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Water Quality Modeling of the Nitra River (Slovakia): A Comparison of two Models

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ABSTRACT

IIASA's Water Resources Project deals with the development of least-cost water quality control strategies for degraded river basins in Central and Eastern Europe, which is an important issue due to the lack of financial resources available for environmental management. The Nitra River basin in Slovakia serves as a case study with collaborative research by IIASA, the Water Research Institute (VUVH, Bratislava), and the Vah River Basin Authority from Slovakia. The Nitra River receives large loads of partially or untreated wastewater mostly of municipal origin. The present paper compares the results of two relatively complex water quality models implemented on the Nitra River (which are important elements of developing ambient criteria based control strategies): QUAL2E and RMA2/4q. The well-known QUAL2E is a result of systematic developments by the US EPA over the past twenty years and solves the steady-state advection-diffusion equation for temperature, biochemical oxygen demand (BOD), dissolved oxygen (DO), nitrogen and phosphorus forms, and algae. RMA2/4q was initially developed in the mid-1970s for the US Army Corps of Engineers and has been consistently maintained and extended by Resource Management Associates and researchers at the University of California - Davis. State variables and reaction terms of RMA4q is practically identical to QUAL2E, but it offers more details in computing the flow and physical transport. For the current case, this model was used to solve the unsteady one-dimensional hydrodynamic and advection-diffusion equations for temperature, BOD, and DO. Model comparisons and calibration results showed similar BOD decay rates. DO was overestimated in both models in comparison with DO observations when the O'Connor-Dobbins reaeration method was used. Fixed reaeration coefficients gave better results, with QUAL2E having a larger value than for RMA4q. Sediment oxygen demand was included in the QUAL2E simulations and resulted in better agreement with observed data. Additional data requirements for improved understanding of water quality processes in the Nitra River system are discussed.

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WATER QUALITY MODELING OF THE NITRA RIVER (SLOVAKIA): A COMPARISON OF TWO MODELS

Stephen A. Breithaupt[1]
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1. INTRODUCTION

One of the focuses of IIASA's Water Resources Project is the management of degraded river basins in the Central and Eastern European (CEE) countries. These problems are frequently characterized by point and non-point source pollution, traditional and priority pollutants, and contamination of sediment and soil. On the financial side the lack or scarcity of resources available for management is typical and stems from the ongoing political and economic transition of these countries. The latter feature calls for the development of short-term least-cost regional control strategies which can be further developed as the economic conditions improve. Such a policy requires the definition of water quality goals according to water uses and in terms of ambient standards (or regionally variable effluent standards). In turn, these call for application of water quality models describing the impact of emissions on the quality of receiving waters. Such a philosophy deviates significantly from the practice of North America and Western Europe, where frequently uniform, technology based effluent standards are used. This uniform emission reduction policy (often based on the best-available-technology (BAT)) does not explicitly consider water quality impacts and economic implications; the overall assumption is that a budget is available for the implementation and that strict effluent standards will insure beneficial uses are maintained. However, such a strategy is frequently expensive; hence it may not be feasible in the short-run for the CEE countries (Somlyódy and Paulsen, 1992).

As a case study for the development of a least-cost approach, the Nitra River basin in Slovakia (Figure 1) has been selected for collaborative research between IIASA, the Water Research Institute (VUVH, Bratislava) and the Vah River Basin Authority. It is a tributary of the Vah River, that is itself a tributary of the Danube. The Nitra River has a catchment area of about 5.000 km^2, with approximately 600.000 inhabitants. The length of the river is 171 km. Its mean annual flow measured at the river's mouth is 25 m^3/s, and its summer low flow 3-5 m^3/s (Somlyódy and Paulsen, 1992).

Numerous point source emissions affect the water quality of the Nitra River (Figure 1) as a result of inadequate treatment, while diffuse loads are unimportant. Municipal emissions play a dominant role. River water quality is among the worst in Slovakia. From a comprehensive water quality sampling campaign performed in August 1992 under low flow conditions (see Masliev et al., 1994 for details) the spatial minimum DO in the Nitra River was about 2 mg/L, while the maximum BOD value was 13.2 mg/L. Maximum concentrations for other constituents were as follows: NH_3-N 6.90 mg /L, NO_3-N 1.90 mg /L and total phosphorus 1.01 mg /L. The river serves primarily for wastewater disposal, with withdrawal for industrial and agricultural purposes (Somlyódy et al., 1993).

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2. STUDY OBJECTIVES

As stated above, one aspect of the least-cost approach is evaluating impact of emissions on ambient water quality. This can be done by using water quality models which allow - under the limitations of these methods - analysis of existing and future situations with different treatment strategies and environmental conditions. Models require for calibration and validation purposes varying amounts of information about the existing system, which is proportional to its complexity. In turn, model complexity depends on the objective of the study. The understanding of basic processes requires more detailed models, while for management purposes generally simpler approaches are used. Data to drive complex models typically includes field and laboratory data of the physical, chemical, and biological characteristics of the system. Models for planning purposes (subsequent to calibration and validation) often use "generated" data characterizing the "design or critical scenario".

The development of a set of relatively simple models for the Nitra River in an uncertainty analysis framework is discussed in Masliev and Somlyódy (1993). A management optimization model based on dynamic programming and its application are presented in Somlyódy et al. (1993). The focus of the present paper is on implementing two relatively complex water quality simulation models for the Nitra River system (utilizing the data collected from the August 1992 sampling campaign).

3. METHODOLOGY

3.1. Water Quality Models Used

The two water quality models applied to the Nitra River are QUAL2E and RMA4q. Both models describe dissolved oxygen, nutrients, and suspended algae of rivers. State variables and reaction terms are nearly identical; RMA4q also incorporates attached (benthic) algae in addition to DO, CBOD, nitrogen and phosphorus forms, and phytoplankton (see Table 1 and Figures 2 and 3). The hydraulic and transport components of the two models as well as their implementation are significantly different (Table 1). QUAL2E is supported by the United States Environmental Protection Agency (EPA). It has a simplified hydraulic submodel which solves Manning's equation for velocity and depth, and it solves the advection-diffusion equations for steady-state flow (Brown and Barnwell, 1987). RMA4q is a dynamic water quality model (King, 1990), which derives its unsteady velocity profiles from a companion model (RMA2) capable of handling both, 1D and 2D (depth averaged) flows[3]. It also solves the unsteady advection-diffusion equations.

QUAL2E is the most recent version of a series of water quality models initially begun in the early 1970s. It has been widely used for waste load studies and has become a "standard" to which other models are compared. Its main utility lies in its relative ease of application and continued user-support by the US EPA. Figure 2 is a diagram of interactions present in QUAL2E.

[3] Resource Management Associates (RMA, Suisun City, CA) originally developed RMA2 for the US Army Corps of Engineers to examine flow conditions before and after construction of Lower Granite Dam on the Snake River, Washington. RMA4q was developed subsequently to RMA2 by RMA and researchers at the University of California, Davis. The numbering system for the various models refer only to the development sequence.

RMA4q has been in existence since the late 1970s. As noted before, its important features are the time dependent solution of 1D and 2D advection-diffusion equations and the superior ability to describe the physical features of the system with variable detail as needed by the user. It is currently being enhanced to include additional water quality processes, which will allow application to a wider variety of water quality problems than is possible with QUAL2E. Figure 3 is a diagram of interactions present in RMA4q. The differences between QUAL2E and the RMA models in handling geometry and its affects on transport are compared in the discussions which follow.

Table 1: Comparison of model structure of QUAL2E and RMA-4.

QUAL2E	RMA-4
1D dendritic systems	1D networks & 2D (vertically averaged) systems
Reaches with uniform characteristics	Detailed spatial description: system geometry water quality processes
Steady-state flows	Steady- and unsteady-state flows[1]
Steady-state and dynamic water quality	Dynamic water quality
Water quality interactions DO-BOD Phosphorus Nitrogen Phytoplankton	Water quality interactions DO-BOD Phosphorus Nitrogen Phytoplankton Attached algae
Conservative and non-conservative constituents	Conservative and non-conservative constituents
Finite difference solution of advection-diffusion equation	Finite element solution of advection-diffusion equations

[1] Velocities and depths are provided by RMA-2, a companion model.

3.2. Model Implementation

3.2.1. River Geometry

Physics-based water quality models must consider the system geometry, which influence the velocities and depths of flow. Both QUAL2E and RMA2/4q solve 1D systems and need bottom width and side slopes. Additionally, QUAL2E needs the bed slope, while RMA2/4q requires a bottom elevation. A major difference between the two models is the level of detail that can be described. QUAL2E requires an averaged geometry over a river reach. RMA2/4q needs geometry data at corner nodes[4]. The former allows easier application while sacrificing detail. The latter provides the capability to describe the river system in detail but is more difficult to implement.

For the whole river, 299 cross-sections have been surveyed. However, the 25-26 August 1992 sampling program began at river kilometer 132.5, and this was used as the headwater

[4] Parabolic, isoparametric elements are used for the 1D solution, giving two corner nodes and one mid-side node per element.

for both water quality models. This gave 266 cross-sections that were analyzed, since neither model uses cross-sectional data directly.

3.2.1.1. QUAL2E

The analysis for QUAL2E geometry input was as follows. The minimum depths for each cross-section were determined and plotted against river kilometer to give a profile of the Nitra River. The profile was examined for changes in bed slope, which was used initially to define river reaches. Each cross-section of a river reach was examined for similarity. If dissimilar cross-sections were found, the reach was subdivided. Also, reaches were subdivided to meet a program limitation of 20 computational elements per reach. For convenience, computational element lengths were set at one kilometer. Finally, a representative cross-section for each reach was used to as input to the model. The model reaches used were as follows:

Table 2: Reach ranges used in QUAL2E for modeling the Nitra River.

QUAL2E Model Reach Locations	River Kilometer Range
1	133.0 - 130.0
2	130.0 - 125.0
3	125.0 - 113.0
4	113.0 - 102.0
5	102.0 - 93.0
6	93.0 - 82.0
7	82.0 - 72.0
8	72.0 - 52.0
9	52.0 - 32.0
10	32.0 - 12.0
11	12.0 - 7.0

The model reaches referred to later in the report are given by this table.

3.2.1.2 RMA2/4q

For the 1D case, both RMA2 and RMA4q require the system geometry to be input as trapezoidal cross-sections. As with QUAL2E, this required taking observed point data for the cross-sections and approximating them as trapezoids, giving sideslopes for each bank and a bottom width. Initially, it was anticipated that all 266 cross-sections would be included; however, due to the dramatic changes in bed width and side slope between adjacent cross-sections, including cross-sectional data at half-kilometer intervals proved impractical. After refinement (described below), the number of elements describing the system would likely have been on the order of 10,000. This was considered an unreasonably large number for the current application; since the computational time would have become significant (likely on the order of days). Without refinement, significant oscillations of mass develop where drastic changes in geometry (bed slope, side slope, or bottom width) exist.

The refinement process involves adding more elements (and nodes) in regions where conditions (in this case geometry) are changing rapidly. This reduces the step size to improve the solution's convergence. Flow continuity is checked to insure oscillations are insignificant. This is done using a constant headwater flow (0.608 m^3/s in the case of the Nitra River), with no tributary or incremental inflows. The objective is to obtain flows throughout the system that deviated no more than ±5% from the headwater flow. This is accomplished by adding more elements in sections where the ±5% criteria were not met.

Ultimately, data were selected at approximately five kilometer intervals from the 266 observed cross-sections, giving a total of 28 trapezoidal cross-sections. Initially there were 27 elements, but refining elements to improve flow continuity, as described above, increased that to near 700.

3.2.2 Hydrologic Balance

The data available for the Nitra River were from river stations, some tributaries, and point source emitters. The river was subject to additional distributed inflows (e.g. through ground water and nonpoint source emissions) which were not monitored. It was necessary to account for these unmeasured flows to set water and (mass) balances between sampling points. To assist in these water balances, the concurrent balance of a conservative constituent was also made. Use of a conservative constituent provides a constraint on estimates of flow. For example, if over a river segment all measured flows indicate a net increase in flow and the conservative constituent's concentration, this can only be accounted for by some unmeasured inflow to that segment. It is frequently useful to account for these unknown inflows, especially if their magnitude is large.

Several approaches to the hydrologic balance are possible. For the Nitra River, a minimization approach was used and is described below.

For the rth river segment between two flow measurement points, the water and conservative constituent balances were set as follows:

$$Q_u = Q_o - Q_i - Q_t - Q_w, \text{ and}$$

$$M_r = \{C_oQ_o - C_iQ_i - C_tQ_t - C_wQ_w\}|_r - C_u\{Q_u\}|_r,$$

where Q = flow (m^3/s), C = total dissolved solids (TDS) concentration(mg/L), a conservative constituent, M_r = residual mass in the hydrologic segment. The indices are u = unknown, o = outflow, i = inflow, t = tributary, and w = wastewater. Q_u is the residual of flow over the hydrologic segment. C_u is found as the concentration which gives the minimum sum over the system of the squares of the mass residuals, i.e.,

$$C_u = \left\{ C_u \middle| \min \sum_{r=1}^{n} M_r^{\ 2} \right\},$$

where r = hydrologic segment number and n = total number of hydrologic segments.

The flows for 25 and 26 August 1992 used in the hydrologic balance are listed in Table A1. The unknown flow and concentration were considered to account for incremental inflows to the river. Hence, the unknown concentration was the value that gave the best overall fit to the observed TDS data. The estimated residual inflow concentration using the method described above was 1525 mg/L. The incremental flow results are given in Table 3.

Table 3: Hydrologic segment ranges and computed residual flows to the Nitra River.

Hydrologic Segment Number	River Kilometer Range	Incremental Flow (m^3/s)
1	132.5 - 123.8	0.217
2	123.8- 115.7	0.370
3	115.7 - 105.5	0.083
4	105.5 - 101.6	0.000
5	101.6 - 98.2	0.000
6	98.2 - 91.1	0.000
7	91.1 - 65.25	0.000
8	65.25 - 47.8	0.000
9	47.8 - 14.5	0.000
10	14.5 - 6.5	-0.308

In the seventh hydrologic segment it was also necessary to estimate withdrawal and tributary flows. Examination of the flow and TDS concentration at the endpoints of this segment revealed a decreasing flow (3.625 to 1.68 m^3/s) with a decreasing concentration (1202 to 962 mg/L). With a conservative constituent, concentration can only decrease by dilution, but since flow is decreasing, there must have been another source with a lower concentration. It was possible that incremental inflow could account for the dilution if its TDS concentration were small. However, from the hydrologic balance above, the incremental TDS concentration was not small (1525 mg/L). In segment seven there were three weirs and a tributary (Radosinka) at river kilometer 67.0, whose flows were not measured during the sampling campaign. One weir was neglected for this analysis, and it was assumed that withdrawals from the other two were proportional to their volume. An estimate of the tributary TDS concentration was made by using the average of upstream tributary concentrations (455 $\pm$ 34 mg/l). Using this information, estimates of each flow was made and are listed in Table A1. These estimated withdrawals and tributary inflows are significant portions of the river flow; their flows need to be measured in any future sampling campaigns.

3.2.3. Initial and Boundary Conditions

The flow and quality boundary conditions used for both models are listed in Tables A1 and A2. For most of the wastewater emissions no dissolved oxygen (DO) data were available. For this a zero DO value was assumed in agreement with the overall (low) level of treatment. The constituent concentrations of the tributary at river kilometer 67.0 were estimated in the same manner as TDS, i.e., by using the average of upstream tributary concentrations.

3.2.3.1. QUAL2E

Since QUAL2E solves the steady-state transport equations, no initial conditions were needed. For flow and constituent boundary conditions it required only the flow, temperature, or concentration at the headwater and load points. For the lower boundary, a constant gradient boundary condition was assumed. The hydrologic segment ranges did not correspond to the model reaches defined by the system geometry (compare tables 2 and 3); hence incremental inflows were linearly distributed over the reaches.

3.2.3.2 RMA2/4q

The initial conditions for RMA2 were a velocity of 0.25 m/s and a depth of 4.0 m. Recall that at the downstream boundary, the Nitra River flows into the Vah River. If the stage of the Vah had been known, the specified elevation boundary condition would have been appropriate through all time. However, after the first time step, the downstream boundary condition was changed from an initial constant head to a stage-flow relation. The constant head boundary condition was needed only to start the solution. The stage-flow boundary condition takes the form $Q = A_1 + A_2(ELEV - E_0)^C$, with Q = flow, A_1, A_2, and C = flow coefficients, ELEV = water surface elevation computed by the model, and E_0 = bed elevation. A_2 and C were computed based on Manning's equation with friction factor equal to 0.03. For this case, the Nitra River was assumed to be a wide, rectangular channel, and the bed slope was computed between the lower two, defined cross-sections. This gave the result of $A_1 = 0.0$, $A_2 = 0.727$, and $C = 1.667$.

The headwater was given a total flow boundary condition equal to 0.608 m^3/s. Tributaries, wastewater emissions, and withdrawals were incorporated into the system as short 'stubs', to which flow boundary conditions were applied at terminal nodes. The flow values used are listed in Table A1. The direction of flow depended on whether it was an inflow or outflow from the system. Concentration boundary conditions are given in Table A2 and were applied at the headwater and terminal nodes of the tributary 'stubs' in RMA4q. All the boundary conditions were held constant over time.

Inflows to elements were determined from the results of the hydrologic balance, so that flows for each hydrologic segment were distributed over the elements in its range. Since element lengths were relatively short, the hydrologic segment data were added as flow per unit length and did not have to be distributed over reaches, as required in QUAL2E. Elemental inflows were held constant over time.

3.3. Calibration

Initially all rates were set to the minimum values of the typical ranges listed in the QUAL2E manual (Brown and Barnwell, 1987). This kept the effect of any process at a minimum and allowed initial examination of simple mass balance effects on water quality. Subsequently, the effects of the rates of processes were modified to account for changes in state variables beyond simple dilution or addition. As an objective measure of the goodness-of-fit between model output and observed data, the residual sum-of-squares was computed for each modeled state variable. The goal was to minimize this residual and keep the rates of processes within the suggested ranges. This general approach was taken with both models for each state variable. Additional details on calibration data are discussed below.

3.3.1. Temperature

Both water quality models have routines which simulate temperature, and use that to correct the rates of other processes. Unfortunately, no climate data were collected for the 25 and 26 August 1992 sampling campaign. This data is necessary to simulate temperature. As a result, it was decided that the most important facet would be for temperature correction and not to accurately simulate observed stream temperature. Climate data were adjusted so the model gave an approximation to the observed temperature data. Required climate data include wet and dry bulb temperature, air pressure, wind speed, and cloud cover. One value for each parameter was applied to the whole river system, though the actual value used differed for each model. This was necessary since QUAL2E used daily averaged values and RMA4q requires time varying data. Since stream temperature varied along the length of the river, climatic data were chosen to give intermediate output values for the whole system.

3.3.2. Biochemical Oxygen Demand and Dissolved Oxygen

Biochemical oxygen demand (BOD) data were collected from the river and emitters during the August 1992 sampling campaign. Dissolved oxygen (DO) was only collected from the river and one emitter (power plant). However, considering the nature of the other emitters, the assumption of zero DO in the discharge is reasonable (as already noted). The river DO was corrected for diurnal fluctuations by fitting diurnal measurements to a sine function and determining the mean DO based on the time and magnitude of a particular observation. This is most appropriately used on steady-state models. However, the sampling times and concentrations were not available for use in dynamic modeling. RMA4q was run to give results at 12 Noon, which was considered an intermediate time during the actual sampling period.

Since decay and settling rates are both first order, and no data were available on settling, the two were combined into one rate. Hence, during calibration, BOD decay rate (k_1) was adjusted over a range of values, while BOD settling rate was kept at zero. This could cause DO results to be too low, but this was not the case, as shown below.

