GIS-based Surface Water Monitoring System and Modelling for Conservative and Reactive Pollutants, Applied on the Upper Basin of Olt River

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This paper presents the problem of pollutant propagation monitoring and modeling on the upper zone of a river. The first model of the pollutant propagation is based on the averaging mass balance, without computing chemical and biochemical transformations. The second model was developed using the Streeter Phelps oxygen balance, including the kinetic terms of re-aeration and oxygen consuming. Experimental data set was obtained on the upper zone of the Olt River; using this data set, the proposed model was calibrated. The development of the model is like a part of a GIS-based environmental information system. Calculated results are depicted by diagrams and thematic maps of the upper Basin of Olt River, representing the water-quality parameters.

Keywords: monitoring, pollutant, GIS-based model

On the upper Basin of Olt River, upstream and downstream to Miercurea Ciuc, the monitoring of water quality is made in regular mode through periodic sampling and analyzing the samples in the specialized laboratories of the Institute of Water Administration - Miercurea Ciuc . The sampling period is 24 h, with few exceptions, when it was found alarming flow growth or imminent danger of river pollution. Taking into account the relatively high speed of flow, it can be considered that the discharging of a large quantity of pollutant with short discharging time can pass unobserved in the sampling point because of the piston flow.

The monitoring of surface waters in the Carpathian–Danube Basin was made using classical systems (intermittent sampling) as well as automatic monitoring stations [1]. For example the parameters tracked in the Tisa Basin with automatic and online monitoring stations are presented in table 1.

The modernization of surface water monitoring stations has been developed after the ecologic accident at Baia Mare where the water of Lãpu°, Sãsar, Some° and Tisa was polluted with cyanides and heavy metals (Cu, Zn) [2]. In the Carpathian–Danube Basin researches has been made for monitoring of running waters quality, developing new methods of observation of environmental factors. Because of low flow speed of rivers, with sedimentations, these sediments are monitored. The pollutant sediments are deriving from human activity – mining of heavy metals (Cu, Zn, Cd, Ba), and metals like La, Ce, Nd. [3]. The dead channels of Danube were monitored determining the concentration of nutrients and turbidity [4]. The running waters of the Carpathian–Danube Basin were also monitored with supervision systems depending on environment and water ministries. The state apparatus makes the monitoring using different algorithms function of endowment and specific of each country from the Basin. Each country elaborates the Yearly Environment Report, which is commonly available on Internet in the site of ministries. As regards the quality of running waters in the Carpathian–Danube Basin, the available data bases were GEMS and CORINE. The Romanian running water monitoring system is formed by 633 section of supervision.

Table 1
BASIC INSTRUMENTATION OF RUNNING WATER MONITORING STATIONS IN TISA BASIN

Water quality parameter	Units	Limits	Measuring
Water temperature	C°	0-50	Digital thermometry
pH		0-14	Potentiometry
Dissolved Oxygen	mg L^{-1}	0-20	Voltametry
Conductivity	µS/cm	0-2000	Conductometry
Turbidity	NTU	0-500	Nephelomerty
Ammonium ion concentration	mg L^{-1}	0-10	Fotometry
TOC	mg L^{-1}	0-20	UV added oxygenation
Mineral oils	—	—	Surface sampling followed by TOC determination
Chlorophyll-a	µg/l	0,1<	Fluorescence
Biomonitoring (toxicity meter)	Index T	0-15	Fluorescence
Heavy metals (Zn, Cd, Pb, Cu)	µg L^{-1}	0-5000	Polarography
Nitrates	mg L^{-1}	0-10	Spectrophotometry
Cyanide	mg L^{-1}	0-2	Polarography

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From these supervision sections result the primary information concerning the characterization of Romania's running water quality. Statistical evaluations have been made on catchments and also on national level for appreciation of water quality degree. New physical, chemical and microbiological methods available for determination of pollution degrees for waters have been also used [5]. The results are presented in form of yearly reports and maps. The real-time monitoring of surface running water quality using an informational system based on GIS (Geographic Information System) with assuring system-user interactivity can be a real utility for decision factors in environment management [6].

Models

The methodology of surface running water-quality monitoring is varied because of the large scale of water characteristics. The monitoring system contains from the psychical part (instruments, hardware) and the soft part (software, models, GIS, procedures, etc.), (fig. 1). Information about environmental factors can be obtained in three ways: on line or in situ, through data acquisition system; through periodical sampling, and sample processing and analyzing in laboratory and from other sources like archives.

