Twenty years' monitoring of Mèze stabilisation ponds: part II – removal of faecal indicators

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Abstract The WSP system serving Mèze and Poussan (French Mediterranean coast) was constructed in 1980 and enlarged and upgraded from 1994 to 1998. Water quality along the waste stabilisation pond to (WSP) system has been monitored over the years, thus allowing us to assess the influence of enlargement and upgrading works. A significant enhancement of the average microbiological quality of the effluent was observed, with respective *E. coli* and streptococci average abatements of 4.1 and 3.4 log. units. Former seasonal variations of microbiological removal have vanished. The contribution of the different ponds to the disinfection performance of the WSP system was analysed. A microbiological quality model was proposed to evaluate the die-off kinetics related to the different ponds and as a tool for the design and management of WSP systems. Though the relationships between die-off coefficients and environmental factors appeared somewhat frail, this modelling is considered a promising approach for the prediction of WSP microbiological performance.

Keywords Die-off kinetics; disinfection; modelling; waste stabilisation ponds

Water Science & Technology **Vol 51 No 12 pp 33–41** © IWA Publishing 2005

Introduction

Twenty five years ago waste stabilisation ponds (WSPs) were recommended for the treatment of wastewater to be reused for unrestricted irrigation (WHO, 1989). About the same years, WSP technology was chosen to protect bathing waters and shellfish breeding areas along the Mediterranean coast of Languedoc and Roussillon (Southern France). A prominent implementation of this policy was the construction of WSPs to treat the sewage of all the municipalities of the northern part of the watershed of the Thau coastal lagoon. Since then, disinfection in WSPs has been much studied, at the real scale, on pilot plants and in the laboratory. However, predicting disinfection performance still remains a somewhat tricky aspect of WSP state-of-the-art. This is due to the high number of factors and interrelated processes involved. Disinfection in WSP can be seen as resulting from the combination of pond hydrodynamics, which control water retention times, and micro-organism decay processes. Both, hydrodynamics and die-off processes, are highly dependent on meteorological factors, mainly temperature, solar irradiation and wind. The natural variability of these factors adds to the difficulty of disinfection investigation and prediction (Brissaud *et al.*, 2003).

Therefore, despite the valuable and promising findings of many investigators, from Racault *et al.* (1984) to Shilton and Harrison (2003) on hydrodynamics and from Mezrioui (1987) to Davies-Colley *et al.* (2003), on micro-organism die-off kinetics, there is still much to be learnt from the monitoring of full-scale plants. Monitoring the impact of upgrading and enlargement works on effluent quality may be highly significant for practical purposes (Archer and Donaldson, 2003). Such was the motivation for this paper, trying to take advantage of data collected over several periods, between the beginning of the 1980s and the year 2004, at the Mèze WSPs. We attempted to link the die-off kinetics in the different ponds of the system to meteorological variables in order to work out a model that could be used as a tool for the design and management of WSP systems.

Materials and methods

Mèze WSP system

The main characteristics of the Mèze WSP system are described in Picot *et al.* (2005). The plant was constructed in 1980 in order to protect the shellfish breeding areas of the Thau coastal lagoon. It was a 3-cell system (M_1, M_2, M_3) with a total surface of 8 ha and a treatment capacity of 8,000 p.e. In the early 1980s, the treated load corresponded to 4,400 p.e., augmented with winery effluents in autumn (Figure 1). At that time, no microbiological criterion was set on the effluents discharged into the Thau lagoon. The plant became overloaded a few years later, due to serviced population growth as well as increasing industrial loads. Nowadays, the load to be treated amounts to up to 1,120 kg BOD_5 per day, being equivalent to 19,000 p.e. Furthermore, the new consent conditions require that WSP effluent *Escherichia coli* and enterococci contents are less than 1,000 cfu/100 mL in summer and 10,000 cfu/100 mL in winter. These requirements may be strengthened in the near future. The plant had to be retrofitted in order to cope with these new conditions.