3.3.3. Nutrients

Nutrient data available for the river and emitters include ammonia (NH_3), nitrite (NO_2), nitrate (NO_3), and phosphorus (PO_4). Unfortunately no organic nitrogen or phosphorus data were available from this sampling campaign. Consequently, concentrations of both organic forms were set to low levels equal to 0.01 mg/L. This was chosen to minimize the effect from hydrolysis of organic forms to inorganic nitrogen or phosphorus. For some of the emitters PO_4 data were not available. Another set of data, reported as total phosphorus, were used in this case[5].

No nutrient (and algae) calibration with the dynamic model (RMA4q) had been completed by the time this report was prepared.

3.3.4. Algae

No algae biomass data are available for the 25. and 26. August 1992 sampling campaign. Thus, only an evaluation of the effect of algae biomass with the given model geometry on nutrient and dissolved oxygen is presented below. The evaluation is based on the residual sum-of-squares between model output and observed nutrient data.

[5] A regression between the two sets of values yielded the relation $P_{tot} = 0.946*(P_{dis})+0.102$, $r^2 = 0.61$, if one "outlier" is eliminated.

4. MODEL RESULTS AND DISCUSSION

4.1. QUAL2E

4.1.1. Flow and Total Soluble Solids

Figures 4 and 5 illustrate the results for flow and soluble solids. Computed flow matches the observations very closely. Flow increases until around river kilometer 90. At river kilometers 80.7 and 71.9, flow declines dramatically due to withdrawals. The dip around river kilometer 53 is caused by a withdrawal to the Stara Nitra and subsequent addition of a wastewater emission. It should be recognized that TDS is not modeled, but reflects the least squares approach taken in the hydrologic balance. The large TDS increase at river kilometer 130.6 results from a wastewater emission, while the decline at river kilometer 67.0 results from dilution by the tributary.

Respective, velocity and depth profiles are shown in Figures 6 and 7. These illustrate the geometry and hydrologic conditions applied to the model. In general, the velocities decreased in the downstream direction, while depth remained relatively constant. For the latter, an exception occurs over the fifth reach (river kilometer 102 - 93), where there was an increase in flow resulting in a discontinuity between reaches upstream and downstream of it. Velocity also increased in this reach. These results have a consequence for dissolved oxygen computations.

4.1.2. Temperature

The climate data used for QUAL2E were as follows: dust attenuation coefficient = 0.06, cloudiness = 0.0 tenths, dry bulb temperature 25.0 °C, wet bulb temperature 20.0 °C, barometric pressure = 1000 mb, and wind speed = 0.0 m/s. Temperature results reflect the steady-state solution and the climate conditions applied to the whole system (Figure 8). The water temperature results were relatively constant over the whole system. However, the observations varied over both the system length and the daily cycle. Steady-state solutions do not reflect the latter influence. It can be seen that any perturbation by inflows rapidly returns to an equilibrium temperature, which is constant over the system's length; since the climate conditions are uniform as well. As stated above, the purpose for simulating temperature was solely for temperature correction, but it also had consequences for dissolved oxygen results. These are discussed below.

4.1.3. Biochemical Oxygen Demand

In hydrologic segment 8 (river kilometer 65.25 - 47.8), if one considers a mass balance of all measured loads, including the Nitra municipal wastewater treatment plant (MWWTP) discharge[6] to the Nitra River, the resulting river concentration immediately after mixing with the Nitra discharge would have been about 41 mg/L. 32 mg/L BOD would have been expected if only the Nitra MWWTP is considered and the upstream concentration equal to zero. The observed BOD level at river kilometer 47.8 (4.7 km downstream from the Nitra MWWTP emission point) was 13.2 mg/L BOD. If the observed value was true, then decay rates of 4.36/day (3.21/day for the zero upstream concentration case) would have been

[6] It has the largest mass discharge of all the emitters measured, with a flow of 0.381 m^3/s, BOD concentration of 138 mg/L, and BOD mass discharge equal to 4,545 kg/day.

expected for each respective concentration[7]. The former value is greater than the maximum in the range of typical values listed in Brown and Barnwell (1987), i.e., 3.4/day, and the latter is in the upper end.

Examination of the residual sum-of-squares when the BOD decay rate was varied revealed a minimum residual in the neighborhood of k_1 = 2.0/day when all points were considered (Figure 9). If the result at river kilometer 47.8 was ignored, a minimum residual was found around k_1 = 1.0/day. These are significantly smaller than that found above. Considering the analysis above, it was appropriate to exclude the BOD value at river kilometer 47.8 and to take k_1 = 1.0/day. The model results for BOD with this decay rate is shown in Figure 10. The sawtooth shape results from the BOD loads from emissions.

While this was the "best fit" to the data, it is questionable whether it is acceptable. Several factors not considered in this analysis that draw into question the accuracy of the result are: (1) the decay rate may vary over the system in response to microflora and fauna changes that are responsible for organic decay; (2) the variability of observed data may be significant; (3) benthic sources and sinks may be influential. Additional research is necessary to determine their importance.

4.1.4. Sensitivity of Nutrient State Variables to Algae Growth

No algal biomass data were available to evaluate the effect suspended algae had on other state variables, so a range of headwater algae concentrations were input to QUAL2E. Also, for comparison, algae growth parameter values were set in one case to give maximum algae growth and in the other to give a minimum, with parameter values falling within the ranges listed in Brown and Barnwell (1987). These were μ = 1.0/day, K_N = 0.30 mgN/L, and K_P = 0.05 mgP/L for the minimum case and μ = 3.0/day, K_N = 0.01 mgN/L, and K_P = 0.001 mgP/L for the maximum case. The effect on nutrients and DO is illustrated in the following tables.

Table 5: Sum-of-square residual values of constituents affected by algae growth with parameter values giving *minimum* algae growth. (NH_3 preference = 0.5).

Headwater Algae Concentration (µg chlorophyll *a*/L)	Residual Sum-of Squares			
	NH_3	NO_3	PO_4	DO
0.1	60.13	3.14	0.801	92.91
1.0	60.13	3.14	0.801	92.91
10.0	60.13	3.14	0.801	92.98

[7] Computations assumed first-order decay and residence times as computed by QUAL2E.

Table 6: Sum-of-square residual values of constituents affected by algae growth with parameter values giving *maximum* algae growth (NH_3 preference = 0.5).

Headwater Algae Concentration (μg chlorophyll *a*/L)	Residual Sum-of Squares			
	NH_3	NO_3	PO_4	DO
0.1	60.13	3.14	0.801	92.91
1.0	60.13	3.14	0.801	92.98
10.0	60.21	3.14	0.801	93.09

It can be seen that algae growth produced by QUAL2E and the geometry describing the Nitra River does not significantly affect nutrient and DO levels. However, there are indications from the 25. and 26. August data set which suggest there is algal growth occurring, particularly in the lower river. Figure 11 shows the diurnal variation in dissolved oxygen at two stations of the Nitra River. The changes in DO at river kilometer 14.5 indicate the autochthonous production of oxygen by algae. One possible cause for QUAL2E not producing algae is the modeled residence times are too small. Residence times can be increased either by a lower flow or by changes in geometry, which decrease the velocity. Since the flows were measured, the geometry of the lower system must be examined.

For the whole river, a hydraulic residence time of 4.37 days was obtained. A plot of the residence time for each computational element (1 km long) shows increasing residence times from the headwater until approximately river kilometer 70, where it becomes relatively constant and greater than double the minimum seen in the upper river (Figure 12). The increased residence times should help promote algal growth by giving long exposure time to nutrients. The spike at river kilometer 54 was caused by a withdrawal and is followed immediately by an emission, so that the flow, velocity, and residence times were changed abruptly. The relative constancy of the residence time below river kilometer 70 is a result of constant geometry imposed by QUAL2E reach description and relatively constant flow. The geometry was held fixed since examination revealed the representative cross-sections of the lower reaches to be similar.

Another effect of the hydrology and geometry of the system was shallow depths with low flows. The rate term for algae settling used in the model is

$$\frac{dA}{dt} = \frac{\sigma_s}{d} A,$$

where A = algae concentration (mg/L), σ_s = settling rate of algae (m/day), and d = depth (m). Shallow depths increased the loss rate of algae due to settling. By setting the rate coefficient to zero, a decrease in the residual sum-of-squares for the nutrients and an increase in DO residual sum-of-squares can be seen (Table 7). This indicates increased algal concentrations, greater uptake of nutrients, and increased oxygen production. Figure 13 shows that settling rates are highest at the upper and lower ends of the system. The lowest values in the neighborhood of river kilometer 100 are a result of the geometry of reach 5, with relatively narrow width and deeper depths. Note that the value for σ_s = 0.5 m/day is the minimum of the typical range listed in Brown and Barnwell (1987), while σ_s = 0 was necessary to produce even minimal growth in the lower river.

Table 7: Sum-of-square residual values of constituents affected by algae growth with parameter values giving *maximum* algae growth (NH_3 preference = 0.5; headwater concentration = 10.0 μg chlorophyll *a*/L).

Algae settling rate (m/day)	Residual Sum-of Squares			
	NH_3	NO_3	PO_4	DO
0.5	60.21	3.14	0.801	93.09
0.0	56.35	2.80	0.578	94.84

The analyses of nutrient and dissolved oxygen sensitivity to algae suggest that the typical ranges of process rates are not necessarily applicable to the Nitra River situation. However, without algae biomass data it was not possible to determine from the existent observations what rates would be appropriate. Hence, intermediate values from the typical ranges for the rates and coefficients were selected. The values used for algal growth were as follows: maximum specific growth rate = 2.0/day, respiration rate = 0.05/day, settling rate = 0.01/day, nitrogen half-saturation constant = 0.15 mg/L, phosphorus half-saturation constant = 0.025 mg/L, light half-saturation constant = 0.50 intensity/min., and a ratio of 50 μg chlorophyll *a*/mg algae. The headwater algae concentration was set at 0.1 μg chlorophyll *a*/L. Figure 14 illustrates the algae concentrations obtained using the parameter values listed above. While concentration increases in the downstream direction, the maximum of 0.3 μg chlorophyll *a*/L is still very small.

4.1.5. Dissolved Oxygen

Dissolved oxygen is influenced by temperature, BOD, reaeration rate, sediment oxygen demand (SOD), and net algal growth. The BOD decay rate was based on a fit with the river data, discussed above. The inputs for the temperature routines were set to provide an intermediate value between observations, also discussed above. In QUAL2E reaeration can be handled by several methods. Each of them considers the depth and average velocity of the river and computes a reaeration coefficient. O'Conner and Dobbins developed an equation which is suitable for low velocity streams (Brown and Caldwell, 1987). Their reaeration rate equation takes the following form,

$$\frac{dO_2}{dt} = k_2\left[O_2^* - O_2\right]$$

$$k_2 = \frac{12.96u^{0.5}}{d^{1.5}}$$

with k_2 = reaeration coefficient, u = velocity, d = depth, O_2^* = oxygen saturation concentration, and O_2 = oxygen concentration.

The DO results from QUAL2E are displayed in Figure 15. The shape of the curve is approximately the same as corrected observations, but the magnitude is too high. Note the decrease in dissolved oxygen at river kilometer 100. This is a consequence of the deeper flow of reach 5. As can be seen in the reaeration rate equation, larger depths result in smaller reaeration rates.

Another approach used was to specify reaeration coefficients, rather than have them computed from velocity and depth. The effect on the residual sum-of-squares is shown in Table 8. Improved results were obtained over those from the O'Connor-Dobbins formulation. The value of k_2 = 7.5/day was chosen; since it gave a better match for the upper river between observations and model output than did the O'Connor-Dobbins method (and other k_2 values). A sampling of k_2 values computed using the O'Connor-Dobbins formulation with velocities and depths obtain along the river from QUAL2E gave a range from 3.06 to 42.2/day, with most values near 20/day. The value of 7.5/day is much lower than this central tendency, indicating the O'Connor-Dobbins method over estimates reaeration.

Table 8: Residual sum-of-squares results for different reaeration coefficients.

Reaeration Method	Residual Sum-of Squares DO
O'Connor-Dobbins	87.92
$k_2 = 5.0$	42.40
$k_2 = 7.5$[§]	50.11
$k_2 = 10.0$	65.04

[§] Chosen value since it gave a better match to the upper river than the other k_2 values.

To improve the fit further, sediment oxygen demand was added to the lower reaches (reaches 5-8; river kilometer 102 - 52). Table 9 shows a comparison of various combinations of SOD examined, but there is no independent data to support the selection of these values. Thomann and Mueller (1987) report a maximum value of 10 g/m^2-day for municipal sewage sludge in the vicinity of an outfall, which is less than the 20 and 15 g/m^2-day used in QUAL2E used to minimize the sum of squared residuals. The QUAL2E DO results with the chosen reaeration coefficient and SODs are illustrated in Figure 15. Overall, the addition of these processes gave an improved fit. Note that DO from algal growth were insignificant at the algal levels output from the model.

Table 9: Residual sum-of-squares results for different sediment oxygen demand (SOD) values.

SOD (reach #) (g/m^2-day)	Residual Sum-of Squares DO
0.0(none)	50.11
10(5)	41.77
10(5,6)	37.42
15(5,6)	33.99
20(5),15(6),10(7)	29.10
20(5),15(6),10(7),20(8)	37.12
20(5),15(6),10(7),5(8)[¥]	17.41

[¥] Chosen combination.

4.1.6. Nutrients

The calibration of nutrient models without organic phosphorus or nitrogen is problematic. For dissolved phosphorus, it makes the process simply a mass balance exercise, since algae uptake cannot also be adjusted. Without some constrain on the concentration of algae, it was impossible to ascertain the significance of algal uptake. However, examination of QUAL2E results in comparison with observations (Figure 16) suggests there may be significant algal uptake of dissolved phosphorus, especially below river kilometer 55; this is the only sink for dissolved phosphorus in QUAL2E.

For the inorganic nitrogen components (NH_3, NO_2, and NO_3), the problem was compounded by the interaction between the different forms. Examination of the ammonia results (Figure 17a) showed that it was underestimated in the upper river, suggesting another source. A probable candidate would be hydrolysis of organic nitrogen, but there were no observations with which to estimate hydrolysis rates. In the lower river, ammonia was overestimated; a possible sink would be uptake by algae. Nitrite results agreed well, except near the lower end of the system (Figure 17b). Nitrate was overestimated throughout the river system (Figure 17c). The only sink for nitrate in QUAL2E is algal uptake, as for ammonia. The results for NH_3 and NO_3 suggest the presence of algal growth, especially in the lower river, as also indicated by diurnal DO observations.

If an ammonia source was added to the upper river, e.g. benthic source, the ammonia results in the upper river were closer to observations (Table 10). Adjustment of the ammonia oxidation rate within the typical range improved the fit throughout the river (Table 11 and Figure 18a). The same was true for nitrite, by changing the rate from 0.2 to 2.0/day the residual sum-of-squares went from 10.0/day to 0.4/day (Figure 18b). However, the end product of ammonia and nitrite oxidation was nitrate, so the whole process further exacerbated problems, i.e., overestimating NO_3 concentrations (Figure 18c). Since an ammonia source was needed in the upper river, either from hydrolysis of organic nitrogen or a benthic source, it was inevitable that an accumulation of nitrate would result. As stated above, the only sink for nitrate in QUAL2E is algal uptake.

Table 10: Residual sum-of-squares results for different benthic ammonia release values.

NH_3 release rate (reach #) (mg/m^2-day)	Residual Sum-of Squares NH_3
0.0 (none)	60.21
5.0 (1-9), 0.0 (10-11)	60.0
1000 (1-5), 0.0 (6-11)¥	46.3

¥ Chosen combination.

Table 11: Residual sum-of-squares results for different ammonia oxidation rates.

NH_3 oxidation rate (1/day)	Residual Sum-of Squares NH_3
0.1	46.3
0.5¥	28.3
1.0	30.0

¥ Chosen value.

4.2. RMA4q

Sampling times of flow and water quality data were not available for the 25. and 26. August 1992 data set. This constrains an unsteady hydrodynamic and water quality model by not allowing calibration over a daily hydrologic and solar period. However, the model was run to a transient steady-state to a time corresponding to 12 Noon, so that solar radiation would be indicative of the time when field observations were made. RMA4q was run in this manner to provide some measure of comparison with the available data and to QUAL2E results

4.2.1. Hydrodynamic Simulation

Initial hydrodynamic simulations gave depths somewhat less than 0.5 m throughout the river system. This caused a drawdown upstream of withdrawals, leading to flow acceleration in the vicinity of the withdrawal. Control structures were placed downstream of the three withdrawals to increase water depth to reduce this acceleration and since weirs were known to exist at these locations. To place weirs in the model, a control structure equation is used in the form $Q = A + B\left(H - C\right)^{GAM}$, with Q = flow, A, B, and GAM = coefficients, C = the crest elevation of the control structure, and H = water surface elevation. These structures were assumed to function as broad-crested weirs with height two meters from the bed to the crest (H - C). This depth was selected arbitrarily, since no weir height data were available. Table 12 indicates pertinent information for each weir. The weirs height had to be added gradually during the transient solution of the hydrodynamic equations starting at bed elevation and increasing the height in successive time steps, else the iteration results did not converge.

Table 12: Locations and lengths of weirs added to the Nitra River system geometry.

Location (river kilometer)	Weir Length (m)	B - conveyance factor
80.8	17.8	30.3
72.0	20.0	34.1
53.1	21.3	36.3

The model was run until a transient steady-state was achieved, which required 360 hours. Figures 19 and 20 show the final respective velocity and depths profiles from RMA2. The influence of the weirs can be seen in both figures, with velocity decreasing and depth increasing behind the weirs. In general, velocity decreased in the down stream direction, which is expected since bed slope decreases as well. In the upper reaches, the depth increased in the downstream direction; however, below the weirs, no general trend makes itself apparent. Near the downstream boundary, the depth increased reflecting the boundary condition applied there.

Computed flow (after 360 hours) is illustrated in Figure 21. Small deviations from continuity can be seen, but they are negligible. Since the mass flows used in both QUAL2E and RMA2 were developed from the same hydrologic balance, the results are very similar. (See the discussion on BOD below.)

4.2.2. Water Quality Simulation

As stated above, the velocity and depth results from RMA2 drive the transport model RMA4q. The analyses performed on QUAL2E were also applied to RMA4q. However, this was only done for biochemical oxygen demand (BOD) and dissolved oxygen (DO). Nutrient and algae analysis were not performed due to time constraints on the project. It must be recognized that the results from RMA4q are not daily averaged values, as they are for QUAL2E with its steady-state solution. RMA4q's results are from the dynamic computation of solar heating and mass transport. The results which follow are for the 12 Noon solar hour. Time limitations precluded comparisons with other solar times to ascertain if this was the best for comparison with QUAL2E. For this particular application, where flows are assumed to be at steady-state, a primary difference between the two models emanates from the geometric description of the river, with the RMA models allowing for greater geometric detail.

4.2.2.1. Temperature

Since RMA4q supplied a dynamic solution, the temperature results reflect the daily variation in climatic conditions, transport through the system, and the system's geometry. Climatic conditions were held constant, except for dry bulb temperature and short-wave radiation. The short-wave radiation computation is from the simulation of solar radiation input to the water body; other climatic data are input by the model user. The results are shown in Figure 22. The portion of the system in the neighborhood of river kilometer 120 was seen to have wide variation in temperature (approximately 9°C over a daily period) and is caused by the shallow, low flows used in RMA2/4q.

The temperature dips at river kilometers 80.8, 72.0, and 53.1 resulted from storage behind the weirs at those locations. Daily heating had not raised the temperature in these sections to the level seen in shallower sections, and the advective flux had not yet flushed out cool, stored water. Immediately downstream of each weir it can be seen that temperature increased rapidly, which is expected for relatively shallow depths (see Figure 19). Temperature tends toward equilibrium and will proceed at a faster rate with shallower depths than deep.