The way followed by different type of information depends on the type, importance and speed of environmental parameter variation. Thus, for a low-speed variation parameter is chosen the cheaper way, in the most cases the periodical sampling. For very important parameters, with high-speed variation, in most cases pollutants of high grade toxicity and for physical parameters (temperature, flow capacity) the method of in situ or on-line determination is chosen, with data transmission to the processing system. After the storing and processing of initial information, results are disseminated in tables, data bases and thematic maps.

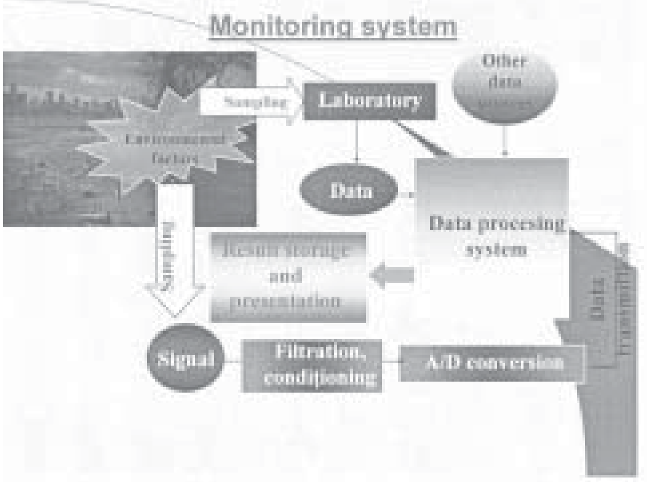


Fig. 1. A complex system for surface running water monitoring

The complexity of the monitoring system is imposed by the number of monitored environmental parameters and by abnormal values of the water quality parameters.

An aspect taken into account, connected to complexity of system, is the cost of investment, the cost of exploitation and the reliability of the information system. Ideally would be the use of a monitoring system with as many as possible sampling points, with the afferent system of data acquisition, which leads sometimes to high growth of costs. Through adequate mathematical modeling of spatial and temporal distribution of environmental parameters, the number of sampling points can be considerably reduced and implicitly the complexity of the hard part of the monitoring system. In this paper the suitability of two mathematical models in the upper flow of Olt River has been tested.

A simple model of materials balance, where the mixing of pollutants (effluents) with receiver (emissary) is considered without taking into account the chemical transformations.

The effluents are considered like point-like sources and as simplifying hypothesis was considered the, perfect mixing at the sloping zone and piston type flow of receiver (zero longitudinal dispersion). The experimental data has been collected together by SGA Miercurea Ciuc, APM Harghita and SC GOSCOM SA Miercurea Ciuc, for resolving some litigation regarding the pollution of Olt River crossing Miercurea Ciuc. In this case the materials balance equations are:

$$\sum_{i=1}^{n} Q_i = 0 \qquad \text{and} \qquad \sum_{i=1}^{n} Q_i \cdot c_{i,j} = 0 \qquad (1)$$

where

Q_i represents the flow rate of effluent i, including the emissary;

$c_{i,j}$ represents the concentration of pollutant j in effluent i, including the emissary;

This model can be applied on short sections, where distances between pollutant sources are minimal and variation of concentration of pollutants in emissary between sources are negligible. This model has been applied on the river zone which absorbs pollutants from the territory of the Miercurea Ciuc city.

A model based on atmospheric oxygen transfer and oxidative decomposition of oxydable pollutants.

On longer sections, where the reaction time is larger, the chemical and biochemical transformations in surface running waters cannot be neglected. The studying of the profile of oxygen deficit and of chemical consumption of oxygen allows the decreasing of sampling points with the condition of identification of critical space and establishing a sampling point in this place. The determination of critical space is made by trials determining the location along the river, where the concentration of dissolved oxygen is minimal. The source of oxygen for running water without meaningful vegetation is the atmospheric air, which is dissolved in water passing through the air-water interface. The speed of oxygen resolution in river's water depends on degree of saturation, on temperature, on relative speed of air at the air-water interface, and on the specific contact surface between air and water. The solubility of oxygen in water depends on water temperature, on salinity and on partial pressure of oxygen above the water. The balance state – saturation – of the dissolved oxygen – air – water system can be described with the Henry law. The temperature factor is included in Henry's constant, and the atmospheric pressure is included in the expression of partial pressure of oxygen in air. The solubility of oxygen in water, function of temperature, can be computed with polynomial relation. Solubility is corrected with correction relations, function of water salinity and pressure. The oxygen transfer from air to water can be described, in some limits, by empirical relations available for some condition as water speed and depth. The BOD - biochemical oxygen demand - model for a channel can be constructed starting from the mass balance, which for a constant debit and constant geometry can be described as:

$$\frac{\partial C}{\partial t} = -U\frac{\partial C}{\partial x} - k_r C \qquad (2)$$

where:

C represents the oxydable organic material concentration - BOD (mg oxygen L^{-1});

U - water velocity (ms^{-1}) and k_r - total speed of oxygen consumption (days^{-1})

$$k_r = k_d + k_s \qquad (3)$$

where:

k_d represents the speed of decomposition of oxydable materials (days^{-1});

k_s -speed of organic sediment expurgation (days^{-1}),

$$k_s = \frac{v_s}{H} \text{ (days}^{-1}\text{)} \qquad (4)$$

where:

ν_s represents the speed of organic sediment BOD sedimentation (m days^{-1});

H - the water depth (m).

In stationary regime:

$$0 = -U\frac{\partial C}{\partial x} - k_r C \qquad (5)$$

Considering a point pollutant source that the charging effluent BOD is perfectly mixed, the charge with organic oxydable materials can be computed with the mixing balance at the discharging point:

$$C_0 = \frac{Q_w C_w + Q_r C_r}{Q_w + Q_r} \quad \text{(mg oxygen L}^{-1}\text{)} \qquad (6)$$

where:

Q_w represents the effluent flow (m^3s^{-1});

Q_r = the emissary flow (m^3s^{-1}),;

C_w - BOD of efluent (mg oxygen L^{-1}),;

C_r - BOD of emissary (mg oxygen L^{-1}).

Using this value as initial condition the equation (5) is solved and it results:

$$C = C_0 e^{-\frac{k_r}{U}x} \qquad (7)$$

where C_0 represents the initial BOD value (mg oxygen L^{-1}).

The rate of decomposition depends on channel depth:

$$k_d = 0.3\left(\frac{H}{8}\right)^{-0.434} \quad \text{for } 0 \le H \le 8 \text{ ft} \qquad (8)$$

$$k_d = 0.3 \qquad \text{for} \qquad H > 8 \text{ ft} \qquad (9)$$

The influence of temperature upon the rate of decomposition can be formulated with the correcting equation:

$$k = k_{20}\theta^{T-20}, \text{ where } \theta = 1.047, \text{ [7].} \qquad (10)$$

Gas transfer and re-aeration can be described with the laws of gas transfer through liquid film (Whitman 1923, Lewis and Whitman 1924). The computing formulas of transfer rate express the speed of re-aeration function of channel depth. As a rule, it is used the O'Connor- Dobbins, Churchill or Owens-Gibbs formulae. Generally the O'Connor- Dobbins formula gives lower values than the Churchill and Owens – Gibbs formulae.

There are other formulae for calculation of re-aeration speed. In simulation module of QUAL2E and QUAL2K are used internal formulae for calculation of re-aeration speed [8].

The oxygen transfer can be intensified much more by constructing obstacles against the flow of water like waterfalls and barrages. Because these obstacles, strongly intervene in oxygen balance, it must be kept an inventory of them on the course of water.

The modeling of dissolved oxygen concentration in water is made with acceptable estimation with the Streeter-Phelps model. Starting from the equation of balance of the dissolved oxygen in water for the perfectly mixing model:

$$V\frac{dC_{ox}}{dt} = -k_d VC + k_a V\left(C_{ox}^{*} - C_{ox}\right) \qquad (11)$$

where:

C_{0x}^{*} represents the concentration of dissolved oxygen in water (mgL^{-1});

C_{0x} - the concentration of oxygen in water at saturation (mgL^{-1}).

