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IN ACTIVATED SLUDGE SYSTEMS

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ABSTRACT

The aim of this research was to show the impact of the flow conditions and variations in the aeration intensity on changes in the predicted dissolved oxygen (DO) concentrations in a full-scale activated sludge reactor. The Activated Sludge Model No. 1 was used to describe the biochemical processes. The studies were performed at the Rock Creek wastewater treatment plant in Hillsboro, OR (USA). The simulation results were compared with data originating from two 24-hour experiments. The model that best reflected the spatial and diurnal distributions of the DO concentrations was the one-dimensional advection-dispersion equation with a variable overall oxygen mass transfer coefficient in the source term for dissolved oxygen.

KEYWORDS

Activated sludge; dispersion; dissolved oxygen dynamics; mass transfer; mathematical modelling; simulation

INTRODUCTION

Dissolved oxygen (DO) concentration is one of the principal control parameters in activated sludge systems. Effective control of the air supply may significantly reduce the operational costs of wastewater treatment. Various aeration strategies can be tested by means of mathematical modelling and computer simulation. The most commonly used model, the Activated Sludge Model No. 1 (ASM1) (Henze et al., 1987), provides a detailed description of biochemical processes including carbon oxidation and nitrogen removal. However, with respect to the DO dynamics the model suffers from several deficiencies that have been reported in the literature (e.g., Lessard and Beck, 1991; Daigger and Nolasco, 1995). These deficiencies are usually a result of non-biochemical processes. The aim of this study was to demonstrate the impact of reactor hydraulics and variable oxygen mass transfer on a predicted DO profile along the longitudinal axis of the activated sludge reactor.

Many important parameters are influenced by the hydraulic flow characteristics in the activated sludge reactor including organic matter removal and settling properties of the activated sludge (Horan, 1990). The extreme patterns of flow in a reactor are described as plug flow or completely mixed. A value of the dispersion number, defined as $\left(\frac{E_L}{u \cdot L}\right)$, indicates which of the two patterns is approached. When the dispersion number is greater than 0.5-4 (Khudenko and Shpirt , 1986; Murphy and Timpany, 1967; USEPA, 1993), completely mixing can be assumed. Long and narrow tanks, for which the dispersion number is smaller than 0.05-0.2 (Khudenko and Shpirt, 1986; Eckenfelder et al., 1985; USEPA, 1993) are considered an approximation of plug flow. In traditional wastewater treatment practice, reactors have generally been designed on the basis of these ideal configurations. However, typical dispersion numbers in wastewater treatment units in practice range between 0.1 and 4 (San, 1994) which suggests that the existing deviations from ideal flow have to be taken into consideration. Several complex models are available to describe these deviations, of which the dispersion model and the tank-in-series model have found widespread application. The tank-in-series model reasonably describes only small deviations from complete mixing (Horan, 1990). However, the dispersion model appears to provide a better representation of the response curve than the equal, or non-equal tanks-in-series models when the variance of the curve is used as the criteria of comparison (Murphy and Timpany, 1967). Moreover, recent studies (San, 1994; Stamou, 1994, Stamou, 1997; M kinia, 1998) have indicated that dispersed flow reactor modelling is most appropriate in the design of full-scale activated sludge systems.

The oxygen mass transfer in the activated sludge systems can be modelled using the two-film theory, unless a constant DO concentration, S_O, is assumed. According to this theory, the transfer rate is proportional to the overall oxygen transfer coefficient, K_La. In the activated sludge systems, this coefficient can be further related to the changes in intensity of aeration. Chen et al. (1980) proposed a power function of the air flowrate, Q_A, to determine the K_La value as follows:

$$K_L a = m \cdot Q_A^b \tag{1}$$

where b and m are empirical constants.

Goto and Andrews (1985) developed a linear relationship with a non-zero intercept:

$$K_L a = m \cdot Q_A - b \tag{2}$$

The expression (2) was used in later studies (Reinus and Hultgren, 1988; Bocken et al., 1989, Holmberg et al., 1989; Huang and Hao, 1996; M kinia, 1998). Huang and Hao (1996) modified the ASM1 by introducing the linear function (2) to predict the DO dynamics in the lab scale AAA (alternating aerobic-anoxic) process. M kinia (1998) used spatially and temporally variable K_La in a model of a full-scale activated sludge reactor.