4.2.2.2. BOD

BOD decay rate (k_1) was varied over a range from 0.1 to 4.0 day^{-1}. The minimum residual sum of squares between model results and observations was found for k_1 = 1.1 day^{-1} (Figure 23). This is slightly larger than the value found for QUAL2E (1.0 day^{-1}). This is not a significant difference, especially since the change in residual sum of squares in the immediate neighborhood of k_1 =1.1 is small. This of course raises the question of uncertainty in parameter estimates. Improved methods for estimating parameters with field data and handling uncertainty have been discussed in Masliev and Somlyódy (1993).

Using a k_1 = 1.1 day^{-1}, the BOD results shown in Figure 24 were obtained. These are quite similar to those obtained for QUAL2E, and the concerns about applicability of the results expressed for QUAL2E are valid for the RMA4q model as well. The peaks caused by the major BOD loads are slightly greater than seen in the QUAL2E outputs. These result from slightly lower flows in RMA4q than computed from QUAL2E; hence a smaller level of dilution. The difference is likely caused by the significantly different ways the two models compute flow, velocity, and depth. QUAL2E uses Manning's equation to compute velocity and depth, with flow being a mass balance of inputs and withdrawal at each grid point. RMA2 solves the one (and two) dimensional, shallow water equations for depth and velocity. Flow (Q) at each grid point is computed from the relation Q = VA, where V = velocity and A = cross-section area for a given depth and geometry. So, assuming the hydrologic balance is correct, QUAL2E should get the flow balance right, whereas RMA2 is first solving a system of non-linear partial differential equations and then computing flow. Considering the task it is accomplishing, RMA2 does very well. But since the dilution is less in RMA4q, a higher BOD decay rate (k_1) is necessary to reduce the residual sum of squares.

4.2.2.3. Dissolved Oxygen

The O'Connor-Dobbins method was used first to compute the reaeration coefficient (k_2). As for QUAL2E, it overestimated the overall DO level when compared to (corrected) observations of dissolved oxygen (Figure 25). In the upper river, reaeration continues in RMA4q, until the oxygen is nearly at saturation; while for QUAL2E oxygen computed by the O'Connor-Dobbins method had leveled off (Figure 15). This difference is caused by variations in simulated temperature between the models, which influence the saturation oxygen concentrations. RMA4q temperature varies dynamically with the solar cycle. Concentration dips occurred behind weirs where reaeration was reduced and biochemical oxygen demand was still being exerted. Near river kilometer 100, DO dropped but not as much as for QUAL2E; since the depths in this region are smaller for RMA2 than for QUAL2E.

Direct input of k_2 was also examined. Table 13 shows the residual sum of square comparison between computed and observed values. $k_2 = 5.0$ gave the minimum residual. The effect of setting it to a constant can be seen in Figure 25. Reaeration was slowed in comparison with the O'Connor-Dobbins method for sections with relatively high velocity and small depth. The opposite was seen behind the upper two weirs. Sediment oxygen demand was not added to the system as for QUAL2E due to time constraints on the project; however it is expected that improved results could be obtained.

Table 13: Residual sum of squares between values computed by RMA4q and observations. The reaeration coefficient was applied to the whole system.

Reaeration Coefficient (k_2)	Residual Sum of Squares
O'Connor-Dobbins	93.28
2.5	72.34
5.0	55.49
7.5	72.50
10.0	88.35

5. SUMMARY AND CONCLUSIONS

Flows produced by QUAL2E and RMA2/4q were almost exactly the same, but velocity and depth differed in several sections, resulting from the greater density of cross-section data used in RMA2/4q, i.e., cross-sections at 5 kilometer interval were used. Much fewer cross-sections were used for QUAL2E; since reach descriptions required only representative cross-sections.

The water quality results from the two models generally agreed. A minimum of the sum of squared residuals between model output and observations was used for selecting suitable rate coefficient values. Differences between the models were due to ways system geometry was defined and whether a steady-state or dynamic solution was solved. This was particularly true for temperature and dissolved oxygen simulations, where the former was depth dependent, and the latter was both velocity and depth dependent.

RMA4q's temperature results reflected the dynamic nature of the solution, while QUAL2E's temperature results illustrated the steady-state solution. Temperature in RMA4q varied longitudinally due to transport and time varying climate input, while QUAL2E reflects the averaging of climate conditions required of steady-state solutions.

BOD decay rates obtained for the two models were very close, with 1.0/day and 1.1/day for QUAL2E and RMA4q, respectively. The accuracy of these values is uncertain since (1) the decay rate may vary over the system in response to microflora and fauna changes that are responsible for organic decay; (2) the variability of observed data may be significant; (3) benthic sources and sinks are influential.

Both models overestimated DO in comparison to observed values when only the O'Connor-Dobbins reaeration formulation was used without any SOD. Using constant reaeration coefficients brought the simulated results closer to observations. The reaeration coefficient values used were 5.0 and 7.5/day for RMA4q and QUAL2E, respectively. These constant coefficients were smaller than typical values computed using the O'Connor-Dobbins relation. Use of SOD improved the fit for QUAL2E results, but the values required were greater than that seen for typical domestic wastewater sludge in the United States. If the O'Connor-Dobbins reaeration method had been used, much higher SOD values would have been necessary. It may be that sediment deposits and biological films have very large oxygen demands. If so, it would not be necessary to resort to specifying reaeration coefficients, but to use a reaeration formula. This needs to be corroborated by field investigations of the river bed.

Nutrient and algal concentrations could not be adequately modeled due to data limitations. However, analyses from observations and comparison with QUAL2E output indicated the presence of algal growth, especially in the lower river. Using information in Brown and Barnwell (1987) to constrain the maximum growth rate of suspended algae, algal growth from the model had little effect on nutrient or DO levels. Longer residence times occurred in the lower system than the upper, which should promote algal growth in the lower river. Setting settling rates to zero also produced minimal effects on DO concentrations of the system. In consideration of these, the typical algal growth rate values do not appear appropriate for the Nitra River. Chlorophyll *a* field data are needed for modeling and evaluating the effect of suspended algae. Benthic films composed of biologically active organisms and non-active material will also affect nutrient and DO levels of the water column. In shallow river system, the bed can dominate the processing of water quality constituents in the river. The role of the benthic community should also be evaluated.

To adequately evaluate the Nitra River system using complex models, additional data are needed, especially if nutrients and algal growth are important factors in controlling the system (which may be the case at the downstream stretch). Field data collection in support of a model needs to consider the data needs of the model. It is recognized that the data collection for this particular date was oriented towards simpler Streeter-Phelps models and their extensions. However, when more advanced models were applied (QUAL2E and RMA2/4q), the data were not sufficient to support these models. In particular, organic nitrogen and phosphorus, dissolved oxygen, and algal data are needed for both river and load stations. Quality control (QC) information for field sample data would be useful in evaluating the uncertainty of model input data, as well as that used in calibration. This includes not only laboratory QC but also field sampling QC. For dynamic modeling, time varying boundary condition data are also needed. If the upstream boundary condition is fixed, while the solution of the interior nodes are varying, this causes the solution in the region of the boundary not to vary as much as other interior points subject to the same daily cycle, i.e., solar radiation. Indeed, the model boundary is at some interior point of the actual system, and the portion of the actual system upstream of the model boundary also varies with time.

The ease of implementing a model for a water quality management project is an important consideration. Experienced users can rapidly implement a simple model, but more effort is required as complexity increases. For detailed analysis of effects from a particular emitter, a complex, dynamic model may be appropriate, especially if an understanding of dynamic effects are important in evaluating water quality for management purposes. For waste allocation and management studies over a whole watershed, a simpler model may be appropriate which can be easily incorporated into optimization schemes. With the least-cost approach in mind, the immediate need is often to use simple linear models (in terms of emissions) leading to linear programming tasks. However, recent work suggests the possibility of using complex, non-linear models within the optimization process based on dynamic programming (DP). The DP approach has been applied to the Nitra River system and is discussed in Somlyódy, *et al* (1993). This is particularly important if algae and associated diurnal DO fluctuations are of concern. Under such conditions phosphorus control becomes a crucial issue and the entire problem becomes inherently non-linear.

Finally, the general conclusion which can be drawn from this modeling effort follow.

1. Use of complex hydrodynamic and water quality models requires significant resources for implementation, including personnel, computational facilities, and data.
2. While problems arise in model calibration, they point to deficiencies in understanding of the actual system. These problems typically arise from having insufficient data. Particularly for the Nitra River system, algae and benthic/sediment processes are not yet clearly understood and need to be examined; since they likely are important.
3. Knowledge of the uncertainty in data used for calibration is needed to ascertain the accuracy of modeling results. This is true for both simple and complex models, though evaluation for complex models would be more difficult due to the greater number of state variable interactions and computation time.

6. REFERENCES

Brown, L.C. and T.O. Barnwell, Jr., 1987. The Enhanced Stream Water Quality QUAL2E and QUAL2E-UNCAS: Documentation and User Model, EPA/600/3-87/007.

King, I.P. 1990. Program documentation: RMA4 - a two dimensional, finite element, water quality model. Department of Civil and Environmental Engineering. University of California - Davis.

Masliev, I. and L. Somlyódy. 1993. Uncertainty analysis and parameter estimation for a class of linear dissolved oxygen models. Working Paper (in press). International Institute for Applied Systems Analysis. Laxenburg, Austria.

Masliev, I., Petrovic, P., Kuníková, M., Zajícová, H, and Somlyódy, L. 1994. Longitudinal water quality profile measurements and their evaluation in the Nitra River basin (Slovakia). International Institute for Applied Systems Analysis. Laxenburg, Austria.

Somlyódy, L and, M. Kularathna, and I. Masliev. 1993 Water quality management of the Nitra River Basin (Slovakia): evaluation of various control strategies. WP-93-63. International Institute for Applied Systems Analysis. Laxenburg, Austria.

Somlyódy, L and C.M. Paulsen. 1992. Cost-effective water quality management strategies in Central and Eastern Europe. WP-92-091. International Institute for Applied Systems Analysis. Laxenburg, Austria.

Thomann, R.V. and J.A. Mueller. 1987. Principles of Surface Water Quality Modeling and Control. Harper and Row. New York.

APPENDIX A - DATA FOR 25 AND 26 AUGUST, 1992 SAMPLING CAMPAIGN

Table A1: Inflows and withdrawals used in modeling the Nitra River.

Station Name	Station Type	Location (rk)	Flow (m^3/s)
Novaky over	Headwater	132.5	0.608
Lehotsky pot.	Tributary	132.30	0.180
Chemical factory Novaky	Wasteload	130.60	0.130
Chemical factory Novaky	Wasteload	129.70	0.170
Power plant Novaky	Wasteload	128.30	0.285
Nitrica	Tributary	112.00	0.196
Partizanske-sewer 1	Wasteload	112.00	0.085
Partizanske-sewer 2	Wasteload	111.00	0.106
Tannery Bosany	Wasteload	100.60	0.150
Bebrava	Tributary	98.30	0.485
Chotina	Tributary	94.00	0.045
Topolcany STP	Wasteload	93.00	0.070
	Withdrawal W	80.726	1.401
	Withdrawal W	71.925	1.084
Radosinka	Tributary W	67.0	0.540
Poultry Luzianky	Wasteload	65.20	0.002
Wine factory Luzianky	Wasteload	63.80	0.006
Ferenit factory Nitra	Wasteload	63.00	0.001
Sugar factory Nitra	Wasteload	60.90	0.013
Freezing factory Nitra	Wasteload	54.05	0.001
Bioveta factory Nitra	Wasteload	54.00	0.007
Stara Nitra	Withdrawal	53.10	0.442
Nitra STP	Wasteload	52.50	0.381
Confluence of Zitava	Tributary	25.40	0.087
Confluence of Stara Nitra	Tributary	22.60	0.382
Nove Zamky STP	Wasteload	8.80	0.152

W estimated flows

Table A2: Inflows concentrations used in modeling the Nitra River.

Station Name	Location (rk)	Temp. (°C)	DO (mg/L)	BOD (mg/L)	NH4 (mgN/L)	NO2 (mgN/L)	NO3 (mgN/L)	Diss.P (mgP/L)	TDS (mg/L)
Novaky over	132.5	19.4	2.5	7.4	3.58	0.112	1.90	1.142	438
Lehotsky pot.	132.30	18.3	6.8	2.6	0.22	0.022	1.90	0.240	418
Chemical factory Novaky	130.60	24.1	0.0*	68.0	0.0*	0.0*	0.0*	0.0*	7030
Chemical factory Novaky	129.70	25.5	0.0*	42.0	0.0*	0.0*	0.0*	0.0*	1060
Power plant Novaky	128.30	26.4	4.0	5.0	0.92	0.113	2.00	0.411	1338
Nitrica	112.00	21.2*	9.3	2.7	0.06	0.032	3.50	0.033	480
Partizanske-sewer 1	112.00	21.4	0.0*	100.0	12.09	0.002	0.30	2.43	590
Partizanske-sewer 2	111.00	21.5	0.0*	38.0	7.66	0.053	0.02	1.86	782
Tannery Bosany	100.60	24.5	0.0*	120.0	0.0*	0.0*	0.0*	0.0*	3074
Bebrava	98.30	22.0	7.3	2.5	0.25	0.109	2.00	0.388	482
Chotina	94.00	22.7	11.8	4.1	0.08	0.119	3.30	0.013	478
Topolcany STP	93.00	22.6*	0.0*	37.0	15.55	0.002*	0.30*	0.82	790
Radosinka	67.0	22.0*	8.5*	3.1*	0.13*	0.087*	2.93*	0.18*	455
Poultry Luzianky	65.20	20.8	0.0*	3.0	0.13	0.510	10.60	0.104	766
Wine factory Luzianky	63.80	21.5	0.0*	170.0	0.0*	0.0*	0.0*	0.0*	2074
Ferenit factory Nitra	63.00	20.9	0.0*	10.2	0.0*	0.0*	0.0*	0.0*	1126
Sugar factory Nitra	60.90	20.9	0.0*	15.0	0.0*	0.0*	0.0*	0.0*	1138
Freezing factory Nitra	54.05	22.3	0.0*	85.0	4.54	0.003	0.70	1.106	896
Bioveta factory Nitra	54.00	21.5	0.0*	8.0	0.64	0.394	8.30	1.8	682
Nitra STP	52.50	22.6	0.0*	138.0	19.56	0.002	0.50	3.0	930
Confluence of Zitava	25.40	26.4	7.2	12.8	0.65	0.026	0.20	0.098	418
Confluence of Stara Nitra	22.60	25.2	9.6	12.0	0.35	0.075	2.40	0.62	1180
Nove Zamky STP	8.80	24.5	0.0*	140.0	15.63	0.002*	1.50	1.68	748

* estimated

APPENDIX B - FIGURES

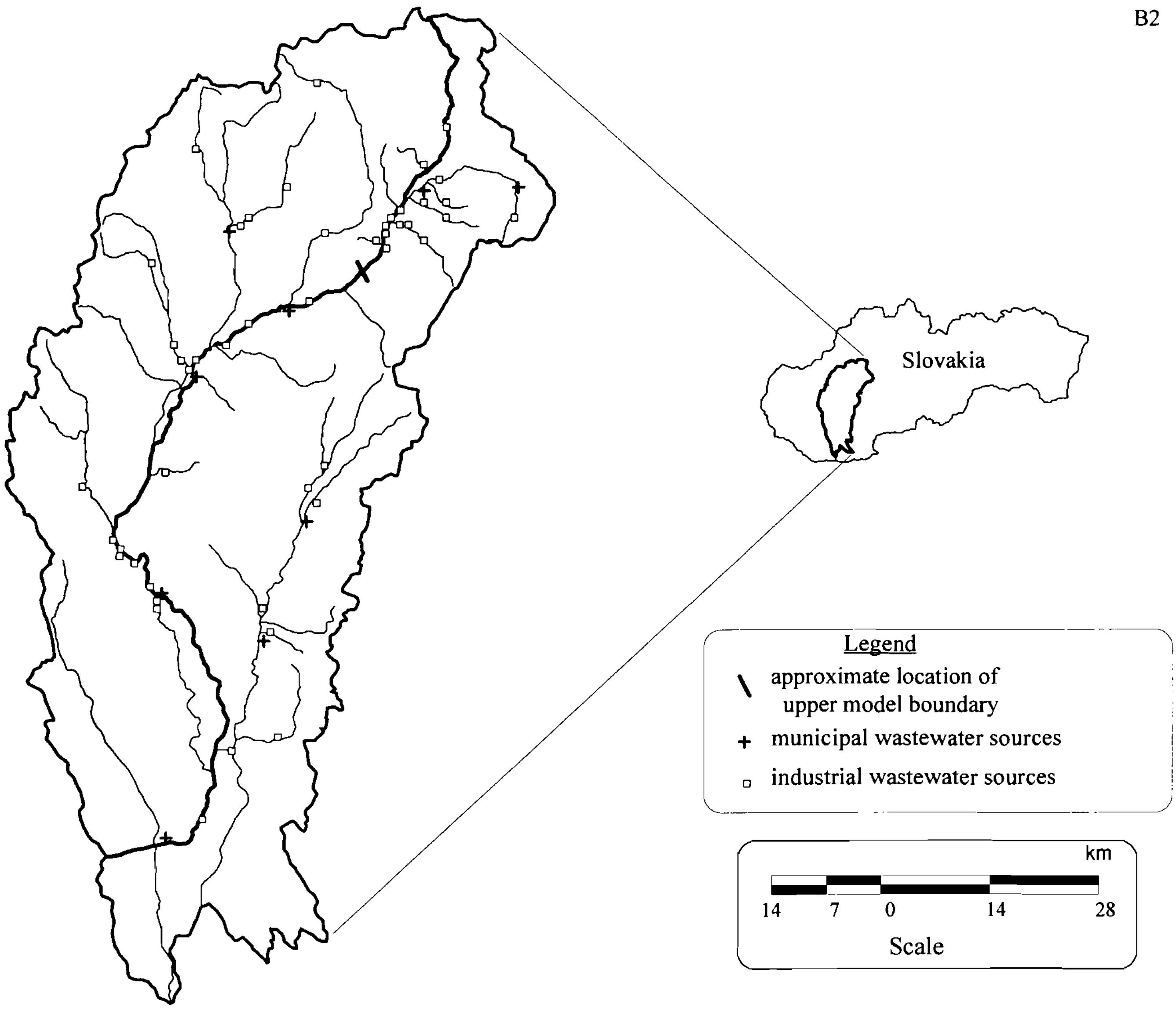

Figure 1: Map of Nitra River basin and its location in Slovakia. The bold line through the center of the basin is the main stem of the Nitra River. Locations of wastewater sources and the model's upper boundary are also indicated.

QUAL2E CONSTITUENT INTERACTIONS

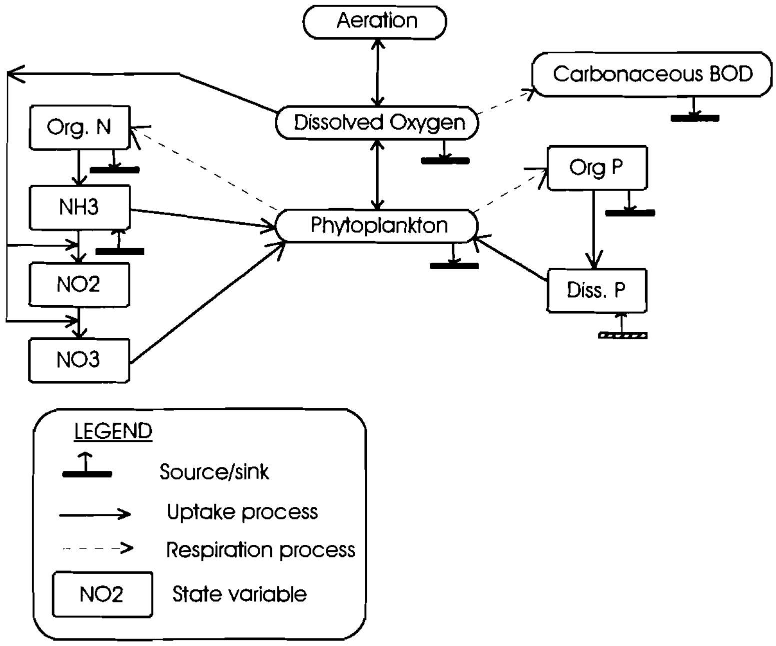

Figure 2: Water quality interactions present in QUAL2E.

