WATER QUALITY MANAGEMENT
OF THE VESZPRÉMI-SÉD RIVER
AND MALOM AND NÁDOR CHANNEL SYSTEM
IN HUNGARY

BY

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A Thesis
Submitted to the Faculty of Graduate Studies
in Partial Fulfillment of the Requirements
for the Degree of

MASTER OF SCIENCE

Department of Civil and Geological Engineering
The University of Manitoba
Winnipeg, Manitoba

ISBN 0-612-16129-3

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A Thesis/Practicum submitted to the Faculty of Graduate Studies of The University

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Abstract

The water quality management of the Veszprémi-Séd River and Malom and Nádor Channel System in Hungary is a formidable task since water quality is poor due to past environmentally incompatible practices and because financial resources are limited in the post-communist transitional economy. The objective of this work is to identify water quality management strategies for the Veszprémi-Séd River and Malom and Nádor Channel System. The principal water quality concerns in the system are the high levels of ammonia and biochemical oxygen demand during spring and the low concentration of dissolved oxygen in the summer. Wastewater treatment alternatives and costs are estimated for the dischargers in the system and the decision support tool DESERT is used to simulate water quality and analyze water quality management policies. Three command and control policies, the uniform effluent limit (UEL), least cost (LC), and individual effluent limit (IEL) policies, are evaluated with respect to cost efficiency, equity, certainty of system outcome, and administrative ease. The UEL policy is equitable, insensitive to uncertainty, and administratively simple, but it is economically inefficient. The LC policy is cost effective, but its administrative burden is high and it is less equitable and more sensitive than the UEL policy. The IEL policy has low information requirements but it also has the worst characteristics of the UEL and LC policies since it is neither cost effective nor equitable. The results of this analysis accentuates the need for reliable water quality and flow data and the distinction between nationally established and regionally achievable water quality criteria.

Table of Contents

Acknowledgement

The author wishes to express her deep appreciation to her advisor, Dr. Barbara J. Lence, for her guidance and support throughout the course of this work. The suggestions and help that the author has received from Mr. János Fehér of the Water Resources Research Center in Hungary (VITUKI) and Mr. Ilya Masliev, Dr. László Somlyódy, and Dr. Mark G. Smith of the Water Resources Project at the International Institute for Applied Systems Analysis (IIASA) are gratefully acknowledged. The author also wishes to express her gratitude to the members of her thesis examination committee, Dr. Donald H. Burn and Dr. John Gray, for reviewing this work and offering helpful suggestions. The financial support provided by the Natural Sciences and Engineering Research Council of Canada (NSERC) Postgraduate Scholarship, IIASA, and the NSERC under award number OGPO41643 are gratefully acknowledged. The author would like to thank all her colleagues and friends in the Water Resources group at the University of Manitoba for their suggestions and moral support. Lastly, the author wishes to thank her family and friends for their understanding, encouragement, and steadfast support throughout her university studies.

List of Figures

List of Tables

Chapter 1

Introduction

Water quality management in regions where water quality is significantly degraded and financial resources are limited is a challenging problem. The countries in Central and Eastern Europe (CEE), which are currently undergoing intensive political and economic transition, face such a formidable task. Past economic activities taken place under central planning left a legacy of environmental problems in many regions in CEE, as revealed after the collapse of communism in CEE in the late 1980s. Now, water quality improvement is to take place in a state of limited financial resources while competing with the many national priorities of countries in transition.

The objective of this work is to identify water quality management strategies for the Veszprémi-Séd River and Malom and Nádor Channel System in Hungary. The Veszprémi-Séd River and Malom and Nádor Channel System serves as an illustrative case study for Hungary, which is currently investigating alternative water quality management approaches and water quality objectives and criteria. This system is polluted by municipal and industrial emissions, yet it also serves as the water supply for aquaculture and agriculture in the lower reaches of the system. The principal water quality concerns in the system are the high levels of ammonia and biochemical oxygen demand (BOD) during spring and the low concentration of dissolved oxygen (DO) in the summer. The Hungarian government seeks to improve the water quality in this region but is concerned about adding to the financial burdens that municipalities and industries already face in the transitional economy.

Mathematical modelling and mathematical programming are applied to aid in the identification and analysis of water quality management polices for the Veszprémi-Séd River and Malom and Nádor Channel System. Wastewater treatment alternatives and costs are generated for the dischargers in the system. Three policies, the uniform effluent limit (UEL), least cost (LC), and individual effluent limit (IEL) policies, are compared on the bases of cost efficiency, equity, certainty of system outcome, and administrative ease.

Chapter 2 provides a review of the literature on water quality management. The topics reviewed are the policies and tools for water quality management and the issues of water quality management specific to CEE. Chapter 3 discusses the characteristics and the water quality problems of the Veszprémi-Séd River and Malom and Nádor Channel System. Chapter 4 presents the approach for the analysis and describes the software tools that are used. In Chapter 5, the water quality modelling of the Veszprémi-Séd River and Malom and Nádor Channel System is presented. This chapter presents the water quality simulation model and the results of the calibration and sensitivity analysis for this model. In Chapter 6, water quality management strategies are analyzed. Only command and control policies are investigated in this work. Chapter 6 discusses the wastewater treatment alternatives and costs that are required under the different water quality management policies and evaluates the efficacy of these policies. Chapter 7 concludes this work with a summary of the results and recommendations for further work for the water quality management of the Veszprémi-Séd River and Malom and Nádor Channel System.

Chapter 2

Literature Review and Background

Managers of water quality typically face two main questions: What are the desired water quality goals in the system?, and How should these goals be achieved?. Water quality management programs are designed to achieve water quality standards while taking into consideration the social and economic costs of pollution control. Since the 1960s mathematical modelling has been used to aid researchers and decision makers in studying and managing water quality (Loucks and Lynn, 1966; Johnson, 1967; ReVelle *et al.*, 1967; Smith and Morris, 1969). Orlob (1992) presents a summary of the use of mathematical modelling in water quality management. Examples of the application of systems analysis to water quality management abound in North America and Western Europe, but in CEE, where until recent years countries have had centrally planned economies, the application of systems analysis to pollution control has been rare (Somlyódy, 1991).

This chapter reviews the literature on water quality management of river systems. Section 2.1 reviews the literature on the polices and tools used in water quality management. Section 2.2 focuses on the water quality management issues in CEE with an emphasis on Hungary.

2.1 Water Quality Management

The basic problem in surface water quality management involves (1) the determination of the levels of point or nonpoint discharge that can be allowed while maintaining a desired level of water quality at some receptor locations in the water body and (2) the selection of a regulatory mechanism to effect the necessary amount of pollution reduction. Typically a water quality simulation model is used to relate the emission of pollutants to the distribution of the impacts of the pollutants in a river system. Using a water quality simulation model, transport coefficients, or functions, that represent the effect of pollution inputs from the dischargers on the water quality at the downstream receptor locations can be developed. Then the regulatory body can determine the degree of waste removal for the dischargers in the system, called waste load allocation (WLA), that is necessary to maintain the desired level of water quality at the receptor locations. WLAs are often determined using water quality management optimization models that consider water quality maintenance and other objectives such as cost minimization and equity.

2.1.1 Policies for Water Quality Management

Water quality management policies are designed to achieve water quality standards while taking into consideration the social and economic costs of pollution control. The river basin approach, which considers river basins rather than political boundaries as the natural unit for water quality management, is gaining increasing acceptance (Scheierling, 1996). Policies for water quality management can be broadly classified into two categories: command and control approaches and incentive based approaches.

2.1.1.1 Command and Control Policies

Command and control approaches are the conventional practice in water quality management where the regulatory body specifies the degree of wastewater treatment, or effluent water quality, that is required of the dischargers in order to meet water quality goals. One conventional policy is the uniform treatment, or UT, program. Under a UT program, all dischargers in the regulated area are required to perform the same percentage of waste removal determined by the regulatory body to meet water quality goals. The extreme case of the UT program is the policy of "best available technology" (BAT), where dischargers are required to use the technology that can effect the greatest degree of waste removal. A policy that is often studied in the literature but rarely used in practice is the least cost, or LC, program. Under a LC program, the regulatory body determines the WLA such that the sum of the wastewater treatment costs of all dischargers in a river basin is minimized. Cost minimization is generally achieved when large dischargers who enjoy economies of scale in wastewater treatment perform high levels of waste removal while the dischargers who are less efficient in wastewater treatment perform low levels of waste removal. The locational effects of dischargers in a river basin are also considered in identifying the solution to a LC policy since the relative contributions to pollution at a water quality receptor location are different for different dischargers depending on their locations on the river.