Eight new ponds were added to the 3 old ones between 1996 and 1998, the total pond surface reaching 14 ha. At first, in the early 1980s, a typical French 3-pond system, it is nowadays a sophisticated system of 11 ponds, some of them being aerated, step fed and with possible recirculation. Moreover, the management of the system has been complicated in order to optimise the performance of the step-fed facultative stage and alleviate the impact of storm events on microbiological water quality. A part of the effluent of the anaerobic ponds is deviated to M_1 without passing through the step-fed ponds (Figure 1). Water depth is controlled in every pond through weir level adjustments. Wastewater is stored during storm events in order to increase water detention times and prevent degradation of the effluent microbiological quality. Raising and lowering the water levels entail

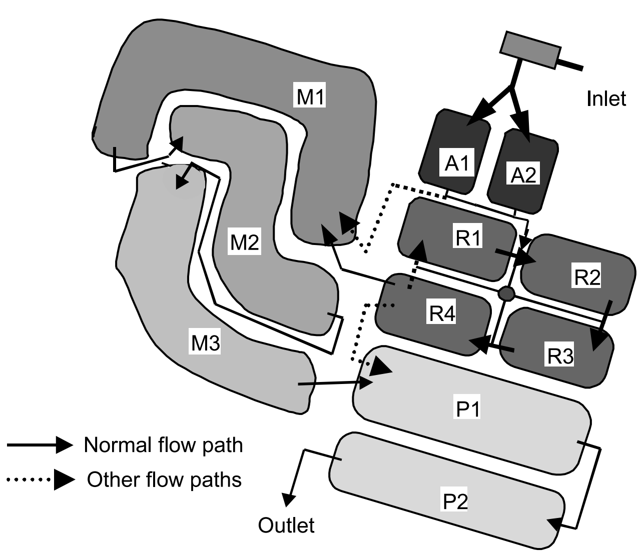

Figure 1 Layout of the current Mèze WSP system

temporary modifications in the hydraulic circuits which tend to affect the microbiological quality.

Monitoring

Water quality was monitored at the inlet and outlet of ponds M_1, M_2, M_3 fortnightly from June 1980 to August 1982 (Trousselier *et al.*, 1986), monthly from November 1984 to June 1986 (Mezrioui, 1987) and from October 1988 to January 1990 (Picot *et al.*, 1992). Water quality was observed at the inlet of M_1 and outlet of M_3 from August 1994 to February 1995 before, during and after desludging of pond M_1, with a time step varying from 1 to 3 weeks (Crabos *et al.*, 1996). From 1980 to 1998, inlet and outlet flow rates were not recorded; only a few measurements are available.

The current facility was completed in 1999. Since then, inlet and outlet flow rates have been recorded and water quality analysed on a fortnightly basis at the plant inlet, the outlet of the deep surface aerated ponds A_1 and A_2, the outlet of R_4 which is the 4^{th} pond of the step-fed facultative aerated stage, the outlet of the maturation pond M_3 and the outlet of the last polishing pond P_2. Additional analyses were made, from July 2003 to August 2004 at the outlets of M_1, M_2 and F_1 in order to identify the part played by each pond in the performance of the system.

Meteorological data were available, recorded either on site or at Sète, on the south bank of the Thau lagoon. Water temperature was measured fortnightly.

Modelling microbial decontamination

The model of microbial decontamination is based on bacterial mass balance, assuming that each lagoon is perfectly stirred and bacterial decay is a first-order kinetic process. The bacterial balance is expressed for each pond as:

$$\mathrm{Vi}\frac{\Delta \mathrm{N(i,j)}}{\Delta \mathrm{t}} + \mathrm{N(i,j)}\frac{\Delta \mathrm{Vi}}{\Delta \mathrm{t}} = \mathrm{Q}_{(\mathrm{i}-1,\mathrm{j})}\mathrm{N}_{(\mathrm{i}-1,\mathrm{j})} - \mathrm{Q}_{(\mathrm{i,j})}\mathrm{N}_{(\mathrm{i,j})} - ki\mathrm{N}_{(\mathrm{i,j})}\mathrm{Vi} \tag{1}$$