RMA-4 CONSTITUENT INTERACTIONS

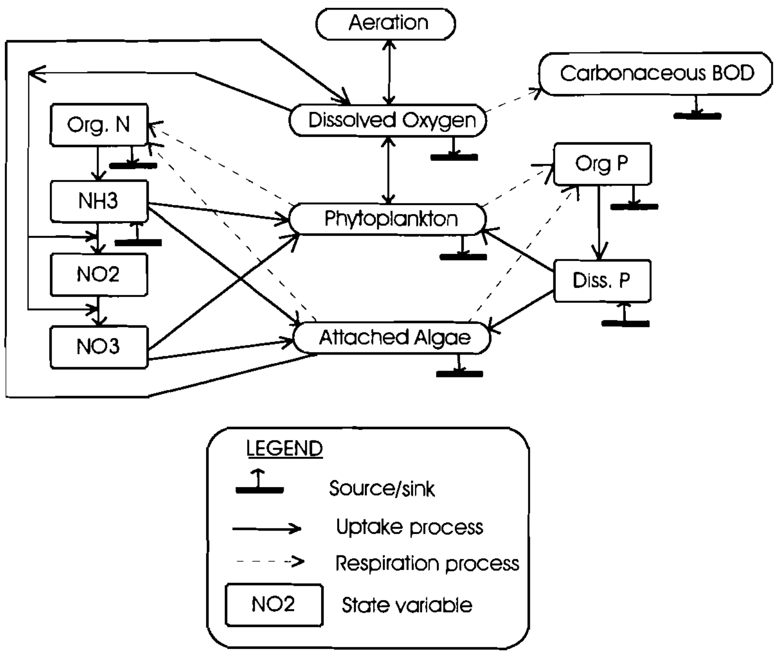

Figure 3: Water quality interactions accounted for in RMA4q v2.

Figure 4: Flow for the Nitra River system from QUAL2E results.

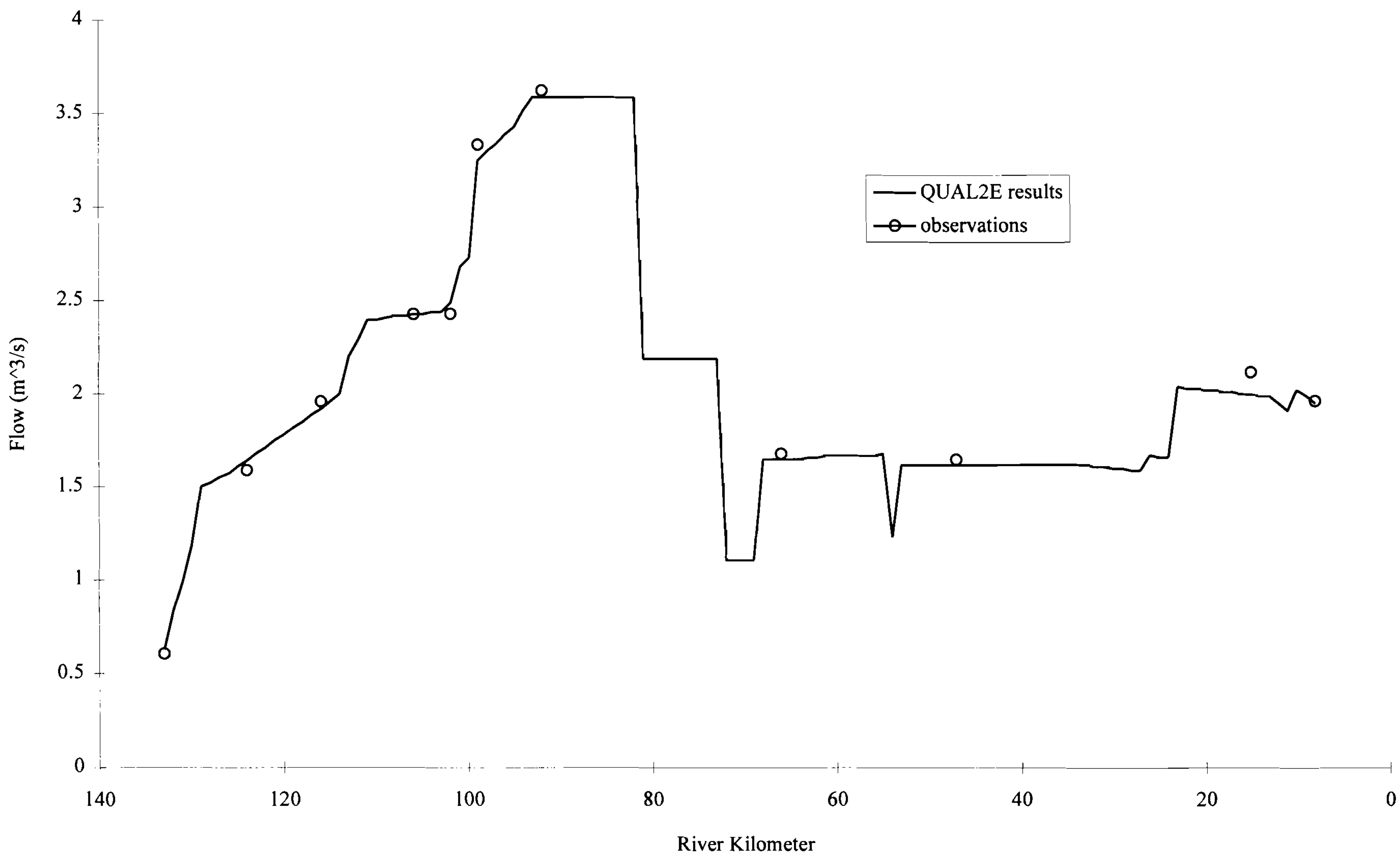

Figure 5: Soluble solids for the Nitra River system and results from QUAL2E.

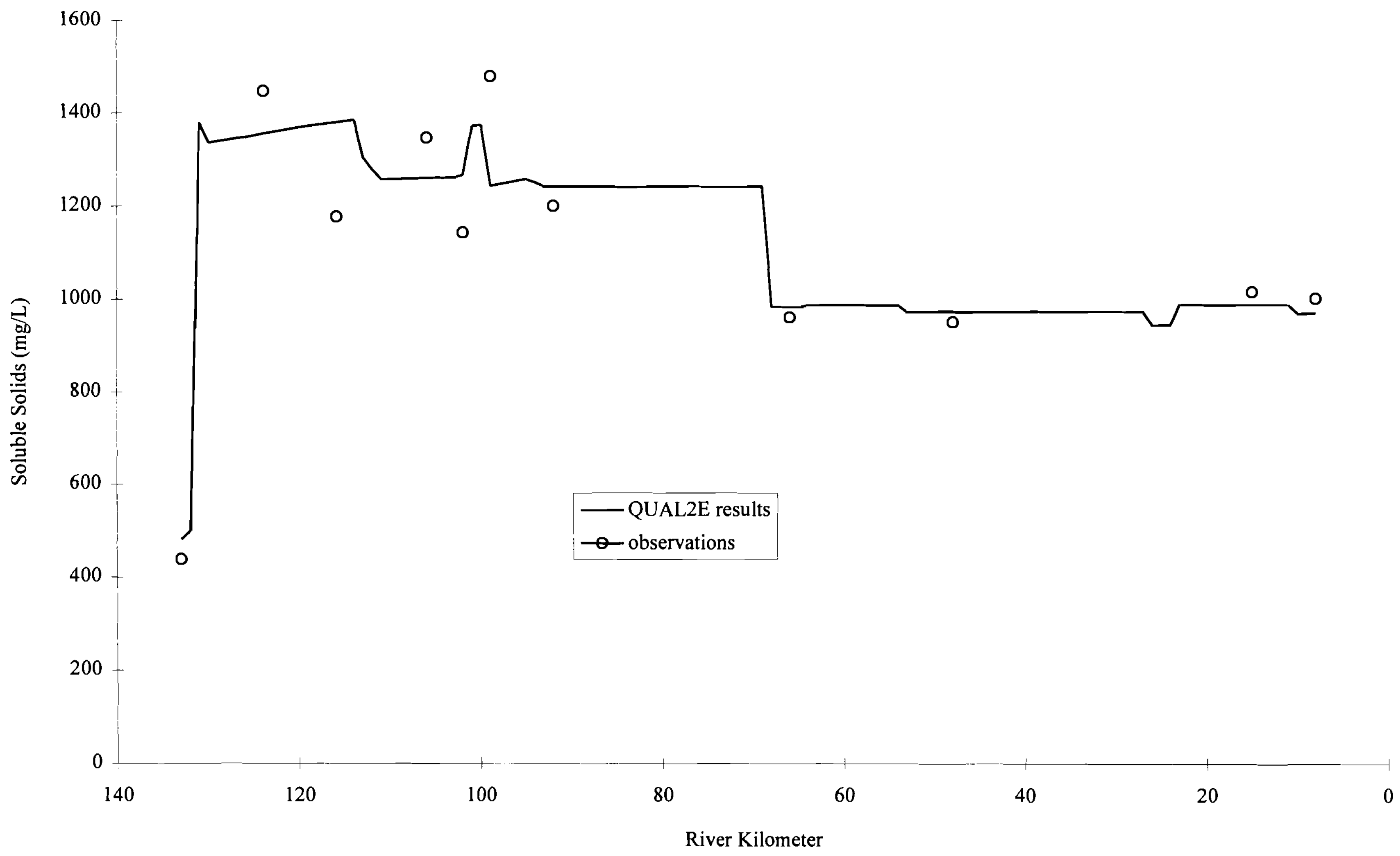

Figure 6: Velocity for the Nitra River system as computed by QUAL2E

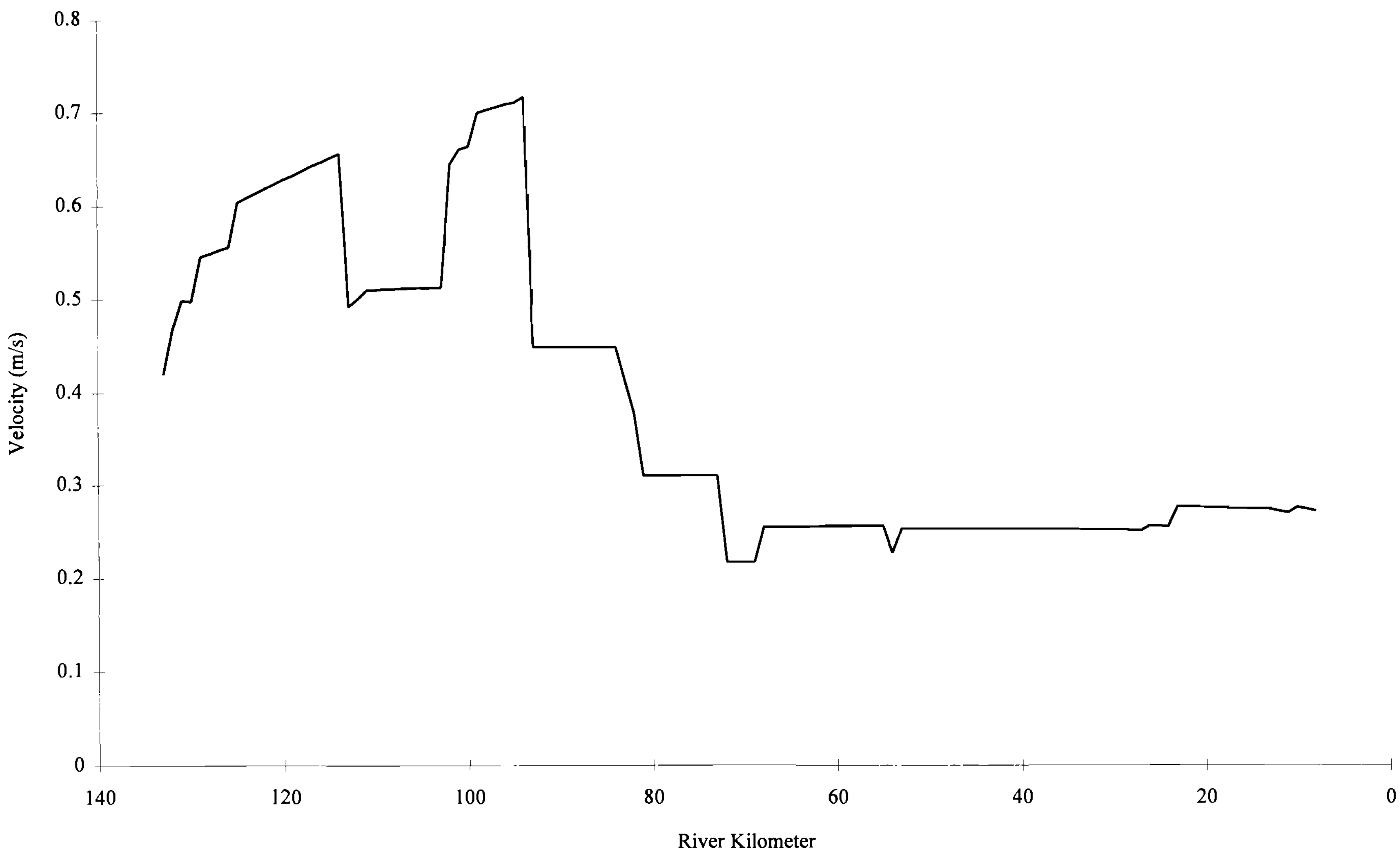

Figure 7: Depth for the Nitra River system as computed by QUAL2E

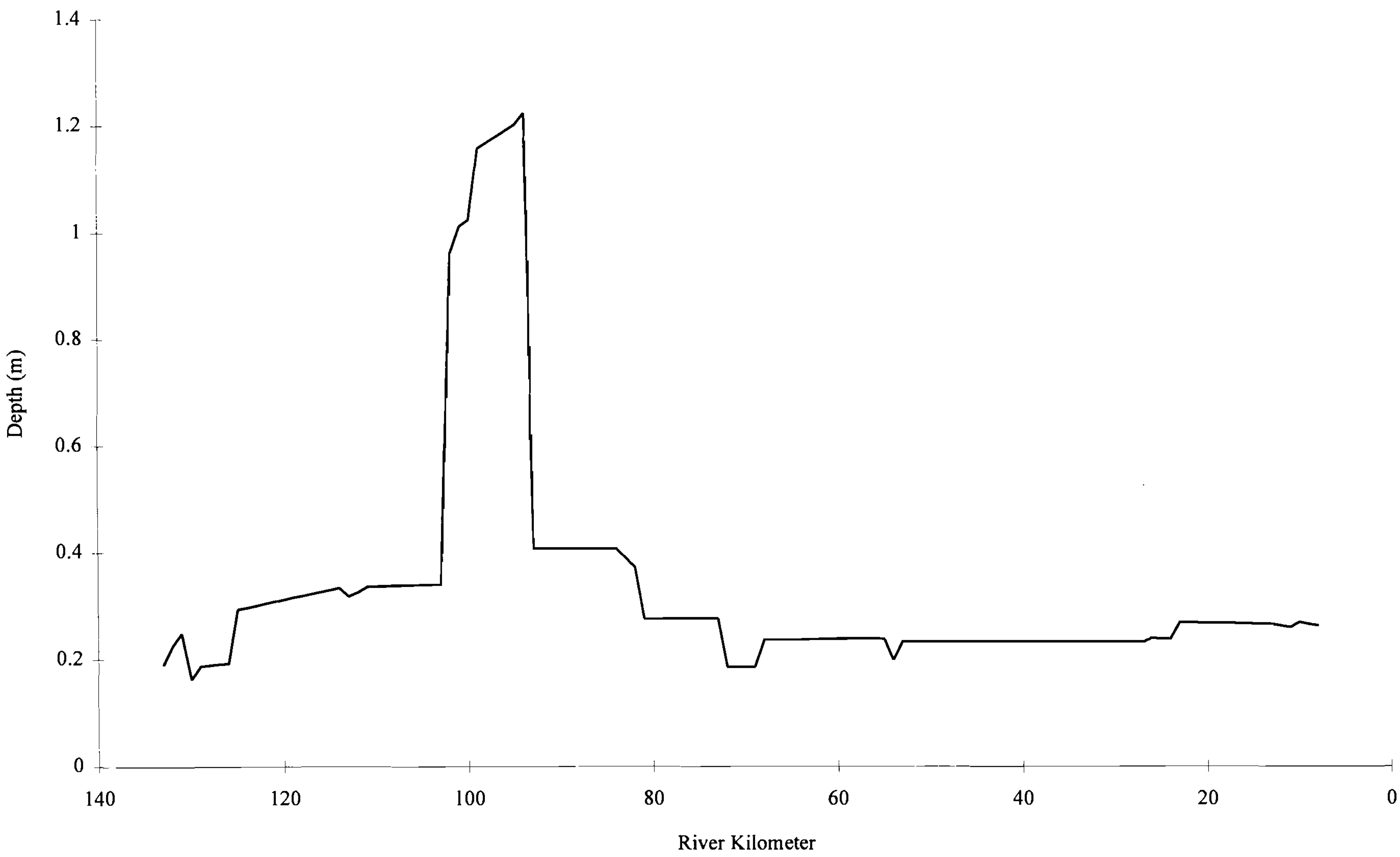

Figure 8: Temperature for the Nitra River system and results from QUAL2E.

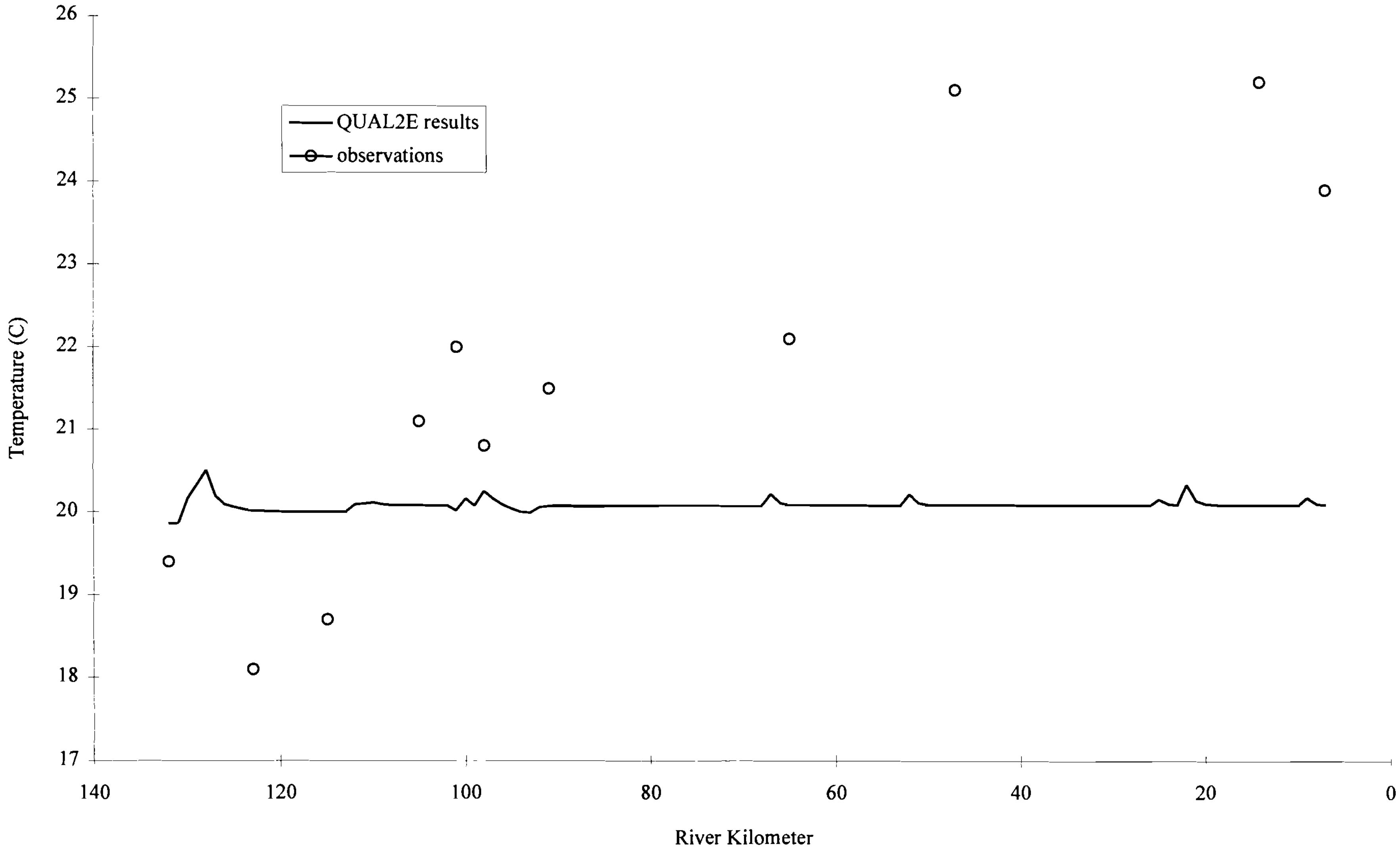

Figure 9: Residual sum of squares for QUAL2E BOD results when compared to observations. BOD decay rates (k1) are varied from 0.02 to 4.5/day.