The efficacy of the UT and LC programs have been investigated by many researchers. Johnson (1967) compares the UT and LC programs for the management of BOD and DO in the Delaware Estuary and shows that the LC program is more cost efficient than the UT program for a range of ambient DO standards. However, he holds that neither the LC nor the UT program is equitable since the LC program does not "treat equals in a like manner" while the UT program does not treat different dischargers differently. In addition to the UT and LC programs, Smith and Morris (1969) investigate a zoned uniform treatment (ZUT) program that is a combination of the UT and LC programs. In a ZUT program, dischargers in a river basin are grouped into zones that may be based, for example, on the types of industries or geographical considerations. Then WLAs are performed such that dischargers within zones face uniform treatment levels while the required treatment levels vary across zones, resulting in a set of zoned uniform treatment levels that minimize the sum of wastewater treatment costs in the river basin. The authors show that the ZUT program results in costs that fall between those of the UT and LC programs and represents a compromise between the UT and LC programs in terms of cost efficiency and equity.

Rossman (1989) investigates seasonal wastewater discharge programs that employ different effluent standards during different times of the year to take advantage of the seasonal variation in a receiving water's assimilative capacity. Using pollutant discharge maximization as a surrogate for cost minimization, he determines seasonal discharge limits that minimize waste removal efforts while maintaining an acceptable annual risk of water quality violation for the single polluter case in the Quinnipac River, Connecticut. Lence *et al.* (1990) extend Rossman's formulation to the multi-discharger case and compare the costs of seasonal and nonseasonal programs. They show that the WLAs based on discharge maximization do not necessarily result in costs that are lower than those of the UT program, since only the locational effects of dischargers, but not cost efficiency, are considered in the process of the discharge maximization WLA.

Management of water quality through command and control approaches is often criticized since such approaches place emphasis on the use of end-of-pipe treatment but do not provide incentives for polluters to reduce waste output through innovation, such as alterations in production input or processes (see, for example, Howe, 1993). Incentive based policies, a different class of policies that uses economic incentives to induce pollution control, is the focus in the next section.

2.1.1.2 Incentive Based Policies

Incentive based policies use economic incentives to induce polluters to remove wastes so that both improved water quality and economic efficiency are achieved. Two incentive based policies that have received wide attention are the effluent charge (ECH) and transferable discharge permit (TDP) programs (Kneese, 1964; Dales, 1968; Eheart, 1994).

Under an ECH program, dischargers pay a set charge, or tax, for each unit of pollution that is emitted. Therefore there is an incentive for dischargers to reduce waste output since the less they discharge, the less pollution tax they pay. Theoretically each discharger, as a rational profit maximizing economic unit, is induced to reduce waste discharge to the point where its marginal pollution abatement costs equal the pollution charge rate (Kneese, 1964). In order for water quality goals to be met, the regulatory body must set the charge rate high enough to induce the amount of waste removal that is required for the river system yet not so high that the program becomes economically inefficient.

Johnson (1967) compares the costs of the UT, LC, and ECH programs for the management of BOD and DO in the Delaware Estuary and shows that the ECH program is capable of achieving water quality goals at a cost approaching that of the LC program. Only the costs of wastewater treatment are included in this economic comparison since the effluent charges are considered to be transfer payments only, not economic costs, although they are financial costs of the dischargers. He presents a discussion of the equity characteristics of the three programs and argues that the ECH program is more equitable provided the charge is determined based on the marginal damage to the environment. However, the author recognizes that the marginal damage to the environment cannot easily be quantified and he uses the marginal costs of wastewater treatment as the basis for the determination of the ECH.

Contrary to Johnson's view, Weitzman (1974) argues that emission standards are generally more suitable for managing water quality than effluent charges. Beginning with the observation that large multi-divisional firms and other governmental organizations almost never operate by setting transfer prices on commodities within the firms for profit maximization, he goes on to present a theoretical model that determines the comparative advantage of emission standards over prices. His model shows that for cases where the benefit function is sharply curved, such as the case when one extra unit of water quality improvement can result in water suitable for recreational activities, or where the cost function is piecewise linear, such as the cost curve of a wastewater treatment plant with discrete treatment options, a program of emission standards functions better than a ECH program.

Miltz *et al.* (1988) compare the costs of the UT and the uniform ECH programs for the management of nonpoint source pollution in the Highland Silver Lake watershed in Illinois. Their results show that the UT program is more cost effective than the uniform ECH program (1) when there is a high correlation between discharge abatement costs and pollution transport coefficients and (2) when there is a high variability in transport coefficients relative to the variability in pollution abatement costs.

Other disadvantages of the ECH programs compared to management through command and control approaches include (1) the increased administrative complication (Johnson, 1967), (2) the increased uncertainty in meeting water quality goals since actual emissions are only indirectly controlled (Weitzman, 1974), and (3) the large transfer payments that render the ECH program more costly from the dischargers' point of view (Brill *et al.*, 1979).

A different incentive based policy that can address the problem of the large transfer payment is the TDP program. The TDP program uses market incentives to achieve cost efficiency in the maintenance of water quality (Dales, 1968). The regulatory body defines a set number of discharge permits, each of which entitles the holder to a certain amount of effluent discharge. The sum of these permits represents the maximum amount of pollutant that the system can tolerate while meeting water quality goals. Once the permits are issued, they can be traded among dischargers as a market commodity. For each profit maximizing discharger, the optimal strategy is to remove pollutants up to the point where its marginal cost of waste removal equals the market price of the discharge permit. Dischargers with high marginal treatment costs buy discharge permits and remove little or no pollutants while efficient dischargers with low marginal costs sell permits for revenue and operate at high removal levels. As for the ECH program, there is a continual incentive for dischargers to reduce waste output since a discharger can sell the permits for profit if it can reduce its own waste output. Trading takes place until a market equilibrium is reached. At this point, cost minimization is achieved as all dischargers operate at the same marginal total cost of pollution abatement (treatment cost plus permit cost).

Eheart (1980) develops TDP programs for the control of BOD. Using the Willamette River in Oregon as an example case, he shows that TDP programs result in waste removal costs lower than that of the UT program. Brill *et al.* (1984) evaluates the locational effects of dischargers on water quality under TDP programs for the management of BOD. They present three restrictions on TDP markets that can be used to prevent water quality violation due to locational effects: (1) limits on the total discharge in sections of the river basin; (2) zoned markets; and (3) revaluation factors applied to permit trades. Lence (1991) develops TDP programs to manage multiple pollutants as a single weighted sum of the various pollutants that have an additive or noninteractive impact on environmental quality. She presents the methodology for selecting weighting factors that achieve adequate environmental protection, costs efficiency, and certainty of system outcome.

While three TDP programs have been implemented in the United States since 1981, very few transactions have actually taken place (Eheart, 1994). This lack of activity on the part of the dischargers to take advantage of permit trading and achieve cost efficiency can be attributed to the thin markets for TDPs and the devaluation that the administrator applies when permits are traded (Eheart, 1994). The devaluation is applied to counter the uncertainty in achieving water quality goals due to locational effects since a TDP program controls only the total amount of discharge into the system but not the distribution of these discharges among the various polluters. Other drawbacks of TDP programs include the potential for collusion due to imperfect competition in markets with few active players (Rose, 1973) and the increased administrative burden of distributing rights and facilitating transactions.

Despite the efficiency and incentive characteristics of incentive based policies, these policies are not widely used for water quality management (Howe, 1993). One important factor is that many people consider that charging a fee for environmental disposal implies a right for continued use, which is morally wrong (Paulsen, 1993). Also, the concept of treating the environment like property is not popular among environmental groups (Howe, 1993).

2.1.2 Systems Analysis in Water Quality Management

Orlob (1992) states that mathematical modelling "has become an accepted part of the process of establishing and evaluating alternative scenarios for water quality management." The earliest tool in water quality management typically is simulation modelling for estimating pollutant transport (Loucks and Lynn, 1966; Warren and Bewtra, 1974). Given a set of background ambient conditions, descriptive simulation models can predict the water quality response to different scenarios of pollutant loading, enabling the analyst to investigate the degree of wastewater treatment required and the possible water quality goals of the river system. Loucks and Lynn (1966) discuss the method of selecting the degree of wastewater treatment as well as establishing stream quality standards using simulation models which predict the probability distribution of minimum DO concentrations. Warren and Bewtra (1974) develop a simulation model to study the effects of varying treatment plant capacity and sewer capacity on stream quality.

