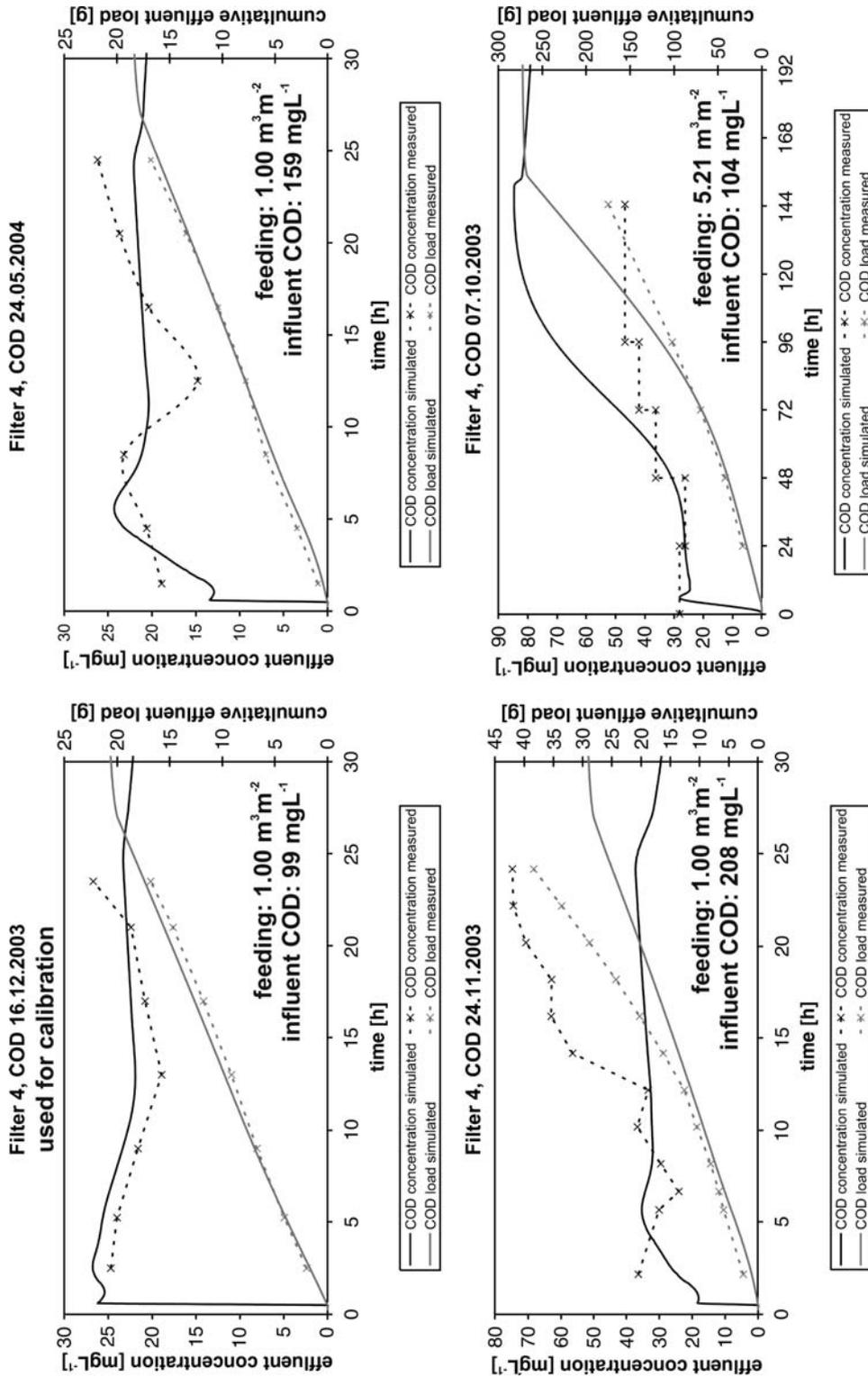


Fig. 7. Comparison of measured and simulated COD effluent concentrations and loads for the simulation of single-events of the lysimeter experiments.