where i is the ith pond and j the jth time; Vi is the water volume in the pond i (m^3); $\Delta N_{(i,j)} = N_{(i,j)} - N_{(i,j-1)}$; Δt is the time step; $N_{(i-1,\ j)}$ and $N_{(i,\ j)}$ are microorganism concentrations at the inlet and in lagoon i respectively in the day j (CFU/100 ml); $Q_{(i-1,\ j)}$ and $Q_{(i,\ j)}$ are the inflow and outflow rates of lagoon i (m^3/day); k_i is the die-off coefficient of lagoon i (Xu *et al.*, 2002). Several inlets with different concentrations can be taken into account by adding corresponding terms in Equation 1. Flow rates have been calculated through water balances, at each time step, from inlet flow rate in the plant, rainfall, evaporation and water levels. Water levels have been measured fortnightly during the last few years.

Die-off coefficient, k_i, is a key parameter for disinfection prediction; it is known to depend on several environmental variables, mainly temperature, solar radiation, suspended solids, pH, DO, … "Observed" k_i values can be calculated directly from Equation 1 provided inlet and outlet microorganism concentrations have been analysed and hydraulic parameters are known. It is desirable for prediction purposes to link k_i to the more influential variables; according to the work of Xu *et al.*, (2002), temperature, solar radiation and SS were considered. Die-off rates were expressed as follows:

$$\mathrm{k} = \alpha \times \beta^{(\mathrm{T}-20)} \times \mathrm{e}^{0.\chi \mathrm{Im}} \tag{2}$$

where T is the temperature, Im, the depth averaged solar intensity – which is a function of SS – and α, β and χ are constants determined through an optimisation process. Values of α, β and χ minimise the difference between values of N (i, j) respectively observed and calculated using Equation 1 where N(i − 1, j) is an observed value and k_i is expressed according to Equation 2.

Results

Average removals of faecal coliforms (or *E. coli*) and streptococci were respectively only 2.9 and 2.6 log. units during the first years of operation. High seasonal variations were observed; winter outlet faecal coliform contents were up to 3 log. units higher than summer values, which meant a lower protection of shellfish breeding areas (Figure 2). During the following years, as the applied load increased progressively, microbiological performances slowly degraded. Measures from August 1994 to February 1995 reflect disruptions due to desludging works.

After the enlargement and retrofitting of the plant at the end of the 1990s, a significant enhancement of the average microbiological quality of the effluent was observed, with respective coliforms and streptococci average abatements of 4.1 and 3.4 log. units. This improvement was not due to a decrease in the applied load: as an example, the average surface organic loading was nearly the same, ≈65 kg $BOD_5 ha^{-1} d^{-1}$ in 1988–1990 and 72 in 2000–2003 – but to the physical design of the upgraded plant, with a dramatic

