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Somlyody, L., Masliev, I., Petrovic, P. and Kularathna, M.

IIASA Collaborative Paper
March 1994

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I. Masliev
P. Petrovic
M. Kularathna

CP-94-2
March 1994

IIASA International Institute for Applied Systems Analysis □ A-2361 Laxenburg □ Austria
Telephone: +43 2236 715210 □ Telex: 079 137 iiasa a □ Telefax: +43 2236 71313

VÚVH Výskumný ústav vodného hospodárstva, Nábr.L.Svobodu 5, 812 49 Bratislava
Tel.: +42.7.343111, Fax: +42.7.315743

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L. Somlyódy
I. Masliev
P. Petrovic
M. Kularathna

CONTRIBUTORS

<u>Slovak Republic: Water Research Institute (VUVH)</u>

V. Fekete	water management
J. Jasan	water supply
J. Jokelová	hydrology and experiments
M. Kuníková	water quality, emission inventory, and water quality profile measurement
A. Lapšanský	GIS support
J. Lehocký	water quality and longitudinal profile measurements
M. Lukác	hydrology, hydraulics, and experiments
J. Matulík	water management and planning
P. Petrovic	Slovak Project Coordinator hydrology, water management
H. Zajícová	water quality, longitudinal profile measurement (participant of IIASA's 1992 YSSP program)

<u>Slovak Republic: Slovak Technical University</u>

J. Námer	wastewater treatment

<u>IIASA</u>

M. Kularathna	water quality control policy
I. Masliev	water quality and its modeling, monitoring, longitudinal profile measurement
L. Somlyódy	principal investigator

<u>International collaborators</u>

S. Breithaupt	water quality modeling	University of California at Davis
H. Koivusalo	water quality modeling	Helsinki University of Technology
S. Murcott	wastewater treatment	Massachusetts Institute of Technology
C. Paulsen	water quality control policy	Resources for the Future
O. Varis	water quality modeling	Helsinki University of Technology

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PREFACE

In the spring of 1992, a collaborative research program funded by the Committee of Systems Analysis of the earlier Czech and Slovak Federal Republic (CSFR) and the Ministry of Forest and Water Management (now Ministry of Soil Management) of the Slovak Republic (together with the Slovak Commission for Environment) was launched to analyze pollution problems in the Váh and Nitra river basins and to prepare associated management strategies. The participants were IIASA, the Water Research Institute in Bratislava (VÚVH), and the Váh River Basin Authority.

In the beginning, only relatively broad goals were formulated for the study, with the aim to define specific objectives during the first phase of the project. As a result, a decision was later made to concentrate only on the Nitra sub-basin. The Nitra is perhaps the most highly contaminated catchment in Slovakia, and it clearly demonstrates the need for the development of a comprehensive pollution control policy. From IIASA's side, the river was not only considered as an important, specific problem but also as a case study for the ongoing project on the Water Quality Management of Degraded River Basins in Central and Eastern Europe (CEE), a project having broader policy and methodology oriented objectives.

Stemming from the current scarcity of financial resources for environmental management in CEE countries like Slovakia, the development of short-term, least-cost strategies which can later be flexibly extended as economy improves was formulated as the major task of the project . The adopted systems approach required the integration of knowledge and data from a number of fields such as hydrology, hydraulics, water chemistry, hydrobiology, wastewater treatment, and environmental economics. The approach also required the application and development of various methodologies, models, and tools.

It was evident from the very beginning that the joint analyses of independently collected data (gathered without the aim of later integration) on the basis of different principles and methods would cause significant difficulties. The often infrequent and error corrupted information plus several other issues called for a special focus on properly handling uncertainties. It was also clear that available information was not sufficient to relate emissions (and their required reduction) and ambient water quality or to calibrate water quality models describing the above load response relationship (a crucial element of any strategy development). For this reason, significant effort was devoted to perform so called longitudinal water quality profile observations. Although the design of such experiments was done jointly (keeping in mind policy oriented goals of the study), the difficult task of the execution relied upon our colleagues from Slovakia. Similarly, detailed experimental work was necessary to test the applicability of some innovative wastewater treatment upgrading methods to be considered as future alternatives.

The project required the application and development of a significant number of methodologies and tools such as river basin hydraulic and water quality models, techniques of parameter estimation and uncertainty analysis, various kinds of optimization, database management, mapping, and interfacing. Whenever possible, the above methods were developed in a generic fashion to also serve the broader goals of IIASA's project on the management of degraded river basins.

Dr. Pavel Petrovic
Slovak Project Coordinator
Water Research Institute (VUVH)

Professor László Somlyódy
Leader
Water Resources Project

ABSTRACT

The Nitra River is a tributary of the Váh which enters the Danube downstream of Bratislava. The watershed area is slightly larger than 5000 km^2, and more than 650000 inhabitants live there. The quality of the river is one of the poorest in Slovakia due to numerous municipal and industrial discharges and the low level of wastewater treatment. The ongoing economic transition and shortage of financial resources for environmental management call for the development of regional short-run, least-cost policies. The development of such policies was the main objective of this joint study with the participation of IIASA, the Water Research Institute (VUVH, Bratislava) and the Váh River Basin Authority.

The present state of emissions and water quality was evaluated on the basis of available, routine types of information (including observations from the basin-wide water quality monitoring network) and additional data collection. It was found that industrial discharges form problems which can be handled mostly locally with a straight-forward strategy. In contrast, the management of municipal discharges - representing about 70% of the total BOD5 emission in the catchment - is a more complex issue requiring the development of a regional policy.

The definition of ambient water quality criteria (or the usage of a combination of ambient and effluent criteria) reflecting water use is a pre-requisite of the establishment of a least-cost policy. Thus, the application of water quality models is necessary to relate emissions to receiving water quality (as well as their changes). Due to the nature of the problem, a number of oxygen and nutrient balance models were used, ranging from the traditional Streeter-Phelps model to the latest version of U.S. EPA's QUAL model family. The models were calibrated and validated on the basis of two comprehensive longitudinal water quality profile observations. These observations were gathered under low-flow conditions to correspond with the design requirements of the strategy development. Due to the presence of uncertainties of different origins, the methodology of Hornberger, Spear, and Young (based on the so-called "behavior definition") was applied for parameter estimation of simpler models which then were directly incorporated into an optimization model. This optimization model was based on dynamic programming, utilizing structural features of river basin water pollution problems.

Elements of the water quality control policy model or decision support system (including the linked hydraulic and water quality model(s), the parameter estimation and uncertainty analysis routines, the dynamic programming, the database, the graphical user-interface, etc.) were developed in a rather generic fashion to allow a transfer from one watershed to another. This philosophy corresponds to the broader goals of IIASA's Water Resources Project dealing with issues of the management of degraded river basins in Central and Eastern Europe and the development of associated methodologies for which the Nitra River served as a case study.

Starting from the existing municipal wastewater treatment facilities, a number of alternatives were developed for each site on the basis of various combinations of well-proven physical, biological, and chemical processes to which different effluent quality (BOD-5, TP, NH4-N, NO3-N, etc.) as well as investment, operation, maintenance, and repair costs belong. The technological alternatives (and their major parameters) serve as input to the management optimization model. A special focus was devoted to phased plant development and innovative,

cost-effective upgrading of highly overloaded plants by adding chemicals in low dosage. The issue of upgrading was also experimentally analyzed by jar tests at different treatment plants.

The objectives of the policy model were formulated in terms of minimizing the total annual cost or the investment cost. Constraints might incorporate ambient water quality (characterized by DO, BOD-5 and NH4-N), effluent criteria, and/or minimum level of treatment. The derived least-cost policies were compared to policies based strictly on effluent criteria and to those based on the application of "best available technology." The effluent criteria based policy stems from the new Slovakian legislation if its ambient criteria element was excluded (the legislation defines the simultaneous usage of effluent and ambient criteria and an eleven-year long transition period after which more stringent standards should be met).

The role of industrial emissions was demonstrated in a sensitivity fashion, while the influence of parameter uncertainty on the developed policies was analyzed by an a posteriori Monte Carlo simulation and a multi-objective assessment. The study shows that significant cost savings are possible in comparison to uniform, effluent standard policies. They also suggest that a long-term strategy should be realized on the basis of a sequence of properly phased least-cost policies corresponding to ambient (or regionally variable) standards to be tightened gradually as financial resources become available.

1. INTRODUCTION

The Nitra River is a tributary of the Váh which enters the Danube downstream of Bratislava. The catchment area is 5140 km^2 and 653300 inhabitants live there. The length of the river is slightly below 200 km. The mean streamflow near the mouth is 22.5 m^3/s, while a typical August low flow is about 3 m^3/s. The region is highly industrialized with a low level of wastewater treatment. Inadequate and/or partial treatment also characterizes the municipalities. Most of the municipal wastewater treatment plants are overloaded significantly (by 100% or more).

The water quality of the river is one of the poorest in Slovakia. According to the existing evaluation system where Class V indicates the worst quality, the Nitra's water quality is categorized as Class IV-V. As a result, water use is restricted to water abstraction for industrial and irrigation purposes. However, the primary utilization of the river system is actually waste disposal.

The economy of the country is undergoing an intensive transition process, and resources for environmental and water management purposes are scarce. As legislation is concerned, a new system of standards was set in 1993. It incorporates both effluent standards (distinguishing two stages - before and after 2005) and ambient water quality standards, without specifying how this mixed system should be used during the future transition period. At the same time, no analysis was made to determine the economic consequences of realizing a strict effluent standard system. In addition, the possible gains of an ambient criteria based least-cost strategy (to which regionally variable effluent standards would belong) was not considered. This is not a surprise since rather detailed river basin studies would have been required to address the above issues.

The effluent quality based legislation used in most Western countries leads to uniform emission reductions at all sites. The development of such a policy is a simple task, and both the ambient water quality and the costs directly follow from the standards. Enforcement is also straightforward. In fact, the actual impacts and costs are often of little interest or unknown in advance. It is generally assumed that receiving water quality will be "good" if stringent effluent criteria were selected and money is available to realize the strategy (i.e. the society is willing to pay for a safe environment). The choice of technology is also a side-effect of this system since standard values are most frequently set on the basis of a few (or one) well proven technologies (e.g. secondary biological treatment in the U.S.).

Western countries achieved remarkable improvements in the state of the water environment during the past two decades or so, but it was not a cheap process. For instance, the management of municipal emissions alone necessitated 1% of the GDP in several countries for the above period. Furthermore, meeting of the requirements of the European Community will need quite an amount of additional, costly actions in most of the member countries.

The situation is rather different in the CEE countries including Slovakia. The per capita costs of addressing rather serious water pollution problems can be estimated to be a couple of thousand USD which exceeds the annual specific GDP (GDP in these countries is roughly 1/5 to 1/10 of Western European values). The economies of the CEE countries are presently in a rather bad shape. National debts are high, the production is still declining, inflation rates are significant, and unemployment is about 15%. Increasing prices and decreasing salaries, breadand-butter worries, and the need to restructure the economy as fast as possible puts environment at a low priority for both the public and various governments. One or two percent of the GDP are simply not available for municipal emission control (and due to the difference in per capita GDPs between the West and East, a much larger with respect to the income investment would be needed to have an equally rapid improvement as in Western countries). Economic recovery will unfortunately be a slow process; it suffices to refer to the slow development of Portugal or Spain or to the fact that a doubling of GDP under a 5% growth rate would require about twenty years.

Thus, our conclusion is that current Western policies are not really feasible for the CEE nations. The East should look for strategies which can be characterized by cost-effectiveness on the short run and can be further expanded as financial conditions improve. This statement also applies to the Nitra River basin. For this reason, the major objective of the present research is to analyze various policies and to identify those which are realistic under the present tight budgetary conditions. A special focus will be devoted to municipal emission strategies which form a main element of the Nitra problem. These least-cost strategies are non-uniform in nature, and their development requires the application of clever methodologies including water quality models relating emissions (and their reductions) to receiving quality (and its improvement). Several other objectives of the study are interrelated to the above policy goal and they will be summarized in Chapter 4.

The report is organized as follows (see also Chapter 5 for further structural details). Chapter 2 offers an overall characterization of the watershed, while Chapter 3 discusses major issues of hydrology and water management. This logically leads to the definition of detailed objectives for the study and the approach to be applied (Chapters 4 and 5). Chapter 6 deals with emissions of different origins since the preparation of an inventory is a pre-condition to developing any sensible control policies. In turn, Chapter 7 considers the impact side, i.e. it gives a comprehensive evaluation of the water quality on the basis of observations of the basin-wide monitoring network. Issues of water uses, water quality classification, various standard systems, and legislation are also treated here in a detailed fashion. The chapter also deals with revision of the water quality monitoring network.

Chapter 8 summarizes results of two longitudinal water quality profile measurements performed in August 1992 and April 1993. The first one covered the entire river basin, while the second one focused on three regions for more detailed sampling. The observations were crucial to calibrate and validate the water quality simulation models. Several model versions (Chapter 9) were applied which describe households of dissolved oxygen and nutrients which form major regional problems for the Nitra River basin. Models ranged from the simple Streeter-Phelps to one of the latest releases in the U.S. EPA's QUAL model family. Chapter 9 discusses the hydraulic sub-component and its calibration, the different water quality models, the methods employed for parameter estimation and uncertainty analysis, and the actual calibration and validation by using data from the longitudinal profile measurements.

Chapter 10 offers a state-of-the-art discussion on water quality management (optimization) models and selects dynamic programming as a particularly well suited method to address river basin problems. The advantages are due to the decomposition features and the discrete nature of the decision variables which represent different wastewater treatment alternatives. Details of the code are also given here.

The subject of Chapter 11 is rather different; it outlines various physical, biological, and chemical municipal wastewater treatment processes and their combinations for both upgrading and constructing treatment facilities. This chapter also summarizes laboratory, jar tests performed at three treatment plants to analyze the applicability of low-dosage chemical enhancement of existing overloaded plants. As a major input to the decision model several alternatives are worked out for each treatment plant. These alternatives are characterized by effluent water quality, and costs of investment, as well as operation, repair, and maintenance.

Chapter 12 introduces the prototype decision support system which integrates all the elements discussed in the previous chapters. Chapter 13 deals with the actual policy development. It analyzes effluent standard strategies and compares them to least-cost policies based on ambient water quality criteria (or a mix of effluent and ambient standards). The influence of major elements and parameters of the decision model is shown by a comprehensive sensitivity analysis. The role of parameter uncertainty is evaluated in a multiple objective or regret analysis fashion. Additionally, an *a posteriori* Monte Carlo simulation is also performed for the optimal policy to illustrate bounds of longitudinally variable water quality components and the violation of ambient standards. The chapter summarizes the major elements and features of the recommended least-cost policy. Finally, the report is completed by conclusions and recommendations.

The report contains a set of appendices with supplementary materials which were kept out of the main text for brevity and clarity of discussion. Appendices are numbered according to corresponding chapters, e.g. Appendix 7.1 is the first Appendix to Chapter 7.

This report had been prepared in English in the Water Resources Project of IIASA and translated to Slovak language in the Water Research Institute in Bratislava.

2. GENERAL CHARACTERIZATION OF THE WATERSHED

The Nitra River is a left tributary of the Váh River, and its basin lies in the western second quarter and southern half of the Slovak Republic (Fig. 2.1). The boundary coordinates are 47°46' and 48°59' N and 17°49' and 18°49' E. The total area of the Nitra catchment is about 5140 km^2.

The Nitra river head is situated on the southern slopes of the Malá Fatra mountains. Originally, the Nitra was a left-sided tributary of the Danube. At present it flows into the Váh river, which in its turn mouths to the Danube near Komárno. A shortcut from Nové Zámky to Komoca on the Váh river was built in 1971. The highest point at the water divide is Vtácnik at 1346 m above Baltic sea level (a.s.l.). The length of the Nitra river to the "new" mouth in Komoca is 196.7 km (as compared to 242.6 km to the original mouth). The total length of all the streams in the basin (including tributaries of various orders) is 7300 km, which represents the mean density of the river drainage 1.42 km per km^2 (Porubsky, 1991).

Total population in the basin is 653300, and it is relatively concentrated with 42% in towns of more than 20000 citizens and 50% in towns of more than 10000 inhabitants (larger municipalities with wastewater treatment plants are illustrated in Figure 2.1). Nearly 40% of the population lives in villages of fewer than 1000 inhabitants and in rural areas of low infrastructural development causing serious strategic problems in a broader sense for the future.

2.1 Geology

The whole drainage basin of the river Nitra is geologically formed and extends on basic rocks of the Danube basin and Central-Carpathian mountains. The oldest formations are crystalline schists of Paleozoic mountains, and the youngest are quaternary sediments (Kollár, 1976).

The geological conditions have very large variability (for details see MLVH, 1976). The oldest rocks in Povazský Inovec are crystalline schists, gneisses, diorites and granodiorites. Crystalline schists occur generally in mountainous regions, together with intrusive granitoids, biotite pad micaceous gneisses, granitoide rocks, and others. The frame of the Central-Carpathian zone is created by Mesozoicum with its rocks and crystalline rocks. Mesozoicum on the map of the Nitra river basin has a rather diversified geological structure and non-uniform development.

Mountain ranges extending along the eastern and southeastern part of the Danube lowland are composed of neogene volcanic zones. The following rocks may be found in neovolcanic complexes: andesites, tuffs, and tuffites of the aforementioned rocks. Lowland and flatland territories are formed of sedimentary neogene and quaternary sediments. In the alluvial plains of the Nitra and its larger tributaries and in the Danube lowland, mainly fluvial sediments occur. Their extent, area, and thickness vary with localities, the largest being found in lower parts of the catchment. They are composed of gravels and sands, covered with all kinds of clays and fine flood loamy-clayey sediments.

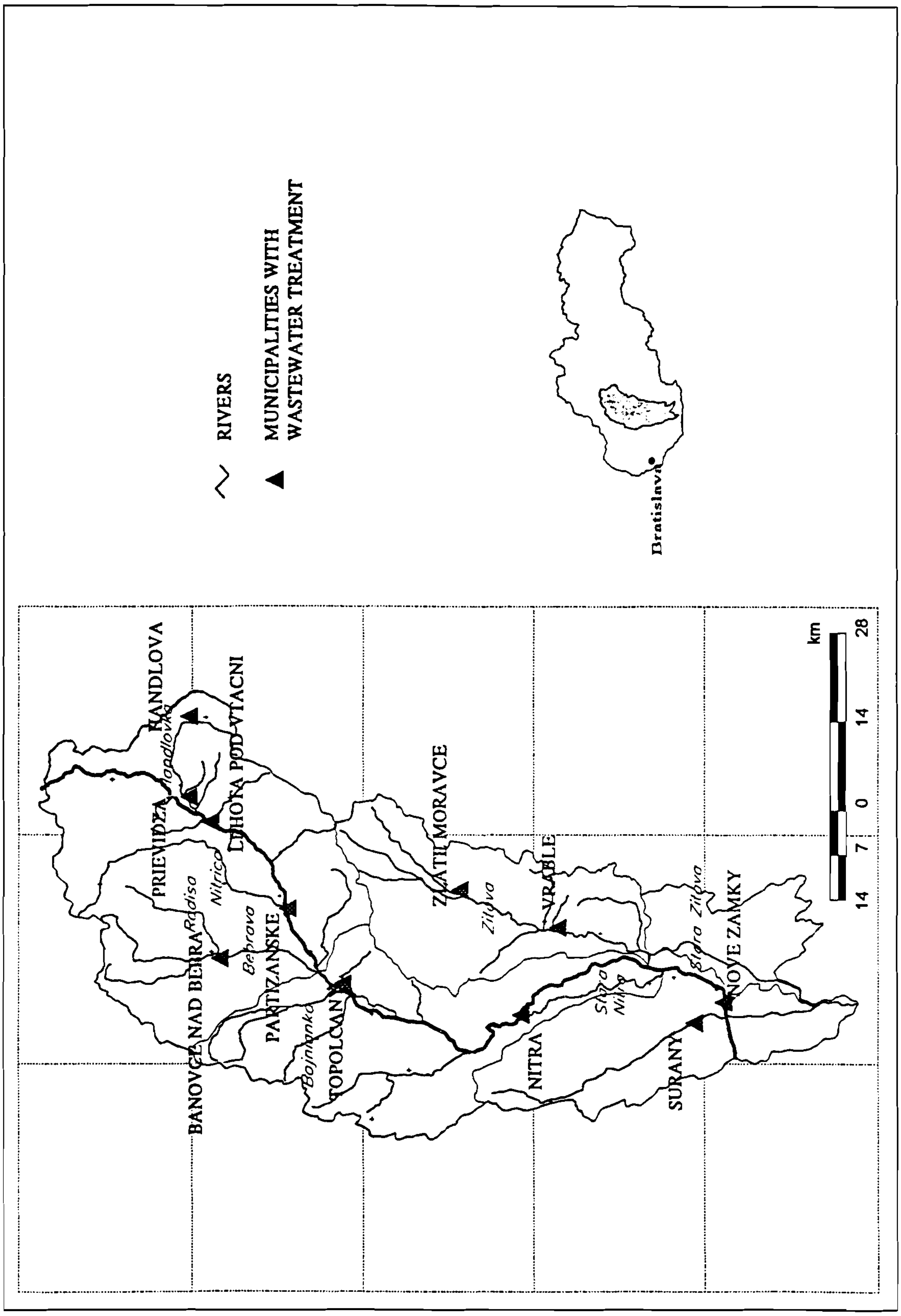

Figure 2.1 Nitra River basin and municipalities with wastewater treatment

The chemical composition of groundwater (and surface waters) in the upper Nitra is primarily influenced by the mineralogical and petrographic character of the rock through which it circulates. The water in the Nitra river basin is a mixture coming from different parent materials. Waters of crystalinic and neovulcanic rock have a silicatogenic composition. Water which is mineralized by limestone and dolomite solution is carbonatogenic, and it belongs to the water of mesosoic carbonate and basal paleogene. Transient silicatogenic-carbonatogenic water is seen in the water of sedimentary neogene and in the majority of the quaternary mountain sediments. A separate genetic category is found in the fluviogenic water of the river valley and in the lower part of the watershed: the mineralization of this water is significantly influenced by the infiltrated water from the river system and the effect on the chemical composition of the water from the underlying rock is reduced.

2.2 Climate

According to the Czechoslovak climatic classification (which is still valid) the Nitra river basin can be divided into three main regions:

1. Warm region: more than 50 days with maximum temperature over 25°C. This region includes a part of the Danube Lowland and the lower part of the Upper Nitra Hollow.

2. Medium warm region: fewer than 50 days with maximum temperature over 25°C and a mean July temperature over 16°C. This region includes the rest of the Upper Nitra Hollow and mountains lower than 800 m a.s.l.

3. Cold region: mean July temperature from 12°C to 16°C. This region includes mountains higher than 800 m a.s.l. The richest precipitation is usually brought by southern Mediterranean air advection.

The smallest total amount of precipitation is observed in the lowland, situated along the lower stream of the Nitra and the Zitava rivers (Fig. 2.1), where the mean annual precipitation is between 540 and 600 mm. In the Upper Nitra Hollow the annual precipitation varies from 650 to 800 mm, while in the mountainous regions of the basin it is between 650 and 900 mm. Maximum values of about 1200 mm/y are observed on the mountains tops. The long term (1931 - 60) areal mean of precipitation is 713 mm; it varies between 486 (1947) and 946 (1939) mm (Petrovic and Venetianerova, 1973). The yearly course has one significant maximum in July and a secondary one in November. Long-term monthly areal averages are between 43 and 66 mm (the driest month was October 1944 with only 2 mm; the wettest was July 1960 with 202 mm).

The mean areal evapotranspiration (Petrovic, 1974) evaluated from meteorological input for the whole watershed (using a model approach with a monthly time step) is 575 mm (with a range of 442 mm to 759 mm). Annual potential evapotranspiration for the lowest elevation of the watershed was estimated as an average of 732 mm (with a range of 653 mm to 867 mm).

2.3 Hydrology

The hydrologic regime is influenced by low precipitation with nonuniform distribution over the year, by high evaporation (especially in the summer), and partially by existing geology in the watershed. Seasonal variation of runoff is affected primarily by snow accumulation and melting. The maximum mean runoff occurs in spring and the minimum values in September and in winter time. The absolute minimum discharge observed at the Nitra mouth was 2.4 m^3/s (compare to values for the Zitava river of 0.015 m^3/s, the Nitrica 0.026 m^3/s, the Bebrava 0.48 m^3/s and the Radosinka 0.07 m^3/s--see Figure 2.1--Molnár, 1993). Floods are mostly consequences of snow melts. The peak flow occurring once in ten years is about 340 m^3/s.

The mean (1931-1960) annual runoff sum is 570 x 10^6 m^3 (measured at Nové Zámky), corresponding to 111 mm and a mean runoff coefficient of 0.16 (Petrovic and Venetianerová, 1973).

2.4 Land use

From the total catchment area of 5140 km^2, about 1400 km^2 is forested, with 15% covered by coniferous forest and 85% by deciduous. The prevailing use of land in the lower part of the watershed is agriculture, occupying 3170 km^2. Primary crops are maize, sugar beets, potatoes, and other plants suitable for the given microclimatic conditions. The intensive agricultural use of soil is supported by irrigation, which is often used on more than 250 km^2 of land. Urban areas cover approximately 150 km^2.

The Nitra catchment has a highly varied industry; nearly a full spectrum of it can be found in the valley of the main river and its tributaries. In the upper part are various mining activities. The middle part has chemical, building, and textile industries, machineries, wineries, tanneries, and thermal power plants. The lower part is characterized by food industries, sugar refineries, and breweries.

Intensive crop production (and irrigation) can be found in the middle and lowland parts of the basin. The restructuring of the former Slovakian socialist economy to a market economy influences the intensity of the agricultural land use. During the last three years reprivatization has started, and agricultural farms and cooperatives are in a strong transition with a significant resulting drop in production. Some cattle and chicken farms are currently using only a part of their full capacity (50-70%). Fertilizer application shows a similar reduction. According to statistical data (for all of Slovakia), in 1990/1991 only 51% (123.1 kg per hectare) of the total active substance of 1989/1990 (about 240 kg per hectare) was used (there was such an overload of fertilizers in the top soil layer that no impact on the crop yield was found until now).

Water-generated electricity forms only a small part of the energy needed in the Nitra river basin. A power plant with a mean annual production of 0.58 GWh is located in Nitrianske Rudno. Water flowing as a coolant from a water reservoir to the thermal power plant in Zemianske Kostolany is used for electricity production. Additionally, two small hydropower plants are installed in weirs in the towns of Nitra and in Jelsovce (with a mean streamflow of about 17 m^3/s). Dolný Ohåj weir at Zitava (with 4 m^3/s streamflow) diverts water to the channel to the Nitra River at Nové Zámky.

2.5 Water availability

Each year the water management administration in Slovakia provides a state water quantity budget and analysis on water quality. The methodology of both evaluations was developed in the VÚVH (see Fekete, 1985 and 1990; MLVH, 1986).

On the basis of criteria used in Slovakian quantitative balance studies, the Nitra River basin is characterized by water stress (see later). The water-stress problem arises from the relatively low runoff coefficient, high demand, and high pollution level. In summer there is a significant water need for irrigation, because soil moisture storage and precipitation do not cover crop water need in intensive agriculture production. In winter the main issue is satisfying industrial water demands during low-flow periods. The quality of surface waters is so poor that it is not suited for drinking purposes, and this water in the Nitra river basin is transferred from different regions of Slovakia (see later).

2.6 Emissions and overall water quality

The Nitra river basin is known as a very polluted area. Municipal and industrial emissions are significant, and past contamination of soil and sediment presents a serious problem. With the exception of the upper part of the river, water quality is classified as poor or very poor (the last two classes of the surface water-quality classification according to the State standard CSN 75 7221). A more detailed description on emissions and the state of water quality is given in Chapters 3, 6, and 7.

2.7 Administrative issues

The administrative management of Slovakia is under intensive re-structuring after creation of a new independent state. The previous "vertical" administrative structure was a three-step control: government, counties (now considered regions), and districts. Often economical management has not been significantly influenced at the local level. However, this situation is likely to change in connection with privatization and decentralization.

The Nitra river basin lies on the territory of two counties - the West Slovak County (Západoslovenský kraj, with districts Galanta, Komárno (this part of the watershed management belongs administratively to the Danube River Authority Office), Levice, Nitra, Nové Zámky, Topolcany, Trnava, and Trencín) and the Middle Slovak County (Stredoslovenský kraj, with districts Prievidza and Ziar nad Hronom). Some basic data of the above regions are given in Table 2.1.

The major institutions at present are the district governments (in greater towns the local authority is considered a district itself). Environmental management is done operationally by a few "regional" offices, which for the territory studied are in Nitra and Topolcany. This structure is going to be changed. Certain administrative simplification will combine all the different authorities under a single state authority at the district/local level. Water management is done on a watershed level (differing from administrative districts). The Váh

River Authority (with headquarters in Piestany and offices in Nitra and Topolcany) is responsible for the Nitra River.

The top decision level in the field of water management includes two ministries. All the problems related to water quantity and quality as well as to legislation are addressed by the Ministry of Environment. Technical tasks dealing with river and water structures (management and control) belong to the Ministry of Soil Management. River authorities are in charge for all the practical works on their territory. They also provide sampling analyses on surface and subsurface waters (within the monitoring program sponsored by the Ministry of Environment). Both ministries rely upon the Water Research Institute (VÚVH) for methodological support. The current basis of water management is the Master Plan proposal from 1976 (MLVH, 1976), which is continuously updated. Starting next year Hydroecological Plans will be also prepared.

Table 2.1 Basic data on the administrative areas in the Nitra river basin

District	number of municipalities	area (km^2)	number of inhabitants
Prievidza	50	923	139,383
Topolcany	113	1,309	160,233
Ziar n. H.	2	59	2,638
Levice	4	85	1,655
Nové Zámky	29	701	107,348
Galanta	3	103	6,145
Trnava	6	73	5,301
Nitra	73	1,434	207,582
Trencín	12	134	8,063
Komárno	7	319	14,937
TOTAL	299	5,140	653,285

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3. HYDROLOGY AND WATER MANAGEMENT

3.1 Hydrology

The Nitra river with its tributaries drains 10.5% of the Slovak territory. The difference in altitude between the river's head and mouth is 690 m. The Nitra River bed gradient is largest in mountainous stretches, where it is 7.3‰. In the Upper Nitra Hollow the gradient is 1.4‰, and in lowland it is 0.3‰. In the Upper Nitra Hollow there are many meanders (caused by the decreasing river bed gradient) and intensive sedimentation. In lower parts of the Nitra and Zitava rivers the sedimentation of finer materials, such as sand and clay, occurs. Due to this sedimentation process the level of the river bed is ever increasing, creating the need for flood levee construction.

3.1.1 Surface Hydrology

As noted in Chapter 2, the length of the Nitra river is 196.7 km, while the total length of streams in the basin is 7300 km, leading to a mean density of the river drainage of 1.42 km/km^2 (Molnár, 1993). The actual density varies between 0.5 km/km^2 and 3 km/km^2, depending mainly on relief features, soil-forming material, and precipitation.

The basic hydrologic characteristics of the major rivers in the basin (see Figure 3.1) are given in Table 3.1.

Table 3.1 Basic hydrologic characteristics of major rivers in the Nitra basin (Porubský, 1991)

River	A	h_p	h_r	c_d	q	Q_a
Nitra	5144	665	151	0.23	4.77	24.50
Nitrica	319	658	128	0.20	4.06	2.85
Bebrava	634	716	194	0.27	6.15	3.90
Radosinka	385	595	98	0.16	3.12	1.20
Handlovka	178	838	333	0.40	10.54	1.88
Zitava	1244	631	128	0.20	4.06	5.05

A - catchment area [km^2]
h_p - mean yearly precipitation total [mm]
h_r - run-off height [mm]
c_d - run-off coefficient
q - run-off per unit of surface [$ls^{-1}km^{-2}$]
Q_a - mean yearly discharge [m^3/s]

The Nitra river has, naturally, the highest runoff in the lowest profile, but the largest runoff coefficient can be found in the Handlovka sub-basin. The highest streamflows usually occur in March due to snow-melt. The flood frequency distributions for the Nitra River and its tributaries are illustrated in Table 3.2. The flow frequency curves for eight sections of the Nitra River are given in Figure 3.2 (refer to Figure 3.1 for the location of the gauging sites).

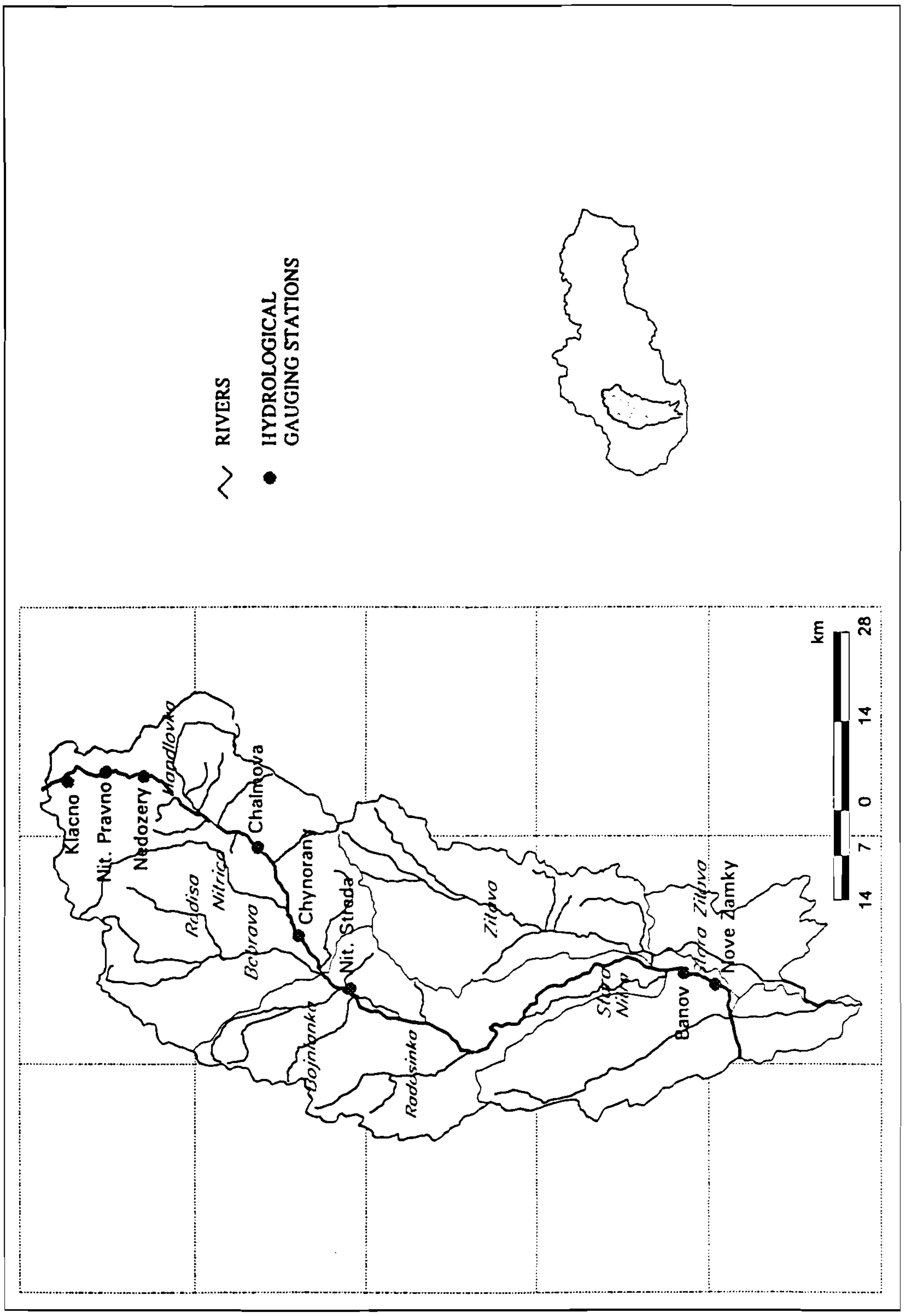

Figure 3.1 Gauging stations on the Nitra River.

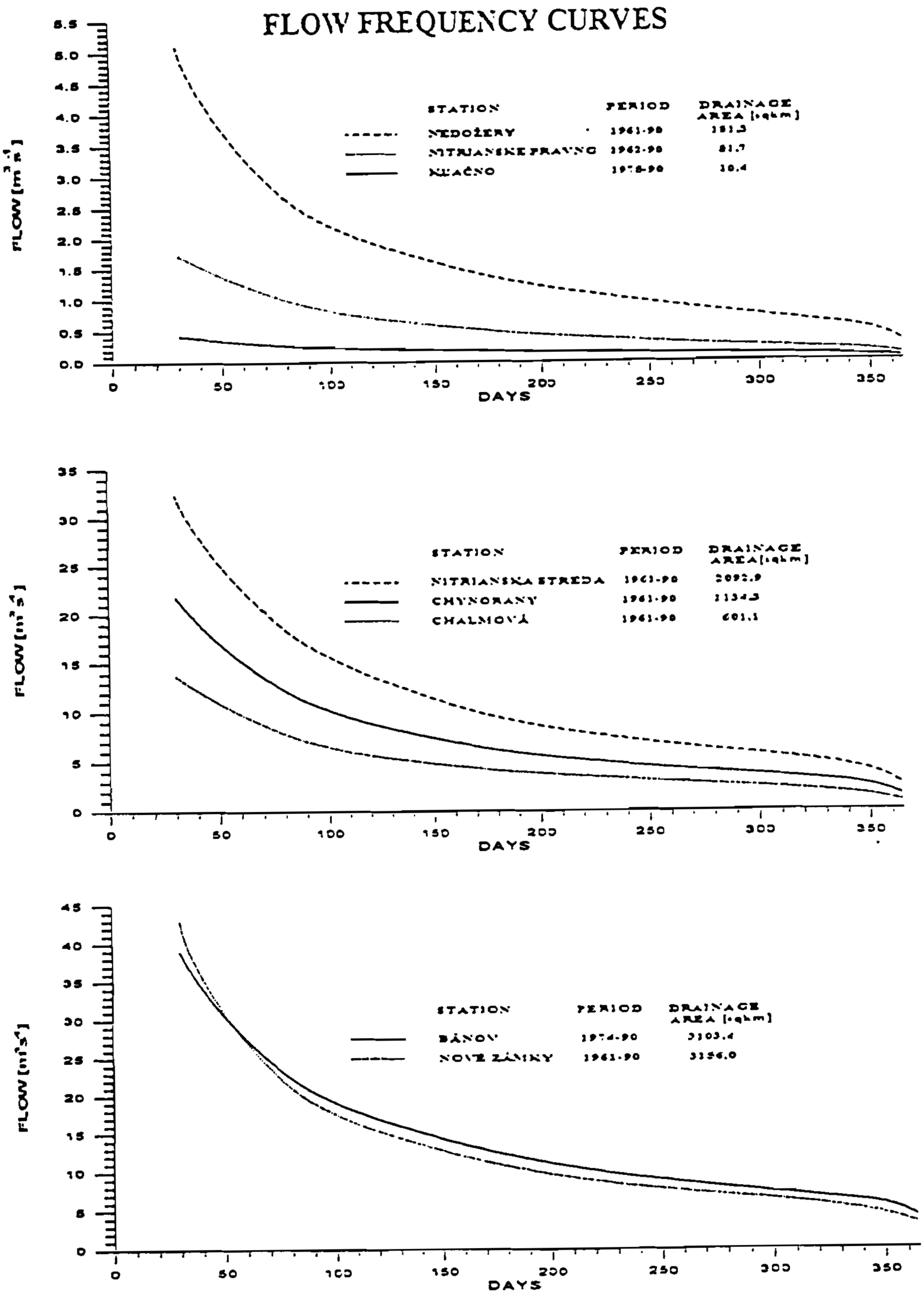

Figure 3.2 Flow Frequency curves for eight sections of the Nitra River.

Table 3.2 Flood frequency probabilities for the major rivers in the Nitra basin (Molnar, 1993)

Streamflow (m^3/s)	Frequency (years)					
	1	5	10	20	50	100
Nitra	170	290	340	375	410	430
Nitrica	51	88	101	113	128	135
Bebrava	73	120	140	160	180	195
Radosina	20	42	50	59	70	80
Handlovka	30	77	92	106	125	140
Zitava	22	53	66	80	100	120

The lowest flows are observed in summer or autumn; the July-October period is the driest. Statistical evaluation of historical data can be used to characterize durability of streamflows at various cross sections. Table 3.3 summarizes flow values reached or exceeded over M days (from 30 to 364 days) in the given (indicated) period. As can be seen, at Nitrianska Streda (being the most downstream hydrologic station with a homogeneous data set, see Figure 3.1) Q_{30} is slightly above 30 m^3/s, while low-flow values Q_{355} and Q_{364}, which can be used as design conditions for water quality control (see Chapters 9 and 13), are 3.7 m^3/s and 2.7 m^3/s, respectively. The low flow frequency curves for Nitrianska Streda and Chalmova (Figure 3.1) can be found in Figure 3.3. The range of flow variation clearly shows the significant impact of dilution on water quality. The seasonal changes of the streamflow are illustrated in Figure 3.3, showing different probability values of the mean monthly flow for Nitrianska Streda on the basis of observations for the period 1961-1992 (1%, 10%, mean, 90% and 99%).

Table 3.3 Mean daily stream flow recurrence intervals for the Nitra river.

Gauging Station	M (days)							
	Period	30	90	180	270	330	355	364
Klacno	75-90	0.43	0.25	0.17	0.13	0.10	0.07	0.05
Nit. Pravno	62-90	1.74	0.90	0.49	0.29	0.20	0.15	0.11
Nedozery	61-90	5.10	2.40	1.34	0.86	0.61	0.46	0.32
Chalmová	61-90	13.80	7.13	4.17	2.88	1.97	1.32	0.88
Chynorany	61-90	21.89	11.13	6.23	4.20	3.14	2.27	1.55
Nit. Streda	61-90	32.50	17.00	9.52	6.50	4.95	3.70	2.70
Bánov	74-90	39.10	20.67	12.30	8.29	6.52	5.36	4.18
Nové Zámky	61-90	43.00	19.20	10.80	7.20	5.48	4.10	3.22

Q_M are given in m^3/s

Period is the time for which Q_M were derivated

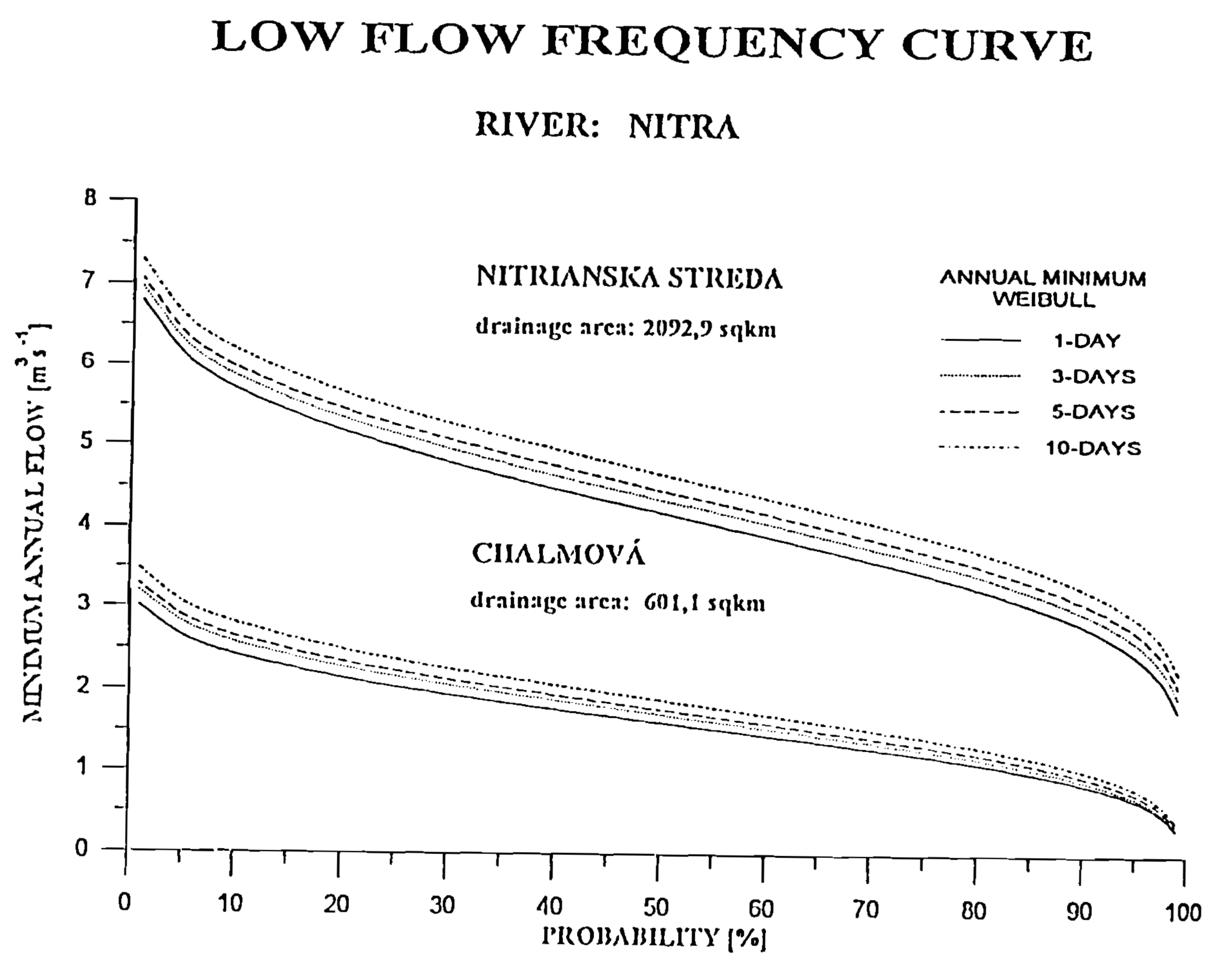

Figure 3.3 Low Frequency curves for two sections of the Nitra River.

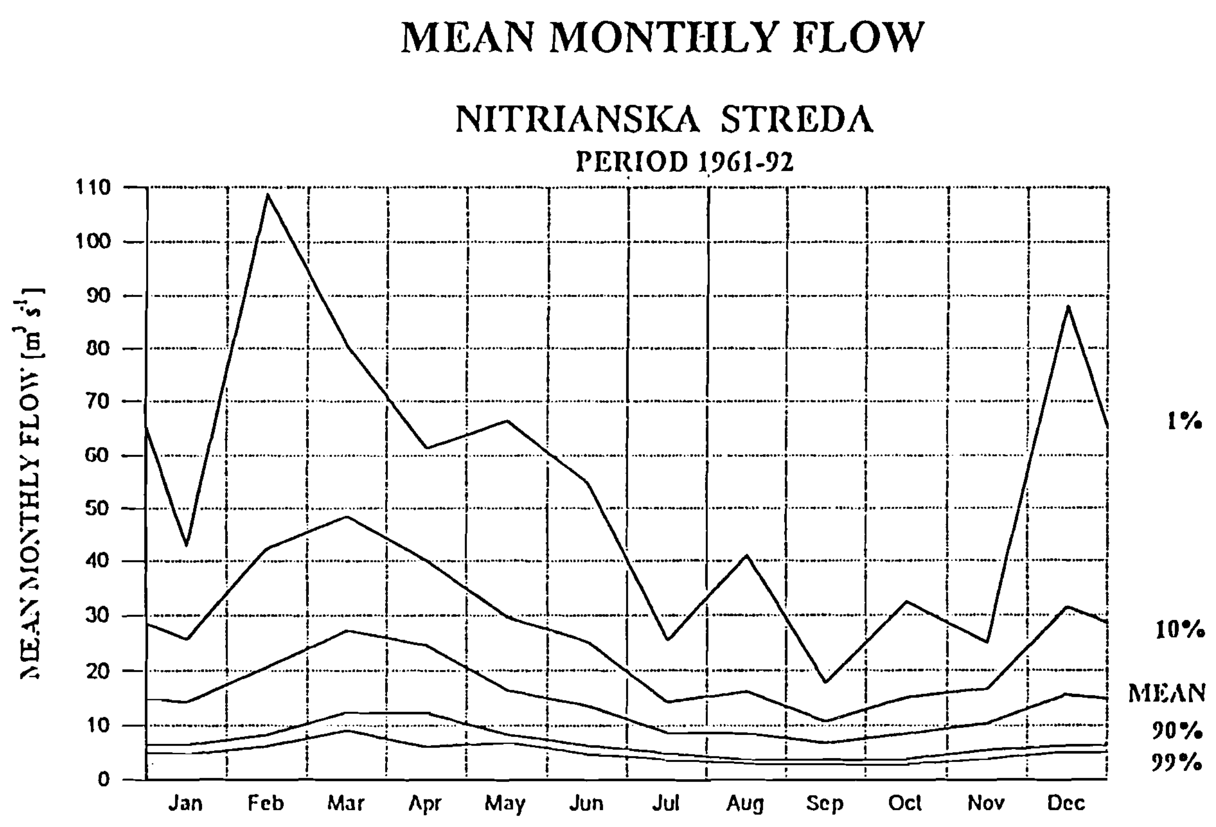

Figure 3.4 Seasonal changes of the mean monthly flow (Nitra River at Nitrianska Streda)

Evaluation of low flow for consecutive days (1 to 10) was performed on the basis of the UNESCO/FREND methodology (Gustard et al., 1989), for each year. Results obtained for the recommended Weibull distribution for a selected number of days (1, 3, 5, 10) are shown in Figure 3.3. They suggest that the 90% probability value of the 10-consecutive-day flow at Nitrianska Streda (which also can be employed for water quality planning purposes) is about 3.2 m^3/s, close to the previously mentioned values.

3.1.2 Subsurface hydrology

Groundwater resources do not play a significant role in the Nitra watershed (though their quality is much better than that of surface waters). Mostly in regions where the thickness of the sediment in the river valley reaches tens of meters the yield can exceed 10-20 l/s. Such areas can be found upstream of the mouth of the Bebrava, near to Vestenice (the Nitrica tributary), between the town Nitra and the mouth of the Zitava, and along the low Zitava. The total amount of water extracted is less than 150 l/s.

3.1.3 Hydraulics and structures

Compared with other Slovak basins (especially the Váh basin) there are only a few hydraulic structures in the Nitra river basin, and they have mostly local significance. Basic hydrological and technical data about major structures in the basin are given in Table 3.4.

Table 3.4 Major hydraulic structures in the Nitra river basin

Location	River	A	Q_{100}	A_F	V_T	V_R	V_S	T
Nitr. Rudno	Nitrica	135.0	110	0.84	3.73	0.45	2.93	ED
V. Uherce	Drahoznica	25.5	22	0.22	1.10	0.19	0.89	ED
Duchonka	Zeleznica	25.1	30	0.14	0.55	0.13	0.11	ED
Prusy	Dubnicka	18.0	40	0.38	1.70	0.10	1.70	ED
Slepcany	Ceresnovy	58.0	30	0.46	1.40	0.20	1.10	ED
Jelsovce	Nitra	2394.3	355	0.20	0.36	-	0.36	W
Nitra	Nitra	2715.4	370	0.15	0.88	0.45	0.43	W
Nováky	Nitra	488.8	190	0.20	0.37	-	0.31	ED

A - catchment area [km^2]
Q_{100}- [m^3/s] (flow exceeded in 100 days of a year)
A_F - maximum flooded area of reservoir [km^2]
V_T - total (maximum) volume of reservoir [$10^6 m^3$]
V_R - retention (minimum) volume of reservoir [$10^6 m^3$]
V_S - storage volume of reservoir [$10^6 m^3$]
T - type of damming construction (ED - earthfill dam, W - weir)

From existing structures only the Nitrianske Rudno reservoir has an important role; it supplies cooling water to the thermal power plant of Novaky at the Nitra River (with an intake from the Nitrica). The other weirs support local water intakes with small storage; thus they are of small importance from the viewpoint of regional water management control and operation.

3.2 Water Availability and Use

The annual average per capita water availability is less than 1000 m^3/cap/y (about 2000 l/cap/d) and an average of 20% of this amount is used. According to different classifications (see e.g. Kulshreshtha, 1993) these values identify a region with marginal water vulnerability, so management is not an easy task. The situation is much worse if we consider summer low-flow periods. For instance, in August 1992 (the second driest August since 1961) the corresponding figures were 650 l/cap/d and 53% usage, which specify a class of "water stress" from the viewpoint of management.

On the availability side the runoff coefficient is smaller than for most other Slovakian rivers, and the low level of quality prohibits many water uses. This lack calls for water transfers from other regions. Public water supply relies heavily upon subsurface waters of high quality distributed from the Gabcíkovo region near Bratislava.

Altogether nearly 1 m^3/s is imported, about 50% from the Gabcíkovo area, and the other half from Jelka and Turcek. Half of the public water is used for domestic purposes. Industry uses 12%, losses are estimated to be close to 20%, and another 20% is unaccounted for.

On the basis of data for 1992, the average water use was 3.1 m^3/s (the return flow represented 67% of it). The distribution according to various sectors was 35% for industry, 6% for agriculture, 15% for energy, and 44% for households. During summer months the proportion of agricultural and household use increases (the latter up to about 50%). Water uses in industry, agriculture (reduced by 50% in 1992 due to ongoing structural changes), and energy production rely almost fully upon surface waters, while public water supply utilizes exclusively groundwater resources. In August 1992 there were several days when more than 70% of the flow monitored at the mouth had already been used. If we also consider availability and demand on smaller scales, it is evident that water management is not a straightforward task under critical low flow conditions.

Both surface and subsurface water intakes are shown (together with emissions) in Figures 3.5 and 3.6.

In summary, surface waters are used for the purposes of industry, agriculture, hydropower, and waste disposal. The quality at most of the locations is so poor that recreational usage and fishing are out of question (though the public may not be aware of the high risk of bodily contact).

3.3 Municipalities

The ratio of public water supply in the entire watershed is somewhat less than 70%. In larger municipalities the coverage is close to 100% (see Chapter 6), while it is still above 80% for settlements in the >500 population domain. The development of the (combined) sewerage network is lower by about 20% than that of supply. The situation is particularly poor in smaller settlements. For instance, wastewater collection is practically absent in villages with fewer than 4000 inhabitants. The major problem, however, is the low level of wastewater treatment (see Chapter 6 for details). Treatment plants exist in no more than eleven municipalities where about 50% of the generated wastewater is treated (see Figure 3.7

showing influent BOD-5 loads and the treated portion). As can be seen from the figure, most of the treatment plants are significantly overloaded; the average annual wastewater flow is double the design capacity. (All the treatment plants are mechanical-biological plants with the high load activated sludge process; that is, no nitrification takes place.) Thus the effluent BOD-5 value can exceed 60-70 mg/l, with only 70% (or smaller) removal rate--a very low figure (these values show one reason for the poor quality of receiving rivers). The contribution

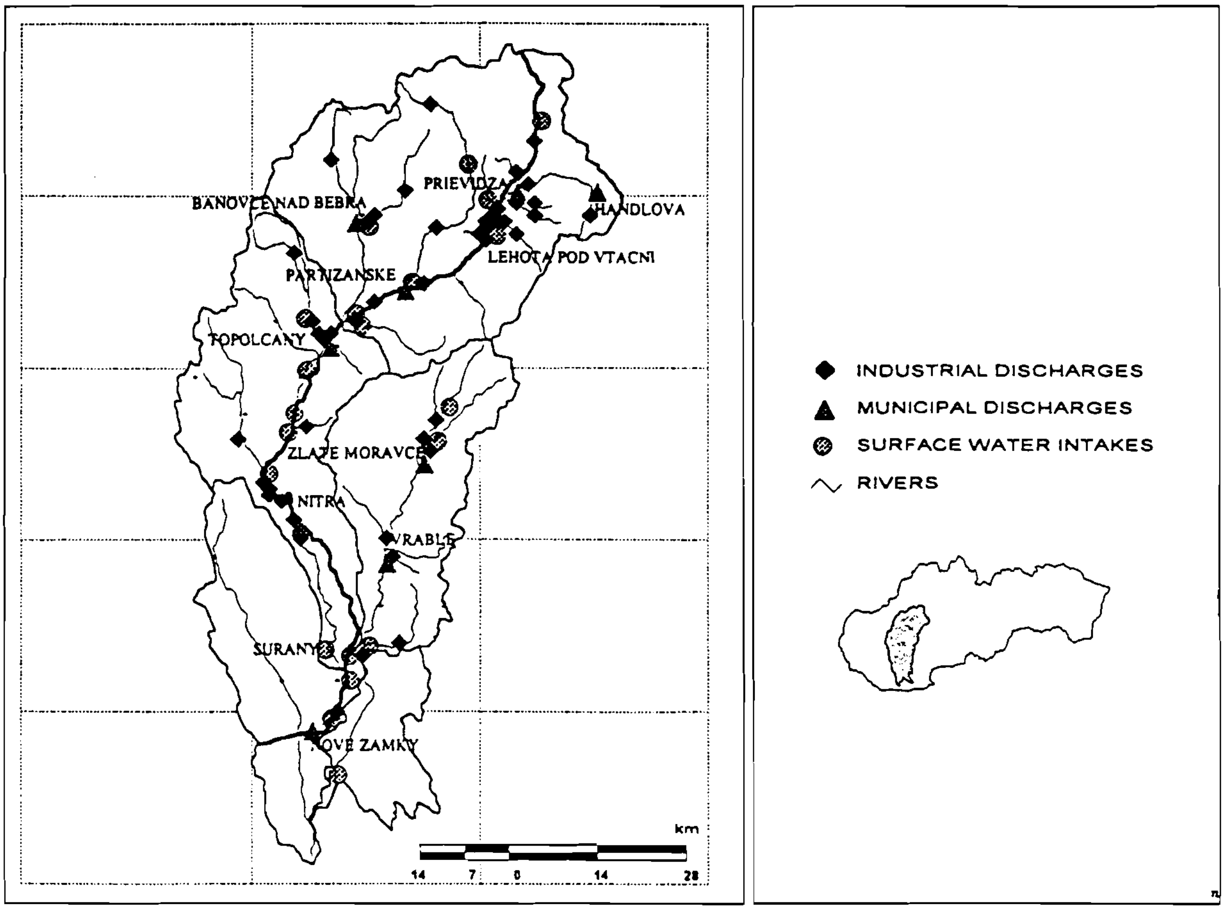

Figure 3.5 Surface water intakes and emissions in the Nitra River basin.

of industrial discharges to municipal wastewaters is high and sometimes close to 50%. It causes problems not only due to the frequent lack of pre-treatment (which leads to difficulties such as sludge disposal), but also because of high flow fluctuations.

Municipalities contribute to about 70% of the total BOD-5 emission in the catchment. This value illustrates their crucial role in water quality management. The rest of the BOD-5 emission comes from industry; the role of agriculture is negligible.

About 70% of public drinking water is used for domestic purposes (~ 200 l/cap/d), while the rest is mostly for industrial uses. As indicated earlier, roughly an equal amount is used from surface intakes, again for industrial purposes. Aged infrastructures leading to high water losses from the distribution network and infiltration to the sewer system, cause general problems in nearly all the municipalities. The needs for stormwater management and sludge treatment and disposal are also issues for most municipalities.

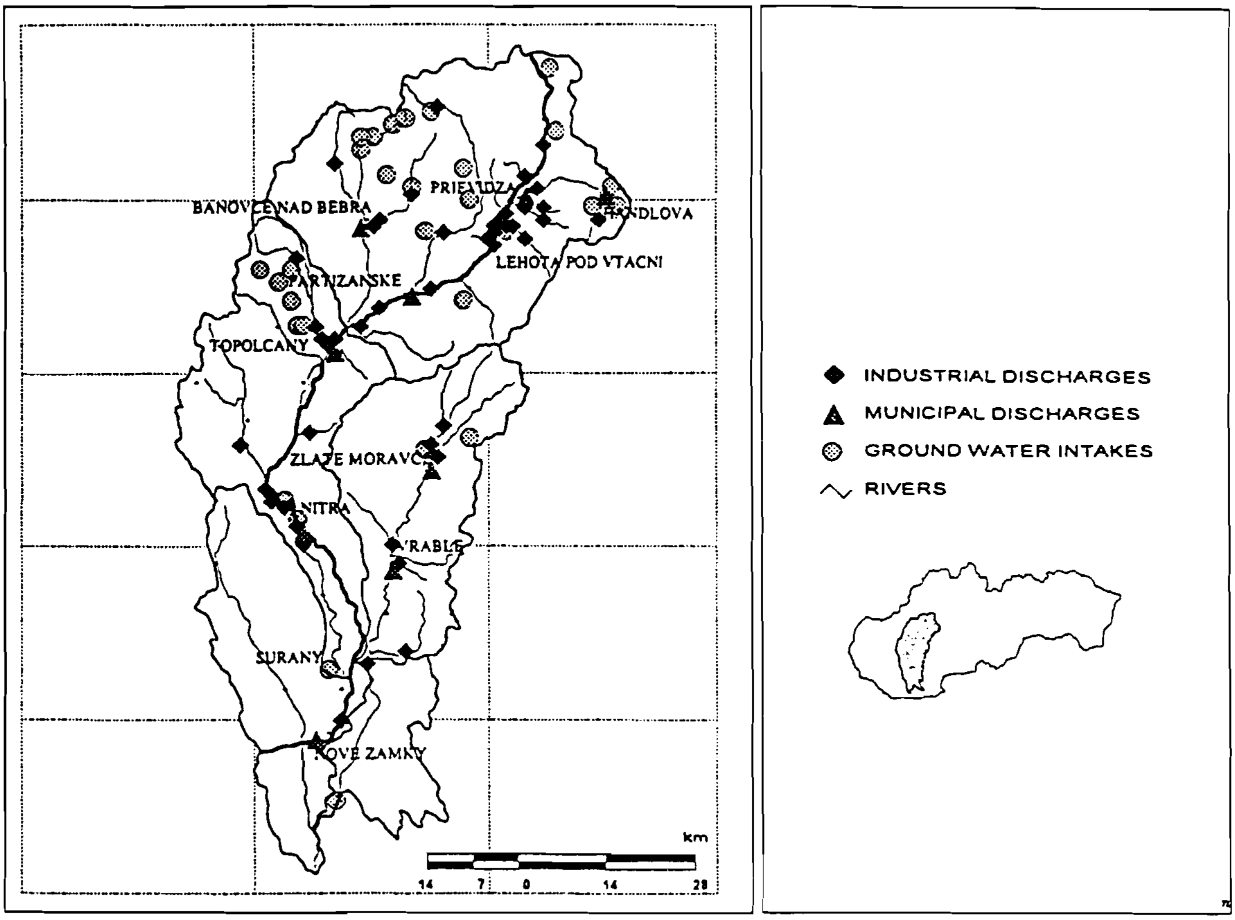

Figure 3.6 Groundwater intakes and emissions in the Nitra River basin.

3.4 Industrial Emissions

Larger industrial emissions of BOD-5 are displayed in Figure 3.8 in a similar fashion to that for municipalities. Major industrial areas/plants include Surany, Nitra, Bosany, Luzianky, Kostolany, D.Ohaj and Chynorany. Several sugar beet factories with seasonal operation (which generally starts in October) can be found in downstream regions of the watershed. Some other industries are tanneries, chemical factories, canning, meat, milk and vegetable processing plants, and wine and spirit production plants. The thermal power plant at Kostolany which uses low-quality brown coal should be mentioned also. Its residual ash has high arsenic content, so this plant is the likely reason for the high arsenic concentrations monitored (see Chapter 7). Wastewater treatment is rather poor, and sometimes missing (e.g. at the sugar beet factories in Surany and Mraziarne Nitra). The treatment is often only mechanical, and discharges frequently go to the municipal sewerage system. In addition to mechanical treatment, a number of mechanical-biological plants can be found. Sometimes chemical treatment steps, oil removal and neutralization are also added. The average BOD-5 contribution of industry to the catchment's total emission is about 30% (though it is somewhat higher in the "campaign" season.

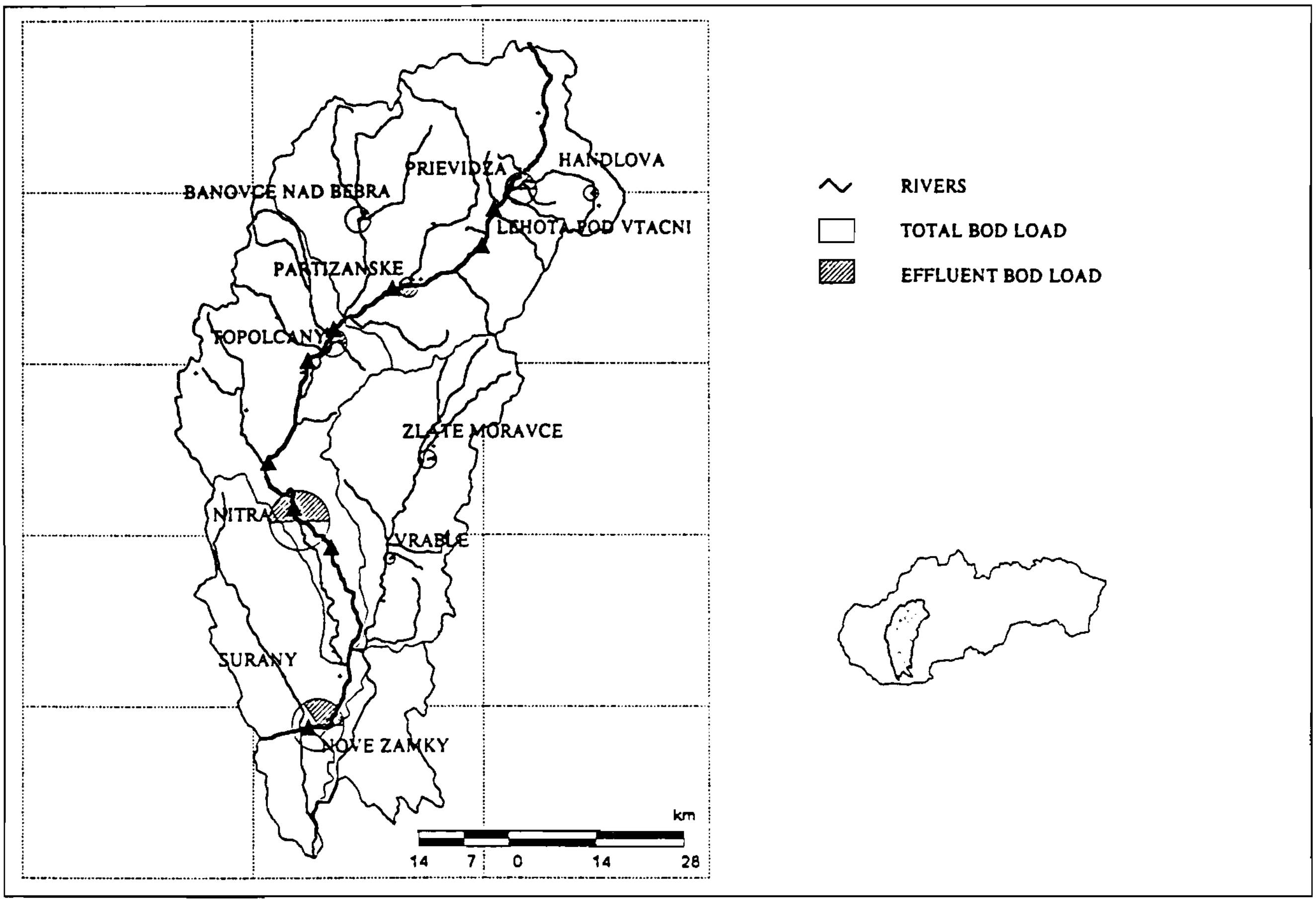

Figure 3.7 Municipal wastewater treatment in the Nitra River basin.

3.5 Water Quality

The history of the water quality monitoring network goes back to the mid-sixties (Chapter 7). There are data available since 1976 which thoroughly cover major dischargers for 26 sampling sites (see Figures 3.2 and 3.4). These data systematically do not show a trend. The quality is permanently poor, indicating that socio-economic changes which negatively affected the environment took place prior to launching the monitoring program.

Water quality is characterized by low (and sometimes depleted) oxygen levels, high BOD-5 values (not rarely close to or above 30 mg/l--see Figure 3.9 showing maximum BOD-5 values taken from the database--characterizing biologically treated wastewater effluents), high ammonia, phosphorus, dissolved and suspended solids, and arsenic concentrations (see Chapter 7 for details). According to the existing evaluation system, the quality of the Nitra River is characterized by the two poorest classes. This is a clear consequence of high waste residuals due to old-fashioned production technologies, improperly developed water infrastructures, and insufficient industrial and municipal treatment levels as discussed before (see Chapter 6 for further details).