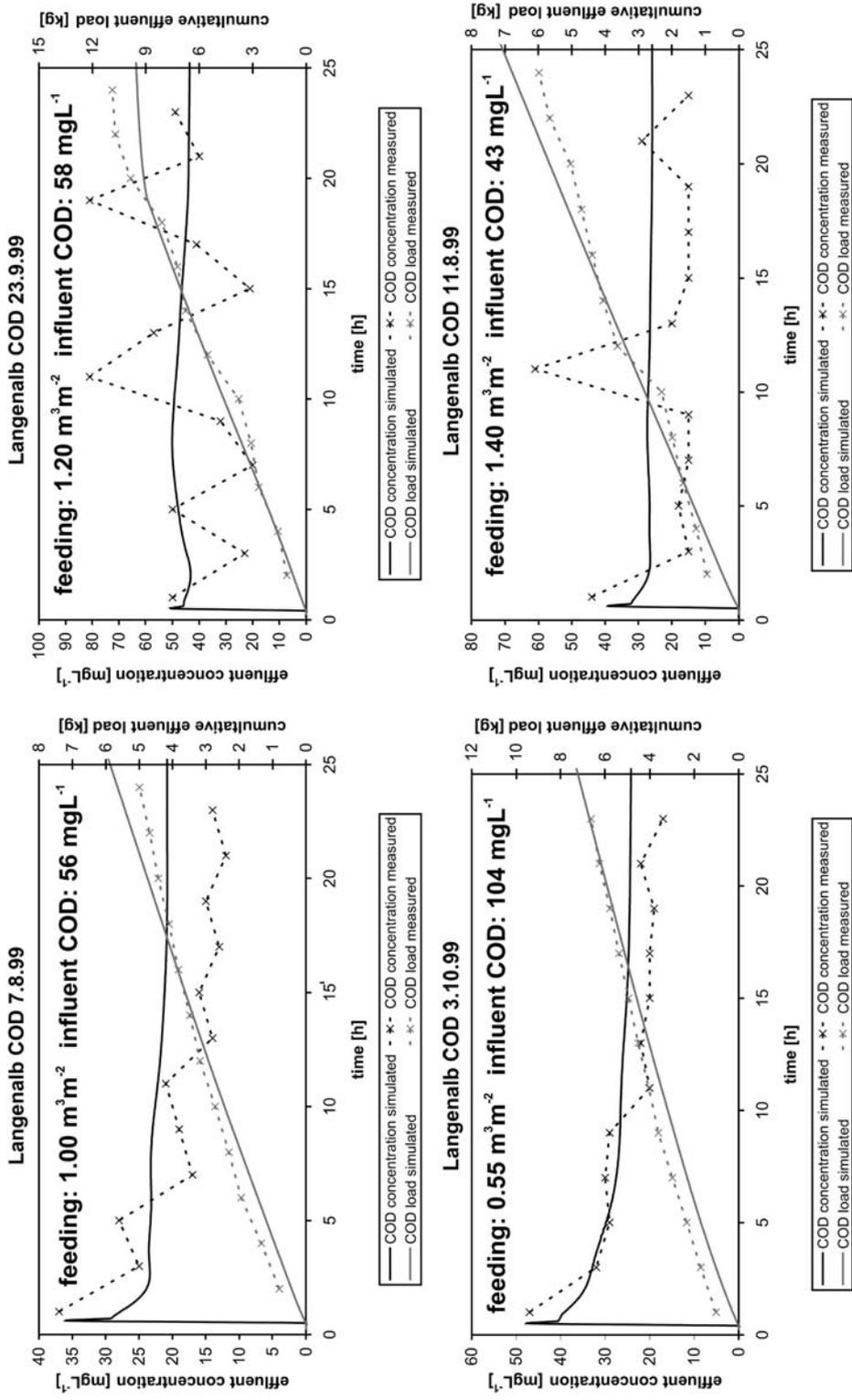


Fig. 8. Comparison of measured and simulated COD effluent concentrations and loads for the simulation of field experiments.

(22.7 mg·L<sup>-1</sup>) COD concentration is about 1.2%. The difference between the sums of loads is about 2.2%. It is obvious that by using an approach with COD adsorption a good match between simulated and measured data can be achieved.

The calibrated parameter set should be verified by using a split sample test. This means that the parameters are transferred to other data sets with (i) the same boundary conditions or (ii) different boundary conditions concerning influent concentrations and feedings.

3.2.4. Validation for single-events of the lysimeter experiments

The results for the calibration run (16.12.2003, upper left picture) and for three verification runs are shown in Fig. 7. The hydrographs of the COD effluent concentrations (measured and simulated) and of the sums of loads are pictured. The CI fraction varies in dependence to the measured COD effluent concentration between 7–16% of the COD influent concentration and the CS fraction varies between 74–83%.

For the feeding from 24.05.2004 (upper right picture in Fig. 7) small deviations between measured and simulated concentrations and sum of loads could be achieved. The visual comparison of hydrographs from 07.10.2003 (lower right picture in Fig. 7) shows a good match until a feeding time of 72 h. The breakthrough of the simulated COD concentration after 48 h is caused by CS adsorption. In contrast to the feeding from 07.10.2003 the COD breakthrough of the feeding from 24.11.2003 (lower left picture in Fig. 7) can be noticed for the measured data after 12 h with an

influent concentration of 208 mg·L<sup>-1</sup>. It is obvious that it is not possible to model both feedings with the same parameters set for CS adsorption and COD fractionation.