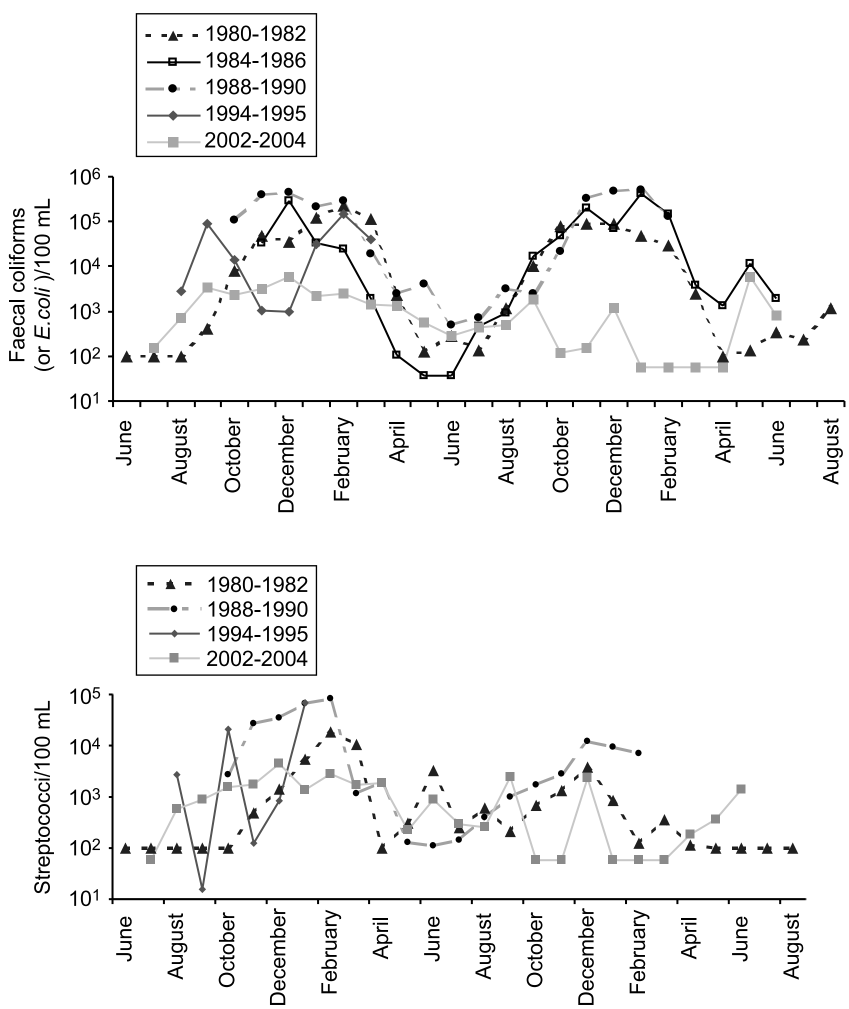

Figure 2 Disinfection performance over the years

increase in the number of ponds, and, indirectly, to the partial aeration of primary and secondary ponds. The improvement is particularly significant in winter and constitutes a valuable asset with respect to oyster breeding. Contrary to what was observed before the enlargement, the bacterial abatement did not show any clear seasonal variation. This can be viewed as in part the consequence of the variations of the water volume stored in ponds M and P in order to mitigate the influence of rain events.

The contribution of the different ponds to disinfection is illustrated in Figure 3. Primary deep aerated ponds eliminated no more than 0.5 log unit *E. coli*. Step-fed aerated facultative ponds, the hydraulic loading of which was limited to 2,500 $m^3 d^{-1}$, were responsible for about 2 log. units removal, with significant variations that might be related to weather conditions and the recirculation rate. The maturation pond M_1 played a fair part in the improvement of the microbiological water quality. M_2 removal performance was poor during the first 6 months and then improved without any clearly identified reason. With the exception of analyses made during or after heavy storm events, *E. coli* content in M_3 effluent fell below 10^3cfu/100 mL, with several values below the detection limit. For most of the year 2003–2004, microbiological quality degraded in pond P_1. A possible contamination by waterfowl droppings was alleged; more convincing is the influence of leakage and temporary by-pass allowing a variable proportion of R_4 effluent to reach P_1 directly, thus jeopardising the benefit of disinfection performed in maturation ponds. The last pond, P_2, provided an additional average abatement of about 1 log unit. From a strictly sanitary point of view, the usefulness of ponds P_1 and P_2 may be questioned. Given the abatement observed in P_2, a better control of flow paths between the ponds should result in a higher microbiological effluent quality.