In addition to descriptive simulation models, management optimization models can be used to determine the "optimal" wastewater treatment efforts for specific water quality goals. The cost of wastewater treatment is often used as a measure of optimality while water quality standards are the constraints in the optimization model (see, for example, ReVelle *et al.*, 1967; Smith and Morris, 1969; Dysart, 1970). In addition to costs and water quality, considerations such as equity and uncertainty can be explicitly incorporated into an optimization model (see, for example, Brill *et al.*, 1976; Lohani and Thanh, 1978).

Linear programming (LP) is one of the most widely used optimization techniques in water quality management, mainly due to the availability of generic LP codes (Kularathna and Somlyódy, 1994). LP models require all the relationships in the objective function and constraints to be linear. Using an objective function of cost minimization and constraints on water quality in their LP models, ReVelle *et al.* (1967) determine LC WLAs for the management of BOD and DO. Johnson (1967) and Smith and Morris (1969) determine UT and LC WLAs for the management of BOD and DO using LP.

Another optimization technique that is used in water quality management is nonlinear programming (NLP). NLP offers a more general mathematical formulation than LP since NLP can accommodate nonlinearity in the objective function and constraints. However, NLP approaches in water quality management remain less popular than other techniques due to their complexity and large computational requirement (Kularathna and Somlyódy, 1994).

An optimization technique that is gaining prominence in water quality management is dynamic programming (DP). The DP technique solves a problem involving a sequence of decisions by dividing the problem into subproblems, or stages, each with a reduced number of decisions. At each stage the results of the decisions, or states, become the inputs to the next stage and the subproblems are solved recursively until the last stage is reached. In a typical DP model for water quality management, the decisions are the levels of waste removal, the stage corresponds to a river reach, and the states are the instream water quality conditions. Dysart (1970) demonstrates the use of DP for pollution control for the Chattahoochee River in Georgia. He solves for the optimal degree of BOD removal and heat reduction in effluents required to satisfy a range of ambient DO standards.

Increasingly complex problems of water quality management and advances in computer technology serve to stimulate greater use of mathematical models such as the ones described above to aid in decision making (Orlob, 1992). This focus of computer-aided support for decision makers has resulted in the development of integrated tools, called decision support systems (DSS), that are "friendly" to the decision makers who often are not experts in mathematical modelling. A DSS typically includes three basic elements: (1) an information manager that receives and stores data and other information in a data base, (2) a collection of analytical tools such as the simulation and optimization models discussed above, and (3) a user interface that allows the user to select and edit data, execute models, and display results.

The proliferation of mathematical models and the more recent DSSs has prompted concerns that these tools may be misused for finding the "optimal" solution rather than to aid in the planning process. Brill (1979) cautions that public-sector problems are social and political in nature and cannot be completely represented or "solved" by mathematical modelling. Water quality management is a public-sector problem involving the allocation of a scarce resource. The determination of water quality goals and regulatory mechanisms that are acceptable to all members of the society is beyond the capabilities of any mathematical tools. However, the use of mathematical tools can provide valuable insights to the decision makers for the effective management of water quality.

2.2 Transitional Issues for Water Quality Management in CEE

The ongoing political and economic transition following the collapse of communism in CEE presents both challenges and opportunities for water quality management. The effect of transition on water quality, the economic and financial challenges faced by CEE countries in transition, and the implication of transition on environmental policies are the subjects of this section.

2.2.1 Water Quality Trends

As information and evidence on the quality of the environment in CEE countries appeared following the collapse of the communist regime, the degraded state of the environment became apparent. Under central planning, the state was responsible for all functions, including both economic development and environmental control, without the necessary checks and balances (Scheierling, 1996). The guaranteed market, the heavy subsidies in agriculture and industries, and the absence of incentives for the efficient use of inputs under a centrally planned economy all served to promote consumption and production at the expense of the environment. In Poland for example, from 1967 to 1986, river water classified as drinkable decreased from 33 to only four percent of the country's total river length while “non-classifiable” water, or water unsuitable even for industrial use, increased from 23 to 42% (Scheierling, 1996).

In the transitional years following the collapse of the communist regime, water quality in many regions in CEE showed improvement, mainly due to the decline in the agricultural and industrial sectors. In Hungary, the termination of subsidies for fertilizer reduced fertilizer use by over 50% while nutrient concentrations in surface water decreased by 20 to 40%. At the same time, restructuring and privatization resulted in the closure of industries, and industrial output in Hungary declined by more than 25% (Somlyódy and Paulsen, 1992).

In contrast to agricultural and industrial pollution, pollution caused by municipal point sources remained unchanged. As municipalities continued to raise the price of water to better reflect the realistic costs of water supply and wastewater treatment, water demand continued to drop. However, municipalities had also launched extensive efforts to connect households to sewer systems, offsetting the decrease of influent into wastewater treatment plants effected by the price increase of water (Somlyódy and Paulsen, 1992).

While water quality has improved since the beginning of the transition without water quality management actions, this trend is not likely to continue into the future. Although the process is slow, economic recovery is taking place in CEE. In Hungary, a growth in gross domestic product (GDP) occurred for the first time in 1994 since 1989 (Scheierling, 1996). As the economy recovers, pollution will likely begin to rise unless management actions are taken (Somlyódy and Paulsen, 1992; Scheierling, 1996).

2.2.2 Economic and Financial Challenges

The necessary reforms for transition was launched in Hungary after the elections of 1990 with popular support. Similar to other CEE countries, the focus had been on concentrating the limited national resources on improving economic efficiency (Donáth, 1995). Environmental protection had to compete with other important considerations such as private sector development, education and health, agricultural restructuring, and infrastructure improvement in transportation and telecommunication for attention and funding (PHARE Information Office, 1995).

In order to improve water quality, Hungary has eight different national programs for subsidizing investment in municipal wastewater treatment and sewage collection. However, there is little coordination among these programs and the priority for funding is not based on water quality considerations. As a result, clever municipal governments can obtain funding from more than one program while others must wait (Fehér *et al.*, 1995). Given the limited national resources, activities in water quality improvement are stalled.

In response to the urgent and considerable needs for financial support in CEE countries, individual countries and international donor organizations provide financial assistance to CEE countries toward a variety of purposes. In Hungary, the largest foreign source of support is the program Poland and Hungary: Assistance to Restructure the Economy (PHARE), which channels the European Community's support to CEE countries. From 1990 to 1994, PHARE funding for Hungary totalled over US$500 million, 11% of which was earmarked for environmental protection and nuclear safety (PHARE Information Office, 1995). While international support significantly contributes to environmental protection in CEE countries, assessments of the performance of international support show that the effectiveness of international support can be improved (Klaassen and Smith, 1995). The main problems are that (1) loans require state guarantees which are limited due to the high national debts in CEE countries, (2) international funds are generally restricted for specified project types which do not reflect the receiving country's priorities, (3) foreign aid is often tied to the use of supplies from the donating country which tends to increase the costs for the receiving country, (4) the processes of the application for and the disbursement of international funds may cost close to 20% of the funds, and (5) some receiving countries have inadequate experience and legislation for the distribution of funds to their projects.

2.2.3 Policy Making and Institutional Change

Hungary submitted a formal application for joining the European Union (EU) before the year 2000 (PHARE Information Office, 1995). In developing its National Environment and Nature Conservation Concept (NKTK), Hungary took into consideration the recommendations of the EU but at the same time recognized that the process of transition would require a long time (Donáth, 1995). The NKTK defines the short-, medium-, and long-term priorities corresponding to the years 2000, 2005, and 2020, respectively. The short-term priorities encompass issues that require immediate action, such as the clean-up of severely deteriorated regions. The medium-term priorities include emission control and energy and material conservation. The long-term priority is the establishment and use of sustainable development concepts.

In 1995, the Hungarian Parliament passed a new environmental legislation although the relevant regulations had yet to be developed. The new legislation embraces the BAT policy but at the same time favours the transition to the ECH policy from the current system of command and control policies with non-compliance fines (Fehér *et al.*, 1995). This contradiction reflects the pressure on CEE countries to conform to EU practice and the need to recognize the economic difficulty.

One possible compromise between achieving the environmental standards of western Europe and adhering to the financial constraints is the stepwise tightening of environmental standards in stages. For example, the Czech Republic and Slovakia will implement their effluent standards in two stages, with the tightening of standards occurring in 2005 (Somlyódy, 1993).