In this paper only events which are investigated in detail were presented. Further simulations were done and the results were compared with 24-h-composite samples. For the single events only small deviation could be observed between measured and simulated COD concentration between 1.2 and 18.4%.

For the choice of the calibration events their representativeness is crucial. The boundaries of influent concentration, influent volume (feeding), and impoundage time are important. The verification of a parameter set which was estimated with a feeding with extreme boundaries failed. The experience showed that matching of simulated to measured data for NH<sub>4</sub>-N is easier than for COD because of higher uncertainties which result from the assumption for COD such as parameters describing adsorption and the COD influent characterisation for CSO.

3.2.5. Validation for single-events of the field experiments

For the bacteria concentration and the COD fractionation the same values as in the lysimeter study were calibrated for data from the CW Langenalb. During the calibration of the adsorption parameters of the field study values of 1.5 kg·L<sup>-1</sup> (k<sub>s</sub>) and 0.15 h<sup>-1</sup> (ϖ) were estimated. Because the long-term simulations failed (cp. lab-scale experiments) for each single event initial conditions for COD fractions had to be calibrated. In dry periods CI is accumulated in the filter because of lysis

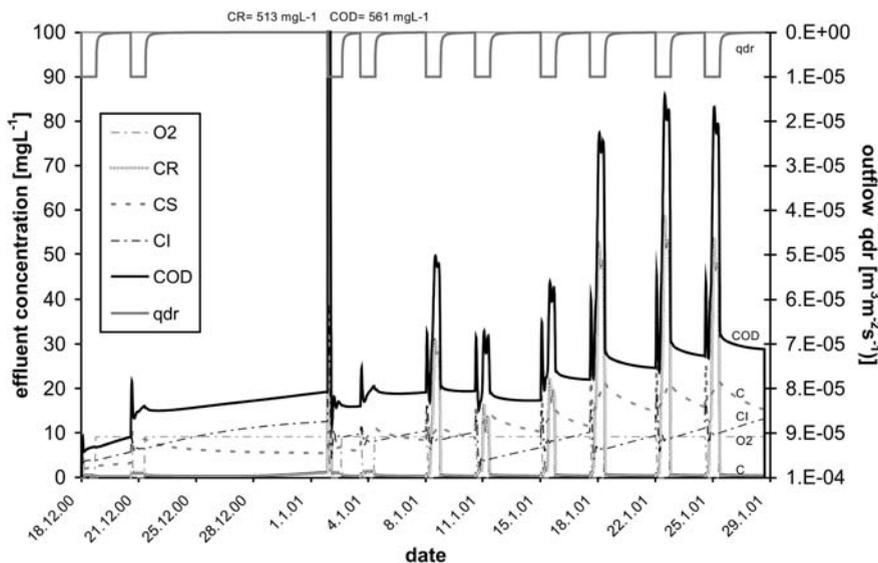


Fig. 9. Effluent concentrations and throttle outflow rate for the long-term simulations.