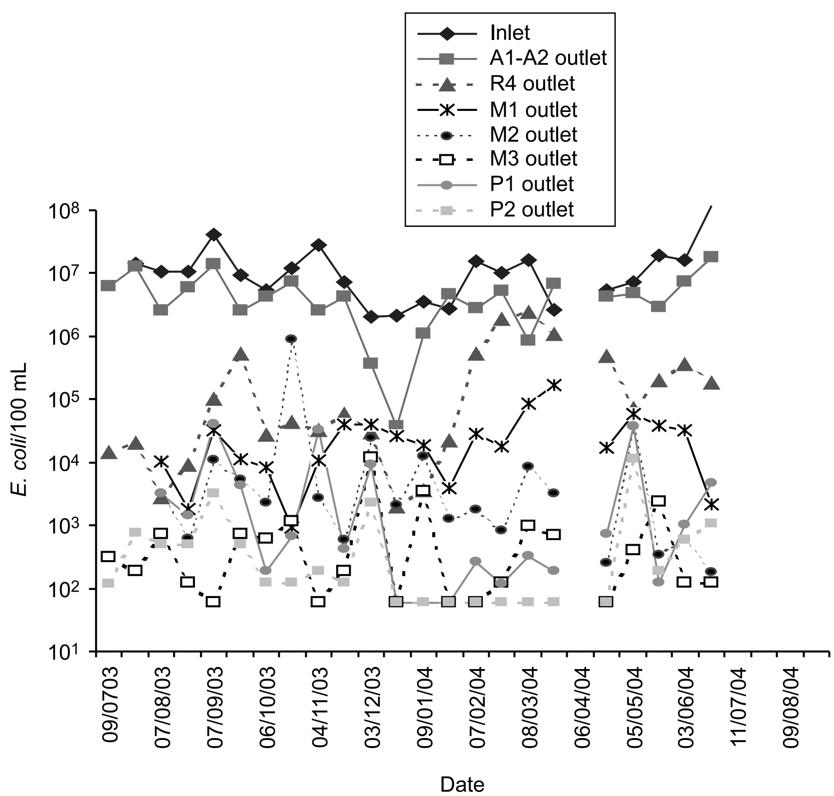

Figure 3 Contribution of the different ponds to *E. coli* removal

Die-off coefficients

E. coli die-off coefficients have been calculated for ponds M_1, M_2 and M_3, applying Equation 1 at a 2-week time step for the period 1980–1982 and a monthly time step for the periods 1984–1986 and 1988–1989 (Figure 4). Mean *E. coli* die-off coefficient of pond M_1, at that time a facultative pond, was found to be about only $0.13\,d^{-1}$ in 1980–1982 and 1984–1986 and $0.46\,d^{-1}$ in 1988–1989; no clear seasonal variation appeared. Though highly variable from one year to another, seasonal effects were more easily identified in maturation ponds M_2 and M_3. M_2 *E. coli* die-off coefficient, k_{M2}, reached $80\,d^{-1}$ in summer 1980, no more than $3\,d^{-1}$ in summer 1981 but $400\,d^{-1}$ in summer 1982. Peak k_{M2} values were respectively 6 and $20\,d^{-1}$ in summers 1985 and 1989. Very low k_{M2} values were observed in winter, with respectively, 0.04, 0.01 and $0.015\,d^{-1}$ in December 1981, January 1986 and November 1988. Variations of M_3 *E. coli* die-off coefficient, k_{M3}, were also shown to depend on seasons, with highest values in summer – respectively 10, 20, 18 and $7\,d^{-1}$ in 1980, 1981, 1982 and 1989 – or in spring (more than $150\,d^{-1}$ in 1985). Winter k_{M3} values could be as low as $0.1\,d^{-1}$ in February 1981 and zero in January 1986 and 1989).

Modelling water balances in each pond allowed assessing unknown outlet flow rates of the different ponds, from A_1/A_2 to P_2. from July 2003 to August 2004. The calibration procedure consisted of optimising the simulation of P_2 outlet flow rate through the adjustment of water depth in the different ponds, the available observed data being taken into account. Then *E. coli* die-off coefficients were calculated using Equation 1.

E. coli die-off coefficient in pond M_1, k_{M1}, showed higher values than before retrofitting and a marked impact of the season, with values up to $8\,d^{-1}$ in August 2003 and $20\,d^{-1}$ in July 2004 while k_{M1} dropped to $0.2\,d^{-1}$ in January (Figure 5).These values and evolution of k_{M1} were somewhat similar to what was observed in maturation ponds M_2 and M_3 before the retrofitting and manifested the new part played by pond M_1; formerly a facultative pond, it acted as a maturation pond after the addition of anaerobic and step-fed aerated stages. The variations of k_{M2} and k_{M3} did not show any clear seasonal effect (Figure 6). The variations of k_{M3} were found too erratic to fall in any consistent correlation. Though the order of magnitude of k_{M2} and k_{M3} was not much affected by the retrofitting of the WSPs, the processes involved in microbial decontamination were influenced by the new operation conditions.