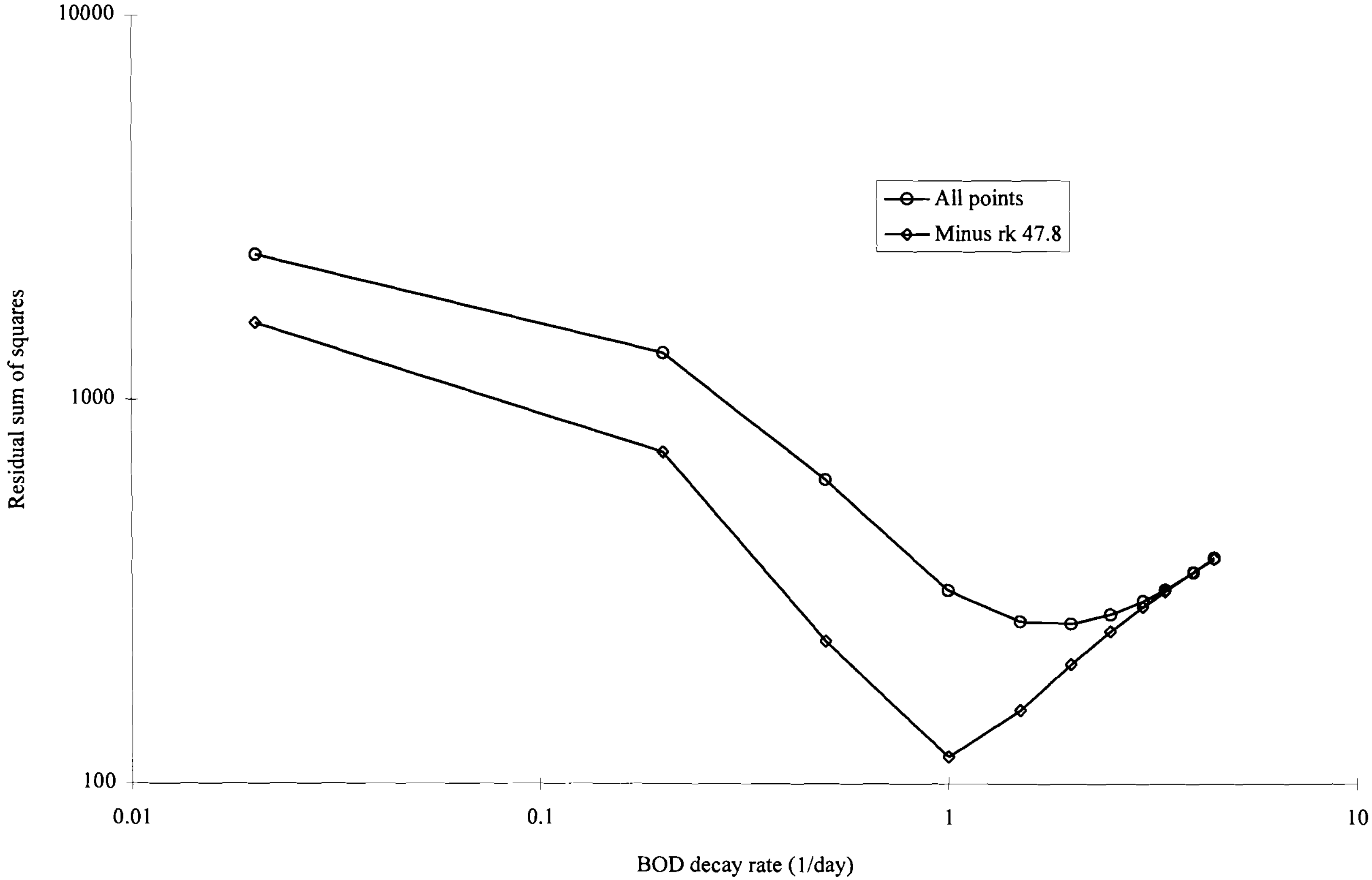

Figure 10: BOD observations and QUAL2E model results.

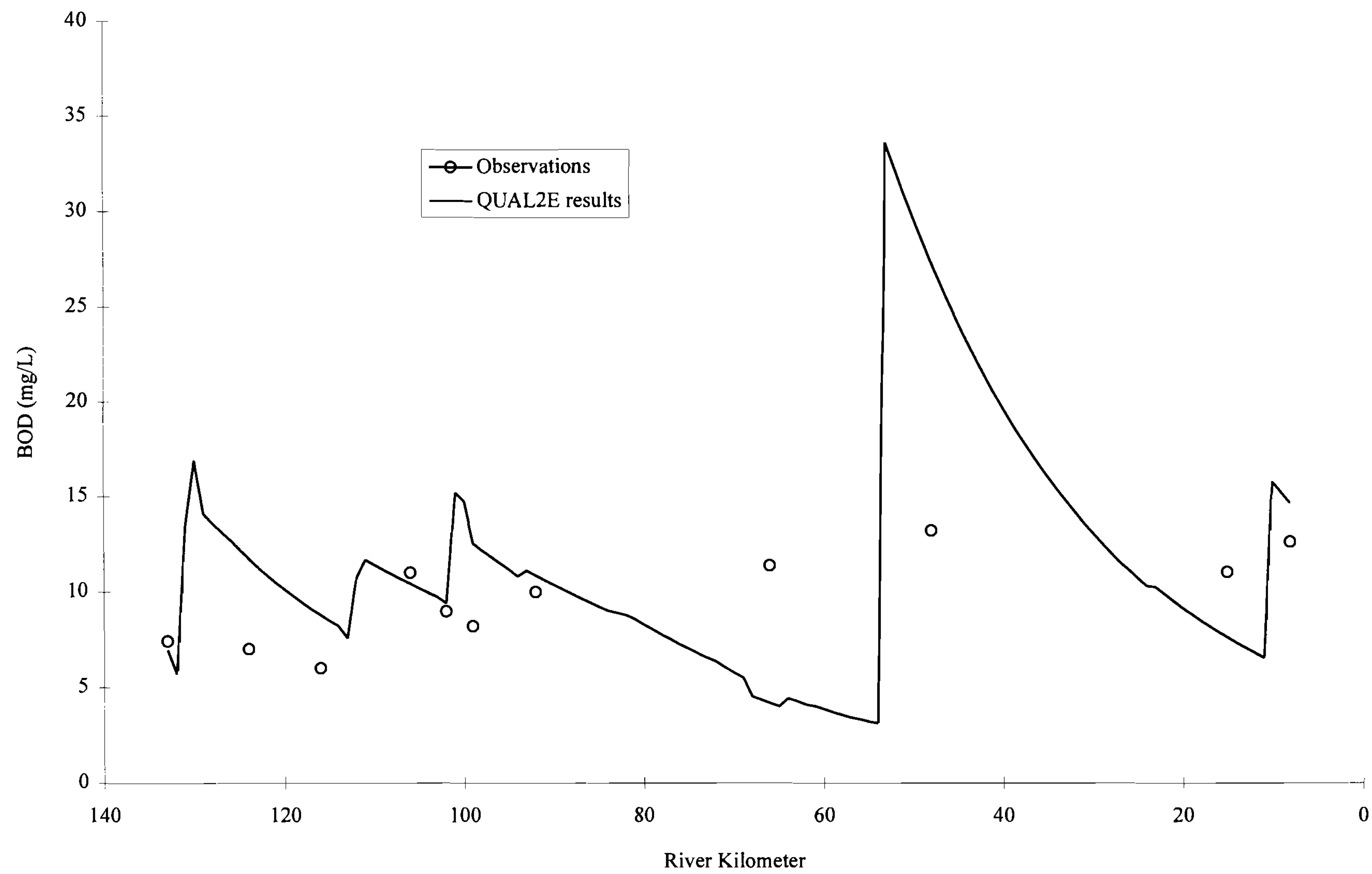

Figure 11: Diurnal variation in dissolved oxygen in the upper and lower Nitra River.

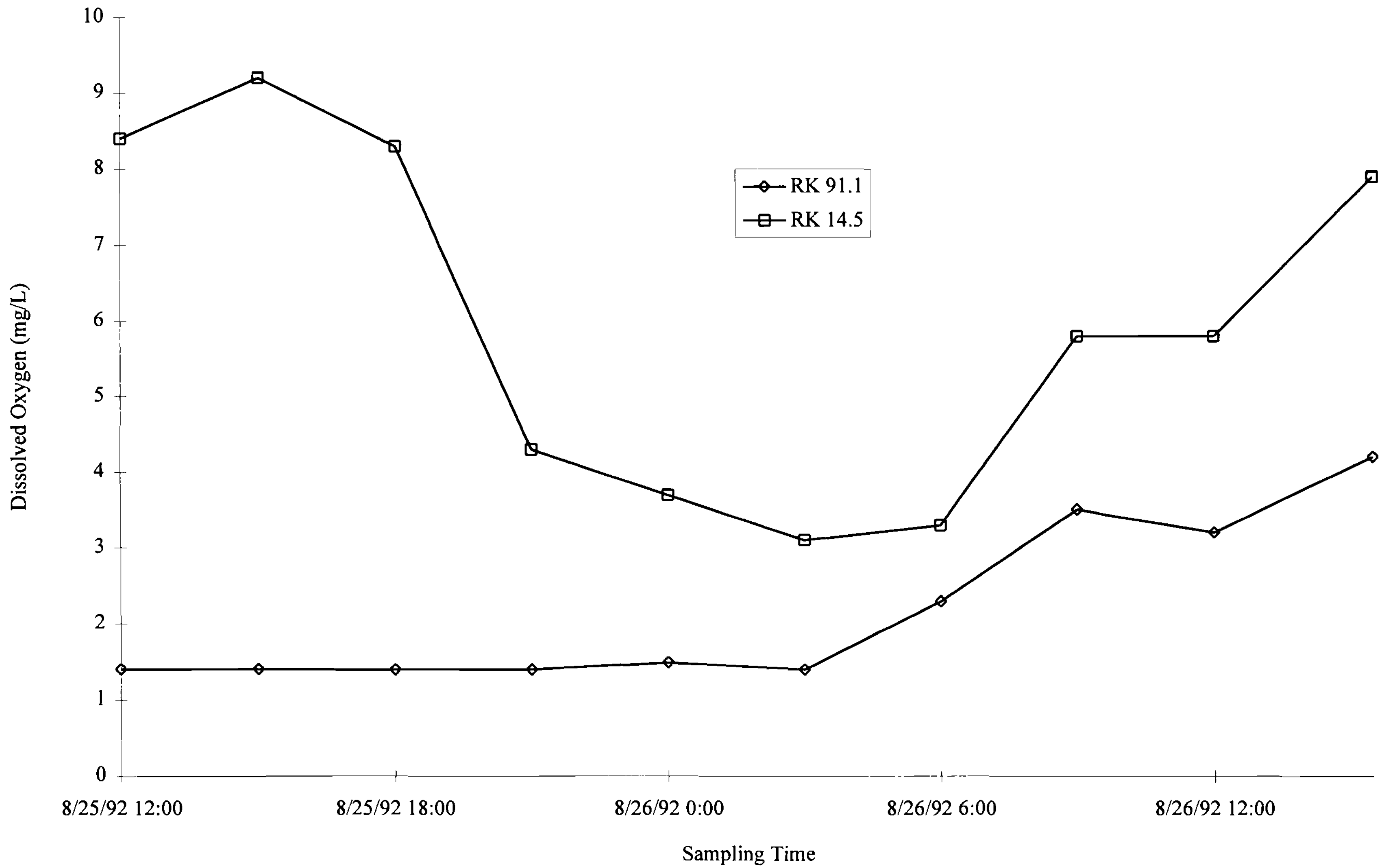

Figure 12: Computed residence times of 1 kilometer long computational elements in QUAL2E.

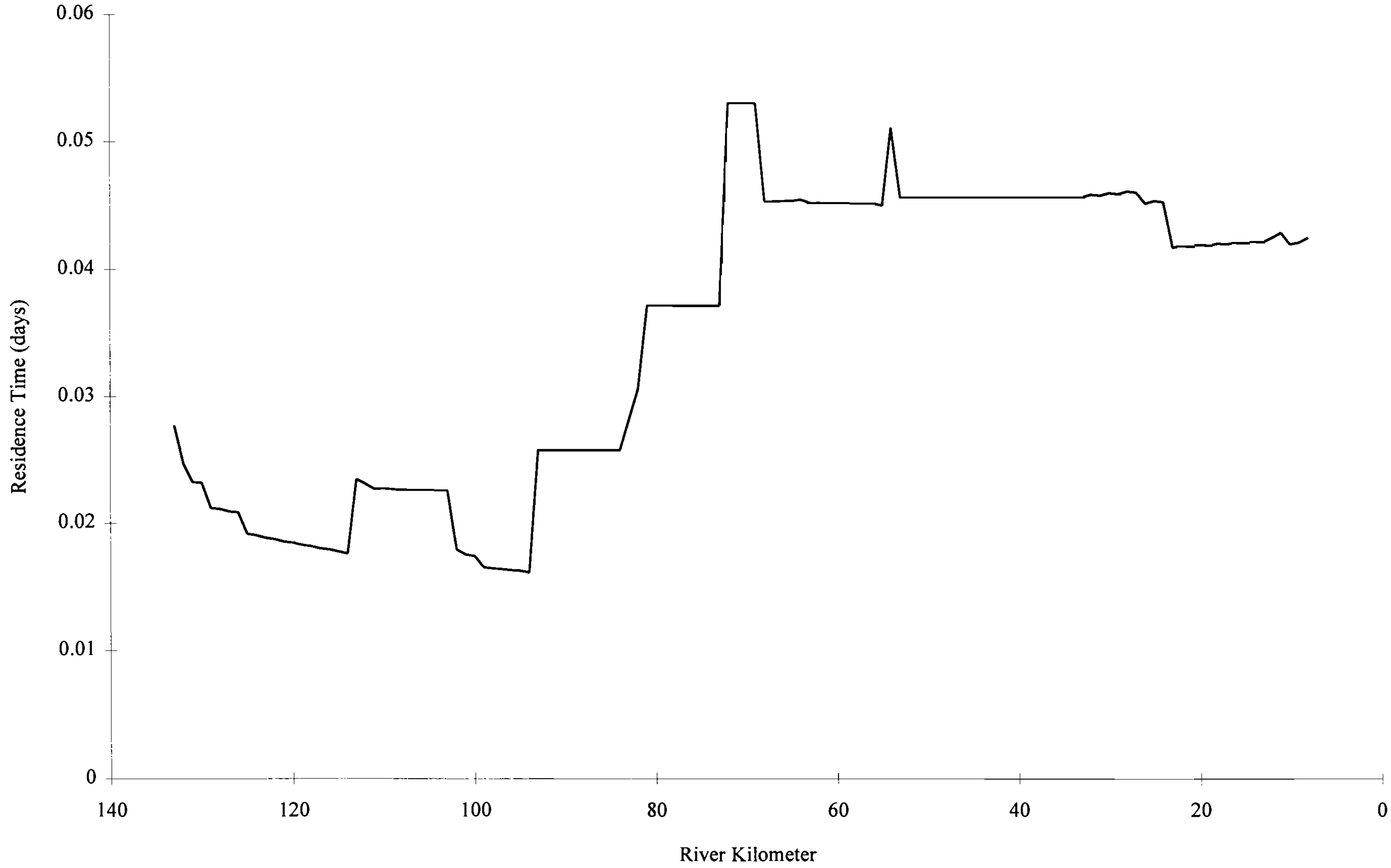

Figure 13: Computed algal settling rates from QUAL2E depths and a settling rate = 0.5 m/day.