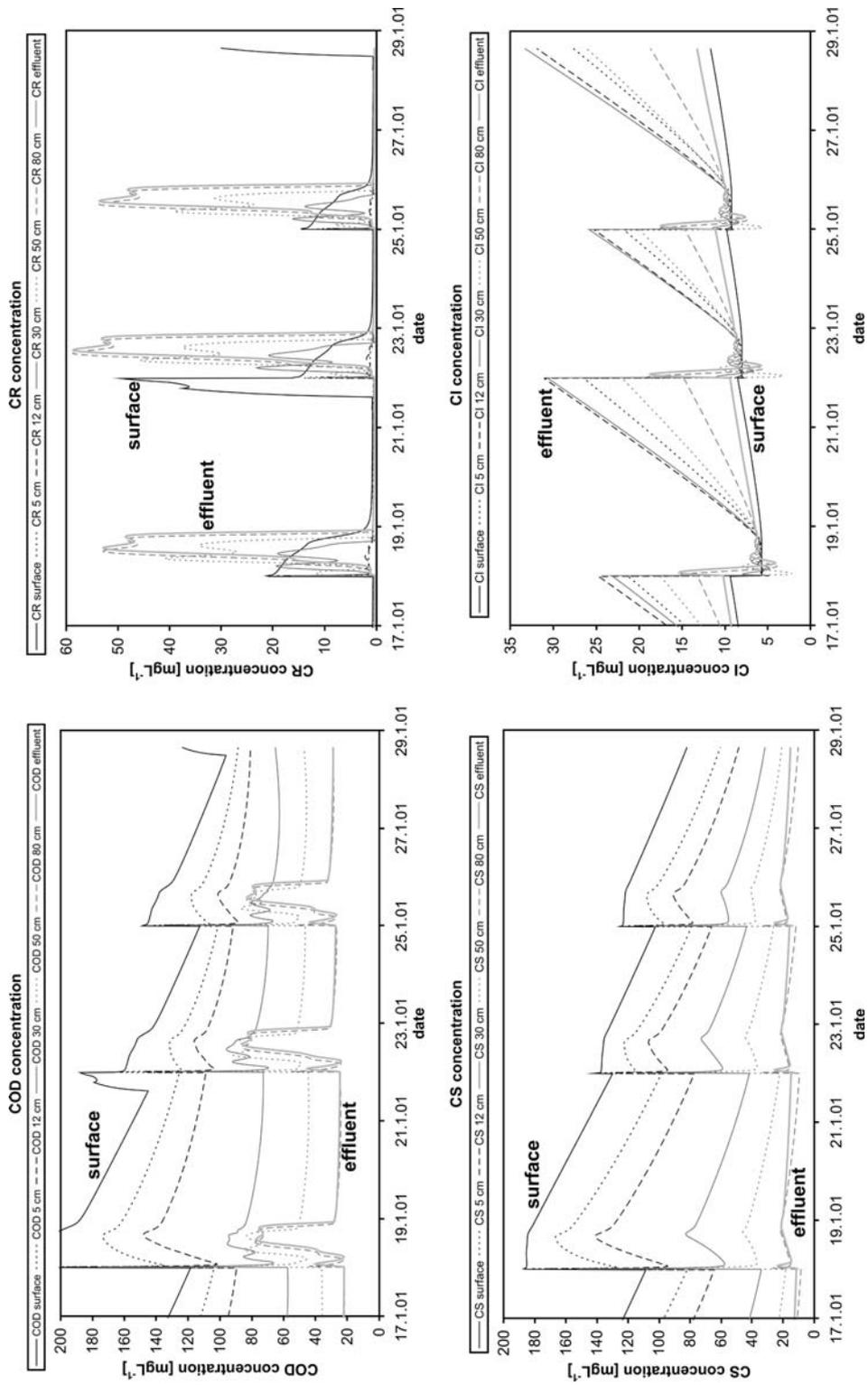


Fig. 10. COD concentrations and its fractions in different filter depths.

processes of bacteria which is leached out when a new feeding starts. To simulate these effect initial concentrations were indicated.

Fig. 8 shows a comparison of measured and simulated COD effluent concentrations and sum of loads for four single-events of field experiments. The events have different feedings and COD influent concentrations. For all events the same parameter set is used. The used parameter set can be regarded as verified. The deviation between measured and simulated COD effluent composite samples has values between 3.1–15.7%.

The COD effluent concentrations measured at the field study Langenalb do not show a breakthrough for COD. The maximum values of effluent concentrations are caused by leaching out of accumulated CI because of biomass lysis or preferential flow. After the leach out effect the effluent concentrations reach constant values which can be well simulated.

### 3.2.6. Long-term simulations of lab-scale experiments

For the simulations of the lab-scale experiments the parameter set of the lysimeter study is used. Fig. 9 shows the hydrographs of O<sub>2</sub>, COD, CR, CS, CI and the throttled outflow rate for a long-term simulation. Detailed pictures of the simulation results are shown in Fig. 10 and Fig. 11, respectively. Fig. 10 shows the COD and Fig. 11 the bacteria concentrations over the filter depths.

For every loading of the filter COD effluent peaks can be observed (Fig. 9). In dry periods CS desorps and is hydrolysed to CR, NH<sub>4</sub>-N and inorganic phosphorous. Degradation of CR is inhibited in CW2D at low NH<sub>4</sub>-N concentration. The CR-maximum value of 500 mg/l (02.01.2001) consists of accumulation of CR in dry periods when degradation is inhibited because of lack of NH<sub>4</sub>-N. Because of lysis processes of heterotrophic and autotrophic bacteria the CI concentration rises and is leached out when a new feeding starts. The simulated composite sample of COD of the first feeding yields acceptable results (measured composite sample 5.2 mg.L<sup>-1</sup>; average of the simulated effluent concentration 6.6 mg.L<sup>-1</sup>).