METHODS

Site description

The Rock Creek Wastewater Treatment Plant (WWTP) is located in Hillsboro, Oregon (USA) upon the Tualatin River. The plant is operated by Unified Sewerage Agency of Washington County and treats wastewater drained by a combined sewerage system from a catchment area of 163 km^2. The secondary treatment system at the Rock Creek WWTP consists of four parallel activated sludge basins. The basin explored during the studies has the following dimensions: length 84 m, width 15.6 m, depth 4.9 m. For operational purposes the reactor is divided into five equal zones, and in the dry season (May-November) the anoxic zone occupies the initial 20% of the reactor volume (Figure 1).

The aeration system is equipped with perforated membrane discs submerged 4.2 m below the liquid surface. Air is supplied from one source (blowers) but each zone has a separate end pipe. The total flowrate is controlled to maintain continuously a set point of the DO probe located in the middle of Zone 3. The proportions of air supplied to each zone can be adjusted by changing the valve settings.

Under normal dry season conditions, over 26,000 m^3/d of wastewater was treated in one reactor. The average concentrations of the principal parameters in the settled wastewater were approximately as follows: total COD = 210 g/m^3, filtered COD = 120 g/m^3, BOD_5 = 105 g/m^3, $N\text{-}NH_4^+$ = 21 g N/m^3. The biodegradable organic fraction was estimated to account for 75–80% of total COD, including 22-26% of the readily biodegradable substrate (M kinia and Wells, submitted). The ratio N-NH_4^+/TKN was approximately 0.67. The dry season effluent limits for the plant equal 7.5 g/m^3 for BOD_5, 4.0 g N/m^3 for N-NH_4^+, and 0.17 g P/m^3 for total P.

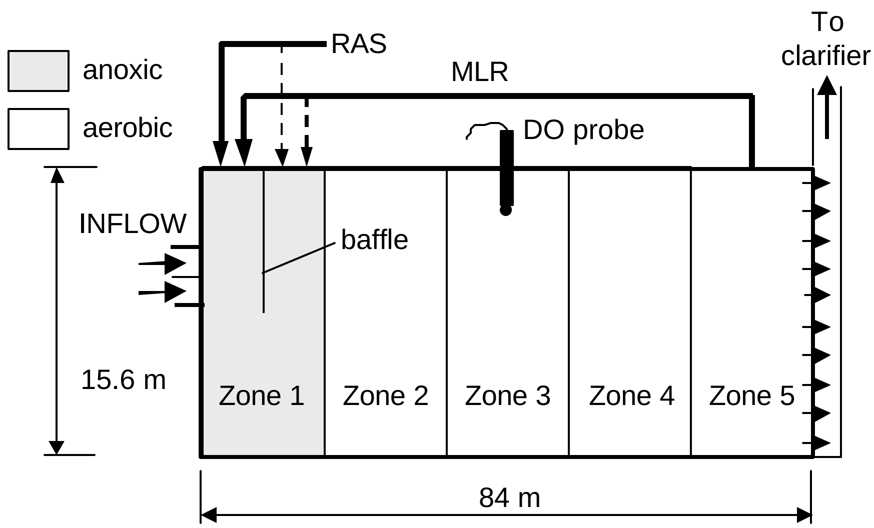

Fig. 1. Scheme of the activated sludge basin at the Rock Creek

Model development

Transport of wastewater constituents in the activated sludge reactor is described by the 1-D (assuming complete mixing in cross-section) advection-dispersion equation with a source term:

$$\frac{\partial C_k}{\partial t} + \frac{1}{A} \cdot \frac{\partial (u \cdot A \cdot C_k)}{\partial x} = \frac{1}{A} \cdot \frac{\partial}{\partial x}\left(A \cdot E_L \cdot \frac{\partial C_k}{\partial x} \right) + r_k \tag{3}$$

where C_k represents a concentration of one of the wastewater constituents (soluble and particulate) included in the ASM1. The source term, r_k, for each non-inert constituent is adopted from the ASM1 matrix. The only exception is the DO concentration, S_O, for which the source term $r_{k,So}$, is completed with the oxygen transfer from the aeration device, as follows:

$$r_{k,So} = -OUR + K_L a \cdot (S_{O,sat} - S_O) \tag{4}$$

where OUR reflects oxygen consumption in the biochemical processes (according to the ASM 1) and $S_{O,sat.}$ is the oxygen saturation concentration. Details of the model development and parameter estimation can be found in M kinia and Wells (submitted).