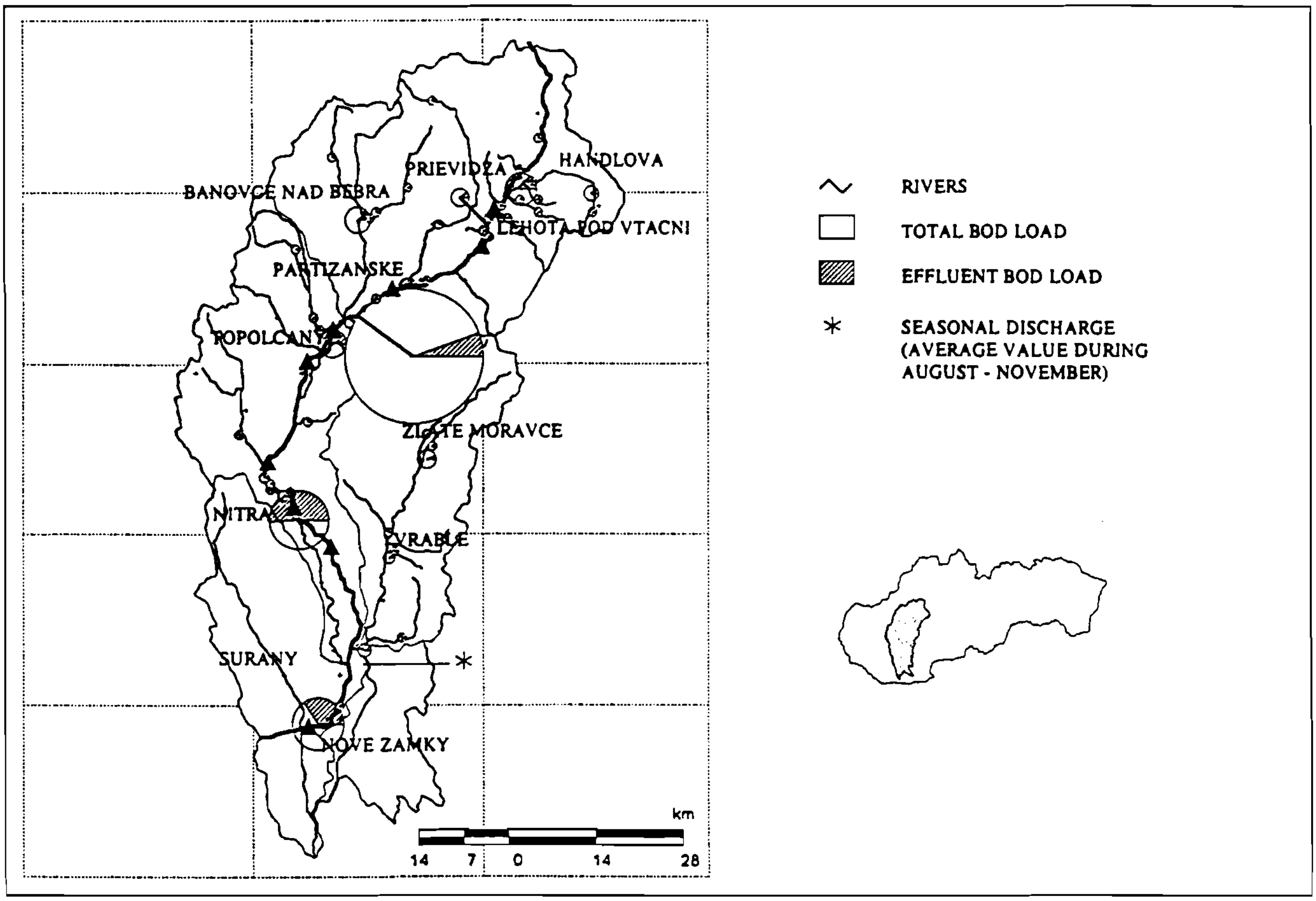

Figure 3.8 Industrial wastewater treatment in the Nitra River basin

Groundwater quality is also rather problematic. For instance, in the Novaky/Nové Zámky region, 98% of the samples analyzed did not meet drinking water standards. Iron, manganese, and ammonium were the components where admissable limits were surpassed most frequently (clearly indicating the absence of oxygen in the course of the filtration process).

3.6 Data Availability

Overall, data availability is acceptable, particularly for climatology, meteorology, hydrology, and river morphology. However, with respect to pollution issues, the situation is worse. To name some, significant improvements are needed in flow measurements at wastewater treatment plants (those with both primary and secondary effluents form crucial problems) and in monitoring emissions and their different components. Only a few emissions measurements per year are available, and, for instance, nutrient data are frequently missing. Water quality of rivers has been monitored systematically since 1976, but the frequency of measurements is far from satisfactory at only one measurement per month. Further difficulty is caused by the fact that the above data (and several others not mentioned here which are needed for a comprehensive management oriented analysis) are not easily available. They are not accessible in an integrated data base and their joint evaluation--after establishing pre-conditions--can lead to a number of inconsistencies, discrepancies, and surprises.

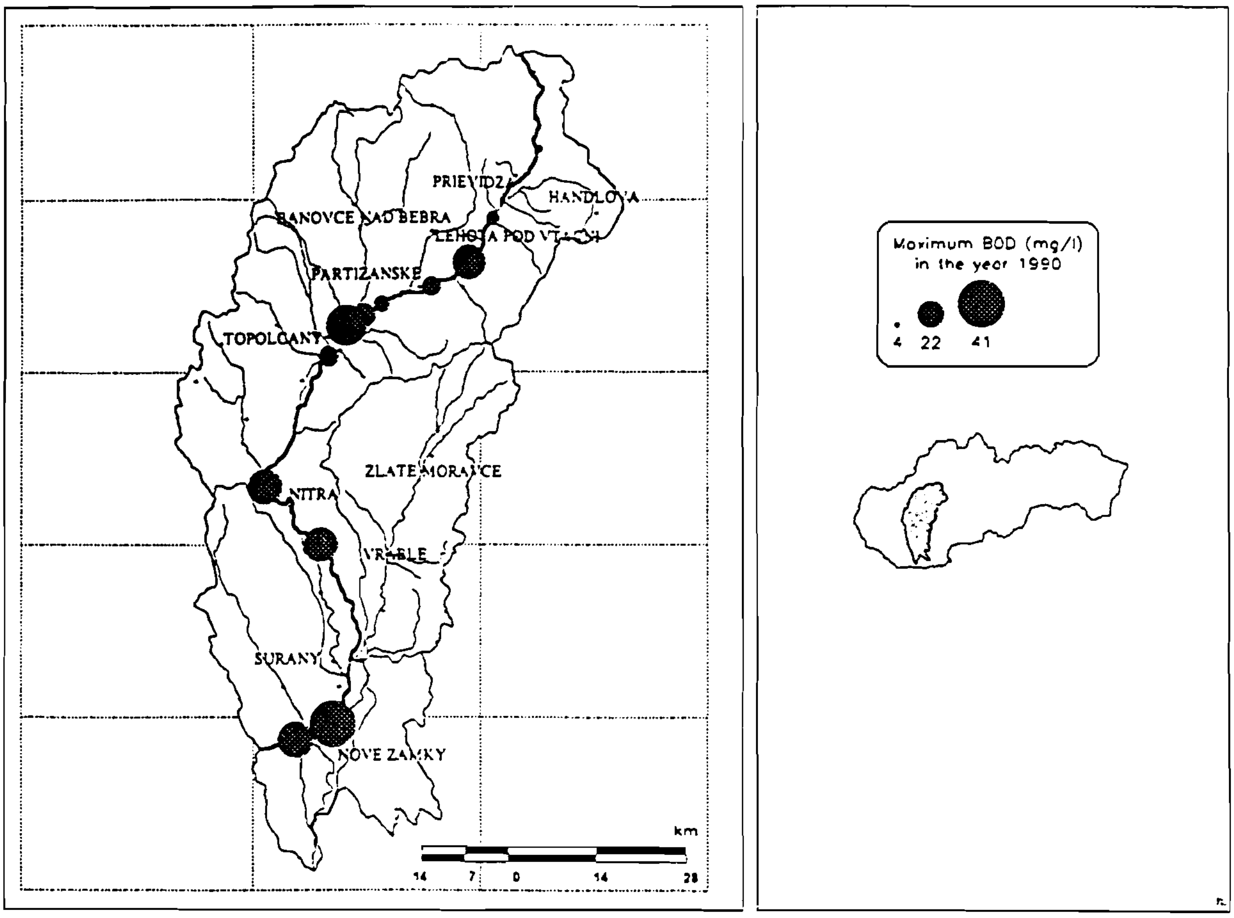

Figure 3.9 Maximum BOD-5 values in the Nitra River basin in 1990.

3.7 Legislation

Both past and new (valid since November 1993) legislation use a mix of effluent and ambient water quality standards. The decree (No. 242/1993) distinguishes two periods--before and after 2005; effluent standards are somewhat tighter on the longer time horizon. The requirements for municipalities resembles those of the European Community, although limit values are somewhat less stringent and there is a stronger focus on the protection of drinking water resources. For instance for the >100000 P.E. range (1 P.E.= 60 mg BOD-5/d) effluent standards for BOD-5, COD_{cr}, NH4-N and TP are 30 mg/l, 110 mg/l, 10 mg/l and 3 mg/l, respectively. The corresponding tightened values subsequent to the beginning of 2005 are 20 mg/l, 90 mg/l, 5 mg/l and 1.5 mg/l. With decreasing P.E. value, standards become more relaxed. For instance, for the smallest cluster (< 50 P.E.), the BOD-5 limit is 80 mg/l, and other component standards are not specified. Actually on the short term TP is not given for the <25000 P.E. domain, while NH4-N is excluded for the < 5000 P.E. range. Industrial emission standards depend on the type of production, and limit values cover a broad range (e.g. 100-800 mg/l for COD-Cr, a range which does not seem to be fully justified (many possible exceptions are also discussed by the decree).

Ambient criteria distinguish two receiving waters, those which are used for drinking water purposes and others. Ranges of standards (to be met under the flow of Q_{355}) are for DO, BOD-5, COD-Cr, NO3-N, organic N, and TP, 5 to 6 mg/l, 4 to 8 mg/l, 25 to 35 mg/l, 3.4 to 7 mg/l, 1 to 2.5 mg/l, and finally 0.05 to 0.4 mg/l, respectively. There is no standard given for the amount of NH3 in drinking water; the standard for other water is 0.5 mg/l. This set of standards is rather strict (see Chapter 7), and meeting them will require significant investments (and may not be realistic in the short term). In principle, both ambient and effluent standards should be satisfied; however, it is not yet clear how well the system will work during the forthcoming transition period (until 2005, the projected date for reaching a consolidated situation).

Dischargers are required to have permits, which are valid for eight years. The decree specifies analysis of emission monitoring practices and accreditation of laboratories for enforcement purposes.

Unfortunately, enforcement was practically lacking in the past. Emission control measurements scarcely were performed, and the amount of wastewater charges collected was (and is) very low. The domestic sewage charge is 3 Sk/m^3 (current exchange rate is 1 USD ~ 30 Sk) which should be paid together with the water tariff (4 Sk/m^3 at present). Values are about double and somewhat negotiable for industry and private enterprises. Municipalities use the income stemming from charges and tariffs to cover operation costs, and a portion of charges may go to the Environmental Fund. Additionally, charges/fines may be paid to the Fund, depending on the final discharge quality (but at present the levels are set so low that they do not act as incentives to improving treatment).

The level and extent of the above economic instruments are not yet properly set (and several other tools are not used at all). It is estimated that the Environmental Fund available in a year is about an order of magnitude smaller than the real needs. During the past three years the Fund ranged between 1 and 1.5 billion Sk, corresponding to about 30-50 million USD, and 50-70% of it (a very high value) came from the state budget. Next year only 40% will come from the state budget, and this percentage should be brought to 0% as soon as possible. One use of the Environmental Fund is partial upgrading and constructing of wastewater treatment plants (funds are used as subsidies up to 30% of the total investment).

3.8 Institutional Issues

As indicated earlier, the institutional setting is a complex one that undergoes changes together with economic, political, and social alteration. For instance, the operation of the water quality monitoring program alone involves the Ministry of Environment, the Slovak Hydrometeorological Institute, River Basin Authorities, and the Water Research Institute.

Decision structures are rather sophisticated. The key actors incorporate the two ministries, the environmental inspectorate and its regional offices, the river basin authorities, regional (and local) water and sewer works (also going through a transition -- for example, at present investments on the municipality level can not be made without the involvement of regional offices), local and district governments, and other offices and agencies at the financing, planning and implementation stages. The smooth and efficient operation of these institutions is necessary to improve environmental and water quality management. This realization of such an operation will be a major challenge of the coming years.

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4. OBJECTIVES OF THE STUDY

The major problem with the Nitra River is its high contamination by municipal and industrial pollutants, which is associated with a medium water quantity vulnerability (particularly during summer months). Considering likely future changes in water use patterns (e.g., a decrease in domestic and industrial consumption, an increasing ratio of re-use and re-cycling, the introduction of advanced production technologies, etc.), water management--if handled carefully and pollution is controlled--will be able to satisfy the needs of the coming decades. Thus, pollution control forms the priority issue for water and environmental management and for the present study.

As is apparent from previous chapters, a significant amount of information seems to be available for the Nitra watershed which characterize its hydrology, morphology, hydraulics, water supply and use, emissions, water quality, and so forth. However, this information has never before been applied in an integrated fashion for policy development. Emissions and ambient water quality impacts cannot be related to each other. Even a detailed pollution inventory illustrating major causes of water quality deterioration is missing. There was no such data collection performed in the past which would have addressed this issue. Finally, several questions should be raised with respect to the usefulness of the existing, basin-wide water quality monitoring network. For instance, is the sampling frequency satisfactory for the purpose of classification or trend detection? Is the number of monitoring sites sufficient, and their distribution well developed? Should not wastewater discharges be observed more frequently? And so forth.

On the basis of the above findings, objectives of the present research can be identified as follows:

(1) To develop a (short-term) regional water pollution control policy with a special focus on cost-effectiveness and minimizing costs (this view stems from the existing economic situation).

(2) To prepare a basin-wide emission inventory (which is a pre-requisite of any strategy development).

(3) To establish relationships between the emissions at various locations and longitudinally (and temporally) variable ambient water quality (by using water quality models and experiments) in order to work out ambient criteria based least-cost strategies.

(4) To develop a decision support system for the actual establishment of the above water quality control policies.

(5) To revise the existing water quality monitoring network.

As can be seen, Objectives 2-4 directly follow from the first, main objective of the study, while the last one is somewhat different in nature. However, it is not at all unrelated to the policy issues considered here.

5. THE APPROACH

As noted earlier, the development of any effluent standard based strategy is a simple task where all the elements and features of the strategy are actually pre-defined by the standards. However, least-cost strategies can be developed only on the basis of selecting ambient water quality criteria reflecting actual water uses. For such cases, the application of water quality (simulation) models which link effluent and ambient water quality is a necessity. At the same time, there are a large number of combinations of possible non-uniform load reductions at various sites each with differing receiving water quality impacts and cost implications. Therefore, the application of an optimization decision model can hardly be disregarded (small catchments with a small number of emissions certainly form exceptions).

The task thus obtained is rather complex. It requires the integration of a river basin simulation model (together with inputs such as the emissions which are subject to the required, unknown level of treatment) into an optimization framework, with possible non-linearities and stochastic features (depending, among others, on the type of the water quality constituent to be considered). An approach should be selected which is not watershed specific (inasmuch as it is possible) and decomposes the problem into smaller, manageable elements.

The scheme developed here is illustrated in Figure 5.1 (which also indicates the chapter in which the particular issue is discussed in detail). The starting point is the preparation of a pollution inventory. For the Nitra river basin, this necessitates a survey of point sources as the role of diffuse loads is negligible. The present state of water quality depends on the emissions, hydrologic and meteorologic factors, as well as physical, chemical, and biological processes of the receiving waters. An overall characterization can be obtained from the analysis of the routine data from the water quality monitoring network. This description will be rather simplified and static due to the scarcity of water quality data available (i.e. one observation, monthly) and the lack of synchronized emission measurements. As the analysis in Chapter 7 will discuss, the present data collection may be sufficient for the purpose of water quality classification. However, it is not sufficient for establishing load response relationships or describing the processes and temporal quality changes induced.

In this manner, the development, calibration, and validation of water quality models should rely upon specifically designed, in situ experiments (Figure 5.1). Their style basically depends on the type of model (or models) to be used for policy purposes. Major questions are whether a static or dynamic model version should be selected and how uncertainties should be treated.

Rivers with dominating point source pollution form relatively simple problems from the point of view of the above issues. Point sources generally do not exhibit significant temporal variability on the time and space scale of interest. Thus, typical low flow conditions are used for "safe" design under the assumption of steady state conditions. For instance, a critical low flow Q355 or the minimum 10-day consecutive flow of a 10-year period can be selected (to which other parameters of the "design scenario" belong). In other words, the basis is an event occurring rather infrequently, and thus, water quality will be better most of the time than the "goal" due to increased dilution. The situation is much more complex if, for example, non-point sources are also important or the system's reservoirs result in differing "critical scenarios" for different water quality components which are interdependent (it suffices to refer to a BOD-DO-N-P-algae problem).

Nevertheless, major features of the Nitra River problem allow the application of the traditional approach based on the low flow design condition. Additionally, the policy issue will be considered as a purely deterministic one, and the role of uncertainties will be analyzed after the strategy development (see Figure 5.1).

Now the conclusion, from the point of view of the in situ water quality experiment, is that it should be performed under low flow conditions which roughly correspond to the design scenario. It should cover both emissions and ambient water quality (and other factors) so that the calibration and validation of water quality model(s) are a meaningful task. These can be performed in a deterministic and/or stochastic fashion. The latter one, as will be shown in Chapters 9 and 13, will allow the performance of an *a posteriori* Monte Carlo simulation to estimate the degree of violation of the set standards or to do a so called "regret analysis" when the consequences of realizing parameters other than those assumed for the design are evaluated (see Figure 5.1).

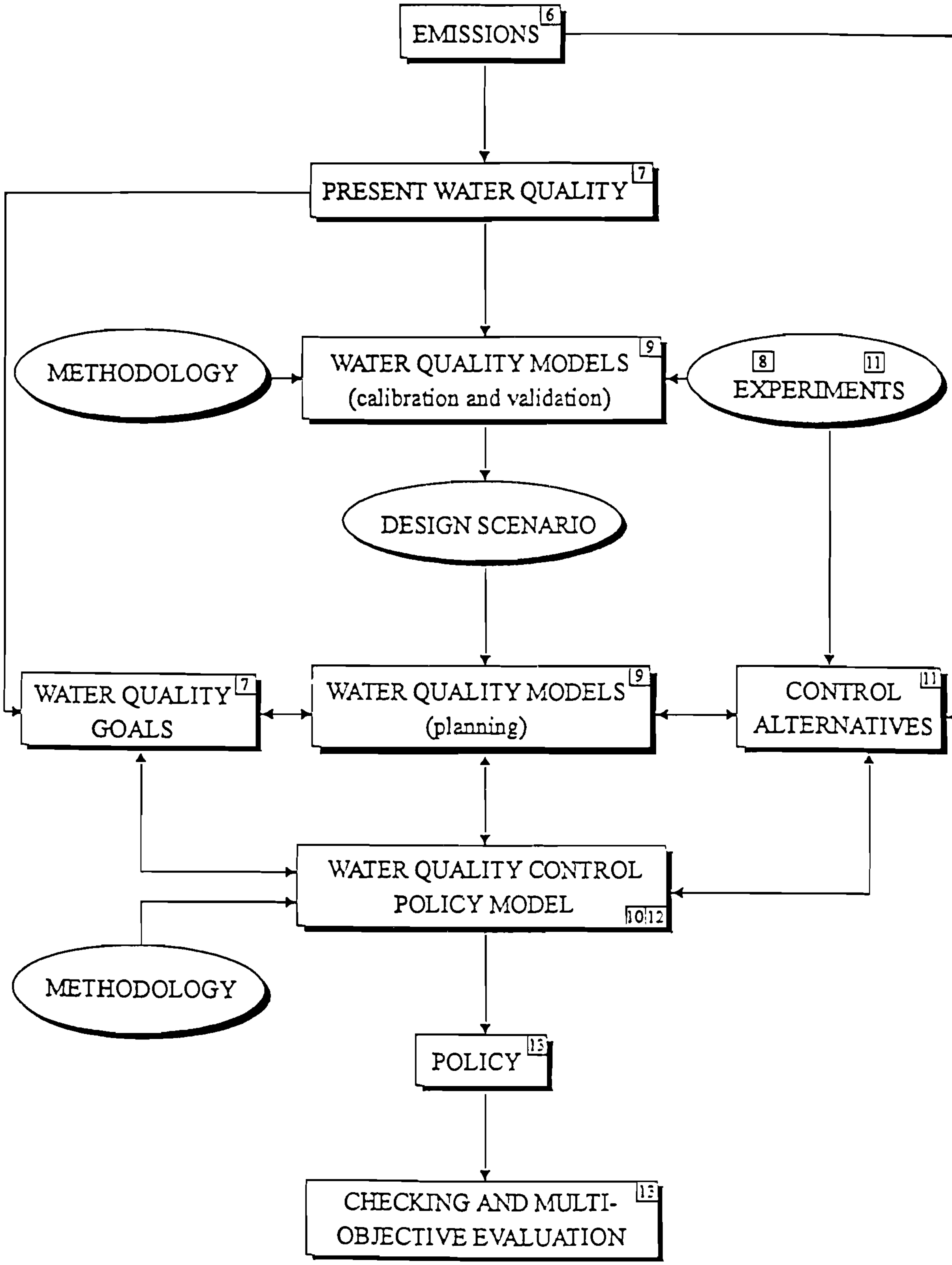
Stemming from the nature of the problem of the Nitra River, the selected quality models consider components of the oxygen and nutrient households. The complexity of the models ranges from the simple, linear Streeter-Phelps (two state variables and two parameters) to the rather sophisticated, non-linear QUAL2E (about ten state variables and more than 30 parameters for each river stretch distinguished in the model). Among these, only the simpler ones were incorporated into a formal parameter estimation and uncertainty analysis framework (Chapter 9). The same models served also for the policy development.

The control policy focused on municipalities which, as noted before, contribute about 70% of the total emission of "traditional" pollutants (e.g. BOD, TN, and TP) in the watershed. Industrial discharges were not directly included into the decision model as no details were available for their control alternatives. Their role was analyzed in a sensitivity fashion. It is also noted that industrial control requires a few, well-defined local actions (Chapter 7).

The water quality goals, the planning-type water quality model, and emission control alternatives enter the policy model on the same level (see Figure 5.1.). Quality goals are formulated on the basis of Slovakian legislation, water uses, and international experiences. On this level, the water quality model (or models) is essentially the same as in the calibration phase. The differences are twofold. First, the parameter values and their statistical distributions are kept from the calibration/validation stage, and second, the flow and other related parameters are taken from the design scenario(s). Control alternatives (in a broader sense) and municipal wastewater treatment options (in a narrower sense) for the particular problem of the Nitra River are represented by a summary table for each emission. Individual lines represent discrete treatment configurations characterized by effluent quality (e.g. BOD, SS, DO, NH4-N, NO3-Nand TP), costs (both investment and operation, maintenance, and repair), economic life of the project, and so forth (Chapter 11). The range of treatment levels may move from "no treatment" to the application of the "best available technology".

The development of the policy model may require significant methodology developments, similar to that of the simulation models. In the frame of the present research, the method of dynamic programming was employed. Its attractiveness comes from the fact that it utilizes major features of river basin pollution problems (i.e. that control actions have no upstream impacts) and that it can be utilized for both linear and non-linear water quality models (i.e. it is a rather generic solution procedure).

Figure 5.1 Approach to the study



The result of the application of the control model is a policy corresponding to its goals and the problem formulation (e.g. meeting set water quality goals under minimization of the total annual cost or the investment cost if the starting budget is really limiting), together with details of the longitudinally variable water quality, treatment configuration and costs. As mentioned before and indicated by Figure 5.1, the entire approach is completed by a checking stage and multi-objective evaluation of the results.

6. MUNICIPAL EMISSIONS

6.1 The present situation of wastewater treatment

A summary for eleven municipalities and plants (see Figure 3.6) is given in Table 6.1. The summary characterizes both the supply and the treatment side for 1990. Approximate data on the origin of the raw wastewater (domestic, industrial and infiltrated) are presented, but admittedly the figures are rather uncertain for several reasons. Age and BOD-5 performance (including information on flows of differing treatment levels) are also included. From the table and other related background materials the subsequent conclusions can be drawn.

(1) The overload can exceed 100% for several wastewater treatment plants (WWTP). The industrial contribution is particularly high for Nové Zámky, Partizanske, and Moravce.

(2) Several WWTPs (including Nitra, Nové Zámky, and Topolcany) operate such that the biological unit is utilized roughly up to design capacity (depending on operational practices which may change from year to year); the rest receives only primary treatment (see the scheme of the Nové Zámky WWTP in Appendix 6.1). The final effluent is obtained by blending the two waters of differing quality (thus the entire process may not have a removal efficiency in terms of BOD-5, COD, or TSS that is higher than 50-70%). The evaluation of such plants is rough because flow measurements for the two treatment lines are generally missing.

(3) The composition of the raw wastewater is within the usual ranges for BOD-5, COD, TSS, and NH4-N, except for municipalities where infiltration is extremely high. In an average it is less than 20%, but for Partizanske, Surany and Handlovka it is close to 30%, and for Lehota it is above 50%. Phosphorus values (e.g. those obtained in the course of our experiments, see Chapter 8) are lower than anticipated, although reasons for these low values are not quite clear (detergent usage and/or analytical difficulties can be possible explanations).

(4) Sludge treatment and disposal are not solved adequately. They are generally based on anaerobic stabilization, dewatering by sludge fields, lagoons, and mechanical facilities. Disposal on landfills and in agriculture is a frequent solution. The latter is increasingly prohibited by tightened land disposal standards. Cadmium causes particular difficulties due to lack of industrial pre-treatment.

(5) The construction of several WWTPs took 5-8 years. To reach full operation often a couple of additional years were needed. Thus, frequently the actual design was done ten years before the operation started (see Table 6.1). It should also be noted that most of the older treatment plants were upgraded during the past 10-15 years to increase the capacity and to partially solve existing, serious operational problems. An example is Nové Zámky where a primary sedimentation tank and two final clarifiers were added recently (see Appendix 6.1).

(6) The design of the plants excluded nitrification. The surface overflow rates were selected conservatively to be around 1 m/h for both the primary and final clarifiers, which turned out to be a wise decision considering the present level of hydraulic overloads.

(7) WWTPs exhibit not only seasonal changes in the flow and loads but also significant diurnal fluctuations, depending on water consumption habits, the impact of industry, and travel time in the collection network. As an illustration, the diurnal variation in the BOD-5 and COD parameters are shown for 17-18 March 1993 in Figure 6.1. It is striking to note that BOD-5 moves in the 100-1000 mg/l range, with a peak around 2-3 p.m. A likely reason is that dairies and meat factories (with slaughter houses) have work shifts between 7 a.m. and 4 p.m. Only rough pre-treatments exist before discharging wastewaters to the sewerage. Peak BOD-5 values can reach 7 000 mg/l, and the travel time to the WWTP is about 2-3 hours.

6.2 The impact of economic transition

Under the present economic and social changes it is of primary interest to determine how the loads of existing plants are changing, to what extent they should be upgraded, and what the longer term capacities of new and/or re-shaped facilities should be. An answer to the above questions requires an analysis of ongoing and likely future changes of water consumption and factors affecting the generation of wastewaters.

Drastic changes in industrial discharges (independent of municipalities) could be seen in the Nitra catchment; often there was a 50% reduction (e.g. at the Tannery Bosany with a new wastewater treatment plant) or full shut down. In contrast, municipal water consumption and wastewater production have shown surprisingly modest changes. For instance, the flow at Nové Zámky diminished by about 15%, but at other localities the difference observed is not significant (under the uncertainty of available data).

Given an analysis of the data of Table 6.1, it is unlikely that the impact of infiltration (and stormwater) would decrease quickly, as the investment requirements of sewer rehabilitation (and stormwater management) are rather high. A further increase in the water price will positively influence residential water consumption, but since its level is not exceptionally high, it is not likely that the future reduction will be greater than 10-20%. Thus, the key element of the system is industry. Economic transformation and the application of effective economic instruments, as well as the introduction of clean technologies and pre-treatment, could lead even to a 50% drop in water consumption if we considered the present low level of re-circulation. Under such conditions, for example, the wastewater flow of the Nové Zámky treatment plant would drop below 20000 m^3/d, which automatically would lead to significant improvements in the plant's performance. A change in the rate of flow and in production technologies would obviously influence also the composition of wastewaters.

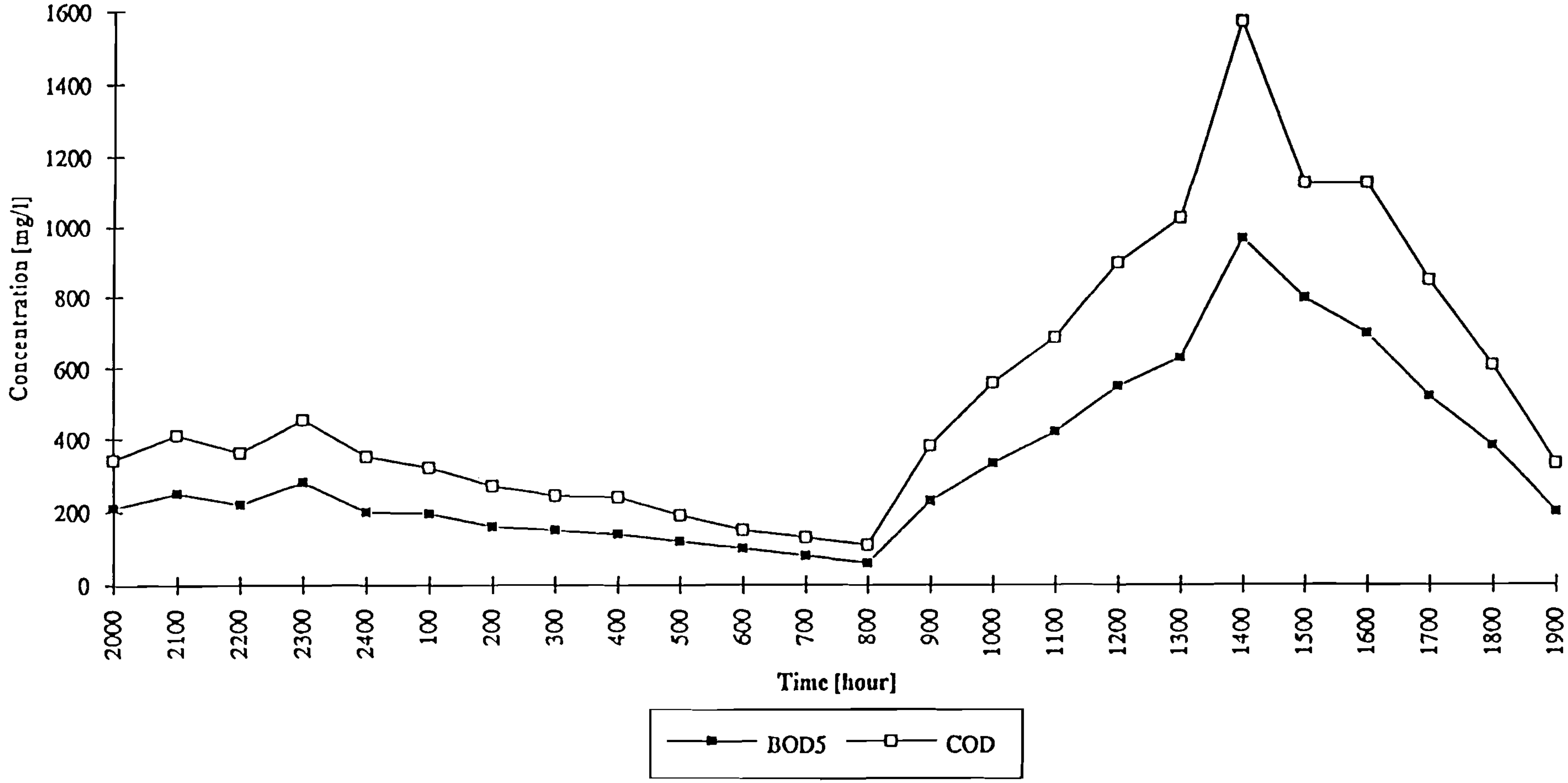
Thus our conclusion is that care should be taken when deciding on future investments. Design conditions can hardly be estimated at present, and there is a danger in launching projects which will not be fully utilized later (i.e. money is wasted). For this reason the recommendation here is to carefully evaluate design conditions of planned control actions and to do flexible development plans. That is, the recommendation is to make investments which are unavoidably needed in the short term with safely defined goals and parameters and expand them after proper monitoring if the needs of the subsequent period can be estimated reliably. This recommendation leads to the idea of multi-stage project development which is discussed in Chapter 11.

Table 6.1 Major parameters of municipalities and their wastewater treatment plants (1990)

Name	Nitra	Nové Zámky	Topolcany	Prievidza	Partizanske	Banovce	Handlova	Zl. Moravce	Surany	Vrable	Lehota
Population	91300	43000	37700	60000	27000	20100	18300	17800	11900	9200	3500
Water supply ratio [%]	99	96	89	100	82	79	100	99	74	97	84
Sewerage ratio [%]	95	91	82	95	83	84	96	57	42	52	43
Wastewater flow [m³/d]	32600	29200	12900	25000	24000	7800	6800	5600	3400	2000	1000
Domestic portion [%]	49	35	48	62	16	51	63	40	30	53	40
Industrial contribution [%]	40	54	37	14	50	48	4	43	41	n.a.	n.a.
Infiltration and Stormwater [%]	11	11	15	24	34	1	33	17	29	n.a.	60
Year of permanent operation	1973	1969	1975	1971[1]	1994[2]	1977					
Design capacity [m³/d]	14200	13000	5100	5300	0	3900	3900	5500	3000	3400	2500
Biologically treated [m³/d]	20000	18000	5900	6000	0	6000	6800	5600	3400	2000	1000
Primarily treated [m³/d]	12600	11200	7000	19000	0	1800	0	0	0	0	0
Untreated [m³/d]	0	0	0	0	0	0	0	0	0	0	0
Influent BOD5 [mg/l]	250	240	280	171	91	350	150	270	170	230	121
BOD5 removal rate [mg/l]	49	65	70	53	0	85	87	91	88	91	91

[1] Test operation of a new WWTP started in 1992.
[2] Construction of a new WWTP will be completed soon.

Figure 6.1 Diurnal variation of influent BOD5 and COD concentrations: the WWTP of Nitra (17-18 March 1993)



7. WATER QUALITY IN THE NITRA RIVER BASIN AND ITS REGULAR MONITORING

7.1 Water Quality Monitoring

7.1.1 Types and objectives of water quality assessment

As one of the most important life sustaining resources, water deserves close attention with respect to its quality and suitability both for man and for natural life. According to the UNESCO/WHO/UNEP guide (Water Quality Assessment, 1992), water quality observation programs could be subdivided to monitoring, survey, and surveillance. The surveillance usually is connected with specific operational and management issues. Survey is an intensive study limited in time and serving specific purpose. Monitoring is a long-term observation program reporting status of environment and trends in its development. In practice, the most common case of water quality management is multi-purpose water quality monitoring, which covers various water uses such as drinking water supply, industrial manufacturing, recreation, aquatic life, etc. This necessitates a large set of monitored parameters. Other common water quality assessment practices would be basic surveys, serving the purpose to identify major water quality related problems, and operational surveillance (for instance, drinking water supply control).

A special case is the water quality surveying for purposes of understanding the processes affecting the water quality and providing the basis for a policy decision. Usually such assessment is aided with mathematical modeling of water quality. Two such studies were conducted in the Nitra River basin twice, August 1992 and in June 1993, in the course of the reported research project (Chapter 8).

7.1.2 The institutional structure of water quality monitoring in Slovakia and in the Nitra River basin

The Ministry of Forestry and Water Management of the Slovak Republic charged the Slovak Hydrometeorological Institute (abbreviated SHMU in Slovak language) with the task of systematic monitoring of the quality of surface and subsurface waters in Slovakia. The regular monitoring of water quality of the Slovak rivers started in 1963; monitoring of the quality of groundwater started later in 1982. For both flow and quality monitoring, SHMU is in charge of overall supervision and management. In addition, it provides storage of the results in data bases, data processing and evaluation, as well as regular publication them in the form of annual booklets (yearbooks) for reference by any concerned organization.

The surface water sampling and laboratory analyses are carried out by individual watershed authorities. For the Vah and the Nitra river this task is performed by the Vah Authority in Piestany. The SHMU is also responsible for the flow measurements.

The surface water quality evaluation is published by SHMU in annual surveys called "The water quality in the streams of Slovakia" according to the acting Czechoslovak classification scheme (from the 1st of July 1990 the valid classification is CSN 75 7221). The evaluation serves mainly for classification purposes and does not have the aim to accomplish standard compliance. For assessment of the groundwater quality, the SHMU applies the drinking water quality standard CSN 75 7111, and this evaluation is also published in annual surveys.

The water quality observation program in the Nitra River basin is a multipurpose monitoring program which reports both the state and trends of the major water quality parameters. As is often the case with multi-purpose programs (Water Quality Assessment, 1992), it exhibits a compromise based upon limited financial resources, the size of the set of parameters, the frequency of sampling, and distribution of the network over a number of important locations. As the analysis will show later, the monitoring system is in a need of reshaping. The most important issues are detection of non-traditional (organics, heavy metals) and non-point source pollution, increase of sampling frequency at important locations (e.g. river mouth), and monitoring the condition of aquatic life. The detailed discussion of the improvement of the monitoring system can be found in Appendix 7.3, and the summary of the recommendations is in Section 7.4.

7.1.3 The monitoring network in the Nitra River basin and its relation to the character and occurrence of the pollution.

The monitoring network in the Vah watershed consists of 63 sampling stations. Of these, 26 sampling stations are located in the Nitra subwatershed (according to the annual survey for season 1989-1990, see Appendix 7.2). Figure 7.1 provides the outlook of the monitoring network in the Nitra River basin with respect to the location of municipal and industrial pollution; the sites where flow measurement are made are also shown.

Figure 7.1 shows that the monitoring points usually have been deployed in pairs, one immediately upstream of municipality and the other downstream, with the purpose to study the effects of the emissions. There are cases when the complete mixing of the effluent with the river water does not occur upstream of the sampling site (see Appendix 7.2). The allocation scheme used in the Nitra River basin would be adequate for checking the overall amount of the effluent material being discharged between the two points. However, in order to receive policy-oriented information, it is necessary to determine the origin of the pollution down to the particular discharge. This is possible only if the emission data are provided by the monitoring scheme, which is not the case at present, or by a specially organized measurement campaign (Chapter 8).

The sampling procedure should take into account high diurnal variability of the pollutant loads (Chapter 6). In order to get an idea about the mean daily average concentration and emission (cf. Figure 6.1), it would be desirable to take composite samples.

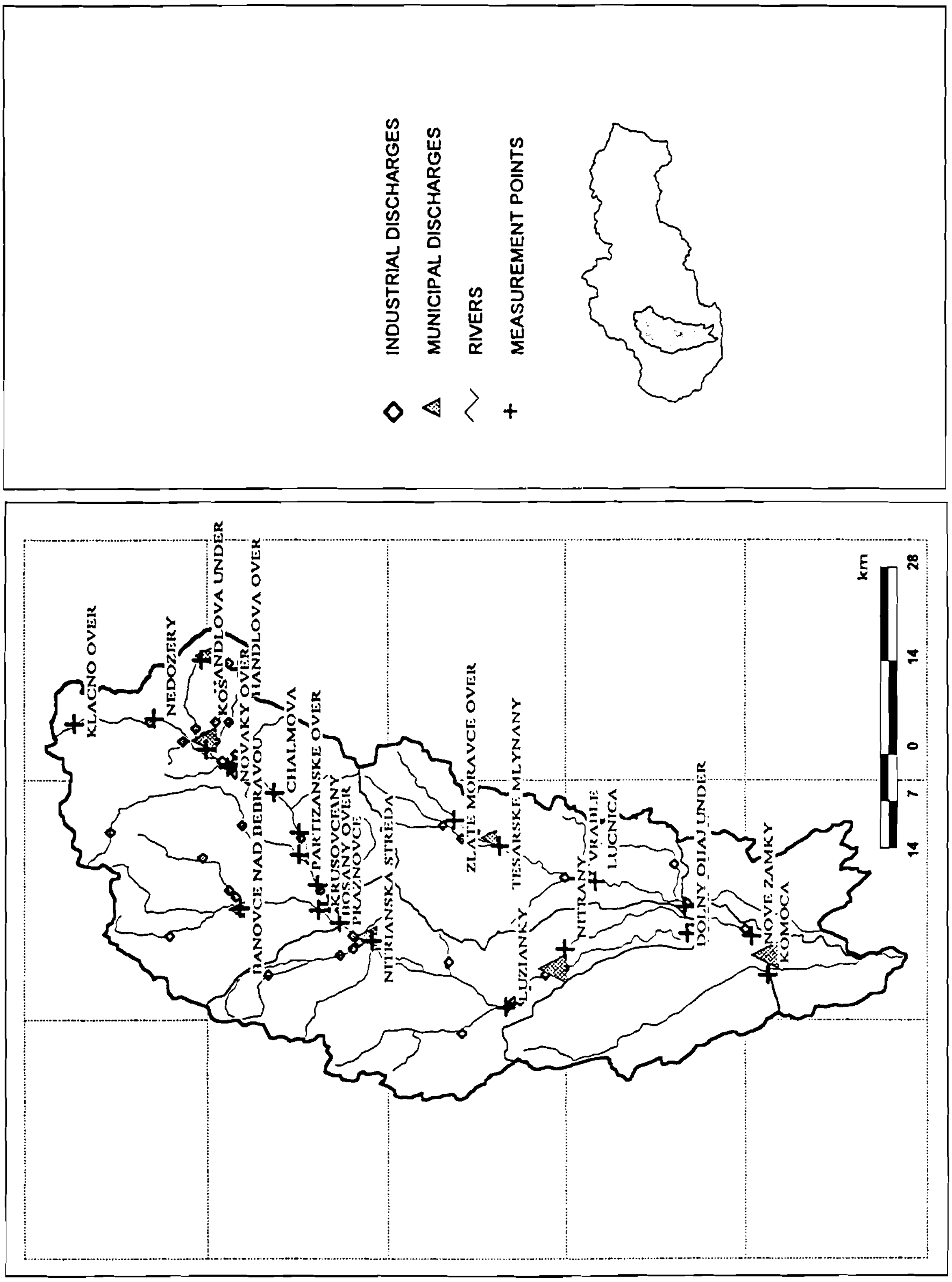


Figure 7.1. The surface water quality monitoring network in the Nitra River basin and the point-source pollution.

7.2 Ambient Water Quality Standards and Classifications

7.2.1 Setting of water quality criteria and standardization with respect to management

Historically, the first concern about the surface water quality came from the necessity to secure a safe supply of drinking water for large European cities suffering from outbreaks of waterborne diseases (Tebbutt, 1992). The uncontrolled disposal of organic waste in nearby watercourses lead to proliferation of numerous bacteria, some of them pathogenic. Later, the role of dissolved oxygen in the process of stabilization of organic material was understood. Incidentally, it was learned that dissolved oxygen is essential not only for aerobic bacteria but also for all higher forms of aquatic life, and methodologies for dissolved oxygen household studies were worked out first by Streeter and Phelps (1925). Biological oxygen demand (with all ramifications), dissolved oxygen, and bacterial cell count thus became the first key parameters in the assessment of anthropogenic pollution of rivers, and are often called "traditional" pollutants. Most early control actions in the field of water pollution control were associated with these parameters.

Later the importance of other elements for the support of aquatic life was understood. Among them, the elements needed in large amounts to build the organic cell matter (nitrogen and phosphorus) were called nutrients. They are normally present in all waterbodies, whether they are affected by human activity or not. Their excessive amount, however, may cause a problem proliferation of aquatic plants or so-called eutrophication. Therefore, special actions may be needed to control the amount of nutrients in waterbodies susceptible to excessive algae growth (EPA Technical Guidance, Book 2 Chapter 2, 1983). For example, excessive nitrogen (usually in the form of nitrates) may appear in the river water due to the application of mineral fertilizers for agriculture and could make the water unsuitable for domestic supply. Since the pathways of the nutrients are affected by aquatic life, a study of the aquatic ecosystem is necessary to provide full insight into dynamics (Somlyódy and van Straten, 1986). The nutrients (especially nitrate) often originate from soil-applied substances and appear in the water in the form of non-point source pollution, i.e. they are associated with the watershed area rather than with a particular location of emission, which has some implications on their control.

As industrialization has occurred, the number and amount of pollutants discharged to surface waters greatly increased. In some cases these "non-traditional" (micro) pollutants were causing more serious problems than the "traditional" ones. Some of the most important non-traditional pollutants are heavy metals, petroleum products, pesticides, and surfactants. Due to the recent appearance of these pollutants, their monitoring and control is much less developed, especially in countries of CEE. It is worth notify here that industrial contamination (not primarily with micropollutants) is also commonly responsible for altering the basic physico-chemical properties of river water such as pH, redox potential, and salinity, which are used routinely for classification of surface waterbodies. Thus the basic physico-chemical parameters of water sometimes can serve as indicators of industrial water pollution in the region.

From the point of view of water quality management, the control of industrial point-source pollution affecting the basic chemical properties of water appears rather straightforward. It is usually a single-location problem and the offending enterprise is easily identified. The emissions should be unconditionally reduced to the levels where they do not hinder the usage of the water for any prospective need. On the other hand, the control of municipal pollution could require certain policy study and planning of investments, because there are many municipalities in the watershed of the river and they affect water quality not in equal measure. Such a policy study is the focal point of this report (Chapter 13). Handling the non-traditional pollutants such as heavy metals and toxic organic materials require from the research side more elaborate experimental and modeling effort and from the control side evaluating the industrial pre-treatment schemes for individual enterprises. This can form a subject of further studies.

The control of water pollution is implemented usually via environmental legislation. Two approaches can be envisaged based on effluent and ambient standards, respectively (or their combination). The former constrain the quality of the effluent water, and the latter direct the quality at the place of its usage. In Slovakia a mixed is in force (see Section 3.7), where the effluent standards seem to be binding. However, the practical usage of the two systems and the scheduling is not clearly defined at the moment.

Water quality standards generally are established with respect to the projected water uses. This structure, for instance, is adopted in the USA (Quality Criteria for Water, 1976) and in the European Community (EC Environmental Directives, 1978). According to the UNESCO/WHO/UNEP survey, the uses of the surfaces water with regard to the quality issued could be divided to the following categories (Water Quality Assessment, 1992):

- drinking water supply
- aquatic life and fisheries
- recreation
- irrigation
- industrial uses
- power and cooling
- transport

The usage of surface water as the source for the domestic purposes can be limited by acute and chronic toxicity caused by pollution. Thus the human health protection issues and/or the reasonability of costs for water conditioning define the standards of the first category. Of those, we will deal exclusively with the standards which prescribe water quality prior to treatment, i.e. quality of water intended for abstraction.

Irrigation intake imposes two notions, the first being prospective harm to the growth and development of the agricultural plants (including the soil deterioration in the presence of certain inorganic ions) and the second being the safety of the yield consumption by humans.

Many of the water quality constituents should be considered both for livestock watering and for human drinking water supply. For uses of bathing and swimming, the most important consideration is the absence of the pathogens in the water. The industrial utilization of water varies with respect to the technological process, but usually does not impose severe quality constraints.

The implications of the pollution to the aquatic life is probably the most difficult to define. The reason lies in the extreme diversity of the species of flora and fauna inhabiting the terrestrial waters and the wide range of its responses to the pollution. The concentration of a pollutant which is lethal for one specie can be virtually harmless for another. Moreover, the aquatic ecosystem is an interwoven network of food chains. Harm to the species which are naturally consumed by the higher elements of the food chain will inevitably concern the latter due to changes in their ration. The consequences of those changes are virtually impossible to predict and classify. Consequently, the standards for aquatic life preservation necessarily should be local and depend upon the proliferation of specific communities in the given waters as well as upon the natural conditions of the aquifer defined by its geography. Usually it is possible to come up with criteria which will ensure the existence and propagation of the essential and significant life in water, as well as protect the life which is dependent on it.

As for the operational definition of water quality parameters, the historically first procedure of concentration-based judgment is still largely in use. In the case of non-traditional pollutants such as heavy metals or organic micoropollutants, the concentration-based assessment is of limited use. First, the measurements of these substances are expensive; second, the number of materials to be controlled is very large and steadily growing; third, the effect of their concentrations on the aquatic life is not well known (see above). However, the recent research in the ecology of aquatic life made it possible to develop a set of integral parameters which directly measure the effect of the pollution on the ecosystem's health. Most of these ecosystem parameters are derived from a variety of biodiversity and ecotoxicology indices. This topic is outside of the scope of this report (see for more detail Water Quality Assessments, 1992). Bioassay tests also measure toxicity of pollution to certain important species of the aquatic life. Bioassay methods for the acute toxicity of polluted water for different groups of aquatic life, including fish, invertebrata, algae, bacteria, protozoa and higher plants are the most developed (US EPA, 1985; OECD, 1987). The sub-lethal or chronic toxicity effects of pollution are much less investigated. Even less is known about the effects of bioaccumulation of micropollutants in the tissue of water organisms, which can be a sensitive indicator of prolonged contamination of the water with persistent and dangerous chemicals. No water quality standards based on the tissue contents of the micropollutants have emerged so far despite of the fact that the issue of bioaccumulation of toxic substances had been acknowledged several years ago (Thomann, 1974).

The concentration-based ambient quality regulation is well suited for controlling pollution in certain watercourses. The impact of pollution on the receiving waterbodies is better expressed by the loads, and in cases when the pollution reaches a lake or inland sea, the loads could be subject to regulation along with the ambient concentrations. Finally, the ecological indicators should be used in cases where concentration control is difficult or of limited use (see above).

7.2.2 The legislation for protection of water environment in Slovakia

The pollution of water environment is regulated via the dedicated legislation. The Water Law 138/1973 and Governmental Decree 30/1975 from 26 March 1975 were valid till November 1993, and since then the Decree 242/1993 is in force (Section 3.6). The regulation covers both the ambient water quality and the emissions. The effluent standards are determined separately for the industrial enterprises of a given branch and for municipalities depending on the population equivalent value. They are compulsory and subject to inspection by authorities (Ministry of Environment). In the new legislation two horizons are discerned for the effluent control, before and after 2005, the latter envisaging tighter restrictions on the admissible load (Section 3.7).

The ambient water quality standards, called "indices of admissible degree of surface water pollution", are determined differently for waters used for domestic supply and for other waterbodies. Standards refer to effluent diluted by Q355 flow of the receiving stream (flow exceeded 355 days a year). Part of the indices are qualitative (color, odor, presence of sludge benches); another part is quantitative (limits of admissible concentrations of polluting substances). Finally, there is mention that the pollution should not cause adverse effects on unspecified water organisms and that coarse fish should be able to live in all waterbodies whereas the waterbodies used for supply should provide proliferation of the salmon fish species. The last part of the requirements (dealing with aquatic life preservation) does not contain any operational definition of the standard and is formulated in very broad terms. The concentration-based part of the standards could, in principle, be enforced, but the procedure defined in the law stipulates only refusal to issue additional emission permits if the water is already polluted beyond the limits. There is no clear way to reach the cleanup in cases when the permits were already issued. The law also makes mention about the possibility of certain exceptions granted for emission in special cases. Recently the Slovak government initiated revision process for earlier granted exceptions in order to reduce their number (at present over 180).

The mandatory levels of pollutants concentrations are not too stringent. The earlier document stipulated, for instance, the BOD-5 limit of 8 mg/l, COD-Mn 20 mg/l, dissolved solids 1000 mg/l, ammonia 3 mg/l, arsenic 0.5 mg/l. These levels would be reached in most cases provided that the effluent standards are enforced literally; they correspond mostly to Class III limits from the Slovak classification system. However, these values are in general higher than corresponding limits for protection of aquatic life established in the USA or in the EC countries (Water Quality Assessment, 1992). The new legislation is more stringent, the limits being closer to Class II.

As mentioned earlier, the practice of water quality management is at present more biased to effluent-based standards. Although the ambient standards are specified, the terms of definition sometimes are broad, and the means of maintaining the receiving waters in proper condition have not been provided (Section 3.7). It is unclear how the two sets of standards will be used for policy purposes till 2005, except that joint incorporation creates an opportunity to formulate least-cost policies for the above transition period. The most common difficulty in the application of ambient water quality standards in management practice is the lack of an obvious relation between the effluent and ambient water quality. This deficiency could be overcome in the course of the specialized study relying on water quality models (Chapter 10).

7.2.3 The classification of surface water quality used in Slovakia

The classification system for the surface water quality used in Slovakia (detailed in the Appendix 7.1) is based on the grouping of the parameters to several sets: parameters of oxygen regime, basic chemical parameters, additional chemical parameters, heavy metals, biological and microbiological parameters, and radioactivity. A similar grouping of the parameters for water quality classification is also adopted in other countries. For instance, the UNESCO/WHO/UNEP document on water quality standards (Water Quality Assessment, 1992) proposes the following divisions: general variables (temperature, color, etc.), nutrients, organic matter, major ions, other inorganic variables, trace elements, organic contaminants and microbiological indicators. The grouping used in Slovakia resembles those used in other countries, although it can be more detailed with respect to the effect of pollution on the water environment and chemical properties, as it is categorized by UNESCO/WHO/UNEP.

The classification scheme used in Slovakia defines water simply as clean or polluted (in five categories) regardless of its prospective uses. This lack of relation with clearly defined water usage goals makes it difficult to apply the classification in the practice of water resources management. It can be overcome only by making limits usage-oriented.

On the other hand, the current classification has some attractive features. One good point is the grouping of the pollutants to five categories, with effects of pollution in each group to be assessed independently of the others. The most serious pollutant is assumed to define the water quality class in each group, thus providing robust and safe worst-case estimation. The other good point (now adopted in much Western European water quality legislation) is the probabilistic manner in which the standard is formulated, namely the threshold level is not to be exceeded in 90% of all the observed cases for the classification to apply (see Appendix 7.1 for details). Unfortunately, the water quality monitoring system in Slovakia at best provides monthly measurements, thus making the mention of the occurrence rates in the standard definition something of a good intention. The frequency of sampling should be increased at some important locations (e.g. river mouth) in order to provide reliable estimates for 90% probability levels, trends and pollution loads to the downstream catchment. An analysis of sampling frequency from the probability theory viewpoint can be found in Appendix 7.5.

7.2.4 The monitored parameters

The list of water quality parameters which are regularly analyzed in the monitoring procedure of the Nitra watershed is presented in Appendix 7.3. The numbers in the cells indicate the beginning year of the regular observations and data collection for the respective parameter. For some parameters/locations, observations prior to 1974 are also available. The list of sampling stations can be found in the Appendix 7.2 (refer to Figure 7.1 for locations).

The set of monitored parameters mainly coincides with the parameters included in the water quality classification outlined above. Not all parameters are monitored at all stations; the analyses which are more expensive or more complex are taken less frequently than the others. The main indicators of domestic wastewater pollution, or so called "traditional" parameters (BOD, COD, nitrate and ammonia nitrogen, coliform bacteria, and basic physico-chemical parameters (temperature, suspended and dissolved solids, pH) are covered comprehensively enough. The indicators of industrial, agricultural, and complex pollution such as heavy metals and organic substances are monitored less frequently. Of the heavy metals, only arsenic is monitored with any regularity in the Nitra River basin. Copper, zinc, cadmium, chromium, and nickel are in the list but monitored only at two stations. And such important components as mercury, lead, chlorinated and polycyclic organics, all of them highly toxic, are not monitored at all. In short, only so-called "traditional" pollution is accounted for in the monitoring program currently performed.

The bioassay tests, while evaluating coliform bacteria, do not provide important information about the bacteria of the fecal group (fecal coliform and fecal streptococci) which are indicative of poorly treated domestic wastewater contamination. Bioassays of water toxicity for particular species of aquatic life can serve as useful integral indicator of industrial and agricultural pollution, but they are not regularly performed in the frame of the monitoring program.

The list of parameters is also incomplete with regard to substances potentially giving rise to eutrophication, such as total and orthophosphate phosphorus. Total phosphorus was absent from the list until 1990, and orthophosphate was monitored only occasionally at some stations. Organic nitrogen was also not sampled until 1990 and since then it has been monitored only from time to time in a few locations. Tests of chlorophyll-a or algae biomass have not been routinely performed, and detailed hydrobiological assessment of the Nitra River or its tributaries has not yet been undertaken yet.

From the evidence of high water pollution, it is to be concluded that the sediment of the Nitra River is contaminated as well. Therefore it is not true that an improvement of water quality will immediately follow the reduction of point-source and non-point-source pollution. A comprehensive program on the river sediment is needed which will estimate the effects of accumulated pollution and approximate the costs of cleaning it up (Appendix 7.3).

All of this calls for certain reshaping of monitoring procedure. The list of parameters should include non-traditional pollutants and analyses of biological materials. The recommended improvements of monitoring system are outlined in Appendix 7.5, a summary of which will be given in the Section 7.4.

7.3 Evaluation of the Multipurpose Monitoring of the Water Quality in the Nitra River Basin

7.3.1 Classification of the current water quality of the Nitra River basin

The water quality classification for the Nitra river for 1989-90 according to CSN 75 7221 is illustrated in Appendix 7.4 as published by SHMU. The table contains columns corresponding to the sampling stations (cf. Appendix 7.2, Figure 7.1) and rows for the water quality parameters, the roman number in each row denotes the resultant water quality class for the given parameter group and the Arabic number denotes the observed parameter value.

As oxygen regime is considered, the most upstream part of the Nitra River (sampling points P1 - P3, see Appendix 7.2 and Figure 7.1) exhibit quite satisfactory conditions (dissolved oxygen concentration does not drop below 9.0 mg/l). The resulting Class III (polluted water) is caused by the chemical oxygen demand measurements. It is known, however, that the chemical oxygen demand is not necessarily related to harmful pollution of water. Some naturally occurring materials such as humic and fulvic acids originating from erosion of soils can contribute to the amount of chemical oxygen demand, although they are biologically inert and pose no threat to the aquatic environment. Comparatively low values of biological oxygen demand (BOD-5) support the hypothesis that organic material giving rise to COD values is of natural origin, since there is no industrial pollution in these upstream reaches. In this case, some drawbacks of the given set of criteria can be seen.

As the river enters industrialized and populated regions downstream (Novaky, Partizanske, Nitrianska Streda, see Figure 7.1), water quality seriously deteriorates. Nitra below Novaky is characterized by several oxygen drops (reaching 3.0 mg/l at Nitrianska Streda and close to zero at the mouth in Komoca), and high BOD values (27 mg/l at Praznovce, 29 at Nove Zamky, see Figure 7.1). It is classified as very intensively polluted (class V) for most of its flow. Both industrial and municipal emissions contribute to the organic pollution of the Nitra River (see Chapter 6).

According to the EPA water quality criteria, dissolved oxygen values below 5.0 mg/l may endanger fish population and are especially dangerous for the larval stages of its development. The value of 4.0 mg/l is the lowest level at which a varied fish community can still be observed. Based on the dissolved oxygen values observed in the Nitra River it may be concluded that the fish community in this river may be adversely affected by oxygen reductions.

The parameters from the Group B (basic chemical parameters, see Appendix 7.1) classify Nitra River upstream of Novaky as polluted on the basis of total dissolved solids concentration (above 500 mg/l). Large amounts of dissolved solids are emitted into the river by the chemical industry at Novaky (see below). From this location down to the mouth, dissolved solids concentration remains high, reaching 1400 mg/l at Chalmova (Figure 7.1). This concentration of dissolved solids (mostly mineral salts) makes water of Nitra River unsuitable for drinking water supply because of adverse taste and physiological effects combined with high costs of repairing the supply equipment (Quality Criteria for Water, 1976). Starting from 500 mg/l, total dissolved solids can cause detrimental effects on the most sensitive crops when this water is used for irrigation.

The parameters in the group C (additional chemical parameters) are mostly cations and anions which compose mineral salts commonly found in terrestrial waters. This group provides essentially no new information in addition to total dissolved salts (total dissolved solids).

The Nitra River downstream of Novaky is classified mostly as very intensively polluted on the basis of high ammonia contents and suspended solids concentration. At Chalmova, the water is too alkaline (pH=8.5), which is the effect of the effluent from sedimentation reservoirs belonging to the chemical industrial complex in Novaky. This is dangerous for the aquatic life itself, but combined with a total ammonia concentration of about 3 mg/l, it can result in extremely high amount of toxic undissociated ammonia (in these conditions as much as 10% of ammonia will remain in undissociated form). The concentration of 0.3 mg/l of undissociated ammonia can be lethal for many freshwater fish species. Ammonia concentrations remain dangerously high further downstream.

High concentrations (above 80 mg/l) of suspended solids also can be found downstream of Novaky, at least partially due to the chemical industry discharges (cf. Figure 8.6). Suspended solids with a concentration above 80 mg/l can adversely influence virtually all kinds of aquatic life, water column and benthic. It also reduces the light available for the phytoplancton species to photosynthesize. Aesthetic considerations require low enough turbidity for the waterbody to be suitable for bathing, swimming and other recreational purposes. Irrigation use of highly turbid water is difficult because of possible crust formation on the soil as well as problems with pumping. In short, the classification of the waters of the Nitra River as extremely polluted (which follows from the Slovak water quality classification B Group) is fully justified.

Virtually no use of the Nitra River water below Novaky other than industrial intakes is possible in the present situation due to ammonia contents, turbidity, and dissolved salts (dissolved solids). Both ammonia and suspended solids enter the river mainly from the wastewater discharged from municipal and industrial plants.

In Group D (heavy metals) measurements were conducted only for arsenic (Appendix 7.3). In several locations, arsenic concentrations reach dangerous levels not only for the human consumption but also for the irrigation of crops (above 100 μg/l). In these locations, the water is appointed Class V (very intensively polluted). This is especially worrisome due to bioaccumulation properties of arsenic, so that adverse health effects will amass with consumption of arsenic-containing food and drinks and thus most likely will affect local population. The origin of the arsenic is not quite clear. It does not seem likely that the arsenic comes from the point-source polluters. One can suppose that the arsenic pollution could be caused by the weathering of the arsenic-containing minerals, i.e. it has geologic origin. Alternatively, it may be contained in large amounts in the river sediment (secondary pollution) if contamination had occurred in the past.

Finally, group E indicators (Appendix 7.1) demonstrate the abundance of bacterial population usually related with discharge of fecal waters (coliform bacteria), obviously caused by insufficient or ineffective treatment of municipal wastewater effluents.

For comparison purposes, the river classification used in the UK (Tebbutt, 1992) gives for the upper part of Nitra River (monitoring points P1 and P2 upstream of Novaky) class 1A. This is high quality water suitable for potable supply abstractions. It should provide favorable conditions for proliferation of fish communities. The river downstream to Novaky can be assigned only a 3rd class rating. This class covers waters which are polluted to an extent that fish are absent or only sporadically present. This water may be used only for low-grade industrial abstractions.

Some additional features of pollution in the Nitra River basin could be gathered from longitudinal profiles, which are based on data from the dedicated measurement program conducted in August 1992 during low-flow conditions (Chapter 8). The dissolved oxygen drops below 4.0 mg/l at several locations in the profile (Figure 8.6). By the European Community regulations (EC Environmental Directives, 1978), this water is unsuitable for coarse fish. As already noted, the US EPA directives likewise require the dissolved oxygen to be above 4 mg/l for protection of aquatic life (Quality Criteria for Water, 1976). The critical level of biological oxygen demand is defined as 6 mg/l by the EC standards and is surpassed at all monitored locations.

Ammonia nitrogen concentration is above 1 mg/l at several locations (Figure 8.7), high enough to endanger the fish life. The mandatory fish protection limit value for the total ammonia adopted in the EC is 1 mg/l. In the USA it is specified that the unionized fraction NH_4OH should not exceed 0.02 mg/l (Quality Criteria for Water, 1976). Under the conditions occurred in the Nitra River during the experiment (water temperature above 20°C and pH around 7.5) it would lead to the same 1 mg/l limit value. The drinking water standards usually are stricter (for example, 0.5 mg/l in the EC).

The levels of phosphorus concentrations (Figure 8.7) are close to 1 mg/l. The Slovak classification puts a value of 1 mg/l of total phosphorus as a limit for Class IV water (intensely polluted).

The pesticide lindane (hexachlor cyclohexane) level (Figure 8.8) exceeds the EPA limit for the aquatic life protection (0.01 µg/l; Quality Criteria for Water, 1976). High lindane concentrations are known to harm both the fish population and the invertebrates. This pollutant is not analyzed in the Nitra River basin on the regular basis. The origin of lindane is most likely agricultural (Chapter 8). Chemical industry complex in Novaky is a source of a range of other organic micropollutants (dichloroethane, trichloroethene, tetrachloride ethene and several others; see Chapter 8).

In short, the Nitra River basin is exhibiting characteristic features of water environment degradation, marked by combination of municipal and industrial waste disposal at clearly insufficient treatment levels.

7.3.2 Long-term changes in the water quality in the Nitra River basin

Besides reporting the state of the environment, the multi-purpose monitoring program should address the issue of temporal changes of that state. For evaluating the long-term changes in water quality in the Nitra River, the annual extreme values which will determine water quality class (annual maximum or minimum where appropriate) were plotted against the location of sampling and the observation year.

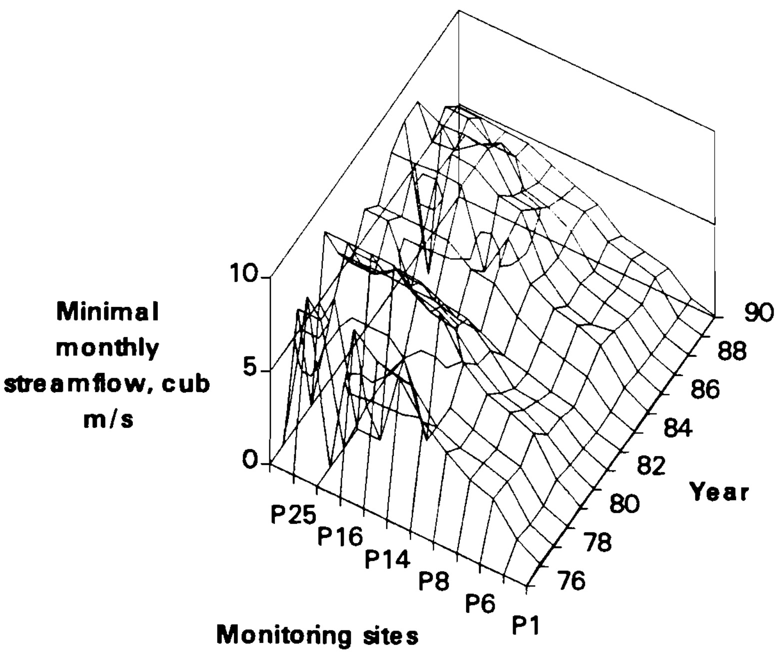


Figure 7.2 Minimal monthly streamflow values in the Nitra River derived from the monitoring database (1976-1990).

Figure 7.2 displays the minimal monthly streamflow registered in the given year plotted with respect to the location (cf. Appendix 7.2, Figure 7.1). This plot can be used to infer what long-term changes, if any, took place in the hydrological regime in the Nitra River over the observation period covering almost fifteen years (1976-1990). There are no visible trends in this plot apart from the usual variability of the river streamflow.

Figure 7.3 likewise shows the minimal annual level of the DO concentration plotted with respect to the location and to the observation year. The spatial changes do follow the above mentioned pattern of deteriorating water quality from upstream to downstream. Three pronounced oxygen sags can be discerned, following the general pattern of sources of organic material emission: Novaky, Partizanske, and Nove Zamky (Chapter 9). These emissions are also distinct in the Figure 7.4, which shows the over decades changes in the annual maximum of BOD-5. The pattern of point-source pollution is visible in the Figure 7.5 (ammonia concentration) as well.

When compared to Figure 7.2, the plots in Figures 7.4 and 7.5 exhibit inverse proportionality, i.e. periods with high river flow correspond to low instream pollution concentrations due to an increase in dilution. Apart from that fact, no pronounced long-term trends are visible from these plots. It is to be concluded that the patterns of organic material pollution did not change drastically over the observation period. Most likely, the pollution of the river reached its contemporary level after intensive development of industry in the region after the Second World War.

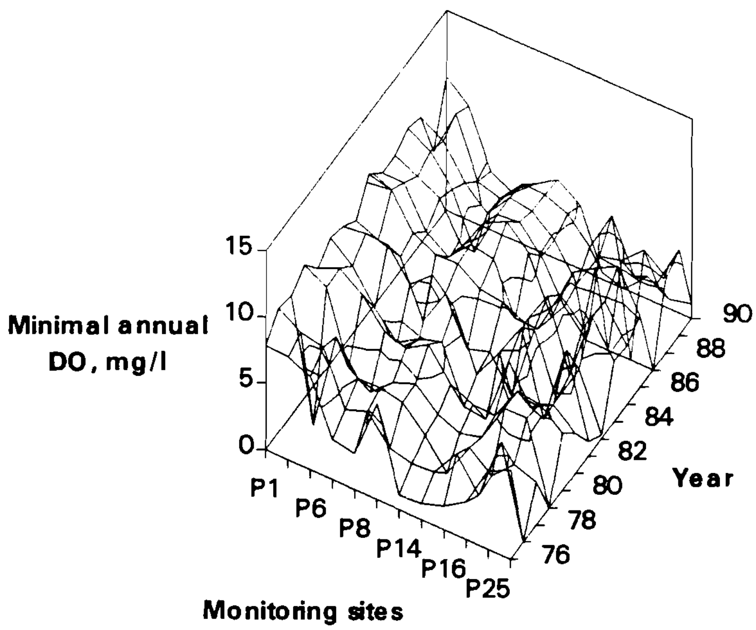


Figure 7.3 Minimal annual DO level in the Nitra River derived from the monitoring database (1976-1990).