Introducing the concept of oxygen deficit (mgL^{-1}):

$$D = C_{ox}^{*} - C_{ox} \text{ and } \frac{dD}{dt} = -\frac{dC_{ox}}{dt} \qquad (12)$$

and making the replacements in equation (11) one obtain:

$$\frac{dD}{dt} = k_d C - k_a D \qquad (13)$$

For limit conditions: at $t=0$, $C=C_0$ and $D=D_0$, one obtain the solution:

$$C = C_0 e^{-k_d t} \quad \text{and} \quad D = \frac{k_d C_0}{k_a - k_d}\left(e^{-k_d t} - e^{-k_a t}\right) \qquad (14)$$

For a spot pollutant source which discharges into a river, approximated with a channel with constant geometry, in stationary regime, the mass balance can be written:

$$0 = -U\frac{dC}{dx} - k_r C \qquad \text{and}$$

$$0 = -U\frac{dD}{dx} + k_d C - k_a D \text{ where } \quad k_r = k_d + k_s \qquad (15)$$

For limit conditions $t=0$, $C=C_0$ and $D=D_0$, the resulting solution is the following:

$$C = C_0 e^{-k_d t} \qquad \text{and}$$

$$D = D_0 e^{-\frac{k_a}{U}x} + \frac{k_d C_0}{k_a - k_r}\left(e^{-\frac{k_r}{U}x} - e^{-\frac{k_a}{U}x}\right) \qquad (16)$$

These equations represent the classic Streeter-Phelps model, which explains the "hole" of dissolved oxygen concentration downstream to the effluent discharge with grown organic material charge (fig. 2).

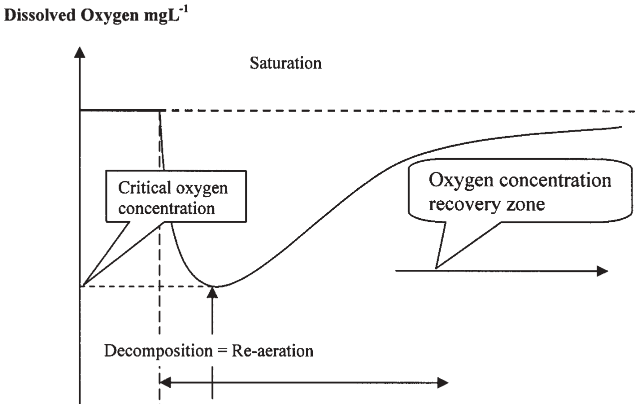


Fig. 2. „Concentration hollow" of dissolved oxygen concentration downstream of the effluent discharge with important organic material charge

Taking into account that $x/U = t$ as stationery time in plug flow reactor, the equation (16) becomes:

$$D = D_0 e^{-k_a t} + \frac{k_d C_0}{k_a - k_r}\left(e^{-k_r t} - e^{-k_a t}\right) \qquad (17)$$

Critical time t_c is obtained differentiating equation (17) for t, equaling to 0 and resolving the equation:

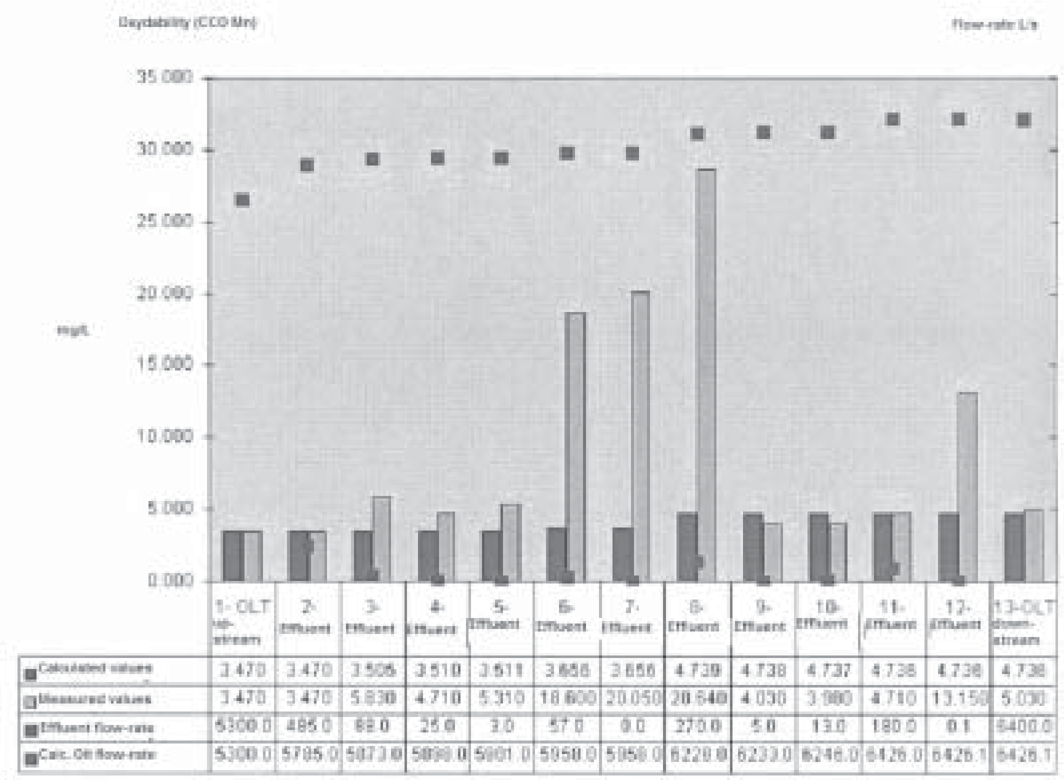
$$t_c = \frac{1}{k_a - k_r}\ln\left\{\frac{k_a}{k_r}\left[1 - \frac{D_0(k_a - k_r)}{k_d L_0}\right]\right\} \qquad (18)$$