Simulation strategy

Using finite difference expressions of Equation (3), a computer program was written in the FORTRAN language and called the DISP model. For comparison of the flow pattern effects, a similar program was written for the tanks-in-series model, termed the TIS model using the same biochemical processes and oxygen mass transfer as the DISP model. Also, simulations were performed using the public domain program SSSP (Simulation of Single-Sludge Processes) (Bidstrup and Grady, 1988). In that program, the options of constant K_La or S_O

concentration can be selected. In this research, the second option was used with the daily average S_O in each zone based on several measurements made during the test.

In all cases, for those kinetic and stoichiometric coefficients that are known to be relatively constant in domestic wastewater, the default values were assumed. For estimating the other coefficients, lab tests using a small batch reactor were performed (M kinia, 1998).

Databases for simulations

Two 24-hour experiments were performed to collect data for testing the three models mentioned. Samples of the settled wastewater were taken automatically every two hours and analysed for total COD, TKN, TSS and TVSS. All flowrate readings were used as input data for the model simulation. During the second experiment (23-24.06.1997), all parameters were recorded with a 15-minute interval. Since the aim of the first experiment (18-19.10.1996) was primarily to test the model input requirements, some parameters (e.g., wastewater flowrate) were recorded with a longer (2-hour) interval.

The only output parameter measured continuously and recorded was the DO concentration in the middle of Zone 3. Occasionally, the DO concentrations were measured and mixed liquor samples were taken (for filtered COD, N-NH_4^+, N-NO_X) in the effluents from aerated zones. Details of both experiments can be found in M kinia (1998).

Estimation of the longitudinal dispersion coefficient

The value of the longitudinal dispersion coefficient, E_L, was estimated from tracer studies performed on the basin studied. A series of the tests were performed during the wet season when all zones of the activated sludge basin were aerated. The mean value of E_L was found for the entire basin by minimising the sum of differences between the measurements and predictions. The resulted E_L values varied in the range 1100 – 1500 m^2/h. Details of the procedure and the study results can be found in M kinia and Wells (submitted).

Alternatively, the E_L value can be estimated from the following empirical formula developed by Murphy and Boyko (1970) and recommended by USEPA (1993):

$$E_L = 3.118 \cdot W^2 \cdot \left(q_A\right)^{0.35} \tag{5}$$

where E_L is in m^2/h, the basin width W – in m, and the air flowrate q_A - in m^3/min/1000 m^3.

Estimation of the overall oxygen transfer coefficient

A series of the oxygen transfer efficiency (OTE) measurements was performed in two activated sludge basins at the Rock Creek WWTP (CH_2MHill, not published). The studies were performed using an off-gas procedure given by Redmon et al. (1983). Based on these results, the K_La coefficient was calculated by rearranging the expression for OTE given by Boyle et al. (1989), as follows:

$$K_L a = \frac{0.23 \cdot Q_A \cdot r_a \cdot OTE}{\left(S_{O,sat} - S_O\right) \cdot V} \tag{6}$$

Since the measurements had been made at 14^{o}C, the K_La values were corrected to a temperature of 20^{o}C using an Arrhenius temperature correction factor of 1.024. Next, the K_La values were correlated to air flowrates using Goto and Andrews' (1985) linear relationship. The following expression for K_La as a function of the air flowrate was derived:

$$K_L a = 0.0081 \cdot Q_A - 2.85 \quad (7)$$

where Q_A is expressed in m^3/h and K_La is expressed as in h^{-1}.Equation (7) was compared with results obtained from Reinius and Hultgren's (1988) method and good agreement was observed for medium air flowrates between 1000 and 2500 m^3/h (M kinia and Wells, submitted). For high air flowrates (over 2500 m^3/h), the off-gas technique was not applied and the K_La values had to be extrapolated.