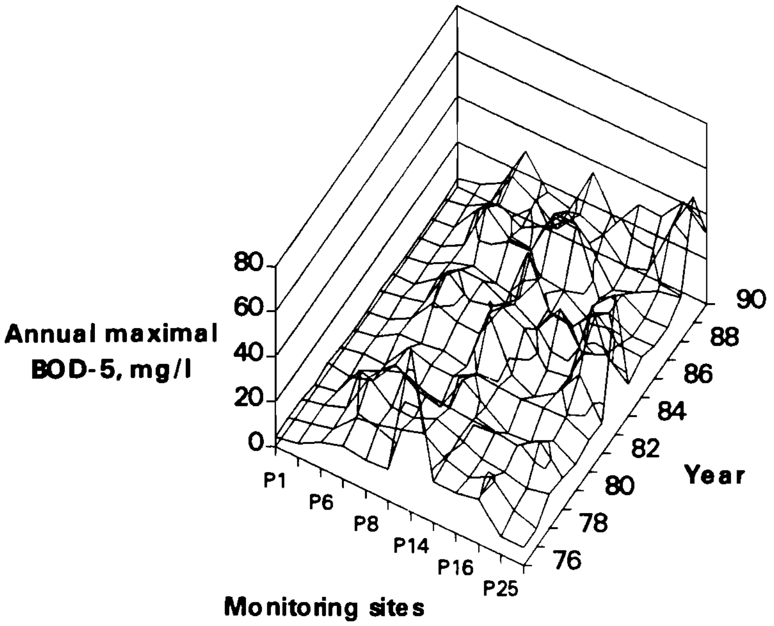


Figure 7.4 Maximal annual BOD-5 level in the Nitra River derived from the monitoring database (1976-1990).

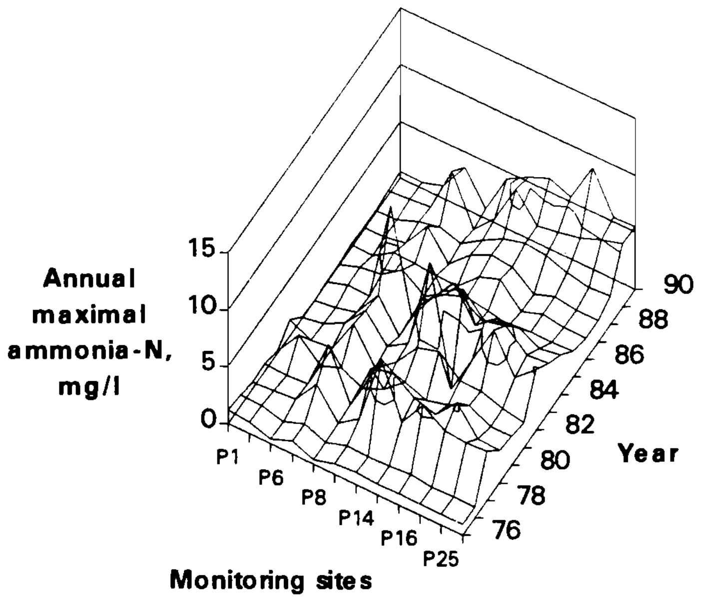


Figure 7.5 Maximal annual NH4-N level in the Nitra River derived from the monitoring database (1976-1990).

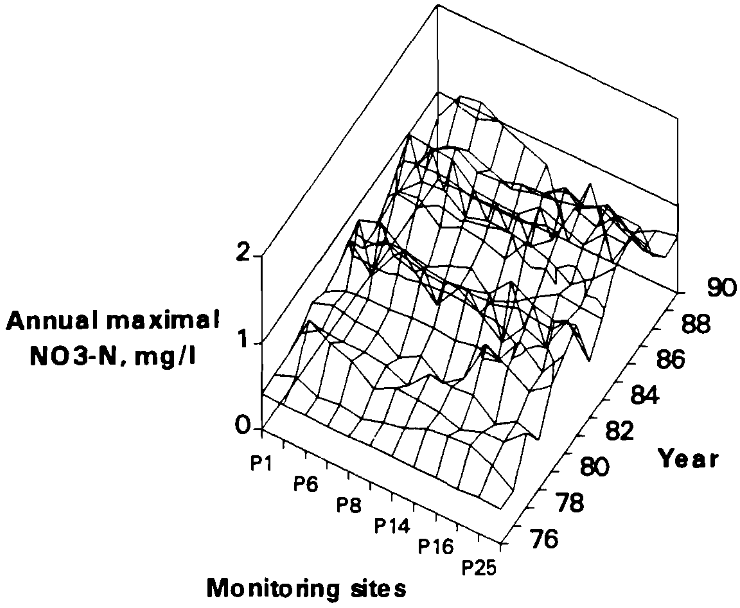


Figure 7.6 Maximal annual NO3-N level in the Nitra River derived from the monitoring database (1976-1990).

The maximal annual nitrates concentration (Figure 7.6) does not clearly follow the pattern of point-source pollution. The spatial distribution of the nitrates concentration is much more uniform and generally decreasing from upstream to downstream. This is understandable, since point-source nitrate emissions are less significant than ammonia point sources (Chapters 6 and 8). Likely source of nitrates is non-point source pollution related to agriculture (Chapter 8). There is a certain increase in the instream nitrates concentration in the period from 1980 to 1988. This is in harmony with the data of fertilizer application (Table 7.1).

Table 7.1 Application of nitrogen in fertilizers in Slovakia since 1950 until 1990 (Statistical Yearbook of Slovakia, 1991).

Year	1950	1970	1980	1989	1990
nitrogen in fertilizers, kg/ha	5.6	51.7	96.6	90.9	83.1

The over-decades changes of maximal annual dissolved and suspended solids concentration are shown in Figures 7.7 and 7.8. The concentration of dissolved solids rise significantly at the monitoring point P7 (Chalmova). This is an effect of the effluent from the sedimentation reservoirs of the chemical industry in Novaky (typical level of TDS 7000 mg/l, see Chapter 6).

Later the river water is diluted with other inflows, and the dissolved solids concentration gradually decreases. However, the water in the immediate vicinity of the chemical industry outflow is not suitable not only for drinking water supply but also for irrigation of most crops (Water Quality Criteria, 1976). The suspended solids concentration falls more rapidly below the sedimentation reservoirs, since the suspended particles concentration is affected not only by dilution, but also by sedimentation. (Figure 7.7-7.8).

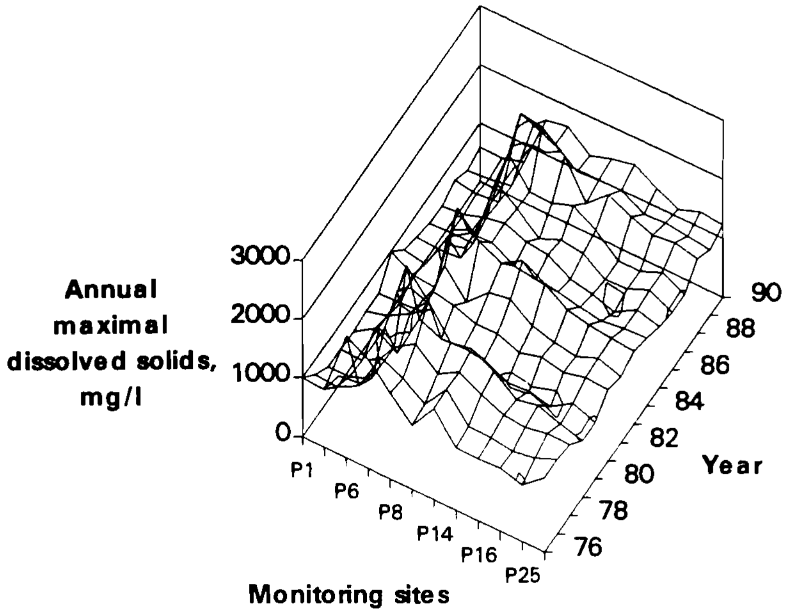


Figure 7.7 Maximal annual dissolved solids concentration in the Nitra River derived from the monitoring database (1976-1990).

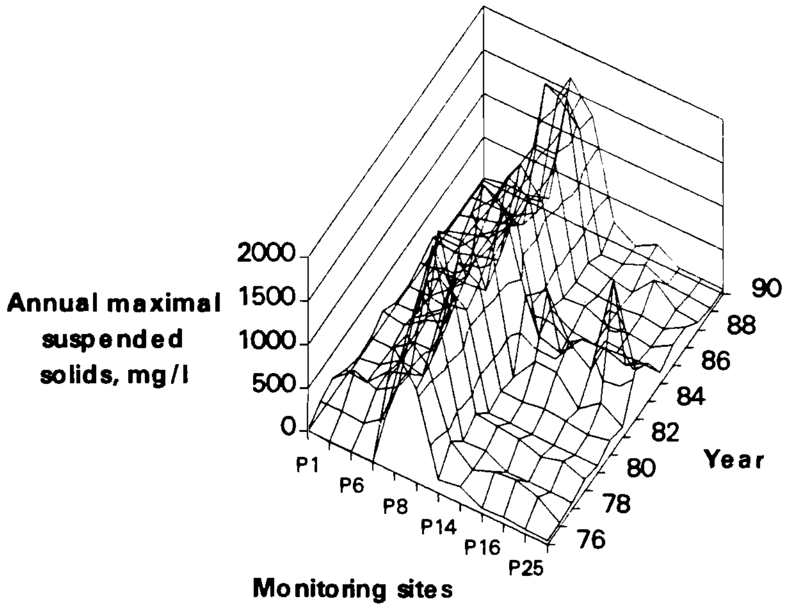


Figure 7.8 Maximal annual suspended solids concentration in the Nitra River derived from the monitoring database (1976-1990).

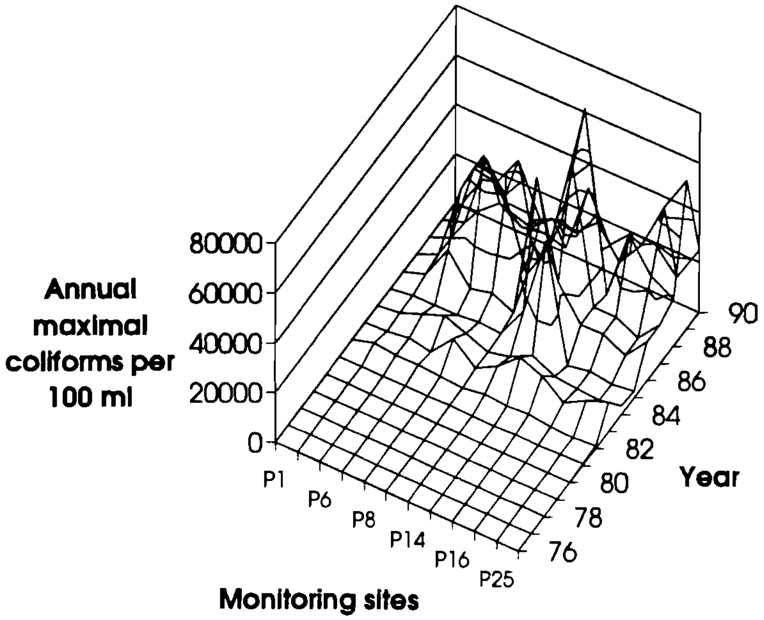


Figure 7.9 Maximal annual coliform bacteria level in the Nitra River derived from the monitoring database (1976-1990).

Coliform bacteria are very useful indicators of water pollution with fecal matter. Temporal and longitudinal changes in bacteriological water quality are shown in Figure 7.9. Longitudinal alterations reflect properly the effect of point sources from the mid-eighties. However, due to the likely changes in the analytical procedure temporal trends can not be established.

Finally, Figures 7.10 and 7.11 give some idea about the changes in the degree of organic pollution of the Nitra River in 1991 (using the 1990 as a basis for comparison). It can be seen clearly from Figure 7.10 that in the upper part of the river, which is affected by industrial emitters, there was a substanial improvement in the oxygen demand concentration. It should be noted that the economy in the region is undergoing economic transition and industrial, enterprises have been experiencing serious financial difficulties since the beginning of the 1990s. Most of the industries were compelled to reduce their production output (tannery factory in Bosany, river km 100; chemical industries in Novaky, river km 130). This fall in production was favorably reflected in the water quality of the river. In the lower part of the river, where most part of organic pollution comes from the municipal emissions, there was no significant reduction of the level of instream oxygen demand.

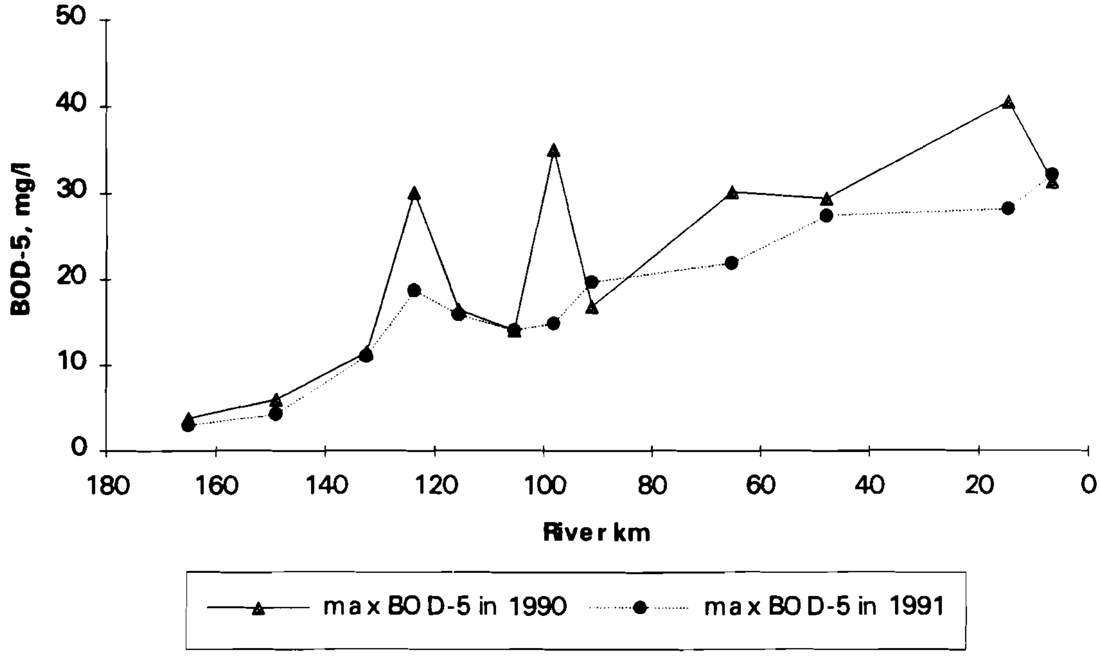


Figure 7.10 Maximal annual BOD-5 concentration in the Nitra River in the years 1990 and 1991.

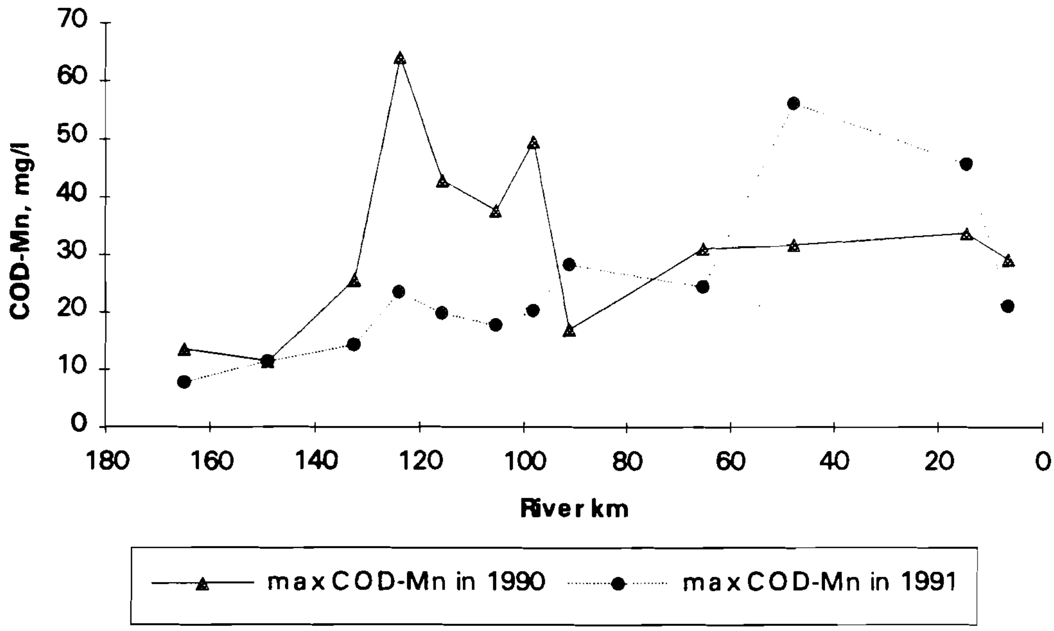


Figure 7.11 Maximal annual COD-Mn concentration in the Nitra River in the years 1990 and 1991.

7.3.3 Seasonal changes

Besides evaluating long-term (inter-annual) changes, the multipurpose monitoring of water quality may be used to establish the seasonal (intra-annual) changes in the observed parameters. For this purpose, a seasonal dataset was created by averaging the observations made in a particular month over the whole observation period. For most parameters, it was justified because there were no distinct inter-annual trends (Section 7.3.2) The seasonal changes in the water quality according to the database for 1976-1990 are presented below.

Figure 7.12 shows the seasonal changes in the dissolved oxygen concentration in the Nitra River. There are two pronounced features which can be observed from this plot. The first is the decrease in the level of dissolved oxygen in the river from upstream (monitoring site P1) to downstream (P26). These changes are caused by the pollution of the river with organic material and its stabilization as discussed above. Secondly, the lowest values of oxygen in the river occur during the low-flow periods at the end of summer and beginning of autumn (months VII-X). The water temperature is high at this time of year, diminishing the solubility of oxygen gas in the river water. This is the critical period from the point of view of the oxygen regime since the proper dilution of waste matter with river water cannot be achieved and a drop in the oxygen concentration occurs as a result imbalance of oxygen consumption and atmospheric supply.

The seasonal pattern of organic pollution can be observed from the Figure 7.13, which shows the seasonal changes in the biochemical oxygen demand (BOD-5). The maximum BOD concentration again occurs during the low-flow period when dilution capabilities are limited. The overall increase of the BOD concentration from upstream to downstream indicates successive emission of the waste materials into the river followed by destruction of the waste by microorganisms. Other indicators of organic pollution such as the permanganate chemical oxygen demand (COD-Mn, Figure 7.14) and ammonia nitrogen (NH4-N, Figure 7.15) follow a similar pattern as the BOD concentration.

The deterioration of the water quality in the Nitra River from upstream to downstream is also demonstrated in the Figures 7.16 and 7.17. The saprobic index (Figure 7.16) systematically increases from 1.5 at the upstream end to values close to 3.0 at the river mouth. The coliform bacteria number per ml of water (Figure 7.17) peaks around the emissions of municipal wastewater. It too is much lower in the upstream locations which are undisturbed by human activity.

It can be noted that peaks of bacterial activity in the lower stretches of the river (coli forms and saprobic index) fall to the late autumn early winter period which is also characterized by high organic pollution. The reason is most likely the operation of sugar beat factories in Surany and Nitra which starts in September (Chapter 6).

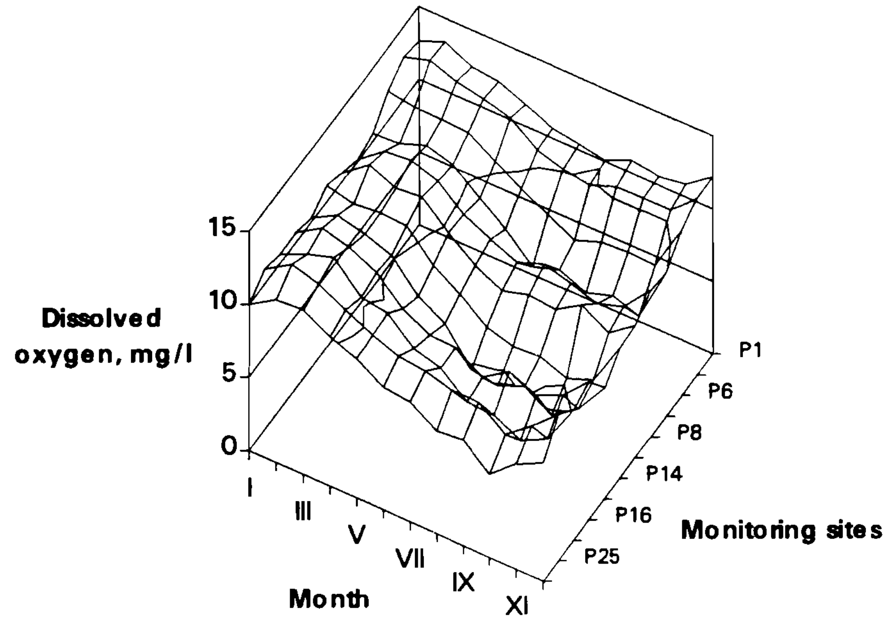


Figure 7.12 Seasonal changes in dissolved oxygen concentration in the Nitra River derived from the monitoring database (1976-1990).

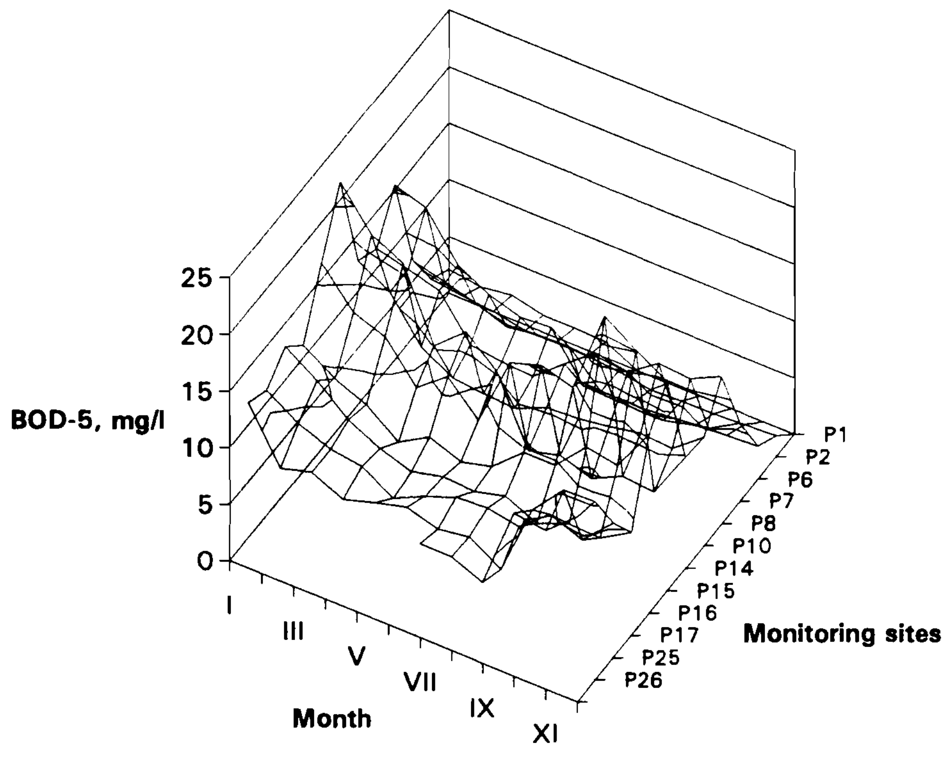


Figure 7.13 Seasonal changes in BOD-5 concentration in the Nitra River derived from the monitoring database (1976-1990).

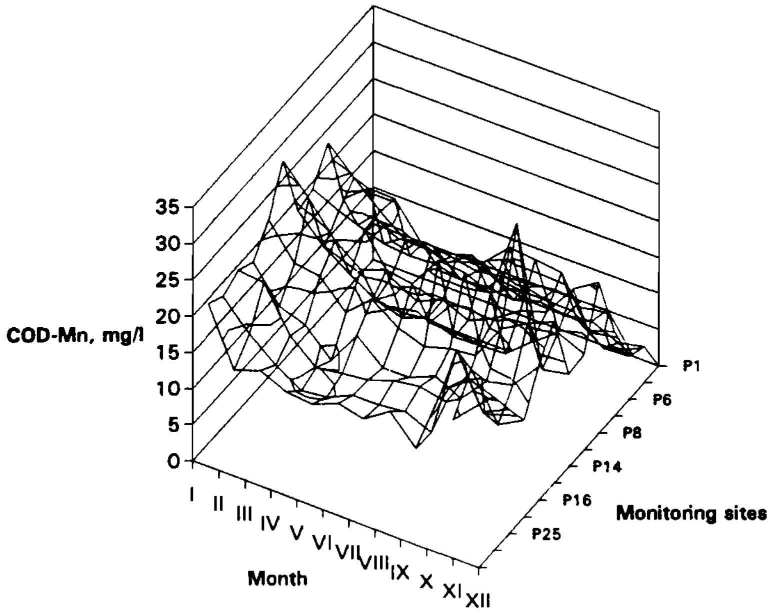


Figure 7.14 Seasonal changes in COD-Mn concentration in the Nitra River derived from the monitoring database (1976-1990).

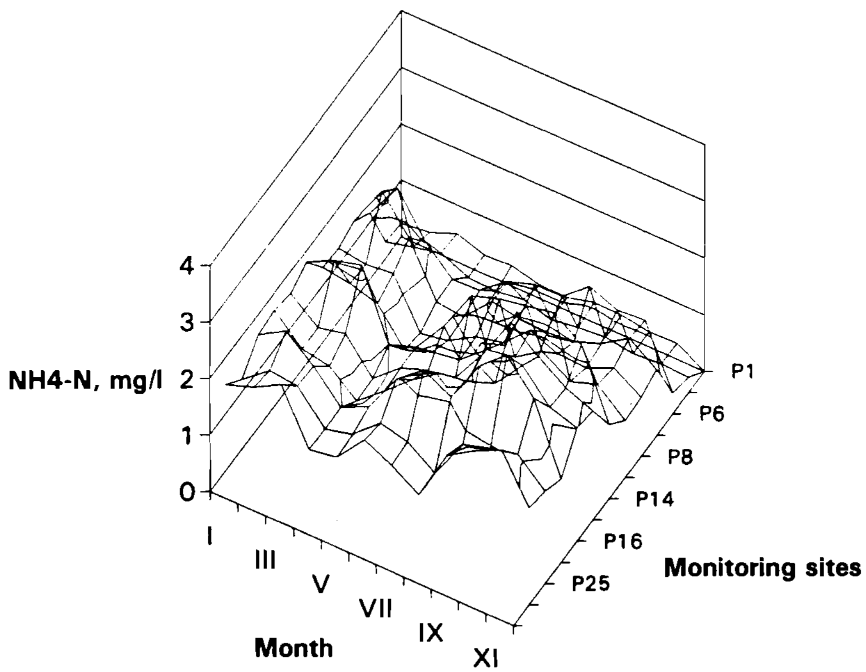


Figure 7.15 Seasonal changes in NH4-N concentration in the Nitra River derived from the monitoring database (1976-1990).

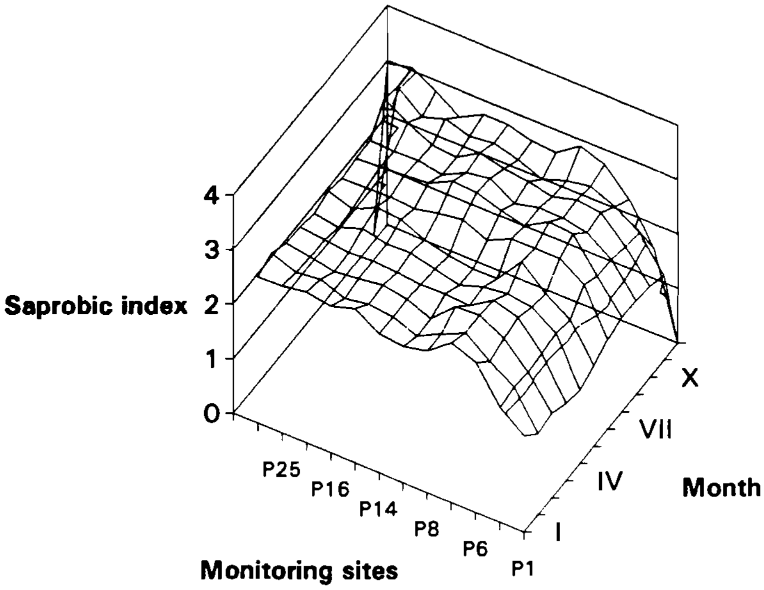


Figure 7.16 Seasonal changes in saprobic index in the Nitra River derived from the monitoring database (1976-1990).

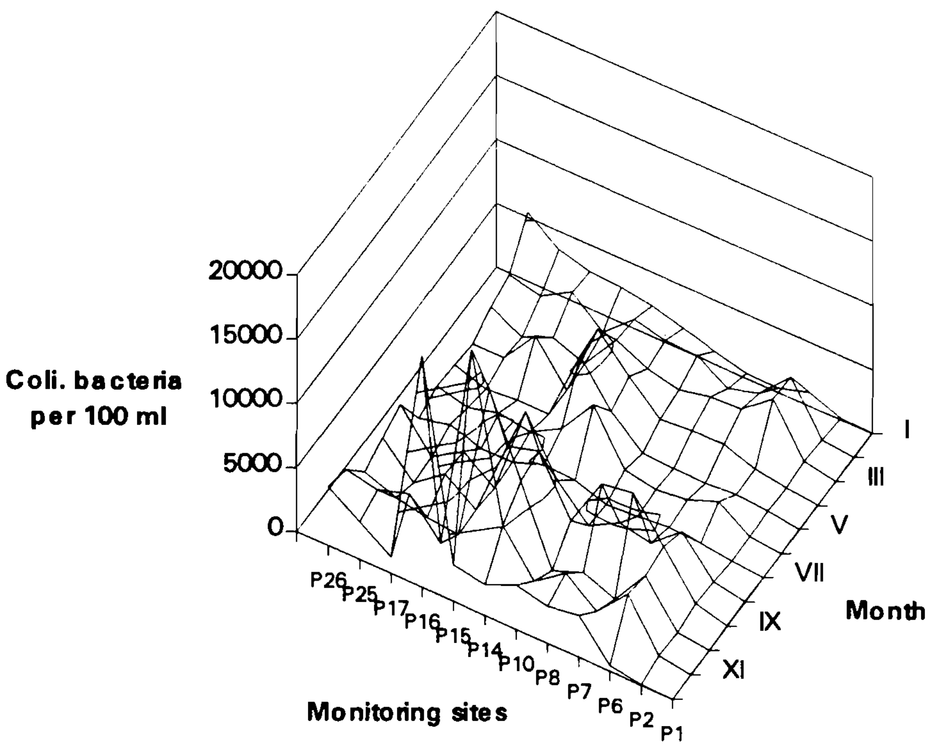


Figure 7.17 Seasonal changes in coliform bacteria in the Nitra River derived from the monitoring database (1976-1990).

7.3.4 Water quality answer sheet

The UNESCO/WHO/UNEP guideline for water quality assessment (Water Quality Assessment, 1992) provides a convenient frame for defining results obtained with a long-term monitoring program. The Table 7.4 summarizes this answer sheet for the most of the Nitra River length where the water can be inexpensively abstracted to satisfy any economical need (downstream to Novaky).

Table 7.4 Water quality assessment answer sheet (continued on the next page).

Question	Answer
Is the water fit for drinking?	No. Reason: frequent excesses of ammonia nitrogen, COD, arsenic, coliforms, and BOD levels over what is required for drinking water. High saprobic index.
Is the water fit for other major uses?	
-agriculture	Conditionally, yes. Sometimes, pH, dissolved solids, and arsenic may limit the suitability of the water for irrigation.
-industry	Yes.
-recreation	No. Body-contact recreation is limited by significant organic pollution and related phenomena: unpleasant odors, color, and high bacteria concentrations. The bathing in the Nitra River below Novaky should be prohibited, and measures should be taken to ensure public awareness of the fact that this activity is dangerous.
Is the quality of the aquatic environment adequate to support the growth of the expected aquatic biota?	No. Organic pollution and related oxygen depletions prevent development of adequate fish populations. Ammonia level can be dangerously high in many stretches.
What are the time variations of water quality?	From the point of view of water quality, the low-flow period is the most critical . Some seasonal food industry units put a stress on water quality when in operation.
What are the long term trends in water quality?	No significant long-term trends were detected over the period of observation. Nitrates concentration in the river follows in general the pattern of fertilizer application, with maximum in mid-1980s. The recent decline in industrial output (1990s) was favorable reflected in the river water quality.

What is the flux of pollutants in the river?	According to the August 1992 experiment (Chapter 8), organic material load expressed as BOD-5 flux (kg/day) is distributed as follows (see Section *.* for more details):	
	total emitted load	11280 kg/day
	of that: municipal emissions	6953 kg/day or 62% of total
	industry	4327 kg/day or 38% of total
Where are the major pollutant sources?	"Traditional" pollutants: wastewater treatment plants located at Partizanske, Topolcany, Nitra, Nove Zamky; sugar beat factories at Surany and Nitra; tannery at Bosany; a number of smaller municipal wastewater treatment plants discharging into the catchment. Non-traditional pollutants: chemical industry complex at Novaky. Heavy metals: The origin of the arsenic in the river water is unclear, possibly due to geologic processes or to secondary pollution	
What is the regional distribution of water quality?	Water quality is acceptable in the upstream stretches (upstream of Novaky) and seriously deteriorates downstream.	
Are pollution control measures adequate?		
Monitoring	Emissions measurements should be included into the monitoring procedure. Sampling locations should make allowance for complete mixing at the site. The list of measured parameters should include non-traditional pollutants, perhaps at a cost of de-emphasizing some of the currently performed measurements to remain in the limits of a reasonable budget. Ecotoxicological tests should be considered.	
Wastewater treatment	Several wastewater treatment plants are overloaded and their efficiency is low. They are in need of upgrading.	

7.4 Summary of recommendations on monitoring

Based on the above material and analysis of present water quality monitoring system (Appendix 7.5), the following recommendations are drawn for enhancement of the water quality monitoring system in the Nitra River basin:

(1) The overall goals of the monitoring program should be defined. They should be ranked with respect to their relative importance and relation to uses of the waterbody. Monitoring should focus on the water quality problems impacting defined uses and

requirements. As objectives change, the monitoring program should be reevaluated to reflect the new goals and policy.

(2) Water quality emission data should be obtained, stored and published along with the data on the quality of receiving waters. This is a necessary precondition to the use of the monitoring information in a policy-oriented way.

(3) The location of monitoring sites should be revised to eliminate the potential of incomplete mixing at the sampling point.

(4) The aquatic life, its major prerequisites such as nutrients (nitrogen and phosphorus) and the biota itself (phytoplankton, invertebrates, fish) are in need of pilot study. Subsequent decisions on the monitoring should be taken on the basis of the study. Total N and total P along with their dissolved fractions should be included in the monitoring program.

(5) The expansion of the list of monitored parameters should be considered to include at least pilot testing of industrial and combined pollution indicators such as toxic heavy metals and organic micropollutants. Toxicity bioassay tests should be considered at least in the form of occasional lethality evaluation of invertebrata (Daphnia). The toxicity tests would be more important at later stages when the acuteness of the organic pollution would decrease due to the emission control. The bioassay test should assess bacteria of the fecal group.

(6) The frequency of the analyses should be adjusted with regard to the statistical properties of the water quality components and the used criteria. Two types of corrections may be needed, first introducing the dependency of the admissible limits on the length of the time series, second taking into account the standard deviation of the particular parameter with respect to its mean value. The frequency should also consider the budget limitations and possibly de-emphasize certain ongoing analyses in favor of more needed new ones (p.1)

(7) A pilot program for the estimation of accumulated pollution in the sediments should be planned and implemented. On the basis of this program the potential of secondary pollution sources should be evaluated and be dealt with in the course of pollution abatement.

7.5 Policy Oriented Conclusions

From the previous material dealing with the intensive industrial pollution caused by the chemical complex in Novaky, no research is really needed since we understand the origin of the pollution and the control action is obvious. The adequate treatment of the effluent from the chemical factory would improve water quality downstream of Novaky from Class V to Class IV class in Group A (the rest of the pollution is caused by organic material emissions from the small wastewater treatment plants on the Handlova River) and from Class V to Class III or IV in Group B.

The pollution caused by municipalities requires a more detailed analysis because there are many more sources (Chapter 6) and their effect on the water quality much more difficult to analyze. The organic material undergoes destruction in the river water, accompanied by consumption of dissolved oxygen. The major source of oxygen in the river is atmospheric reaeration, which is strongly dependent on the physical parameters of the flow. Therefore, the resulting oxygen balance is difficult to evaluate without use of mathematical models. On the other hand, the elaboration of a concise plan for the upgrading of municipal wastewater treatment plants also could be facilitated greatly by the use of operation research tools available on computers (Chapter 10). Those topics are a focus of the reported work (Chapters 11-13).

The study of non-traditional pollution of the river by heavy metals and persistent organic substances should start with the comprehensive survey of the occurrence and origin of such pollution. For the moment, we can do little more than recommend that such a survey be conducted covering both the effluent and receiving waters (including toxicity tests for the small invertebrata and fish species). After finding the hot-spots of pollution, the control is usually an installation of dedicated pretreatment facilities removing the harmful substances before the effluent water enters the routine treatment system.

The non-point pollution (not covered by the reported work) most often is caused by agriculture. This again requires more study, focused not as much on receiving waters as on the practice of applying fertilizer and other chemicals to the fields and ways of manure disposal. The proper control actions would consist of regulating the application and promoting environmentally sound agricultural techniques.

Finally, the accumulated pollution residing in sediments could become important when control actions aimed at lowering the instream concentrations of pollutants begin to be effective. The release of pollution from sediment is dependent on the hydraulic conditions and on the amount of stored pollution. Essentially, the same is true for the inadequately maintained dump sites, which can be a source of a number of hazardous substances like, for instance, heavy metals. Dealing with this pollution can admittedly prove rather costly. In this case, a proper pilot study can be made to estimate of the stored pollution and evaluate the potential runoff and leaching from the dump sites. Control actions such as relocation of the dump sites and/or providing necessary isolation, introduction of sludge incineration, improving the sludge treatment, and even dragging the river sediment could be envisaged on the basis of such research.

On the basis of the analysis made above, the water quality control problem of the Nitra River basin is characterized by the type of pollution, the parameters affected, the pollution's origin and the corresponding control action in the Table 7.2.

Table 7.2 Summary of the water quality control problem of the Nitra River basin.

Description of pollution	*Affected parameters of river water quality*	*Source of pollution*	*Analysis required prior to control action*	*Type of control action*
intensive pollution causing changes in the basic physico-chemical parameters of the river water	pH, dissolved and suspended solids, low-molecular organics	chemical industry in Novaky	no research is required	adequate treatment facilities
organic pollution	dissolved oxygen reduction, bacteria growth, odor, and smell	municipal emissions and food-processing industry	policy-oriented study aimed at elaboration of cost-effective control plan	consecutive upgrade of municipal wastewater treatment facilities
non-traditional pollution	heavy metals, high-molecular persistent organics	industry	study of individual emitters	industrial pretreatment
non-point source pollution	nitrates, pesticides	agriculture	studies of non-point source pollution and agricultural practice	control of fertilizer application (amount and methods of distribution)
accumulated pollution	nutrients, heavy metals, nonvolatile organics	river sediment and dump sites	study of sediment pollution and leaching from dump sites	adequate isolation of dump sites, dragging of sediment

Note: Shaded boxes contain items focused on by the present research

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8. WATER QUALITY LONGITUDINAL PROFILE MEASUREMENTS AND MASS BALANCE ESTIMATIONS

8.1 Experimental Setup

The main purpose of the present study (see Chapter 4) is the evaluation and selection of feasible water quality management strategies. The focus is on the development of least-cost policies which take into account all of the particular features of the region. However, to develop such ambient criteria based policies it is necessary to translate the changes in emissions to the respective changes in the ambient water quality. Water quality models can provide the necessary link between the control action and the reaction of the ambient water quality (Chapter 5).

A variety of models were developed to describe the pathways of pollution in the aquatic environment. The use of these models involves a parameter estimation procedure which should be based on an appropriate set of measured data (Chapter 9). The regular monitoring routine does not provide all the necessary information. A mass balance cannot be completed without data on emissions, which has not been available with the required frequency (Chapter 7). Therefore, a well-designed study is needed in order to provide the necessary set of data for the calibration of models and setting the mass balances.

Two experiments were prepared jointly by Water Research Institute in Bratislava, the Váh River Authority and IIASA in the Nitra River basin. The first was executed in August 25-26, 1992. The June 7-10 1993 experiment was developed on the basis of the August 25-26 experiment results (see for details Masliev et al, 1994). Both experiments considered a mass balance approach and estimation of the key parameters of the water quality processes, which provided the basis for the subsequent decision-making process.

The parameter estimation procedure usually consists of two steps: calibration and validation (Beck, 1976). Calibration can use data collected in one region and the parameters obtained can be validated on the basis of data from another region (validation in space). Another procedure requires samples collected during two (or more) measurement campaigns, in which case the data from the second set of measurements provides the basis for validation in time. The data from the second experiment thus serves as a validation criteria (Chapter 9).

The timing of the first experiment was selected to cover the extreme low-flow event within the year. Under such conditions the diluting of wastewater effluent is minimal, and therefore this event is considered the most critical from the viewpoint of water quality management of source point pollution (see Chapter 5). The period of low-flow usually is defined as the reference time frame for checking ambient water quality standards (Chapter 7). The flow exceeding 355 days in an average year (Q_{355}) is used for checking ambient water quality envisaged by the Slovak Water Act (Chapter 7). The actual stream flow during the August 1992 experiment was less than the Q_{355} value everywhere in the river (Figure 8.5).

All stations of the water quality monitoring scheme of the Nitra River were included in the experiment, and three other sampling sites were added to the scheme. The emission data (concentration and flow) on both the industrial and municipal wastewater discharges were registered and stored. Several tributaries which are not regularly monitored were included in the program as well. Altogether, 48 sampling locations were selected. Concurrently, the Váh River Basin Authority carried out its regular monthly sampling, thus providing a possibility for comparison (Section 8.2). At two sampling stations on the Nitra river, namely Nitrianska Streda (river km 91) and Nové Zámky (river km 14, see Figure 7.1), the diurnal changes were monitored for 24 hours (Figure 8.28).

The entire watershed was divided into four regions to allow simultaneous sampling with four teams working independently. All selected sites were visited in advance and site photos were taken. The flow measurements were conducted at all measurement points. At some locations the streamflow was measured in advance in order to save time. Since the weather during the experiment was extremely dry, the streamflow changes were not significant. The Water Research Institute (VÚVH) was in charge of the streamflow measurements and sampling. The laboratory analyses of samples were done by the River Authority in Piestany. Some of the analyses were also repeated by the VÚVH laboratory for further intercomparisons of the results (Section 8.2).

The list of measured parameters was composed of:

- The main physical and chemical parameters (temperature, conductivity, pH, dissolved and suspended solids, major ions);
- Oxygen regime parameters (dissolved oxygen, chemical and biological oxygen demand)
- Inorganic nitrogen and phosphorus forms;
- Heavy metals (copper, zinc, cadmium, mercury, etc.);
- Industrial organic micropollutants (chloroform, 1,2 dichloroethane, 1,1,1 trichloroethane etc.);
- Pesticides (DDT, aldrin, dieldrin, DDE).

The complete list can be found in (Masliev et al, 1994).

The selection of parameters had been dictated by the most serious water quality problems of the region (Chapter 7). The major chemical and physical parameters are affected by the effluent of the chemical complex in Novaky, which is also a source of industrial organics. The dissolved oxygen problem is caused by the organic pollution from municipal wastewater treatment plants. Agriculture provides non-point source pollution of nitrates and pesticides.

In the course of the preparation of the June 1993 experiment, the Nitra River was subdivided to three stretches with respect to water quality. The water of the most upstream stretch (from the headwater to Novaky, river km 132) is practically pristine water (Classes I-II, see Section 7.1). The upstream stretch from Novaky to Nitrianska Streda (river km 91.1) is affected by industrial pollution caused by the chemical complex at Novaky and the tannery at Bosany (see Chapter 7). Several smaller municipalities are adding their discharges to this part of the river as well (Partizanske, Prievidza etc.) Finally, the pollution of the downstream stretch of the river (from Luzianky, river km 65 to the mouth) is mostly originated by comparatively large municipalities such as Nitra and Nove Zamky (Chapter 7).

Three regions were covered by four-day sampling programs (9-12 June 1993), two of them in the upstream stretch (Novaky-Nitrianska Streda) and one in the downstream stretch (Nove Zamky-Komoca). The daily repeated samplings provided information on the temporal variability of the water quality parameters as well as on the margins of analytical inaccuracy. Several additional locations were involved with the aim to understand the particular water quality processes in more detail. Moreover, the set of analyzed parameters was extended to provide more detailed information on the organic pollution composition (discerning the dissolved and particulate fractions, measuring total organic carbon, total phosphorus and nitrogen etc.). For the purposes of quantitative evaluation (e.g. estimation of the parameters of the water quality models) the data obtained from the particular locations were averaged and the resulting value was used according to the steady-state assumption of the analysis (Chapter 9). The accuracy of the steady-state representation is significantly higher when an average from several value is used to estimate the mean condition (Chapter 9). Because of the detailed spatial and temporal resolution of the second experiment, it is not possible to include all the analysis in this report (see for detail Masliev et al, 1994).

The chapter is organized as follows. First, the comparison between the regular sampling and the experiment data in August 1992 is given to illustrate the margins of experimental errors and the variability of the measured data (Section 8.2). Next the longitudinal profiles of major components and the overall water quality situation at the time of experiment are covered for both surveys (Section 8.3). Mass balances from the August 1992 experiment are discussed in detail in Sections 8.4-8.5. The detailed analysis and discussion of the mass balances for the June 1993 can be found in Masliev et al (1994).

8.2 Intercomparison of the Regular Monitoring and Experiment Sampling (August 1992)

During the experiment the regular monitoring samples were taken from the river independently by the River Authority. This setup gave an opportunity to perform a comparison between the two sets of measurements and to get an idea about the margins of possible experimental error and data variability. Comparison charts for some important water quality parameters are shown in Figures 8.1-8.4.

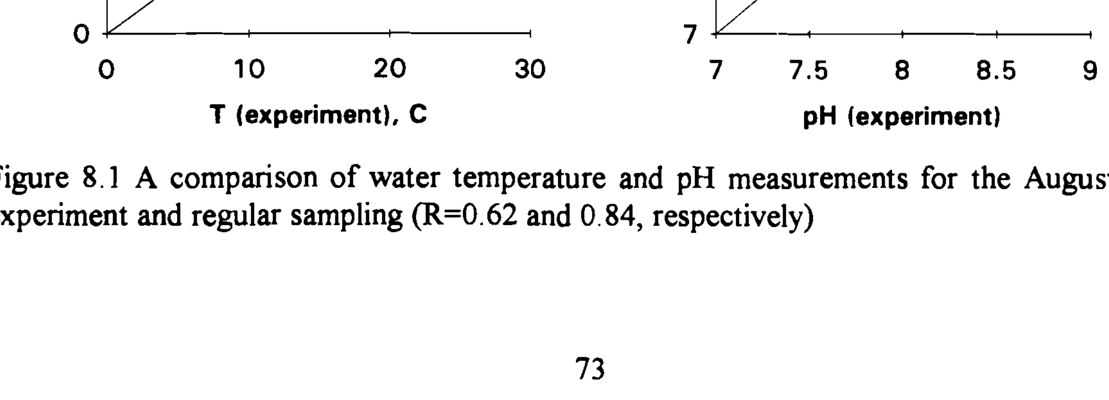


Figure 8.1 A comparison of water temperature and pH measurements for the August 1992 experiment and regular sampling (R=0.62 and 0.84, respectively)

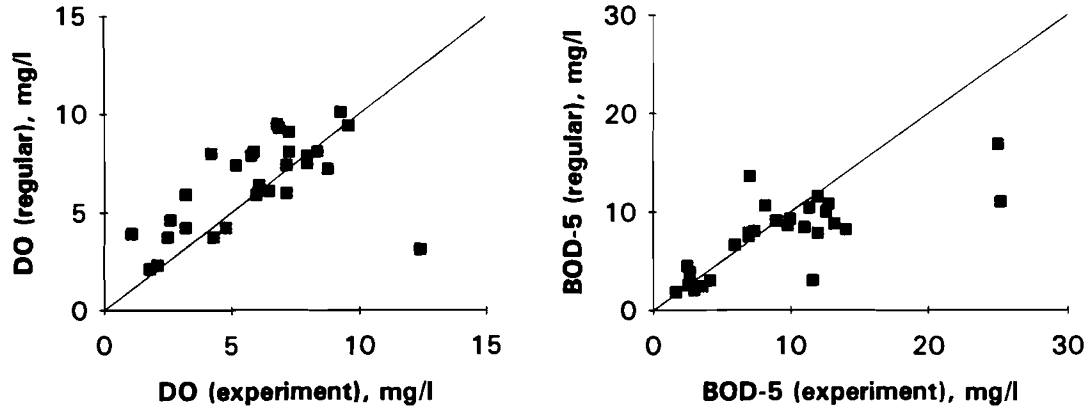


Figure 8.2 A comparison of dissolved oxygen and biological oxygen demand measurements for the August 1992 experiment and regular sampling (R=0.58 and 0.76, respectively)

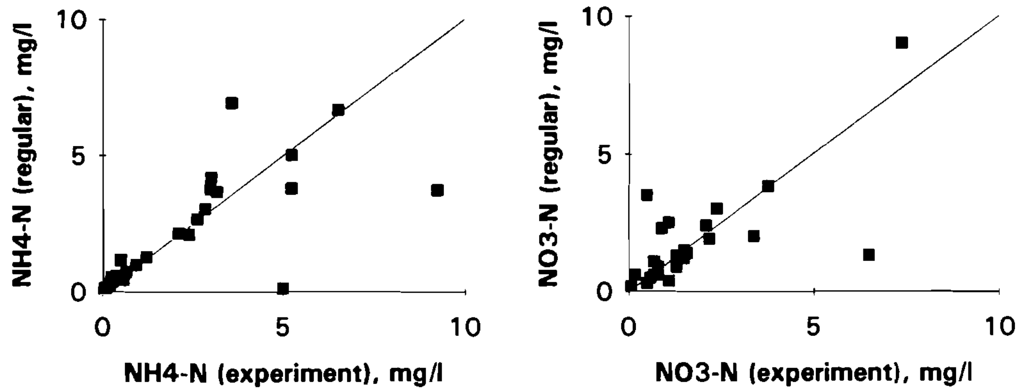


Figure 8.3 A comparison of ammonia and nitrate nitrogen measurements for the August 1992 experiment and regular sampling (R=0.78 and 0.7, respectively)

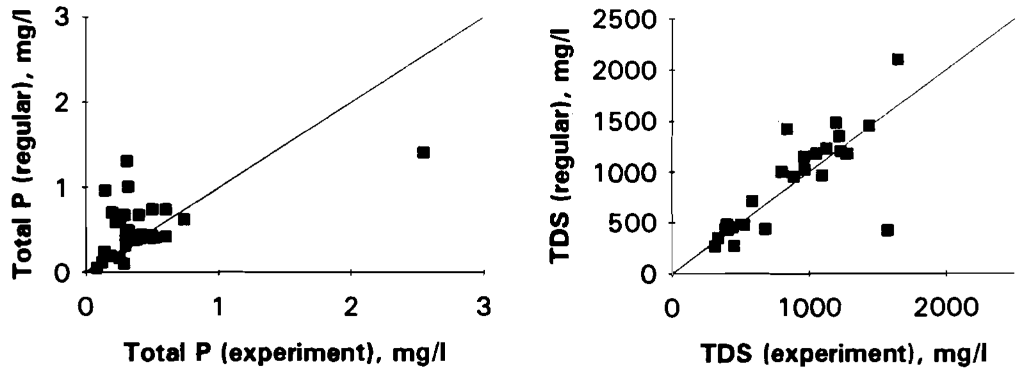


Figure 8.4 A comparison of total phosphorus and total dissolved solids measurements for the August 1992 experiment and regular sampling (R=0.53 and 0.79, respectively).

Water temperature was essentially in the same domain (20-25°C), but correlation is relatively poor (Figure 8.1). pH appears to correlate reasonably well. The dissolved oxygen measurements were in relatively good agreement, with the exception of one outlying value (Figure 8.2). The BOD-5 and nitrogen forms were measured with reasonable correlation

(Figure 8.2 and 8.3), the same could be said about total dissolved solids (Figure 8.4). The phosphorus concentration data, on the contrary, show relatively poor agreement (Figure 8.4). This suggests the need to look into the procedure of phosphorus measurement and to find out the cause of the observed discrepancy.

8.3 Longitudinal Profiles: the August 25-26 Experiment

The streamflow of the Nitra River is shown in Figure 8.5. The daily flow values exceeded in 355 and 364 days in an average year are plotted along with the actual ones from the experiment. It can be seen that the streamflow in the upper part would be overreached 355 days, in the lower part it is even less than the 364 day limit. The 355 day value is defined as a reference low-flow condition for water quality assessment by the Slovak environmental legislation (Chapter 7). Thus, the experiment corresponds to the design needs.

The drop in the streamflow between measurement point Nitrianska Streda (river km 91) and Luzianky (river km 65) is not explained by the known water intakes (Figure 8.5). The most likely cause is the irrigation withdrawal from one of the small reservoirs in this stretch at river km 142 or 156 (Jelsevce or Preselany). The flow difference was within the range of daily observations performed in 1993 and thus the (not well monitored) reservoir operation gives full evidence for the discrepancy found.

For the above reason the river was subdivided into two stretches (Figure 8.5), both having consistent flow data. These regions were then used for the purpose of calibration and validation (see Chapter 9 for details).

The longitudinal profile of dissolved oxygen for the whole Nitra River (August 25-26 1992 experiment) is shown in Figure 8.6. In this figure, biological and chemical oxygen demand levels are also given, which are closely related to the oxygen household of the river. In the middle part of the chart, the locations of emissions, tributaries and intakes are indicated with respective symbols. Although at discharge locations the instream concentration should experience an abrupt change (assuming complete mixing), they are not illustrated in the plot. The measured concentrations are connected with lines as if the concentrations were continuous along the course of the river. Subsequent longitudinal profile plots (Figures 8.7-8.8) follow the same fashion. A longitudinal profile showing the "real" concentration changes interpreted one-dimensionally is presented e.g. in Figure 9.6.

One can observe that dissolved oxygen drops below 4.0 mg/l at several locations in the profile, and sometimes it is close to 2.0 mg/l. High oxygen demand levels indicate that significant amounts of organic material were being discharged into the river (BOD-5 level is above 10 mg/l at most measurement points). According to the Slovak classification of the surface water quality, the river water at most sampling sites corresponds to Class IV (intensely polluted water) for oxygen related parameters.

The longitudinal distribution of the nitrogen forms is shown in Figure 8.7. The ammonia nitrogen originates mostly from the effluent of the wastewater treatment plants. The ammonia nitrogen in the lower part of the river is most likely subject to the nitrification process, since the drop in ammonia concentration is accompanied by the simultaneous rise in the nitrate concentration (see also Figure 8.2).

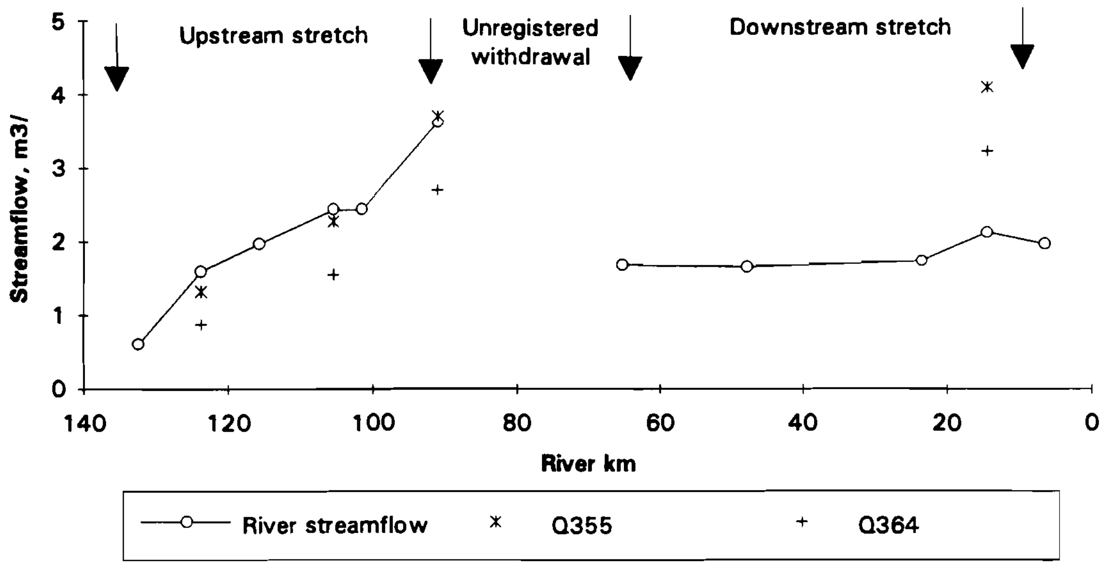


Figure 8.5 Longitudinal profile of the streamflow in the Nitra River (August 25-26 1992 experiment)

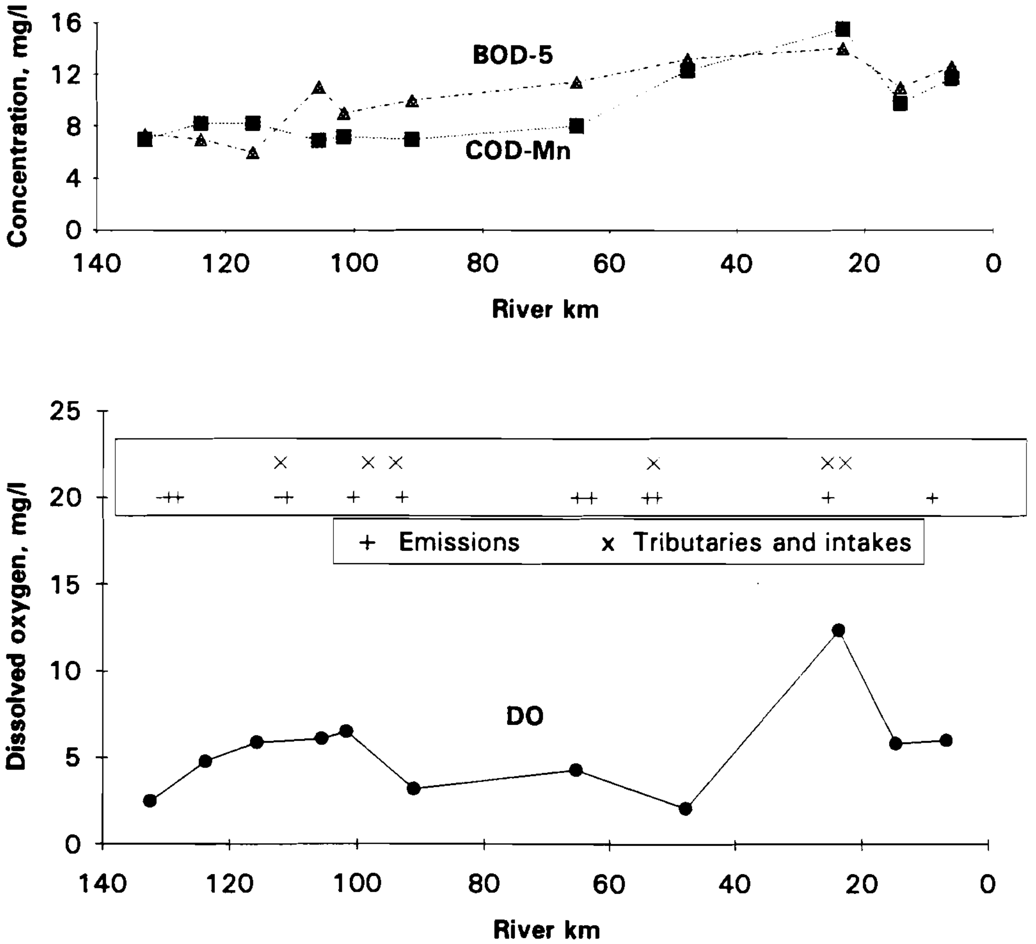


Figure 8.6 Longitudinal profile of the oxygen demands and dissolved oxygen in the Nitra River(August 25-26 1992 experiment)

The longitudinal profile of the phosphorus forms (total and ortophosphate) in the river is shown in Figure 8.7. It can be seen that there were some discrepancies in the analytical procedure, since for some samples the ortophosphate fraction appears to be less than the total phosphorus. The concentration of phosphorus is rather high (Chapter 7).

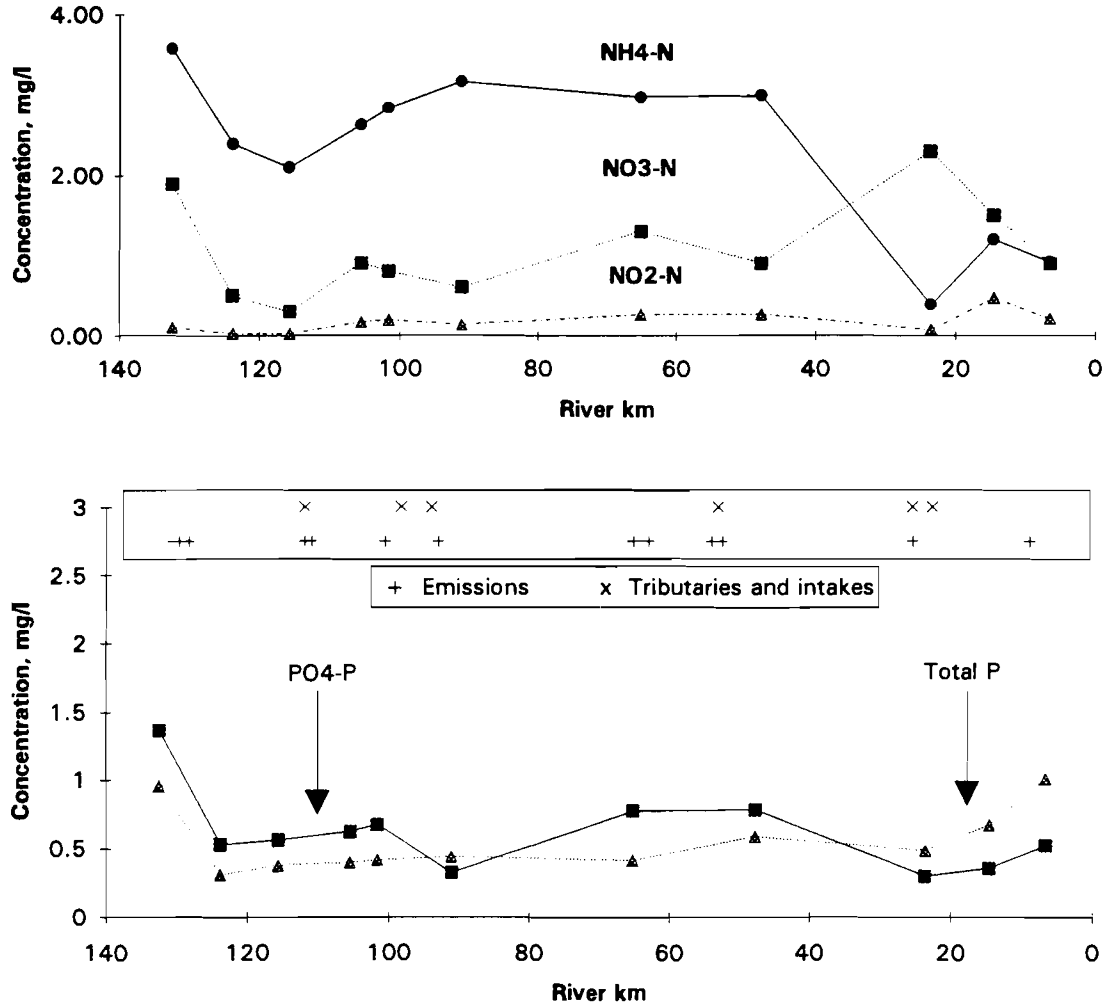


Figure 8.7 Longitudinal profile of the phosphorus and nitrogen forms in the Nitra River (August 25-26 1992 experiment)

Figure 8.8 shows the concentration of the pesticide lindane (hexachlor cyclohexane) in the water of Nitra River during the August 1992 experiment. It is logical to conclude that occurrences of the pesticide would be related to areas of agricultural activity and less to point-sources. The general pattern of the concentration profile in Figure 8.8 confirms this hypothesis, since the areas adjacent to the middle flow of the Nitra River are used for growing irrigated crops. The level of lindane is high enough to cause negative effects on aquatic life (Chapter 7).

The chemical industry complex in Novaky is a source of multiple organic micropollutants. In Table 8.1, the concentrations of seven of them are listed in the following order: concentration upstream of the complex, concentration in the factory effluent, concentration at the sampling point immediately below the discharge and further downstream. The pattern of occurrence of these micropollutants is the same in each case: below the sensitivity limit upstream of the complex, significant amount in the effluent and in the river downstream to the discharge and very low further downstream. Therefore, the origin of the micropollutants could be traced, without a doubt, to the chemical factory at Novaky.

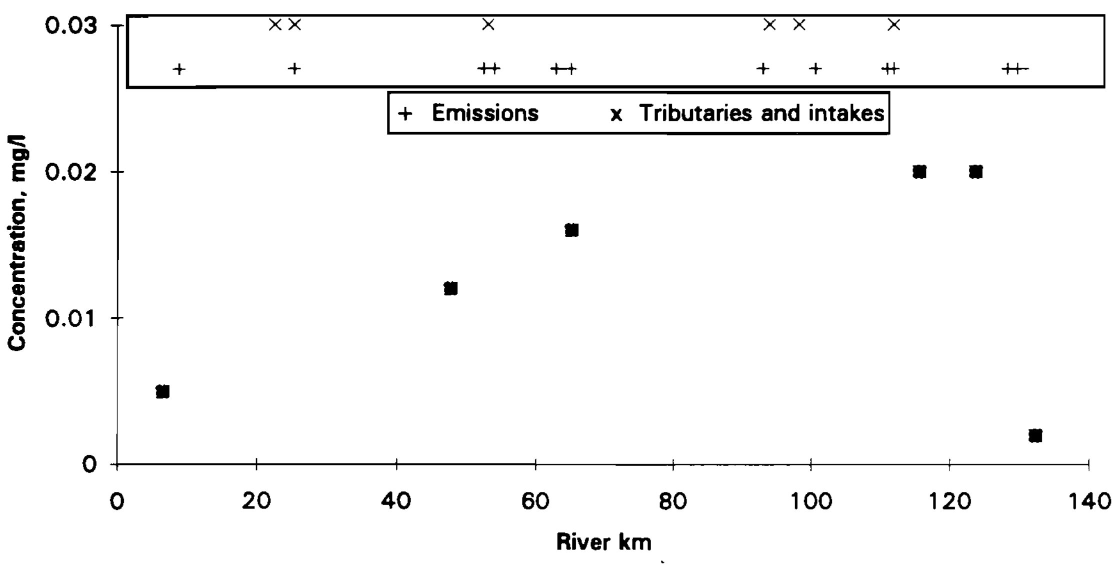


Figure 8.8 Longitudinal profile of lindane concentration in the Nitra River (August 25-26 1992 experiment)

Table 8.1 The organic micropollutants impact of the chemical industry in Novaky in August 1992

Location	River km	mg/l Chloroform	mg/l 1,2 Dichloroethane	mg/l 1,1,2 Trichloroethene	mg/l 1,1,2,2 Tetrachl. ethene
Novaky over	132.50	*	*	*	*
Chem.fac.Novaky	129.70	0.004	0.86	0.060	0.008
Chalmova	123.80	0.004	0.84	0.016	0.021
Partizanske over	115.70	0.001	0.55	0.006	0.008
Praznovce	98.20	*	0.01	*	*

Location	River km	µg/l 1,2 Dichlorobenzene	µg/l 1,3 Dichlorobenzene	µg/l Hexachlorobenzene
Novaky over	132.50	*	*	*
Chem.fac.Novaky	129.70	0.190	9.400	Not measured
Chalmova	123.80	0.013	24.000	0.005
Partizanske over	115.70	0.050	10.300	0.003
Praznovce	98.20	*	0.380	*

Note: * - below the analytically detectable level

Figure 8.9 shows the longitudinal profile of the river streamflow and the dissolved oxygen based on the results of the June 1993 experiment. It can be seen that the streamflow is significantly higher than during the August 1992 (cf. Figure 8.5). This period is usually associated with summer precipitation, producing surface runoff which contributes to an increase in streamflow.

The higher streamflow increases dilution of the wastewater, consequently, the concentrations of the pollutants are lower than during the low-flow (cf. Figures 8.6 and 8.10). Therefore it is not surprising that there are no sites with major dissolved oxygen reductions (Figure 8.9). As mentioned earlier, the higher streamflows during the June experiment allowed for the checking of the parameter estimation under different hydraulic conditions, therefore providing validation in time (Chapter 9).