Many researchers agree that water quality management policies should focus on cost efficiency, at least in the short-term, due to the severe financial constraints currently faced by the CEE countries (see, for example, Somlyódy and Paulsen, 1992; Scheierling, 1996). As a result, the LC, ECH, and TDP policies, which are noted for economic efficiency, receive increasing attention in CEE. Some CEE countries had ECH programs in the past but they were not effective since it was far easier for state-owned firms to pay the charge and then be compensated from the state budget than to reduce waste output. While decentralization may cause firms to be more responsive to incentive based policies, the stigma of past failure may limit the use of ECH policies in CEE (Paulsen, 1993).

Regardless of the mechanisms that CEE countries choose for managing water quality, the institutions responsible for water quality management need to be strengthened during the transition (PHARE Information Office, 1995). Many countries set up new ministries with broad responsibilities in environmental management (Scheierling, 1996). In Hungary two ministries, the Ministry for Environment and Regional Policy and the Ministry of Transport, Telecommunication and Water Management, are responsible for water quality management. The responsibilities and jurisdictions of these two ministries are not clearly delineated, causing inconsistent management actions and difficulties in enforcement of water quality standards (Fehér *et al.*, 1995). Decentralization places increasing responsibilities on municipalities, but municipal authorities are inexperienced and in many cases have yet to develop the necessary tools for dealing with their new responsibilities (Scheierling, 1996). Finally, financial institutions also need to be strengthened to effectively handle the financial arrangements for environmental improvement (PHARE Information Office, 1995).

In 1992, the Water Resources Project at the International Institute for Applied Systems Analysis (IIASA) launched the Degraded River Basins in Central and Eastern Europe Study. One of the main goals of the study is to identify alternative water quality management strategies that are feasible in the face of the environmental, economic, financial, and institutional challenges that characterize the current situation in CEE. Four illustrative case studies are analyzed in the Degraded River Basins in Central and Eastern Europe Study. The water quality management problems investigated in these case studies include the cost effective upgrade of municipal wastewater treatment plants (MWWTPs) in the Nitra River in Slovakia, the control of nonpoint pollution in the Morava River in the Czech Republic, the potential use of incentive based policies for the Narew River in

Poland, and the control of industrial and municipal pollution in the Veszprémi-Séd River and Malom and Nádor Channel System in Hungary. The latter case study is the focus of this work.

Chapter 3

Characteristics of the Veszprémi-Séd River and Malom and Nádor Channel System

The Veszprémi-Séd River and Malom and Nádor Channel System is located southwest of Budapest, west of the Danube River, and east of Lake Balaton as shown in Figure 3.1. The Veszprémi-Séd River branches into the Malom and Nádor Channels downstream of the city of Berhida, although currently the entire flow from the Veszprémi-Séd River is diverted to the Malom Channel. The Malom Channel is an artificial channel constructed for irrigation purposes and it runs parallel to the Nádor Channel for 72 km before converging with the Nádor Channel. The Gaja Creek is a tributary to the Nádor Channel, about 43 km upstream of the confluence of the Malom and Nádor Channels. The length of the main stem of the river system, the Veszprémi-Séd River and the Nádor Channel, is 167 km and the mean daily discharge at the mouth of the Nádor Channel is 4.5 m^3/s. This system provides water for aquaculture and agriculture and assimilates wastewater from 14 polluters. There are nine industrial dischargers and five MWWTPs in the upper reaches of the system in the Veszprémi-Séd River, the Nádor Channel, and the Gaja Creek, and water withdrawals take place in the lower reaches of the Malom and Nádor Channels. In order to satisfy the water demand in the Malom Channel, water can be pumped from the Nádor Channel to the Malom Channel near the city of Szabadbattyán. Figure 3.2 is a schematic representation of the Veszprémi-Séd River and Malom and Nádor Channel System.

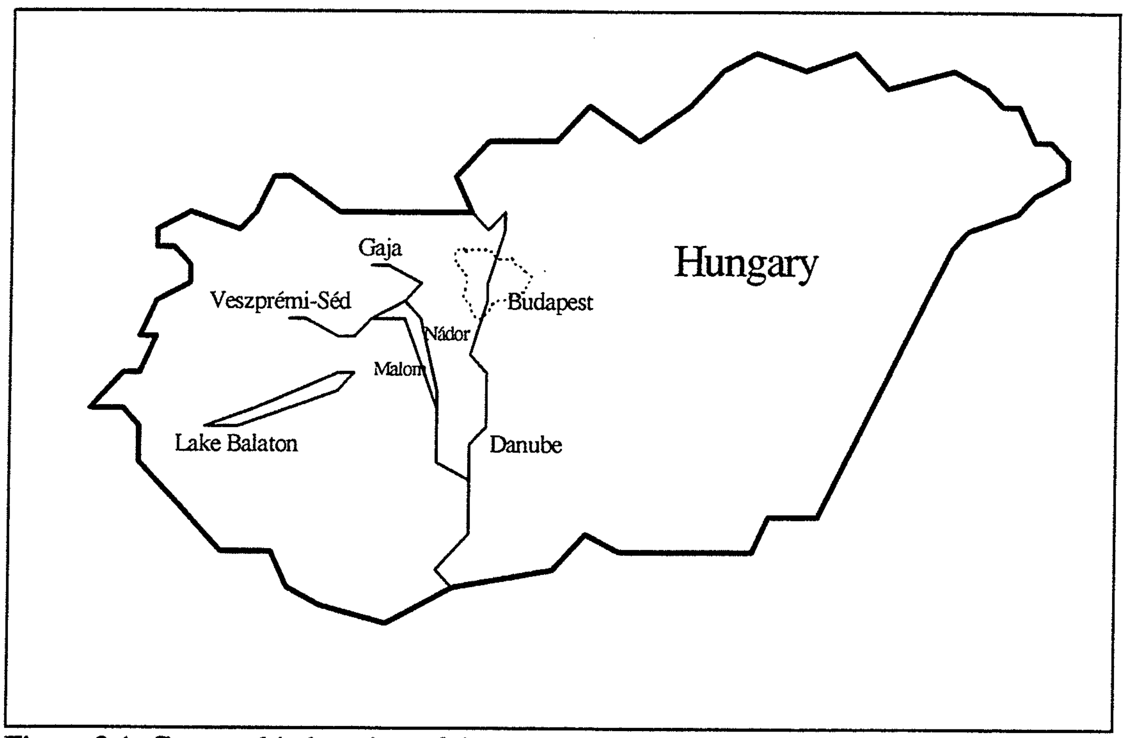

Figure 3.1 Geographic location of the Veszprémi-Séd River and Malom and Nádor Channel System