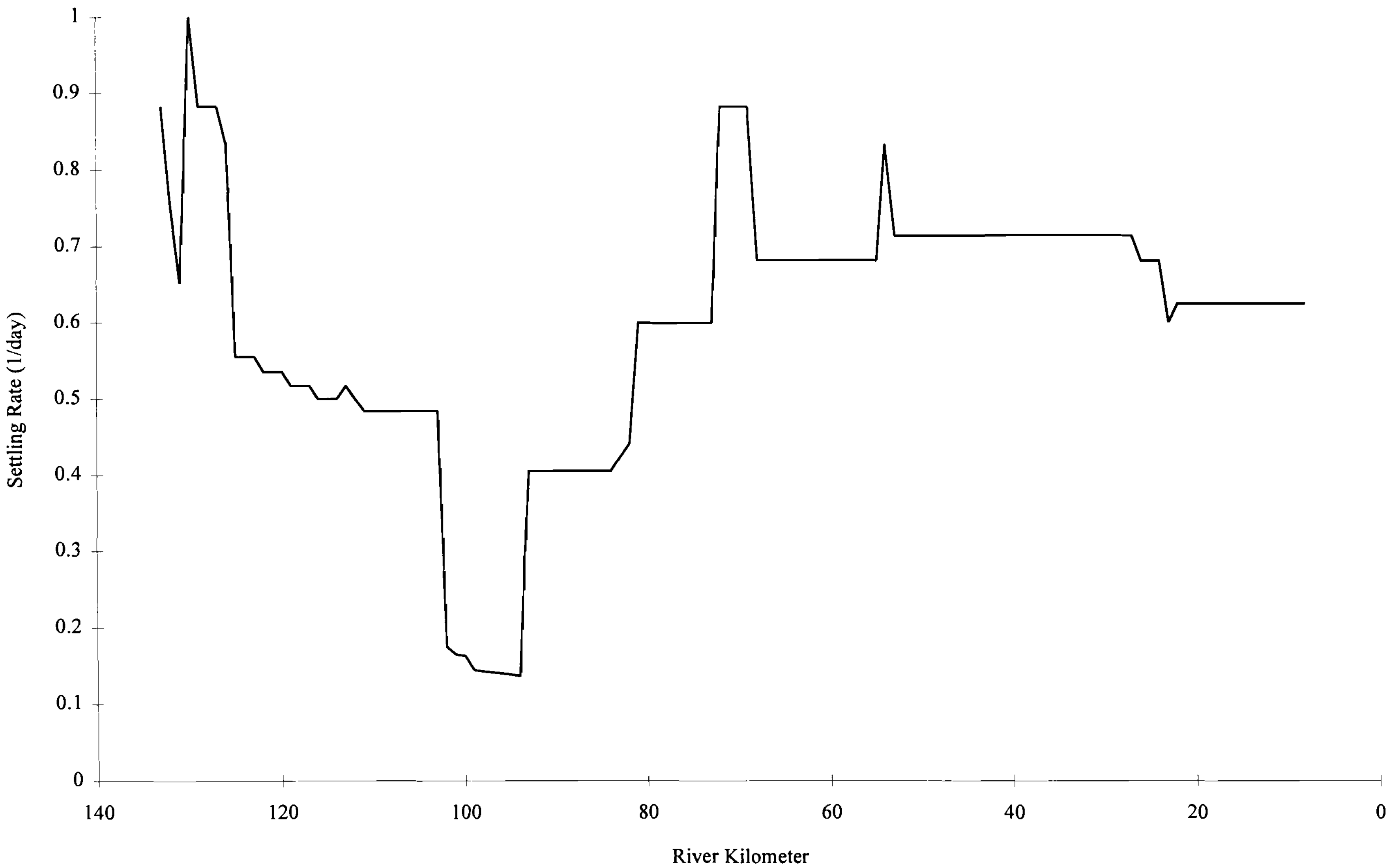

Figure 14: Algae concentration results for the Nitra River from QUAL2E

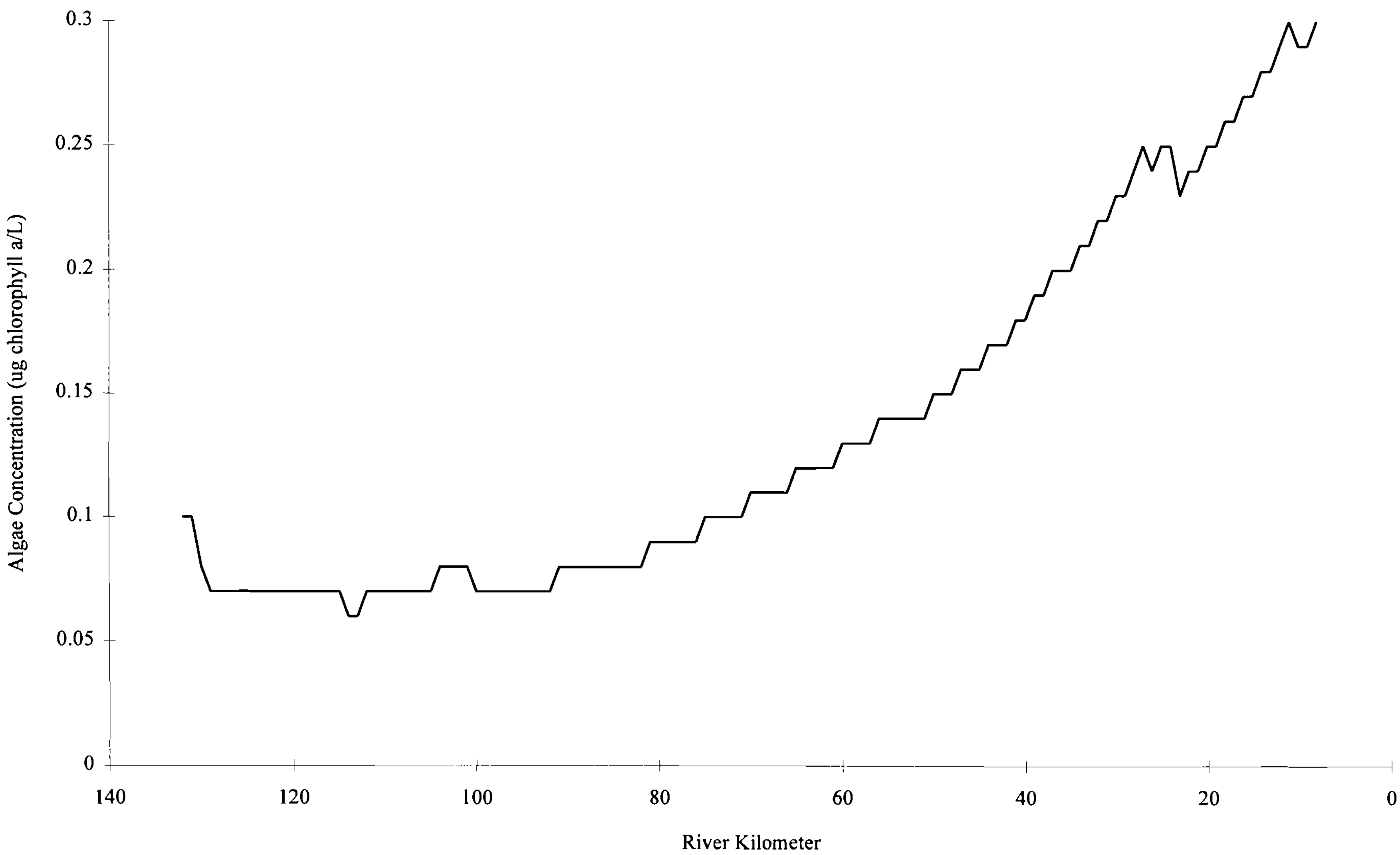

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Figure 15: Dissolved oxygen observations and QUAL2E model results, with a comparison between methods for specifying reaeration coefficients.

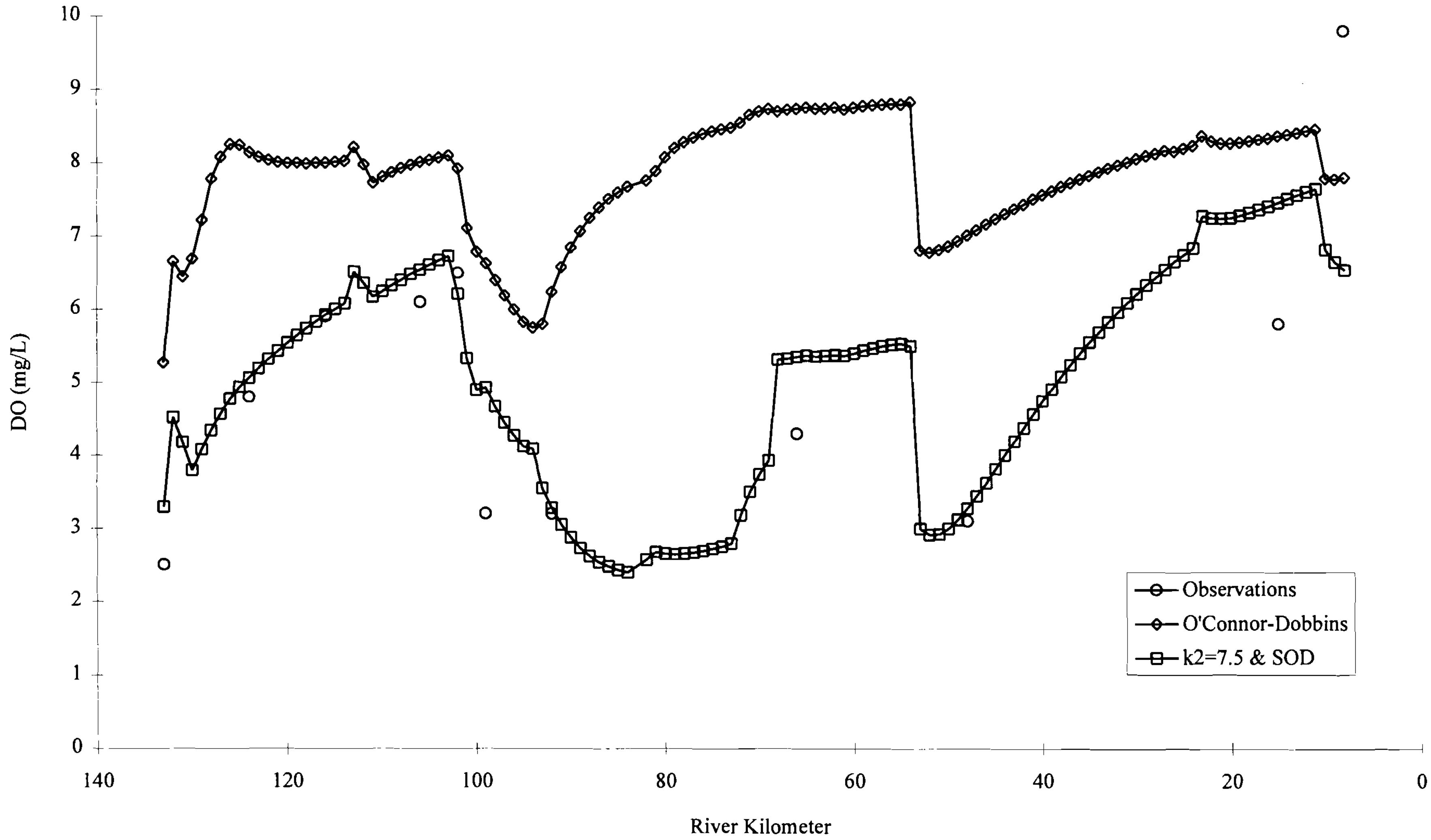

Figure 16: Dissolved phosphorus observations and QUAL2E model results.

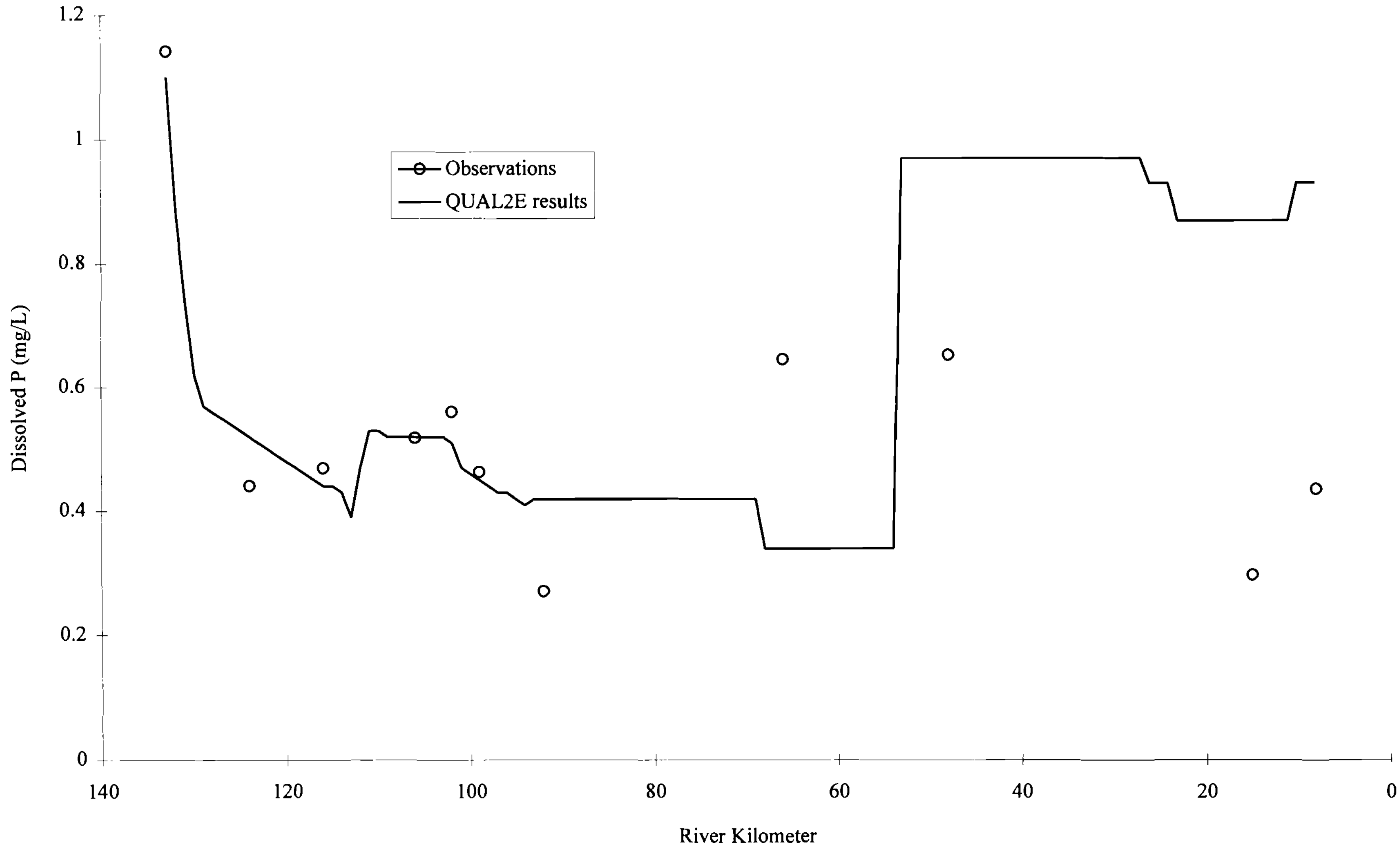

Figure 17a: Ammonia nitrogen observations and QUAL2E model results for the NItra River system.

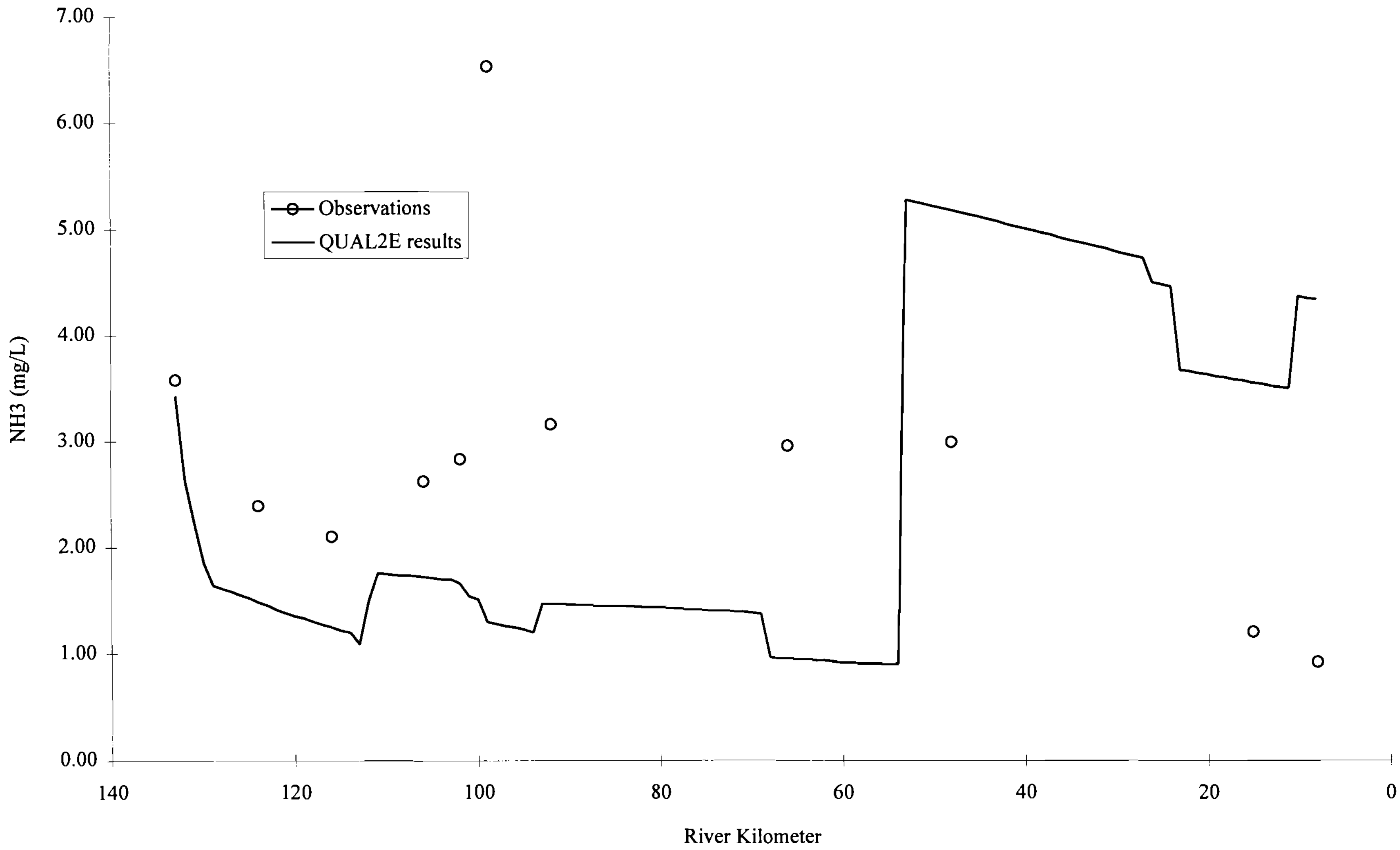

Figure 17b: Nitrite nitrogen observations and QUAL2E model results for the NItra River system.

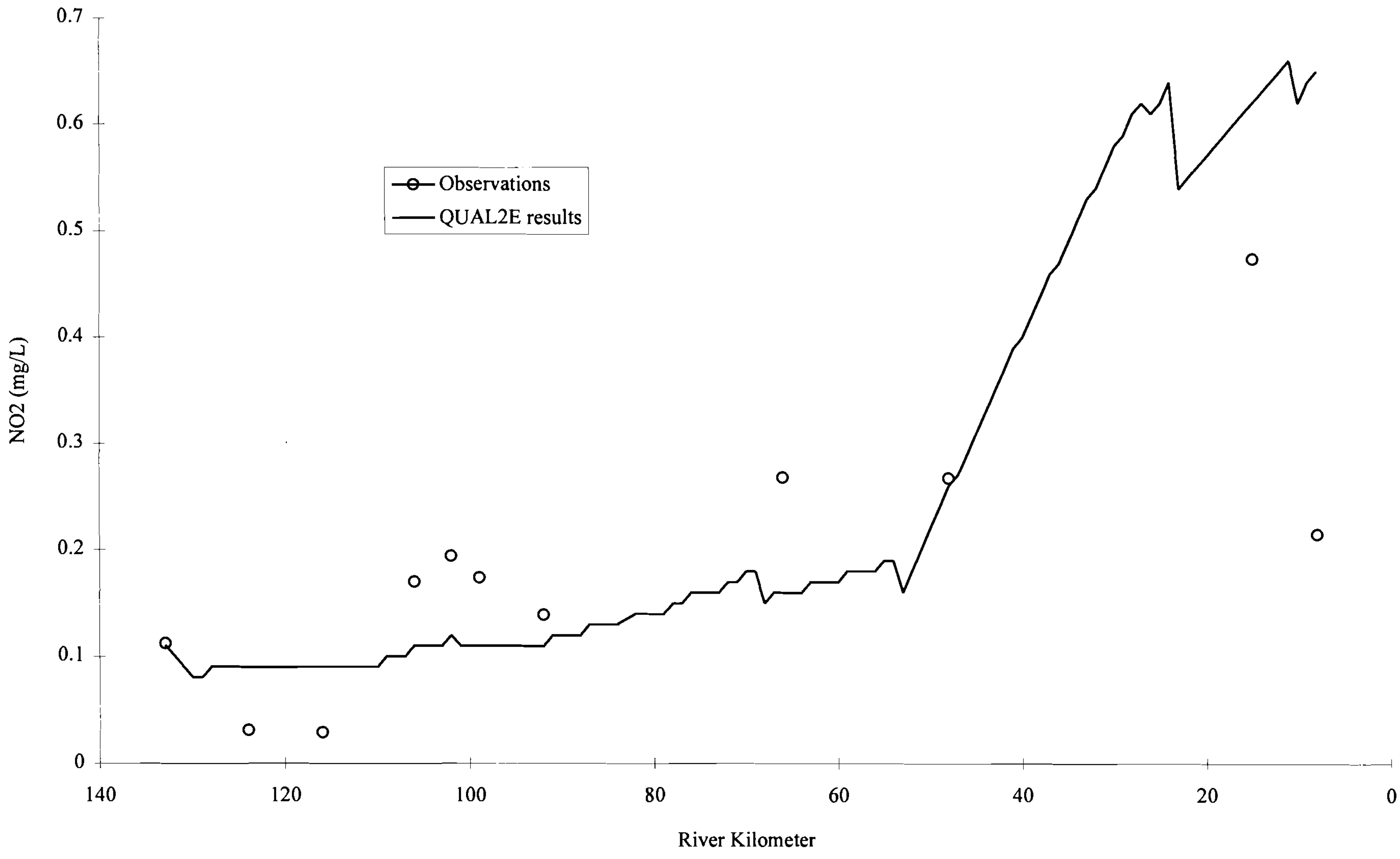

Figure 17c: Nitrate nitrogen observations and QUAL2E model results for the NItra River system.