Fig. 10 shows a section from 18.01.01 to 25.01.01 with nearly steady-state conditions between loading rates and degradation processes. The concentrations of total COD and its fractions are shown for 7 depths of the filter (just at the surface, in 5, 12, 30, 50, 80 cm depth, and just above the drainage layer, respectively).

Another simulation problem is caused by heterotrophic bacteria. Because of the missing steady state between growth and lysis of bacteria the biomass

accumulates especially near the surface (Fig. 11). The high bacteria concentrations near the surface lead to a complete consumption of CR by respiration and therefore a depletion of oxygen. Because of hydrolysis in deeper areas the CR concentration rises.

## 4. Conclusion and outlook

In CWs for CSO treatment it is assumed that degradation processes in the dry phases between two loadings play a major role in the overall system. This is generally agreed for ammonia (e.g. Uhl and Dittmer, 2005; LFU, 2002) where high effluent nitrate concentrations can be measured indicating that nitrification occurred during the dry period. Therefore for ammonia during saturation of the filter adsorption is a crucial process. In the dry periods ammonia is desorbed and finally nitrified.

The presented simulation results showed good matching between measured and simulated effluent concentrations and COD effluent load for single events of the lysimeter and field experiments. It is assumed that mainly due to the inaccuracies for the description of COD degradation processes in the dry periods long-term simulations failed. The main reason is that too many assumptions have to be made. The three main assumptions for modelling of COD degradation in CWs for CSO treatment causing uncertainties are:

- the influent COD fractionation,
- the concentration and distribution of micro-organisms (heterotrophic bacteria), and
- the degradation processes in dry periods (organic matter degradation and nitrification).

There is a lack of detailed understanding of the coupled process of degradation, filtration and adsorption occurring. These uncertainties lead to a high research need. On the one hand model specific data like continuous measurement of effluent concentrations had to be estimated. On the other hand crucial processes had to be understood. Methods for measurement of lysis rates and growth of biofilms and bacteria in vertical subsurface flow CWs must be developed. There is especially a lack of measured data on concentrations in different filter depths in the dry periods that give information about rates of degradation processes.

In further studies the sediment layer at the top of the filter which plays an important role for adsorption and degradation processes has to be considered. The clogging effect which influences the water and solute transport was also not considered. Also the effects of

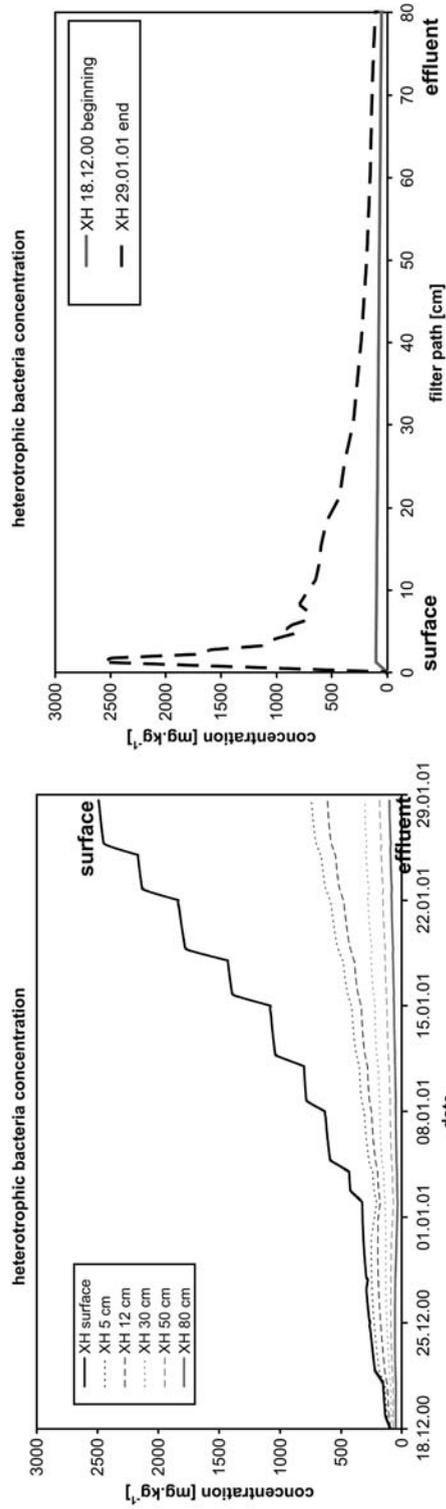


Fig. 11. Concentration of heterotrophic bacteria in different filter depths over time (left) and along the filter path for two days (right).

plant roots and root water uptake will be regarded in further studies.

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