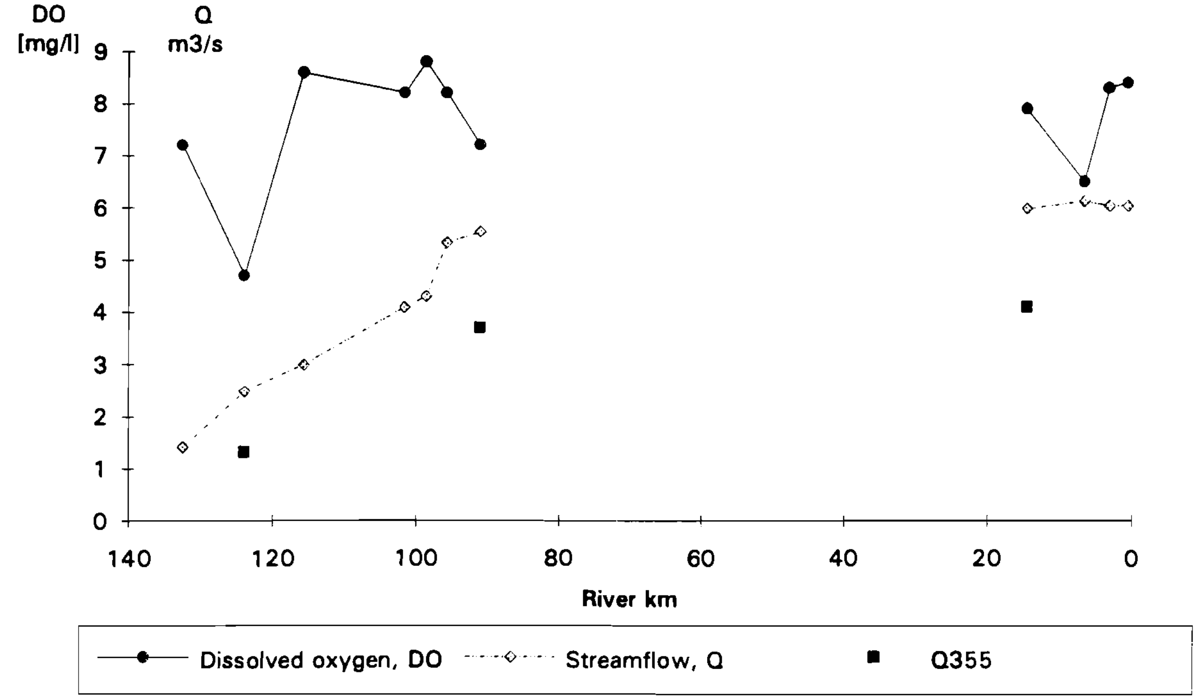


Figure 8.9 Longitudinal profile (partial) of the streamflow and dissolved oxygen in the Nitra River (June 1993 experiment)

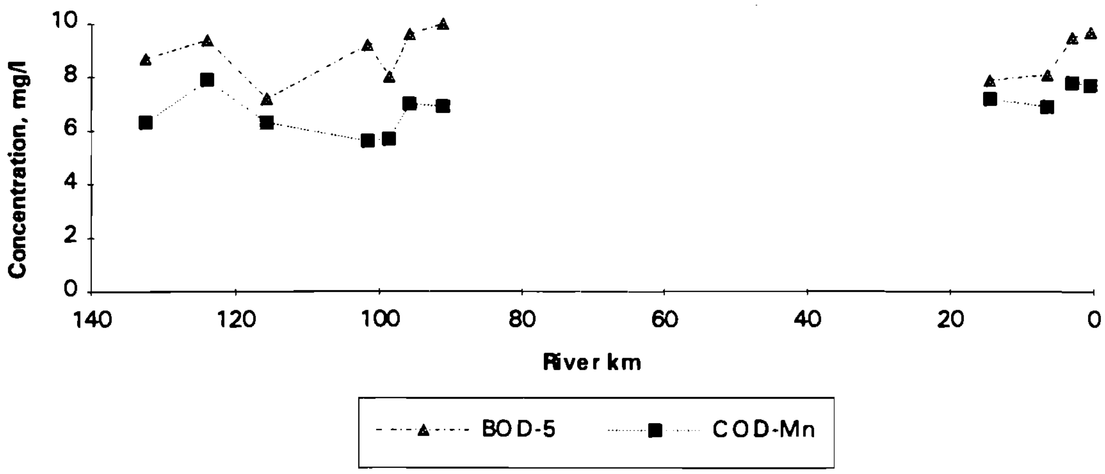


Figure 8.10 Longitudinal profile (partial) of the biological and chemical oxygen demand in the Nitra River (June 1993 experiment)

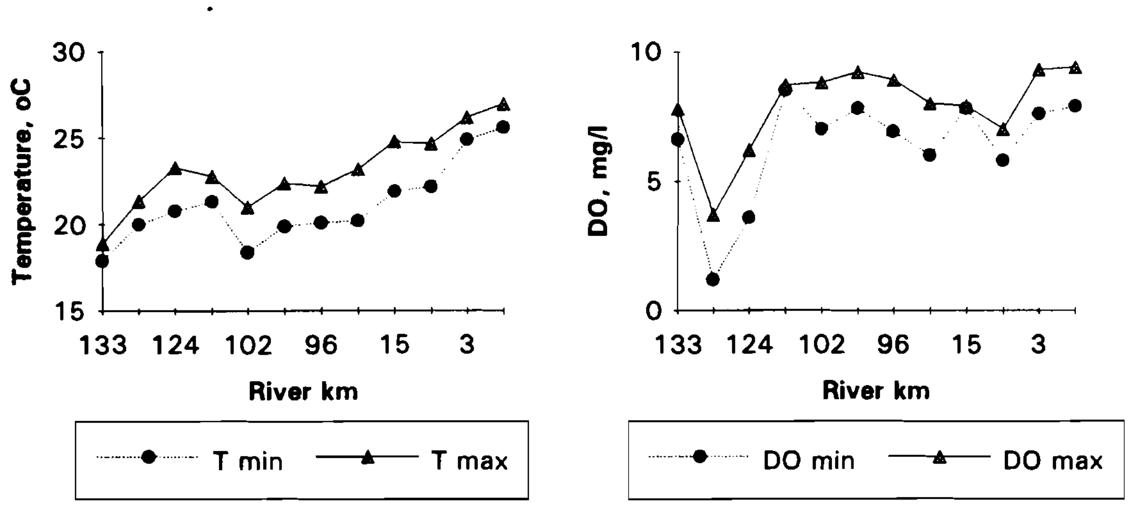


Figure 8.11 Extreme values from water temperature and dissolved oxygen data (June 1993 experiment)

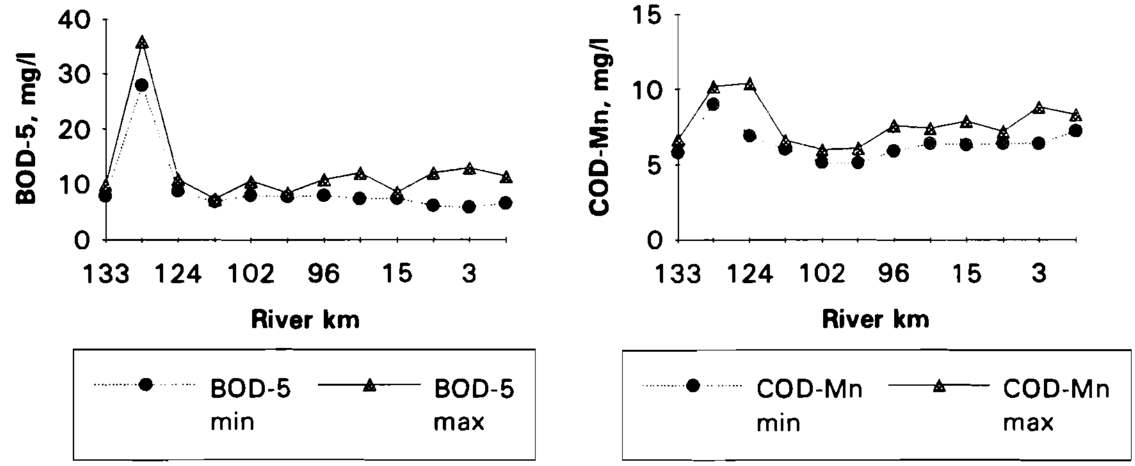


Figure 8.12 Extreme values from biological and chemical oxygen demand data (June 1993 experiment)

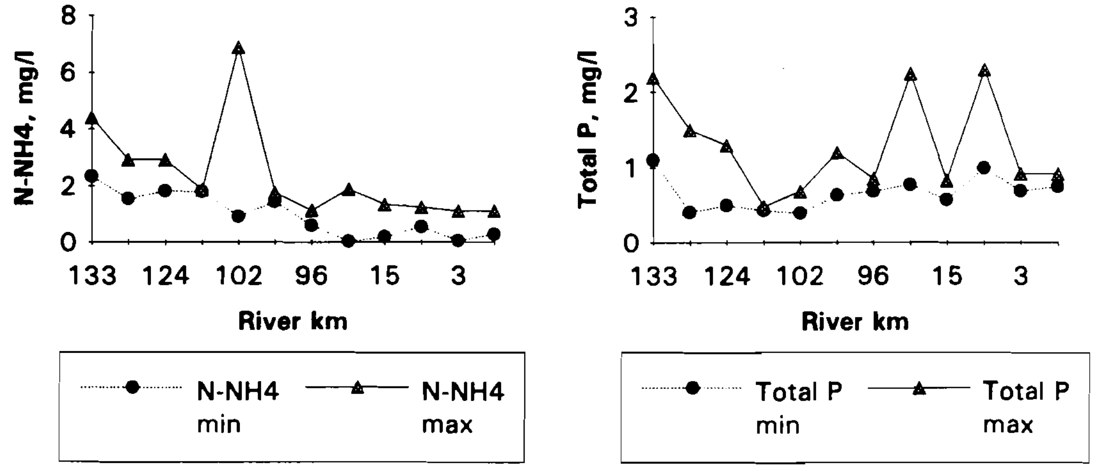


Figure 8.13 Extreme values from ammonia nitrogen and total phosphorus data (June 1993 experiment)

Finally, Figures 8.11-8.13 detail the scale of the temporal changes registered during the 4-day sampling program in June 1993. The minimum and maximum registered values are presented on the plots. For temperature, dissolved oxygen, chemical and biological oxygen demand the margins are narrow, i.e. the changes were not profound. However, for nitrogen and phosphorus (Figure 8.13) the changes are significant. The difference between the oxygen demand and the nitrogen-phosphorus data can be partially explained by variations in the phytoplankton uptakes subject to diurnal changes (Figure 8.29).

8.4 Mass Balances for the Upper Part of the Nitra River

The mass balance procedure accounts for material fluxes within a certain river region. Under presumed conditions, flux of a certain substance at a given location is equal to the flux at the upstream location plus all the emitted loads minus all the losses along the stretch considered. The loads and losses are specific for the substance under consideration, some of them being the result of discharge of the substance at certain emission points, others originate from non-point pollution and some may be caused by the chemical and biological processes in the river water. Thus, one can check the inventory of emitted loads and gain insight to the water quality processes.

The mass balance calculations were made in the following way. Starting from upstream, the flux of the substance was obtained and at any discharge point, the emitted amount was added. This cumulative mass represents the current flux in the river and was plotted as current mass flow. Mass flux was calculated at measurement points as concentration times the streamflow was subtracted from the cumulative mass flow. This difference was plotted as the balance residual which will be indicated by crosses in the accompanying figures.

A closed balance of water flow is a prerequisite for all subsequent balance calculations, since the flux of any substance is calculated on the basis of the flow measurements. The streamflow balance for the upper part of the Nitra River is shown in Figure 8.14. The cumulative river streamflow is plotted with the line connecting diamonds. The magnitude of the discharges (tributaries or emission points) are shown in filled triangles, and the balance residuals calculated at the measurement points are plotted in crosses. The streamflow showed a systematic increase in balance residuals, and therefore an incremental inflow of 0.027 m^3/sec per river km was introduced to provide a closed balance. The water quality of the incremental inflow usually is assumed to be close to that of groundwater. Smaller streams are presumed to be in good contact with the groundwater. For subsequent calculations, the concentration of substances in the incremental inflow was taken from the data on the small tributary, Nitrica, whose water quality parameters were close to those for other small creeks.

To check the validity of the assumptions used in order to close the streamflow balance (e.g. the amount of introduced lateral inflow), the mass balance of conservative substances (dissolved solids or chloride) can be evaluated. The chloride mass balance is plotted in the Figure 8.15 in a similar fashion as the stream flow balance (Figure 8.14).

The chloride-ion mass balance has some local deviations, but the overall balance error for the entire stretch is less than 10% of the mass flow. Bearing in mind the inherent uncertainties in the measurement procedures and the temporal changes of emissions (Chapter 6), the chloride mass balance can be regarded as adequately closed. The streamflow values obtained would be used as a basis for the subsequent estimation of other substance fluxes .

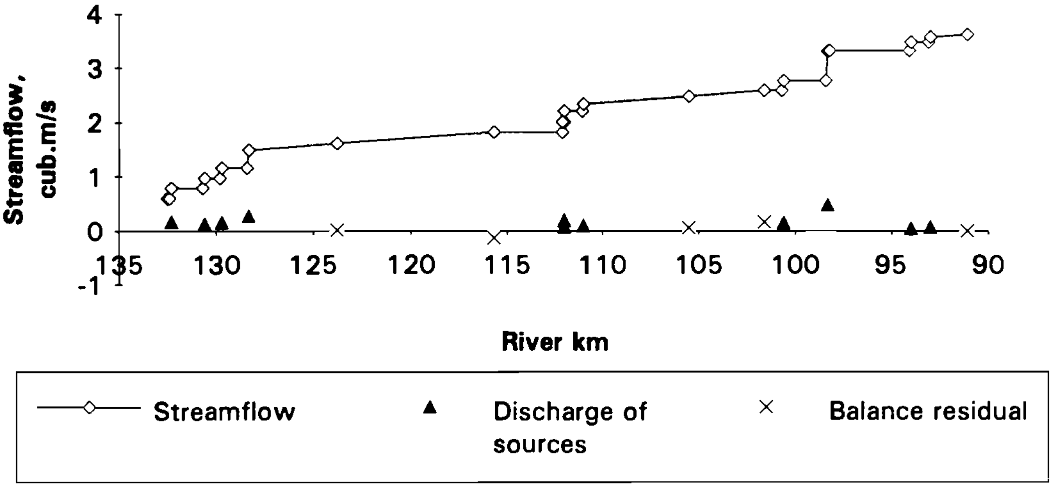


Figure 8.14 Streamflow balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

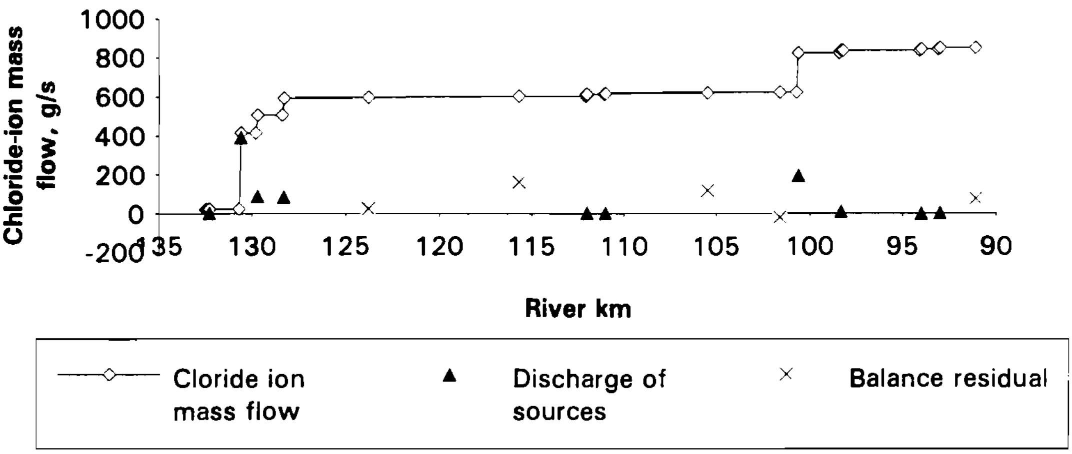


Figure 8.15 Chloride-ion mass balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

With regards to biodegradable organic material, river pollution is characterized by the amount of biological oxygen demand (BOD). The mass balance of BOD-5 (shown in the Figure 8.16) is affected not only by external sources, but also by the instream destruction of organics by heterotrophic bacteria.

As the organic material undergoes microbial decomposition, the balance residual in the mass flow is systematically decreasing, indicating a (instream) loss of mass. From the beginning to the end of the upper stretch of the river, the loss in mass flow is 40% of the total emitted amount. If we roughly estimate the average travel time of pollution in this stretch as 13 hours (half of the travel time for the whole stretch), the corresponding decay rate will be 1.0 1/day. This is close to the value 1.1 1/day obtained from the same data with the help of more sophisticated parameter estimation technique (see Chapter 9).

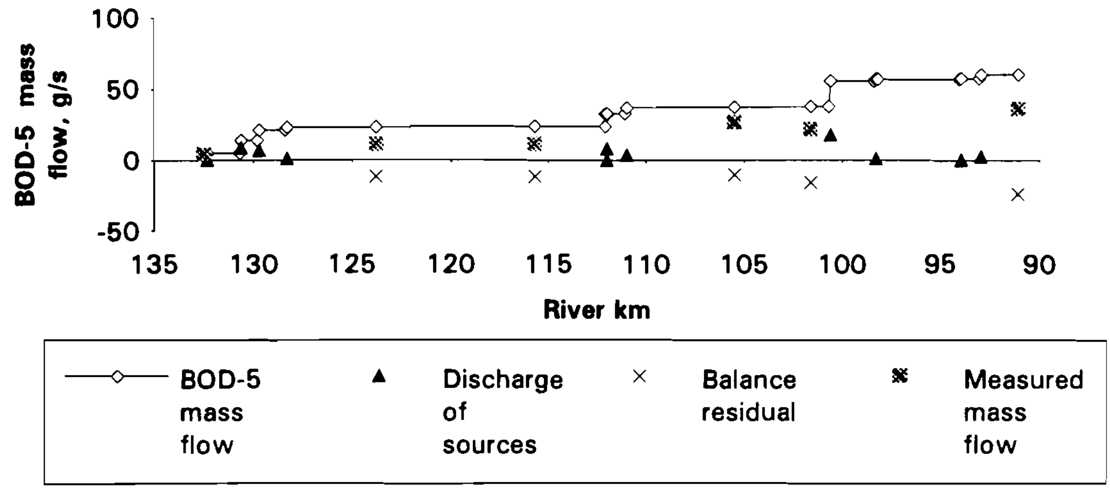


Figure 8.16 BOD-5 mass balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

The major emitters of BOD-5 and, consequently, organic material in this stretch of the Nitra River are (from the upstream to the downstream) chemical industries in Novaky, municipal sewage works at Partizanske, the tannery factory in Bosany and the municipal wastewater treatment plant in Topolcany. Of the total emitted mass of BOD-5 effluent, the tannery factory accounts for 35% of the emission, (the most intensive single polluter), the municipalities, together, account for 30% of total BOD emission and other industries comprise the rest.

The ammonia nitrogen mass balance is shown in Figure 8.17. One can see that the balance residuals are sometimes positive, indicating the presence of unaccounted-for sources of ammonia. These discrepancies are relatively small and could be caused by significant diurnal changes of emission intensity (see Chapter 6). The overall loss of ammonia mass constitutes 14% of the emitted amount, which leads to a 0.25 1/day estimation of the removal rate. The estimation from Chapter 9 is 0.24 1/day.

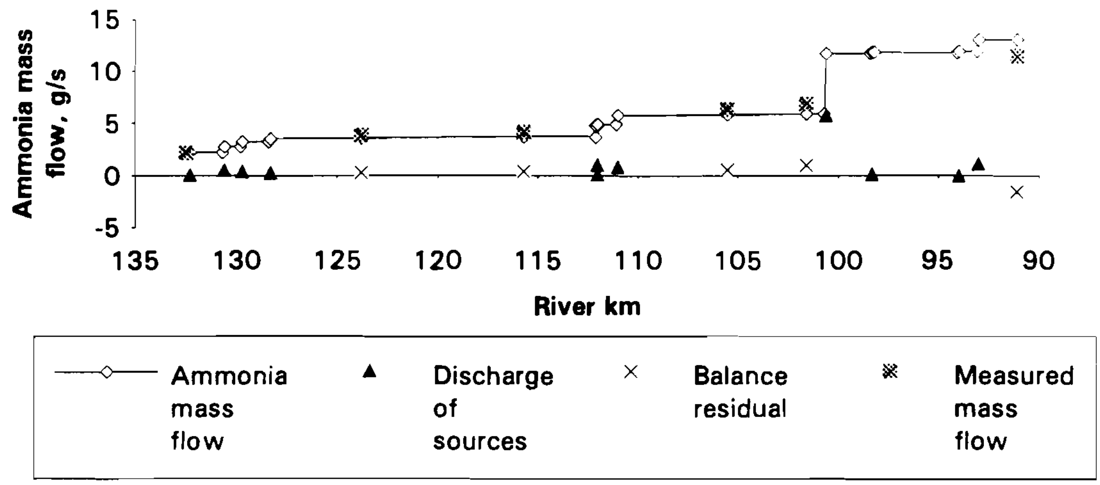


Figure 8.17 Ammonia nitrogen mass balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

The distribution of the load between the emitters is as follows: the tannery factory in Bosany provides 55% of the emitted mass, municipalities account for 30% of pollution and other industrial emitters comprise the rest of the mass.

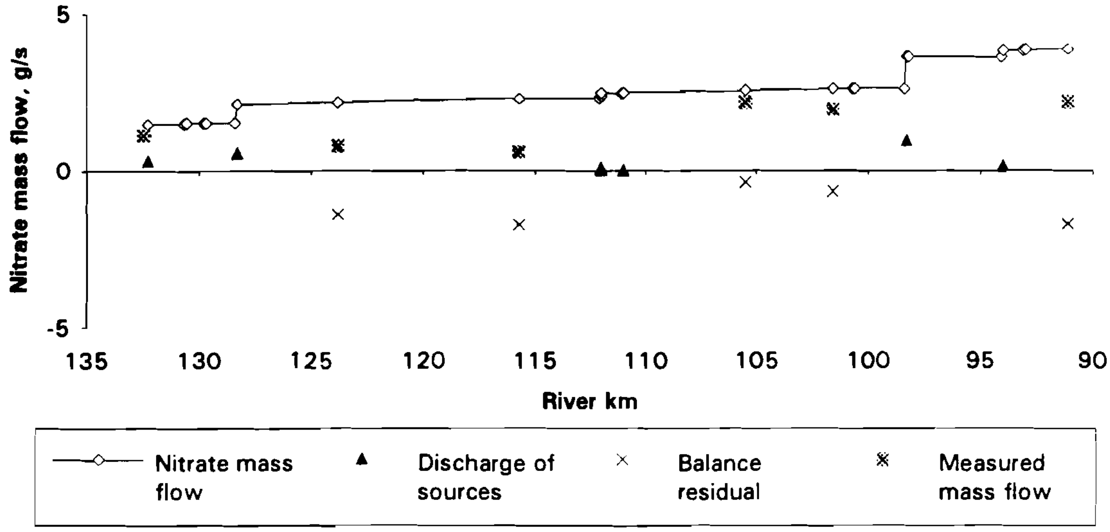


Figure 8.18 Nitrate nitrogen mass balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

The nitrate mass balance is shown in Figure 8.18. The residual analysis suggests that the nitrate nitrogen is removed from the system. Since the dissolved oxygen concentration in the river is low, it is possible that the process of denitrification is taking place, being favored by anaerobic conditions. This process is typical for water polluted with organic materials during low-flow conditions (Water Quality Assessment, 1992). This hypothesis can be further verified if microbiological analysis would reveal the presence of the denitrifying microorganisms in the river and interstitial waters of the sediment. The estimated removal rate of the nitrate nitrogen is about 1.0 1/day.

The total phosphorus mass balance for the upper part of the Nitra River, shown in Figure 8.19, indicates loss of a mass from the system. This loss can be explained by the consumption of reactive phosphorus by the aquatic biota (e.g. phytoplankton) and sedimentation of the particulate phosphorus to the river bed. The ortophosphate phosphorus mass balance, shown in Figure 8.20, indicates a certain loss of reactive phosphorus from the river. The rest of the total phosphorus loss is likely to be explained by the sedimentation process. The estimations for the removal rates (1.3 1/day and 1.7 1/day for the total phosphorus and for the ortophosphate fraction, respectively) are unrealistically high. In this case, a more involved routine should be used to obtain the rate parameters (see Chapter 9).

The distribution of the phosphorus load into the upper part of the river is as follows: 20% from the industrial complex at Novaky, 60% from the tannery in Bosany, and the rest (20%) from municipalities. The principal input of phosphorus into the river is provided by industrial emissions, although municipalities also play a role. It is unclear how much phosphorus is bound within the river sediment, but if this amount is significant, then there will no be immediate effect on the instream phosphorus concentration if emissions are reduced.

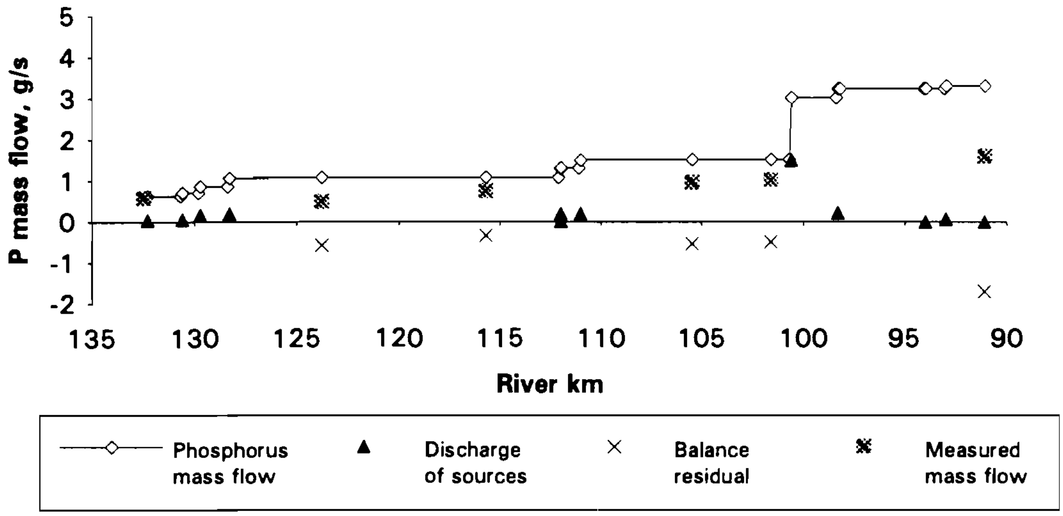


Figure 8.19 Total phosphorus mass balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

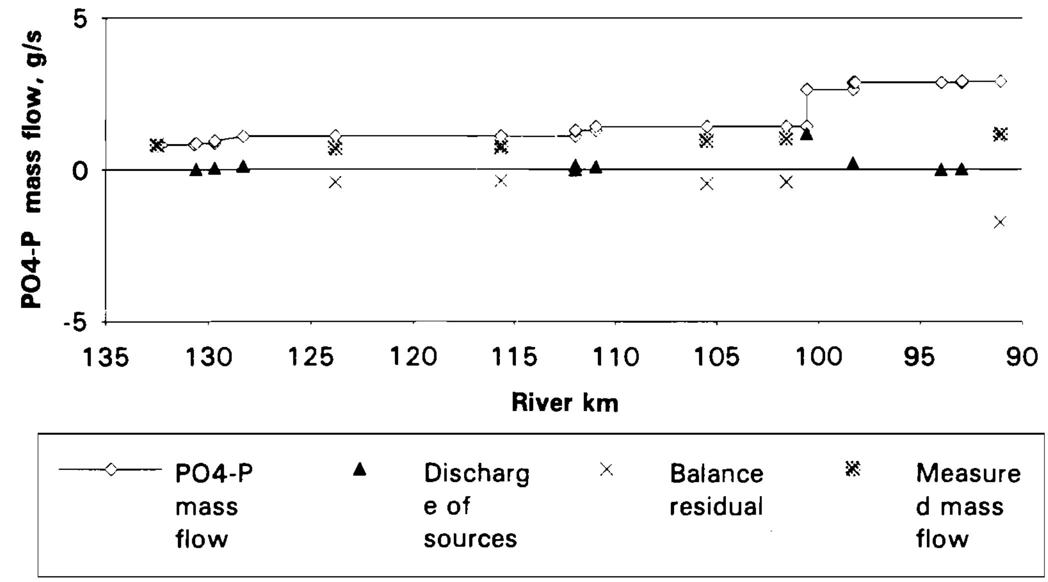


Figure 8.20 Ortophosphate phosphorus mass balance for the upper part of the Nitra River (August 25-26, 1992 experiment)

8.5 Mass Balances for the Lower Part of the Nitra River

The river streamflow mass balance does not show any major discrepancies (Figure 8.21). By observing the chloride-ion mass balance (Figure 8.22) one can verify that the mass balance is closed.

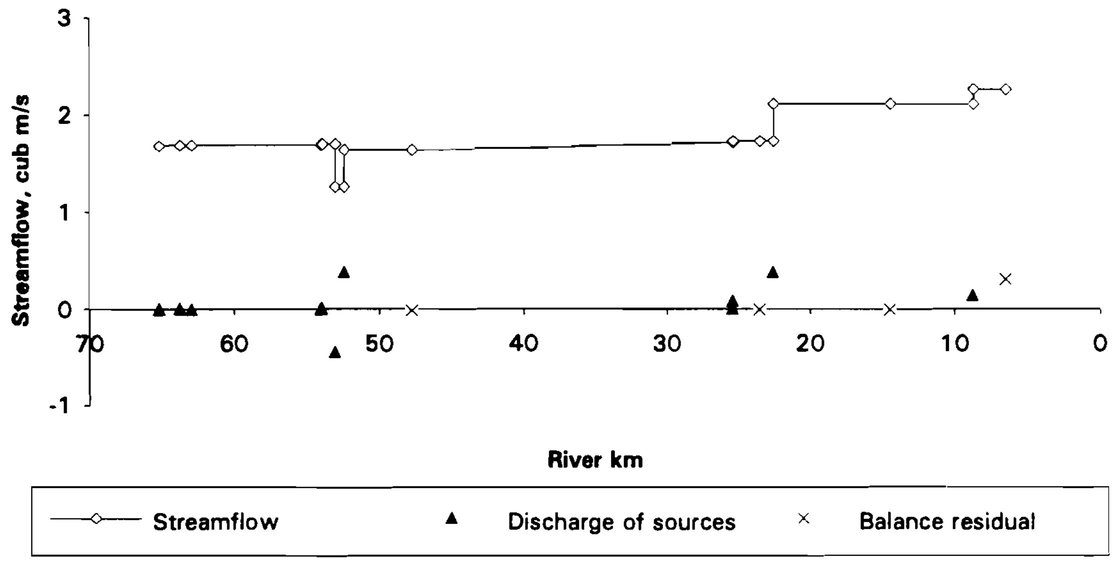


Figure 8.21 Streamflow balance for the lower part of the Nitra River (August 25-26, 1992 experiment)

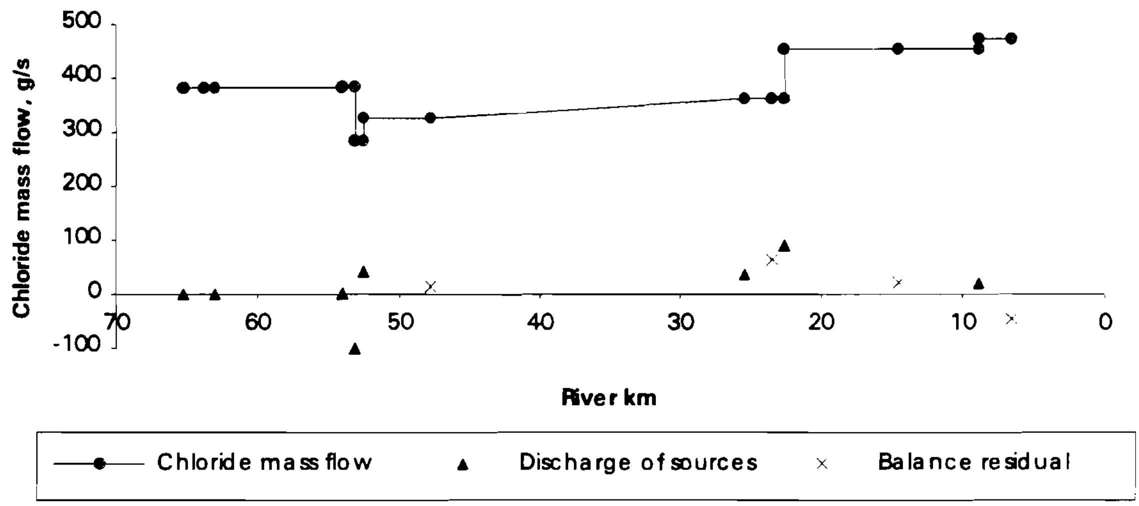


Figure 8.22 Chloride-ion mass balance for the lower part of the Nitra River (August 25-26, 1992 experiment)

The mass balance of BOD-5 is shown in Figure 8.23. Two major emitters are providing nearly all (92%) of the organic material load to the river, namely the Nitra and Nove Zamky sewage treatment plants; the rest is emitted by industrial enterprises. The balance residuals are systematically more and more negative, indicating progressive destruction of the waste material in the river water. Of all the organic material flowing into the river at the lower stretch (flow from the upstream, emissions and tributaries) only 60% reaches the mouth of Nitra River, and 40% is destroyed in the river by heterotrophic bacteria. This indicates high microbiological activity in this river stretch, caused by high levels of pollution. This river stretch, in fact, serves as a secondary treatment facility by stabilizing waste material which was not oxidized during the initial municipal treatment process. This can be also observed by high in-stream BOD-5 concentrations (exceeding 10 mg/l), providing the necessary food source for the heterotrophs. The saprobic index of water in this stretch (measured in June 1993) exceeds

2.5, characterizing an abundance of waste-stabilizing microorganisms. The rough estimation for the removal rate is 1.4 1/day. This is two times more than the value obtained in the Monte-Carlo procedure (Chapter 9). In this case the rough estimation procedure does not provide the correct parameter value.

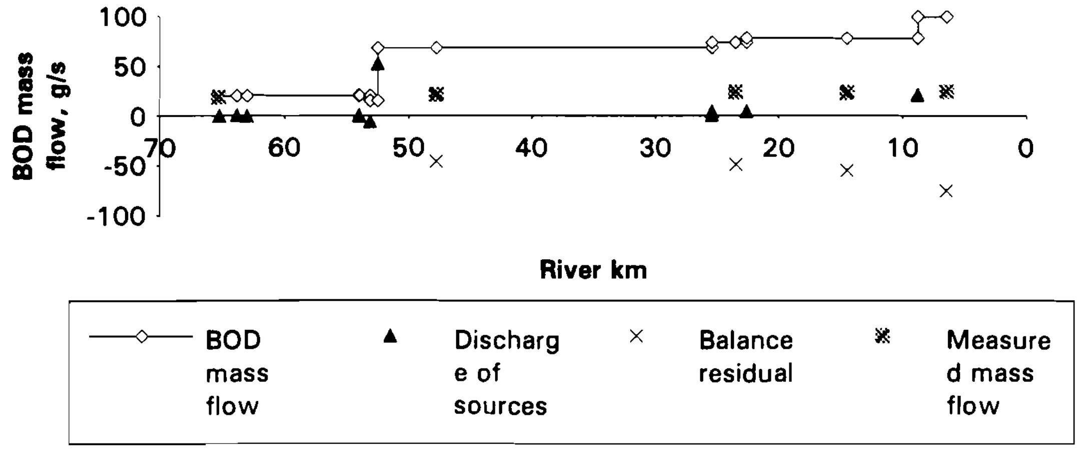


Figure 8.23 BOD-5 mass balance for the lower part of the Nitra River (August 25-26, 1992 experiment)

The ammonia nitrogen mass balance (shown in Figure 8.24) exhibits significant losses in the emitted amount along the course of the river. The estimation for the loss rate is 2.0 1/day. This can signify, again, increased microbiological activity in this stretch, namely proliferation of nitrifiers (Nitrosomonas and Nitrobacter). This conclusion can be further verified by the analysis of nitrate nitrogen (Figure 8.25). The positive mass balance at the Nitrianski Hradok location (river km 25) signals an unaccounted for nitrate source. This source can be related to nitrate release as the result of nitrification process. Further downstream, the nitrate content in the river again diminishes, possibly due to the phytoplanktonic uptakes or denitrification processes.

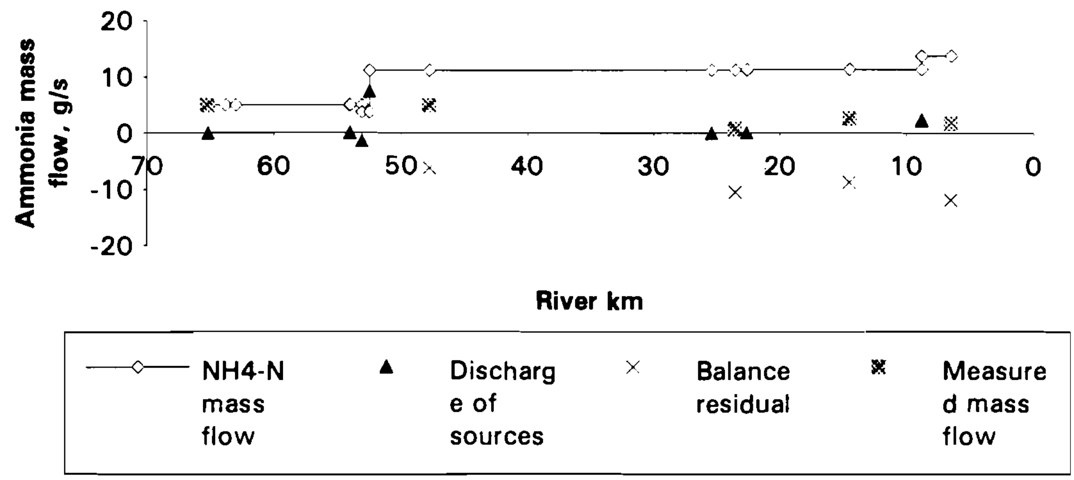


Figure 8.24 Ammonia nitrogen mass balance for the lower part of the Nitra River (August 25-26, 1992 experiment)

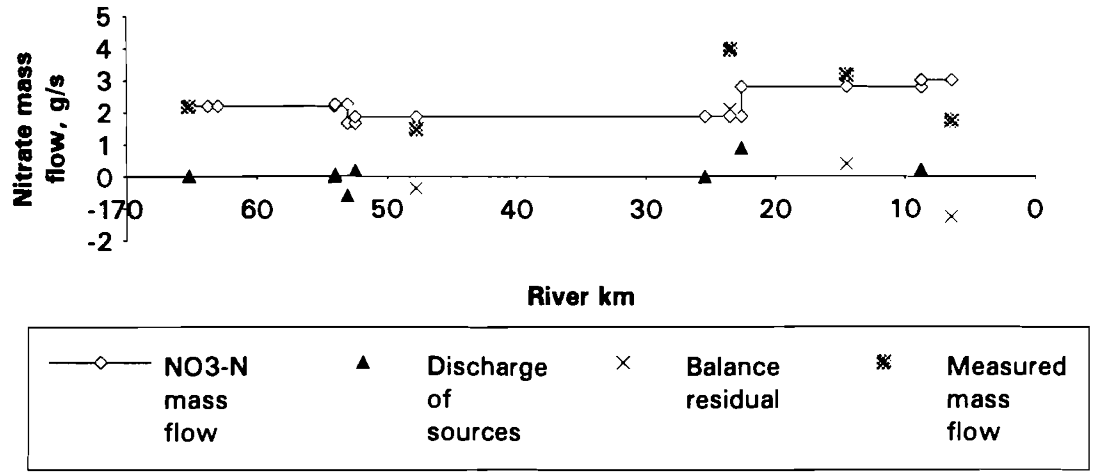


Figure 8.25 Nitrate nitrogen mass balance for the lower part of the Nitra River (August 25-26, 1992 experiment)

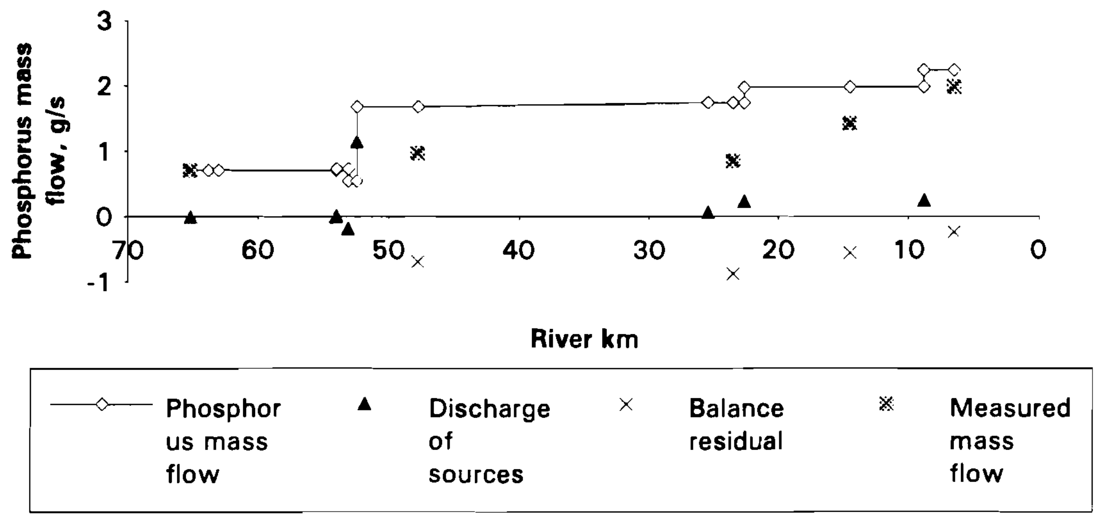


Figure 8.26 Total phosphorus mass balance for the lower part of the Nitra River (August 25-26 1992 experiment)

The total phosphorus mass balance can be analyzed on the basis of Figure 8.26 and the ortophosphate phosphorus mass balance is shown in Figure 8.27. The principal input of phosphorus into the river comes from two municipal treatment plants, Nitra and Nove Zamky, while industry provides about 5% of the load. It can be inferred that the losses of total phosphorus mass flow, as in the case of the upper stretch, are caused by the uptake of aquatic plants and sedimentation. The estimated removal rate is 0.12 1/day. The loss in ortophosphate phosphorus mass from the lower stretch of the river (Figure 8.27; estimated removal rate is 0.7 1/day) can be explained by development of aquatic plants in the lower stretch with slow water currents. Relatively high chlorophyll-a concentrations in the lower stretch (in the range 20-25 mkg/l), obtained from the samples taken in June 1993, further support this conclusion. High diurnal variations of dissolved oxygen concentrations (from 3 to 9 mg/l with afternoon maximum), observed during the August 1992 experiment at Nove Zamky (river km 14.5), also speaks in favor of photosynthetic algae activity in the lower stretch of the Nitra River (Figure 8.28).

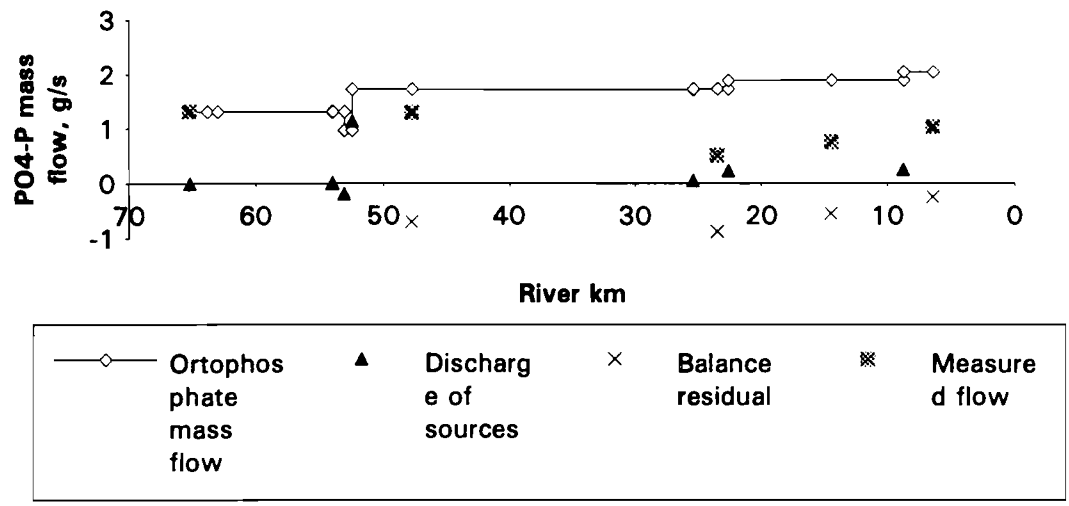


Figure 8.27 Ortophosphate phosphorus mass balance for the lower part of the Nitra River (August 25-26, 1992 experiment)

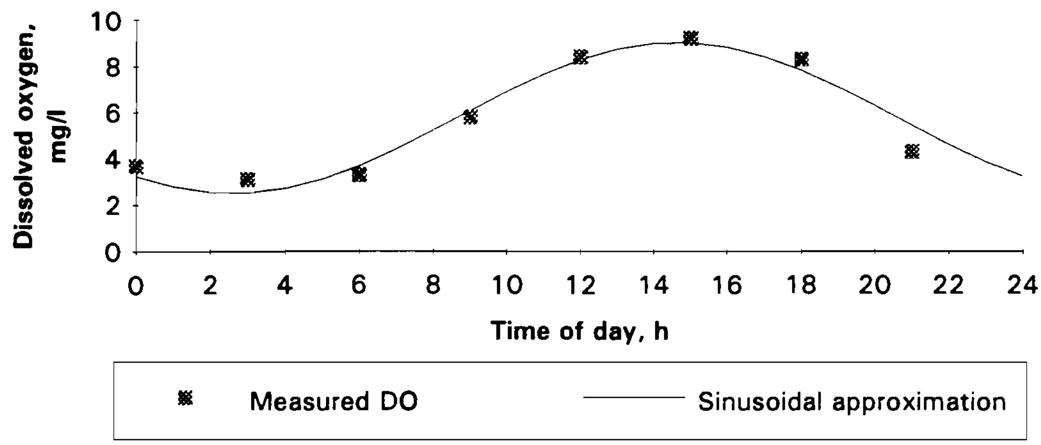
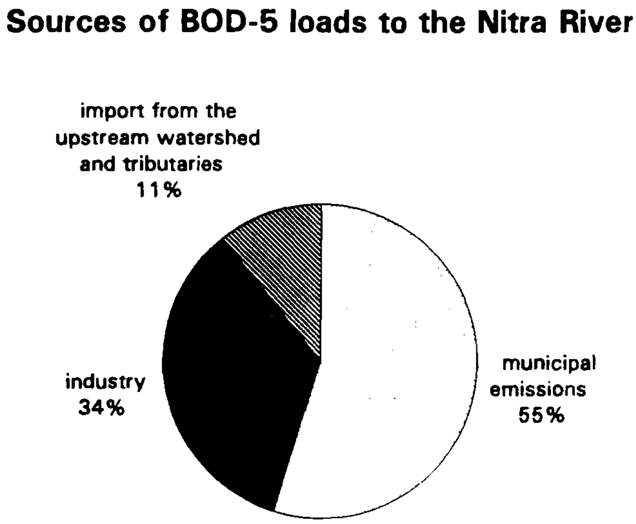


Figure 8.28 Diurnal variations in dissolved oxygen at the measurement point Nitrianska Streda (river km 91.1) during August 25-26, 1992

8.6 Origin of organic pollution in the Nitra River

Lastly, Figure 8.29 and Table 8.1 summarize the balance of organic material in the Nitra River derived on the basis of the data from the August 1992 experiment. It is important to note that the majority of the organic material is discharged into the river by municipal wastewater treatment plants, while industry accounts for 34% of the load. Tributaries and upstream stretches contribute an order of magnitude less. Most part of the emitted organic material (60%) is decomposed in the river by microorganisms, and only 17% of the emitted organics reaches the mouth of Nitra. This indicates that current wastewater treatment does not provide sufficient removal of organic material, and heterotrophic organisms in the river have enough substrate for growth.



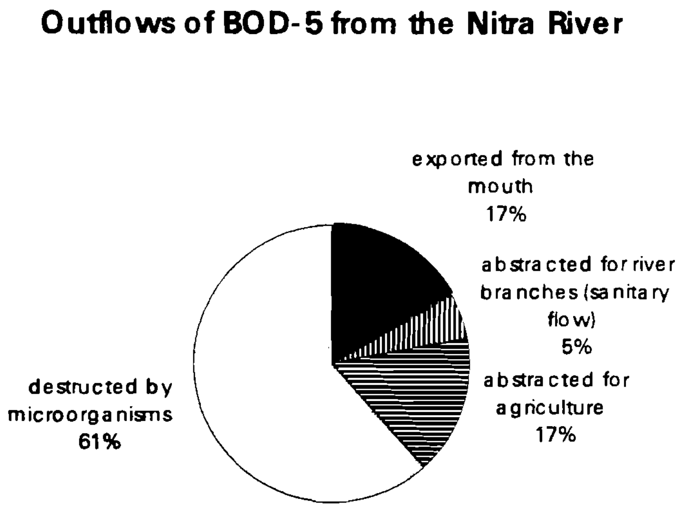


Figure 8.29 The overall balance of BOD-5 mass flows in the Nitra River (August 1992 experiment).

Table 8.1 The overall balance of BOD-5 mass flows in the Nitra River

Load	kg/day	Removal	kg/day
total emitted load	11280	export from the mouth	2134
of that:		removed from the river	10493
municipal emissions	6953	of that:	
industry	4327	abstracted for river branches (sanitary discharge)	646
import from the upstream catchment and tributaries	1348	abstracted for agriculture	2090
		destructed by microorganisms	7757

* assuming the loss of mass between P15 and P16 to be caused by irrigation abstraction

References

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 (IIASA), Working Paper (forthcoming).

9. WATER QUALITY MODELS AND THEIR USE

9.1 Introduction

The development of least-cost water quality management policies requires an analysis of performance under a variety of design conditions and emission inputs. Thus, a model is needed to translate the control action to the ambient quality response (Chapter 5). This translation should take into account the processes which affect the instream concentrations of the constituents under consideration and route the input (emissions and the design natural conditions) to the desired output parameters (the ambient water quality).

The water quality constituents are subject to the following most important processes which may lead to a change in the instream concentrations (see also Somlyódy and Varis, 1992):

- Physical transport (advection and diffusion)
- Chemical and biological reactions

The transport process is caused (mainly) by the motion of water in which the substances are dissolved or suspended. It is therefore necessary to have a description of the flow field (in the case of river flow, it is called river hydraulics).

There is a variety of mathematical models available for the description of water quality phenomena in waterbodies under different conditions (Somlyódy and Varis, 1992) which incorporate descriptions of the key processes. We will focus our attention on the models of rivers and river systems. Because cross-sectional mixing usually is intensive in rivers, one-dimensional (along the stream) mathematical models are most commonly used for description of river flow, advective-diffusive motion, and water quality processes in the river. The temporal changes may be taken into account (dynamic models), or the model can be aimed at the description of the steady-state situation. In both cases, the model is composed of the following equations or the equation sets (Somlyódy and Varis, 1992):

- Equation of water motion (hydraulics)
- Transport equation
- Chemical and biological processes equation or reaction terms extending the transport equation

Selection of a model depends on the water quality problem addressed by the study. Focus of this study (Chapter 7) is organic pollution causing dissolved oxygen deficits, and the model should also simulate the processes affecting the oxygen household of the river. The key parameters related to this problem are oxidizable waste materials (biochemical oxygen demand, ammonia nitrogen, and organic nitrogen), dissolved oxygen, and to a lesser degree, phosphorus and nitrogen, which control algae proliferation (Thomann and Mueller, 1987). The number of the water quality constituents considered by an oxygen model can range from 2-3 (simple BOD-DO oxygen models) to 20-30 (comprehensive ecosystem models). The reported work will be based on simple dissolved oxygen models with two or three components. The application of a more complex stream water quality model with about ten components (QUAL2) will also be discussed (for more detail , see Breithaupt and Somlyódy, 1994).

Modeling procedure involves the following common steps (Beck, 1983): selection of the appropriate model, parameter estimation (calibration), validation and the simulation under the desired conditions (scenario analysis). In this chapter, we will cover the model selection, calibration and validation issues. The set of data used for the model calibration was described in Chapter 8. First, the hydraulic model will be outlined (Section 9.2). Then, simple dissolved oxygen models and their calibration will be covered. The application of a more sophisticated model QUAL2E will be discussed later (Section 9.6). The "final" selected model (Section 9.5) will be used for control policy development (Chapters 10 and 13). Figure 9.1 illustrates use of water quality models in the framework of water quality management and planning.

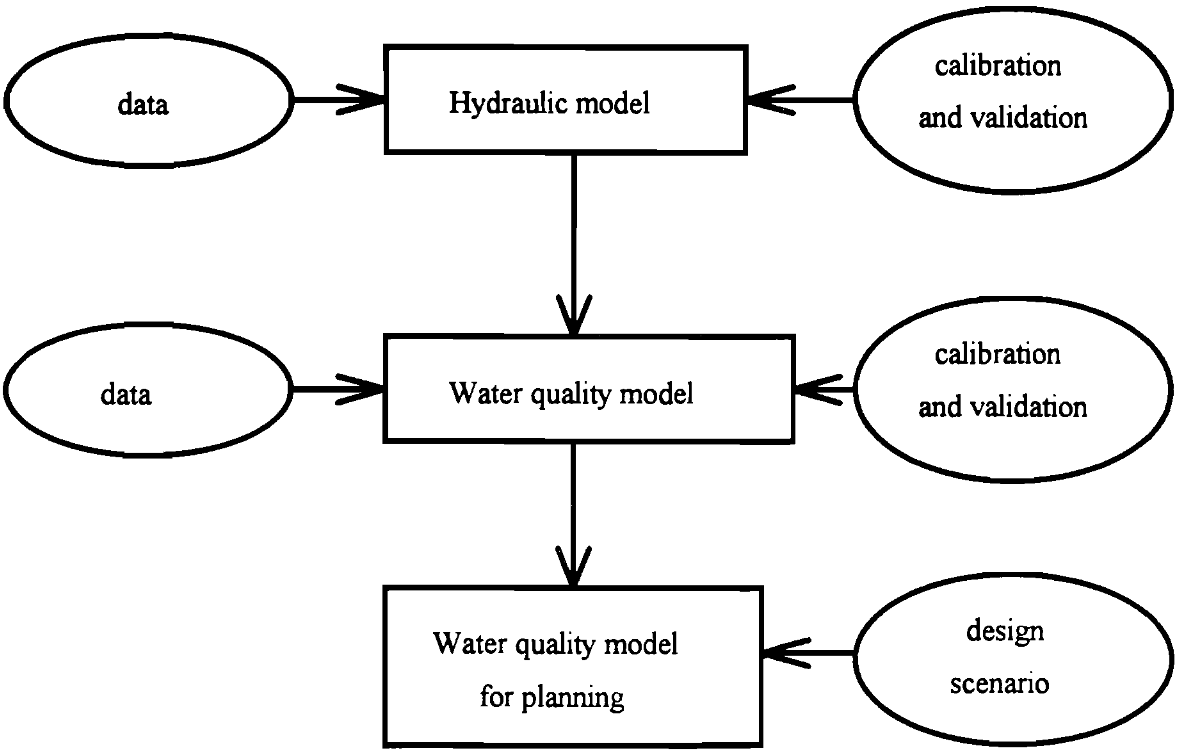


Figure 9.1 Use of water quality models in the policy analysis framework.

9.2 Hydraulic Models and Their Calibration for the Nitra River

Water motion along a river or channel is often described by the set of one-dimensional equations of continuity and momentum, known also as Saint-Venant equations (Mahmood and Yevjevich, 1975):

$$\frac{\partial A}{\partial t}+\frac{\partial Q}{\partial x}=q(x,t) \tag{1}$$

$$\frac{\partial Q}{\partial t}+\frac{\partial}{\partial x}\left[\frac{Q^2}{A}\right]+gA\frac{\partial z}{\partial x}+\frac{A}{rh}\tau_{bx}-\frac{A}{rh}\tau_{sx}=0 \tag{2}$$

where:

x is the coordinate along the river or channel, Q is the stream flow rate,

z is the elevation of water surface,

h is the water depth,

ρ is the water density,

A is the cross sectional area,

q is the rate of the lateral inflow per unit of river length,

g is the gravity constant (9.81 m/sec^2),

τ_{bx} is the bottom shear stress, and

τ_{sx} is the surface shear stress.

The surface shear (or wind) stress τ_{sx} is rarely significant for the river flow, which is being driven mainly by the gravitational forces. If the quadratic law of resistance is applied, one can write for the bottom shear τ_{bx}:

$$\tau_{bx} = \frac{C_f \rho}{A^2} Q|Q| \tag{3}$$

For the resistance coefficient C_f a number of empirical formulations are available (Somlyódy and Varis, 1992). The Manning's equation for the resistance coefficient was used in the model:

$$C_f = \frac{n^2 g}{h^{1/3}} \tag{4}$$

where n is the Manning's roughness coefficient.

For the water quality problems, the local fast movements of the fluid are generally not essential (Somlyódy and Varis, 1992). In this case, the first two terms in the Eq. (2) could be disregarded, and a diffusive wave approximation equation can be derived from Eqs. (1) and (2) (Mahmood and Yevjevich, 1975). However, the diffusive wave approximation requires a lot of input data as initial and boundary conditions, and simulation time is rather long. The diffusion wave approximation is justified when the temporal changes in the river hydraulic parameters (streamflow and water depth) are significant over the time frame of interest. When the main objective is water quality, the steady-state approximation often is acceptable (see Chapter 5 and Somlyódy and Varis, 1992). The system of hydraulic equations for the steady state can be presented as follows:

$$\frac{\partial Q}{\partial x} = q(x,t) \tag{5}$$

$$\frac{\partial z}{\partial x} = -\frac{Q|Q|}{K^2} \tag{6}$$

and, for simplified evaluations, the following equation can be used instead of Eq. (6):

$$i_0 = \frac{Q|Q|}{K^2} \tag{7}$$

where i_0 is the local slope of the river bottom. For simulating the hydraulics of the Nitra River, the last model was used .

The morphometry data were available in the form of 299 cross-section profiles covering the major part of the Nitra River flow (from river km 155 to the mouth). The profiles are located at an average 0.5 km from each other, forming a comprehensive representation of the river morphometry. One of the typical cross-section profiles (at the river km 21.8) is shown in the Figure 9.2.

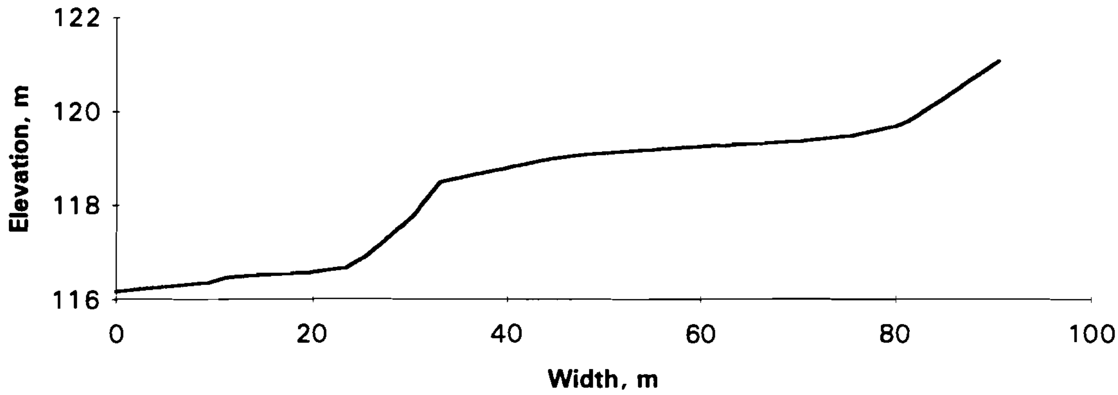


Figure 9.2 Cross-section profile at the river km 21.8 (Bánov)

The calibration of the hydraulic model was based on the rating curves (elevation-streamflow curves) available at seven locations. Both the Manning's roughness coefficient and the bottom elevation were calibrated to fit the elevation and stream flow data. The fitting was facilitated with the minimization procedure for the following function:

$$\Phi = \frac{1}{N}\sum_{i=1}^{i=N}\left|z_i^{obs} - z_i^{calc}\right| \tag{8}$$

where N is the number of data points in the rating curve,

z_i^{obs} is measured elevation,

z_i^{calc} is calculated elevation.

The elevation was calculated from the local bottom slope with Eq. (7). The function Φ was minimized with Powell's method, using z_{bot} and n as the parameters to fit. Keeping in mind that the main purpose of the model is to describe low-flow conditions, only the parts of the rating curves corresponding to a river depth 0-2 m were used for calibration. The calibration results for the Bánov location are shown in Figure 9.3 for illustration.

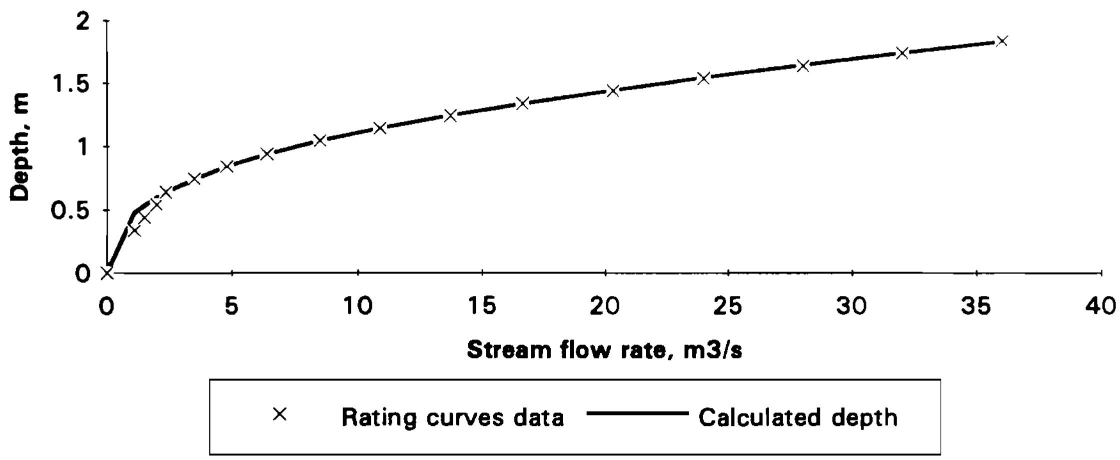


Figure 9.3 Calibration of the hydraulic model for the Bánov location (river km 21.8)

To verify the calibration procedure, the steady-state hydraulic model (Eqs. (5) and (6)) was integrated keeping the streamflow rate constant along the river, and the results were compared with the rating curves' data points (Figs. 9.4-9.5). The results show certain irregularities in the water depth profile, caused by changes in the river morphometry. The rating curves' data points can be considered to adequately agree with the modeled depth profile, taking into account the mentioned irregularities.

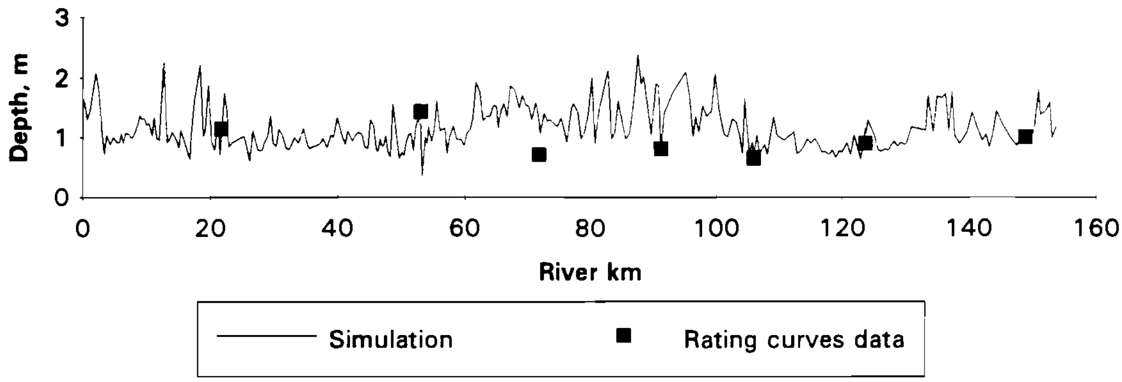


Figure 9.4 Steady state hydraulic model of the Nitra River and the rating curves data. Streamflow rate is 10 m^3/s along the river

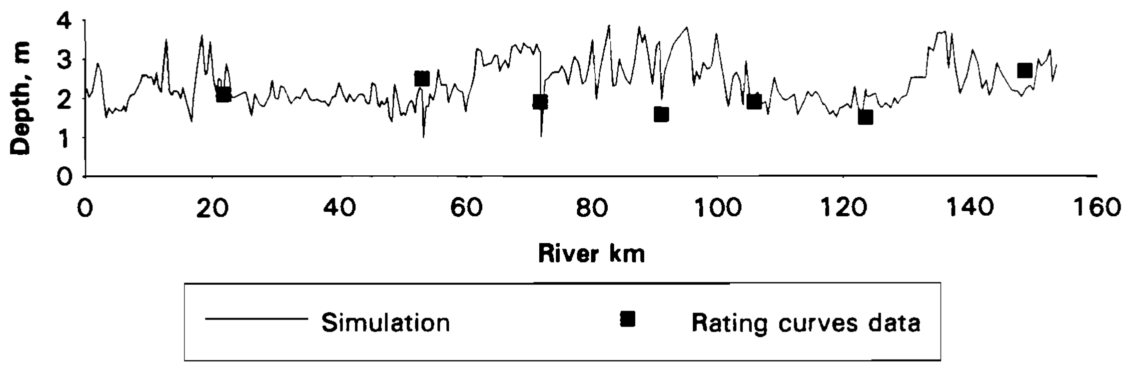


Figure 9.5 Steady state hydraulic model of the Nitra River and the rating curves data. Streamflow rate 50 m^3/s along the river.

9.3 Simple Dissolved Oxygen Models

The formulation of river water quality models of this class is rather conventional. The number of state variables representing the household of dissolved oxygen ranges between one and three, while the number of parameters varies between one and five. More specifically, the applied models include (see Thomann and Mueller, 1987):

- The original DO-BOD Streeter-Phelps model (two parameters)
- The same model with the incorporation of sedimentation of the particulate organic material (three parameters)
- As above, but with sediment oxygen demand (four parameters)
- A three state variable model with nitrogenous BOD (five parameters)

The set of partial differential equations for the three state variable/five parameter model can be written as follows (see assumptions in Somlyódy and Varis, 1992):

$$\frac{\partial(A\,L)}{\partial t}+\frac{\partial(QL)}{\partial x}=-K_r\ A\ L \tag{9}$$

$$\frac{\partial(A\,N)}{\partial t}+\frac{\partial(Q\,N)}{\partial x}=-K_n\ A\ N \tag{10}$$

$$\frac{\partial(A\,C)}{\partial t}+\frac{\partial(Q\,C)}{\partial x}=k_a B(C_s-C)-K_d\ A\ L-K_n\ A\ N-BK_{SOD} \tag{11}$$

Where: L- carbonaceous biological oxygen demand (CBOD) in mg/l
N - nitrogenous biological oxygen demand (NBOD) in mg/l
C - dissolved oxygen concentration in mg/l
x - coordinate along the river, m;
t - travel time in days
Q- streamflow in m^3/d
A - cross-section area in m^2
B - stream width in m
K_r - carbonaceous BOD removal rate in 1/d
k_a - oxygen exchange coefficient (see later) in m/d
K_d - CBOD oxygenation rate in 1/d
K_r - CBOD decay rate in 1/d
K_n - NBOD oxygenation rate in 1/d
K_{SOD} - sediment oxygen demand in $g/m^2/d$
C_s - saturation concentration of dissolved oxygen in mg/l

The exchange coefficient across the water-atmosphere boundary, k_a was calculated using the O'Connor and Dobbins (1956) empirical relationship:

$$k_a = k_{a0}\ f(T)\ \sqrt{(U/H)} \tag{12}$$

where: k_{a0} - the reaeration coefficient in m $s^{1/2}/d$
f(T) - dimensionless temperature correction factor
U - flow velocity expressed in m/s
H - aeration depth in m, defined as the A/B ratio

The reaeration rate K_a (1/d) is defined as k_a/H and is dependent on the flow and stream morphometry at the current location.

For every river stretch with relatively uniform characteristics, Equations (9)-(11) are solved analytically, from the most upstream location to the river's mouth. At confluence points. an assumption of immediate, complete mixing is utilized. The computation time is small enough to allow the model to be incorporated into relatively sophisticated parameter estimation and policy analysis frameworks (Figure 9.1; see also Section 9.4 and Chapter 11, respectively).

9.4 Model Calibration Using Data from the Longitudinal Water Quality Profiles

The selection of proper parameter values for water quality models is a crucial step in developing a catchment-wide control policy. For the parameter estimation, the results of two experiments were used to produce longitudinal profiles and estimate mass balances (Chapter 8).

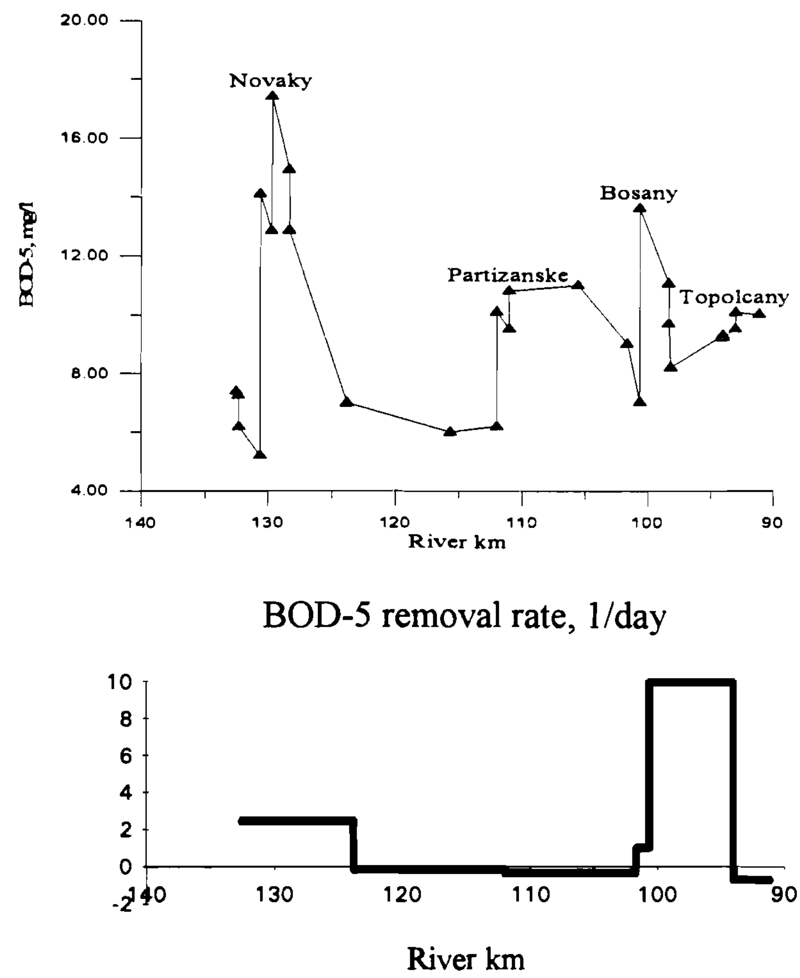


Figure 9.6 Longitudinal profile and the evaluation of the removal rate of BOD-5 for the upper part of the Nitra River.