Critical oxygen deficit D_c is obtained as follows:

$$D_c = \frac{k_d C_0}{k_a}\left\{\frac{k_a}{k_r}\left[1 - \frac{D_0(k_a - k_r)}{k_d C_0}\right]\right\}^{-\frac{k_r}{k_a - k_r}} \qquad (19)$$

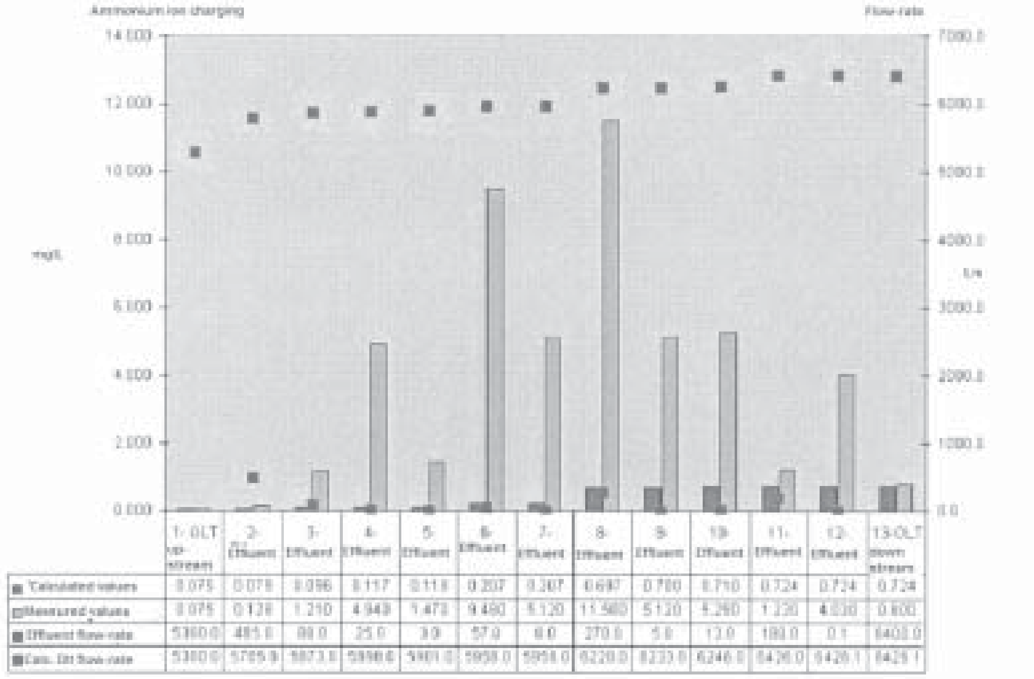
The relations (18) and (19) can be used in model calibration. Knowing the values of D_0 and C_0, from flowing regime and river bed geometry, the speed of re-aeration is determined and from the experimental determination of critical time (critical distance) the maximum (critical) oxygen deficit can be also determined. With these set of values k_r and k_d are determined [8]. Knowing the kinetic constants k_a, k_r and k_d, the profiles of oxygen deficit and chemical oxygen consumption can be determined. The mathematical model has been solved in Matlab 7.0.

The determination of dissolved oxygen concentrations has been realized with a Consort C933 instrument, determining in parallel the temperature of water and atmospheric pressure, for the calculation of dissolved oxygen deficit. The oxydability has been determined using the titration with potassium permanganate (CCOMn).