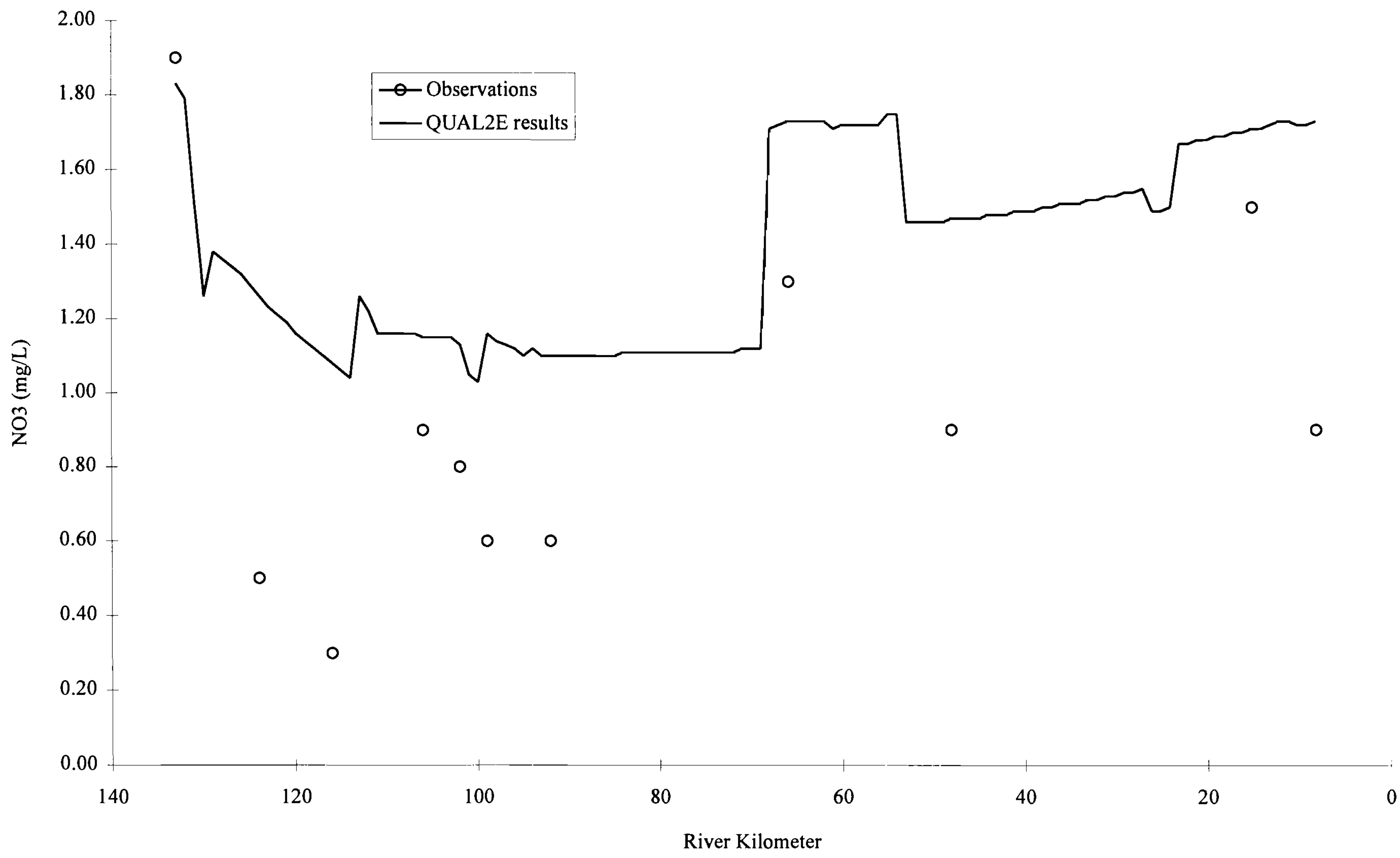

Figure 18a: Ammonia nitrogen observations and QUAL2E model results after adding benthic sources of ammonia and adjusting the ammonia rate.

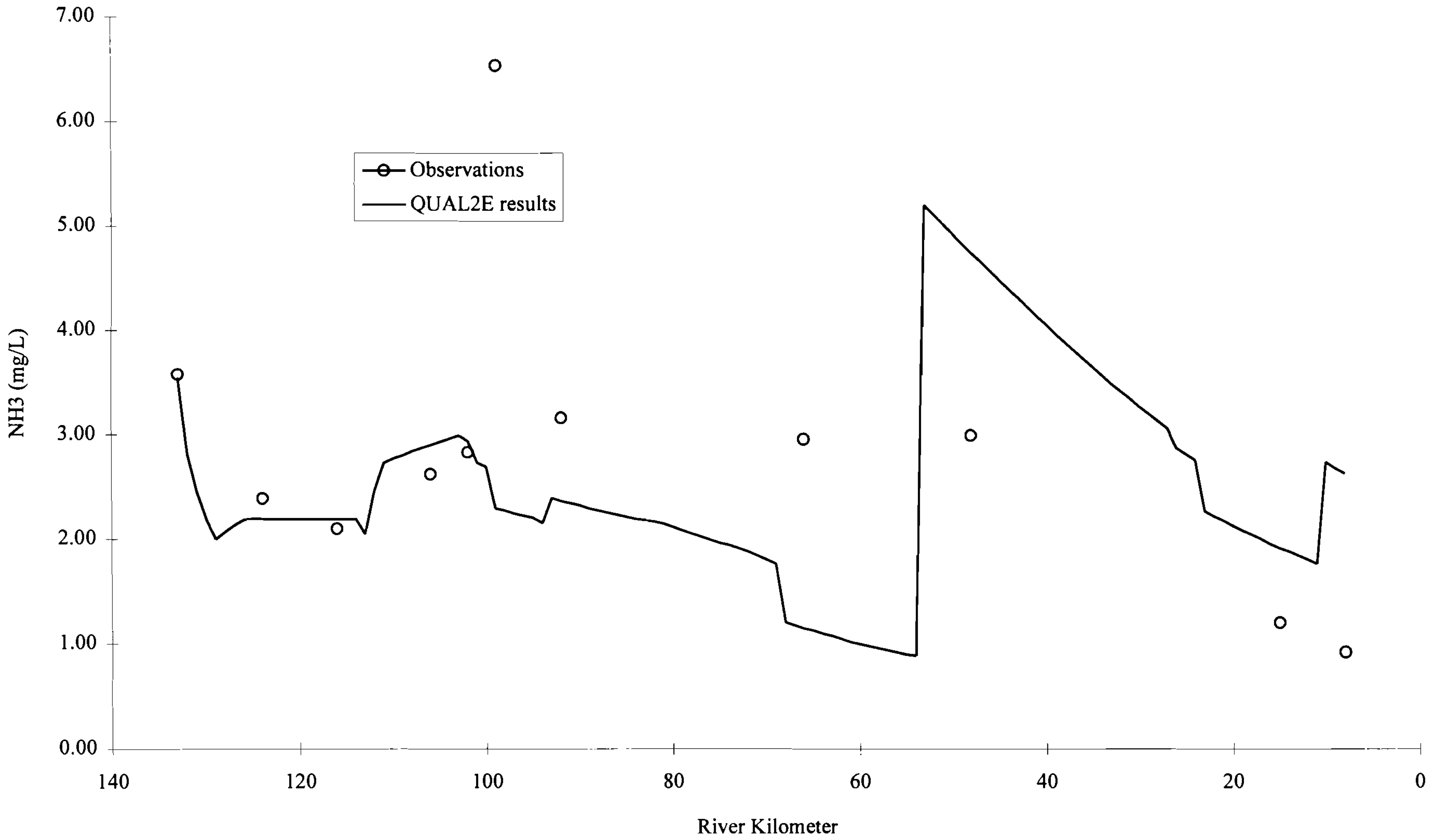

Figure 18b: Nitrite nitrogen observations and QUAL2E model results after adding benthic sources of ammonia and adjusting the ammonia and nitrite oxidation rates.

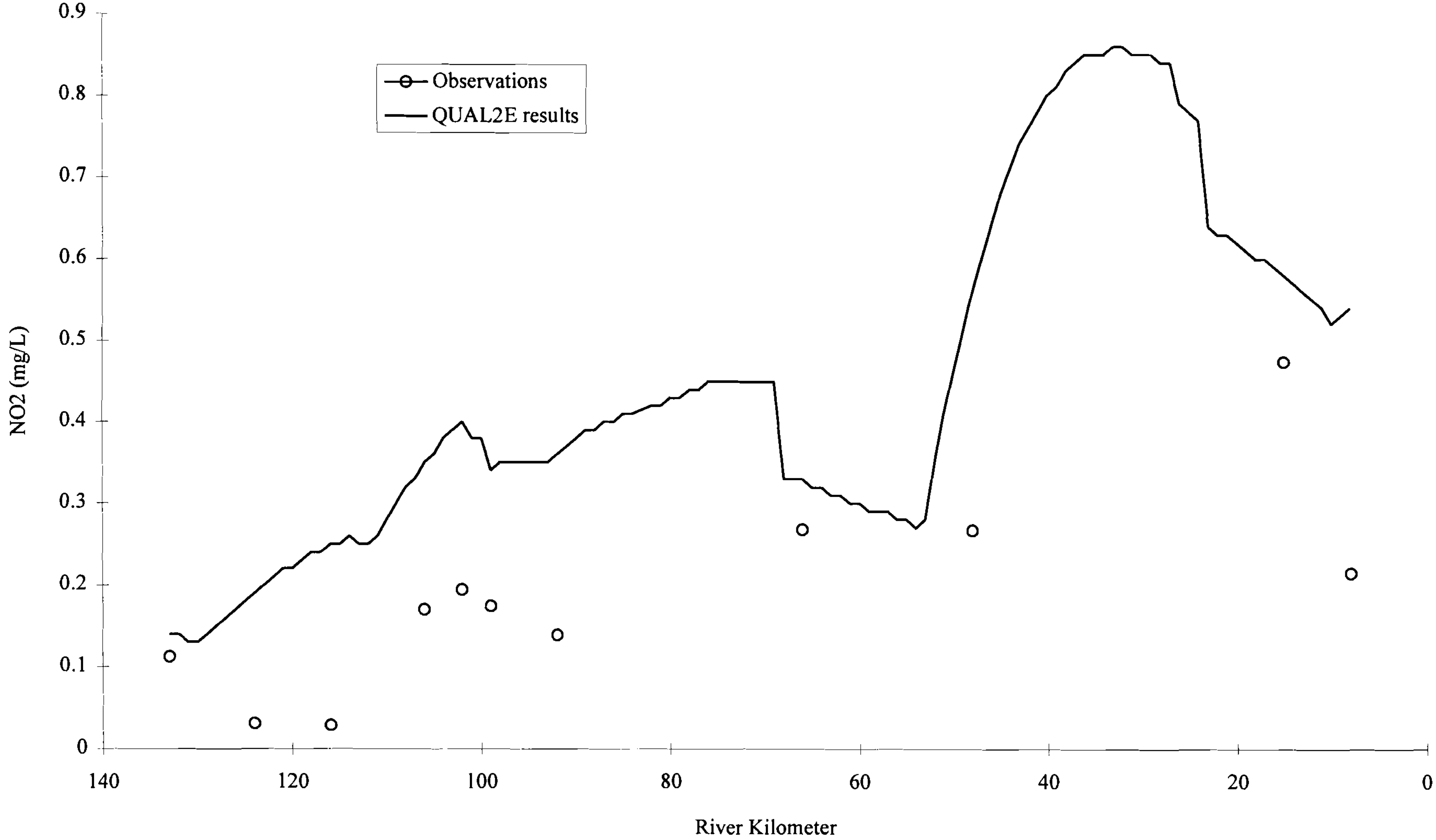

Figure 18c: Nitrate nitrogen observations and QUAL2E model results after adding benthic sources of ammonia and adjusting ammonia and nitrite oxidation rates.

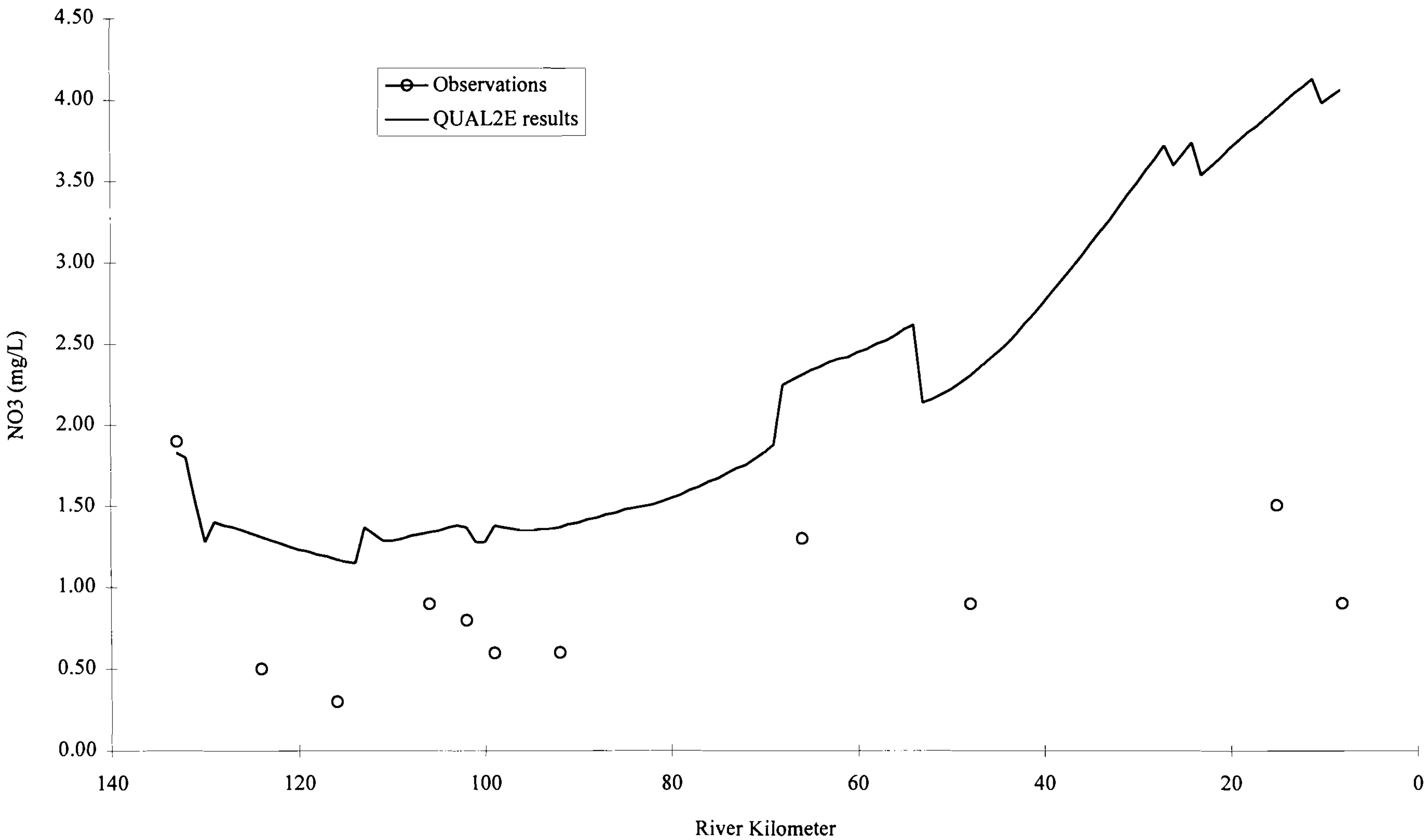

Figure 19: Depth results for the Nitra River system obtained from the RMA2 hydrodynamics model after 360 hours of simulation.

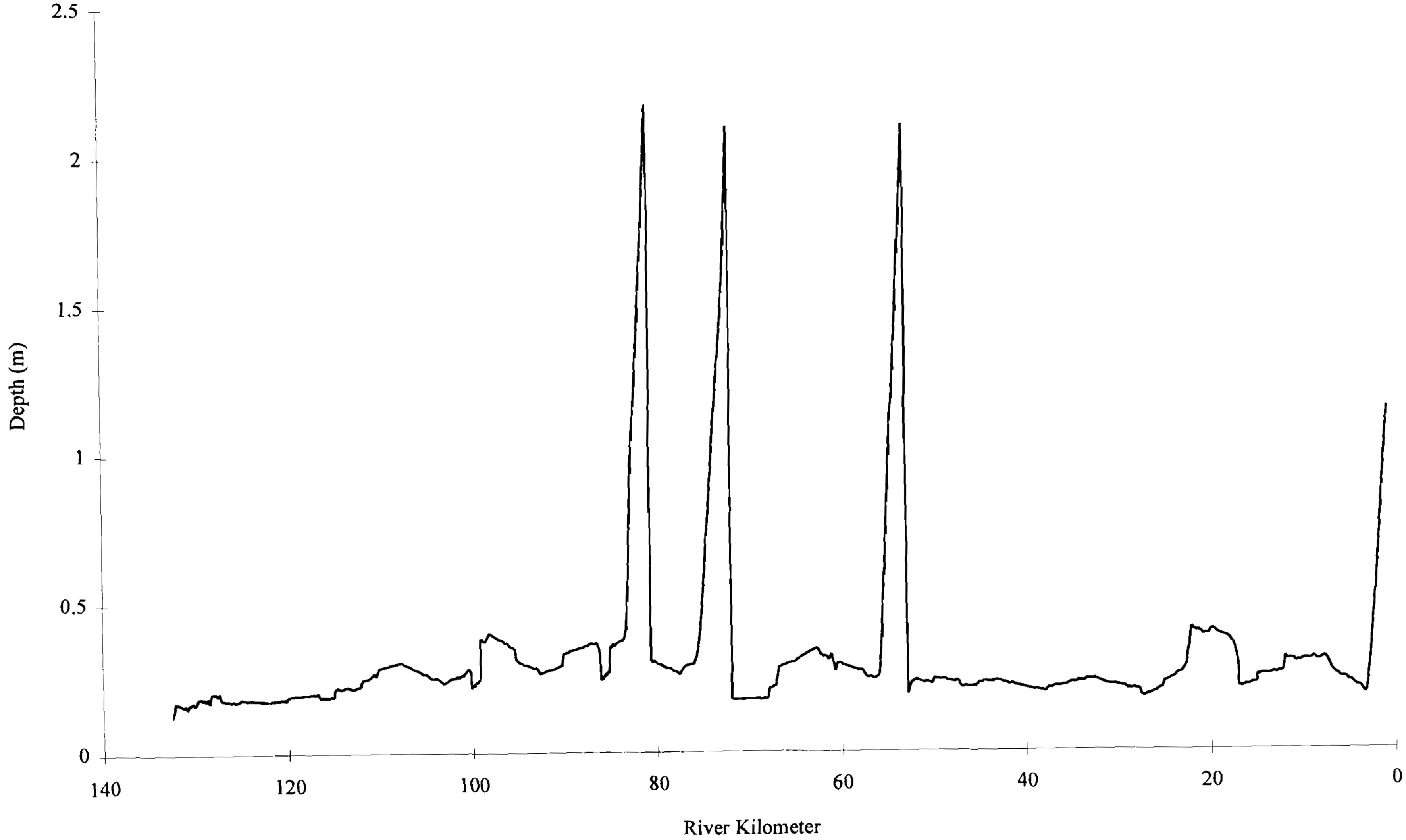

Figure 20: Velocity results for the Nitra River system obtained from the RMA2 hydrodynamics model after 360 hours of simulation.