The conventional methodology is based upon the analysis of longitudinal profiles and mass balances on a reach-by-reach basis (see e.g. Technical Guidance, 1983). The loss of mass in a given reach should be explained by the process(es) assumed (e.g. first order decay for selfpurification). Knowing the travel time for a given reach, it is a seemingly straightforward procedure to come up with the value of the relevant parameter (e.g. the decay rate). However, due to large uncertainties and scarcity of data and to simplifications of the model, this procedure often produces misleading and even confusing results.

Figure 9.6 illustrates typical difficulties arising in the implementation of this procedure for the upper 50 km long reach of the Nitra River. As can be seen, the BOD-5 removal rate fluctuates within an unrealistically broad domain, differing significantly from the values reported in the literature (the plot shows even negative estimates). Since this part of the river is comparatively uniform with regard to morphometry, bed composition, slope, etc., these hypothetical, abrupt changes in the microbiological activity cannot be caused by some changes in external conditions.

As the evaluation of the longitudinal profiles clearly shows, there is a need for the application of more advanced methodologies which are able to properly handle the inherent uncertainties of the system.

A number of methods were developed for parameter estimation and uncertainty analysis in water quality modeling (e.g. Beck, 1979a; Beck and van Straten, 1983, Beck, 1987). Most of the methods use minimization procedures with corresponding loss functions such as the least-squares method (Beck, 1979a). The family of recursive methods such as Kalman filter were also applied for parameter estimation in water quality modeling (examples include Beck, 1979b; Rinaldi et al., 1979). These methods came up with a single "best" set of parameter values which was then used for forecasting the system's behavior. It has been recently understood that, given all the uncertainties mentioned above, our ability to uniquely estimate the key model parameters can be questioned in many cases. Hornberger and Spear (1980) developed the Monte Carlo methodology based on the so-called behavior definition which derives sets or ensembles of parameter vectors rather than a single "best" value. Lately, this approach was extended (Fedra et al, 1981) to allow those ensembles to be treated as samples from a probability distribution to be used in a stochastic fashion for the forecasting purposes. This methodology, referred to as the Hornberger, Spear and Young (HSY) approach (similar to Beck (1987)), was used as the basic parameter estimation tool for this study.

The idea of this method is to look for an ensemble of parameter vectors rather than a unique, "best" one. The vectors are accepted or rejected based on knowledge about the system's performance, or "behavior definition" (e.g. lower and upper bounds of state variables). This knowledge can be vague, allowing large uncertainties to be explicitly incorporated in the calibration procedure. Such a method is in harmony with the scarcity of generally available water quality data and the relatively poor understanding of the processes affecting water quality. All of the parameter vectors (elements of which are selected randomly from pre-specified uniform distributions of initial domains of feasibility) by which the model simulations were compliant with the defined behavior are considered acceptable and can be used later for forecasting of future system responses. Other probabilistic methods such as Bayesian estimation of probability distribution in the parameter space also can account for the uncertainties in the modeling process (see Masliev and Somlyódy, 1993).

The probability distribution obtained will be applied for a policy-oriented risk analysis (Chapter 11) similar to (Fedra et al, 1981). However, some caution is required in the interpretation of the results, because the parameter set obtained depends on the specification of the *a priori* distribution in the parameter space and the behavior definition alike.

The implementation of the HSY approach, in this case, logically breaks down into two distinct tasks: generating the samples in the parameter space and screening resulting simulations. In the sampling unit, both the emission data and the river water quality observations are disturbed by a random component with the normal distribution and a zero mean. This should effectively model our uncertainty. The extent of this disturbance is controlled by the analyst (model user) according to his/her experience and intuition. Samples of the model parameters are uniformly distributed within user-defined bounds, which can be deduced from the literature. Software implementation of the calibration routine is described in Chapter 12.

In the screening unit, as noted before, only those scenarios are selected which correspond to the "behavior definition" specified by the analyst. The extensive set of parameter values can be treated as a sample having a certain probability distribution (with the provisions mentioned above).

There can be several application strategies for the HSY approach to the Streeter-Phelps model. One of them is to estimate parameters in a sequential fashion, utilizing the fact that Eq. (9) can be solved independently from the rest of the system. Thus, we can estimate the BOD removal rate first and then use the average of the resulting sample for substituting into Eq. (11). The second possibility is to sample from the two-dimensional parameter space (BOD removal rate, K_d, and the reaeration coefficient, k_{a0}). The third possibility is to fix the ratio of the two parameters and to estimate one of them (or the opposite way around). In this study, we apply the first two methods.

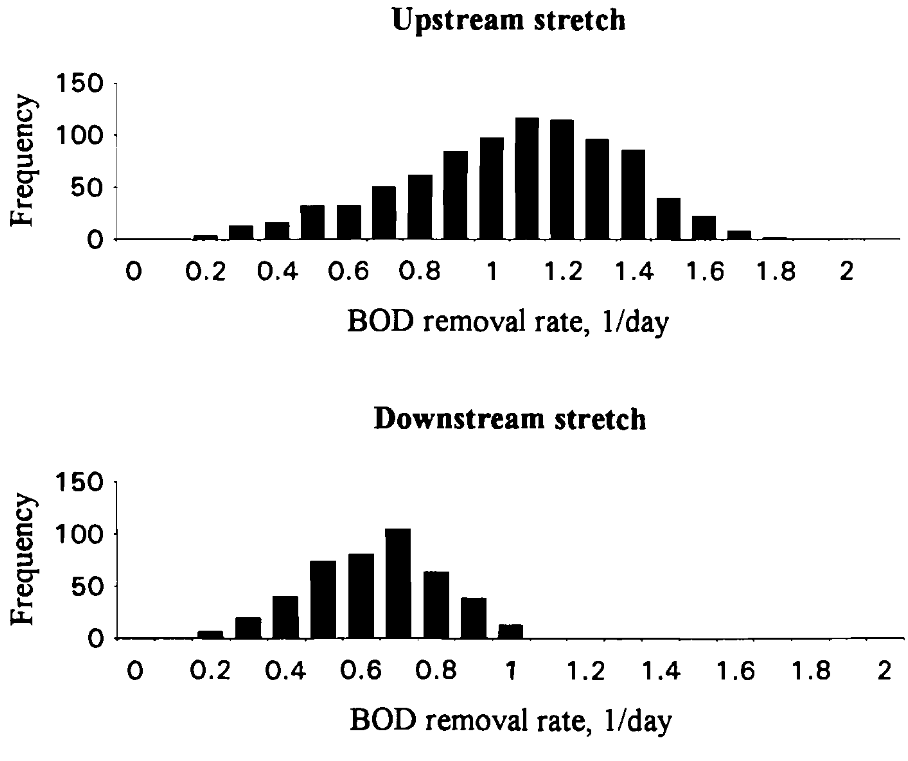


Figure 9.7 Frequency plots for the BOD removal rate in the upstream and downstream stretches of the Nitra River obtained with HSY technique (August 1992 experiment)

For estimation of the BOD removal rate, the coefficients of variance for the BOD measurements in the mainstream and emissions were set to 30%, reflecting our degree of (un)certainty. The "filtering window" was set to 30% of the mean value as well. The resulting histogram for BOD removal rate is plotted in Figure 9.7 for the upper and lower stretches of the river.

The probability distributions of the BOD removal rate for the upper and lower reaches have means of 1.1 1/day and 0.7 1/day, respectively. These values appear to be higher than the usually assumed 0.2-0.3 range for biologically treated wastewater effluents (Thomann and Mueller, 1987). The explanation can be twofold. One of the reasons is the presence of highly overloaded municipal sewage treatment plants (often by 100% or more), resulting in partial treatment only. Additionally, systematic analysis of BOD removal rates performed by O'Connor and others (Technical Guidance, 1983) points to a relationship between stream morphology (water depth in particular) and the rate of microbiological utilization of the organic waste. Shallow streams tend to have higher BOD removal rates due to better contact of water with the microbiota attached to the bottom. Further, one can notice that this correlation also explains the difference between BOD removal rates in the upper and lower stretches of the Nitra River; the lower stretch has a smaller slope and the river is approximately twice as deep as upstream. The subsequent discussion is focused on the upper part of the river. The findings are similar for both upstream and downstream stretches, the latter case serving validation purposes.

The HSY method in a sequential fashion was used first. For the original Streeter-Phelps model (assuming $K_r = K_d$ in Eqs (1) and (3)), the parameter k_{a0} (see Eq. (12)) was calibrated with the BOD removal rate set to the previously found value of 1.1 1/day. The mean value of the reaeration coefficient was found to be 0.37 (corresponding to a reaeration rate of 1.9 1/day at the Chalmova measurement point, river km 123.9), which is less than some literature values (Thomann and Mueller, 1987). The ratio of the two parameters is 1.7. The underestimation of reaeration coefficient suggests that some dissolved oxygen sinks were overlooked.

The incorporation of the sediment oxygen demand of 1.5 $g/m^2/d$ into the oxygen balance equation led to a mean reaeration coefficient of 0.63. Finally, for the model with nitrogenous BOD and sediment oxygen demand, a mean value of 0.77 for the reaeration coefficient was estimated (see Figure 9.8, Table 9.1).

Table 9.1 Summary of mean parameter estimates for the upper stretch of the Nitra River (HSY method, sequential calibration)

Mean parameter values	K_r, 1/day	k_{a0} -	K_n, 1/day	K_{SOD}, $g/m^2/day$
Streeter-Phelps model	1.1	0.37	-	-
Model with SOD	1.1	0.63	-	1.5
Model with NBOD	1.1	0.48	0.24	-
Model with SOD and NBOD	1.1	0.77	0.24	1.5

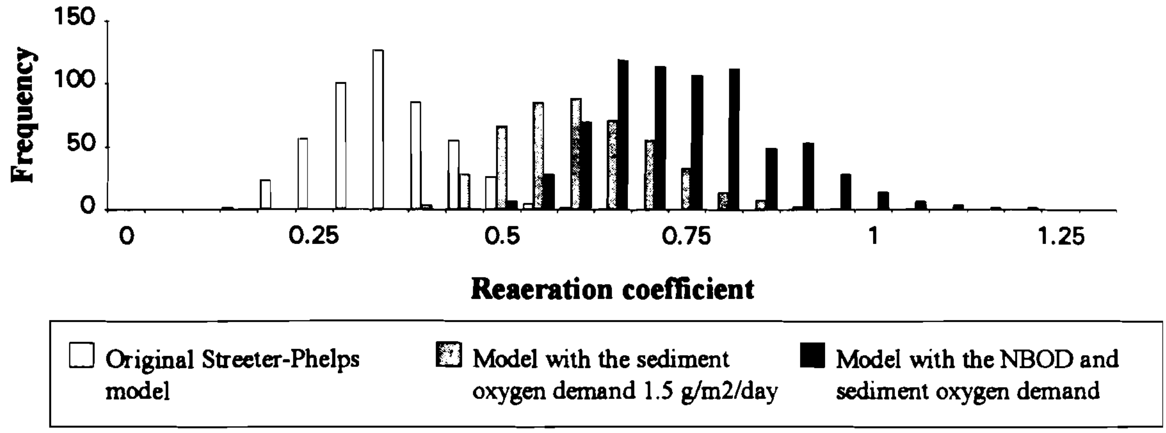


Figure 9.8 Frequency plots for the reaeration coefficient of the three DO water quality models (upper part of the Nitra River).

If the BOD removal rate and reaeration coefficient are estimated simultaneously rather than in a sequence, the margins of acceptance criteria should be somewhat broadened (because of the increase in the dimensionality of the event space). This analysis for the original Streeter-Phelps model was performed with acceptance boundaries set to 50% of the measured DO and BOD concentrations. This procedure leads to a small change in the mean reaeration coefficient and mean BOD removal rate.

From the scatterplot in Figure 9.9, one can see that the estimates for the BOD removal rate and reaeration coefficient are correlated, which is in harmony with the structure of Eq. (3). A linear regression explained this dependency reasonably well (the R^2 value is about 0.6). The K_a/K_d ratio for Chalmova location is 2.1; this is well within the literature's bounds for rivers (e.g. see Jolánkai, 1992).

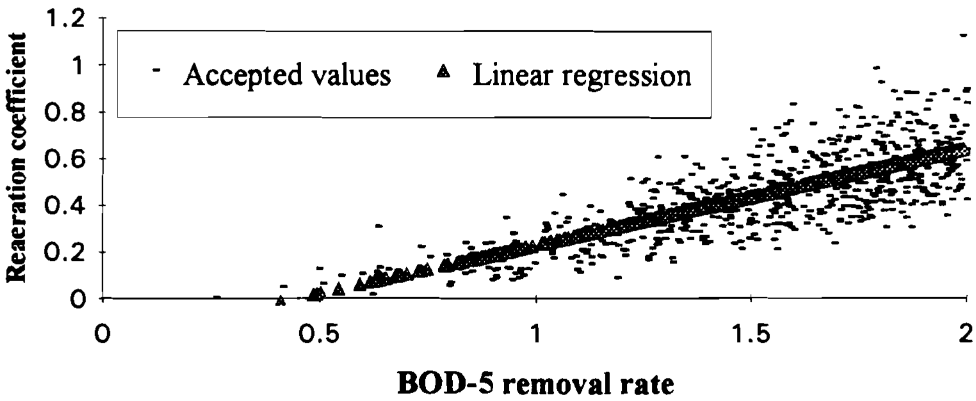


Figure 9.9 Correlation of the two major parameters from the joint Monte Carlo estimation (upper part of the Nitra River)

The four calibrated dissolved oxygen models were used to simulate water quality in the Nitra River. All of them produced dissolved oxygen profiles which fit the measured data reasonably well (Figure 9.10).

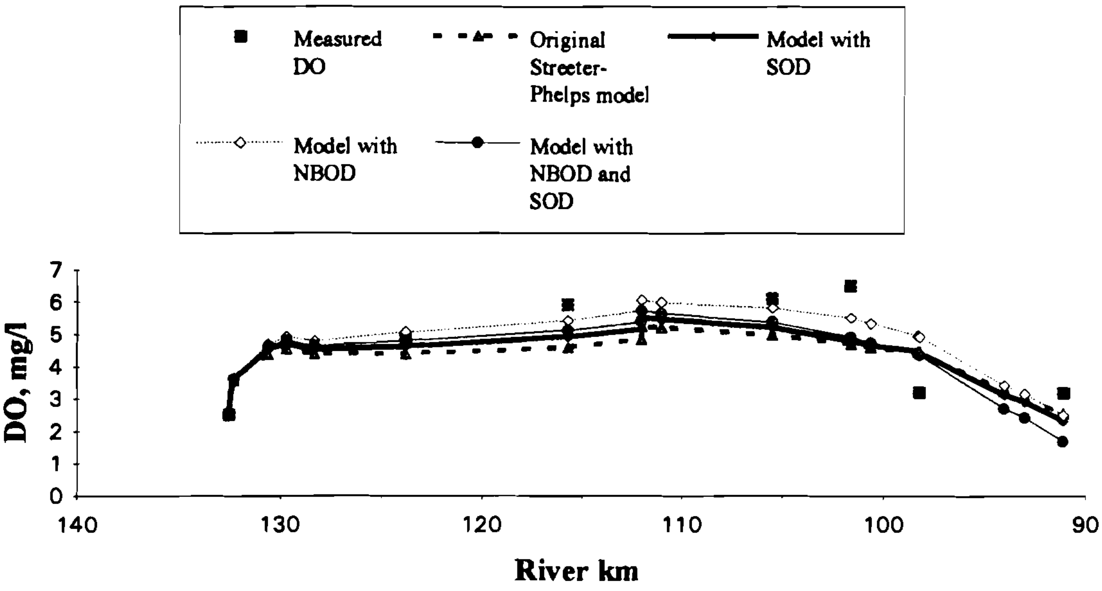


Figure 9.10 Simulation of DO for the upper stretch of the Nitra River by a sequence of calibrated models

Similar parameter estimation procedure was repeated for the lower stretch of the Nitra River, details of which will not be covered here. The results of estimation are summarized in Table 9.2 (see also Figure 9.14).

It can be seen that the estimation for the BOD removal rate is somewhat lower than for the upper part. This can be related to an increase in water depth in the lower part of the river (approximately 50% due to the changes in the streamflow). For the Streeter-Phelps model, the reaeration rate is only slightly higher than for the upper part (0.4 and 0.6, respectively). However, for the 3-component model the estimation is significantly higher, which shows more significant role of ammonia nitrogen in the oxygen household.

Table 9.2 Summary of mean parameter estimates for the lower stretch of the Nitra River (HSY method, sequential calibration)

Mean parameter values	K_r, 1/day	k_{a0} -	K_n, 1/day	K_{SOD}, g/m^2/day
Streeter-Phelps model	0.7	0.57	-	-
Model with NBOD	0.7	2.48	0.8	-

For the purpose of validation in time, parameters of the Streeter-Phelps model were estimated on the basis of the June 1993 experiment (Chapter 8). The data for the upstream stretch of the Nitra River (Novaky - Partizanske) were processed with the help of the HSY approach outlined above.

The frequency plot for the BOD removal rate is shown in Figure 9.11. The mean value is 0.5 1/day, less than the value obtained during the evaluation of the August experiment (1.1 1/day). One of the reasons for this difference could be the significant increase in the streamflow (1.4 m^3/s compared to 0.6 m^3/s in August). The larger streamflow implies increase in the water depth. Therefore, benthic microorganisms can less easily consume organic matter the river water (see above), and the BOD removal rate decreases.

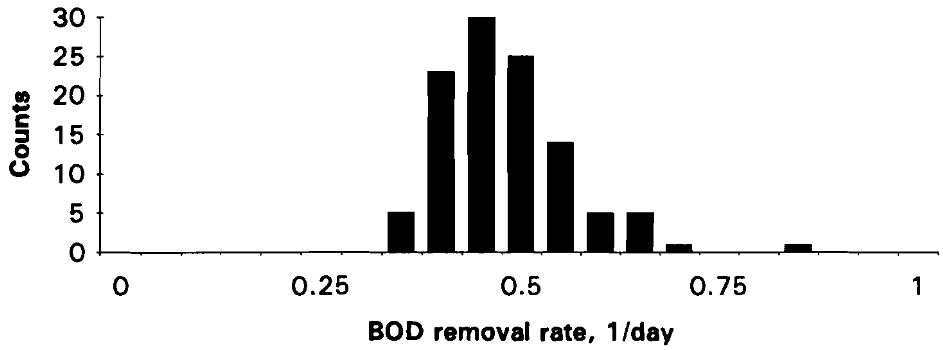


Figure 9.11 Frequency plot for the BOD-5 removal rate estimate for the upper stretch of the Nitra River (June 1993 experiment)

The reaeration coefficient for the Streeter-Phelps model was evaluated using the sequential approach, i.e. setting the BOD removal rate to 0.5 1/day. The resulting frequency plot of the estimation for the reaeration coefficient is shown in figure 9.12. The August 1992 estimation is also shown on the same plot for comparison. The shapes of the two distributions look rather similar. The mean value of the reaeration coefficient is 0.41 (the estimation for the August 1992 dataset is 0.37). Overall, it looks that the two estimates reasonably pass the estimation-validation test.

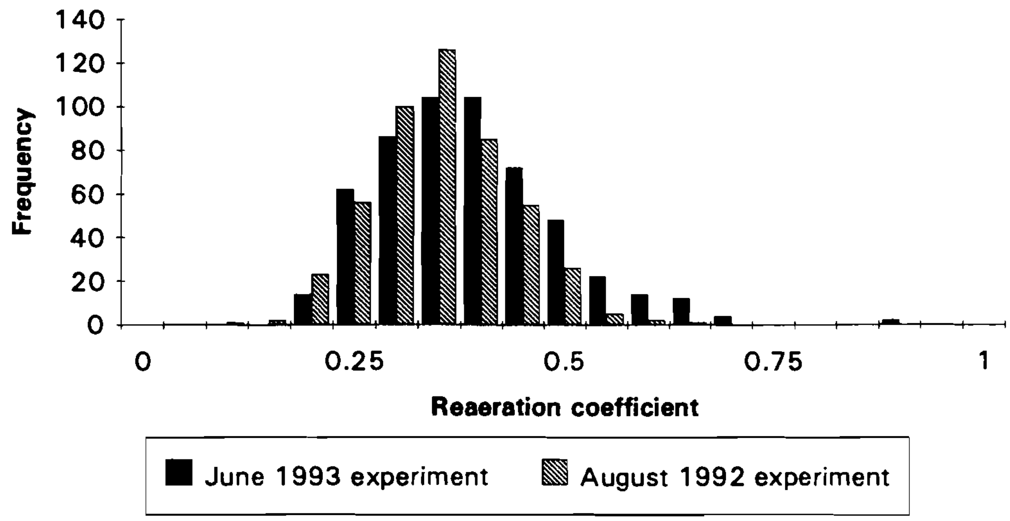


Figure 9.12 Frequency plots for the reaeration coefficient estimation for the upper stretch of the Nitra River. The June 1993 and August 1992 experiments

9.5 The Dissolved Oxygen Model Selected for the Policy Analysis

On the basis of both analysis of the coefficient distributions for both stretches and comparison with the literature ranges, the following model and parameter values were selected for the policy-oriented studies:

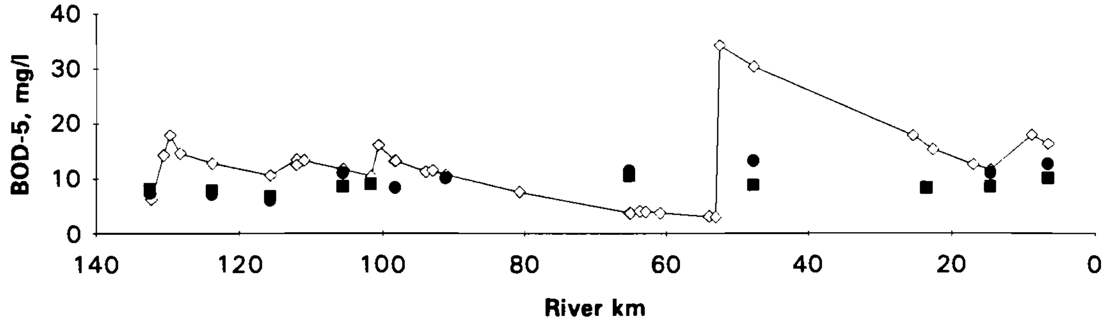
Model:

- Three-component model dissolved oxygen with carbonaceous and nitrogenous oxygen demand.

Parameter values:

- BOD decay rate is 0.8 1/d
- Ammonia decay rate is 0.8 1/d
- Reaeration coefficient is 2.0 m $s^{1/2}$/d

Using the selected model under the calibration conditions (August 1992), a deterministic simulation was performed. The simulation results are shown in Figure 9.13 together with the experiment and regular measurements for the respective water quality parameter. The discrepancies are high in all three plots, but the general character of the longitudinal water quality changes is reflected realistically enough (Figure 9.13).
Finally, Figure 9.14 shows the probability distributions to be used in the uncertainty framework for the policy purposes derived from the 3-component model calibration for the lower stretch of the Nitra River.



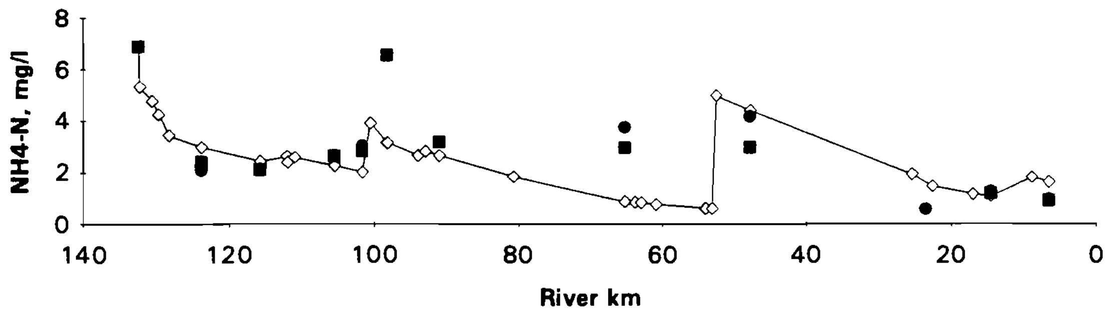


Figure 9.13 A comparison of the selected model with measurements from the August 1992 experiment and regular monitoring (continued on the next page).

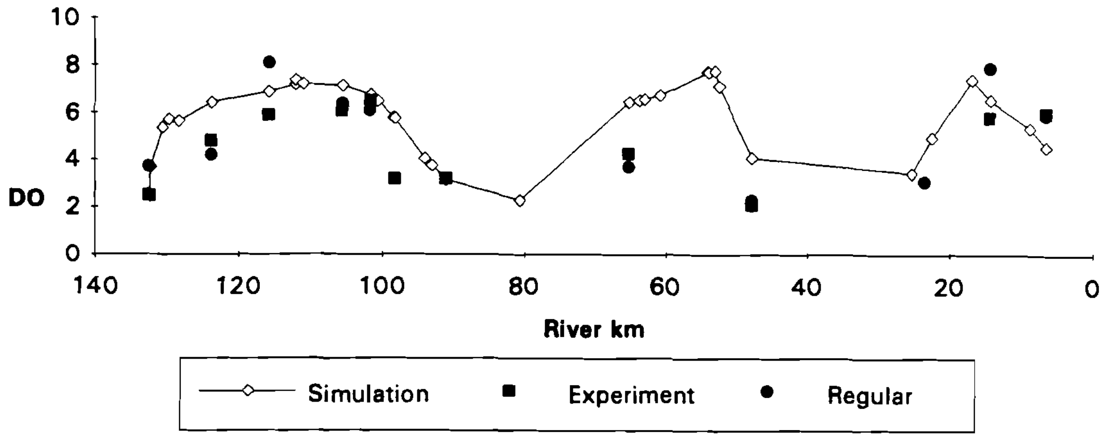


Figure 9.13 A comparison of the selected model with measurements from the August 1992 experiment and regular monitoring (continued).

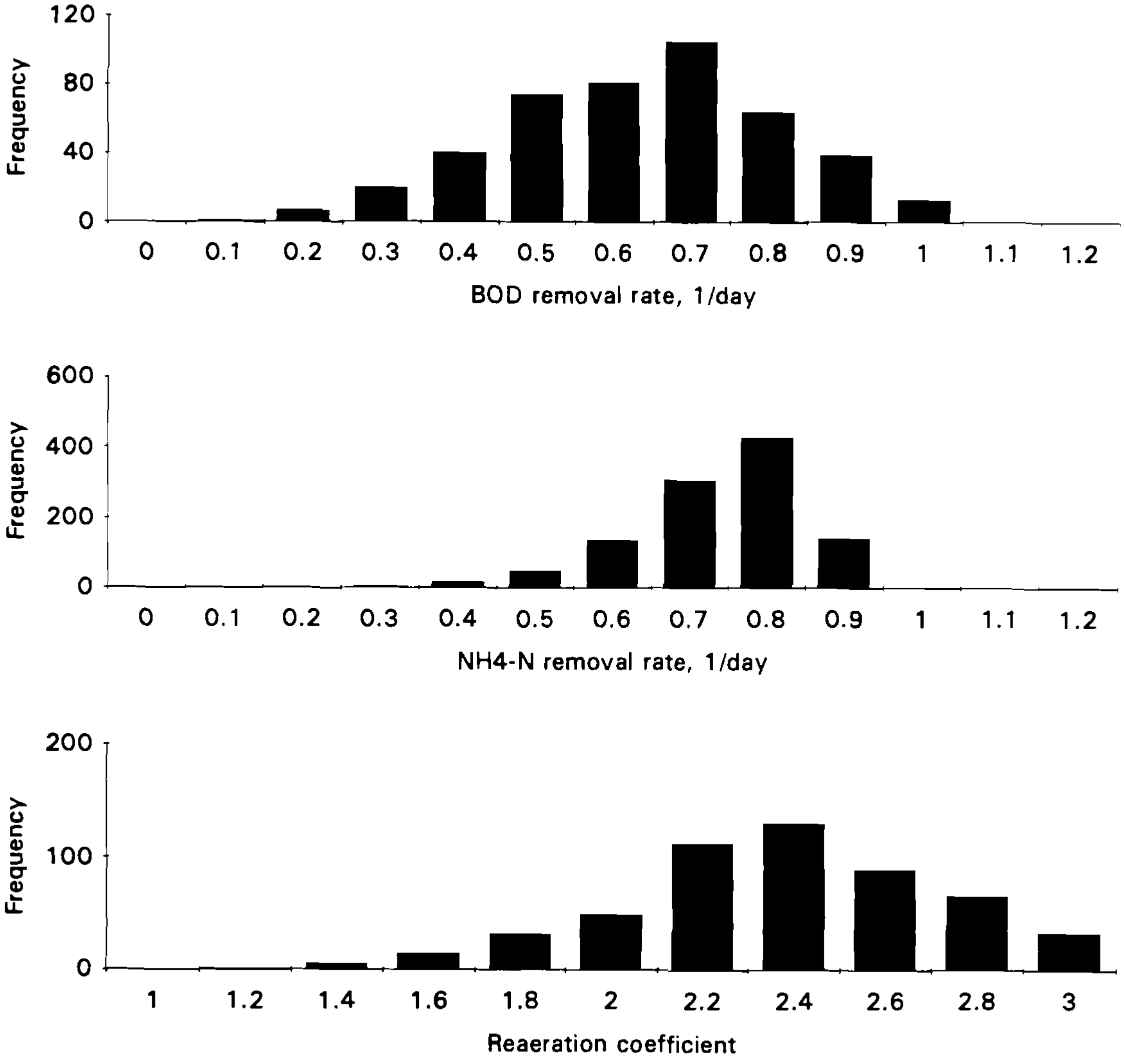


Figure 9.14 Frequency distributions for model coefficients to be used in uncertainty analysis.

9.6 Application of a Complex Water Quality Model (QUAL2E) to the Nitra River

QUAL2E is the most recent version of a series of water quality models initially begun in the early 1970's (Brown and Barnwell, 1987). QUAL2E has been widely used for waste load studies and has become a "standard" to which other models are compared. Its benefits utility lie in its relative ease of application and continued user-support by the US EPA.

The QUAL2E model is essentially a steady state model, with a possibility to simulate diurnal variations in a quasi-stationary approach. The model can simulate up to 15 water quality constituents, and it describes transport and water quality processes in considerable detail. The transport simulation includes both advection and diffusion. The biochemical processes include carbonaceous BOD decay, oxidation of ammonia and nitrite nitrogen, algae growth (followed by uptake of nutrients) and respiration (followed by release of organic phosphorus and nitrogen), settling of algae, carbonaceous BOD, organic phosphorus, and nitrogen. The interrelation of the water quality constituents in QUAL2E is shown in Figure 9.15. The model can also include coliforms and arbitrary conservative and nonconservative substances (not shown in the Figure 9.15).

QUAL2E CONSTITUENT INTERACTIONS

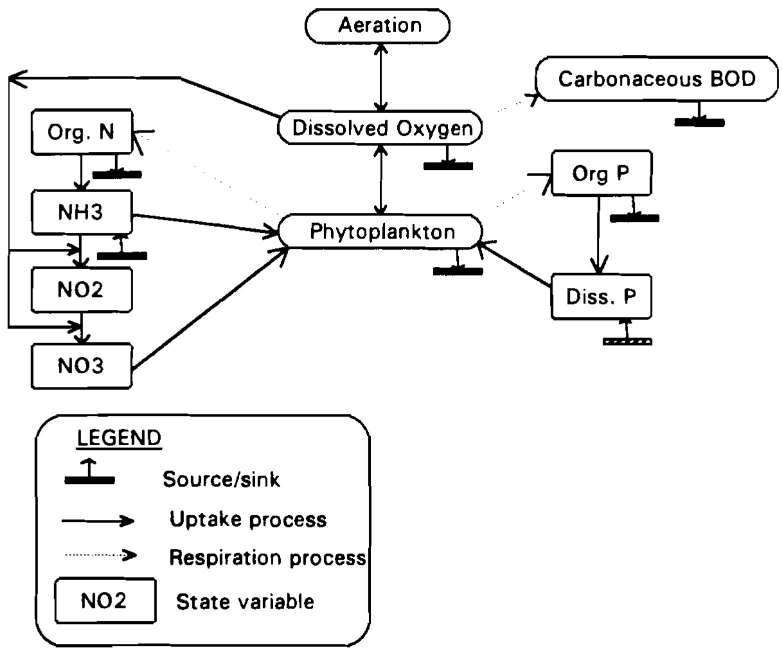


Figure 9.15 An overview of the constituents of QUAL2E and their interaction

The calibration procedure for QUAL2E used the dataset from the August 1992 experiment (Breithaupt and Somlyódy, 1994). Initially, all rates (e.g. BOD decay rate, nitrification rate) were set to the minimum values of the typical ranges listed in the QUAL2E manual (Brown and Barnwell, 1987). This keeps the effect of any process at a minimum and allows initial examination of simple mass balance effects on water quality. Subsequently, the effects of the process rates are 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 followed for each state variable.

As the BOD decay rate (k_1) is varied (Figure 9.16), the minimum residual is in the neighborhood of k_1= 2.0/day, which is unrealistically high. Consider the BOD concentrations in the river in the vicinity of the municipality Nitra (river km 52.5). The resulting BOD concentration in the river immediately after mixing with the effluent from Nitra STP would be about 41 mg/l (32 mg/l if only the Nitra STP effluent is considered). The observed BOD level at Nitrany (river km 47.8, i.e. 4.7 km downstream from the Nitra STP emission point) was 13.2 mg/l. If the observed value is accurate, then decay rate of 4.36 1/day or 3.21 1/day would be expected for each respective concentration. However, if the measurement at Nitrany is disregarded, a minimum residual is found around k_1 = 1.0/day (Figure 9.16). Considering the above analysis, it seems appropriate to take k_1 = 1.0/day. The simulation of BOD with this decay rate value is shown in Figure 9.17. The sawtooth shape results from the BOD loads from emissions.

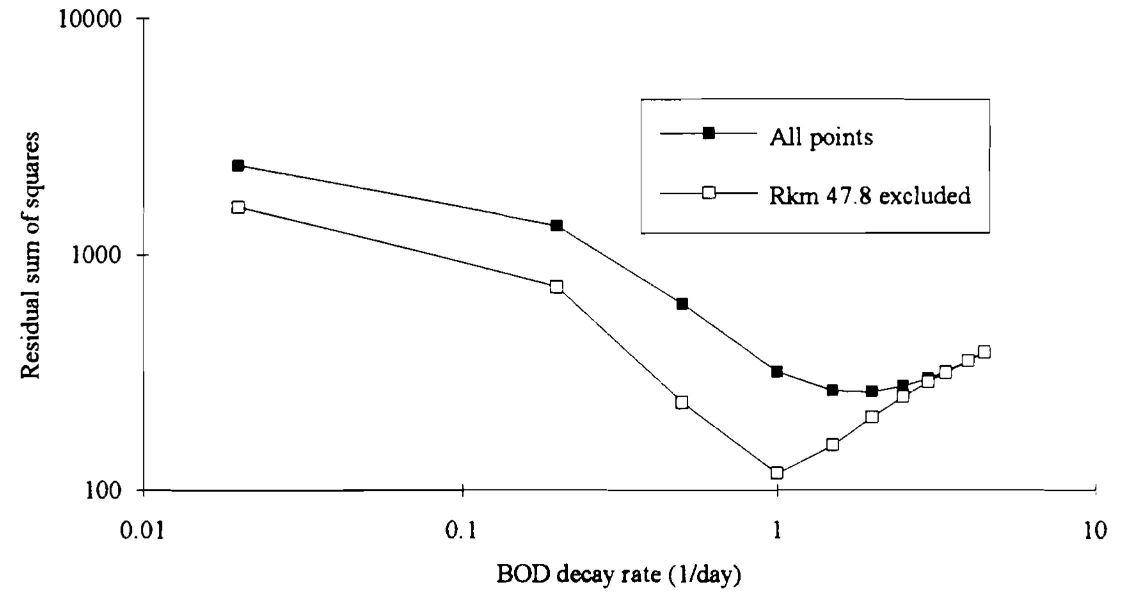


Figure 9.16 Residual sum-of-squares for QUAL2E BOD results when compared to observations. BOD-5 decay rates (k_1) are varied from 0.02 to 4.5/day

Dissolved oxygen is influenced by BOD, reaeration rate, and temperature. 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'Connor and Dobbins (1958) developed an equation for reaeration suitable for low velocity streams, and it was initially used for simulation (Figure 9.18). The reaeration rate obtained for the Nitra River by the O'Connor-Dobbins method falls into the range from 3 to 42 1/day, with the most typical value near 20 1/day. The shape of the dissolved oxygen curve is approximately the same as the observations, but the results are too large. Another approach where reaeration coefficients are directly input was attempted instead. A reaeration rate of 7.5 1/day was chosen since it matched the observation for the upper river stretch better than the O'Connor-Dobbins method. To improve the fit further, sediment oxygen demand (SOD) was introduced in the lower reaches. The QUAL2E DO results with the chosen reaeration coefficient and SOD are illustrated in Figure 9.18. Overall, this gives a good fit, even without significant contributions to DO from algal growth. However, in order to obtain the fit the reaeration rate was set to a level approximately two times lower than the value recommended by literature.

There were no measurements of algae biomass during the August 1992 experiment. Therefore, it was decided to perform several sensitivity tests to evaluate the effect of suspended algae on other state variables. A range of headwater algae concentrations were evaluated. To analyze the effects of algae, model coefficients such as growth and respiration rates were set to extreme limits giving either maximum algae growth or giving a minimum. The effect of varying of algae concentration on the sum-of-square residual is illustrated in Tables 9.3 and 9.4.

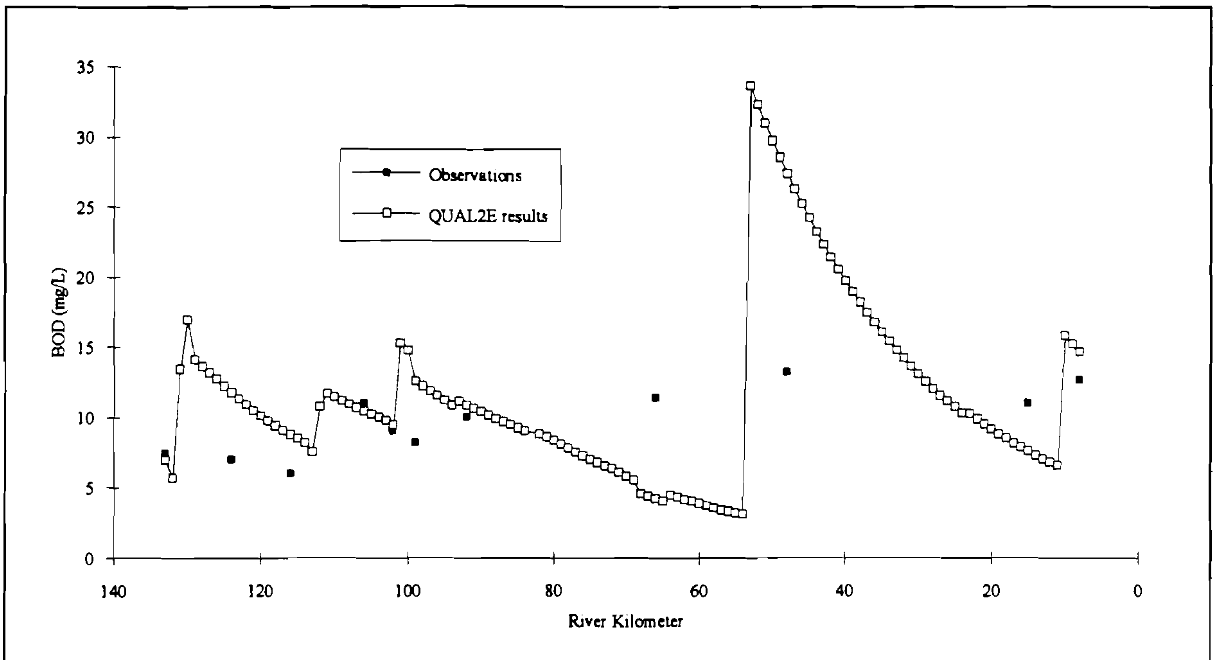


Figure 9.17 BOD-5 observations and QUAL2E model results

Table 9.3 Sum-of-square residual values of constituents affected by algae growth with extreme parameter values giving *minimum* algae growth

Headwater Algae Concentration (mg/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

Table 9.4 Sum-of-square residual values of constituents affected by algae growth with extreme parameter values giving ***maximum*** algae growth

Headwater Algae Concentration (mg/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 from this analysis that suspended algae growth simulated by QUAL2E does not significantly affect nutrient and DO levels. However, there are indications from the August 1992 data set which suggest that the algae is growing occurring, particularly in the lower part of the river (Chapter 8). Without the algae biomass or chlorophyll-a measurements, it is difficult to discover the reason for the noted discrepancy.

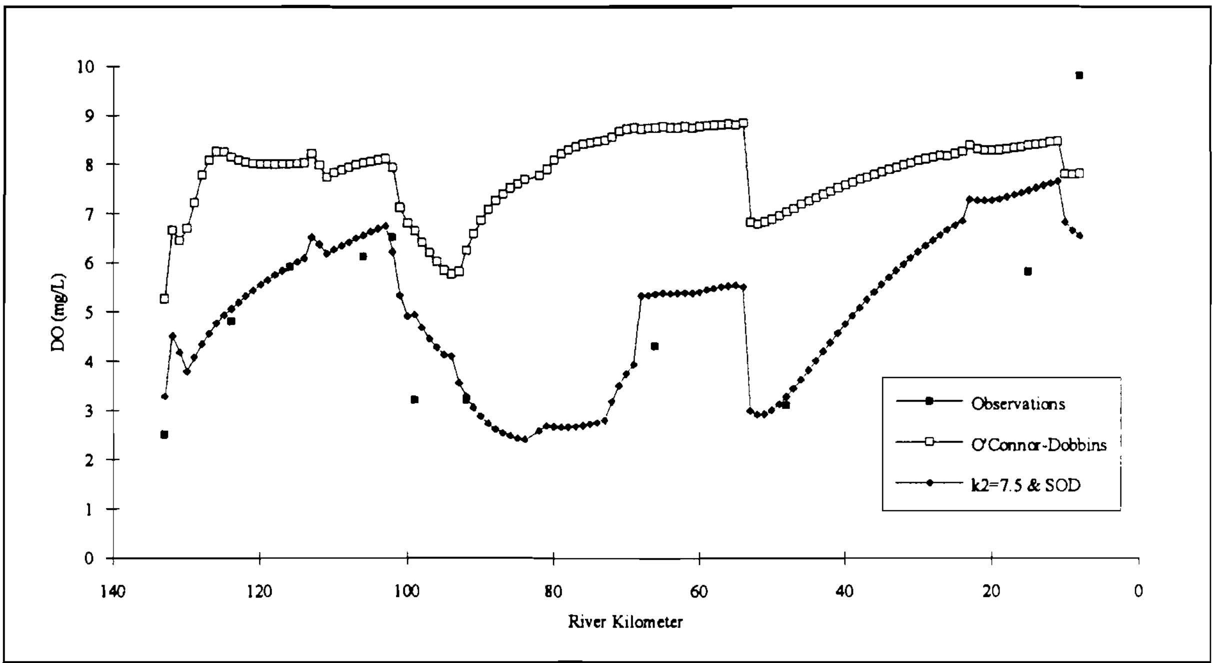


Figure 9.18 Dissolved oxygen observations and QUAL2E model results

The calibration of nutrient models without organic phosphorus or nitrogen is problematic. For dissolved phosphorus, it makes the process simply a mass balance exercise, especially since algae uptake cannot also be adjusted. However, examination of QUAL2E results and comparison with observations (Figure 9.19) suggests there may be significant algal uptake of dissolved phosphorus (this is the only sink for dissolved phosphorus in QUAL2E). But without data on the concentration of algae, it is impossible to ascertain the effect.

For the inorganic nitrogen series (NH_3, NO_2, and NO_3), the problem is compounded by the interaction between the forms of nitrogen. QUAL2E's ammonia results (Figure 9.20) show that it is underestimated in the upper river, suggesting another source of ammonia (cf. Chapter 8). It is probable that hydrolysis of organic nitrogen is a likely source, but there are no observations with which to estimate the rates of hydrolysis. In the lower river, ammonia is overestimated; a possible sink is the uptake by algae. Nitrite results agree well, except at near the end of the system. Nitrate results are overestimated throughout the river system. Like ammonia, the only sink in QUAL2E is algal uptake.

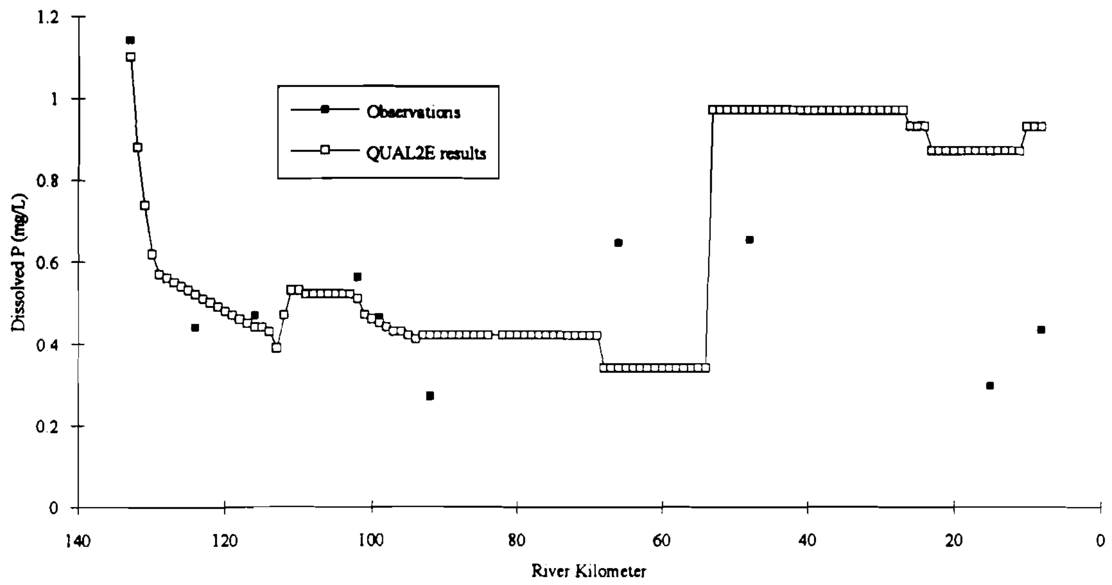


Figure 9.19 Dissolved phosphorus observations and QUAL2E model results

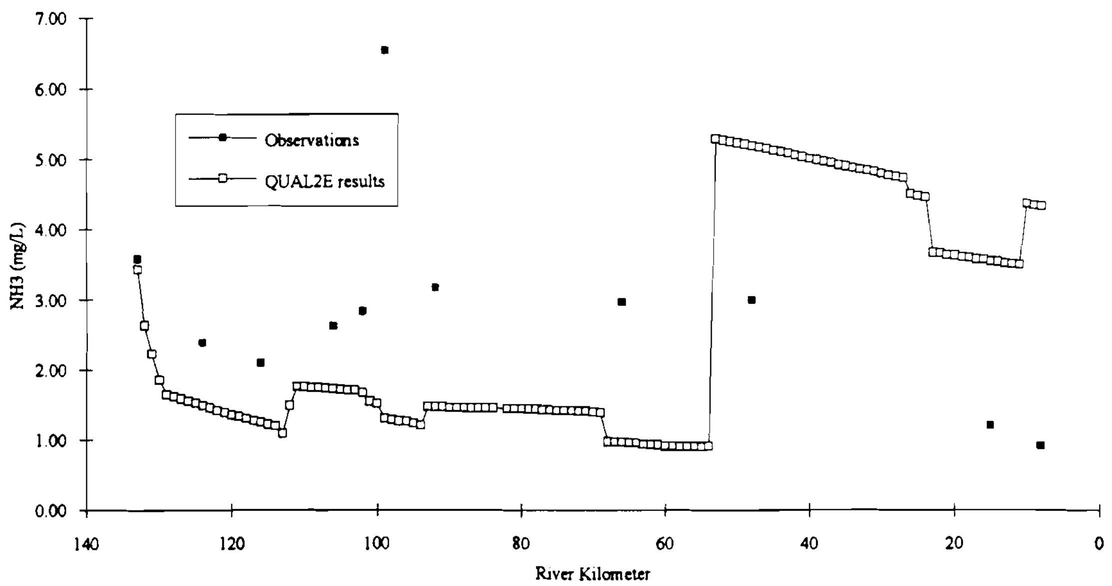
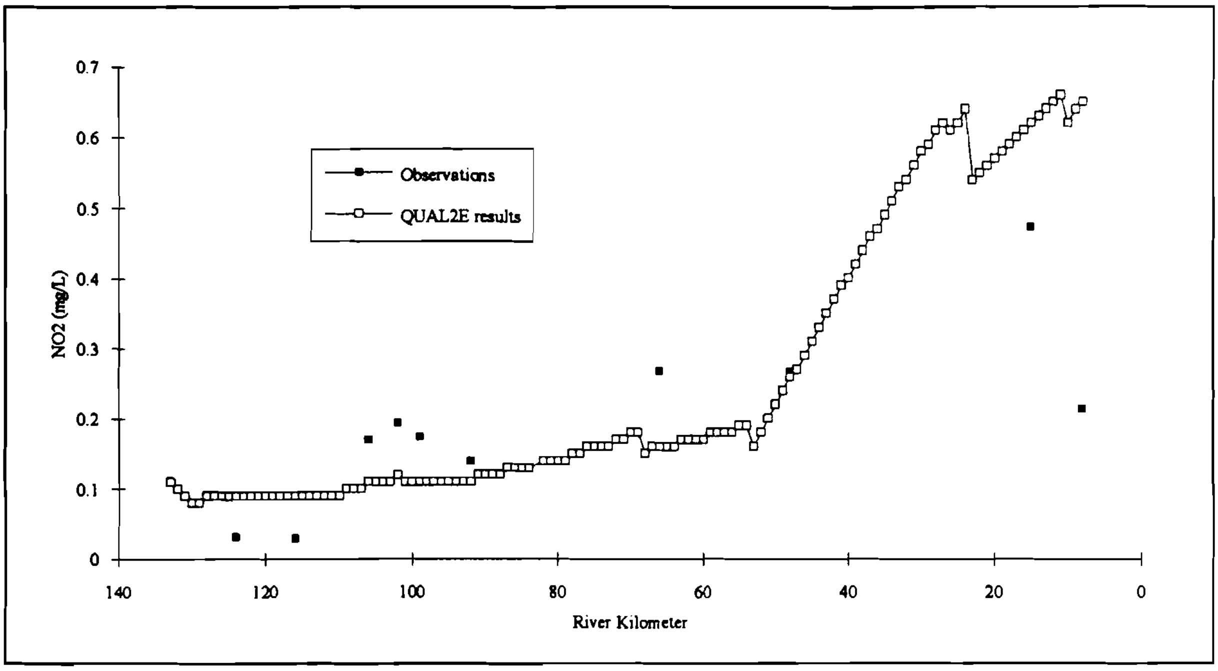


Figure 9.20 (continued on the next page). Inorganic nitrogen observations and QUAL2E model results: ammonia, nitrite, and nitrate.



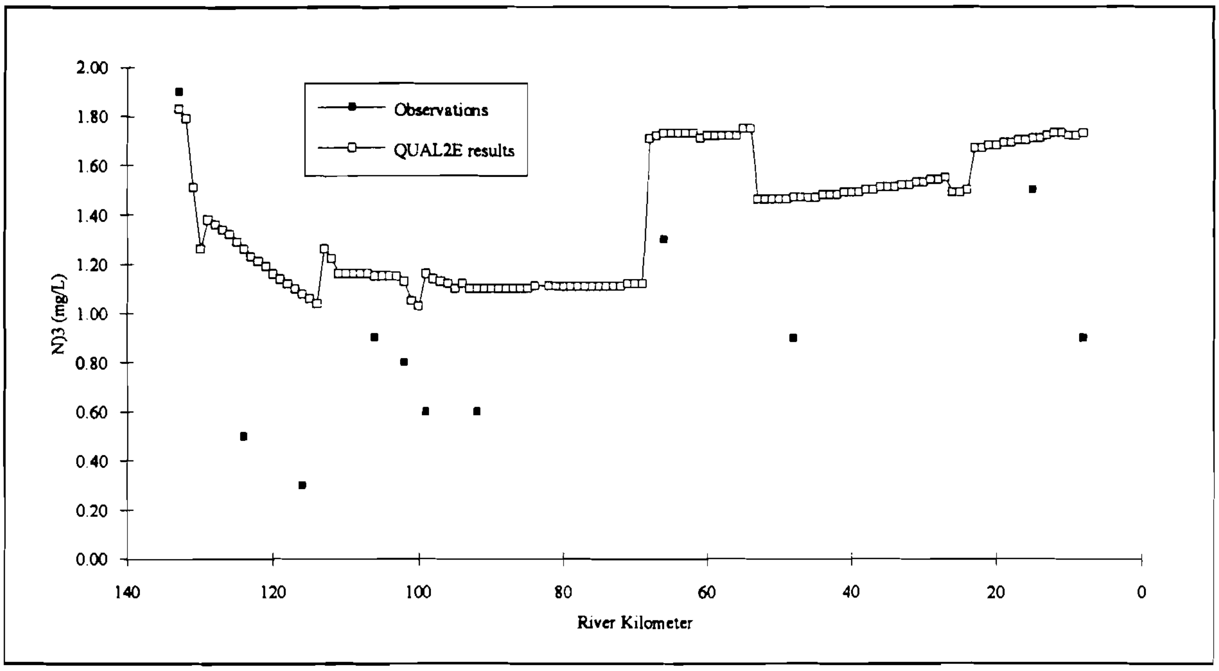


Figure 9.20 (continued) Inorganic nitrogen observations and QUAL2E model results: ammonia, nitrite, and nitrate .

Overall, it can be said that the parameters estimated for QUAL2E are close to those selected for the simple dissolved oxygen model (section 9.5). However, the application of complex models is hindered by limited availability and accuracy of data. Further, use of a complex model in a decision support framework requires unfeasible computational resources. (Further discussion on use of different types of the water quality models in the policy analysis framework see in Section 9.7).

9.7 Implications for the Control Policy and Conclusions

Both the simple and complex water quality models were calibrated by using data from the longitudinal water quality profile measurements (Sections 9.4 and 9.6, respectively) The comparison of the calibration results shows overall agreement in the coefficients included in both models. Both BOD-5 removal rates are close to 1.0 1/day, and the reaeration rates are both approximately two times smaller than the literature values. However, calibration of the complex water quality model QUAL2E was hindered by absence of some measurements (e.g. organic phosphorus and nitrogen, algae biomass) as well as the limited accuracy of the data. Results on nitrogen, phosphorus, and algae show poor agreement with measurements and estimations. Application of complex water quality models require significant data collection and analysis effort. Furthermore, computation time is significant due to the large number of grid elements involved. All these features make it rather difficult to incorporate complex models into the policy decision support framework. Complex water quality models could be used much more effectively to check results of simpler models or in cases where a more detailed analysis is required by the nature of the problem (for instance, eutrophication of lakes). On the contrary, simple models (similar to those described in Section 9.3) can be used effectively in the policy framework due to the modest computation and data requirements (Chapter 10 and 13).

The findings of the modeling efforts discussed in this chapter could be summarized as follows:

- Deterministic evaluation of longitudinal water quality profile observations can often lead to unrealistic parameter values.
- The Hornberger-Spear-Young (HSY) approach forms an attractive, robust, and generic methodology to account for uncertainties.
- The BOD decay rate obtained for the Nitra River was rather high due to inadequate and partial biological waste water treatment and small water depth. Parameter values of the Streeter-Phelps model and its extensions were in overall harmony with the recommendations of the literature. The estimation for the reaeration rate, however, is approximately two times smaller than suggested in the literature.
- Different model versions could be adequately calibrated to the available data set.
- The PDFs of the model parameters were characterized by rather broad ranges. This calls for an application of a risk analysis framework for the water quality control policy models.
- The application of the complex water quality models for the policy analysis is limited by the availability and accuracy of data, as well as by significantly greater computational requirements. The simpler dissolved oxygen models appear to be an effective and adequate tool for aiding in the selection of control policies for organic pollution, while complex models (QUAL2E) are appropriate for checking purposes and handling complicated situations.

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10. WATER QUALITY MANAGEMENT MODEL: A DYNAMIC PROGRAMMING APPROACH

The management problem addressed in this study is the development of a regional wastewater treatment policy, which involves identification of an optimal, least-cost management strategy out of many feasible alternatives. This chapter focuses on management tools which are applicable, while Chapter 13 provides a comprehensive account of the policy analysis performed for the Nitra River basin.

A brief description of different techniques used to identifying "optimal" water quality control strategies is presented in the first portion of this chapter. The dynamic programming (DP) based optimization approach employed in the present study is described next. Possible improvements of the DP approach are also included. Recommendations for further analysis (e.g. alternative formulations of the problem, incorporation of uncertainty, and scheduling) are given at the end.

10.1 Optimization/Simulation Techniques Applicable for Water Quality Management Models

Many of the water quality management models found in literature are optimization models. Linear programming (LP) and dynamic programming (DP) have been the most commonly used techniques. Nonlinear programming (NLP) applications have been less popular in this field.

Optimization models perform a screening function by identifying "optimal" management alternatives. Simulation models also have been used to evaluate the effects of various feasible management alternatives. Systematic heuristic search techniques (e.g. simulated annealing, and taboo search; see Pirlot, 1993 for details) which differ significantly from simulation approaches have not received much attention in water quality management studies. A brief description of the LP, NLP and simulation approaches is provided below. Section 10.2 addresses the applicability of the DP approach employed to solve the problem presented in this paper.

10.1.1 Linear programming

Linear programming (LP) has been one of the most widely used optimization techniques in water quality management, and its wide popularity is partly due to the availability of general-purpose LP packages.

LP solves a special type of problem in which all relations among the variables are linear. This requirement should be fulfilled by the objective function and the constraints which make up the model formulation. It is possible, however, to solve nonlinear problems by linear programming. This requires linearization of the original problem under certain assumptions. A typical objective function for an LP application would be to minimize the cost that satisfies all the water quality standards in the river system. Convex cost functions are required for the successful implementation of an LP model. These requirements, together with the integer properties of water quality management decisions, force a reformulation of the problem so that it is solvable by the LP method. Such reformulations, however, may lead to suboptimal solutions. LP applications for water quality management problems have been presented by many researchers, including Loucks et al. (1967), ReVelle et al. (1968), Biswas (1981), and Burn and Lence (1992).

Linear mixed-integer programming (MIP) is also appropriate in solving the water quality problem, as the feasible decisions of the problem comprise a discrete set of wastewater treatment alternatives. The advantage in applying the MIP method is that the problem does not need to be reformulated as a continuous one. However, MIP imposes severe limitations on problem size because of its high computational load. Non-linear MIP programs are also available, although their computational requirements are high. Hughes (1971), and Loucks et al. (1981) have presented MIP formulations for water quality management problems.

10.1.2 Nonlinear programming (NLP)

Wastewater treatment cost functions, as well as "transformation functions" which relate waste discharge to river water quality, are generally nonlinear. Therefore, a nonlinear optimization model is a more accurate representation of the physical system than the linear model. Nevertheless, NLP approaches have been less popular, due to their complexity and large computational requirement. Furthermore they cannot easily incorporate uncertainty in the model formulation. Hwang et al. (1973), Bayer (1974), and Pratishthananda and Bishop (1977) have applied nonlinear programming for river water quality management.

NLP does, however, offer a more general mathematical formulation. It can handle nonseparable objective functions in addition to the nonlinear objective functions and constraints. NLP includes quadratic programming, geometric programming and separable programming as special cases.

10.1.3 Simulation methods

Simulation in the present context implies the mathematical evaluation of various processes and management alternatives and their effect on the river water quality. One advantage of simulation over optimization is the ability of the simulation approach to describe, in more detail, the various physical, chemical and biological processes that characterize a river system. For most problems, optimization techniques require considerable simplifications of the processes which occur in the system.

In order to select an "optimal" management strategy, a simulation model must estimate the consequences of a variety of feasible alternatives. The number of feasible combinations of wastewater treatment alternatives increases rapidly as the number of treatment plants and treatment levels rises. Consequently, a large number of simulations are required in order to select an "optimal" set of alternatives. For example, if a river basin has 10 wastewater discharge locations, each having 5 feasible treatment alternatives, the total number of feasible combinations will be 5^{10} (this number corresponds to a deterministic case). In order to incorporate the uncertainty of various inputs and parameters into the analysis, a Monte Carlo approach would be needed and would require an impractical number of simulations.

Simulation models in water quality management include those presented by Warren and Bewtra (1974), Orlob (1982), and Thomann and Mueller (1987). The role of BOD dischargers in the Nitra river basin was analyzed using a simulation model by Koivusalo et al. (1992). A state-of-the-art review of water quality simulation models, including a discussion on decision support systems, can be found in Somlyódy and Varis (1992).

10.2 Applicability of Dynamic Programming to Solve the Present Problem

Dynamic programming (Bellman, 1957) is an optimization method for a multistage decision problem. It decomposes a problem with a sequence of decisions into a sequence of sub-problems each having one or a reduced number of decisions. These subproblems are solved recursively, by considering the sub-optimal solution(s) of one subproblem as input(s) to the subsequent subproblem.

The selection of optimal wastewater treatment alternatives in a river basin is a sequential decision problem in space and in time. Spatially, the decisions are made for a series of locations in a river basin. Due to the downstream-only propagation of pollutants in a river system, the water quality at a particular location in a river is fully determined by the water quality at the immediate upstream discharge/control point (or by several discharge/control points in the special case where the location is below a confluence). Similarly, when investigating investments over the planning horizon, decisions are made at points in time. Decisions made at one time point directly affect those made at the next time step. These special sequential attributes make the dynamic programming approach a suitable method for solving the river water quality management problem.

Model linearity is not a requirement for DP. This opens up the possibility to incorporating complex non-linear water quality models within the optimization process. Most of the other optimization techniques would require significantly simplified forms of such models. Furthermore, DP can incorporate stochastic features as well. In discrete DP, constraints that reduce the state or decision space (e.g. pre-specified water quality standards) are advantageous because they reduce the computational requirements of the model. Such constraints, in other optimization techniques, increase the computational loads.

The main limitation of DP is the rapid increase in computational load as the number of state variables increases. This is appropriately known as the "curse of dimensionality" of dynamic programming. Separability of the objective function is one requirement of a problem which is to be solved by DP. However this requirement can be relaxed by alternative formulations (by including additional state variables), although this increases the problem size.

Applications of dynamic programming (DP) for water quality management problems have been reported by Newsome (1972), Hahn and Cembrowicz (1981), and Cardwell and Ellis (1993). A discussion about the DP approach and the reasons for its selection in the present study are presented below. A more detailed description of the state-of-the-art of water quality management models is provided by Kularathna and Somlyódy (1994).

10.3 Description of the Optimization Problem

Two different optimization formulations are considered in the present analysis. They are the minimization of total annual cost (TAC) and the minimization of total investment cost (IC) of the wastewater treatment strategy which is required to satisfy various water quality standards (ambient and/or effluent). The TAC comprises the operation, maintenance, and replacement cost (OMRC) and the annual component of IC. The ambient water quality standards are imposed at selected points along the river. They specify the minimum limits on DO, and maximum limits on BOD and NH4-N at those specific locations.

The above formulations can be mathematically expressed as:

$$\underset{\eta_i \in T_i}{\text{Minimize:}} \quad \sum_i C_i(\eta_i)$$

$$\text{Subject to:} \quad Q_{j,k} \geq S_{j,k} \quad \forall\ j,k$$

$$E_{i,k} \geq E_{standard,\,k} \quad \forall\ i,k$$

where $C_i(\eta_i)$ is the treatment cost (TAC or IC) required to achieve a treatment efficiency η_i at the i^{th} wastewater discharge. T_i denotes the feasible set of treatment alternatives for the i^{th} discharge. $Q_{j,k}$ is the water quality at the standard location j, expressed by the quality indicator k. The corresponding quality standard is denoted by $S_{j,k}$. $E_{i,k}$ and $E_{standard,k}$ stand for the effluent quality (expressed by indicator k) at the i^{th} discharge, and the effluent standard for k^{th} indicator respectively.

10.4 DP Formulation

The river system was subdivided into a number of reaches, which were further divided into "stages". The river network has been defined by the interconnection of different reaches. A "stage" was considered as part of the river from a point immediately upstream of a "point of action" (P1) to a point immediately upstream of the next "point of action" (P2) which is located downstream. A "point of action" can be: a wastewater discharge, an abstraction point, a measurement point, a point with a pre-specified water quality standard, a weir or an artificial point which is introduced in order to maintain the computational procedure.

10.4.1 Discretization of quality states

Application of dynamic programming in the present problem requires a sub-division of the feasible water quality range into a number of discrete intervals. The feasible quality range is defined by the various water quality indicators that are considered. In the case of single quality indicator, the term "interval" is self-explanatory. If two indicators are to be considered (e.g. DO and BOD), the "interval" would be an "area" defined by the two indicators. Three indicators form quality "intervals" which can be visualized as three-dimensional blocks. Three water quality indicators were explicitly included in the present analysis. They are the ambient concentrations of DO, BOD and NH4-N. Depending on the number of intervals considered for each quality indicator, the total number of feasible "joint" quality intervals can be quite large. The discretization used in the present study considered 40 levels for DO, 120 levels for BOD, and 80 levels for NH4-N. The practical ranges considered for DO, BOD and NH4-N were 0-10, 0-30, and 0-20 (mg/l) respectively. In fact, the last interval of each indicator was of variable size, in order to accommodate concentrations which may be larger than the predefined ranges. All other intervals were of equal size for a particular quality indicator.

10.4.2 Stage-by-stage computations of DP

DP computations are started from the most upstream point of the river system. The water quality at this point is known, which implies only one quality state at that point. Using Bellman's (1957) principle of optimality, the DP calculations are performed stage by stage, proceeding towards the most downstream point of the system. As it proceeds, it makes use of river water quality models which estimate the feasible water quality states at the beginning of the subsequent stage. There can be an increase in the number of possible water quality "states" at a "subsequent" stage, depending on the number of management alternatives available at the current stage. As described above, the allowable water quality range at each stage is divided into a discrete number of "quality intervals". If two or more quality states fall within an interval, the quality state that corresponds to the best value of the objective function within the interval is retained for further computations downstream. This indicates that the maximum number of quality states at any stage is equal to the number of discrete quality intervals.

For each of the quality states that are retained for further computation, the objective function values (cumulative costs up to the particular stage) and the current decision (if any) are recorded. It is also necessary to record, for each of the quality states, the previous stages' quality state which produced the current state. The values of the three quality indicators for each quality state are also kept on record. At a water quality standard (constraint) point, it is possible to eliminate some of the discrete states from further consideration, if they do not satisfy the standards. Therefore this reduces the computational load of the DP problem. However, in the present study, the optimization is performed repeatedly with different quality constraints; so that the quality constraints are not considered as a predetermined set of limiting factors.

After performing the DP computations to the most downstream stage of the system, the feasible water quality states at the downstream stage will produce various values for the cumulative objective function. Each value indicates an alternative solution which satisfies the water quality standards imposed. It is necessary to select the state which has the best value for the objective function, so that the optimal solution can be traced upstream from this starting point. The non-optimal solutions (nonoptimal in terms of the single objective considered) that are generated by the DP computations, can be analyzed in a multiobjective decision framework if necessary. This is another advantage of the DP approach, because it generates a set of solutions which cannot be obtained by other optimization techniques. Figure 10.1 displays the main elements of the DP-based computational procedure (see Appendix 10.1 for a detailed flow diagram). A graphical representation of the computations is presented in Figure 10.2.