	1- OLT up-stream	2- Effluent	3- Effluent	4- Effluent	5- Effluent	6- Effluent	7- Effluent	8- Effluent	9- Effluent	10- Effluent	11- Effluent	12- Effluent	13-OLT down-stream
■ Calculated values	3.470	3.470	3.506	3.510	3.511	3.656	3.656	4.739	4.738	4.737	4.736	4.736	4.736
□ Measured values	3.470	3.470	5.830	4.710	5.310	18.600	20.050	28.640	4.030	3.980	4.710	13.150	5.030
■ Effluent flow-rate	5300.0	485.0	88.0	25.0	3.0	57.0	0.0	270.0	5.0	13.0	180.0	0.1	6400.0
■ Calc. Olt flow-rate	5300.0	5785.0	5873.0	5898.0	5901.0	5958.0	5958.0	6228.0	6233.0	6246.0	6426.0	6426.1	6426.1

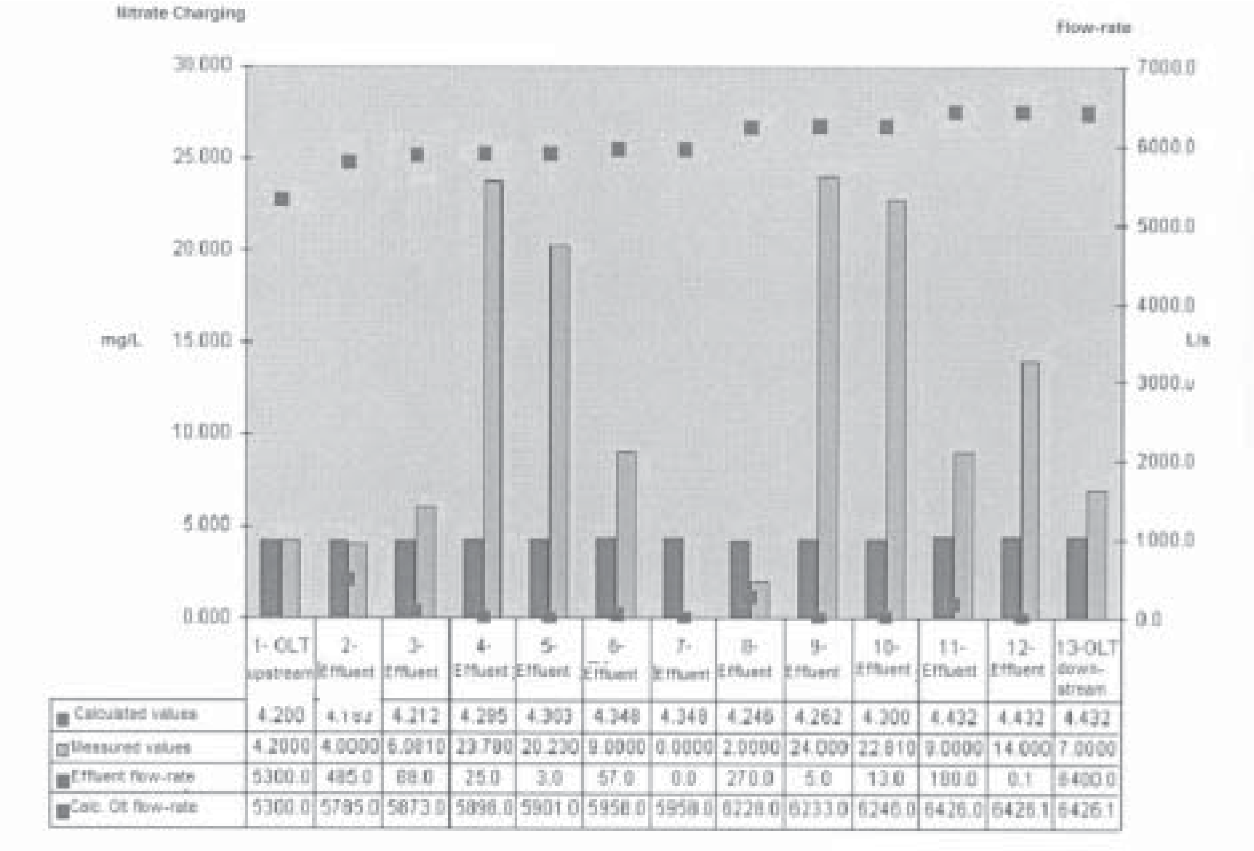
Fig. 3. Chemical oxygen consumption (CCOMn) modeled on the course of river Olt near Miercurea Ciuc