Relationships between die-off coefficients and meteorological conditions were established through an optimisation procedure; α, β and χ were determined for ponds M_1, M_2 and M_3 for the periods January to June 1986 and November 1988 to December 1989 and for A_1/A_2, $R_{1,2,3,4}$, M_1, M_2 and M_3, P_1 and P_2. Die-off coefficients were then calculated from meteorological data recorded from July 2003 to August 2004. *E. coli* die-off coefficient in ponds A_1/A_2, k_A, was found to be remarkably constant over the year, with

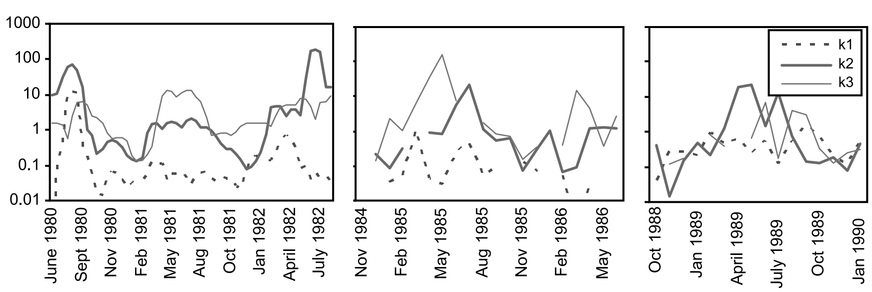

Figure 4 *E. coli*/Faecal coliforms die-off coefficients in ponds M_1, M_2 and M_3 before retrofitting

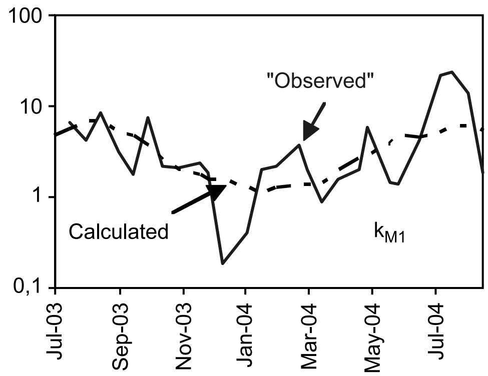

Figure 5 *E. coli* die-off coefficient in pond M_1 (July 2003 to August 2004). "Observed" values were obtained from Equation 1 while calculated values were derived from Equation 2

a mean value of $0.4 \pm 0.07\,d^{-1}$. *E. coli* die-off coefficient in ponds $R_{1,2,3,4}$, k_R, showed clear seasonal variations: from $1.1\,d^{-1}$ in July and August, k_R decreased steadily to $0.17\,d^{-1}$ at the end of January, started increasing at the end of April to reach $0.8\,d^{-1}$ at the end of May. Despite abrupt variations, a similar trend was observed in pond M_1, with k_{M1} values around $4.5\,d^{-1}$ in July and August 2003, a steady decrease to a minimum of $0.15\,d^{-1}$ at the end of December. k_{M1} started increasing at the end of March, from 0.17 to more than $20\,d^{-1}$ in July 2004 (Figure 5). The optimisation procedure found variations of k_R and k_{M1} mainly related to water temperature.

Die-off coefficient variations in ponds M_2 and M_3 were hardly matched by calculated values (Figure 6). Both variations observed and calculated from the optimisation procedure using data collected in the period 2003–2004 did not reflect any seasonal effect on *E. coli* removal. On the contrary, k_{M2} values obtained from Equation 2, using meteorological and SS data of the period 2003–2004 and α, β and χ values derived from the optimisation procedure applied to the data of the periods 1986 and 1988–1989, displayed a clear seasonal trend. The same observation applies to the k_{M3} variations calculated from α, β and χ values optimised over 1988–1989 data. Though differences in operation conditions before and after retrofitting can be evoked, it should be stressed that different monitoring periods provided significantly different sets of α, β and χ values. Therefore, this suggests that Equation 2 failed to report accurately the mechanisms involved in *E. coli* removal, either because inappropriate representation of the impact of meteorological variables or for other influential factors were not taken into account.