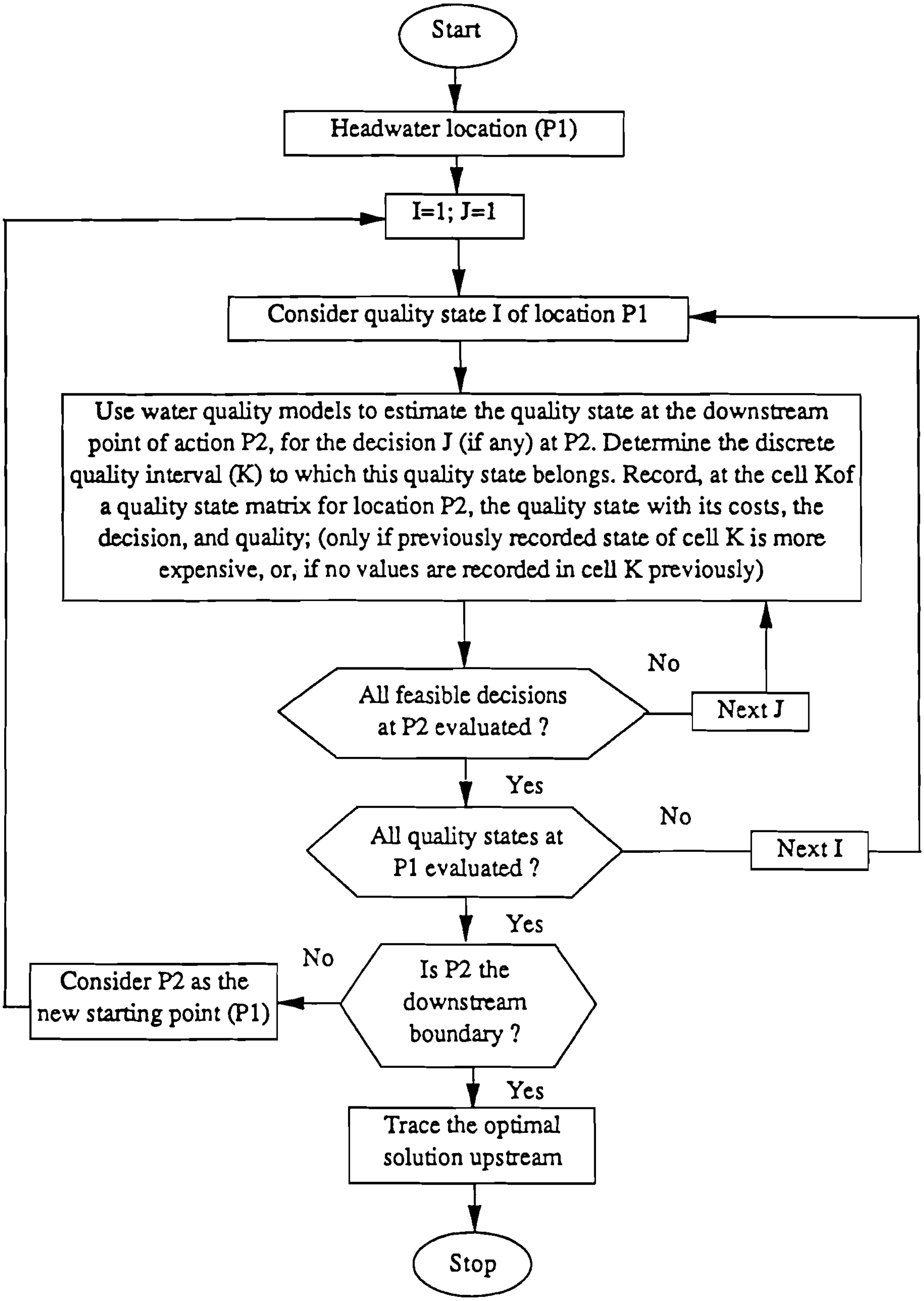
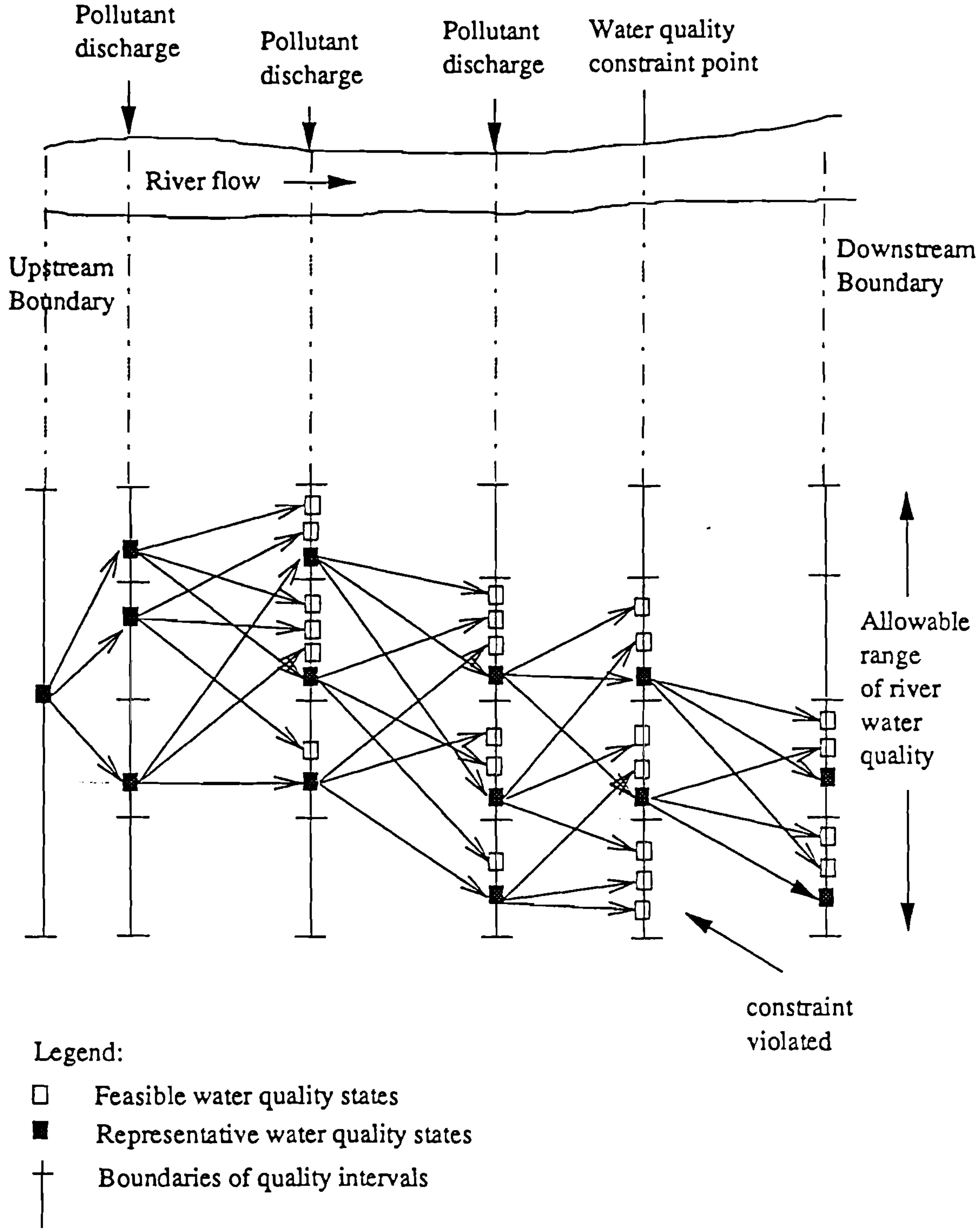


Fig. 10.1 Flow diagram for the computational procedure of DP-based optimization



Notes: For the sake of simplicity, this illustration assumes the following

1. Only one quality indicator to define the river water quality
2. Only three feasible treatment alternatives for each wastewater discharge (which can be different from one discharge to the other)

Figure 10.2 Graphical representation of DP computations

As indicated above, only one quality state is retained from each discrete quality "interval" at each stage, for further computations. This particular state will subsequently represent the quality interval, hence it can be termed the "representative quality state" for the quality interval at that stage. The elimination of all but one state from each interval considerably reduces the amount of computations. However, it can easily lead to a non-optimal solution, especially if the discretization is not fine enough.

10.4.3 Mathematical formulation of the DP computations

The mathematical formulation of the DP solution procedure is given below, for a simplified case of a river without tributaries. An extended form of this procedure was employed to analyze the Nitra system comprising several tributaries. The optimization problem for the most upstream stage can be expressed as:

$$f_1(Q_1) = \min_{\eta_1} C_1(\eta_1) \tag{1}$$

$$Q_1 = T_1(\eta_1, Q_0) \tag{2}$$

where, $f_n(Q_n)$ (n=1 for the most upstream stage) is the cumulative optimal cost required to achieve the allowable water quality state Q_n at the end point of stage n. $C_n(\eta_n)$ is the treatment cost required (which is zero if there is no wastewater discharge or if no treatment is to be made at stage n) to achieve an efficiency of η_n at stage n. This efficiency η_n (the removal rate) is the decision variable at stage n of the problem. For the most upstream stage, Q_0 indicates the headwater quality. $T_n(\ )$ represents the transfer function at stage n, and is a water quality simulation model.

The efficiency of a particular treatment alternative is characterized by the influent and effluent quality. For each WWTP, a set of feasible treatment alternatives is developed. Each alternative is represented by its IC, OMRC, and the effluent concentrations (see Chapter 11). The optimization task is to make a (0,1) decision with regard to all feasible treatment alternatives at each of the WWTPs. These decisions should sum-up to one at each WWTP location. For the subsequent stages (from n=2 up to the most downstream stage), the recursive relation takes the following form:

$$f_n(Q_n) = \min_{\eta_n, Q_{n-1}} [C_n(\eta_n) + f_{n-1}(Q_{n-1})] \tag{3}$$

$$Q_n = T_n(\eta_n, Q_{n-1}) \tag{4}$$

It should be noted that the standard approach in using discrete DP is to discretize the state space and represent each discrete quality state by a representative point. This representative point is usually fixed, the center of the discrete state, for example. Some discontinuity occurs at this point, because the simulated water quality may not coincide with the central point of the quality state. However, the representative point used in the current approach is the simulated quality state which corresponds to the best objective function value within the current state. This allows a correct representation of the river water quality profile (for the selected control actions), as the rounding off of the quality values is not involved.

As indicated above, the current decision (treatment alternative), the previous state (at the previous stage) and the cumulative cost are recorded for each allowable water quality state at each stage of the computation. Having reached the most downstream point, the optimal solution can be traced-back upstream.

10.4.4 Possible improvements

In the present study, uniform ranges of water quality indicators were considered for the DP computations. However the feasible quality range at different locations along the river can be significantly different. An improvement in the accuracy of the results may be obtained by considering location-specific ranges instead of uniform ones. A narrower quality range would lead to a finer discretization, assuming the same number of quality intervals as considered in the uniform case.

An equal number of feasible "quality intervals" was considered at each stage of computations in the present study. Nonetheless, it is advantageous to discretize the quality range into a different number of intervals at different places. This is important due to the fact that it is not practical to use a large number of discrete intervals at each stage. Such a discretization would lead to an impractical computational requirement. However, certain stages need fine discretization, while some other stages need a smaller number of intervals because there are only a few feasible quality states. Significant reductions in computational load can be realized if it is possible to have a non-uniform discretization, combined with non-uniform quality ranges.

For each quality interval, only the "best" quality state falling within that interval was taken as the representative quality state. All the others within the interval were eliminated from further computation. However, this introduces an error in the optimization procedure, eliminating some non-identical quality states (and consequently certain feasible decisions) on the basis of a suboptimal objective function value. This error might be reduced if the objective function is not the sole criteria used to eliminate feasible states which are different from each other in terms of water quality. It may be worthwhile to test other heuristic approaches for selecting the representative quality states. Such an approach can be based on the comparison of the objective function value as well as one or more quality indicators.

10.5 Alternative Formulations of the Problem

Optimization of a Selected Water Quality Measure

The optimization formulations of the present study were aimed at finding the least-cost solutions subject to certain water quality standards. It is also possible to formulate the objective function based on water quality indicators, thus the aim is to achieve the best water quality under given financial constraints. Using the notation introduced in Section 10.3, an example for such a formulation is provided below.

$$\text{Minimize:} \quad \sum_j \sum_k \{\max [0, (Q_{j,k} - S_{j,k}) / S_{i,k}]^p\}$$

$$\text{Subject to:} \quad TAC \leq \text{Maximum allowable TAC}$$

$$IC \leq \text{Maximum allowable IC}$$

This formulation attempts to minimize the relative violations of water quality standards. The relative violations are raised to the power p ($p>1$), and therefore larger violations will be penalized more than small ones. Consequently, the solution might be obtained solely based on a few locations with very poor quality water. This could have a negative impact on the locations at which quality standards are violated to a lesser degree. As the exponent p increases, the above formulation approaches a Min(Max) problem, which would attempt to minimize the maximum violation of quality standards.

Cost as an Additional State Variable

The DP approach produces an array of feasible solutions that fulfill the model constraints. In the case of the above formulation of optimizing a water quality measure, all the solutions generated (if any) will satisfy the particular upper limits specified for the costs. However if the cost was considered as an additional state variable, a set of solutions which correspond to different cost constraints can be obtained.

Multiobjective Considerations

Water quality management objectives are inherently multiobjective. These multiple objective can be categorized, but are not limited, to costs and quality. These categories have multiple objectives within themselves. The trade-off required between IC and TAC is one such example. A solution which considers only one or a few of these various objectives may be undesirable in light of the other objectives. The approach of the present study was to generate different solutions for two different objectives and various water quality standards. An *a posteriori* evaluation was needed to select the best compromise solution, or solutions.

From the view point of a decision maker (DM), multi criterion decision making (MCDM) include three types of techniques. This classification is based on the stage at which DM's preferences are incorporated into the solution procedure. Accordingly, the three types correspond to (1) *a priori*, (2) progressive, or (3) *a posteriori* incorporation of preferences. Although there is no hard line that demarcates the boundaries of these three types, they are considered here as a basis for the present discussion. The drawback of the first type is that the preferences of the DM are needed before the alternative solutions are available. The second type can be time consuming, and it generally assumes that the DM is actively involved during the solution process. The third type involves the generation of a set of Pareto-optimal solutions so that the DM can choose the preferred alternative. This does not imply the presentation of a complete set of solutions, since there are a number of techniques which aid the subsequent decision making process. One disadvantage of such an approach is the need to generate a comparatively large number of solutions.

10.6 Incorporation of Uncertainty

When the inputs and parameters of the model are uncertain (see Chapter 9), which is usually the case, a more comprehensive analysis is needed to evaluate the expected performance of the system, and/or to estimate the probability/magnitude of possible failures. Uncertainty can be taken into account using a number of approaches, which can be broadly categorized as explicit or implicit. Explicit approaches include stochastic linear programming (SLP), stochastic dynamic programming (SDP) and the use of chance constraints. Implicit approaches comprise Monte Carlo simulation combined with optimization, and scenario analysis.

SLP and SDP are generally associated with very high computational requirements. In the case of SLP, various alternatives scenarios are explicitly incorporated into the model formulation, which increases the number of variables and constraints significantly. The SDP computation suffers from the inherent "curse of dimensionality". Incorporation of many state variables and probability matrices which are needed to handle hydrologic and parameter uncertainty is therefore impractical. The chance constraint approach transforms the stochastic optimization problem into a corresponding deterministic one. This is done by defining constraints based on certain reliability levels, which is considered as a pessimistic approach. The selection of the appropriate reliability values (which can even be a part of the optimization problem) is also not straightforward. Sobel (1965) proposed the use of stochastic linear programming for the optimizations in water quality problems. Deininger (1965) applied a chance constrained LP method using an approximation of the Streeter-Phelps equations. Lohani and Thanh (1979) incorporated the random nature of streamflow into the waste load allocation problem by using chance constraints. Fujiwara et al. (1988) presented a study on stochastic water quality management. A stochastic dynamic programming approach was presented by Cardwell and Ellis (1993).

The main criticism against the Monte Carlo simulation technique is the large number of simulation runs that would be needed to make probabilistic conclusions on the performance of the system. Such approaches have been documented by deLucia and McBain (1981), and Burn (1989). An alternative is to perform a scenario analysis, which involves the identification of several representative scenarios. The applicability of the solution obtained for each scenario can be subsequently evaluated within a regret analysis framework. An approach based on multiple scenarios (reflecting alternative hydrologic, meteorologic and pollutant loading conditions) was presented by Burn and Lence (1992). Somlyódy and Wets (1988) described a linearized expectation-variance approach together with a quadratic stochastic optimization for a lake problem.

10.7 Scheduling Problem

The DP approach can be theoretically extended to handle the scheduling problem, although it would result in a high computational load. It would be necessary to determine a set of acceptable strategies for each time step separately, and it would be necessary to identify the allowable transitions of each strategy to those in the next time step. The resulting problem is a standard discrete dynamic programming problem, with selection of an optimal plan that is characterized by one treatment strategy at each time step.

However, in reality, the planning problem is much more complicated. The incorporation of uncertain scenarios would further increase the amount of work. Future wastewater discharges, river flows, interest rates, costs, and other input parameters which are not necessarily known with certainty, affect the solution. The essence of the planning problem would therefore be in selecting the appropriate scenarios and parameters which are to be incorporated into the model formulation.

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11. MUNICIPAL WASTEWATER TREATMENT ALTERNATIVES

The goal of municipal wastewater treatment is to preserve or improve the quality of receiving waters expressed in bacteriological parameters, suspended solids, and indicators of oxygen and nutrient balances. Dissolved oxygen problems are controlled primarily by removing carbonaceous BOD (CBOD). If, however, organic nitrogen compounds and their oxidation (characterized as nitrogenous BOD, or NBOD) play a significant role as well, then nitrification is also required in wastewater treatment. Because nitrogen compounds may have toxic effects that compromise drinking water quality, nitrification may be needed to reduce ammonia and nitrite. If the nitrate level is high (whether from nitrification in the treatment plant or in the receiving water), nitrogen removal is required to ensure a safe drinking water supply. Finally, phosphorus and nitrogen removal may be required to control eutrophication of inland waters and coastal seas, respectively.

A large number of wastewater treatment technologies have been developed to address the above needs. They combine different physical, chemical, and biological processes leading to different removal rates of BOD, TP, TN, and their fractions, and different investment and operation, maintenance, and repair costs.

11.1 A Brief Overview of Treatment Methods

The following is a brief description of basic wastewater treatment technologies (see e.g. Ødegaard and Henze, 1993, Henze and Ødegaard, 1994).

(1) Mechanical treatment

Mechanical treatment is the first step in almost every wastewater treatment plant. It includes a screen, grit chamber, and settling tank. The objective is to remove particles of various sizes and composition. About 60% of influent suspended solids are removed, but only about 30% of the influent BOD5. The process is not complicated and is applicable at all community levels. The treatment needed for the sludge includes stabilization and dewatering.

(2) Chemical treatment

Chemical treatment of wastewater is done in order to remove suspended solids, BOD and COD, and phosphorus. This process is always combined with others. In the process a precipitant (normally a metal salt) is added to cause the formation of a precipitate. The precipitate is a mixture of suspended solid particles, colloidal matter, a phosphorus metal compound, and a metal hydroxide compound. The precipitate is removed in a separation process such as settling or flotation.

There are many variations on the basic chemical treatment concept, with differences in the location of chemical addition (precipitation, simultaneous precipitation, post-precipitation, and contact filtration), the composition of the chemicals, and chemical dosage. If chemical treatment is used as the sole treatment, two methods are distinguished:

(a) Chemically enhanced primary (mechanical) treatment (CEPT)

(b) Primary (or direct) precipitation chemical treatment (PC)

The first type is used in the U.S. to increase the capacity and/or BOD removal at existing mechanical plants, while the second is employed extensively in Scandinavia primarily to remove phosphorus.

In CEPT plants, a metal coagulant is dosed prior to the settling tank, for example in the grit chamber. Because the primary intention is to coagulate suspended matter, the dosage is relatively low (less than 50 mg $FeCl_3$/l). An organic anionic polymer is added after the metal salt to cause flocculation. A cationic polymer may also be applied if coagulation is insufficient under the low metal salt dosage. CEPT significantly improves removal rates compared to conventional primary treatment. Under normal conditions, TSS removal is increased from 60% to 80%; BOD5 from 30% to 50-60%; and TP from 15% to 60-80%.

In primary precipitation plants a metal coagulant is added prior to the flocculation tanks with a relatively high dosage (150 to 250 mg $FeCl_3$/l or 100 to 200 mg $Al_2(SO_4)_3$/l), because the goal is to remove phosphorus in addition to suspended solids. Flocculation is accomplished by mechanical stirring and may be enhanced by adding 0.25 to 0.5 mg/l of anionic polymer. As a result of the higher chemical dosage, a primary precipitation plant achieves higher removal rates than a CEPT plant: about 90%, 70% and 90% for TSS, BOD-5, and TP, respectively. The higher chemical dosage leads to greater production of sludge, with the chemical-dependent portion of the sludge essentially linearly proportional to the chemical dosage (see below). The energy consumption of chemical plants is very low, but the cost of chemicals may be considerable. The sludge contains large quantities of organic matter and requires stabilization. There is also a considerable potential for gas production at larger plants (although there is no indication that the use of aluminum or iron has any adverse effects on the properties or production of the gas).

(3) Biological treatment

The purpose of biological treatment is to reduce the organic load. The treatment can be done on raw, or mechanically or chemically treated wastewater (i.e., not all biological treatment plants include a primary clarifier). The process, where organic matter is partly oxidized to carbon dioxide and partly converted to biomass, may be by either of two different technologies: activated sludge or biofilm. The microbiological processes are identical, but the reactor designs are quite different. In either design, the reactor provides for oxygen transfer to supply the oxygen needed for the oxidation of organic matter. In both processes, the wastewater is brought into contact with a large biomass, from which the wastewater must be separated after treatment (usually by settling).

Activated sludge plants (ASP) can be designed for high or low loading (low and high sludge age). Low load plants also provide nitrification (except during cold weather). The surplus sludge is stabilized, and thus no digestion is needed. In contrast, high load plants require sludge digestion as well as closer controls of treatment operations. An activated sludge plant can also be designed as a two-stage process, with a high-load first stage (biosorption) and a low-load second unit. The second unit requires a smaller volume than it would otherwise, because the load is reduced in the first stage.

Biofilm plants can be configured as trickling filters, submerged aerated filters, or rotating disks. Pretreatment is always needed. Biofilm plants are preferred for medium to high organic load. An advantage of biofilm reactors is that they require smaller residence times and thus smaller area for reactors. A disadvantage is their greater sensitivity to load fluctuations.

(4) Biological/chemical treatment

Combining biological and chemical treatment significantly improves phosphorus removal (to 90-95%) and also slightly improves TSS and BOD-5 removal. However, the quantity of sludge increases, and it requires concentration, stabilization (if pre-precipitated), and dewatering.

(5) Nitrogen removal

None of the methods discussed above remove nitrogen efficiently. Nitrogen is removed in two steps: first, nitrogen is oxidized to nitrate in the presence of oxygen (nitrification), and second, it is reduced to free nitrogen gas in the absence of oxygen (denitrification). A prerequisite for biological denitrification is an organic carbon source, which can be supplied either from the wastewater itself or from organic chemicals like methanol or acetate (these are sometimes supplied by industrial wastes). If wastewater is the carbon source, prenitrification is usually specified; if organic chemicals, postnitrification is typical. Both alternatives can be designed for use with the biofilm or activated sludge process. Biological phosphorus removal can also be incorporated into the denitrification process design.

Biological denitrification can be used to improve the performance of nitrifying (low-load) treatment plants. Denitrification restores about half of the alkalinity removed by nitrification and energy is also partially regained (since nitrate acts as an electron acceptor during denitrification).

11.2 Sludge Disposal and Treatment

Wastewater treatment sludge has a high content of suspended solids, organic matter, nutrients, and bacteria. The sludge may also contain undesirably high concentrations of heavy metals and organic chemicals, depending upon the amount of wastewater contributed by industry and the degree to which it is pretreated. For example, the cadmium content of a sludge even without a heavy industrial input can vary between 4 and 12 mg Cd/kg DS for all types of treatment, causing disposal problems in agriculture (e.g. the limit value is 12 mg Cd/kg DS in Slovakia).

There are three basic possibilities for sludge disposal:

- disposal in landfill
- use in agriculture
- incineration with disposal or reuse of ash

Landfill disposal is still widely used, but in many places, it creates a considerable nuisance due to odor and secondary pollution. Often land is unavailable. However, if suitable disposal sites such as closed mining areas exist, then landfilling may be the optimal sludge disposal option and may require only sludge dewatering as pretreatment.

Use of sludge in agriculture is a more sensible disposal method, as municipal wastewater sludge can be a good soil conditioner and contains valuable soil nutrients. Its use may be limited if the sludge is contaminated by heavy metals, organic chemicals, other toxic materials, bacteria, viruses, or other undesirables, or if land is unavailable. Legislation also plays a significant role. In many countries, contamination by undesirable components has led to an unfortunate general ban on sludge use in agriculture. Sludge should in fact be used on farmland provided that its content is well monitored and controlled within acceptable limits. A particularly crucial need in this respect is control of industrial discharges to the municipal sewer system. Depending on the type of agricultural use, sludge may need to be disinfected to prevent microbial pollution.

Sludge incineration is becoming increasingly common in Western countries—despite its substantial cost—because landfill sites are lacking and agricultural use is prohibited. Disadvantages of incineration include the potential for air pollution and the problem of ash disposal.

Treatment requirements for sludge depend on its final disposal. Thickening and dewatering (to achieve at least 20% dry solids) are required for land disposal. Additional stabilization and disinfection is a prerequisite for use in farming. If incinerated, sludge must be thickened, dewatered, and dried. Incineration itself can be thought of as both a treatment and a disposal process.

Lack of adequate sludge handling is one of the major shortcomings in CEE countries including Slovakia. This situation developed in the past mainly because the flocculants used in sludge treatment were expensive and were often purchased with scarce hard currency. The solution of the sludge problem is rather straightforward today if money is available. Limited construction would be required to accommodate sludge treatment in existing plants and dewatering equipment (centrifuges, belt-filter presses, etc.) is available and thus can be replaced quickly. Flocculants of good quality are now being produced in several CEE countries.

Thus, the major strategic question concerns instituting industrial pretreatment. This is the precondition for using sludge in agriculture. Agricultural use is the most cost-effective solution for sludge disposal except in large cities, where incineration appears to be the best alternative. In the absence of proper pretreatment, the cadmium and chromium content of the sludge can exceed 100 mg/DS and 1000 mg/DS respectively—one to two orders of magnitude greater than realistically set standards for agricultural use.

11.3 Removal Rates and Costs of Technology Alternatives

Five different groups of treatment process combinations can be considered in light of past developments and current needs in Slovakia and the Nitra basin:

(1) Mechanical (primary) treatment
(2) Chemical treatment

(3) Biological treatment
(4) Biological/chemical treatment without nitrogen removal
(5) Biological/chemical treatment with nitrogen removal

Except for mechanical treatment, two variations on the process are considered for each of the groups. They are described in Table 11.1, and corresponding treatment efficiencies (for BOD-5, SS, TP, and TN) and sludge production rates are summarized in Table 11.2. Two influent wastewater scenarios were considered based on a specific pollution load (BOD-5 = SS = 62.5 g/cap/d, TP = 3.0 g/cap/d, and TN = 12.0 g/cap/d) and two hydraulic loads (250 l/cap/d and 400 l/cap/d). Approximate, "generic" cost data based in Western European experiences can be found in Ødegaard and Henze (1993) for three capacities (2000, 10000, and 100000 P.E.). It is noted that U.S. data are rather similar. Sludge treatment costs are also given by the same authors for three options: dewatering only, anaerobic stabilization and dewatering, and dewatering and incineration.

Although absolute costs in Central and Eastern European countries are still somewhat lower, the cost ratios of various methods are practically the same. Thus, the above data can be used as good approximations for relative costs as a basis for technology selection.

A quick analysis of the cost data illustrates that:

- the treatment costs vary approximately in a range of 1:3 depending on treatment goals and levels
- the unit cost of nutrient removal is about an order of magnitude higher than that of BOD (see below)
- the effect of economies of scale is significant (note that when the size of the treatment plant is selected the economy of transportation costs should also be accounted for)

A more detailed evaluation follows based on cost data of Ødegaard and Henze (1993) and other estimates. The outcome of this evaluation is a slightly modified set of treatment alternatives and cost estimates which are used in further analyses in this report.

11.4 Comparison of Various Cost Estimates

The following three issues are considered in this section:

(1) the relative cost of different technology alternatives which has significant impact on the effectiveness of multi-stage treatment plant development (i.e. if a plant is not constructed in a single step due to budgetary restrictions)
(2) the effect of treatment plant scale
(3) the range of absolute cost estimates

(1) Relative investment costs of technology alternatives.
Technologies can be developed stepwise over time. For instance, if there is an existing primary treatment plant, it can be upgraded to chemically enhanced or biological treatment. The upgrade decision depends primarily on the investment cost of the proposed upgrade, its

Table 11.1 Treatment alternatives

P — PRIMARY TREATMENT

Pretreatment (screening and grit removal) and settling
Overflow rate settling tank at q_{design} = 2 m/h

CEPT — CHEMICALLY ENHANCED PRIMARY TREATMENT

Pretreatment (screening and grit removal), chemical addition (main coagulant plus flocculant), and settling.
Low dosage of main coagulant ≤ 50 mg/l iron chloride
Polymer dosage for coagulation/flocculation ≤ 1 mg/l
Overflow rate settling tank at q_{design} = 3 m/h

PC — PRIMARY PRECIPITATION CHEMICAL TREATMENT

Pretreatment (screening and grit removal), chemical addition of main coagulant, mechanical flocculation (stirring), and settling.
High dosage of main coagulant = 150 mg/l iron chloride
No polymer (stirring in flocculation tank)
Overflow rate settling tank at q_{design} = 1.5 m/h

B1 — BIOLOGICAL TREATMENT (HIGH LOAD)
(Activated sludge with F/M = 0.5 kg BOD5/kg SS/day)

Pretreatment (screening and grit removal), activated sludge process, and settling.
Organic sludge load, aeration tank = 0.5 kg BOD7/kg SS/day
Overflow rate settling tank = 1.0 m/h

B2 — BIOLOGICAL TREATMENT (LOW LOAD)
(Activated sludge with F/M = 0.5 kg BOD5/kg SS/day)

Pretreatment (screening and grit removal), pre-settling, activated sludge process, and post-settling.
Organic sludge load, aeration tank = 0.2 kg BOD7/kg SS/day
Overflow rate settling tank = 1.0 m/h

BC1 — BIOLOGICAL/CHEMICAL TREATMENT
(Simultaneous precipitation in normally loaded activated sludge plant)

Pretreatment (screening and grit removal), activated sludge process with coagulant addition (simultaneous precipitation), and settling.
Organic sludge load, aeration tank = 0.2 kg BOD7/kg SS/day
Overflow rate settling tank = 1.0 m/h

BC2 — BIOLOGICAL/CHEMICAL TREATMENT
(Pre-precipitation followed by normally loaded activated sludge plant)

Pretreatment (screening and grit removal), coagulant addition, flocculation settling (pre-precipitation), activated sludge process, and post-settling.
Organic sludge load, aeration tank = 0.2 kg BOD7/kg SS/day
Overflow rate pre-settling tank = 1.5 m/h
Overflow rate post-settling tank = 1.0 m/h

BCDN1 — BIOLOGICAL/CHEMICAL TREATMENT WITH N-REMOVAL
(Predenitrification/simultaneous precipitation in activated sludge plant with total sludge age = 13 days)

Pretreatment (screening and grit removal), activated sludge process designed for nitrification and including predenitrification zone (recirculation system), coagulant addition (simultaneous precipitation), and final settling.
Total sludge age in activated sludge tank = 13 days
Residence time in activated sludge tank = 15 hours
Overflow rate settling tank = 1.0 m/h

BCDN2 — BIOLOGICAL/CHEMICAL TREATMENT WITH N-REMOVAL
(Pre-precipitation followed by biofilm process with post-denitrification and external carbon source)

Pretreatment (screening and grit removal), coagulant addition, flocculation, settling, biofilm process (submerged biofilter) for nitrification and denitrification, methanol addition for denitrification, and final settling.
Overflow rate pre- and post-settling tank = 1.0 m/h
Residence time in biofilm reactor = 3 hours

Table 11.2 Effluent concentrations, removal rates, and sludge production for technology alternatives.

	BOD5			Suspended Solids			Total Phosphorus			Total Nitrogen			Sludge Production		
	Percent	Effluent Conc.		Percent	Effluent Conc.		Percent	Effluent Conc.		Percent	Effluent Conc.		Sludge Conc.		Dry
	Removal	250 l/d/P.E.	400 l/d/P.E.	Removal	250 l/d/P.E.	400 l/d/P.E.	Removal	250 l/d/P.E.	400 l/d/P.E.	Removal	250 l/d/P.E.	400 l/d/P.E.	250 l/d/P.E.	400 l/d/P.E.	Solids
Influent Wastewater	0	250	150	0	250	150	0	12	7.5	0	48	30	–	–	–
Mechanical (P)	30	175	105	60	100	60	15	10	6.5	15	40	25	125	75	4
Chemical															
High load (PC)	50	125	75	80	50	30	70	3.6	2.5	25	36	23	250	200	3
Low load (CEPT)	70	75	45	90	25	15	90	1.2	0.8	30	34	21	350	300	3
Biological (B)															
High load activated sludge (B1)	70	75	45	80	50	30	30	8.4	5.3	25	36	23	185	125	2
Low load activated sludge (B2)	~90	20	20	90	25	15	30	8.4	5.3	30	34	21	205	125	2
Biological/chemical (BC)															
Simultaneous precipitation (BC1)	~90	20	20	~90	20	20	~90	1.0	1.0	35	31	20	250	160	2
Pre-precipitation (BC2)	~90	10	10	~95	15	15	~95	0.5	0.5	35	31	20	380	330	2
Biological/chemical with N removal (BCDN)															
Pre-denitrification with simultaneous precipitation based on activated sludge (BCDN1)	~95	10	10	~97	10	10	~90	1.0	1.0	70	15	10	275	180	1.5
Post denitrification with pre-precipitation based on biofilm process (BCDN2)	~97	5	5	~97	10	10	~95	0.5	0.5	85	7.5	5	380	330	2.5

Concentrations in mg/l except sludge which is in grams dry solids per cubic meter.

cost-effectiveness expressed, for example, in USD/kg BOD removed (see below), and the water quality improvement achieved. Thus, one needs to know the ratio of costs of technologies, which can be consecutive elements in a multi-stage development. A summary is provided in Table 11.3 which illustrates costs of six technologies relative to traditional activated sludge biological treatment (including primary treatment). Costs of sludge treatment (anaerobic stabilization and dewatering) are included. The comparison is made for a 100000 population equivalent plant. Data were used from Ødegaard and Henze (1993), Harleman and Murcott (1993), the U.S. EPA control technology cost digest (1984), the pre-feasibility study of the Vistula River basin (SWECO, 1992), the Danube study (USAID/WASH, 1992a), Roman (1992), and Hungarian design practice.

Table 11.3 Comparison of relative investment costs of different treatment plants (100,000 P.E. domain)

Source	P	CEPT	PC	B	BC	BCDN	Comments
(1) Odegaard and Henze (1993)	0.61	0.74	0.89	1.00	1.00	1.45	Mean for BC and BCDN
(2) Harleman and Murcott (1993)	0.43	0.53	-	1.00	-	-	Mean values
(3) U.S. EPA	-	-	-	1.00	1.20	-	
(4) Danube	0.44	0.50	-	1.00	1.02	1.37	
(5) Vistula	-	-	0.77	1.00	1.05	1.60	
(6) Roman (1992)	0.55	0.65	0.80	1.00	-	1.70	Estimates from cost vs. BOD removal rate function
(7) Hungarian practice	0.50	0.57	-	1.00	-	-	
(8) Estimate Accepted	0.55	0.59	0.73	1.00	1.00	1.45	Mean

P - primary treatment, CEPT - chemically-enhanced primary treatment,
PC - primary precipitation plant, B - primary and biological treatment (low load),
BC - biological/chemical treatment, BCDN - biological/chemical treatment with denitification

As can be seen, the cost estimates deviate from each other by 10 to 15% depending on the technology, local conditions, whether internal infrastructure is included (approximately 25 to 30% of the total investment cost), and several other factors. Subsequent to careful analysis, the estimates of line (8) were accepted for developing treatment alternatives for policy purposes.

(2) The effect of scale

A comparison of the effects of scale is presented in Figure 11.1. In addition to previously cited sources, the figure uses results of the Oder/Odra River basin pre-feasibility study (BCEOM, 1992) and the Czech and Slovakian studies (Grau and Nesmerak, 1993, and Námer and Hyanek, 1993). As can be seen, there is good overall agreement between the various estimates with somewhat greater scatter for plants of less than 40000 P.E.

(3) Comparison of various cost estimates

The expectation is that estimates will vary substantially depending on the country, local conditions, degree of price competition, and so forth. A summary is given in Table 11.4. As can be seen, the first four values are similar. Local estimates obtained from the Slovak and Czech Republics show lower unit costs, reflecting the exclusion of internal infrastructure, different and changing price structures, and market conditions.

Table 11.4 Comparison of unit treatment costs (100,000 P.E., traditional primary and biological treatment

Source	Unit Investment Cost (USD/Cubic Meter)	OMR Cost (USD/Cubic Meter)	Comments
Odegaard and Henze (1993)	1.65	0.11	
Harleman and Murcott (1993)	1.77	0.13	Average values
Vistula Study	1.26	0.09	
Hungarian practice	1.35	-	
Namer and Hyanek (1993)	0.73	0.07	Excluding internal infrastructure
Gran and Nesmerak (1993)	0.71	-	

11.5 A Summary of Treatment Technologies

Among the technologies outlined before, we selected only those which include a primary sedimentation tank. Primary sedimentation is needed because combined sewer systems are prevalent in Slovakia (and the CEE countries), leading to high solids content in the raw wastewater and the need for solids removal. The activated sludge process was assumed for all biological treatment plants, as this is a widely used methodology in all of the countries at present. The technologies chosen are all well-designed. None would be considered to be 'high-tech," and all are relatively simple to operate and therefore do not require unrealistic levels of training.

The summary of treatment methods including removal rates, effluent concentrations, and costs is given in Table 11.5. The individual technologies are all discussed above with the exception of BC1DN and BC2DN. These are the same as BC1 and BC2, but with only partial denitrification and thus lower costs than BCDN (Table 11.4).

Table 11.5 Unit costs of different treatment technologies.
(100,000 P.E.; BOD5 = SS = 250 mg/l, TP = 12 mg/l, TN = 48 mg/l in influent; q = 400 l/cap/d)

Technology	BOD % mg/l	SS % mg/l	TP % mg/l	TN % mg/l	Unit Investment Cost [USD/cubic meter]	Unit OMR Cost [USD/cubic meter]	Process Combination
(1) P	30 175	60 100	15 10	15 40	0.90	0.064	Primary treatment
(2) CEPT	55 113	80 50	75 3	25 34	0.98	0.103	Chemically enhanced primary treatment (low dosage)
(3) PC	70 75	90 25	90 1.2	30 34	1.20	0.122	Primary precipitation (high dosage)
(4) B	90 25	90 25	30 8.4	30 34	1.65	0.106	Primary treatment and activated sludge (low load), ASP
(5) BC1	90 25	90 25	90 12	35 31	1.54	0.143	CEPT + B (ASP)
(6) BC2	95 12	95 12	95 0.6	35 31	1.73	0.165	PC + B (ASP)
(7) BC1DN	95 12	90 25	90 1.2	60 19	1.92	0.168	CEPT + B + partial denitrification
(8) BC2DN	95 12	95 12	95 0.6	60 19	2.11	0.200	PC + B + partial denitrification
(9) BCDN	97 7	95 12	95 0.6	85 7	2.37	0.210	PC + B + denitrification

Note that all the technologies incorporate a primary sedimentation tank, and, for biological treatment, the activated sludge process (ASP) was assumed.

In the course of preparing the cost estimates, down-sizing of the primary sedimentation tank was considered for the CEPT and PC processes due to their higher overflow rates (3 m/h versus 1.5 m/h). Lower costs than estimated may be possible, but cost calculations would require more detailed analyses of design standards and methodology with special attention to handling stormwater overflows. Costs obviously incorporate increased sludge treatment as a result of adding chemicals which enhance BOD, SS, and TP removal. Costs of technologies BC1 and BC2 involve moderate down-sizing of the aeration tank due to increased BOD removal by the upgraded first stage (this is why, for instance, BC1 is less expensive than B). The actual down-sizing can be greater. It depends on the type of chemical treatment (low or high dosage) and the P removal required. PC allows greater reduction in the volume of the aeration tank; however, increased sludge treatment cost can counteract this effect. Thus a detailed design is needed for each individual case.

It is noted that for chemical upgrading of existing treatment plants all the time pre-precipitation will be assumed with low dosage (see Table 11.6 and Appendix 11.1). In Section 11.9 low load activated sludge method (with nitrification) will be denoted by B_n, while partial denitrification will be also specifically identified (thus e.g. B_nCDN_p is a biological/chemical treatment with nitrification and partial denitrification).

Sludge treatment costs are included in Table 11.5; anaerobic digestion and dewatering were assumed for all treatment technologies. In the forthcoming analyses, the values in Table 11.4 were employed together with Figure 11.1 to represent the economic effect of treatment plant scale.

11.6 The Application of CEPT to Upgrade Existing Treatment Plants

As was demonstrated in Chapter 6, there are many plants with overloaded secondary treatment in Slovakia. Chemical enhancement is an attractive, cost-effective method to upgrade these plants, and it is therefore discussed here in more detail (see also Section 11.7 for a discussion on some of the treatment plants in the Nitra River basin).

CEPT has been used frequently in the U.S. during the last 15 years to retrofit existing primary treatment plants. The main goal has been to increase the BOD removal rate and thus the chemical dosage is small. In contrast, in Scandinavian countries a dosage that is four to five times higher is applied in order to meet stringent effluent standards for TP.

U.S. experience with CEPT has shown significant improvements in BOD, SS, and TP removal rates compared to conventional primary treatment (Table 11.5). Surface overflow rates can be practically doubled to 3 to 4 m/h.

The amount of sludge produced is 1 kg solids/kg TSS removed for primary treatment and approximately 50% more for CEPT due to increased removal of BOD, TSS, and the chemical dosage. Whereas for conventional primary treatment, the settled solids consist of only the TSS removed, for CEPT the settled solids include both the solids captured and the added chemicals. The amount of sludge can be estimated by stoichiometry as a function of influent and effluent concentrations of TSS and TP and the type and concentration of metal salt added. The U.S. experience shows that the specific sludge production by CEPT is approximately equal to that of an activated sludge plant (roughly 1.8 kg solids/kg TSS removed or 1 kg

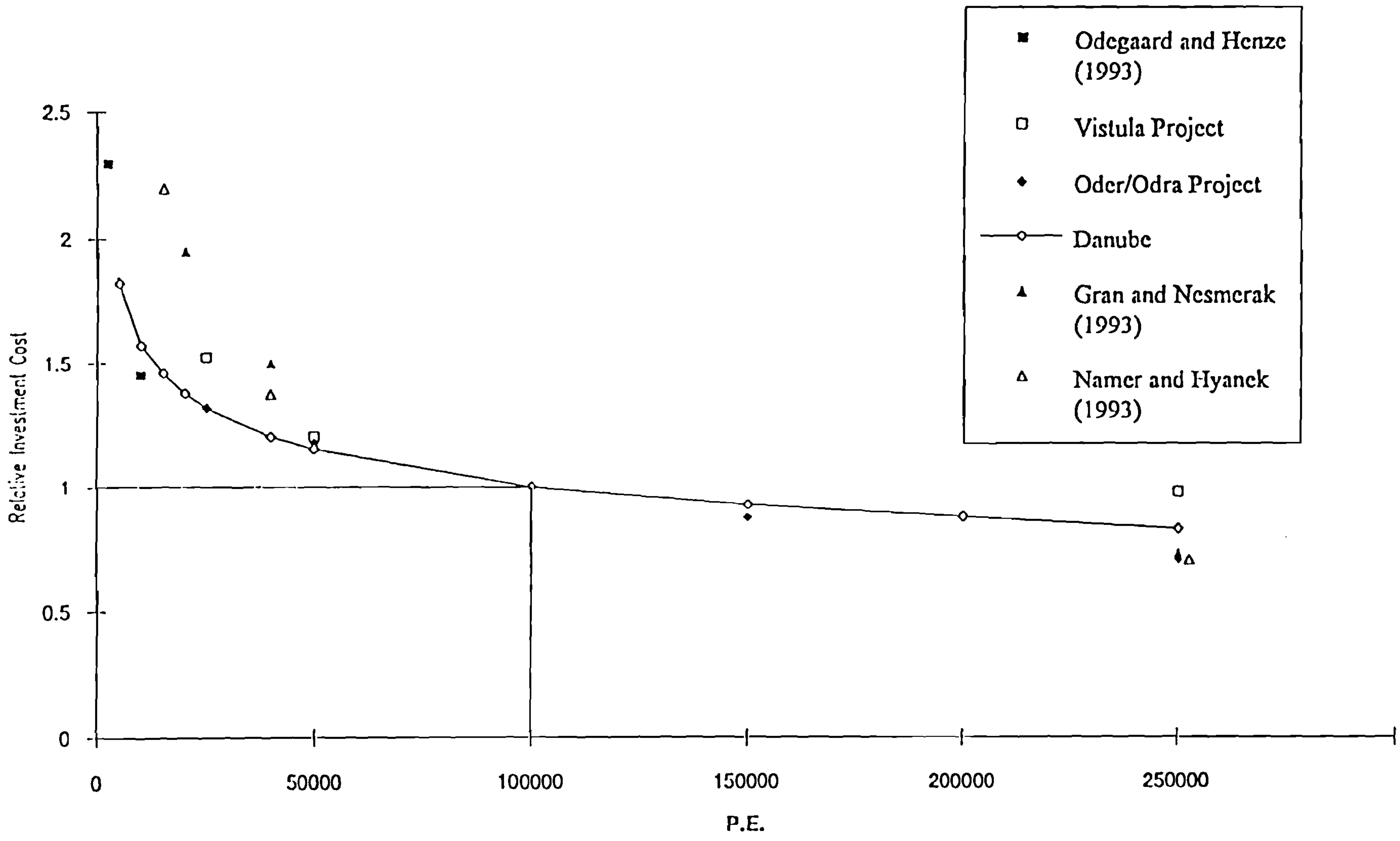


Figure 11.1 The effect of scale on the cost of traditional biological wastewater treatment

solids/kg BOD5 removed), and the contribution of low dosage chemicals used is about 10-15%.

Due to increases in the surface overflow rate and BOD removal efficiency in primary treatment as a result of low dosage chemical addition, the capacity of highly overloaded treatment plants can be practically doubled with only minimal investments. The limiting factor of the extension is generally formed by the final clarifier and sludge treatment where upgrading methods cannot really help and new units can be needed irrespective of the method of plant upgrading.

11.7 Cost-Effectiveness and Multi-Stage Development

Cost-effectiveness is usually expressed in terms of cost per mass of pollutant removed, where cost may be either the total annual cost (TAC) and/or the investment cost (IC). TAC is the sum of the annualized investment cost and the annual operations, maintenance, and repair cost (OMRC):

$$TAC = (CRF) * IC + OMRC$$

where the capital recovery factor is:

$$CRF = \frac{r(1+r)^T}{(1+r)^T - 1}$$

T is the economic life of the project, and *r* is the interest rate (which may be replaced by an apparent interest rate if the effect of inflation is also considered).

The multi-objective nature of cost-effectiveness is obvious: if money is scarce the indicator IC/kg mass removed is much more important than TAC/kg mass removed. This is the situation currently faced in Slovakia, the Nitra basin, and the CEE.

The problem is also multi-objective in the sense that different treatment technologies serve different purposes. For instance, the goal of traditional biological treatment is BOD removal (with or without nitrification). Chemically enhanced treatment (CEPT) also efficiently removes phosphorus while in a primary precipitation plant (PC), phosphorus removal is considered more important than BOD removal. BCDN treatment (Table 11.4) more or less balances the attention given to BOD, P, and N removal. Thus, when evaluating cost-effectiveness, appropriate weights should be applied to the different pollutant removal capabilities.

On the basis of a detailed analysis performed by Somlyódy (1993) the following conclusions can be drawn on cost-effectiveness.

- Primary treatment alone is expensive.
- Traditional biological treatment is more effective, but investment requirements are high.
- CEPT, PC, and BC are attractive from the standpoint of both investment cost and total annual cost.

- Unit removal costs of nutrients are much higher than for BOD (this particularly applies to N).
- The upgrading of a primary plant to a chemically enhanced one is more than three times more effective than upgrading to a traditional biological treatment plant (however, BOD effluent quality remains poorer). Similarly, the upgrading of a biological plant to a biological/chemical one is a cost-effective action.
- Cost-effectiveness can be further increased if treatment plants are built in a multi-stage fashion (by minimizing the present value of future expenditures).
- As the technical realization of multi-stage development is considered an existing primary treatment plant can be upgraded to CEPT (or PC), or a new CEPT plant can be constructed as a first step. The second step is an extension to a biological/chemical plant. (It is the first step if a biological plant exists). The final stage is the addition of a post-denitrification unit (or the introduction of partial denitrification), depending on receiving water quality needs.

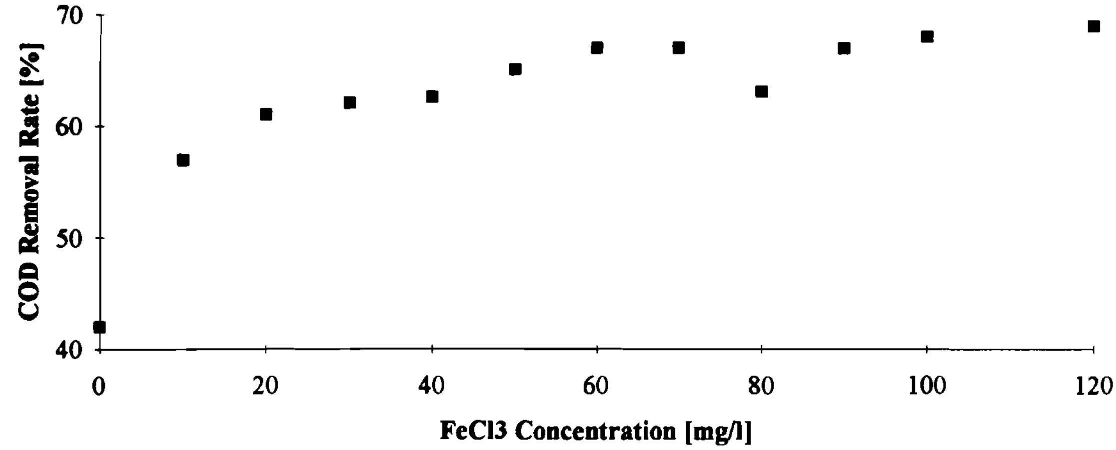
11.8 Experiments to Test the Applicability of Chemical Upgrading in the Nitra River Basin

To test the applicability of low dosage chemical upgrading, so called "jar tests" were performed during the summer of 1993 at three overloaded municipal WWTPs, namely Nitra, Nové Zámky and Topolcany (see Chapter 6). The jar test equipment is actually a (number of) stirred reactor(s) where the speed of the paddle mixer (and its sequence) can be adjusted/changed to resemble the major parameters of the primary treatment unit. Samples for the test are generally taken from the raw water, and chemicals are added in the dosage range to be tested. A generic procedure consists of three stages: a rapid mix stage when the primary coagulant (typically a metal salt) is added, a flocculation stage with slow mixing, and a settling phase with no mixing. The duration of these consecutive stages is about 0.5, 1-5 and 5 minutes, respectively. The analysis made from the sample characterizes the "simulated" effluent quality and removal rate of CEPT for a given dosage. A comparison with those results obtained without adding chemicals indicates the improvement in the performance which can be achieved. Generally, jar test equipment consists of a number of parallel reactors, and thus a large number of tests with differing raw wastewater and chemical compositions, dosages, and mixing procedures can be performed relatively quickly.

Several chemicals were tested with a focus on their local availability and costs. Among others, these included ferric chloride, ferric chloride sulfate, ferrous sulfate and alum, with and without polymers. The dosage was varied systematically in the 0-200 mg/l domain for metal salts and the 0-1.0 mg/l range for polymers. Routine parameters incorporated BOD5, COD, SS, color, and turbidity.

A sample result is shown in Figure 11.2 for Nové Zámky WWTP. The plot illustrates the COD removal rate of the "primary unit" as a function of the dosage of $FeCl_3$ (alum has a similar impact and polymers--even in a rather small amount--further improve the process efficiency). From the figure and the experiences collected, the following conclusions can be drawn.

Figure 11.2 Results of jar tests for the Nove Zamky WWTP (19 June 1993, no polymers added) Raw COD = 630 mg/l



(1) The results obtained are similar to the U.S. ones (see e.g. Morrissey and Harleman, 1990). Although the zero dosage removal rate is higher than the average value monitored at the WWTP (most likely due to the irregular sludge return from the final clarifier to the primary one), the experiment suggest that about a 20% increase in COD (and BOD5) removal rate is achievable.
(2) Fig 11.1 shows a saturation nature. This means that it is not feasible to increase the dosage above 50 mg/l (and even 20-30 mg/l addition can be sufficient).
(3) Similar results were also obtained for the other WWTPs but are not discussed here (see Murcott, 1994).

Jar tests are obviously not enough to make a decision for upgrading a treatment plant. They should be used as a first step in the evaluation and are restricted to the primary sedimentation basin. The next step is a full scale experiment, assuming that the laboratory tests lead to promising results. Such an undertaking was not in the frame of the present study, although some preparations were made.

For the purpose of illustration, we summarize conclusions of a full scale test performed at the North Budapest WWTP (140000 m^3/d design capacity; mechanical-biological plant with high load activated sludge process) subsequent to similar jar test results as shown in Figure 11.2. One of the treatment trains was hydraulically overloaded by about 70%, and 25-30 mg/l $FeClSO_4$ (available locally) was added. The effluent quality remained unchanged in spite of the increased flow, except for TP and heavy metals which improved. The dissolved oxygen level in the aeration basin increased under non-altered compressor operation. This clearly indicates the increased removal of organic material in the primary settling tank. Sludge production did not show detectable changes under the limitation of the available measurement methodology. Final clarifiers (designed as conservative as in Slovakia) were also able to cope with the increased flow, and in fact, the plant became less vulnerable against shock loads than before (see Murcott et al, 1994, for further details).

In summary, it is concluded that low dosage chemical enhancement is a promising, cost-effective upgrading method for highly overloaded mechanical-biological treatment plants.

11.9 Development of Treatment Alternatives for Municipalities in the Nitra River Basin

The basis for the development of the approximate treatment alternatives is actually a number of technological calculations together with the estimation of costs for each plant. The starting point is the existing treatment scheme (see Appendix 6.1 for an example) with all the important geometric, hydraulic, and other information. The procedure is summarized in a step by step fashion as follows.

(1) Selection of the flow of the upgrade and /or new design is required. Here we assumed that a small decrease will take place rather than an increase (see above).
(2) Selection of effluent standards or technological principles for the upgrading/design is the next step. The options are rather large; traditional biological treatment with various effluent criteria is one of them. Another is the combination of biological and chemical processes as discussed earlier. Further alternatives can be obtained depending on the decision on the future utilization of existing facilities (demolishing the current plant and the construction of a new plant, full upgrading to the required capacity, upgrading to a feasible capacity and the addition of a new (smaller) plant, etc.).
(3) The estimation of the maximum capacity of each treatment unit (or new design) is then required. For upgrading, a decision is needed on the type of changes. For a biological enhancement, new volumes for the clarifiers and the aeration basin (with oxygen input) as well as an analysis of the sludge line is needed. For chemical upgrading, no changes are generally needed in the size of the primary clarifier and the aeration unit (chemical storage, transportation, dosage, and mixing devices, however should be solved and implemented), but the final clarifier and sludge processing units may require adjustments. Further alternatives are developed if we combine a chemical dosage and a volume increase of the biological unit, simultaneously, to achieve, for instance, nitrification.
(4) For a new design the situation is relatively straightforward, since data for the river basin planning can be obtained directly on the basis of Table 11.4 and Figure 11.1 for both effluent quality and costs (including IC and OMRC). In contrast, an upgrading necessitates more detailed calculations, in the course of which cost data of Table 11.4 are used in a sequential fashion. Here, among others, approximate unit specific cost of upgrading from a given level of treatment (e.g. primary one) to the desired one is utilized. This step requires the estimation of the amount of chemicals to be added, the increase in the sludge production, the volume of possible new units and the costs implication of all these. The information which can be used at this phase incorporates, in addition to the summary of this chapter, details of cost estimates from Ødegaard and Henze (1993), conventional procedures to design activated sludge plants, and information on the costs of individual units (e.g. settling and aeration basins) and facilities of chemical enhancement.

An example for the above procedure is given in Table 11.6 for Nové Zámky. The table summarizes costs and effluent qualities of ten altogether different alternatives, and it is used directly by the water quality control policy model (see Chapters 10 and 13). Among the alternatives listed in Table 11.6, the first two correspond to the no-treatment ("0" level) option and the current treatment option, respectively. The third one is biological upgrading (B), which is relatively inexpensive in this case because clarifiers had already been added to the plant. BOD-5 removal efficiency improves significantly as a result of this action. The next alternative applies to chemical upgrading (BC with pre-precipitation). It gives a marked improvement in phosphorus removal, but nitrification cannot be achieved. This necessitates the increase in the aeration basin volume (B_nC corresponding to low load, line 4). Option 5 incorporates partial denitrification (B_nCDN_p, which is still relatively inexpensive). Alternatives 6 and 7 utilize the existing plant and its chemical upgrade, respectively, up to a capacity which allows nitrification to treat a portion of the wastewater. The rest is assumed to be processed in a newly-constructed plant ("B+new", and "B_nC+new", respectively) Alternative 8 adds partial denitrification to the previous option, and finally, the last alternative is a new WWTP satisfying the most stringent requirements of the European Community. Among the technologies listed in Table 11.6 options 4 through 9 satisfy requirements of the Slovak legislation.

Table 11.6 Characteristics of the Feasible Treatment Alternatives for Nove Zamky WWTP

Alternative	Cost (10^6 USD)		Effluent Concentrations (mg/l)				Type of technol.
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	240	11	48	0	-
1	0	1.0	60	9	40	0	B
2	3	1.2	15	7	4	30	B
3	2	1.4	20	1.5	34	0	BC
4	5	1.6	15	1.0	4	30	B_nC
5	6	1.9	15	1.0	2	17	B_nCDN_p
6	13	1.2	15	7	4	30	B+new
7	11	1.6	15	1	4	26	B_nC+new
8	14	1.9	15	1	0	19	B_nCDN_p+ new
9	21	2.0	10	0.8	0	10	new B_nCDN_p

Alternatives for other municipalities are summarized in Appendix 11.1 in a similar fashion. The number of technology options depends on the level and development of existing facilities. For example, it is not more than four for Partizanske (new plant) and for small plants such as in Vrable or Lehota. The WWTPs of Nitra and Topolcany, however, are characterized by even larger numbers of alternatives than Nové Zámky, as their problems and shortcomings are practically identical.

It is stressed that the above estimates are used for river basin planning (see Chapter 13 for details) for which the approximations employed are acceptable. Design on the treatment plant level, however, obviously requires a more detailed engineering analysis (with possible and desired feedback to the planning and strategy level).

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12. PROTOTYPE DECISION SUPPORT SYSTEM

12.1 Introduction

One of the outputs of this project is a prototype decision support software (PDSS) which significantly eases the application of advanced water quality modeling and analysis tools for policy analysis The system provides the user with an intuitive and user-friendly interface which hides the complexity of underlying tools. However, the user still should possess a basic understanding of the problem and be capable of interpreting the outcomes of the analysis.

A major element of water management decisions is the spatial distribution of pollutant loads and the subsequent spatial distribution of water quality. A recently software development to provide for spatial display and analysis of data is the Geographical Information System or GIS. The GIS allows the storage of related data in layers or coverages that represent spatial information about a particular problem. Several GIS coverages have been developed for the Nitra River basin during this study. The sources of GIS data and their processing are discussed in Section 12.3. GIS software can be commercial purchased and then linked into a DSS. For this PDSS, a GIS interface was developed to display the spatial representation of the major elements of the river system. The GIS interface software is capable of displaying maps and schematic representations of the basin developed by commercial mapping software such as MapViewer (TM), but does not contain all the analysis and display features of commercial GIS.

12.2 The Decision Support Software

The prototype decision support system was developed in the C programming language to provide maximum flexibility and portability. The target operation system is Windows 3.0/3.1 (TM) by Microsoft Inc. (R). for personal computers. Windows 3.0/3.1 is renowned for its rich user interface capabilities, comprehensive set of system functions and low cost. This operating system is also known to have certain deficiencies. The most pronounced limitations are a 16-bit basic data size and its lack of inherent multitasking and network support. The next version of Windows (called Windows NT, for New Technology) was designed to overcome the shortcomings of Windows 3.1 and also to be a cross-platform operating system, usable on personal computers as well as on workstations. Unfortunately, current release of Windows NT requires extensive hardware enhancement over a typical PC is not wide used at the moment. Therefore we did not consider its use for the PDSS.

The PDSS is composed of several modules, each of which is designed to perform a specific task or several related tasks. The modules and their interrelationship are showed in Figure 12.1 and described below. The main functional capabilities of the distinct modules of the software are as follows:

Data management unit:

- Read the tables with data about the river system in question, parse river distances and reconstruct the tributaries tree.

Data transfer unit:

- Transfer the desired data to Microsoft Excel spreadsheet for analysis and display through the Object Linking and Embedding client-server library (OLE).

Display unit:

- Displays the river tree in a symbolic way for checking purposes and for visibility. The display unit is capable of handling the vector files produced by the commercial MapViewer mapping software.

Optimization unit:

- Performs the least-cost analysis for the given set of alternatives and constraints (Chapter 10).

Calibration unit:

- Performs the pseudo-random disturbances in coefficients, emissions and observations data, letting the user modify the variability extent
- Performs the water quality routing through the river system network
- Selects/rejects routed trajectories as they pass the measurement points on the basis of user-defined "windows of acceptability"

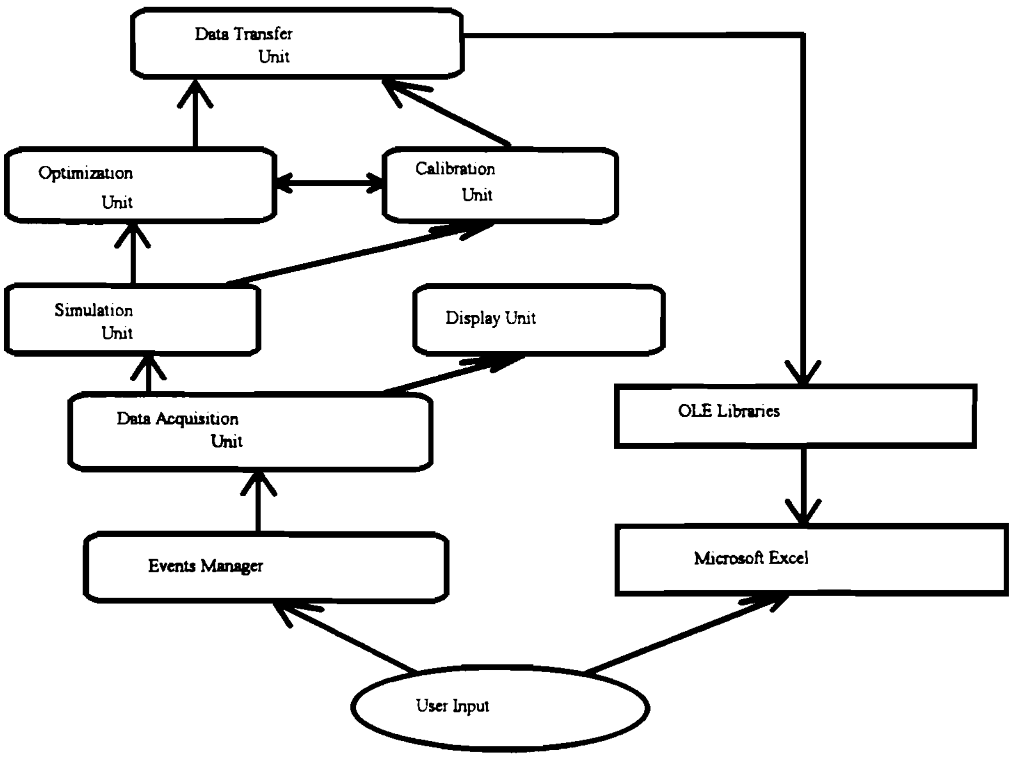
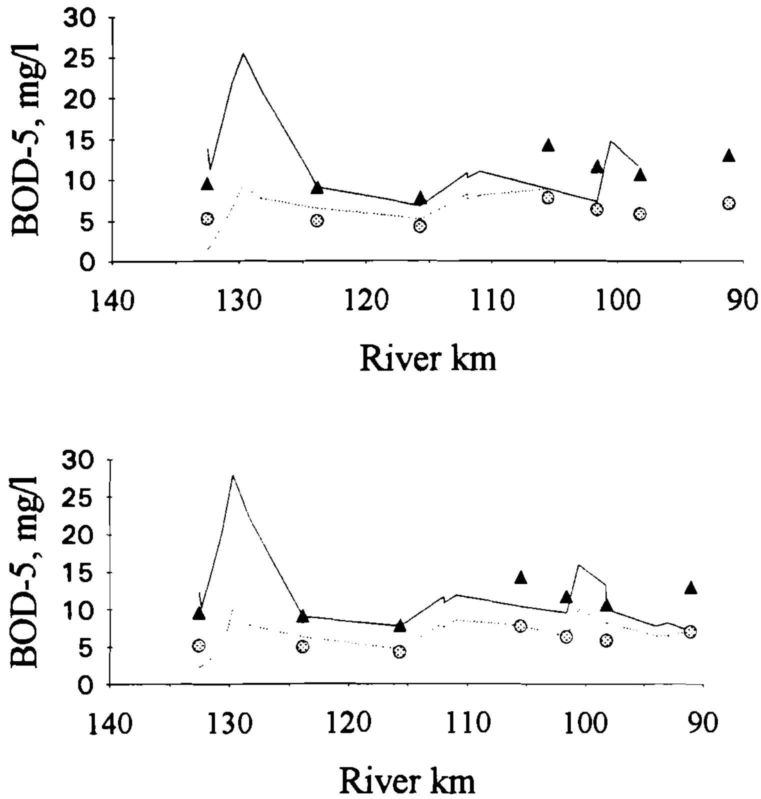


Figure 12.1 Outline of decision support system modules and their interaction with the user.

The data management in the system is aided by a C library implementing basic operations on files with dBase (TM) format. This library is a reasonable compromise between the functionality and overhead. To reduce the software overhead, the library is supplied as a dynamic loadable module.

Microsoft Excel charts are used to display the output of the simulations. The data transfer is facilitated with the Object Linking and Embedding client-server library, which is part of the Microsoft Windows software. Sample charts produced during a calibration of model coefficients are shown in Figure 12.2 (the calibration procedure is described in detail in Chapter 9).



—— and ······ - bounds of the set of accepted simulations (compliant behavior)

● and ▲ - bounds of behavior definition (acceptance windows)

Above: all generated simulations failed to comply with specified behavior definition.

Below: one trajectory passed the behavior definition windows and will participate in the final analysis.

Figure 12.2 Display of Microsoft Excel charts with longitudinal profiles of BOD-5 generated during the HSY procedure

12.3 The Application of GIS

For the purpose of the project, the Base Map of Slovakia (in 1:200000 scale) was digitized by the Water Research Institute (Bratislava) and transformed to appropriate coordinates system.

Geographic coordinates (latitude, longitude) were used as a coordinate system for the decision support software (section 12.2). All the spatial data are also available in UTM coordinate system (zone 34). The main elements of the river basin were digitized by means of the vector module of the ILWIS software developed by ITC Enschede, the Netherlands. The processing of raw digitized data, the building of topology and linking to databases was performed by the use of PC ARC/INFO (developed by ESRI, U.S.A.). Two types of Windows applications were used for the purpose of presentation of all the data:

MapViewer (Golden Software, U.S.A.) - the mapping presentation software which is able to create and store attribute maps;

ArcView (ESRI, U.S.A.) - software developed for the 'user-friendly' presentation of PC ARC/INFO coverages with direct access to attribute tables and the possibility to display an information query.

The digital spatial data of the following elements of the river basin was prepared:

- The boundaries of the Nitra river basin as a part of the Váh river basin
- The boundaries of a partial catchments which divides the Nitra river basin into its smaller parts
- The boundaries of the districts which falls into the Nitra river basin
- The detailed river network; the location of settlements with inhabitants above 2000
- The location of monitoring sites used as a regular (weekly) sampling site
- The sources of pollution caused by municipal waste water discharges
- The sources of pollution caused by industrial waste water discharges
- The location of the surface water intakes
- The location of the groundwater intakes
- The location of the weirs
- The location of sampling sites used in June 1993

Each data layer is linked to the table of additional information in database format .DBF (ARC/INFO) or in spreadsheet format .WK1 (MapViewer). These layers were presented earlier in Chapter 3.

It is possible to use the terrain information derived from Digital Elevation Model (DEM) of the Nitra river basin, which is part of the DEM of Slovakia (grid resolution 100 m) generated by ILWIS raster module. Since MapViewer or ArcView haven't got the ability to handle with raster maps, and raster-vector transformations are memory consuming with poor quality results, the mentioned raster maps are accessible only for presentation purposes in bitmapped format (.BMP). The screen resolution 1024*768 pixels with 256 colors is necessary to display all images properly (Figure 12.3).

12.4 Use of the Prototype Decision Support Software

The user interface of the PDSS is driven with a series of menu. The main system window is used to display the river network in schematic representation. Additionally any number of overlays can be added to illustrate other important objects such as rivers, settlements etc. (Figure 12.4). The information necessary to construct the river network for the analysis is stored in the configuration file, which could be loaded via the File/Open menu selection. After the desired river system is selected and data are processed, the scheme of the river network is rendered and the image can be zoomed or scrolled (Figure 12.4).