	1- OLT up-stream	2- Effluent	3- Effluent	4- Effluent	5- Effluent	6- Effluent	7- Effluent	8- Effluent	9- Effluent	10- Effluent	11- Effluent	12- Effluent	13-OLT down-stream
■ Calculated values	0.075	0.079	0.096	0.117	0.118	0.207	0.207	0.697	0.700	0.710	0.724	0.724	0.724
□ Measured values	0.075	0.128	1.210	4.949	1.473	9.490	5.120	11.580	5.120	5.260	1.220	4.030	0.800
■ Effluent flow-rate	5300.0	485.0	88.0	25.0	3.0	57.0	0.0	270.0	5.0	13.0	180.0	0.1	6400.0
■ Calc. Olt flow-rate	5300.0	5785.0	5873.0	5898.0	5901.0	5958.0	5958.0	6228.0	6233.0	6246.0	6426.0	6426.1	6426.1

Fig.4. Concentration of NH_4^+ ion modeled on the course of river Olt near Miercurea Ciuc

Fig .5 The profile of NH_4^+ ion charging near Miercurea Ciuc in online GIS environment



	1- OLT upstream	2- Effluent	3- Effluent	4- Effluent	5- Effluent	6- Effluent	7- Effluent	8- Effluent	9- Effluent	10- Effluent	11- Effluent	12- Effluent	13-OLT down-stream
■ Calculated values	4.200	[illegible]	4.212	4.285	4.303	4.348	4.348	4.246	4.262	4.300	4.432	4.432	4.432
▨ Measured values	4.2000	4.0000	6.0810	23.780	20.230	9.0000	0.0000	2.0000	24.000	22.810	9.0000	14.000	7.0000
■ Effluent flow-rate	5300.0	485.0	88.0	25.0	3.0	57.0	0.0	270.0	5.0	13.0	180.0	0.1	6400.0
■ Calc. Olt flow-rate	5300.0	5785.0	5873.0	5898.0	5901.0	5958.0	5958.0	6228.0	6233.0	6246.0	6426.0	6426.1	6426.1

Fig.6. Concentration of nitrates, modeled on the course of river Olt near Miercurea Ciuc

Table 2

INITIAL AND CALCULATED DATA

Critical distance	Water speed	h	D_0	L_0	D_{cr}	D_{27400}		L_{27400}	
						calculated	measured	calculated	measured
m	m/s	m	mgO/L	mgO/L	mgO/L	mgO/L	mgO/L	mgO/L	mgO/L
4600	0,3	1,2	2,49	4,84	2,719	1,2841	0,87	0,3302	0,94

Results and discussions

The obtained results are presented as graphical tables and maps, using online GIS environment (fig. 3-8).

Applying the Streeter Phelps model downstream to Miercurea Ciuc, the obtained results are presented in table 2.