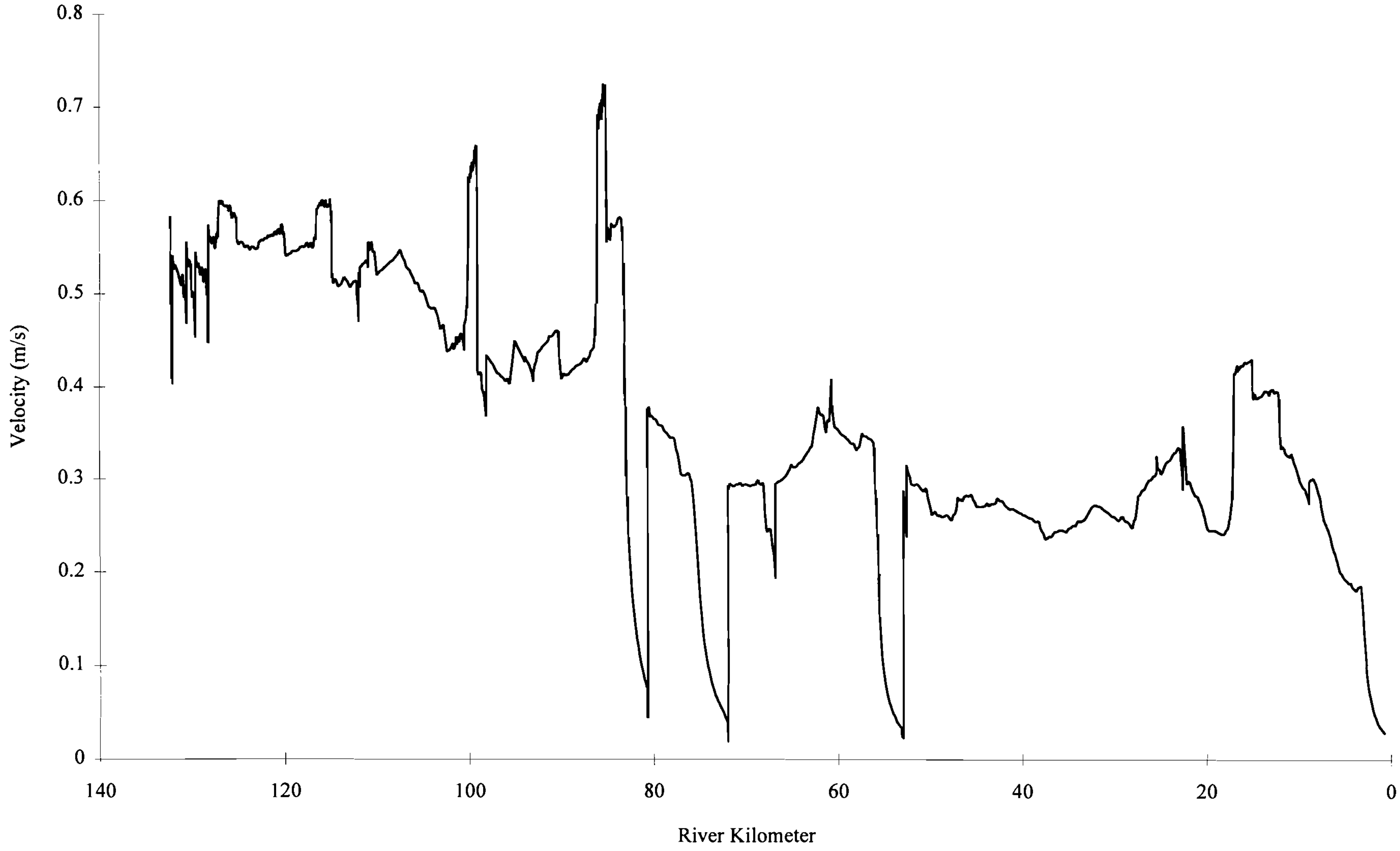

Figure 21: Results from flow computations by RMA2 after 360 hours of simulation and comparison with observations in the Nitra River system.

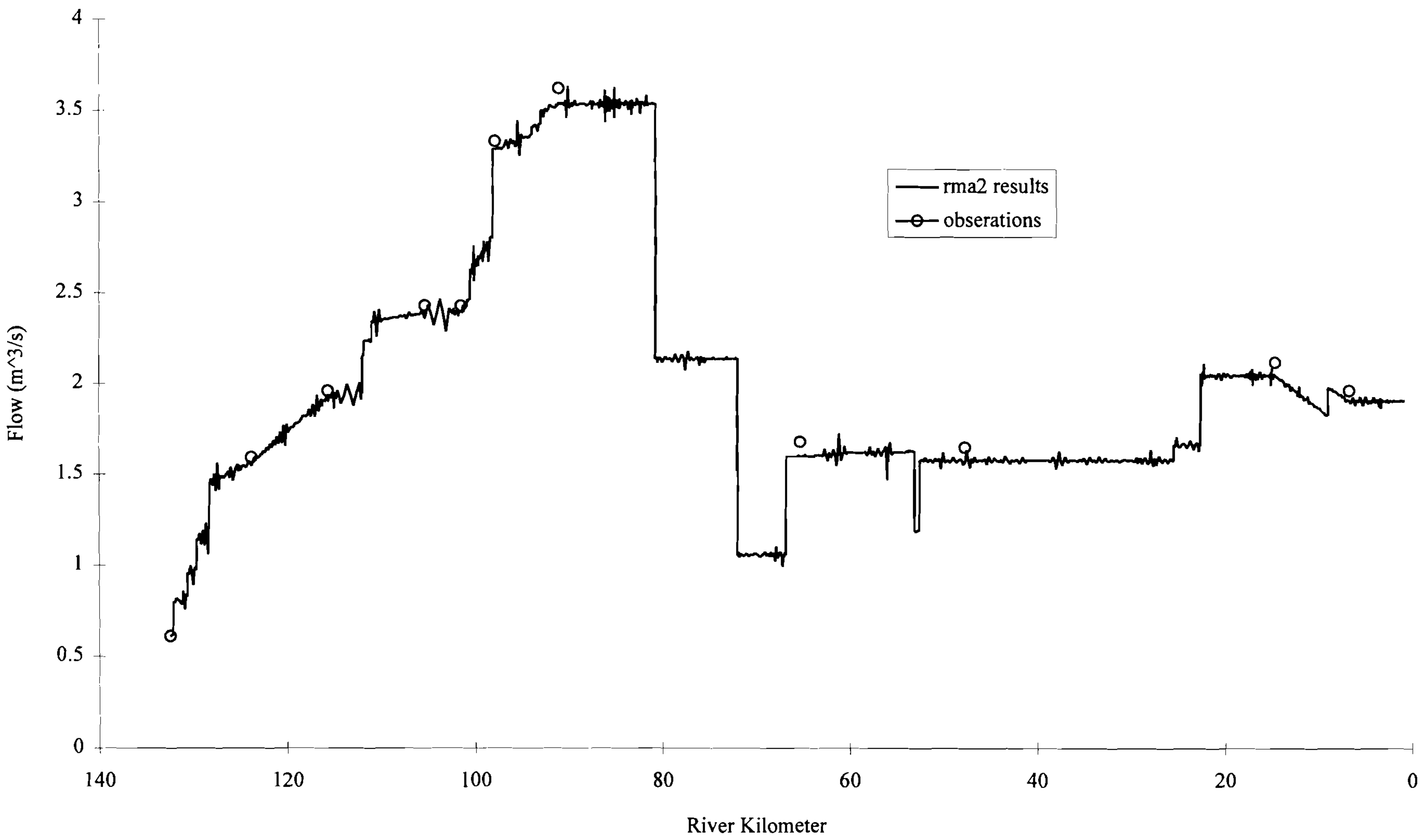

Figure 22: Temperature results from RMA4q after 180 hours of simulation. The solar hour is 12 Noon.

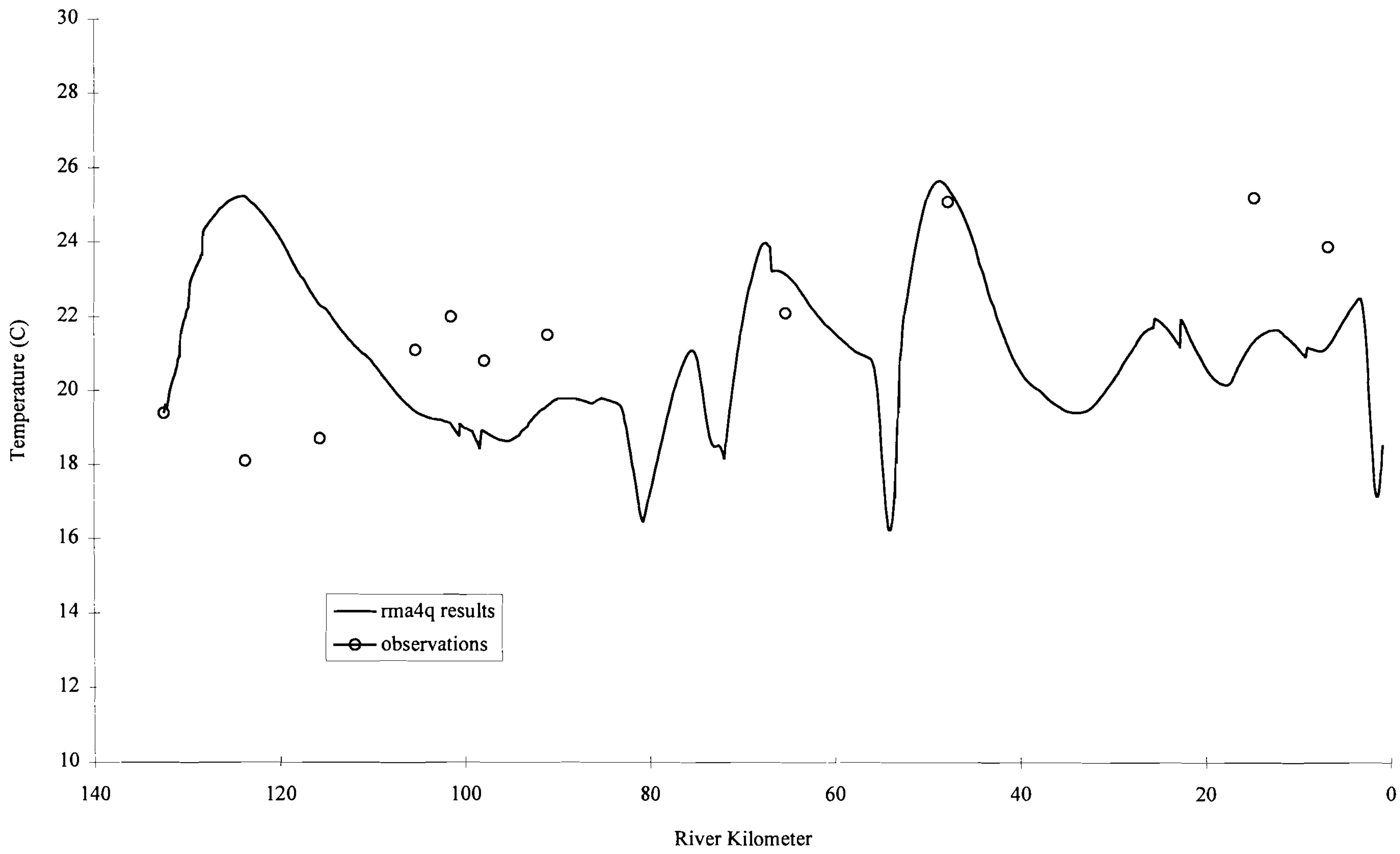

Figure 23: Residual sum of squares for BOD results from RMA4q. BOD decay rate (k1) was varied from 0.1 to 4.0 /day.

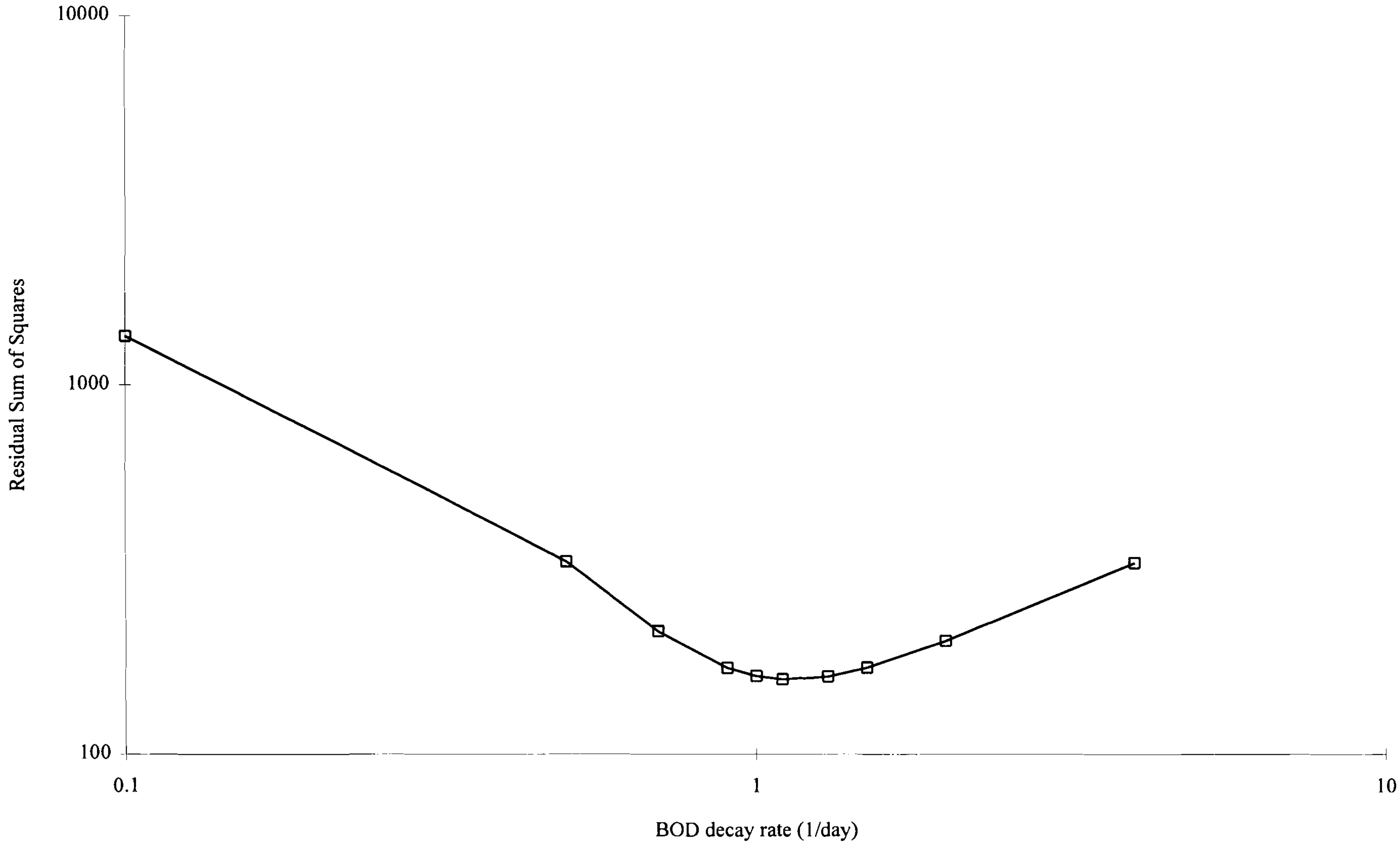

Figure 24: Comparison of RMA4q results after 180 hours of simulation and observations for the Nitra River system. Solar hour is 12 Noon.

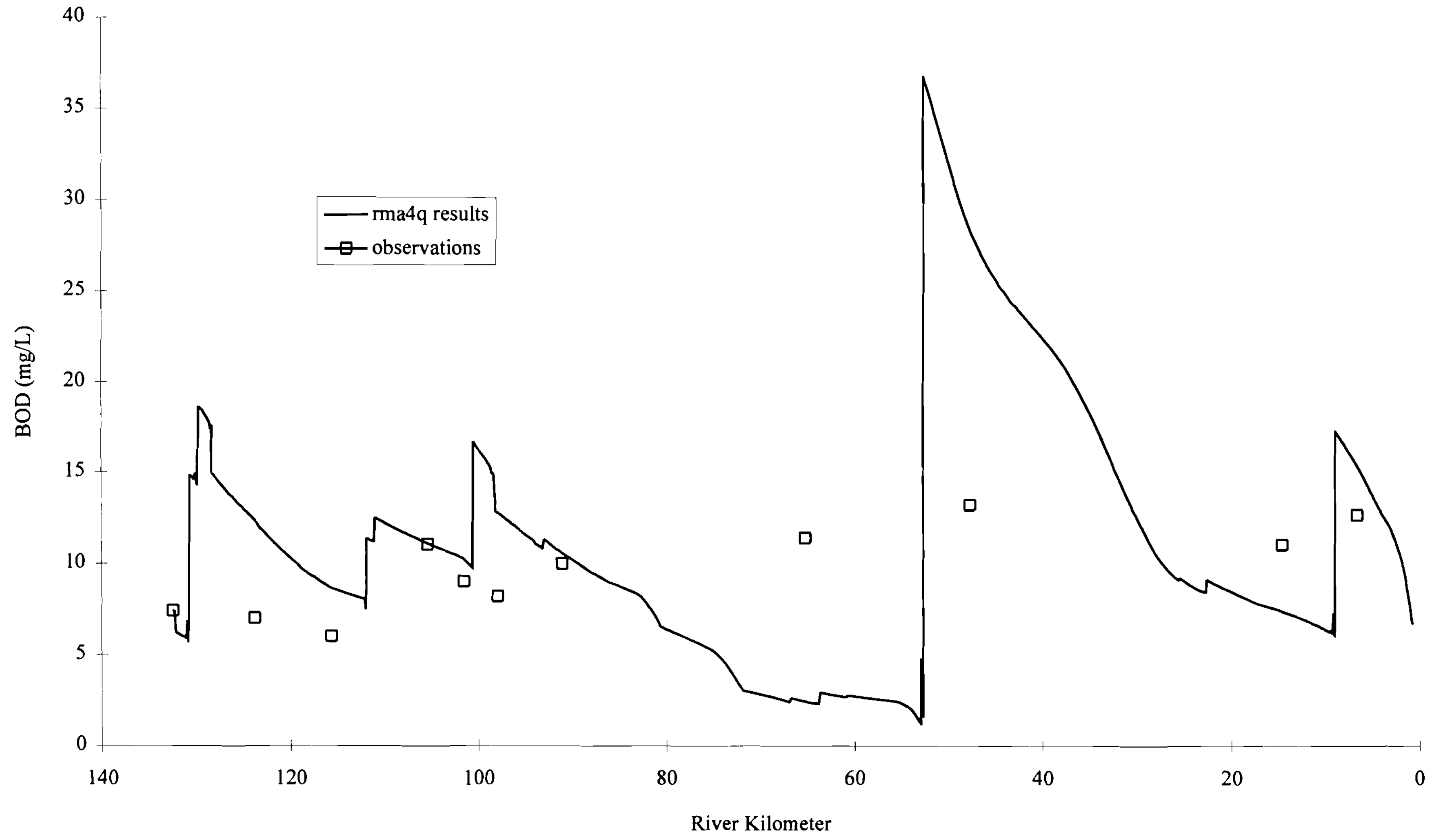

Figure 25: Dissolved oxygen results obtained from RMA4q using two methods for reaeration: O'Connor-Dobbins and direct input of k2. Results are after 180 hours of simulation. The solar hour is 12 Noon.