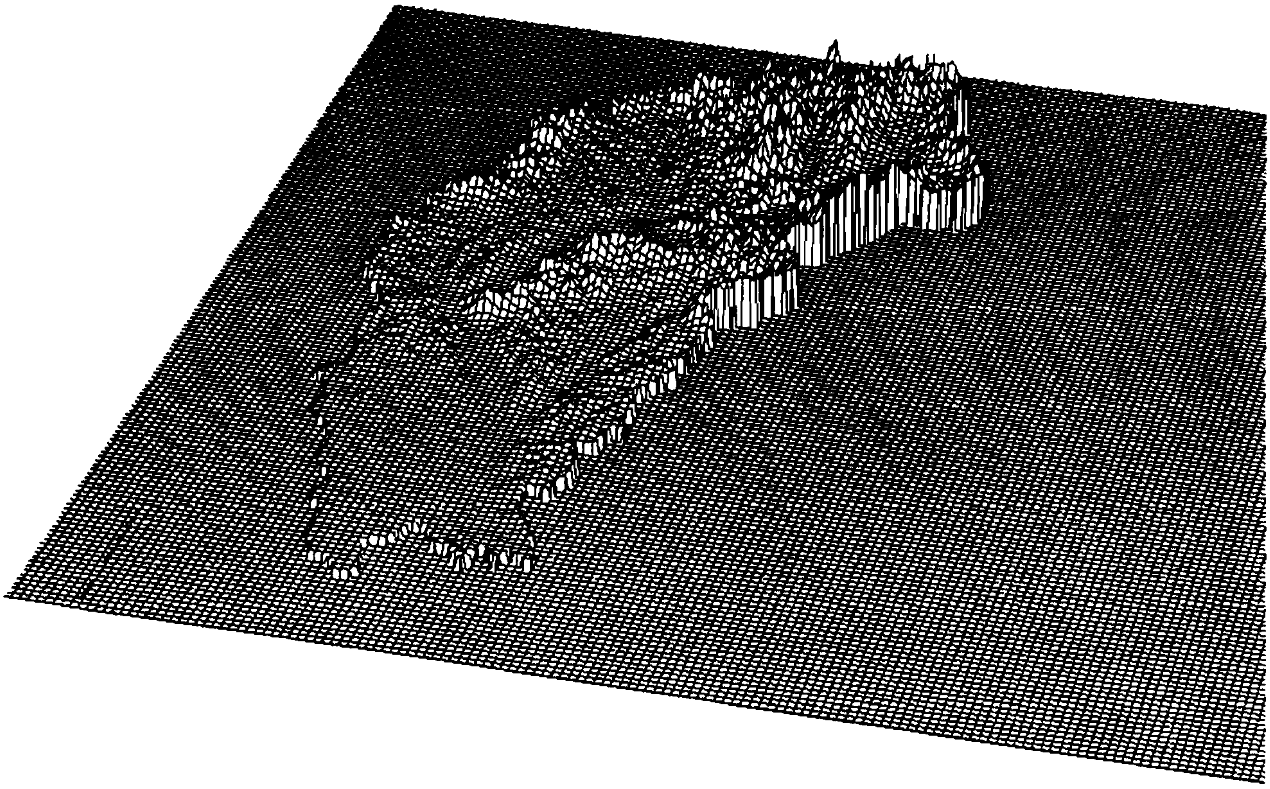


Figure 12.3 The digital elevation model of the Nitra River basin.

The model coefficients are modified via the menu selection "Model Coefficients" (Figure 12.5). For calibration, a coefficient can be set to a constant value, or can be varied within a range (Figure 12.5). Coefficients can also be set to the literature values.

After the desired model and coefficients are selected, the user can begin the calibration procedure (menu selection "Monte Carlo/Monte Carlo simulation for the entire river"). The dialog box contains the controls necessary to set the calibration options such as the margins of uncertainty added to measurements and the acceptance window parameters (Chapter 9). The simulation results are transferred to the Excel table processor via the OLE libraries (Section 12.2). The selected parameters are stored and can be saved with the button "Calibration/Save".

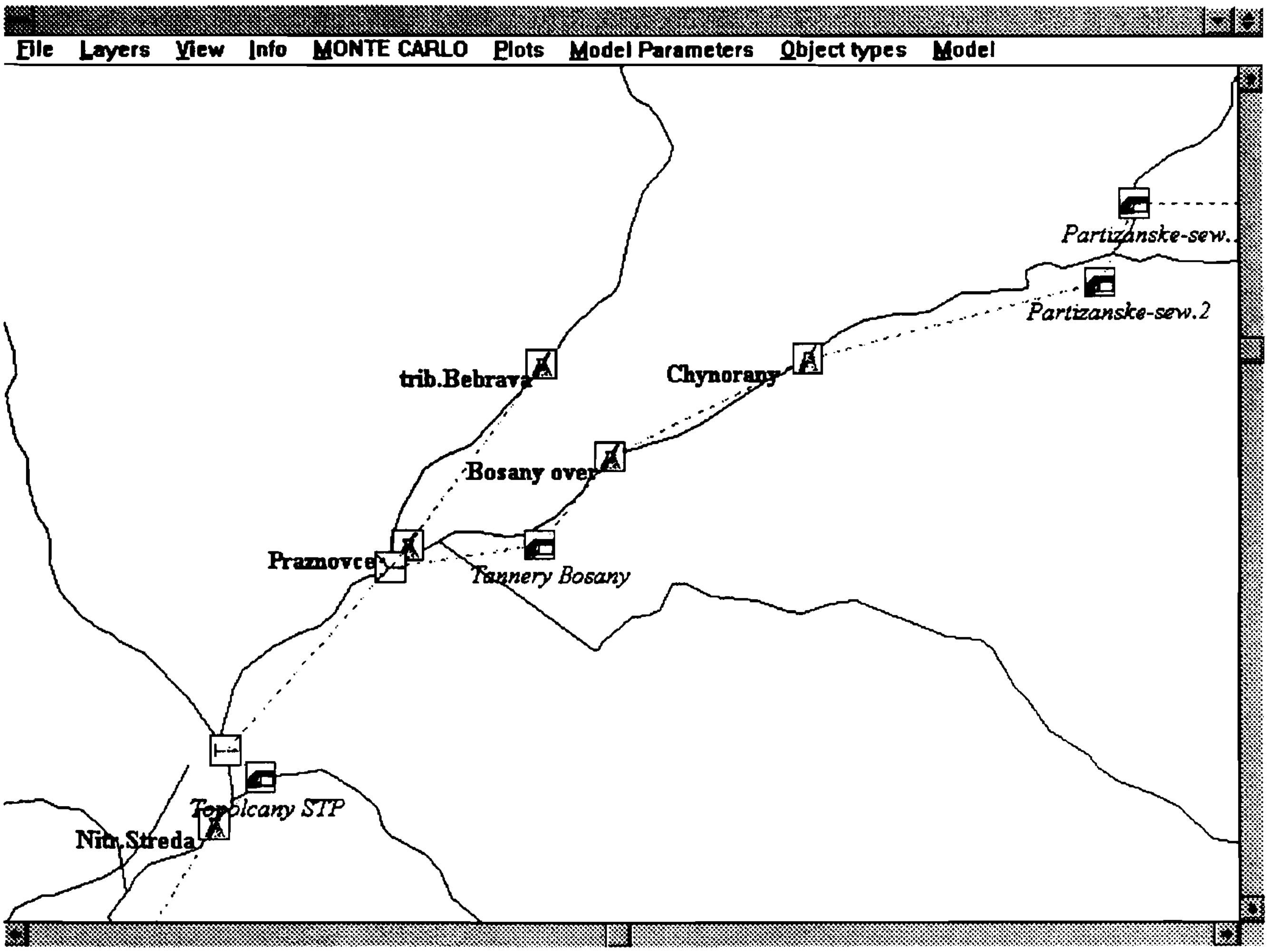


Figure 12.4 The main window of the decision support system software

After calibration the parameters found can be fixed for the policy/scenario analysis (Coefficient/Saved button in the parameters dialog, Figure 12.6). The desired scenario can be loaded (File/Open menu) and optimization for least-cost policy can be performed (Model/Dynamic Programming). The results are again sent to Microsoft Excel table processor and could be analyzed immediately after the simulation.

The set of menu and dialog controls provides a simple and efficient interface for accessing the simulation, calibration and optimization tools and results. It can be used by a person not skilled in programming. However, for setting up river network data and scenarios, understanding results of simulation and selecting policy options, significant background knowledge and expertise in appropriate fields is required.

Further development of the software is planned to include tools for rendering water quality databases, illustration of water quality conditions on the map, and handling more sophisticated water quality models.

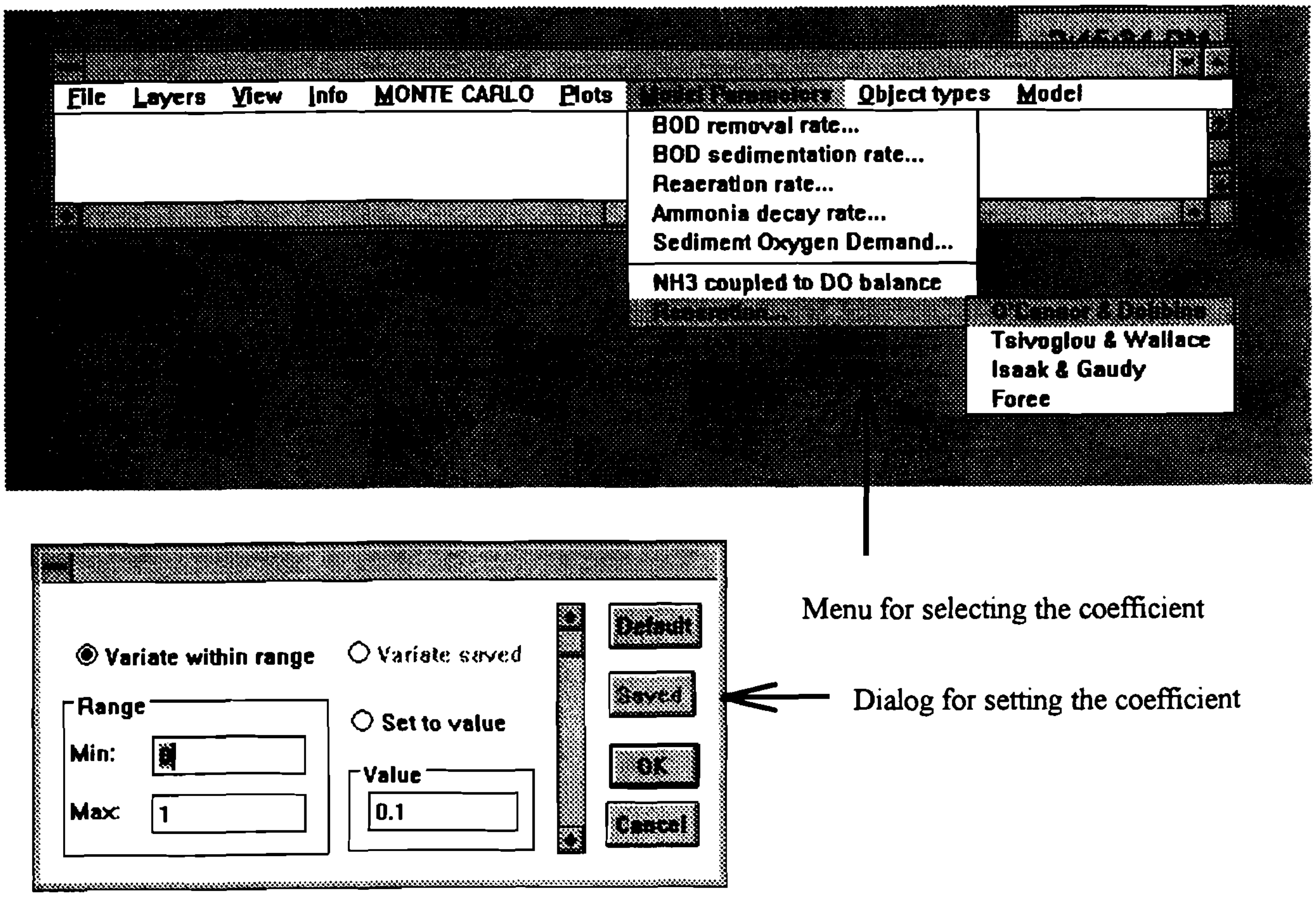


Figure 12.5 Model coefficients menu.

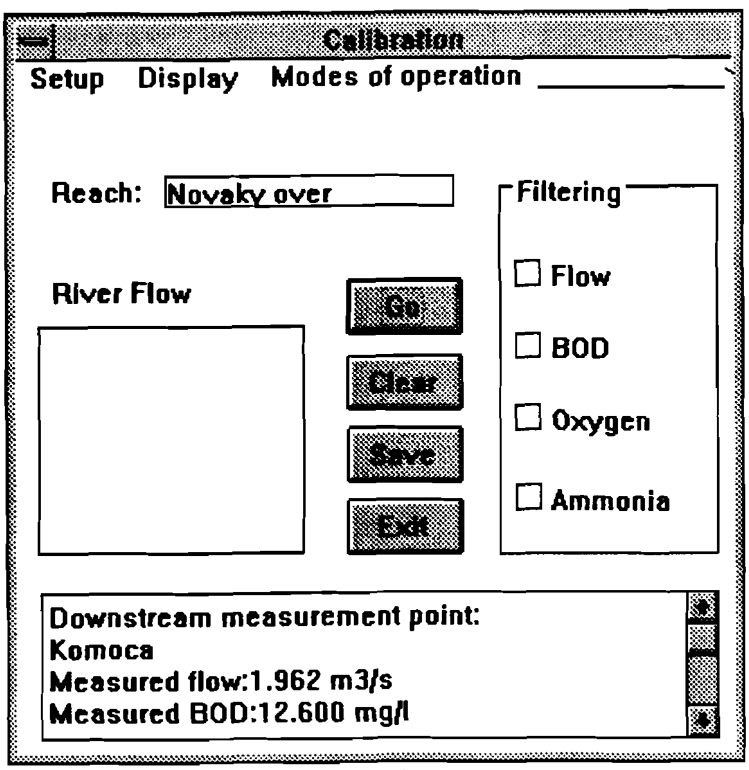


Figure 12.6 Calibration dialog box.

13. WATER QUALITY MANAGEMENT STRATEGIES FOR MUNICIPAL EMISSIONS

13.1 Introduction

The water quality management task for a river basin comprises the identification of the appropriate control alternatives needed to satisfy various water quality, economic and other goals. Water quality goals are normally expressed by various standards (see Chapters 3 and 7) which have strong implications on the cost needed to comply with them. (the cost of wastewater treatment increases exponentially with increasing removal rate).

As noted earlier, the emphasis of this study is on the development of a regional policy, for controlling municipal discharges (see Chapters 3-5 and 7) which contribute to about two-thirds of the total BOD5 emission in the catchment. The rest is represented by industry and other sources considered as non-controllable, not only because of their magnitude but also due to lack of information on realistic management alternatives and their costs. Industrial emissions were handled in the frame of a sensitivity analysis to study the effect of their possible reductions. As will be shown below, the exclusion of industrial emissions from decision variables leads to results that show the river water quality cannot be improved beyond certain limits, irrespective of the treatment employed for the municipal discharges. A joint optimal policy for both municipal and industrial emission control will be different from the "sub-optimal" municipal policy (to be discussed here), and it would offer more flexibility for water quality improvements and economic savings alike.

If the water quality standards are specified by the maximum allowable effluent concentrations, the management task is relatively easy. In that case, the problem is to find the most economical treatment alternative at each discharge point, that fulfills the requirements of the standards at that location. On the other hand, if the standards are specified for the ambient quality, that requires consideration of the joint effect of various discharges on the river water quality. A detailed discussion on the actual legislation in Slovakia and setting effluent and ambient quality standards is provided in Chapters 3 and 7. The point we wish to make here, however, is that the development of a least-cost policy crucial under the present economic conditions requires the usage of ambient criteria, for which the actual legislation (Chapter 3) opens an avenue to explore during the transition period till 2005.

There can be a number of feasible treatment technologies that can be implemented at a discharge location. If the effect of different treatment alternatives at various discharge points are to be evaluated jointly, this leads to a very large number of feasible combinations. The analysis of each of these strategies one by one is therefore impractical. Nevertheless such an analysis is not impossible, particularly if the watershed and the number of emissions is small, or, very long computational times are acceptable to obtain a solution. Such an approach requires only a water quality simulation model of the river, which is to be run for each feasible treatment strategy. The large number of water quality profiles obtained as a result of these simulations will have to be examined together with the associated costs, to make a decision as to which strategy to select.

Another approach is to use an optimization technique, which was the case for this study. An optimization model (which will be subsequently referred to as a policy model) serves as a supporting tool to identify the appropriate management alternative. The purpose of the policy model is to screen the alternative strategies in an efficient way, in order to identify the optimal strategy that satisfies preconditions set for the water quality levels. Such a model will achieve this purpose in a much shorter time than a multiple-simulation model. Chapter 10 reviews various management models and provides a description of the dynamic programming technique used in the present study.

13.2 Policy Model

The policy model selects the optimal treatment strategy from a set of feasible alternatives for each discharge (see Chapter 11). Consequently, the task of the policy model is to choose the optimal configuration by selecting one treatment alternative for each discharge. In other words, the policy model has to make a yes/no (1/0) decision for each treatment alternative at each municipal discharge. As only one alternative will be selected for each location, all but one alternative at a discharge point will receive "no" decisions. Chapter 10 contains a description of the algorithmic aspects of the policy model.

The other inputs to the policy model include the water quality goals or standards defined by the legislation. With these inputs, the policy model makes use of the water quality simulation model (see Chapter 9) estimations to identify the appropriate least-cost policy. The outcome of the policy model is characterized by a treatment configuration, resulting water quality in the river, and the associated costs. The role of the policy model and its interactions with the other modules of the decision support system is presented in the Fig. 5.1 of Chapter 5.

In order to estimate the robustness of the developed policies under different realizations of the assumed parameters etc., *a posteriori* analyses were performed. To this end, a "regret analysis" explores how the derived policy is affected by uncertain water quality model parameters, showing the gains and losses in terms of water quality (and risks associated) and costs. An *a posteriori* Monte Carlo simulation estimates possible variations of water quality, and provides probabilistic conclusions on standard violations.

13.3 An Overview of the Policy Analyses

13.3.1 Quality Indicators Selected for Setting of Standards

The policy analyses can be classified into three main groups, depending on the type of water quality standards considered (see earlier). They are based on effluent, ambient, and mixed standards. Due to the multiple pollution nature of river basins problems, the standards deserve to be defined in terms of each of those pollutants. However such an approach complicates the problem beyond practical limits. An alternative is to select only the most important quality indicators and pollutants for the purpose of setting standards. The present analysis relies mainly on ambient DO level which is the key element in maintaining the balance of the river ecosystem. Without proper DO concentrations, it is not possible to improve the water quality in terms of the other indicators. The pollutant levels measured by BOD and NH4-N also play an important role in this respect, and they are considered by the existing legislation in Slovakia. Therefore the policy decisions of this study will be based on these three quality indicators (extensions are possible), out of which ambient DO is considered as the most important one. The analysis was not limited to a single set of quality standards. Instead, the economic impacts of setting various quality standards were estimated.

13.3.2 Location of Quality Standards

Quality standards are of no value unless they are imposed (and enforced) at the right locations in the river. For example, as it is well known, the DO concentration shows a non linear variation along the river. Therefore it is important to select the critical locations where quality is poor, instead of setting a large number of standard points at random locations. Therefore the standards were mainly set, for the policy exercise, at locations that have very low DO levels at present. In addition, some other locations were also selected as a precaution against a possible shifting of the critical locations when the treatment configuration is changed. Quality standards were set at measurement points P6, P7, P9, P14, P15, P16, P17, P18, and P26 (see Appendix 7.2 and Fig. 7.1 for the names and locations respectively).

13.3.3 Economic Objectives

From the point of view of the costs, two different types of objectives were considered separately. The first was to formulate a policy with a minimum total annual cost (TAC) of waste treatment (see Chapter 11 for treatment costs). The total annual cost comprises the operation, maintenance and replacement costs (OMRC) and the annual repayment of the investment cost. The second goal was to find a policy which requires the minimal basin-wide investment cost (IC), which stem from the tight budgeting situation in CEE countries including Slovakia. A detailed discussion of these policies is given in Section 13.8.

The benefits of water quality management are rather intangible; this is the reason in many cases for expressing the objective of a management exercise in terms of costs.

13.3.4 Life Time and the Interest Rate of Treatment Plants

The TAC depends on the period of repayment and/or life time and the interest rate at which funds were secured. Both of these are not necessarily known with certainty in advance. Therefore, this was handled in a sensitivity framework, by performing separate analyses with different values for them.

13.3.5 Industrial Discharge Reductions

As indicated above, the industrial discharge control was not an explicit part of this policy analysis. The effects of the possible discharge reductions were estimated in a sensitivity fashion, by considering three different industrial discharge scenarios (in harmony with structural changes observed nowadays). The three scenarios considered were the discharges in

1990 and reduced discharge conditions corresponding to 50% and 75% reductions of 1990 amounts.

13.3.6 Uncertainty of the Water Quality Parameters

The decisions of the policy model are based on estimations of the quality simulation model, of which the parameters are highly uncertain (see Fig. 5.1 for the interactions between the two models). This issue is covered in detail in Chapter 9. These uncertainties can have a strong effect on the outcome of the policy model. Different ways to incorporate such uncertainties into a policy model exists. As described in Chapter 10, that can be done explicitly with a huge increase in the problem size or in an implicit way. The latter include generation of a large number of scenarios by Monte Carlo simulation, followed by policy analysis for each of them; a rather impractical procedure.

A more convincing approach is to perform a scenario analysis, by selecting several sets of design conditions that might be expected. The present analysis of uncertainty is biased towards this alternative. The reaeration rate was found to be the most important model parameter (see Chapter 9), with regard to their effect on the optimal treatment configuration. Therefore the *a posteriori* sensitivity analyses were mainly focused on this parameter. One of these analysis was done in a "regret analysis" framework, which examined the cost and quality implications of realizing parameters other than the design ones. Another *a posteriori* analysis, based on Monte Carlo simulations, was aimed at obtaining probabilistic conclusions on water quality when least-cost strategies are implemented under uncertain parameters.

13.3.7 The Base Case

The above discussion indicates that a policy decision cannot be obtained with a single run of the policy model. This is due to the possible variations of the parameters (and other settings) which affect the policy decisions. It is not possible to select some of them in advance by neglecting the others, as they can have a strong impact on the policy decisions. The approach used is to consider a particular set of parameters as the base case, and to analyze the sensitivity with respect to the possible changes separately. This allows a detailed comparison of the effect of different variables on the optimal policy derived for the base case. The base case is described below.

Results of the August 1992 experiment (Chapter 8) performed under critically low flow conditions (satisfying requirements of the legislation; see Chapter 3) served as the basis for the design scenario (including the derivation of parameters of the simulation models). Background pollution of the tributaries and the temperature profile (showing an increase downstream from 19°C to about 25°C) were obtained from the observations. For the generation of design emissions for municipalities, annual average values available for the period 1990-1992 (Chapter 6) were used in addition to measured ones. Industrial emissions were kept at their 1990 level, mainly as a worst-case scenario.

The life time of a treatment plant and the interest rate were assumed to be 20 years and 12% respectively (see Chapter 11 for the feasible treatment alternatives). River water quality standards were assumed to be set at nine selected locations described above. They were chosen to include the most critical locations in terms of the water quality. No minimum level of treatment was defined as a constraint for the base case. This was done mainly to see the largest flexibility of theoretically justified solutions. Strategies of practical importance with primary treatment or the current one as the lowest level are discussed later on. The water quality parameter values selected for the coefficients of reaeration, BOD decay and ammonia decay were 2.0, 0.8 and 0.8 (1/d) respectively (Chapter 9 provides details on parameter selection, while the performance of the water quality model under these parameters are displayed in Fig. 9.13).

13.4 The Current, Best Available, and Effluent Standard-based Treatment Strategies

A set of simulation runs were performed to obtain the effect of specific management alternatives that are relevant for the subsequent comparisons. In these runs, a predefined set of treatment alternatives were assumed for the various municipal discharge locations. The strategies included, those correspond to the legislation in Slovakia, and the best available treatment (BAT) which corresponds to the most stringent recommendation of the European Community. The resulting river water quality profiles were obtained by simulation (by considering the base case parameters for the water quality model). Therefore no policy model was involved in these analyses. The specific alternatives considered were:

(1) No treatment,
(2) Current treatment,
(3) Treatment strategy defined by the present effluent standards in Slovakia ($BOD \leq 30$, $NH4\text{-}N \leq 10$, $TP \leq 3$; denoted by Slovak-1 in Table 13.1)
(4) Treatment strategy corresponding to the future effluent standards in Slovakia ($BOD \leq 25$, $NH4\text{-}N \leq 5$, $TP \leq 1.5$; denoted by Slovak-2 in Table 13.1)
(5) The best available technology (BAT) corresponding to the most stringent recommendations of the European Community (which necessitates construction of new treatment plants).

The implications on river water quality, of implementing these strategies, are summarized in Table 13.1 together with the associated costs and treatment configurations. The minimum and maximum values of water quality indicators of Table 13.1 display the corresponding worst water quality in the river. Treatment configurations are presented with the treatment level numbers selected at each discharge location. A "0" indicates that there is no treatment. The treatment level numbers corresponding to the BAT strategy indicate the highest treatment level available for a particular discharge. They are indicative of the number of treatment alternatives available at each discharge point. It is also to be noted that the treatment level numbers of different discharge points have no relation to each other. For example, the treatment level 8 at Topolcany (To) and the level 4 at Handlova (Ha) both refer to the same type of treatment technology (see Chapter 11 and Appendix 11.1 for details of treatment alternatives).

Each of these treatment alternatives is characterized by the flow, the effluent quality expressed by DO, BOD, SS, NH4-N, nitrate nitrogen (NO3-N) and TP, as well as the estimated costs, IC, OMRC and TAC. The alternatives were obtained by considering existing units (primary sedimentation tank, aeration basin, final clarifier and sludge processing), their various upgrading possibilities, and the design of new treatment plants. A description of the alternative wastewater treatment technologies defined for different locations can be found in Chapter 11.

The following abbreviations for the names of municipal discharges have been used in Table 13.1 and in the subsequent tables. (see Figure 7.1 for the locations).

Ha =	Handlova	Ni =	Nitra
Le =	Lehota	Zl =	Zl. Moravce
Pr =	Prievidza	Vr =	Vrable
Pa =	Partizanske	Su =	Surany
Ba =	Banovce	No =	Nove Zamky
To =	Topolcany		

Table 13.1 Performances of the Current, BAT and Slovakian Effluent Standard Based Treatment Strategies

Strategy	IC (mil. USD)	OMRC (mil. USD)	DO Min (mg/l)	BOD Max (mg/l)	NH4-N Max (mg/l)	Treatment alternatives selected for different locations *										
						Ha	Le	Pr	Pa	Ba	To	Ni	Zl	Vr	Su	No
None	0.0	0.0	0.1	34.6	7.9	0	0	0	0	0	0	0	0	0	0	0
Current	0.0	5.7	2.3	30.6	7.7	1	1	0	1	1	1	1	1	1	1	1
Slovak-1	32.1	8.4	5.4	11.3	2.3	2	2	4	3	3	6	5	2	1	2	4
Slovak-2	35.2	8.7	5.4	11.3	2.3	2	2	4	3	3	6	6	2	2	2	4
BAT	95.5	11.1	5.7	11.1	2.1	4	3	5	3	5	8	7	4	3	3	9

* discharge locations are indicated by the first two letters of their names

Table 13.1 shows that the feasible ranges of water quality are relatively narrow. The maximum BOD levels corresponding to no treatment and the best available treatment are 35 (very poor quality) and 11 mg/l respectively. However the corresponding difference in cost is 95.5 million USD. The reason for the high BOD that occurs even with the BAT, violating the standards of legislation (see Section 3.6), is the industrial and non-controllable discharges which were assumed to be unchanged. The same reason applies for the minimum DO level of 5.7 mg/l observed with the BAT strategy.

The present and future effluent standards of Slovakia, in spite of their cost difference, are identical from the view point of ambient water quality.

13.5 Least-Cost Policies on the Basis of DO Alone

The setting of ambient quality standards opens up the possibilities to formulate least-cost policies. Table 13.2 displays several strategies that were obtained by specifying DO standards between 3 mg/l and 6 mg/l (without setting limits to the minimum treatment level). The "cost" in this case was taken as the total annual cost (TAC).

Table 13.2 Least-Cost Treatment Strategies Obtained by Specifying Ambient DO Standards

Policy	IC (mil. USD)	OMRC (mil. USD)	DO Min (mg/l)	BOD Max (mg/l)	NH4-N Max (mg/l)	Treatment alternatives selected for different locations										
						Ha	Le	Pr	Pa	Ba	To	Ni	Zl	Vr	Su	No
DO≥3	3.2	3.8	3.0	18.7	4.2	0	0	2	0	1	2	2	0	0	1	0
DO≥4	4.0	5.6	4.2	13.8	4.1	0	0	2	1	3	1	1	0	0	1	2
DO≥5	15.0	6.2	5.0	13.8	3.2	0	0	2	1	3	6	2	0	1	1	2
DO≥6	33.4	7.7	5.6	11.3	2.2	2	1	3	1	3	8	5	0	0	1	2

Table 13.2 shows that substantial improvements are attainable without incurring heavy investments (in contrast to Table 13.1). The IC shows an exponential increase (see Fig. 13.1) with tightening minimum DO level. The performance of the strategy corresponding to DO≥6 differs only slightly from the BAT strategy of Table 13.1. On the contrary, the difference in the investment costs of those two alternatives is 62 million USD.

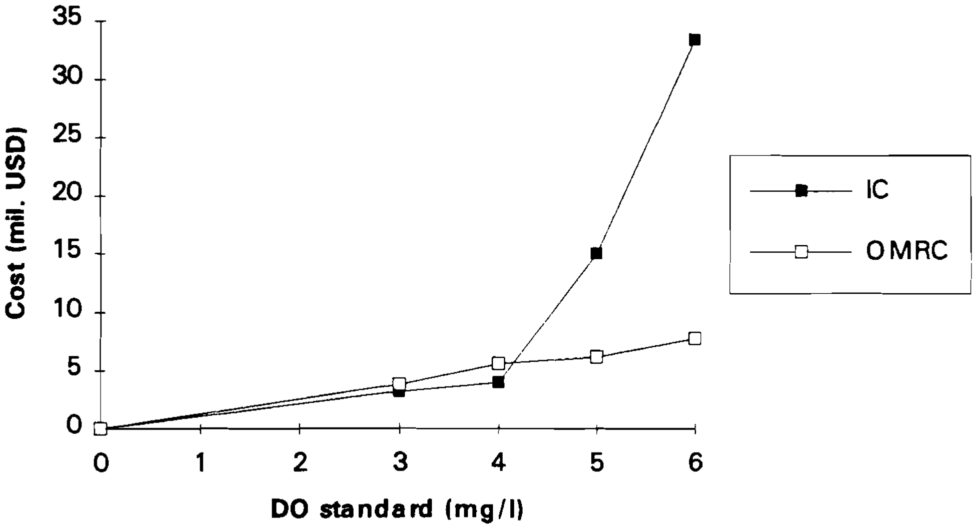


Figure 13.1 Cost implications of setting different ambient quality standards

The quality improvements under least-cost policies is further illustrated in Fig. 13.2, in which longitudinal DO profiles along the Nitra River are displayed. These DO profiles correspond to those resulting from current treatment, Slovakian effluent standards (present) based strategy, BAT, and the least-cost policies obtained with DO standards of 3 and 5 respectively.

13.6 The Performance of Mixed Strategies; Incorporation of BOD and NH_4-N Standards

In addition to the standards set for the ambient DO, the analysis was repeated with ambient standards for BOD and NH4-N as well. The policies (1) and (2) of Table 13.3 are the results of such analyses. Mixed policies (3) and (4) include quality standards for both ambient and effluent water quality. The definitions are as follows.

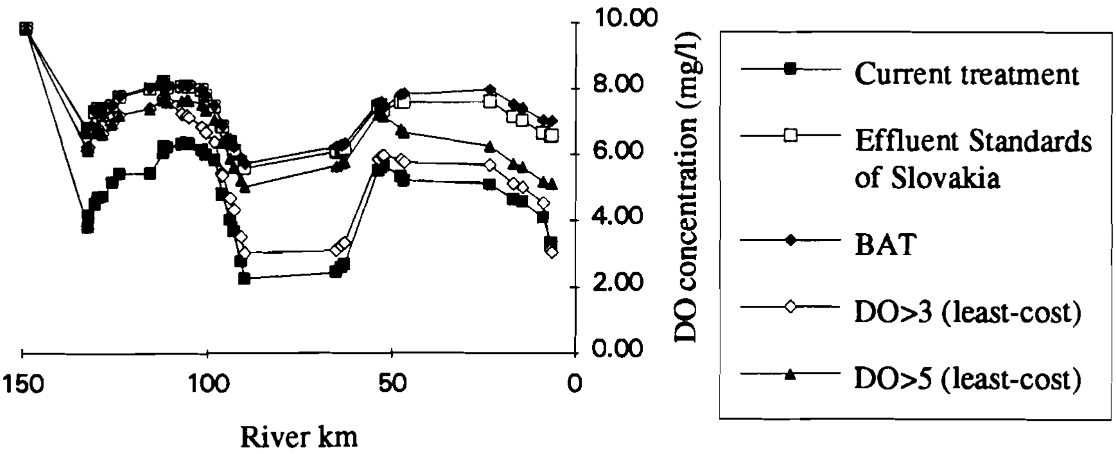


Figure 13.2 Longitudinal profiles of DO concentration for five different treatment strategies

Least-cost policies based on ambient quality standards:

Policy 1 $DO \geq 5$, and $BOD \leq 10$

Policy 2 $DO \geq 5$, and $BOD \leq 10$, $NH4\text{-}N \leq 2$

Least-cost policies based on mixed (ambient and effluent) quality standards:

Policy 3 $DO \geq 5$, and $BOD \leq 10$, Effluent $NH4\text{-}N \leq 10$

Policy 4 $DO \geq 5$, and $BOD \leq 10$, Effluent $NH4\text{-}N \leq 10$, Effluent $TP \leq 3$

Table 13.3 Performance of Least-Cost Strategies by Specifying DO, BOD and NH4-N Standards

Policy	IC (mil. USD)	OMRC (mil. USD)	DO Min (mg/l)	BOD Max (mg/l)	NH4-N Max (mg/l)
1	20.0	7.0	5.0	11.2	3.6
2	29.5	7.1	5.6	11.3	2.2
3	31.0	7.9	5.6	11.1	2.2
4	33.6	8.5	5.5	11.3	2.3

Policy (1) indicates that by specifying ambient BOD standards in addition to DO, the IC requirement increases by 5 million USD (see Table 13.2). However setting of BOD standards is not necessarily justified professionally. Incorporation of NH4-N has an even stronger influence. Policies (3) and (4) that were obtained with effluent standards in addition to the ambient ones produce less attractive results. The water quality remains almost the same as those of the policy (2), although the costs show an increase. This is due to the rigid restrictions imposed on the type of feasible treatment alternatives. The ambient standards, on the other hand, allow sufficient flexibility with regard to the treatment technology selected at individual discharge points.

13.7 The Effect of Reductions of Industrial Discharge on the Least-Cost Policies

Table 13.4 demonstrates the effect of industrial discharges. The first two rows of Table 13.4 corresponds to the least-cost treatment strategy (1) of Table 13.3, but with reduced industrial emissions (by 50% and 75% respectively), the influence of which is evaluated by simulation.

Rows (3) and (4) of Table 13.4 contain the DO≥5 municipal least-cost policies obtained under the assumption that industrial discharges were already reduced (again, by 50% and 75% respectively, and the cost the reduction is excluded).

Table 13.4 Effect of Industrial Discharge Reduction

Case	IC	OMRC	DO Min	BOD Max	NH4-N Max
1	20.0	7.0	6.0	8.5	3.5
2	20.0	7.0	5.5	8.2	3.4
3	7.2	6.2	5.1	9.7	3.6
4	6.2	5.7	5.1	11.4	3.8

Table 13.4 indicates improved water quality as a result of reductions of industrial discharges. Nevertheless the improvement is not in proportion to the discharge reduction. This is partly due to the "non-controllable" pollutant sources which prevents river water quality improvement beyond certain limits (the saturation DO level is about 8 mg/l). The least-cost strategies formulated under reduced discharge conditions suggest cost "savings" where the actual saving obviously depends on expenditures of the industrial control. The issue clearly calls for the development of an integrated least-cost strategy covering all the discharges of different origins. The impact of the industrial emission control on cost of municipal treatment is further displayed graphically in Fig. 13.3. It summarizes the least-costs required to maintain DO and NH4-N ambient standards for the Nitra River basin under three different industrial discharge scenarios.

13.8 Effect of Different Economic Objectives on the Least-Cost Strategy

All the least-cost policies presented so far were obtained so as to minimize the total annual cost (TAC) of municipal treatment in the river basin. Although they are associated with the minimum TAC's, their IC requirements can be high. This can be observed from the following relationship of the TAC and the IC (see Chapter 11):

$$TAC = (CRF) * IC + OMRC, \qquad (1)$$

where CRF is the capital recovery factor.

According to (1), a strategy with a low TAC can have a high IC and a low OMRC. Such strategies with high investment needs may not be desirable due to the lack of initial funds. In such cases, a strategy with a relatively low investment cost, even if associated OMRC is relatively high, would be preferred. Such a strategy can be obtained by specifying the IC as the economic objective of the policy model. The former strategy gives a solution which is desirable in the long run, and the latter one gives immediate benefits. These two conflicting goals can be best handled in a multiobjective framework. Although the present study does not explicitly include such analyses, the strategies obtained by setting these two different objectives are compared and contrasted.

Least-investment-cost treatment strategies are summarized in Table 13.5 (see Table 13.2 for the corresponding results with the TAC objective). Fig. 13.4 shows a comparison of the two strategies (for the DO≥5 case). It is observed that the IC formulation resulted in consistently higher TACs than the corresponding solutions obtained with the TAC objective. In general, the IC objective selected more uniform treatment levels at different locations. This was due to two main reasons. First, the no-treatment alternative was not considered as a solution in the case of the IC objective (which would be also the case in practice). Instead, the new economic objective obviously forces the selection of existing treatment without additional investments as the lowest feasible treatment level. Second, the exponential increase of the IC's in selecting high levels of treatments leads to more uniform set of lower treatment levels.

With the TAC objective, the solution is observed to be rather non uniform. It is not only non-uniform, but also can be quite unstable with regard to slight changes in the quality standard locations and other parameters. This is mainly due to the fact that two alternative solutions that are very different in terms of the IC, can produce only a marginal difference in TAC. Another reason is that selection of no-treatment as a decision was not restricted by the model formulation, because of theoretical reasons (which would be never the case in the reality).

Table 13.5 Least-Cost Treatment Strategies with Minimum Investment Costs

Policy	IC (mil. USD)	OMRC (mil. USD)	DO Min (mg/l)	BOD Max (mg/l)	NH4-N Max (mg/l)	Treatment alternatives selected for different locations										
						Ha	Le	Pr	Pa	Ba	To	Ni	Zl	Vr	Su	No
DO≥3	0.0	5.9	3.5	12.7	5.7	1	1	1	1	1	1	1	1	1	1	1
DO≥4	2.0	6.3	4.0	12.5	3.9	1	1	2	1	1	1	2	1	1	1	1
DO≥5	15.0	6.9	5.2	11.4	3.2	1	1	2	1	3	6	2	1	1	1	2
DO≥6	33.4	8.2	5.6	11.3	2.2	2	1	3	1	3	8	5	1	1	1	2

13.9 The Effect of the Locations of Water Quality Standards

Quality standards influence the least-cost strategy in two ways. First, the standards themselves have a direct impact on the required treatment levels. Second, the locations of standards also affect the optimal strategy. Unless they are set at the most critical locations, the resulting strategy would fail to improve water quality at such locations. Quality standards assumed to exist at locations with good quality, will have a minimum effect on the treatment configuration. They can even become redundant standards.

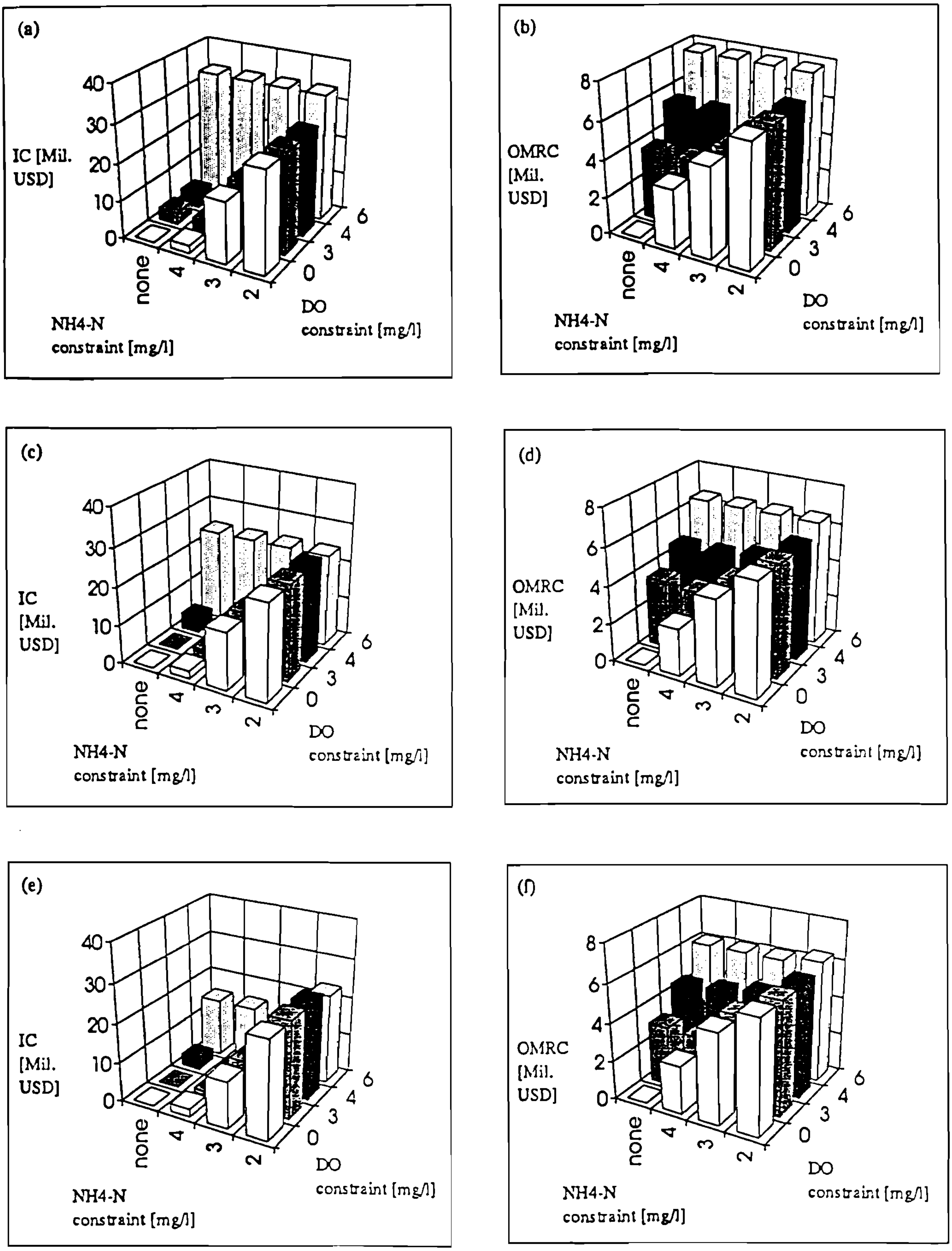


Figure 13.3 Cost Implications of setting DO and NH4-N standards under different industrial discharge scenarios. (a),(b) IC and OMRC respectively, corresponding to 1990 discharges; (c),(d) IC and OMRC respectively, for a 50% reduction of ind. discharges; (e),(f) IC and OMRC respectively, for a 75% reduction of ind. discharges

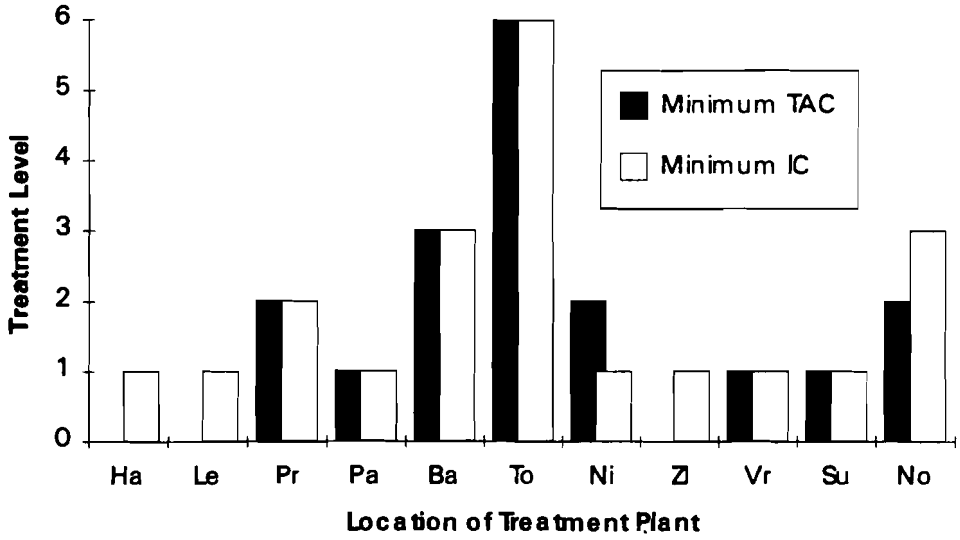


Figure 13.4 Comparison of the least-cost treatment strategies for two different economic objectives: minimization of TAC, and minimization of IC, (DO≥5)

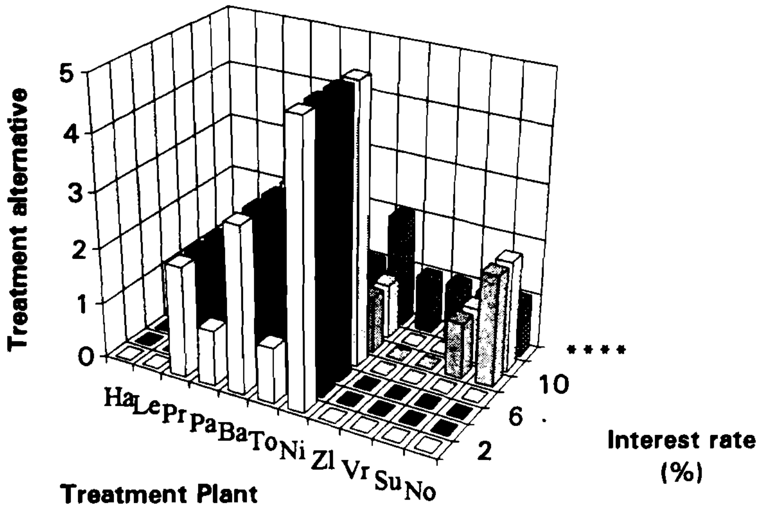
The least-cost strategies presented so far were obtained with the ambient quality standards specified at locations identified as critical. Sensitivity of the resulting strategy to these locations were evaluated by considering two more different sets of quality standard locations. These sets comprised only three locations each.

Although a fewer standard locations may seem to lead to cheaper least-cost strategy, this did not happen for one of the new location sets. Instead, a slight cost increase was observed, which may indicate that the original standard locations are not quite suitable. However, this increase in cost was relevant only for low levels of ambient standards (DO≥3 and DO≥4). The strategy corresponding to DO≥5 happened to be the same as that obtained with the original standard locations (see Table 13.2). For the DO≥6 case, a "cheaper" strategy was identified, implying the new locations are unsuitable. The increased cost with the lower standards and the reduced cost with the higher standards indicate the shifting of the critical water quality location under different levels of treatment. In other words, the new locations become critical when low levels of treatment is applied, but they are not critical under higher levels of treatment. However, the indicators did not change significantly under modified locations. In view of these, the original locations are observed to be acceptable for setting of standards in the Nitra River basin.

13.10 The Effect of Possible Changes of Interest Rate

The interest rate has a direct impact on a policy decision based on the minimization of the TAC. It does not, however, influence decisions based on the IC objective. The base case of this study assumes an interest rate of 12%, which, in reality, may vary up or down within certain practical limits. In order to estimate the effect of such changes on the least-cost strategies, different interest rates were employed. The results are displayed graphically in Fig.

13.5, for an ambient standard of DO≥4 mg/l. In addition, Fig. 13.5 also displays the treatment strategy obtained with the minimized IC objective.



**** The least-cost strategy obtained with the objective of minimum IC.

Figure 13.5 Effect of the interest rate on the least-cost strategies obtained with the objective of minimum TAC (ambient quality standard: DO≥4).

Figure 13.5 shows that the lower interest rates (2% - 8%) provide a strategy with high levels of treatments at certain locations, with no treatment at some others (without prescribing the minimum level of treatment). This is accompanied by a higher investment cost as well. High interest rates (10% and 12% in this case) provided a more uniform set of treatment alternatives, and consequently imply a lower investment cost. A similar behavior was observed for the DO≥5 case as well. In that case, IC of the least-cost strategy was increased by 9 million USD under lower interest rates. The strategies corresponding to DO≥6 case were found to be unchanged under different interest rates, except for the lowest rate considered (2%). It is important to note that the costs of treatment strategies obtained under the base case (12%) are stable under the realistic range of interest rates varying from 9% to 12%.

The impact of the two conflicting objectives and variations of interest rates on the least-cost treatment strategy is further illustrated in Figure 13.6. It displays the range of treatment alternatives that would be identified by the policy model, depending on the interest rate and the economic objective. Treatment strategies formulated with ambient standards of DO≥4 are summarized there (least-cost strategies obtained with DO≥6 standards, however, show much less variation). It is noted that in practice, plants would be operated on the current level and thus the variation is much smaller than suggested by Figure 13.6.

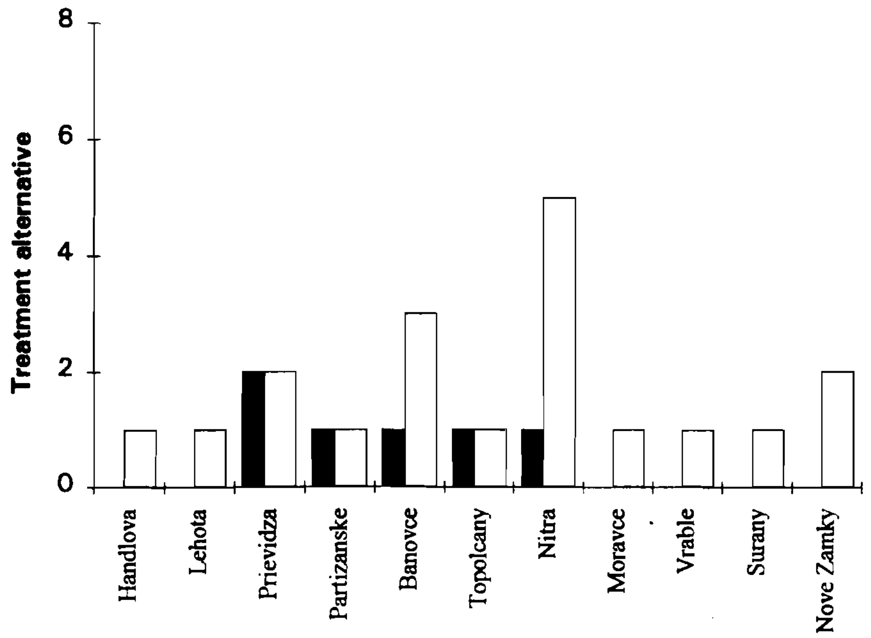


Figure 13.6 The range of treatment levels identified for interest rates from 2% to 12% (with ambient standard DO≥4)

13.11 The Effect of the Life Time of Treatment Plants

In the preceding policy analyses, the life time of each treatment alternative was assumed to be 20 years. This assumption was made irrespective of the type of the treatment plant that can be new, upgraded, or a mixture of those two. If the actual life times are different, that might make the least-cost policies based on traditionally formulated TAC objective inappropriate for the real situation. In order to analyze this effect, a sensitivity analysis was performed with four sets of life times for the three categories of treatment alternatives. They contained life times ranging from 10 to 40 years.

These sensitivity runs were performed with three different interest rates: 12%, 6%, and 2%. In the case of the 12% interest rate, the life times did not significantly affected the least-cost strategies or the cost requirements. With low interest rates, some changes in the strategies were observed. However they were quite irregular and insignificant, and therefore will not be discussed further in detail.

13.12 Strategies with Primary Treatment as the Minimum Treatment Level

Policy analysis for the four DO standards were repeated, by specifying primary treatment as the worst feasible treatment. Previous analysis allowed no-treatment, thus assuming some of the plants can be shut down under least-cost strategy. The results show that new policies are obviously more "expensive", for both objectives. Although the ICs are unchanged, the new ones are expensive in terms of the TAC, due to their high OMRCs. Figure 13.7 compares the two types of treatment strategies obtained for DO≥5 mg/l ambient standard.

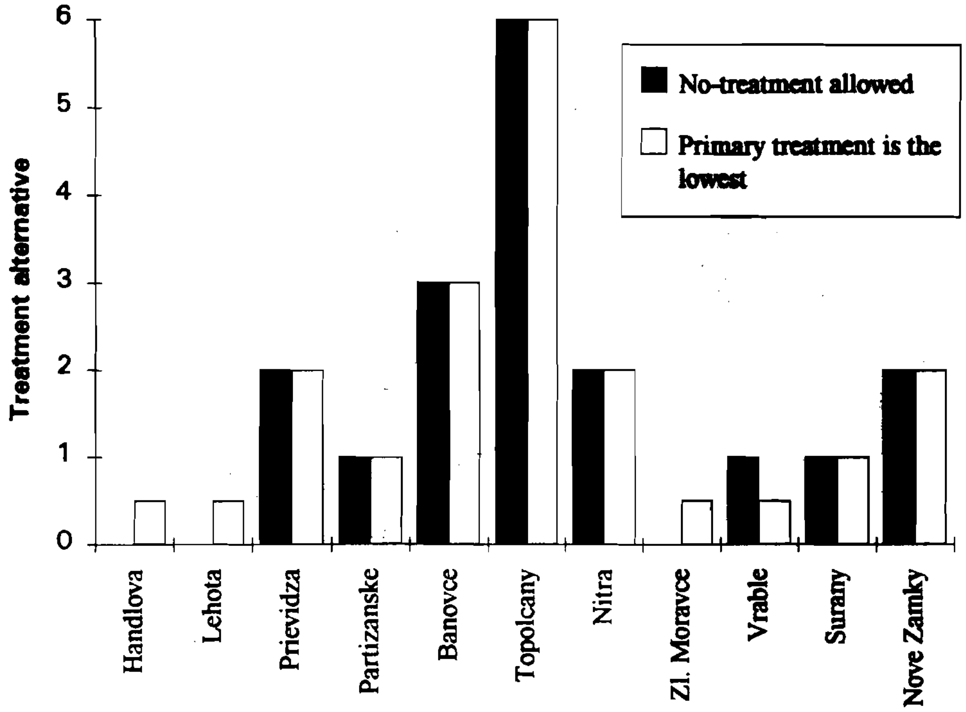


Figure 13.7 Least-cost (TAC) strategies (for DO≥5) which specify no-treatment and primary treatment, respectively, as the lowest alternative

13.13 Strategies with Current Treatment as the Lowest Treatment Alternative

The analysis of Section 13.12 was extended one step further, by specifying current treatment as the lowest feasible alternative. Only the policies with TAC objective need to be obtained with this formulation, since the IC objective incorporates it automatically (see Section 13.8). Even the TAC policies were observed to be identical to those with IC objective, which have been discussed in Section 13.8.

13.14 Uncertain Water Quality Model Parameters - Their Impact on the Least-Cost Policies

Deterministic conditions were assumed for obtaining the least-cost strategies of this study. Uncertain water quality parameters were assigned predetermined values for the optimization process. The effect of parameter uncertainty on the optimal solution was analyzed *a posteriori*, in two ways, which are described below.

The aim of the first approach was to evaluate the effect of using a design parameter which will not be realized. This implies that a mistake has been done in the design, and the magnitude of the mistake, in relation to the realized parameter, can be expressed in terms of the over-investment or under-investment, and the associated improvement or worsening of water quality. This can be termed as a "regret-analysis", although the "regret" can be positive or negative (see Burn and Lence 1992 for a broader regret analysis). Reaeration rate coefficient, the most important model parameter, was considered for this analysis, which is presented in Section 13.14.1.

Second approach was to estimate the feasible range of water quality under specific treatment-strategies. For this purpose, water quality estimation was done by Monte Carlo simulation, and the results lead to probabilistic conclusions on the quality impacts of least-cost strategies. Section 13.14.2 describes the Monte Carlo analysis and its results in more detail.

13.14.1 "Regret-analysis" for quantifying the effects of parameter uncertainty on the control strategies

As indicated in Section 13.14, the "regret-analysis" estimates the cost and quality implications of not realizing the parameter used for the design. Therefore, this necessitates evaluation of the performance of the least-cost policy under different parameter values. In the policy analysis, reaeration coefficient (K_{ao}, see Equation 9.12) was assumed to be 2 1/d. The regret-analysis assumes that a deviation of +/- 0.5 1/d from this value is possible, which corresponds to a deviation of +/- 2 times standard deviation (see Chapter 9). This range would, therefore, cover the most probable range of the parameter.

The difference in the design and realized values will lead to a deviation in ambient quality (the realized quality will be different to the design value), which can be indicated by the minimum DO level. Quality estimations of the regret-analysis relies solely upon this indicator.

If the deviation in ambient water quality is negative, standards will be "violated", indicating an under-investment, while a positive deviation indicates an over-investment. The over or under investment is the cost difference of least-cost policies corresponding to the two different parameter values (design and realized).

The analysis can be further explained with respect to Table 13.6, which summarizes the results obtained for DO≥3 policy. As Table 13.6 indicates, three design scenarios (K_{ao}=1.5, 2.0, 2.5 1/d) were considered to evaluate their behavior under three "realized" scenarios. Investment cost of each design is indicated by the last line. The other cells in the table contain two numbers each, which indicates the cost and quality implications of each design under different realizations of parameters. The first number indicates the over or under investment. As an example, if the design was done with K_{ao}=1.5 and K_{ao}=2 was realized, the table indicates an over-investment of 1 million USD. This is because the K_{ao}=1.5 design costs 4.2 million USD, while K_{ao}=2 design costs 3.2 million USD. If it was known *a priori*, that K_{ao}=2 will be realized, the design could have been done accordingly, thus indicating the 1 million USD (4.2-3.2) over-investment. Under-investments are indicated by negative signs. The second number shows the improvement or worsening of the water quality. These are the results of quality simulations performed by assuming that a designed treatment strategy is in operation, while a different parameter is realized. The cell pointed above (design K_{ao}=1.5, realized K_{ao}=2.0) indicates 1.5 mg/l for the improvement in DO level (which also imply that the minimum DO level is 4.5 mg/l , because the design was done for DO≥3). A negative number for the second value indicates a violation of quality standard.

Table 13.6 The Role of Uncertainty of Reaeration Coefficient: Its Implications on the Investment Cost and the Ambient DO Concentration (for DO≥3 policies)

Actual K_{ao}[l/d]	DO≥3 Design K_{ao}[1/d]		
	1.5	2.0	2.5
1.5	0/0*	-1.0/-1.8	-4.2/-3.0
2.0	1.0/1.5	0/0	-3.2/-1.4
2.5	4.2/1.6	3.2/0.6	0/0
IC	4.2	3.2	0.0

* over-investment and improvement in water quality, respectively

Large negative deviations of ambient DO criteria indicate high vulnerability of the treatment policy decision and, consequently, significant risks. If that risk is to be avoided, one can look for the "safe" solution, which is more costly. Large investment differences under different parameter values imply that a "safe" policy decision is expensive, consequently the over-investment, if so becomes, will be high. The regret analysis thus exposes decision risks and helps to determine whether the safe policy is affordable, and vice versa. Tables 13.7 and 13.8 summarize the regret analysis results for DO≥4 and DO≥5 mg/l respectively.

Figure 13.8 further illustrates the minimum DO concentrations which may occur when the actual reaeration rate is different to the design values (the effect on nine designs done by assuming different K_{ao} values and setting various DO constraints are presented). Each curve of the Figure 13.8 correspond to one set of design conditions. The point corresponding to the design conditions are indicated on each of them. The other four points on each curve were obtained by subsequent simulations. For these simulations, the set of "optimal" treatment alternatives found for the respective design condition was assumed to be in operation. The simulations was performed for the four different K_{ao} values, in order to obtain the four other points of the curve. It is observed that the curves of two of the design conditions are coinciding (designs conditions: K_{ao}=2, DO_{min}=5.6; and K_{ao}=1.5, DO_{min}=5). That implies the possibility to choose the cheaper one (in this case K_{ao}=2, DO_{min}=5.6), when considering a selection from those two alternatives.

Tables 13.6-13.8 also illustrate the high sensitivity of investment cost to the value of K_{ao} (the last line). A higher K_{ao} leads to increased natural O_2 input to the river water, and thus, a lower level of treatment is necessary.

Table 13.7 The Role of Uncertainty of Reaeration Coefficient: Its Implications on the Investment Cost and the Ambient DO Concentration (for DO≥4 policies)

Actual K_{ao}[l/d]	DO≥4 Design K_{ao}[1/d]		
	1.5	2.0	2.5
1.5	0/0	-17.0/-1.5	-19.5/-3.0
2.0	17.0/0.6	0/0	-2.5/-1.1
2.5	19.5/1.0	2.5/0.7	0/0
IC	21.0	4.0	1.5

Table 13.8 The Role of Uncertainty of Reaeration Coefficient: Its Implications on the Investment Cost and the Ambient DO Concentration (for DO≥5 policies)

Actual K_{ao}[l/d]	DO≥5 Design K_{ao}[1/d]		
	1.5	2.0	2.5
1.5	0/0	-16.4/-0.8	-24.2/-2.1
2.0	16.4/0.5	0/0	-7.8/-0.6
2.5	24.2/0.9	7.8/0.5	0/0
IC	31.4	15.0	7.2

As can be seen from the Tables 13.6-13.8, the drop in the DO level caused by the reduction in the reaeration rate exceeds the improvement under equal parameter increase. This occurs due to the saturation character of the problem (the upper limit of DO_{min} under municipal emission control is less than 6 mg/l, see Table 13.2.) Similarly, missing investments exceed overexpenditures at higher DO levels.

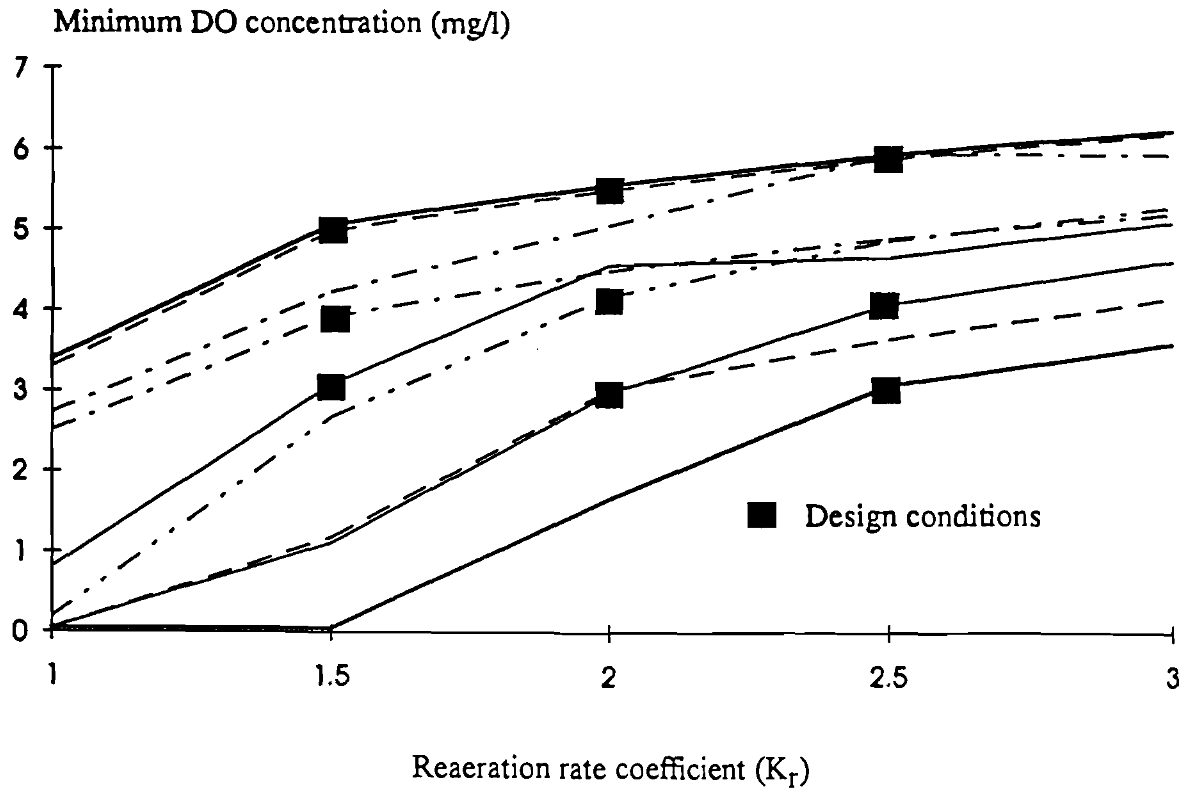


Figure 13.8 Minimum DO concentrations which may occur when the actual K_r value is different to that considered in the design

For the DO criterion of 3 mg/l, the vulnerability of ambient water quality is high, while possible overexpenditures and additional investment needs are small. This calls for a safe policy decision to invest 4.2 million USD which guaranties DO above 3 mg/l.

For target DO levels 4 and 5 mg/l, the over and underexpenditures become much higher. The smaller deviations of the DO criteria makes the policy selection even more difficult. Interestingly, the policies aimed at DO level 4 mg/l do not look too attractive and the DO=5 mg/l strategy of 15 million USD investment is perhaps the best compromise. The additional investment requirement may be rather high if K_{ao}=1.5 l/d is "realized", but the respective improvement in DO level (from 4.2 mg/l to 5 mg/l) is not in proportion with the expenses. Thus, the above policy can be considered as relatively cheap one which (as contrasted to the present DO_{min} level around 2 mg/l) guarantees DO between 4.2 mg/l and 5.5 mg/l (and the "worst" DO levels may be observed under low flow conditions only, occurring for at most one or two weeks in a year).

Finally, DO=6 mg/l strategies (not presented in this paper) do not offer too many interesting features. They are expensive (25-35 million USD, depending on K_{ao}) and safe (DO=5.5-6.4 mg/l, K_{ao}=2 l/d). In addition, over or underexpenditures are smaller (10 million USD) than for the DO=5 mg/l case.

13.14.2 Results of the Monte-Carlo Simulation Approach Used to Estimate the Implications of Parameter Uncertainty

The performance of two treatment strategies, under uncertain parameters (coefficients of reaeration, K_{ao}; BOD decay, K_r; and ammonia decay, K_n), were evaluated by *a posteriori* Monte Carlo simulations. The strategies considered were the current treatment and the least-cost (IC) one corresponding to ambient DO≥5 mg/l. Although the policy analysis employed a three-component water quality model (DO, BOD and NH4-N), the Monte Carlo analysis was done under three-component as well as two-component (DO and BOD only) model formulations. Three-component model require consideration of all three parameters, while ammonia decay rate is excluded in the two-component formulation.

Parameter sets used for these *a posteriori* Monte Carlo simulations were those appropriate for predicting the system behavior under present conditions (see Chapter 9 for details of the parameter estimation procedure). The *a posteriori* analysis does the opposite, by using the pre-selected parameters to predict water quality. Parameter identification has been done in two ways; jointly, and marginally; depending on whether the joint effect of parameters or the individual effect was considered during the identification process. Identification of joint parameter sets, although more appropriate, require heavy computations especially if the number of parameters is large. Therefore joint parameters have been identified only with the two-component model formulation, for which a marginal set also has been identified. Parameters for the three-component model were estimated only on the marginal basis.

Accordingly, water quality estimations of the *a posteriori* Monte Carlo analysis also can be categorized into marginal or joint sensitivity analysis, depending on how the parameters were generated. In the marginal sensitivity analysis, water quality is analyzed with respect to the variations of each individual parameter separately, while joint analysis considers the joint effect. An outline of the analysis is given in Table 13.9.

Table 13.9 Outline of Quality Estimations of Monte Carlo Simulations

Quality Model	Treatment strategy and the type of analysis	
	Current treatment	Least-cost (IC) strategy (DO≥5)
Three-component	Marginal	Marginal
Two-component	Joint	Joint
	Marginal	Marginal

Extreme values of DO, BOD and NH4-N in the river (which may occur at different locations), obtained under the least-cost strategy, using three-component marginal sensitivity analysis, are summarized in Fig. 13.9-13.11 respectively. The DO plot shown in Fig. 13.9 is related to the reaeration coefficient. However, estimated DO values are also affected, to a lesser extent, by BOD decay rate and ammonia decay rate coefficients as well; although they are not presented in this paper.