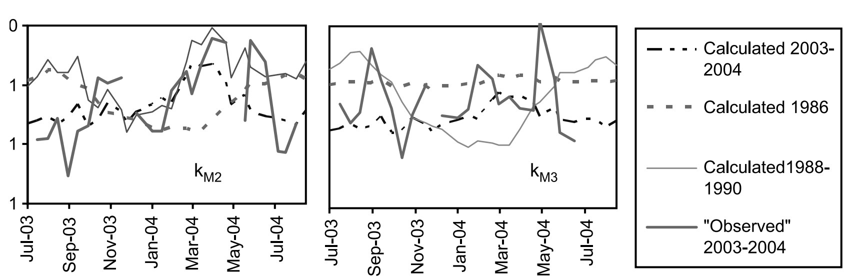

Figure 6 *E. coli* die-off coefficients in ponds M_2 ad M_3 (July 2003 to August 2004)

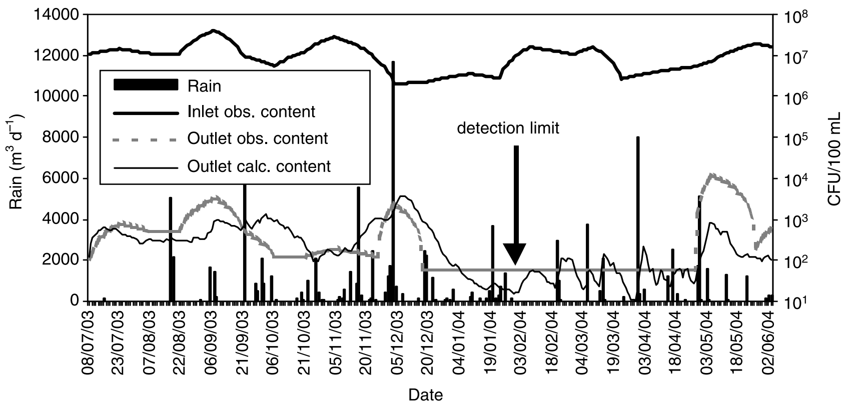

Figure 7 Modelling *E. coli* content in the effluent of Mèze WSP system

Uncertainties related to the amount of R_4 effluent passing directly to P_1 made any relationship between k_{P1} and the meteorological conditions illusive. Die-off coefficient, k_{P2}, appeared to be season dependent with average values of $0.64\,d^{-1}$ from July to September, $0.28\,d^{-1}$ from October to February and $0.78\,d^{-1}$ from March to June.

Effluent quality was calculated through the application of Equation 1 in each cell, one cell after the other, k values being derived at each time step from the relationships established between α, β and χ and water temperature, solar radiation and SS content. Microorganism contents in each cell were thus calculated from the observed microorganism content observed at the inlet of the WSP system. Despite discrepancies between the observed and calculated k values, particularly after the storm of early May, and the above mentioned deficiencies of Equation 2, the simulation was considered satisfactory (Figure 7).

Conclusions

Retrofitting of the Mèze WSP system resulted in a significant enhancement of the average microbiological quality of the effluent, with respective *E. coli* and streptococci average abatements of 4.1 and 3.4 log. units. This improvement was not due to a surface loading decrease but to the subdivision of the plant into a high number of cells and to the step-fed deep aerated pond system. The contribution of the polishing ponds to the disinfection performance was not without ambiguity. Therefore, it might be considered that, owing to the microbiological requirements of the present regulations, the total pond surface could have been significantly reduced. This is considered as a beneficial effect of the design of the first two stages of the retrofitted plant. However, as they play a significant part in the mitigation of storm events, the polishing ponds are not likely to be decommissioned.

Former seasonal variations of microbiological removal have vanished; this is considered the consequence of the new design and the plant operation aimed at mitigating the impact of storm events.

A microbiological quality model was proposed to evaluate the die-off kinetics in the different ponds and as a tool for the design and management of WSP systems. Though relationships between die-off coefficients and environmental factors appeared somewhat frail, this modelling is considered a promising approach for the prediction of WSP microbiological performance and the design of management strategies, provided our knowledge of disinfection mechanisms improves.

Acknowledgements
The authors would like to thank the CCNBT, the Conseil Général de l'Hérault, the Agence de l'Eau RMC for their financial support and SATESE 34, ENTECH and Ecosite de Mèze for their kind assistance.

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