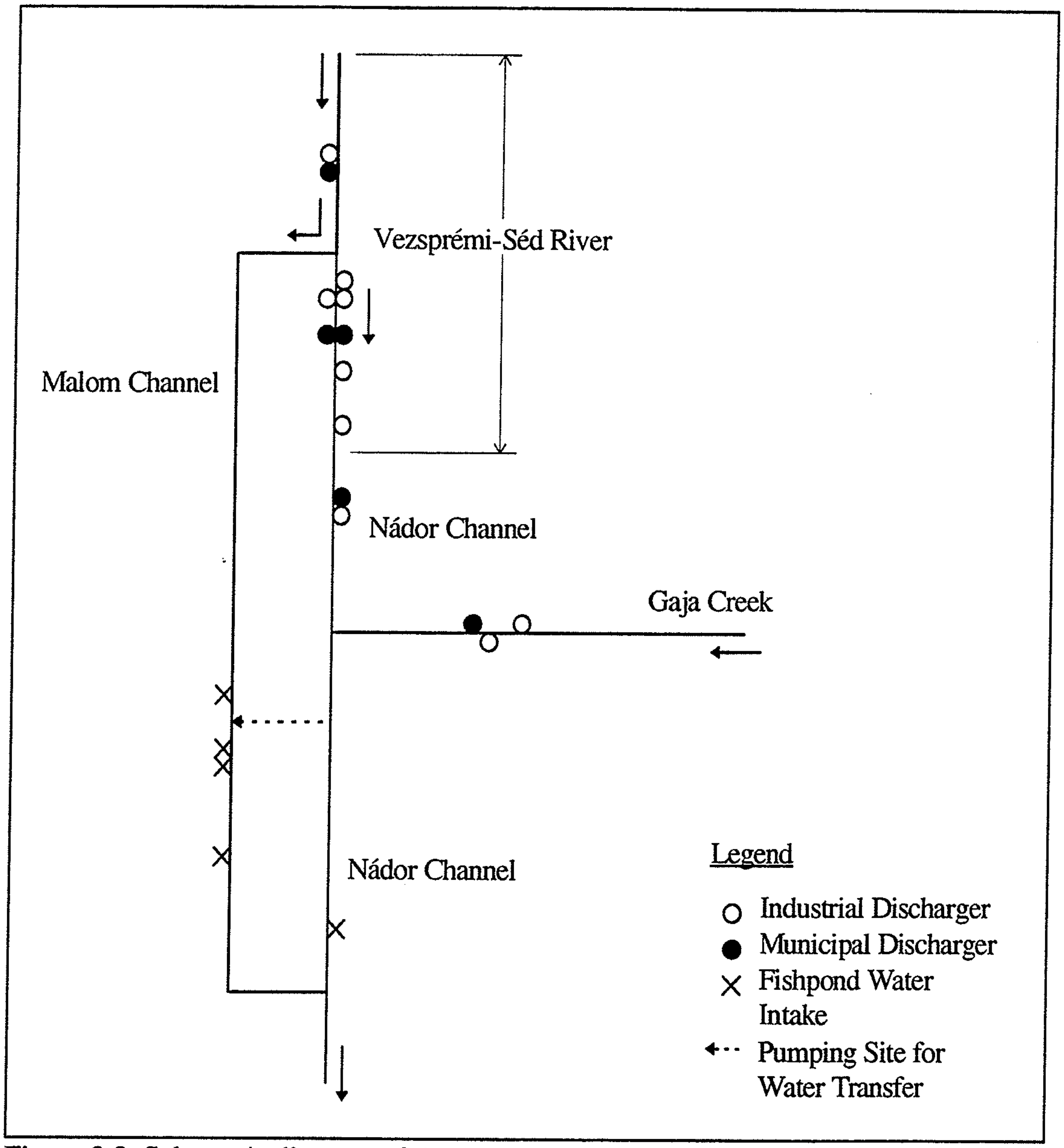

Figure 3.2 Schematic diagram of the Veszprémi-Séd River and Malom and Nádor Channel System

3.1 Water Quality

Due to the discharge of partially treated industrial and municipal wastewater, the water quality of the Veszprémi-Séd River and Malom and Nádor Channel System has been deteriorating since the 1950s (Fehér *et al.*, 1995). The Veszprémi-Séd River and Malom and Nádor Channel System is currently categorized for industrial water use under the Hungarian National Classification. Table 3.1 shows the average water quality of the effluent of the point sources in the Veszprémi-Séd River and Malom and Nádor Channel System and the corresponding national emission standards applicable to polluters discharging into receiving water classified for industrial use. All but two dischargers meet the current national emission standards for ammonia nitrogen, total suspended solids (TSS), and pH. However, the receiving water of emissions meeting the national emission standards for industrial water use may not be suitable for fishpond or irrigation use.

Table 3.1 Effluent water quality of point sources in the Veszprémi-Séd River and Malom and Nádor Channel System

Discharger	Type[1]	Receiving River	Distance from Mouth of River (km)	Discharge (m^3/s)	Ammonia Nitrogen (mg/l)	BOD_5 [2] (mg/l)	TSS [3] (mg/l)	pH
Bakony Works Emission I	I	Veszprémi-Séd	33.600	0.012	9.8	19	50	8.3
Veszprém MWWTP	M	Veszprémi-Séd	32.900	0.169	23.7	25	41	7.8
Nitrokémia Rt. Plants I-IV	I	Veszprémi-Séd	16.502	0.200	27.4	55	113	7.3
Nitrogén Müvek Rt.	I	Veszprémi-Séd	16.501	0.210	5.7	20 [4]	32	8.1
Nitrokémia Rt. Plant V	I	Veszprémi-Séd	16.500	0.037	4.4 [4]	18 [4]	38	8.3
Balatonfüred MWWTP	M	Veszprémi-Séd	13.801	0.033	23.3	17	24	7.8
Balatonfüzfö MWWTP	M	Veszprémi-Séd	13.800	0.056	12.8	20	26	7.8
Nitrokémia Rt. Plant VI	I	Veszprémi-Séd	12.300	0.010	4.4	18	24	8.4
Peremarton Chemical Works	I	Veszprémi-Séd	4.000	0.020	33.3	113	27	8.0
Várpalota MWWTP	M	Nádor	109.000	0.044	2.2	14	41	8.6
Bakony Works Emission II	I	Nádor	108.500	0.0004	14.9	22	23	7.9
SZIM Köszörügépgyár	I	Gaja	5.000	0.003	1.0	45 [4]	50	8.4
ALBA Textil Kft.	I	Gaja	4.002	0.022	0.7	73	87	9.4
Székesfehérvár MWWTP	M	Gaja	4.000	0.351	22.2	17	25	8.0
Hungarian National Emission Standard for Receiving Water Classified for Industrial Use					30	-	200	5 - 9

[1] "I" denotes industrial dischargers; "M" denotes municipal dischargers.
[2] 5-day biochemical oxygen demand
[3] Total suspended solids
[4] Estimated concentrations

Table 3.2 shows the average ambient water quality at the water quality monitoring location at Úrhida, 7 km upstream of the most upstream fishpond water intake in the Malom Channel, and the corresponding Hungarian water quality criteria for fishery use.

The Hungarian ambient water quality criteria for irrigation use is not shown since the criteria for irrigation are equal to or less stringent than the criteria for fishpond use. The data show ammonia to be the major concern in the Veszprémi-Séd River and Malom and Nádor Channel System as the annual average ambient ammonia concentration is close to 20 times the Hungarian criterion at the tolerable level. The Hungarian ambient ammonia criterion at the tolerable level is violated during the entire year, but the situation is particularly severe in early spring and late fall where instream concentrations of ammonia nitrogen over 20 mg/l, 40 times the tolerable limit, have been recorded. This seasonal variation in ambient ammonia level is attributed to the fact that macrophytes, primarily reed grass which utilize ammonia, grow in the Malom and Nádor Channels during the growing season but are absent during early spring and late fall. In April, 1993, fish kills occurred in fish ponds that received water from the Malom Channel, resulting in a loss of more than two tons of carp fish (Fehér *et al.*, 1995). Analysis of the routine water quality monitoring data during that period suggests that the high concentration of unionized ammonia resulting from the high instream concentration of ammonia, the high pH characteristic of the water in the region, and the lack of macrophytes which utilize instream ammonia during early spring could have caused the fish kills (Fehér *et al.*, 1995). Although the instream concentrations of the 5-day BOD (BOD_5) and DO meet the Hungarian ambient water quality criteria for fishery use on average as shown in Table 3.2, these criteria are not satisfied throughout the year. The ambient BOD_5 criteria are often exceeded during early spring while the ambient DO criteria are often violated in the summer when flow is low and temperature is high.

Table 3.2 Ambient water quality and Hungarian ambient water quality criteria for fishfarming

Water Quality Constituent	Ambient Concentration	Criteria Values	
		Tolerable Limit	Objective
Ammonia nitrogen (mg/l)	9.6	0.5	0.05
Nitrite as nitrogen (mg/l)	0.3	0.1	0.01
Phosphate as phosphorous (mg/l)	2.1	2.0	0.3
BOD_5 [1] (mg/l)	9.8	12	6
Dissolved Oxygen (mg/l)	6.0	4	6
COD [2] permanganate (mg/l)	12.2	15	8
TSS (mg/l)	60	1000	25
pH	8.1	6.0-9.0	6.5-8.5

[1] 5-day biochemical oxygen demand
[2] Chemical oxygen demand

The upper part of the Nádor Channel upstream of the confluence with the Gaja Creek acts as a sewer for the partially treated wastewater of nine dischargers since the entire flow from the Veszprémi-Séd River is diverted to the Malom Channel. The flow from the Gaja Creek dilutes the wastewater in the Nádor Channel, but the Gaja Creek itself receives partially treated wastewater from three dischargers, one of which is the MWWTP of Székesfehérvár, the largest city in the watershed of the Veszprémi-Séd River and Malom and Nádor Channel System. Therefore the water in the upper reaches of the Nádor Channel is unsuitable for fishpond and irrigation use.

3.2 Water Availability

The District Water Directorate allocates 27 million m^3 of water per year for agricultural use in the Veszprémi-Séd River and Malom and Nádor Channel System (Fehér *et al.*, 1995). The majority of this amount of water is supplied to the fishponds and the irrigation users from the Malom Channel. However, the flow in the Malom Channel is often less than the demand, particularly during the summer when there is high demand but low flow. In the past, the District Water Directorate could pump water from the Nádor Channel to the Malom Channel near the city of Szabadbattyán to satisfy the demand in the Malom Channel which occurs in the spring and summer. Due to the deteriorating water quality of the Nádor Channel, water transfer from the Nádor Channel is no longer a viable option (Fehér, 1995a).