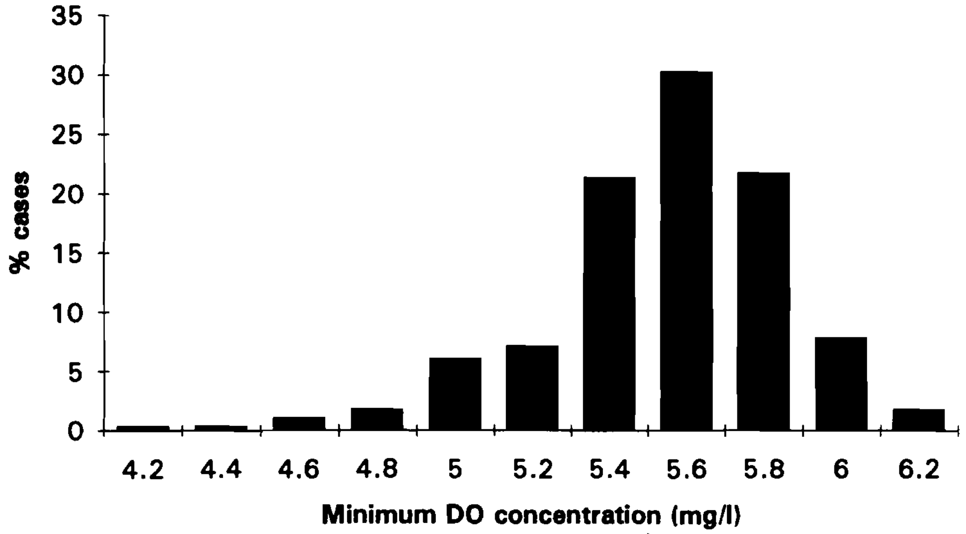


Figure 13.9 Distribution of the minimum DO concentration in the Nitra River under the least-cost strategy

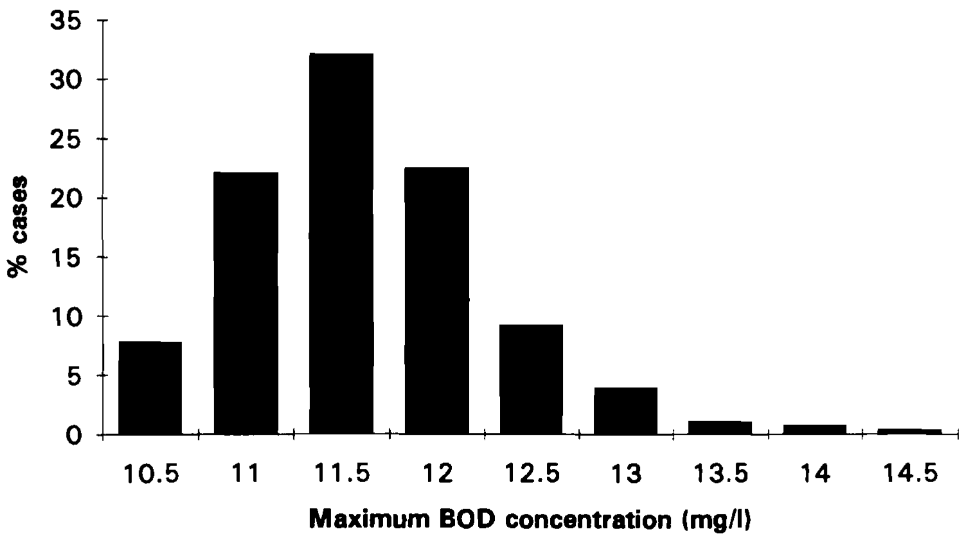


Figure 13.10 Distribution of the maximum BOD concentration in the Nitra River under the least-cost strategy

Figures 13.9-13.11 show that the extreme quality levels are centered around the deterministic estimates done in the policy analysis (see Table 13.5). The extreme quality range is relatively small, and even a much larger deviation may not be regarded as undesirable, because the extreme values will occur only at one or two locations within the river basin.

Table 13.10 presents the mean values and standard deviations of minimum DO levels (for the least-cost strategy), as estimated by the three analyses outlined in Table 13.9. The results of Table 13.10, in the case of marginal analyses, refer to the variation of estimated DO with respect to each individual parameter. Minimum DO levels resulting from the current treatment strategy are summarized in Table 13.11.

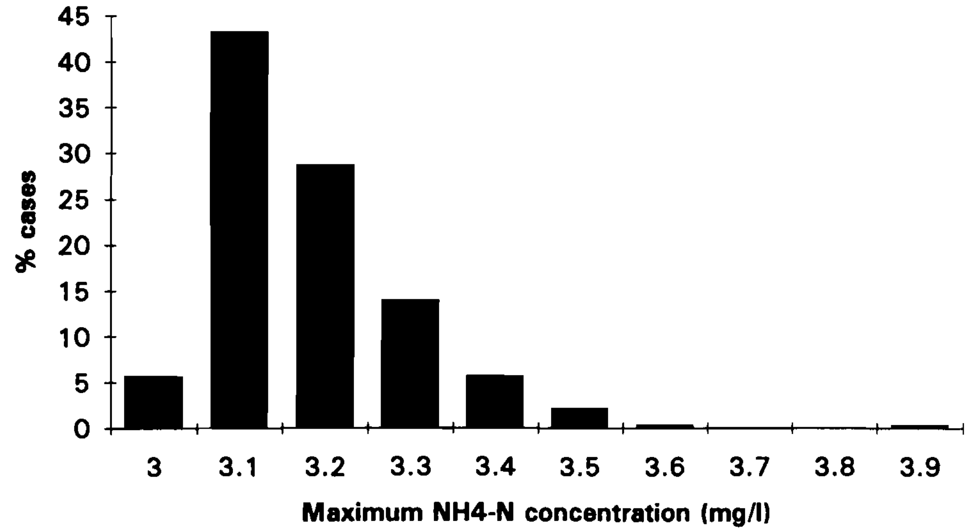


Figure 13.11 Distribution of the maximum NH4-N concentration in the Nitra River under the least-cost strategy

Table 13.10 Statistics of minimum feasible DO levels for the least-cost treatment strategy (estimated by three-component water quality model)

	Parameter of the marginal analysis		
	K_{ao}	K_r	K_n
Mean (mg/l)	5.6	5.2	5.2
Std. deviation	0.3	0.1	0.03
Prob(DO≥4)*	100	100	100
Prob(DO≥5)	96.4	98.0	100

*probability (%) of having a minimum DO level of more than 4 mg/l

Table 13.10 shows a difference in the mean DO levels estimated by different models. The improvement of water quality resulting from the least-cost strategy can be seen by comparing Tables 13.10 and 13.11. It should also be noted that, with the current treatment, the probability of having a minimum DO level which is more than 4 was estimated as zero by each model.

Table 13.11 Statistics of minimum feasible DO levels for the current strategy (estimated by three-component water quality model)

	Parameter of the marginal analysis		
	K_{ao}	K_r	K_n
Mean (mg/l)	2.9	2.4	2.3
Std.deviation	0.5	0.3	0.3

Although the mean DO levels of least-cost policy indicate an improvement, the variations do not show a consistent improvement. Even an increase in the variation is noted with the two-component model. This stems from the fact that wastewater treatment alternatives do not reduce fluctuations of effluent concentrations, although they reduce the concentration on the average. A control structure such as a reservoir, on the other hand, will have an impact on such fluctuations of river water quality.

13.15 Policy Recommendations

As mentioned several times earlier, the new Slovak legislation uses jointly effluent and ambient water quality standards and specifies 2005 as a target year after which criteria are tightened. Thus approximately a decade is kept in mind to realize feasible strategies, i.e. strategies which improve the state of the water environment and are affordable from a financial viewpoint. In the light of the above transition period and policy dilemma, the question is now which recommendation should be given on the basis of the analyses performed and discussed in earlier sections.

The starting point of our answer is a rough comparison of costs and impacts of different policies developed. On one side, if we excluded the uniform application of the "best available technology," BAT (see earlier), we may select the strategy based exclusively on the effluent standards of the new legislation. This would significantly improve water quality (see Table 13.1). However, it would also be rather expensive, requiring more than 30 million USD (roughly equivalent to thousand million Sk).

On the other side, DO-based least-cost policies show significant saving possibilities. Some of them--as pointed out--are too vulnerable and risky. However, the DO>5 mg/l strategy proved to be rather robust and acceptable. The investment cost requirement is less than half of the previous effluent based policy and the achievable water quality is nearly the same (see Table 13.5). The difference in minimum DO and maximum BOD5 levels is less than 5%, i.e. non-significant and non-detectable (NH_4-N obviously exhibits somewhat larger deviation).

A more detailed comparison of the two policies can be seen in Table 13.12 including not only costs and receiving water quality but also treatment configurations (note that Level 1 indicates present treatment for most of the sites and the highest number refers to BAT, see Chapter 11 and Appendix 11.1). It can be clearly seen that the effluent standard based strategy nearly always leads to higher level of wastewater treatment. An exception is formed by the middle stretch of the river where the extremely poor present quality forces the same technology for the regional least-cost strategy as given by effluent standards.

Uniform effluent standards generally lead to uniform technologies. However, this is not the case if the upgrading of existing facilities overloaded to different extents serve also as viable alternatives. In this sense, both strategies resulted in technologies which may vary from site to site. The least-cost policy did not lead to a particularly preferred technology, although most of the up-grading would be based on chemical enhancement (compare Table 13.12 and Appendix 11.1). However, more or less the same statement is also valid for the effluent standard based strategy, suggesting higher cost-effectiveness on the treatment plant level without having a regional consideration. This is an obvious consequence of our discussion in Chapter 11.

In summary, our recommendation is to implement the DO>5 mg/l least-cost policy as a short-term policy which can then be further expanded as financial resources become available. For the purpose of enforcement, regionally variable effluent standards can be used - like in several countries - which also belong to the results of the strategy development.

Table 13.12 Comparison of treatment strategies: Slovak effluent standard based strategy vs.DO≥5 least-investment-cost policy

Hand-lova	Lehota	Prie-vidza	Parti-zanske	Bano-vce	Topol-cany	Nitra	Zl.Mo-ravce	Vrable	Surany	Nove Zamky
										9
					8					8
					7	7				7
					6	6				6
		5		5	5	5				5
4	4	4		4	4	4	4			4
3	3	3	3	3	3	3	3	3	3	3
2	2	2	2	2	2	2	2	2	2	2
1	1	1	1	1	1	1	1	1	1	1
0	0	0	0	0	0	0	0	0	0	0

Legend for Table 13.12:

	Min. DO (mg/l)	Max. BOD (mg/l)	Max NH4-N (mg/l)
Slovak effluent standard based policy (32.1 million USD)	5.4	11.3	2.3
DO≥5 mg/l least-cost policy (15 million USD)	5.2	11.4	3.2

14. CONCLUSIONS

(1) The major problem of the Nitra River basin is the extremely poor water quality (Class IV-V according to the existing classification system) prohibiting most of the water uses (including drinking water supply, recreation and fishery) due to municipal and industrial emissions coupled with a low level of existing wastewater treatment. The water quality problem is associated with relatively scarce water resources which may cause management difficulties particularly in summer months. Industrial discharges cause mostly local water quality changes which can be solved by well-defined actions without significant research efforts. In contrast, municipal emissions call for the development of a regional control policy. The new legislation is based on a mix of effluent and ambient water quality standards (resembling the requirements of the European Community) opening avenues for the development of least-cost policies crucial under the present economic conditions. It specifies a transition period up to 2005 when standards will be less stringent subsequently water quality standards will be tightened.

(2) An emission inventory was prepared on the basis of existing information and additional data collection. The role of non-point sources was found to be negligible at present. Municipal emissions contribute to about two-third of the total BOD5 discharge in the catchment. They primarily affect oxygen and nutrient households in the river system.

(3) Water quality has remained at about the same poor quality for the past twenty years (i.e. its deterioration took place earlier). At present, it is characterized by low dissolved oxygen (sometimes close to depletion) and high levels of coliform bacteria, BOD-5 (around 30 mg/l), COD, NH4-N, TP, dissolved solids and arsenic. The existing monitoring network consists of 26 locations with a monthly one sampling frequency. A revision of the system is recommended to include the co-monitoring of emissions, the introduction of increased frequency at the most important locations (e.g. at the mouth) to be able to properly estimate trends, annual averages and certain probability levels (e.g. for classification), the detection of non-point sources (of growing importance in the future), nutrients (with their detailed fractions), micropollutants, sediment contamination, as well as a detailed biological assessment.

(4) Two longitudinal water quality profile observations were performed and evaluated. A load response relationship expressed by alternative water quality models was developed to relate ambient water quality to emissions, as well as to describe their changes. Such a relation is the pre-requisite to establishing an ambient criteria-based regional water quality management strategy. Detailed measurements were done under low flow conditions to be able to determine elements and parameters of the critical, design scenario for the policy analysis.

(5) A number of water quality models describing the balance of dissolved oxygen and nutrients were applied; from the simple Streeter-Phelps to the latest, complex version of U.S. EPA's QUAL model family. Model versions were calibrated and validated by using data of the longitudinal profile experiment. The models performed equally well, leading to the selection of a relatively simple, three state variable, extended Streeter-Phelps model for policy purposes. This selection is justified for two reasons: the large, accurate data need of complex models and methodological difficulties to incorporate them into a policy, optimization framework.

(6) The robust and generic method of Hornberger, Spear and Young based on the so called behavior definition was employed to parameter estimation and uncertainty analysis. It offers parameters together with their distribution (being model specific) which can be used for a Monte Carlo simulation and a risk analysis as contrasted to a deterministic procedure. Model parameters obtained were in overall harmony with broad recommendations in the literature. BOD-5 decay rates were higher than usual due to the presence of often only partial biological wastewater treatment and the small water depth, while the reaeration rate was approximately half that suggested in the literature.

(7) Existing municipal wastewater treatment plants--most of them significantly overloaded--were analyzed in detail. A number of well-proven technologies were selected (such as primary treatment, chemically enhanced primary treatment, CEPT, primary precipitation, biological treatment with the activated sludge process - low or high load, biological/chemical treatment, BC, and its extension with (partial) denitrification, BCDN) and generic "cost functions" were developed for them on the basis of Western experiences. The functions express the relationship of effluent quality (BOD-5, DO, TSS, NH4-N, NO3-N, TP etc.) and investment costs and operation, maintenance and repair costs (including of sludge processing). It was shown that mechanical/chemical treatment and biological/chemical-one are particularly cost-effective. The innovative, low dosage chemical upgrading of highly overloaded existing biological plants is especially attractive requiring minimal investments (the excess amount of sludge produced remains small due to the low dosage). The applicability of low dosage chemical upgrading was analyzed by laboratory experiments at several treatment plants. The marginal cost of nitrogen removal is high and thus such a technology should have a lower (and later) priority (obviously depending on needs of receiving waters). All these suggest upgrading or constructing treatment plants in a phased fashion, further increasing cost-effectiveness (subsequent steps of such a development can be CEPT, BC and BCDN). On the basis of the generic cost functions and site specific features a number of alternatives were proposed for each municipality (their number ranged between four and ten) which served as direct inputs to the policy, optimization model.

(8) The joint hydraulic-water quality model, the parameter estimation and uncertainty analysis routine, and information on treatment alternatives, tools and methodologies (e.g. data base, graphical interface) were integrated in a multiple pollutant water quality control policy model or (prototype) decision support system. For the purpose of optimization dynamic programming was applied. Its advantage is that it can handle both, linear and non-linear water quality models (in terms of the load responses) and it utilizes structural properties of river basin pollution problems (a control measure has no upstream impact). Objectives of the model can be formulated to minimize the total annual cost or the initial investment cost (which is of crucial importance if financial resources are scarce). The major constraints to be specified in the present version of the model to meet ambient water quality standards are DO, BOD-5 and NH_4-N. Additional constraints can incorporate effluent standards and the prescription of the minimum level of treatment (e.g. primary treatment or the current level) the usage of which can lead to mixed policies. The system is completed by two different types of *a posteriori* analyses. The first simply evaluates the degree of violating ambient criteria by using the water quality simulation model together with distributions from the HSY estimation procedure in a Monte Carlo framework. The second is a "regret analysis" which assesses the economic and

environmental consequences (e.g. over-expenditures and violating DO levels set, respectively) if a scenario deviates from the design one (critical low flow in our present case together with parameters associated). For the Nitra River, a strong focus was put on the role of the reaeration coefficient, due to it being the most important parameter. Extensions of the present methodology is underway in several directions (e.g. the direct incorporation of uncertainties, economic instruments and emissions as well as water quality components other than considered here, multiobjective assessment and scheduling) to assure its broad applicability.

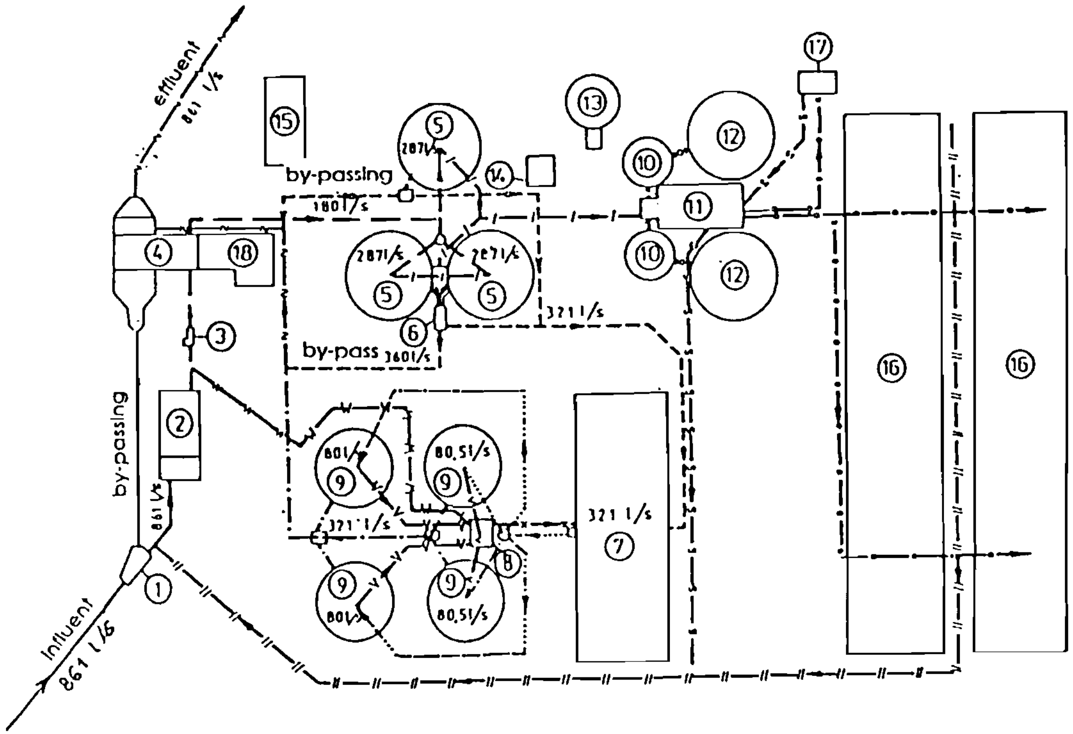
(9) A large number of strategies were developed and analyzed, which were based on different ambient criteria, effluent standards and their mix. The range of realistic expenditures was extremely broad (between 3 and 95 million USD) depending on the policy formulation. These results indicate the possibility of significant saving . In contrast, the variation of receiving water quality was much smaller. The most expensive solution is to replace all the treatment plants with new ones satisfying the most stringent recommendations of the European Community (corresponding to nearly 96 million USD or 3000 million Sk). The present (and future) Slovak effluent standard system implies an investment of 32-35 million USD. A least-cost policy leading to Class III water (in terms of DO, BOD-5 and NH4-N) is roughly equivalent to the former one. The water quality is identical for all cases and the BAT policy provides little improvement (due to non-linearities and the presence of industrial as well as other, "non-controllable" discharges). This feature clearly shows the extreme importance of selecting water quality goals together with evaluating the economic implications (and the need to develop an integrated strategy covering all the emissions of various origins). Often a small improvement in ambient water quality (say 1 mg/l in the DO level) can lead to ten million USD in additional investment.

(10) The first step of managing water quality of a river is to restore dissolved oxygen conditions. Least-cost policies developed on the basis of ambient DO criteria showed a number of attractive features. An investment of about 15 million USD would improve the minimum DO level from about a poor quality of 2 mg/l, the current level, to 5 mg/l characterized as medium or good quality in terms of DO. Since control actions influence several water quality constituents simultaneously (stemming from the multiple pollutant nature of the problem), DO based least-cost policies lead to significant improvement with regard of other components as well. Although this policy is much cheaper than that based exclusively on Slovak effluent standards, the difference in receiving water quality (DO, BOD-5 and NH4-N) is negligible. The strategy and possible overexpenditures are not sensitive to uncertainties in parameters of the water quality model. As contrasted to effluent standard based policies, the least-cost ones are rather non-uniform. Technologies can vary from site to site together with the desired effluent quality. These latter belong, however, to results of the policy and can be used as regionally variable standards for enforcement (several countries follow this practice). The DO>5 mg/l policy is the recommended short-term, least-cost strategy. A long-term strategy can be obtained by a sequence of least-cost policies under gradually tightened criteria as proposed by the new Slovak legislation. The realization of such policies is fully in harmony with the idea of multi-stage waste water treatment development.

APPENDICES

APPENDIX 6.1	**Technology Scheme of the Nové Zámky wastewater treatment plant**
APPENDIX 7.1	**The System of Water Quality Standards Used in Slovakia**
APPENDIX 7.2	**The Monitoring Stations in the Nitra Subwatershed**
APPENDIX 7.3	**List of Surface Water Quality Parameters Regularly Monitored in the Nitra Subwatershed**
APPENDIX 7.4	**Classification of Water Quality of the Nitra River Basin in 1989-1990**
APPENDIX 7.5	**Recommendations for the Enhancement of the Existing Surface Water Quality Monitoring System**
APPENDIX 11.1	**Summary of Municipal Wastewater Treatment Alternatives**

APPENDIX 6.1 TECHNOLOGY SCHEME OF THE NOVÉ ZÁMKY WASTEWATER TREATEMENT PLANT



L E G E N D

1 Lightening chamber
2 Rough pre-treatment
3 Flow measurement flume
4 Pumping station
5 Settling tanks
6 Lightening chamber
7 Activation tanks
8 Sludge pumping station from the secondary settling ta
9 Secondary settling tanks
10 Digestion tanks
11 Boiler room, mechanical hall
12 Sludge collector
13 Gas container
14 Mechanical hall, gas compression plant
15 Workshops
16 Sludge fields
17 Mechanical sludge dewatering
18 Operational building

- raw waste water
- waste water after rough pre-treatment
- waste water after mechanical treatment
- waste water after biological treatment
- waste water - efluent
- sludge water from the sludge collector and from the mechanical sludge dewatering
- sludge water from the sludge fields
- activated sludge
- flocculated sludge from the secondary settling
- recirculated sludge
- waste sludge
- raw sludge
- digested sludge
- digested sludge to the sludge fields and to the mechanical sludge dewatering

Q24 = 151,0 l/s
Qmax = 260,0 l/s
Qmax.rain.= 680,0 l/s

Appendix 7.1 The system of water quality standards used in Slovakia

From the 1st of July 1990 the Czechoslovak standard CSN 75 7221 has become valid and it replaced the previous Czechoslovak standard CSN 83 0602.

The classification of surface water quality according to CSN 75 7221 is based on the evaluation of selected water quality parameters which this standard divides into 6 groups:

A - parameters of oxygen regime
B - basic chemical parameters
C - additional chemical parameters
D - heavy metals
E - biological and microbiological parameters
F - radioactivity parameters

Every individual parameter of a group is assigned a class separately from the others. The resulted class in each group is determined according to the most unfavorable water quality parameter in the group.

Classification of surface water quality has to be based at least on the parameters listed as follows:

A. dissolved oxygen, BOD-5, COD-Mn or COD-Cr
B. pH, water temperature, soluble substances or conductivity, suspended solids, ammonia nitrogen, nitrate nitrogen, total P
C. calcium, magnesium, chloride, sulfate, detergents, nonpolar extractable matter, chlorinated organic components
D. mercury, cadmium, arsenic, lead
E. saprobity index, coliform bacteria or fecal coliform bacteria
F. total volume activity τ, total volume activity

According to the standard CSN 75 7221 surface waters are divided into 5 classes:

I class - very clear water
II class - clear water
III class - polluted water
IV class - intensively polluted water
V class - very intensively polluted water

The number of surface water samples to be used as a basis for the classification must be at least 24. The shortest evaluated time interval should be 1 year and the longest one usually ought not to be longer than 5 years. The 90% probability value is to be compared with the limits provided by the standard. On the basis of this comparison the appropriate class is to be assigned to the water quality parameter in question. Finally the resulted class of a group is to be determined according to the most unfavorable individual parameter class within a group.

In the case that the frequency sampling is 12 times a year, it is needed to merge values from two monitoring years. If there is a need to evaluate water quality for the yearly time interval with only 12 monthly values an average from 3 most unfavorable parameter values is to be used as a basis for classification.

The class limits in the CSN 75 7221 are strict, especially in group E - biological and microbiological parameters. For example, with respect to the parameter psychrophile bacteria even the upper parts of rivers are evaluated as polluted water. The previous standard did not include this parameter altogether. The limit values for the parameters groups are as follows according to the CSN 75 7221:

A - Parameters of Oxygen Regime

		Class				
Parameter	Unit	I	II	III	IV	V
Dissolved oxygen	mg/l	>7	>6	>5	>3	<3
BOD-5	mg/l	<2	<5	<10	<15	>15
COD-Mn	mg/l	<5	<10	<15	<25	>25
COD-Cr	mg/l	<15	<25	<35	<55	>55
TOC	mg/l	<5	<8	<11	<17	>17
S^2	mg/l	*	*	*	<0.02	>0.02

B - Basic Chemical Parameters

		Class				
Parameter	Unit	I	II	III	IV	V
pH	-	6-8.5	6-8.5	6-8.5	5.5-9	<5.5,>9
Water temperature	°C	<22	<23	<24	<26	>26
Soluble subst.	mg/l	<300	<500	<800	<1200	>1200
Conductivity	mS/m	<40	<70	<110	<160	>160
Suspended solids	mg/l	<20	<40	<60	<100	>100
Total Fe	mg/l	<0.5	<1.0	<2.0	<3.0	>3.0
Total Mn	mg/l	<0.05	<0.1	<0.3	<0.8	>0.8
N-NH4	mg/l	<0.3	<0.5	<1.5	<5.0	>5.0
N-NO2	mg/l	<0.002	<0.005	<0.02	<0.05	>0.05
N-NO3	mg/l	<1.0	<3.4	<7.0	<11	>11
N-org.	mg/l	<0.5	<1.0	<2.5	<3.5	>3.5
Total P	mg/l	<0.03	<0.15	<0.4	<1.0	>1.0

C - Additional Chemical Parameters

	Class					
Parameter	Unit	I	II	III	IV	V
Cl	mg/l	<50	<200	<300	<400	>400
SO4	mg/l	<80	<150	<250	<300	>300
Ca	mg/l	<75	<150	<200	<300	>300
Mg	mg/l	<25	<50	<100	<200	>200
Phenols	mg/l	<0.002	<0.01	<0.02	<0.5	>0.5
Tenzides anion.	mg/l	*	<0.5	<1.0	<2.0	>2.0
Total Mn	mg/l	<0.05	<0.1	<0.3	<0.8	>0.8
Non-polar ext. m.	mg/l	*	<0.05	<0.1	0.3	>0.3
CN^-	mg/l	*	*	<0.2	<0.5	>0.5

D - Heavy Metals

		Class				
Parameter	Unit	I	II	III	IV	V
Hg	ug/l	<0.1	<0.2	<0.5	<1.0	>1.0
Cd	ug/l	<3.0	<5.0	<10	<20	>20
Pb	ug/l	<10	<20	<50	<100	>100
As	ug/l	<10	<20	<50 <	100	>100
Cu	ug/l	<20	<50	<100	<200	>200
Cr	ug/l	<20	<100	<200	<500	>500
Cr^{VI}	ug/l	*	<10	<20	<50	>50
Co	ug/l	<10	<20	<50	<100	>100
Ni	ug/l	<20	<50	<100	<200	>200
Zn	ug/l	<20	<50	<100	<500	>500
Va	ug/l	<10	<20	<50	<100	>100
Ag	ug/l	*	<10	<20	<50	>50

E - Biological and Microbiological Parameters

		Class				
Parameter	Unit	I	II	III	IV	V
Bios. saprobity index	-	<1.2	<2.2	<3.2	<3.7	>3.7
Psychrophil bac.	KTJ/ml	<500	<1000	<5000	$<10^4$	$>10^4$
Coli bac.	KTJ/ml	<1	<10	<100	$<10^3$	$>10^3$
Fec. coli bac.	KTJ/ml	<0.2	<2.0	<20	<200	>200
Enterococci	KTJ/ml	<0.1	<1.0	<10	<100	>100

NOTE: * means below the sensitivity level

Appendix 7.2 The Monitoring Stations in the Nitra Subwatershed

Station	Code	Name	River	River km
P1	V388000D	Klacno	Nitra	165.0
P2	V393000D	Nedozery	Nitra	149.0
P3	V405510D	Handlová over	Handlovka	29.0
P4	V400510D	Handlová under	Handlovka	22.7
P5	V410510D	Kos	Handlovka	1.2
P6	V414000D	Nováky over	Nitra	132.5
P7	V416000D	Chalmová	Nitra	123.8
P8	V423500D	Partizánske over	Nitra	115.7
P9	V439010D	Partizanske	Nitrica	0.2
P10	V441500D	Chynorany	Nitra	105.5
P11	V442500D	Bosany over	Nitra	101.6
P12	V457000D	Bánovce n.Bebravou	Radisa	0.5
P13	V487500D	Krusovce	Bebrava	3.4
P14	V495020D	Práznovce	Nitra	98.2
P15	V497000D	Nitrianska Streda	Nitra	91.1
P16	V538000D	Luzianky	Nitra	65.3
P17	V544500D	Nitrany	Nitra	47.8
P18	V554000D	Zlaté Moravce over	Zitava	88.2
P19	V559000D	Tesárske Mlynany	Zitava	78.0
P20	V575210D	Vráble	Telinský p.	0.5
P21	V580000D	Lúcnica n.Zitavou	Zitava	55.2
P22	V590000D	Dolný Oháj under	Zitava	32.7
P23	V598510D	Surany over	Stará Nitra	4.0
P24	V598D00D	Surany under	Stará Nitra	0.8
P25	V599500D	Nové Zámky	Nitra	14.5
P26	V775500D	Komoca	Nitra	6.5

Appendix 7.3 List of surface water quality parameters regularly monitored in the Nitra subwatershed

(continued on the next page)

Number of monitoring station		*1*	*2*	*3*	*4*	*5*	*6*	*7*	*8*	*9*	*10*	*11*
GENERAL PARAMETERS												
1	Sampling dates	70	74	74	74	70	70	70	70	74	74	90
2	Sampling hours	70	74	74	74	70	70	70	70	74	74	90
3	Q daily average	70	74	74	74	70	70	70	70	X	74	90
4	Q immediate	X	74					X		74	74	
A.- OXYGEN REGIME												
1	Dissolved oxygen	70	74	74	74	70	70	70	70	74	74	90
2	BOD-5	70	74	74	74	70	70	70	70	74	74	90
3	COD-Mn	70	74	74	74	70	70	70	70	74	74	90
B.- BASIC CHEMICAL PARAMETERS												
1	pH	70	74	74	74	70	70	70	70	74	74	90
2	Water temperature	70	74	74	74	70	70	70	70	74	74	90
3	Total soluble sub	70	74	74	74	70	70	70	70	74	74	90
4	Conductivity	70	74	74	74	70	70	70	70	74	74	90
5	Total suspended so	70	74	74	74	70	70	70	70	74	74	90
6	Total iron	70	75	75	75	70	70	70	70	75	75	90
7	Total manganese	70										
8	Ammonia nitrogen	90	90	90	90	90	90	90	90	90	90	90
9	Nitrite nitrogen											
1	Nitrate nitrogen	90	90	90	90	90	90	90	90	90	90	90
1	Organic nitrogen											
C.- ADDITIONAL CHEMICAL PARAMETERS												
1	Chloride	70	75	75	75	70	70	70	70	75	75	90
2	Sulfate	70	75	75	75	70	70	70	70	75	75	90
3	Calcium	70	75	75	75	70	70	70	70	75	75	90
4	Magnesium	70	75	75	X	X		70	70	75		
5	Detergents		74		74	70				74	75	
6	Nonpolar extractable matter											
7	Total cyanide											
D.- HEAVY METALS												
1	Cadmium					90	90					
2	Arsenic				70	70		70	70		74	
3	Copper											
4	Chromium (VI)											
5	Nickel											
6	Zinc											
E.- BIOLOGICAL AND MICROBIOLOGICAL PARAMETERS												
1	Bioseston saprobity index	78	78	78	78	78	78	78	78	78	78	90
2	Psychrophil. bacteria	70	74	74	74	70	70	70	70	74	74	90
3	Coliform bacteria	70	74	74	74	70	70	70	70	74	74	90
F.- RADIOACTIVITY PARAMETERS												
1	Total alpha-activity											
2	Radium 226											

NOTES:

X - sampling stations or parameters were monitored occasionally

* - parameters were monitored only up to the year 1989

List of surface water quality parameters regularly monitored in the Nitra subwatershed (continued on the next page).

Number of monitoring station		*12*	*13*	*14*	*15*	*16*	*17*	*18*	*19*	*20*	*21*	*22*
GENERAL PARAMETERS												
1	Sampling dates	74	70	70	74	70	74	70	74	90	74	70
2	Sampling hours	74	70	70	74	70	74	70	74	90	74	70
3	Q daily average	75	70	70	74	70	X	70	74		74	70
4	Q immediate	X			74		X				X	
A.- OXYGEN REGIME												
1	Dissolved oxygen	74	70	70	74	70	74	70	74	90	74	70
2	BOD-5	74	70	70	74	70	74	70	74	90	74	70
3	COD-Mn	74	70	70	74	70	74	70	74	90	74	70
B.- BASIC CHEMICAL PARAMETERS												
1	pH	74	70	70	74	70	74	70	74	90	74	70
2	Water temperature	74	70	70	74	70	74	70	74	90	74	70
3	Total soluble sub	74	70	70	74	70	74	70	74	90	74	70
4	Conductivity	74	70	70	74	70	74	70	74	90	74	70
5	Total suspended so	74	70	70	74	70	74	70	74	90	74	70
6	Total iron	75	70	70	75		75	70	75		75	70
7	Total manganese			70	75							
8	Ammonia nitrogen	90	90	90	90	90	90	90	90	90	90	90
9	Nitrite nitrogen				90		90					
1	Nitrate nitrogen	90	90	90	90	90	90	90	90	90	90	90
1	Organic nitrogen					90						90
C.- ADDITIONAL CHEMICAL PARAMETERS												
1	Chloride	75	70	70	75	70	75	70	75	90	75	70
2	Sulfate	75	70	70	75	70	75	70	75	90	75	70
3	Calcium	75	70	70	75	70	75	70	75	90	75	70
4	Magnesium				75		75	70			75	70
5	Detergents	75	70	74	74		74	75	74			
6	Nonpolar extract. matter											
7	Total cyanide							74			74	
D.- HEAVY METALS												
1	Cadmium				90						90	
2	Arsenic			70	75		75					
3	Copper										90	
4	Chromium (VI)							89	74		90	
5	Nickel										90	
6	Zinc										90	
E.- BIOLOGICAL AND MICROBIOLOGICAL PARAMETERS												
1	Bioseston saprobity index	78	78	78	78	78	78	78	78	90	78	78
2	Psychrophil. bacteria	74	70	70	74	70	74	70	74	90	74	70
3	Coliform. bacteria	74	70	70	74	70	74	70	74	90	74	70
F.- RADIOACTIVITY PARAMETERS												
2	Total alpha activity									81		
3	Radium 226									81		

NOTES:

X - sampling stations or parameters were monitored occasionally

* - parameters were monitored only up to the year 1989

List of surface water quality parameters regularly monitored in the Nitra subwatershed (continued).

Number of monitoring station		23	24	26
GENERAL PARAMETERS				
1	Sampling dates	X	74	72
2	Sampling hours	X	74	72
3	Q daily average			73
4	Q immediate		74	
A.- OXYGEN REGIME				
1	Dissolved oxygen	X	74	72
2	BOD-5	X	74	72
3	COD-Mn	X	74	72
B.- BASIC CHEMICAL PARAMETERS				
1	pH	X	74	72
2	Water temperature	X	74	72
3	Total soluble sub	X	74	72
4	Conductivity	X	74	72
5	Total suspended so	X	74	72
6	Total iron	X	75	72
7	Total manganese			
8	Ammonia nitrogen	90	90	90
9	Nitrite nitrogen			90
1	Nitrate nitrogen	90	90	90
1	Organic nitrogen			90
C.- ADDITIONAL CHEMICAL PARAMETERS				
1	Chloride	X	75	72
2	Sulfate	X	75	72
3	Calcium	X	75	72
4	Magnesium		75	72
5	Detergents		74	75
6	Nonpolar extract. matter			
7	Total cyanide			75
D.- HEAVY METALS				
1	Cadmium			90
2	Arsenic			89
3	Copper			
4	Chromium (VI)			89
5	Nickel			
6	Zinc			
E.- BIOLOGICAL AND MICROBIOLOGICAL PARAMETERS				
1	Bioseston saprobity index	X	78	78
2	Psychrophil. bacteria	X	74	75
3	Coliform. bacteria	X	74	72
F.- RADIOACTIVITY PARAMETERS				
2	Total alpha activity			72
3	Radium 226			72

NOTES:

X - sampling stations or parameters were monitored occasionally

- parameters were monitored only up to the year 1989

Appendix 7.4 Classification of water quality of the Nitra River basin in 1989-1990

(Continued on the next page)

	River			Nitrica	Nitra	Nitra	Radica	Bebrava	Nitra	Nitra	Nitra
	Sampling station			Nitrica	Chyno	Bosany	Banov	Kru	Praz	Nitr	Luzi
					rany		ce	covce	novce	Streda	anky
							nad				
				P9	P10	P11	P12	P13	P14	P15	P16
	Parameter	Symbol	Unit								
	A.- OXYGEN REGIME			III	IV	V	III	IV	V	V	V
1	Dissolved	O2	mg/l	I	II	IV	I	I	III	IV	II
	oxygen			9,9	6,9	4,5	8,0	8,6	5,3	3,0	6,6
2	BOD-5	BOD 5	mg/l	II	IV	V	III	IV	V	V	V
				5,0	13,4	15,3	7,7	11,0	27,1	16,7	16,4
3	COD-Mn	COD	mg/l	III	IV	IV	III	IV	V	IV	IV
				12,8	21,1	22,8	14,9	19,4	34,7	17,9	24,1
	B.- BASIC CHEMICAL PARAMETERS			IV	V	V	III	IV	V	V	V
1	pH	pH		I	I	I	I	I	I	I	I
				8,3	8,1	7,9	8,3	8,3	8,0	8,1	8,1
2	Water	T	0C	II	II	I	I	I	II	I	II
	temp.			22,6	22,1	20,2	16,3	21,5	23,0	20,0	22,5
3	Total so-		mg/l	III	V	V	III	III	V	V	V
	luble subst.			638,2	1298,9	1290,0	633,4	633,2	1387,8	1244,2	1221,2
4	Conduc-		mS/m	III	IV	IV	II	III	IV	IV	IV
	tivity			72,2	129,7	134,7	60,3	85,54	141,2	125,5	123,7
5	Total susp.		mg/l	IV	III	V	II	IV	IV	IV	V
	solids			67,4	55,6	111,3	38,0	62,2	63,0	71,0	142,0
6	Total	Fe	mg/l	I	II	II	II	I	II	II	II
	iron			0,44	0,72	0,93	0,58	0,48	0,65	0,55	0,77
7	Total	Mn	mg/l	-	-	-	-	-	-	I	-
	manganese									0,0	
8	Ammonia	N-NH4+	mg/l	II	IV	IV	III	IV	V	IV	IV
	nitrogen			3,5	2,88	0,46	2,92	2,89	0,61	2,19	5,09
9	Nitrite	N-NO2-	mg/l	-	-	-	-	-	-	V	-
	nitrogen									0,21	
10	Nitrate	N-NO3-	mg/l	III	II	II	II	III	II	II	II
	nitrogen			4,3	2,8	3,1	2,6	4,1	2,4	2,9	2,7
	C.- ADDITIONAL CHEMICAL			II	IV	IV	II	II	V	III	III
	PARAMETERS										
1	Chloride	CL-	mg/l	I	IV	IV	I	I	IV	III	III
				44,3	364,6	356,0	39,9	39,0	387,4	270,7	263,5
2	Sulfate	SO42-	mg/l	I	III	III	I	I	III	III	III
				59,5	182,3	207,7	48,0	64,3	180,4	154,1	165,6
3	Calcium	Ca	mg/l	II	IV	IV	II	II	IV	III	III
				113,2	208,5	215,7	107,3	117,2	203,9	174,3	194,3
4	Magnesium	Mg	mg/l	II	-	-	-	-	-	II	-
				32,2						41,9	
8	Deter-		mg/l	II	II	-	II	II	II	II	-
	gents			0,11	0,19		0,18	0,17	0,26	0,17	
9	Non-polar			-	-	-	-	-	V	-	-
	extr.		mg/l						0,9		
	matter										
10	Total	CN-	mg/l	-	-	-	-	-	-	-	-
	cyanide										

Classification of water quality of the Nitra River basin in 1989-1990.
(Continued on the next page)

	River			Nitra	Zitava	Zitava	Tel. p.	Zitava	Zitava	Stara N
	Sampling station			Nitra	Zlatne	Tes	Vrab	Lucnica	Dol	Surany
				ny	Morav	Mlyna	le	ca	Ohaj	nad
					ce	ny				
				P17	P18	P19	P20	P21	P22	P23
	Parameter	Symbol	Unit							
	A.- OXYGEN REGIME			V	III	IV	V	IV	IV	V
1	Dissolved	O2	mg/l	IV	I	I	II	I	I	III
	oxygen			3,9	9,5	8,4	6,5	9,2	8,2	5,5
2	BOD-5	BOD 5	mg/l	V	III	IV	V	III	III	V
				18,4	6,0	10,6	16,3	9,1	9,7	17,0
3	COD-Mn	COD	mg/l	IV	III	IV	V	IV	IV	IV
				20,6	13,4	17,6	25,6	17,2	17,4	19,3
	B.- BASIC CHEMICAL PARAMETERS			V	III	IV	V	IV	V	V
1	pH	pH		I	I	I	I	I	I	I
				8,0	8,2	8,3	8,1	8,3	8,3	8,0
2	Water	T	0C	II	I	I	I	I	I	III
	temp.			22,8	16,5	17,0	17,2	18,5	20,5	21,0
3	Total so-		mg/l	IV	II	III	V	III	V	V
	luble subst.			1147,0	469,2	620,2	1465,3	685,2	1280,9	1206,0
4	Conduc-		mS/m	IV	II	II	IV	III	V	IV
	tivity			134,2	54,2	65,0	154,7	73,9	178,4	125,7
5	Total susp.		mg/l	III	III	IV	V	III	IV	V
	solids			58,1	59,9	61,0	100,0	50,6	89,7	146,7
6	Total	Fe	mg/l	II	II	I	II	I	I	II
	iron			0,51	0,7	0,33	0,71	0,47	0,45	0,71
7	Total	Mn	mg/l	-	-	-	-	-	-	-
	manganese									
8	Ammonia	N-NH4+	mg/l	V	III	IV	IV	IV	III	V
	nitrogen			5,02	1,04	2,82	2,61	2,01	1,50	6,89
9	Nitrite	N-NO2-	mg/l	V	-	-	-	-	V	-
	nitrogen			0,23					0,137	
10	Nitrate	N-NO3-	mg/l	II	III	III	V	III	III	II
	nitrogen			2,0	3,06	3,7	15,1	5,0	4,0	2,3
	C.- ADDITIONAL CHEMICAL			V	II	II	III	III	V	III
	PARAMETERS									
1	Chloride	CL-	mg/l	III	I	I	II	II	V	III
				271,7	32,8	47,8	174,0	64,7	570,2	276,7
2	Sulfate	SO42-	mg/l	III	II	II	III	II	III	III
				159,0	85,9	96,0	192,3	110,5	179,3	174,7
3	Calcium	Ca	mg/l	III	I	II	III	II	II	III
				188,5	72,5	89,2	192,0	106,3	125,2	149,3
4	Magnesium	Mg	mg/l	II	I	-	-	II	II	-
				37,0	15,0			32,0	49,9	
8	Deter-		mg/l	II	II	II	-	-	-	-
	gents			0,29	0,16	0,19				
9	Non-polar			V	-	-	-	-	-	-
	extr.		mg/l	0,5						
	matter									
10	Total	CN-	mg/l	-	I	-	-	III	-	-
	cyanide				0,0			0,003		

Classification of water quality of the Nitra River basin in 1989-1990.
(Continued on the next page)

	River			Stara N	Nitra	Nitra
	Sampling station			Surany	Nove	Komo
				pod	Zamky	ca
				P24	P25	P 26
	Parameter	Symbol	Unit			
	A.- OXYGEN REGIME			V	V	V
1	Dissolved	O2	mg/l	IV	II	V
	oxygen			4,1	6,2	0,0
2	BOD-5	BOD 5	mg/l	V	V	V
				196,1	28,7	33,0
3	COD-Mn	COD	mg/l	V	V	V
				87,5	43,3	34,4
	B.- BASIC CHEMICAL PARAMETERS			V	IV	V
1	pH	pH		I	I	I
				8,3	8,1	8,1
2	Water	T	0C	III	I	I
	temp.			24,0	21,8	21,8
3	Total so-		mg/l	V	IV	IV
	luble subst.			1200,6	1119,5	1126,7
4	Conduc-		mS/m	IV	IV	IV
	tivity			141,3	127,7	123,0
5	Total susp.		mg/l	V	III	III
	solids			212,0	57,0	44,0
6	Total	Fe	mg/l	III	II	II
	iron			1,06	0,56	0,52
7	Total	Mn	mg/l	-	-	-
	manganese					
8	Ammonia	N-NH4+	mg/l	IV	IV	V
	nitrogen			3,81	4,43	5,3
9	Nitrite	N-NO2-	mg/l	-	-	V
	nitrogen					0,38
10	Nitrate	N-NO3-	mg/l	II	II	II
	nitrogen			2,9	2,5	2,7
	C.- ADDITIONAL CHEMICAL			IV	III	III
	PARAMETERS					
1	Chloride	CL-	mg/l	IV	III	III
				318,5	287,6	233,7
2	Sulfate	SO42-	mg/l	III	III	III
				178,0	157,6	169,3
3	Calcium	Ca	mg/l	II	III	III
				186,2	158,3	156,5
4	Magnesium	Mg	mg/l	II	II	II
				36,4	39,5	41,3
8	Deter-		mg/l	II	-	II
	gents			0,32		0,32
9	Non-polar			-	-	-
	extr.		mg/l			
	matter					
10	Total	CN-	mg/l	-	-	III
	cyanide					0,003

Classification of water quality of the Nitra River basin in 1989-1990.
(Continued on the next page)

	River			Nitra	Nitra	Handl.	Handl.	Handl.	Nitra	Nitra	Nitra
	Sampling station			Kla	Nedo	Hand	Hand	Kos	No	Chal	Parti
				cno	zery	lova	lova		vaky	mova	zanske
						nad	pod		nad		nad
				P1	P2	P3	P4	P5	P6	P7	P8
	Parameter	Symbol	Unit								
	D.- METALS			-	-	-	I	I	I	V	V
1	Mercury	Hg	mkg/l	-	-	-	-	-	-	-	-
2	Cadmium	Cd	mkg/l	-	-	-	-	I	I	-	-
								0,0	0,0		
4	Arsenic	As	mkg/l	-	-	-	I	I	-	V	V
							10,0	0,0		170,0	115,2
5	Copper	Cu	mkg/l	-	-	-	-	-	-	-	-
7	Chromium	CrVI	mkg/l	-	-	-	-	-	-	-	-
9	Nickel	Ni	mkg/l	-	-	-	-	-	-	-	-
10	Zinc	Zn	mkg/l	-	-	-	-	-	-	-	-
	E.- BIOLOG. AND MICROBIOL.			IV	V	V	V	V	V	V	V
	PARAMETERS										
1	Bioseston			II	III	II	V	V	III	V	IV
	index			1,86	2,49	2,14	3,94	3,85	2,95	3,72	3,54
2	Psychroph.		thous.	III	V	V	V	V	V	V	V
	bacteria		cells/ml	5,0	63,3	31,3	185,4	242,6	90,6	106	92,6
3	Coliform		thous.	IV	V	IV	V	V	V	V	V
	bacteria		cells/ml	0,105	2,7	0,6	135,2	41,68	11,1	7,6	5,6

Classification of water quality of the Nitra River basin in 1989-1990.
(Continued on the next page)

	River			Nitrica	Nitra	Nitra	Radica	Bebrava	Nitra	Nitra	Nitra
	Sampling station			Nitrica	Chyno	Bosany	Banov	Kru	Praz	Nitr	Luzi
					rany		ce	covce	novce	Streda	anky
							nad				
				P9	P10	P11	P12	P13	P14	P15	P16
	Parameter	Symbol	Unit								
	D.- METALS			-	V	-	-	-	IV	IV	-
1	Mercury	Hg	mkg/l	-	-	-	-	-	-	-	-
2	Cadmium	Cd	mkg/l	-	-	-	-	-	-	1	-
										0,0	
4	Arsenic	As	mkg/l	-	V	-	-	-	IV	IV	-
					142,6				90,0	100,0	
5	Copper	Cu	mkg/l	-	-	-	-	-	-	-	-
7	Chromium	CrVI	mkg/l	-	-	-	-	-	-	-	-
9	Nickel	Ni	mkg/l	-	-	-	-	-	-	-	-
10	Zinc	Zn	mkg/l	-	-	-	-	-	-	-	-
	E.- BIOLOG. AND MICROBIOL.			V	V	V	V	V	V	V	V
	PARAMETERS										
1	Bioseston			III	III	III	III	III	IV	IV	III
	index			2,55	3,0	2,96	2,81	2,83	3,4	3,4	2,9
2	Psychroph.		thous.	V	V	V	V	V	V	V	V
	bacteria		cells/ml	28,5	72,6	102,3	123,4	41,3	144,9	146,7	85,2
3	Coliform		thous.	IV	V	V	V	V	V	V	V
	bacteria		cells/ml	0,5	9,3	2,1	2,6	4,8	18,6	14,3	7,3

Classification of water quality of the Nitra River basin in 1989-1990.
(Continued on the next page)

	River			Nitra	Zitava	Zitava	Tel. p.	Zitava	Zitava	Stara N
	Sampling station			Nitra	Zlatne	Tes	Vrab	Lucnica	Dol	Surany
				ny	Morav	Mlyna	le	ca	Ohaj	nad
					ce	ny				
				P17	P18	P19	P20	P21	P22	P23
	Parameter	Symbol	Unit							
	D.- METALS			IV	I	I	-	II	-	-
1	Mercury	Hg	mkg/l	-	-	- .	-	-	-	-
2	Cadmium	Cd	mkg/l	-	-	-	-	I	-	-
								0,0		
4	Arsenic	As	mkg/l	IV	-	-	-	-	-	-
				55,2						
5	Copper	Cu	mkg/l	-	-	-	-	I 3,3	-	-
7	Chromium	CrVI	mkg/l	-	0,0	0,0	-	0,0	-	-
9	Nickel	Ni	mkg/l	-	-	-	-	30,0	-	-
10	Zinc	Zn	mkg/l	-	-	-	-	40,0	-	-
	E.- BIOLOG. AND MICROBIOL.			V	V	V	V	V	V	V
	PARAMETERS									
1	Bioseston			III	III	III	III	III	III	III
	index			3,04	2,50	2,70	2,66	2,63	2,69	2,67
2	Psychroph.		thous.	V	V	V	V	V	V	V
	bacteria		cells/ml	108,1	19,2	41,5	75,0	34,1	58,9	140,7
3	Coliform		thous.	V	IV	V	V	V	V	V
	bacteria		cells/ml	21,0	0,6	1,3	3,39	3,89	11,4	1,11

Classification of water quality of the Nitra River basin in 1989-1990.
(Continued)

	River			Stara N	Nitra	Nitra
	Sampling station			Surany	Nove	Komo
				pod	Zamky	ca
				P24	P25	P 26
	Parameter	Symbol	Unit			
	D.- METALS			-	-	IV
1	Mercury	Hg	mkg/l	-	-	-
2	Cadmium	Cd	mkg/l	-	..	I
						0,0
4	Arsenic	As	mkg/l	-	-	IV
						52,6
5	Copper	Cu	mkg/l	-	-	-
7	Chromium	CrVI	mkg/l	-	-	0,0
9	Nickel	Ni	mkg/l	-	-	-
10	Zinc	Zn	mkg/l	-	-	-
	E.- BIOLOG. AND MICROBIOL.			V	V	V
	PARAMETERS					
1	Bioseston			V	IV	IV
	index			4,0	3,32	3,21
2	Psychroph.		thous.	V	V	V
	bacteria		cells/ml	402,9	377,8	379,9
3	Coliform		thous.	V	V	V
	bacteria		cells/ml	181,3	26,16	28,7

Appendix 7.5 Recommendations for the Enhancement of the Existing Surface Water Quality Monitoring System

1. Introduction

As it was outlined in the Chapter 7, the water quality monitoring system can serve many purposes: reporting the state of the water environment, using means, percentiles or other statistical criteria; detecting temporal trends; checking the suitability of water for specific purposes; checking the pollution load to the receiving waterbody. The monitoring procedure itself should be set up according to the priorities of different monitoring goals. Thus, if the emphasis is on estimating the load to the receiving waterbody, more analyses should be done at the mouth of the stream. Likewise, if the main purpose is to control the domestic water supply, the sampling sites immediately upstream of the corresponding uptakes should receive the most attention. As the requirements in many cases are often controversial, the setting of monitoring scheme often is a result of a compromise (Chapter 7).

In the case of monitoring of water quality in the Nitra River basin, the objectives of the existing monitoring scheme were set up approximately twenty years ago. It is desirable to reevaluate the procedure, perhaps to de-emphasize certain past priorities and define new ones keeping in mind changes both in the human activity in the region and developments in the water quality management since that time. The data collected in the course of the monitoring should serve as a basis for this reevaluation. This comprehensive analysis should be performed by the responsible national institutions and agencies, and therefore it is not the objective of this work to take all the details into consideration. However, some of the important issues of setting up the monitoring program in the Nitra River basin will be discussed below.

The monitoring program in the Nitra River basin as it is currently set up serves the following goals (in order of importance):

- Reporting the general information about the state of environment (on the basis of the classification scheme)
- Overall checking of the amount of pollution (mostly conventional pollutants are considered)
- Detecting accidents causing considerable pollution of the river water

It can further be used for the additional purposes of:

- Estimating averages
- Detecting trends
- Estimating the load to the receiving waterbodies (the Danube and the Black Sea)
- Checking the possibility of water uptake for different purposes
- Checking the conditions of aquatic life preservation
- Providing feedback information for the water quality management (upgrading of the wastewater treatment plants, selection and adjustment of the policy instruments)

Some particular issues of the setup of the monitoring system are discussed below. Summary of the recommendations can be found in the Section 7.4.

2. Location of Monitoring Sites

From Figure 7.1 it can be seen that in most cases the monitoring points have been deployed in pairs, one immediately upstream of the municipality and the other downstream, with the purpose of studying the effects of the emissions. Sometimes the sample of the water from the downstream site is done before the complete mixing of the effluent with the river is achieved, and the sample therefore cannot represent the conditions in the river. This is true, for example, for the monitoring point Praznovce, which is located less than a kilometer downstream from the mouth of the tributary Bebrava.

A special care is the absence of emissions control in the monitoring scheme. The monitoring provided by the emitters themselves is not procedurally and temporally coherent with the ambient water quality sampling, and its results are unavailable in compound and consistent manner for the purposes of any sensible generalization. Thus to use the information provided in the policy-oriented way the inclusion of emission monitoring is a necessary precondition. Flow measurements should supplement the sampling of the effluent water to provide the notion of emitted load.

3. Set of Water Quality Measurements

Currently, mostly so-called "traditional" pollutants are monitored in the Nitra River basin (see Chapter 7). There has been no detailed assessment made of the river ecosystem, and therefore at the moment no clear understanding of how pollution affects aquatic life. Therefore, it is not possible to use the monitoring data for the control of aquatic life preservation. To address this problem, a hydrobiological study should be undertaken to compare the condition of river biota in the polluted and relatively clean river stretches in the basin. This issue is closely connected to that of nutrient assessment and control. Biogenic elements such as nitrogen and phosphorus can give rise to excessive growth of algae (eutrophication). Chlorophyll-a, nitrogen and phosphorus fractions analyses should be made a part of monitoring system, on a less frequent basis to reduce costs.

The analysis of the suitability of the river to support aquatic life can be facilitated with toxicity tests (Chapter 7). They serve as an indicator of the presence of the hazardous pollutants originating from industry or agriculture. Their usage can be especially helpful in cases when the traditional parameters are within the "admissible" limits. When organic waste loads decrease, the toxicity tests will provide valuable information on the water quality problems related to non-traditional pollutants.

The analysis of current monitoring results shows indicate the presence of certain indicators of industrial pollutants such as arsenic and dissolved solids. Of the heavy metals only arsenic is monitored more or less regularly. Copper, zinc, cadmium, chromium and nickel are in the list but monitored at two stations only. Mercury, lead, chlorinated and polycyclic organics, all of them highly toxic, are not monitored at present. The analyses of these substances could be performed less frequently than the others to save costs.

Organic micropollutants at present are practically outside of the monitoring activity. The low-molecular organic micropollutants such as dichlorobenzene, dichloroethane, etc. in the Nitra River originate mainly from the industrial emissions (most notably from the chemical complex at Novaky; Chapter 7 and 8). It would make sense to perform a regular analysis of the samples taken from the locations Chalmova and Partizanske (river km 124 and 116, respectively) and from effluent of the chemical plant. Another consideration is the excessive concentration of pesticides (e.g. lindane), originating from agriculture. To control this pollution it is necessary to include pesticide analysis in the monitoring program in the middle and lower parts of the river, the locations of intensive agriculture. Finally, insufficient coverage of the high-molecular nonvolatile persistent organics (such as oil products) is found in the monitoring scheme.

The pollution of the water column and high suspended solids concentration (Chapter 7) can lead to the accumulation of many of the pollutants in the river sediments. As pollution control measures are undertaken, concentrations in the water column will drop, and certain portion of the stored pollution can be released from the sediment. It is therefore necessary to perform an analysis of the river sediments in order to evaluate the potential of secondary pollution.

3. Sampling Frequency

The directive for the evaluation of the water quality time series adopted in Slovakia (Chapter 7) suggests that a minimum of 24 values should be used for classification of water quality in the given observation period (usually from 1 to 5 years). However, it is well known that the accuracy of estimation depends on the number of samples. US Corps of Engineers guidelines for the flood protection risk estimations take this into account (Chow, 1988). If we adopt the 90% percentile value as a guideline, it can be shown using the statistical tables (Handbook of tables, 1982) that 35 samples are needed to guarantee the 90% percentile to fall into the 95% confidence interval. With 24 samples we may reliably obtain only 85% percentile, and with 12 samples only 75%. With the current sampling frequency 3 years of sampling is needed to border the 90% percentile referenced in the standard, and there is no way whatsoever to classify biennial or annual intervals. Probability based standards should be build using statistical criteria, otherwise their application may lead to inconsistent results. In other words, the admissible limits should depend on the length of the series which was used for calculating the ambient water quality criteria.

In addition, the statistics of analyses are also dependent on the statistical properties of the individual water quality component. As previous work in this field shows, one of the key parameters which governs the accuracy of the estimation of the mean from infrequently sampled time series is the ratio of the mean and standard deviation (Somlyódy et al, 1986). Establishing a trend in water quality should also take this ratio into account.

Based on this methodology, an analysis was performed to see how many samples are needed to estimate mean concentration of BOD-5 and ammonia in the Nitra River. Since the empirical distribution of the parameters is different at each location, the analysis was made location-specific. The desired reliability of the estimation was set to the commonly used 95% level. The admissible error of the estimation was expressed as a multiple of the mean itself. This ratio of admissible error and the mean serves as a measure of accuracy of estimate and is denoted as alpha (following the notation convention in the original work).

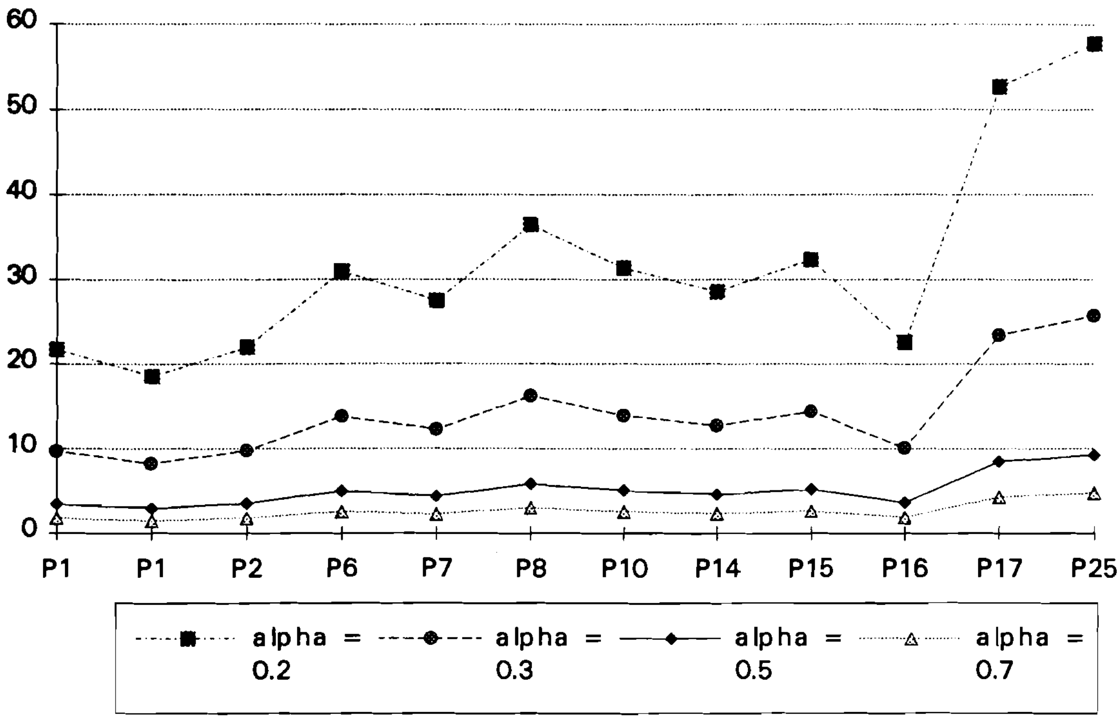


Figure 1 Number of samples necessary to estimate BOD-5 mean value

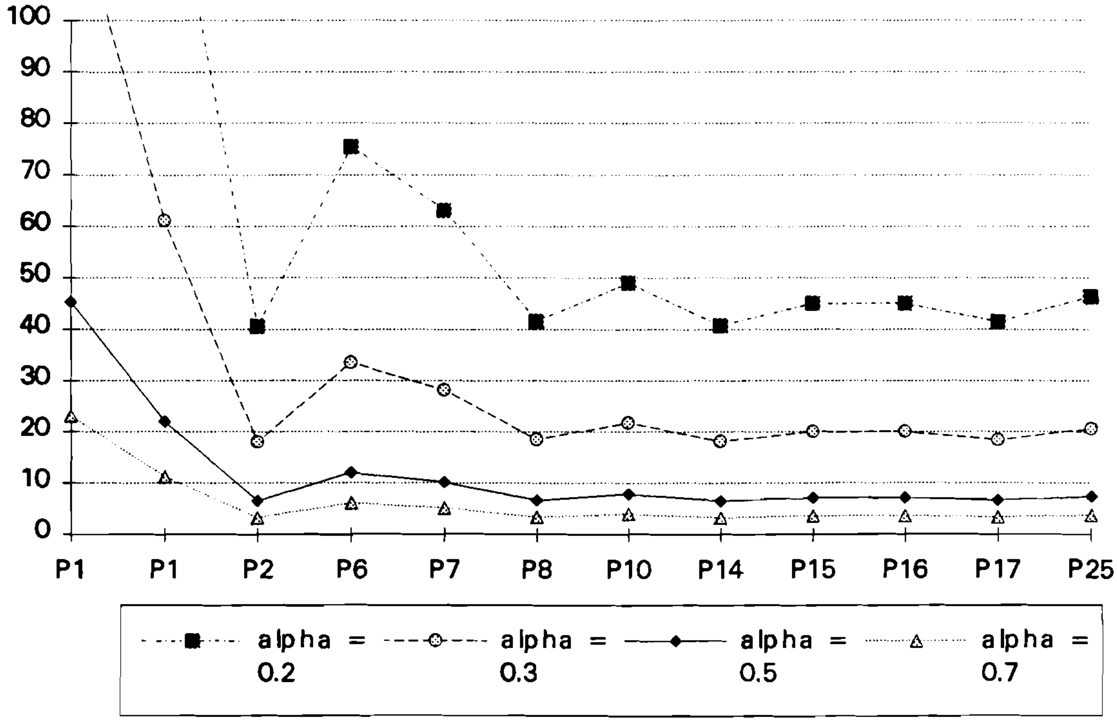


Figure 2 Number of samples necessary to estimate the NH_3-N mean value

It can be seen that the desired accuracy influences the number of samples very strongly (see Figures 1 and 2). It is relatively easy to decrease alpha from 0.7 to 0.5, but to achieve alpha 0.2 requires an order of magnitude more samples. The specifics of the particular water quality pollutant also plays a role. The estimation of BOD-5 concentration with a given accuracy generally requires more samples to be taken in the lower part of the river, but for the ammonia nitrogen quite the opposite is true.

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APPENDIX 11.1 SUMMARY OF MUNICIPAL WASTEWATER TREATMENT ALTERNATIVES

Table 11.1 Characteristics of the Feasible Treatment Alternatives for Handlova WWTP (700 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	150	8	30	0	None
1	0	0.35	18	6	21	0	B
2	1	0.46	15	1	4	17	BnC
3	1.3	0.55	12	1	0	12	BnCDNp
4	7.8	0.7	6	0.5	0	5	New BnCDN

Table 11.2 Characteristics of the Feasible Treatment Alternatives for Lehota WWTP (1000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	130	8	30	0	None
1	0	0.07	12	6	2	19	B
2	0.1	0.1	10	1.5	0	19	BnC
3	0.15	0.12	10	1.0	0	10	BnCDNp

Table 11.3 Characteristics of the Feasible Treatment Alternatives for Prievidza WWTP (25000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	170	10	40	0	None
1	0	1.0	17	7	28	0	B
2	0	1.2	17	7	4	24	B
3	3	1.5	15	7	0	16	BnDNp
4	5	1.6	10	1	0	10	BnCDNp
5	2	1.4	15	1	4	24	BnC

Table 11.4 Characteristics of the Feasible Treatment Alternatives for Partizanske WWTP (15000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	100	8	20	0	None
1	0	0.7	10	6	0	14	Bn
2	1.5	0.8	10	6	0	8	BnDNp
3	2.0	0.95	10	1	0	8	BnCDNp

Table 11.5 Characteristics of the Feasible Treatment Alternatives for Banovce WWTP (8000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	360	13	50	0	None
1	0	0.4	26	9	35	0	B
2	0.5	0.52	24	1.5	35	0	BC
3	1.0	0.58	20	1.5	5	30	BnC
4	2.5	0.7	18	1.0	0	20	BnCDNp
5	9.0	0.8	16	1.0	0	8	New BnCDN

Table 11.6 Characteristics of the Feasible Treatment Alternatives for Topolcany WWTP (13000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	310	12	48	0	None
1	0	0.6	110	8	40	0	B
2	1.2	0.7	75	3	40	0	BC
3	9	0.6	25	7	34	0	New B
4	12	0.7	25	7	0	34	New Bn
5	8.8	0.82	25	1.5	34	0	New BC
6	9.0	0.9	20	1.5	0	31	New BnC
7	10.8	1.08	20	1.5	0	19	New BnCDNp
8	14.4	1.45	10	0.6	0	8	New BnCDN

Table 11.7 Characteristics of the Feasible Treatment Alternatives for Nitra WWTP (30000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	240	11	48	0	None
1	0	1.1	100	9	40	0	B
2	2	1.3	60	2.7	34	0	BC
3	9	1.2	24	7	30	4	B+New B (15000 m^3/d)
4	15	1.4	18	7	4	30	Bn+New Bn(22000 m^3/d)
5	11	1.6	20	1.5	7	27	BnC + New BnC(15000 m^3/d)
6	14	1.9	15	1.5	0	25	BnCDNp + New BnC (15000 m^3/d)
7	25	2.3	8	0.8	0	10	New BnCDN

Table 11.8 Characteristics of the Feasible Treatment Alternatives for Zl.Moravce WWTP (6000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	250	12	48	0	None
1	0	0.3	20	8	24	10	B
2	0.5	0.4	20	1.5	4	34	BnC
3	1.0	0.5	15	1.0	0	20	BnCDNp
4	7.0	0.6	10	0.6	0	8	New BnCDN

Table 11.9 Characteristics of the Feasible Treatment Alternatives for Vrable WWTP (3000 m^3/d)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	230	3	30	0	None
1	0	0.17	20	2	2	19	B
2	0.1	0.22	15	1	2	19	BnC
3	0.2	0.25	12	0.5	0	10	BnCDNp

Table 11.10 Characteristics of the Feasible Treatment Alternatives for Surany WWTP ($3000\ m^3/d$)

Alternative	Cost (10^6 USD)		Effluent concentrations (mg/l)				Type of treatment
	IC	OMRC	BOD	TP	NH4-N	NO3-N	
0	0	0	200	10	45	0	None
1	0	0.17	20	7	24	8	B
2	0.5	0.22	20	1	4	28	BnC
3	3.9	0.34	10	0.6	0	8	New BnCDN