RESULTS AND DISCUSSION

Examples of the measurements and model predictions of DO concentrations are shown in Figures 2 to 5. The principal parameter compared was the sum of differences between the continuous readings of the DO probe and predicted values in Zone 3 (Figures 2 and 4). In both cases, the model predictions oscillated around the set points equal 5 g O_2/m^3 and 4 g O_2/m^3 during test 1 and test 2, respectively, but each time the sum was smaller for the DISP model than for the TIS model. Moreover, the DISP model generated much more realistic longitudinal profiles of the DO concentrations (Figures 3 and 5). The TIS model predictions remained closer to the actual values and the DISP model predictions when the DO gradient in the vicinity of the DO probe was small, i.e., when completely mixed conditions were approached.

Another pertinent observation deals with the impact of the flow conditions and variable K_La on the predictions of other parameters. As an example, the results for N-NH_4^+ are shown in Figure 6. They confirm that dispersion affects the concentration distribution inside the reactor. The predictions obtained using the DISP model remained much closer to the actual values than the predictions obtained using the models based on the tanks-in-series approach (TIS and SSSP). To verify the impact of K_La variations, the results obtained using TIS and SSSP models were further analysed. At higher DO levels, the biochemical process rates were hardly affected by the K_La variations. Therefore, the N-NH_4^+ predictions in Zone 2 and Zone 3 were comparable for both models (TIS and SSSP). At low DO concentrations, such as in Zone 5, the impact of the variable K_La increased with respect to the biochemical process rates. Therefore, the TIS model (with variable K_La values) did and the SSSP model (with constant DO concentrations) did not reflect the effluent peak occurring as a result of the increasing ammonia load to the reactor.

The shift in strategy of controlling the activated sludge process by changing the DO profiles brought significant economic benefits . The average air input to the reactor was reduced by approximately 20% from 8500 m^3/h (1996) to 6800 m^3/h (1997). Simultaneously, the TKN loads were relatively constant, i.e., 595 kg N/d (1996) and 630 kg N/d (1997). Effluent concentrations of ammonia remained almost unchanged, and N-NO_X concentrations were even lower due to more favourable conditions for denitrification in Zone 5.

CONCLUSIONS

Flow conditions and variable aeration intensity were important factors in modelling a full-scale activated sludge systems. The 1-D advection-dispersion equation with a variable K_La in the source term for dissolved oxygen best reflected the spatial and temporal changes in concentrations of this parameter and ammonia. As an operational tool, the model developed can allow for a more flexible and effective control of the activated sludge systems by taking longitudinal and diurnal changes in the aeration intensity into account.