Presently, the operation of a bauxite mine close to the Gaja Creek supplies 20 000 to 30 000 m^3/day of water to the Gaja Creek which flows into the Nádor Channel (Kling, 1995). However, the mining activities near the Gaja Creek are expected to be terminated in five to seven years and the clean water from the bauxite mines will no longer be available to the Veszprémi-Séd River and Malom and Nádor Channel System (Fehér *et al.*, 1995).

Chapter 4

Approach for the Analysis of the Veszprémi-Séd River and Malom and Nádor Channel System

4.1 Approach

The objective of this analysis is to identify water quality management strategies for achieving an acceptable level of water quality in the Veszprémi-Séd River and Malom and Nádor Channel System, given existing dischargers and withdrawals in region. Only command and control policies, which are considered by Hungarian water quality management authorities to be less complicated in practice than incentive based policies, are investigated (Fehér, 1995a). Incentive based policies are not examined because (1) they tend to have high administrative costs compared to command and control policies (Johnson, 1967), (2) the stigma of past failure may limit their use in CEE (Paulsen, 1993) and (3) the indirect control of pollution based on economic incentives may not be suitable for Hungary's unstable transitional economy where market economy is still in its infancy.

The specific tasks to be undertaken include (1) the determination of alternative WLAs for the Veszprémi-Séd River and Malom and Nádor Channel System for the management of ammonia, BOD, and DO and (2) the comparison of these WLAs on the bases of cost efficiency, equity, certainty of system outcome, and administrative ease.

WLAs are determined for three policies: (1) Uniform Effluent Limits, UEL; (2) Least Cost, LC; and (3) Individual Effluent Limits, IEL. Under the UEL policy, the effluent limits on water quality are set so that ambient water quality criteria are achieved while each discharger is required to produce the same level of effluent water quality. Under the LC policy, effluent limits are set so that the ambient water quality criteria are achieved at the minimum total wastewater treatment costs for the river basin. The IEL policy is a water quality management option in Hungarian legislation which allows the Regional Environmental Inspectorate to stipulate stricter effluent limits, regardless of cost efficiency, for polluters that have significant effects on the ambient water quality of a region, resulting in nonuniform effluent limits (Fehér *et al.*, 1995). The Hungarian IEL policy does not clearly define how waste loads should be allocated. In this analysis, a policy that maximizes the total pollutant discharge without cost considerations is used as the basis for determining the WLA under the IEL policy.

Ambient water quality criteria for four water quality constituents are used for the WLAs: (1) ammonia nitrogen, (2) unionized ammonia, (3) BOD_5, and (4) DO. Unionized ammonia criteria are used in addition to the ammonia nitrogen criteria because ammonia toxicity is caused by the unionized fraction of ammonia nitrogen (World Health Organization, 1986). Table 4.1 shows the Hungarian ambient criteria for fishfarming and the EU ambient criteria for fisheries. Although the EU criteria do not apply to water for intensive fishfarming, both Hungarian and EU criteria are used for comparison and because Hungary is a potential EU member. The Hungarian and EU criteria specify two levels of water quality. The lower levels are the tolerable level and the mandatory level for the Hungarian and EU criteria, respectively. These are the minimum ambient water quality recommended for the specific use. The higher levels are the objective and guide for the Hungarian and EU criteria, respectively. Hungarian criteria are more stringent than EU ones for ammonia but are less stringent for DO. Hungarian and EU criteria for unionized ammonia and BOD_5 are the same except for the absence of the mandatory level of the EU criterion for BOD_5. Besides the criteria shown in Table 4.1, the U.S. Environmental Protection Agency (USEPA) unionized ammonia criterion, which is a function of pH and temperature, is used since the proportion of unionized ammonia is affected by the pH and temperature (Heber, 1992; Emerson *et al.*, 1975).

Table 4.1 Ambient water quality criteria

Water Quality Constituent	Hungarian Criteria (mg/l)		EU Directive (mg/l)	
	Tolerable	Objective	Mandatory	Guide
Ammonia Nitrogen	0.5	0.05	0.8	0.2
Unionized Ammonia as Nitrogen	0.021	0.004	0.021	0.004
BOD_5	12	6	-	6
DO	4	6	7	8

Source: Fehér (1995b), Environment for Europe (1993)

4.2 Tools

The software tools used to develop and analyze water quality management strategies for the Veszprémi-Séd River and Malom and Nádor Channel System include DESERT, a DSS for river basin water quality management (Invanov *et al.*, 1995), and a spreadsheet code for generating wastewater treatment alternatives and costs (Masliev, 1995a). Both of these tools were developed in the Water Resources Project at IIASA.

4.2.1 DESERT

DESERT is a DSS for river basin water quality management. It includes a data management unit, an hydraulics simulation unit, a water quality simulation unit, and a water quality management optimization unit. Water quality simulation in DESERT is one-dimensional and includes the pollution transport mechanisms of advection and reaction. Longitudinal dispersion is not considered for water quality simulation. No reaction equation or parameters are pre-specified in DESERT. Theoretically, any pollutant can be modelled since the user enters the functions that describe the reactions of the pollutants. The pollutant transport equations can be solved analytically or numerically, and the model can be steady- or non-steady-state. DP is used in the optimization unit to minimize the total wastewater treatment costs for the river basin. The decisions are the discrete levels of waste removal, the stage corresponds to a river reach, and the states are the instream water quality conditions. Only ambient water quality constraints are allowed in the optimization model; other constraints such as uniform treatment constraints and budget constraints cannot be accommodated in DESERT.

The data required for simulation in DESERT include river cross-sections, locations of river objects such as dischargers and headwaters, flow, and water quality data such as temperature and concentrations of pollutants. Cost minimization using the optimization unit requires the effluent concentrations and costs of the wastewater treatment alternatives in addition to the required data for simulation.

4.2.2 Wastewater Treatment Alternatives and Costs Estimation

The alternatives that can be generated for the treatment of conventional pollutants such as BOD_5 and ammonia range from primary mechanical treatment to tertiary treatment for nutrient removal:

(1) primary treatment, P,

(2) chemically enhanced primary treatment, CEPT,

(3) primary precipitation, PC,

(4) P and secondary treatment with high load activated sludge, B1,

(5) P and secondary treatment with low load activated sludge, B,

(6) CEPT and secondary treatment with low load activated sludge, BC1,

(7) PC and secondary treatment with low load activated sludge, BC2,

(8) BC1 and partial denitrification, BC1DN,

(9) BC2 and partial denitrification, BC2DN, and

(10)BC2 and denitrification, BCDN.

The computer program developed to estimate wastewater treatment alternatives and costs is based on the design criteria and unit cost estimations given by Somlyódy *et al.* (1994). The design criteria for wastewater treatment alternatives, such as overflow rates and retention times, and unit cost estimations are given in Appendix A. The investment cost of an upgrade to a higher treatment level is calculated as the difference between the investment cost of that new treatment technology and the worth of the existing infrastructure and volumes, such as the building, concrete works, and simple mechanical equipment. The worth of the existing facility is calculated based on the equipment costs given by the software WAWTTAR, Water and Wastewater Treatment Technologies Appropriate for Reuse (Gearheart *et al.*, 1994). The costs of sludge handling is included in the estimated costs of wastewater treatment. The operation and maintenance costs (OMC) include labor, chemicals, and electricity usage associated with the operation and maintenance of wastewater treatment plants.

The required information includes the quantity and quality of the influent to the wastewater treatment plant and the number and dimensions of the existing wastewater treatment units. The output of this code includes the investment costs, the OMC, and the effluent concentrations of BOD_5, ammonia nitrogen, TSS, and total phosphorous.

Chapter 5

Water Quality Modelling of the Veszprémi-Séd River and Malom and Nádor Channel System

A water quality simulation model in DESERT is used to relate the emissions of pollutants to the ambient concentrations of the pollutants along the stream. The water quality constituents that are modelled are ammonia and BOD_5 in the March critical period and DO in the August critical period.

5.1 Water Quality Information

Depending on the given station, three to five years of daily flow data are available at nine flow stations in the Veszprémi-Séd River and Malom and Nádor Channel System. The location of the flow stations and the length of the records are listed in Table 5.1. Fourteen to 26 years of routine water quality monitoring data are available at seven water quality monitoring points depending on the given location as listed in Table 5.2. At each of the seven water quality monitoring points, at least one and up to four water quality measurements are recorded for each month by the Regional Environmental Inspectorate. The measured water quality indicators include pH, temperature, ammonia, BOD_5, chemical oxygen demand (COD), DO, total phosphorous, heavy metals, and fecal coliform. The Regional Environmental Inspectorate measures the effluent water quality of the dischargers three to five times per year. Effluent water quality data for 1991 and 1992 are available. The measured water quality indicators include pH, ammonia, BOD_5, and COD.