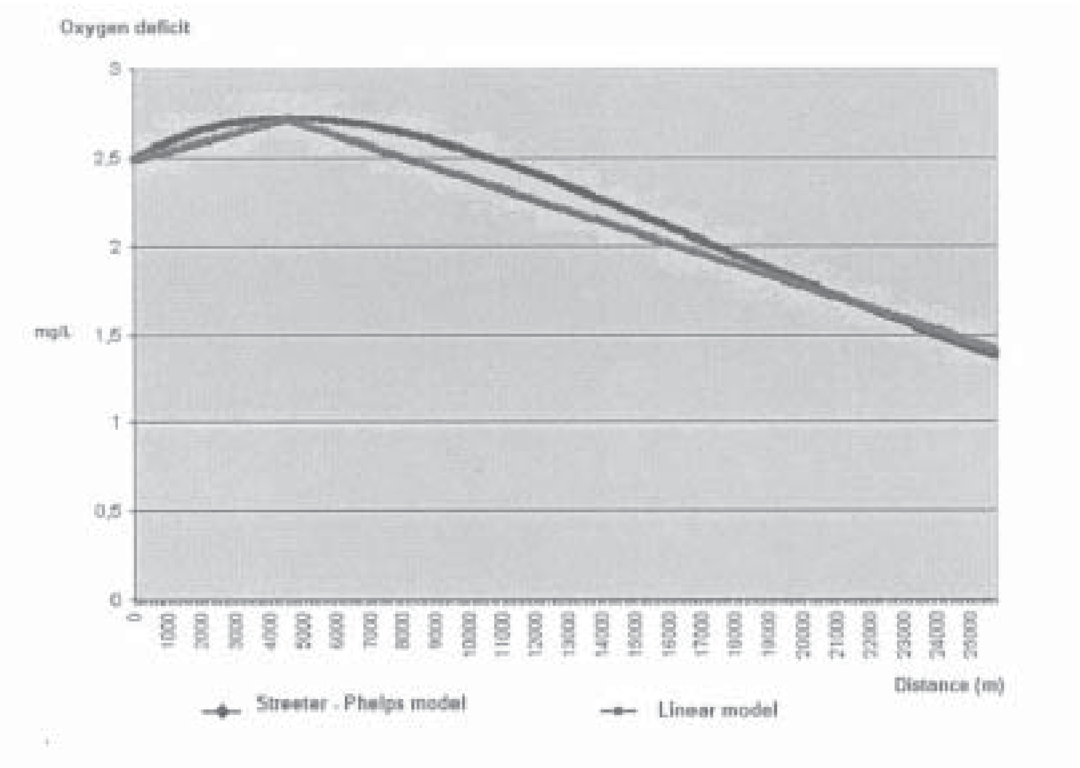


Fig .7. The profile of oxygen deficit downstream of Miecurea Ciuc, obtained with Streeter - Phelps model, compared with the model obtained through linear interpolation

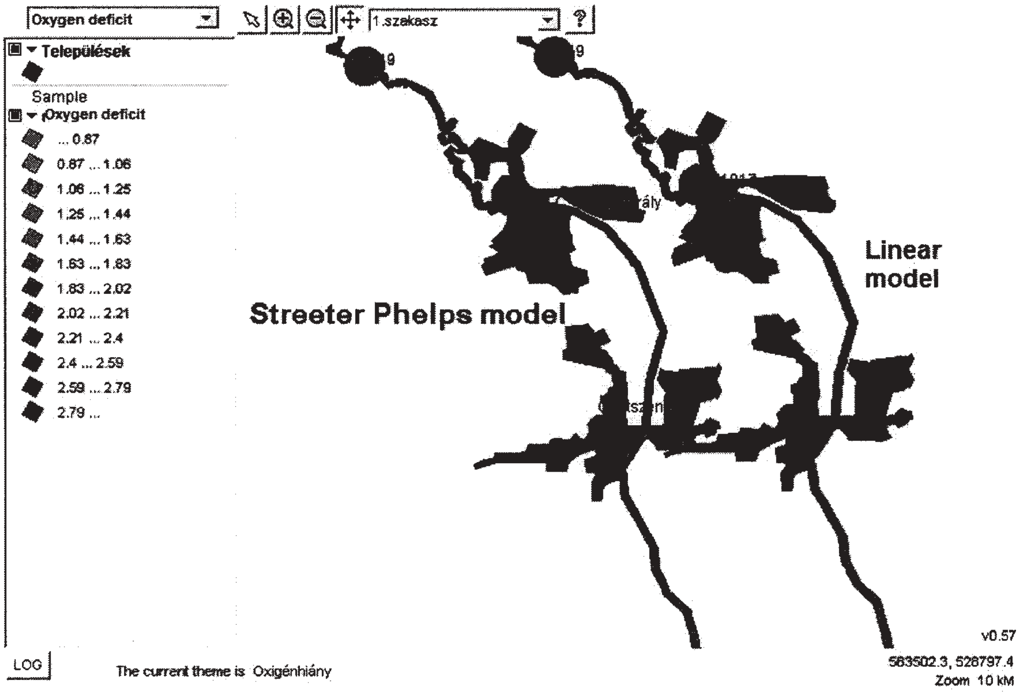


Fig. 8. Comparative representation in online GIS environment of profiles of oxygen deficit obtained with Streeter - Phelps model (at left) and linear interpolation (at right)

Conclusions

The operation with the simple mixing model, without considering the chemical reactions proved to be efficient in case of modeling the oxydability, and in case of ammonium and chlorine ions charging, the differences between calculated values and determined values being insignificant. In case of nitrate charging the differences between calculated values and determined ones were significant, the error being over 50%. In case of modeling the profile of dissolved oxygen deficit, for the simplification of model , important parameters intervening with appreciable weight has been neglected. Although there are differences between the measurement results and the results obtained using mathematical models, these differences are situated in reasonable limits, 0.5 mgL^{-1} for oxygen deficit, and 0.7 mgL^{-1} oxygen for CCOMn. In conclusion it is justified the use of mathematic model for the calculation of dissolved oxygen regime in Olt river with condition of more precise determination of the flow regime and of the input parameters.

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