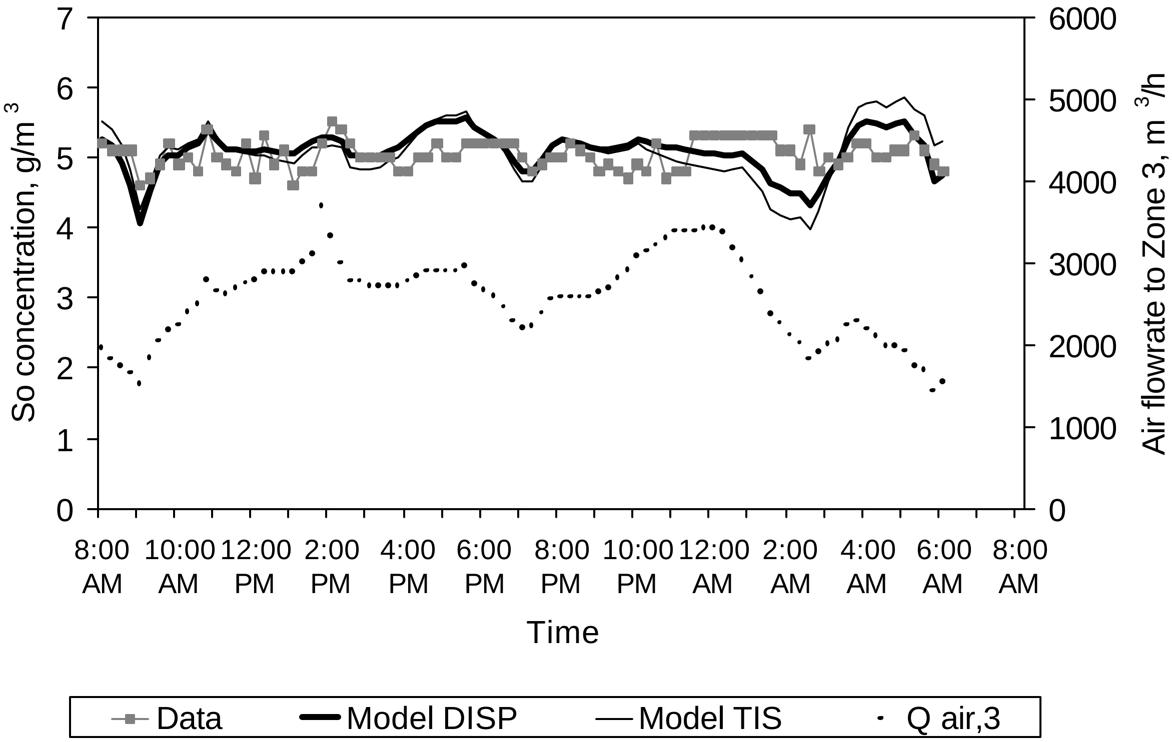

Fig. 2. Observed and predicted dissolved oxygen concentrations in Zone 3 of the activated sludge basin at the Rock Creek WWWT during the continuous experiment performed on 18-19 October, 1996

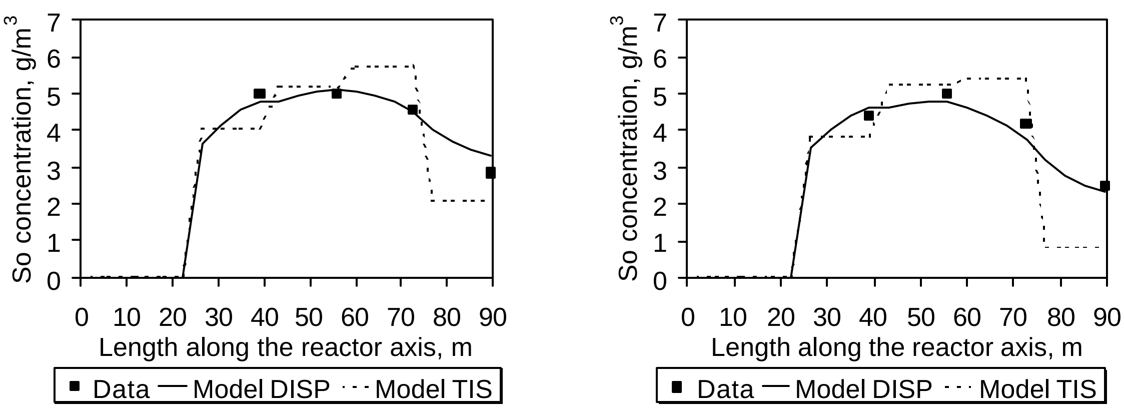

Fig. 3. Predicted dissolved oxygen profiles at the beginning and at the end of the continuous experiment performed on 18-19 October, 1996