Table 5.1 Location of flow stations and length of records

Waterway	Location	Distance from Mouth of River (Rkm)	Years of Available Data
Veszprémi-Séd	Veszprémkülsö	32.700	1990-93
Veszprémi-Séd	Sóly	20.500	1990-93
Malom	Ösi	54.630	1989, 91-93
Malom	Úrhida	38.776	1989, 91-93
Malom	Kálóz	15.775	1989, 91-93
Nádor	Ösi	110.500	1990-93
Nádor	Sárszentmihály	95.000	1990-93
Nádor	Cece	46.700	1991-93
Gaja	Székesfehérvár	6.300	1990-93

Table 5.2 Location of water quality monitoring points and length of records

Waterway	Location	Distance from Mouth of River (Rkm)	Years of Available Data
Veszprémi-Séd	Sóly	20.700	1968-93
Malom	Ösi	54.630	1969-93
Malom	Úrhida	38.776	1969-93
Nádor	Úrhida	93.700	1968-93
Nádor	Cece	46.700	1969-93
Nádor	Sióagárd	1.400	1968-81
Gaja	Sárszentmihály	0.200	1968-81

In November, 1993, field data were collected by the Hungarian Water Resources Research Center (VITUKI) for flow, ambient water quality, effluent water quality, and sediment quality. Portable equipment was used in this exercise to obtain more detailed flow and water quality profiles in the river system than those provided by the regular gauging and monitoring points. Ambient water quality measurements were made at 17 locations along the Veszprémi-Séd River, the Malom and Nádor Channels, and the Gaja Creek and at four tributaries to the Nádor Channel and the Gaja Creek whose flow and water quality were not measured routinely by the Regional Environmental Inspectorate. Table 5.3 lists the locations where ambient water quality data were collected during this sampling program. The eight point dischargers whose effluent water quality were measured during the sampling program were Nitrokémia Rt. Plants I-IV, Nitrokémia Rt. Plant VI, Peremarton Chemical Works, ALBA Textil Kft., Bakony Works Emission II, Székesfehérvár MWWTP, Balatonfüzfö MWWTP, and Veszprém MWWTP.

Table 5.3 Sampling locations for the field sampling program

Waterway	Location	Distance from Mouth of River (Rkm)
Veszprémi-Séd	Veszprém, upstream of gauging station	42.000
Veszprémi-Séd	Veszprém, downstream of gauging station	31.000
Veszprémi-Séd	Sóly, downstream of gauging station	18.000
Veszprémi-Séd	Ösi	0.000
Malom	Ösi	70.395
Malom	Úrhida, near gauging station	55.825
Malom	Kálóz, at gauging station	32.806
Malom	Örspuszta	18.970
Péti, tributary of Nádor	Mouth of Péti	106.000
Hidegvölgyi, tributary of Nádor	Mouth of Hidegvölgyi	105.500
Nádor	Upstream of confluence with Gaja	97.000
Nádor	Úrhida	88.500
Nádor	Felsöszentiván	79.987
Nádor	Örspuszta	56.484
Nádor	Cece	46.000
Dinnyés-Kajtori, tributary of Nádor	Mouth of Dinnyés-Kajtori	25.500
Nádor	Sióagárd	0.500
Gaja	Hármashidi	6.500
Gaja	Upstream of Aszalvölgyi	4.100
Aszalvölgyi, tributary of Gaja	Mouth of Aszalvölgyi	4.000
Gaja	Upstream of confluence with Nádor	0.100

5.2 Water Quality Simulation

5.2.1 Design Conditions

Ambient stream conditions that represent a critical situation are chosen as the design conditions. The critical periods selected for design are the months of March and August. Ammonia concentrations in the Malom and Nádor Channels are high in early spring and late fall compared to the concentrations during the rest of the year due to the absence of macrophytes in early spring and late fall (Fehér *et al.*, 1995). Ambient BOD_5 concentrations often exceed the Hungarian ambient BOD_5 criteria in early spring as well. The critical period for DO, on the other hand, is the summer when stream temperature is high and stream flow is low. Since the critical periods for ammonia, BOD_5, and DO do not coincide, multiple critical periods are investigated. March, July, and August are the months when the fishponds in the Veszprémi-Séd River and Malom and Nádor Channel System require water for their operation (Kling, 1995). March is considered to be a month in spring while both July and August are summer months. The lowest daily flow in the history of record occurs in August. Ambient water quality records also show that both the minimum monthly flow and the maximum monthly temperature often occur in August. Thus, the critical periods selected for design are the months of March, for the management of ammonia and BOD, and August, for the management of DO.

The natural variability in stream flow is unlikely to be adequately represented in the limited flow data available, hindering the selection of flow values that represent critical design conditions. The mean monthly flows and the mean monthly seven-day average flows in March and in August are similar to one another, therefore the mean monthly flows are used in the analysis. Since the amount of unionized ammonia present is highly dependent on pH (World Heath Organization, 1986), the 90^{th} percentile pH value over all the measured values obtained in March at each water quality monitoring point is used in the analysis, representing a severe but not extreme condition. Temperature affects the rates of decay and reaeration, as well as the amount of unionized ammonia present in the stream. The mean monthly temperatures for March and August are used in the analysis.

5.2.2 Water Quality Model

A steady-state, one-dimensional model which includes the transport mechanisms of advection and reaction is used to simulate the distribution of ammonia, BOD_5, and DO in the Veszprémi-Séd River and Malom and Nádor Channel System. Inputs to the system are considered to be continuous; therefore the effect of longitudinal dispersion is assumed to be small and could be excluded (Thomann and Mueller, 1987). Discharges are assumed to be completely mixed across the river cross-sections. The instream removal of ammonia and BOD_5 are modelled as first order reactions (Thomann and Mueller, 1987). The one-dimensional transport equations representing the changes in the concentrations of ammonia and BOD_5, respectively, are:

$$\frac{\partial N}{\partial t} = -v\frac{\partial N}{\partial x} - K_n N \tag{5.1}$$

$$\frac{\partial L}{\partial t} = -v\frac{\partial L}{\partial x} - K_r L \tag{5.2}$$

where N = concentration of ammonia (mg/l);

L = concentration of BOD_5 (mg/l);

t = time (day);

x = distance (m);

v = velocity (m/day);

K_n = removal rate of ammonia (/day); and

K_r = removal rate of BOD (/day).

The removal rate of BOD, K_r, is the sum of the BOD decay rate, K_d, and the loss rate of BOD due to settling, K_s (Thomann and Mueller, 1987). The removal of BOD due to settling is minor relative to BOD decay; therefore K_r is assumed to be equal to K_d. Ammonia reacts with water to form unionized and ionized ammonia. The equilibrium constant for the partitioning of ammonia is given by:

$$pka = 0.09018 + \frac{2729.92}{T + 273.2} \tag{5.3}$$

where pka is the equilibrium constant and T is the water temperature in degrees Celsius (Emerson *et al.*, 1975). The fraction of unionized ammonia present is given by:

$$f = \frac{1}{10^{(pka - \mathrm{pH})} + 1} \tag{5.4}$$

where f is the fraction of unionized ammonia present and pH is the pH of the water (Emerson *et al.*, 1975).

The simulation of DO includes the processes of the oxidation of carbonaceous BOD (CBOD) and nitrogenous BOD (NBOD), and atmospheric reaeration. CBOD is exerted due to the oxidation of organic carbon while NBOD results from the oxidation of ammonia to nitrite and then to nitrate nitrogen in the nitrification process. The oxidation of one gram of ammonia into the final product of nitrate requires 4.57 g of oxygen, therefore the amount of NBOD can be estimated to be 4.57 times the amount of ammonia (Thomann and Mueller, 1987). The one-dimensional transport equation representing the change in the concentration of DO is:

$$\frac{\partial c}{\partial t} = -v\frac{\partial c}{\partial x} - K_d L_u - K_n(4.57 \cdot N) + K_a(c_s - c) \tag{5.5}$$

where c = concentration of DO (mg/l);

L_u = ultimate CBOD (mg/l);

K_a = atmospheric reaeration rate (/day);

c_s = saturation concentration of DO (mg/l);

and all other variables are as described above. The ultimate CBOD, L_u, is the total oxygen demand exerted for the oxidation of carbonaceous material. The ratio of L_u to the 5-day CBOD, L_5, can be expressed as:

$$\frac{L_u}{L_5} = \frac{1}{1 - \exp(-5K_d)} \tag{5.6}$$

The rate of atmospheric reaeration as a function of internal mixing and turbulence has been studied extensively (Thomann and Mueller, 1987). Three commonly used formulations are the empirical equations developed by Churchill *et al.* (1962) and Owens *et al.* (1964), and a theoretical equation developed by O'Connor and Dobbins (1958). These three formulations are shown in Table 5.4.