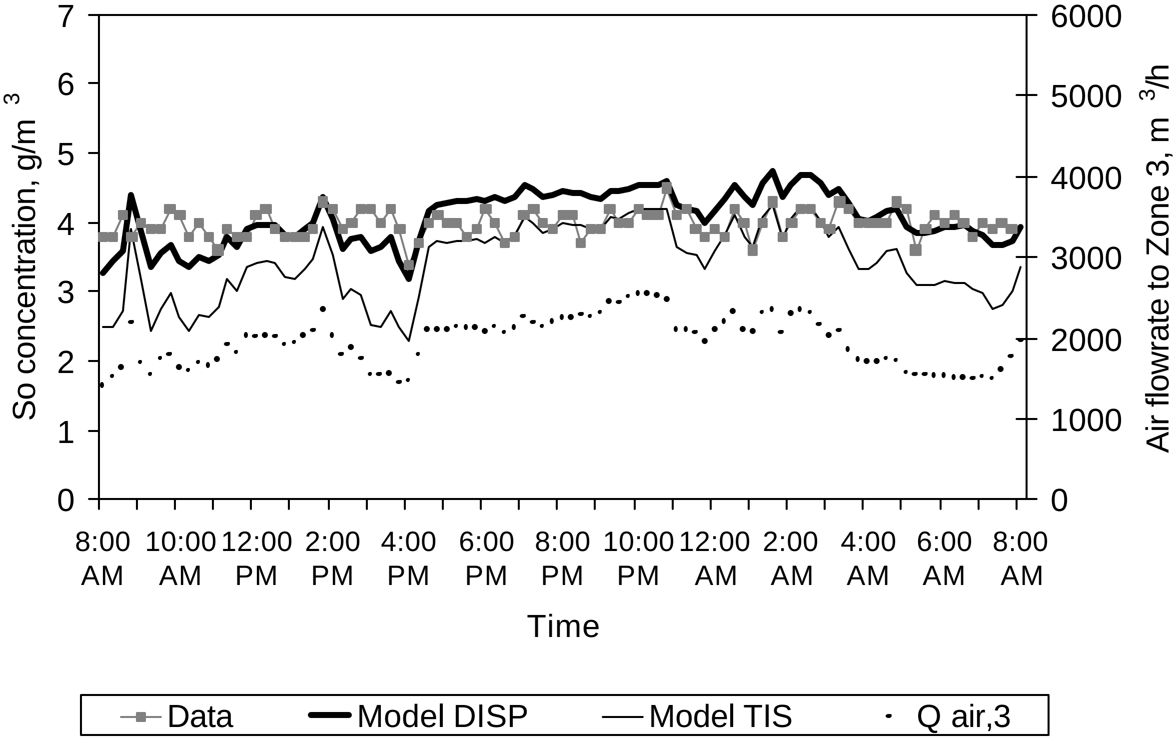

Fig. 4. Observed and predicted dissolved oxygen concentrations in Zone 3 of the activated sludge basin at the Rock Creek WWWT during the continuous experiment performed on 23-24 June, 1997

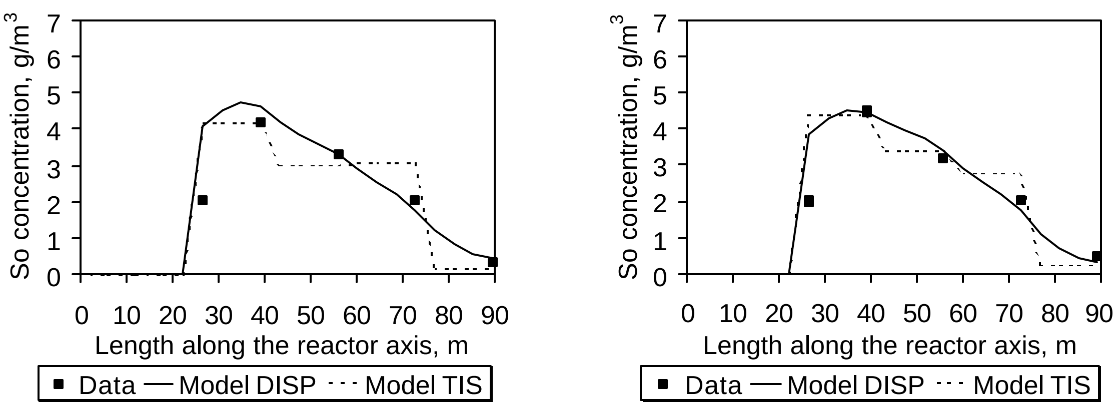

Fig. 5. Predicted dissolved oxygen profiles at the beginning and at the end of the continuous experiment performed on 18-19 October, 1996

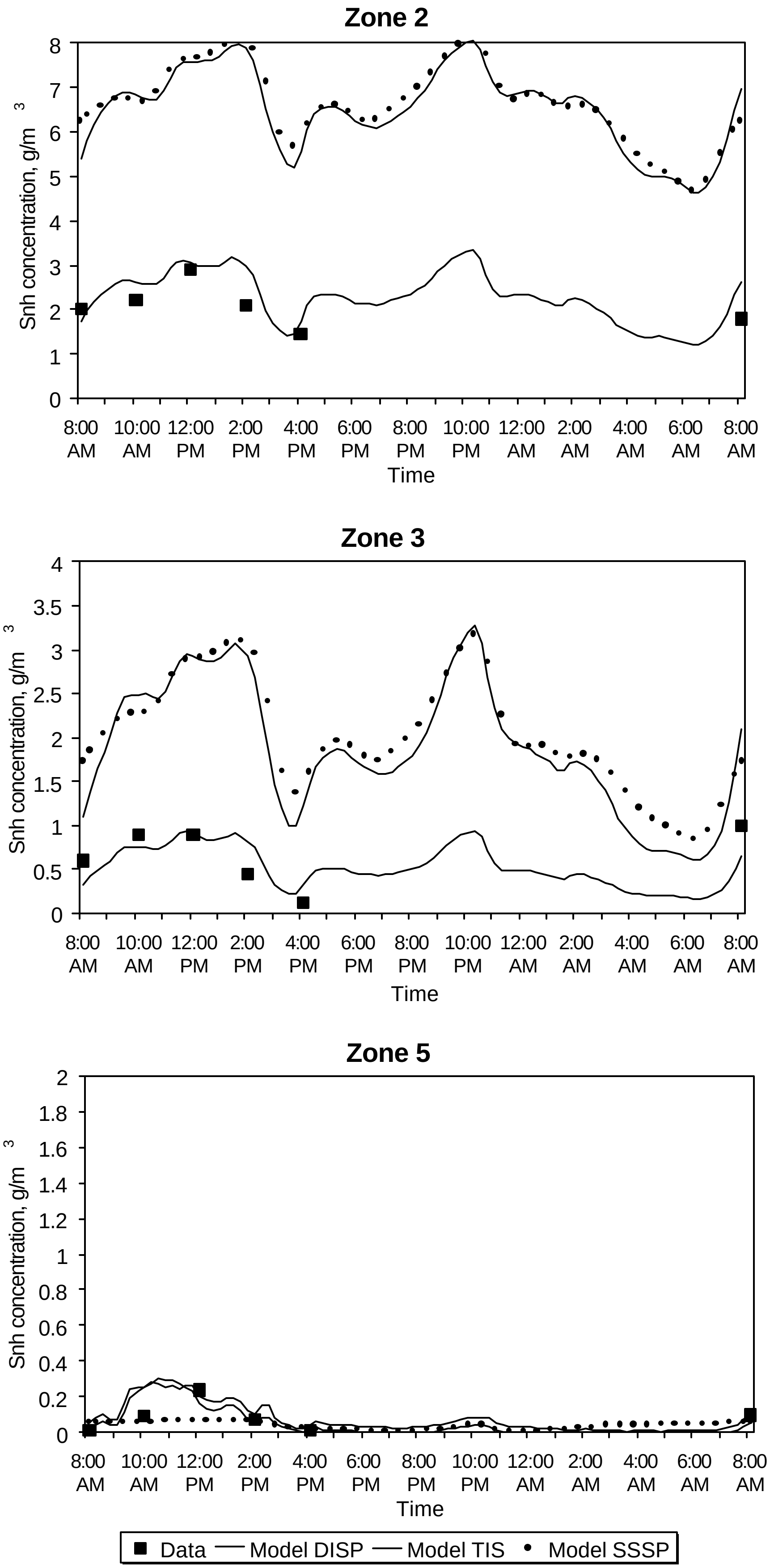

Fig. 5. Observed and predicted ammonia concentrations at various points of the activated sludge basin

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