Table 5.4 Common equations for calculating reaeration coefficients

Investigators	Equation[1]	Applicable Velocity Range (fps)	Applicable Depth Range (ft)
O'Connor & Dobbins	$K_a = \frac{12.9U^{0.5}}{H^{1.5}}$	0.5 - 1.6	1 - 30
Churchill *et al.*	$K_a = \frac{11.6U}{H^{1.67}}$	1.8 - 5	1 - 11
Owens *et al.*	$K_a = \frac{21.6U^{0.67}}{H^{1.85}}$	0.1- 5	0.4 - 11

[1] K_a expressed in day^{-1}, U = velocity (fps), H = average depth (ft)

Source: Thomann and Mueller (1987)

The rate of atmospheric reaeration, as well as the rates of biological reactions, are dependent on temperature. The temperature dependence of the atmospheric reaeration rate, the BOD decay rate, and the ammonia removal rate is given approximately by:

$$K_T = K_{20} \cdot \theta^{T-20} \tag{5.7}$$

where K_T = rate (/day) at temperature T (°C);

K_{20} = rate (/day) at 20°C; and

θ = temperature coefficient.

The values of the temperature coefficient, θ, for reaeration, CBOD decay, and NBOD removal, are 1.024, 1.047, and 1.08, respectively (Thomann and Mueller, 1987).

The Veszprémi-Séd River and Malom and Nádor Channel System are divided into 21 reaches, each of which is considered to be uniform from the point of view of hydraulics. The reaches are separated at locations where hydraulic characteristics change considerably, such as at locations of discharge, tributary inflow, withdrawal, and water transfer. A reach may also be defined to end at a point in the stream where water quality criteria are to be met. The water quality receptor locations of concern are the five locations of water abstraction for fishponds and the location in the Nádor Channel where water can be pumped from the Nádor Channel to the Malom Channel for supplementing the water supply. In the Malom Channel, the water quality receptor locations are the fishpond water intakes at 111.140, 109.225, 108.975, and 100.657 river km upstream of Sióagárd. In the Nádor Channel, the most upstream water quality receptor location is the pumping site and the last receptor location is a fishpond water intake, located at 109.864 and 81.764 river km, respectively, upstream of Sióagárd.

5.2.3 Assumptions

In order to simulate the water quality for the Veszprémi-Séd River and Malom and Nádor Channel System, several assumptions are made:

(1) The entire flow of the Veszprémi-Séd River is diverted to the Malom Channel. This assumption is made because the majority of the water demand in the Veszprémi-Séd River and Malom and Nádor Channel System is supplied through the Malom Channel. This assumption is also consistent with the current practice of the District Water Directorate.

(2) Water is not transferred from the Nádor Channel to the Malom Channel at the pumping site in the simulation model. This assumption is consistent with the current practice of the District Water Directorate.

(3) Water balances conducted for the March and August critical periods indicate that there is more flow in the downstream end of the Veszprémi-Séd River and Malom and Nádor Channel System than in the upstream end after taking into consideration all estimated discharges and withdrawals. This difference is distributed evenly along the Veszprémi-Séd River and the Malom and Nádor Channels, resulting in lateral flows of 0.003 $m^3/s/km$ for March and 0.001 $m^3/s/km$ for August.

(4) The flow in the most upstream location of the sections of the Veszprémi-Séd River simulated is very small and no reliable flow data are available for this location. The most upstream gauging station in the Veszprémi-Séd River is located downstream of two dischargers, Bakony Works Emission I and Veszprém MWWTP. The flow upstream of these two dischargers is estimated to be the flow of the most upstream

gauging station minus the discharges from Bakony Works Emission I and Veszprém MWWTP: 0.003 m^3/s in March and 0.001 m^3/s in August.

(5) The mean values of the available effluent water quality data collected by the Regional Environmental Inspectorate for effluent flows, ammonia concentrations, and BOD_5 concentrations of the all the industrial point dischargers are used to represent their emissions. The mean values of the available effluent water quality data collected by the Regional Environmental Inspectorate are used also for three of the five municipal dischargers. The remaining two municipal dischargers, Balatonfüred MWWTP and Balatonfüzfö MWWTP, serve the communities around the tourist area of Lake Balaton and therefore have seasonal operations. During the tourist season in the summer the monthly effluent discharges from these two MWWTPs are nearly two times those in the rest of the year. Therefore for the Balatonfüred and Balatonfüzfö MWWTPs, the average monthly discharge for the summer is used for the August critical period while the average monthly discharge for the whole year is used for the March critical period. Since these two MWWTPs use additional clarification and activated sludge units to handle the increased flow in the summer, the effluent water quality is assumed to be the same for the two critical periods. The effluent flows and water quality of the Balatonfüred and Balatonfüzfö MWWTPs are shown in Table 5.5. The effluent flows and water quality of the other dischargers are the same as those shown in Table 3.1.

Table 5.5 Effluent characteristics of the Balatonfüred and Balatonfüzfö MWWTPs

Discharger	Effluent Characteristics	March	August
Balatonfüred MWWTP	Discharge (m^3/s)	0.027	0.037
	Ammonia (mg/l)	23.3	23.3
	BOD_5 (mg/l)	17	17
Balatonfüzfö MWWTP	Discharge (m^3/s)	0.043	0.084
	Ammonia (mg/l)	12.8	12.8
	BOD_5 (mg/l)	20	20

(6) The effluent BOD_5 concentration of the industrial discharger Nitrogén Müvek Rt. is not available. Since Nitrogén Müvek Rt. is a manufacturer of fertilizer products and it uses stabilization ponds to treat its wastewater, its effluent BOD_5 concentration is assumed to be 20 mg/l (Nemerow, 1978; Masliev, 1995b).

(7) There are four creeks discharging into the Nádor Channel and the Gaja Creek for which there are no routine flow and water quality data. The flow and water quality measurements obtained during the field sampling program conducted in November, 1993, are used in the water quality simulation for March and August. The ammonia concentration of the Aszalvölgyi Creek, which flows into the Gaja Creek, measured in the sampling program is 72 mg/l, more than two times the highest ammonia concentration among all the point sources in Veszprémi-Séd River and Malom and Nádor Channel System. This single measurement is assumed to be invalid and the highest ammonia concentration among the other three inflow creeks, 8 mg/l, is used as the ammonia concentration of the Aszalvölgyi Creek. If future water quality measurements in the Aszalvölgyi Creek show high concentrations of ammonia, one

should consider the possibility that there may be an unknown or unregulated source upstream that should be monitored and included as a polluter in the WLA process.

5.2.4 Calibration and Sensitivity Analyses

The calibration of the water quality simulation model for ammonia, BOD_5, and DO is performed using the ambient and effluent water quality data collected in the field sampling program conducted during November, 1993. For the six dischargers whose effluent water quality was not measured during the sampling program, the effluent water quality data measured in November, 1992, were used. The parameters that are calibrated are the rates of reaeration (K_a), ammonia removal (K_n), and BOD removal (K_r). One constraint on water quality simulation in DESERT is that the calculation of the values for each parameter in all the reaches must be based on only one equation. Therefore, only one formulation for the calculation of K_a can be used for the entire system even though different formulations are suitable for the velocities and depths in different reaches. For the sake of generality, another constraint is that the same values of K_n and K_r at 20°C and the same formulation for K_a are used in both the Malom and Nádor Channels.

As part of the calibration process, the sensitivity of the simulation to the parameters K_n, K_r and K_a are evaluated. The parameters K_n and K_r are varied within the common ranges of values found in the literature. The typical ranges of values for K_n and K_r at 20°C are 0.1 to 0.5/day and 0.5 to 3.0/day, respectively (Thomann and Mueller, 1987; U.S. Environmental Protection Agency, 1985). K_a is typically calculated using the formulae developed by O'Connor and Dobbins (1958), Churchill *et al.* (1962), and Owens *et al.* (1964).

Ammonia simulation is affected by changes in the parameter K_n. The ranges of the ammonia concentration profiles of the Nádor and Malom Channels are shown in Figures 5.1 and 5.2, respectively. The ranges of the profiles increase in the downstream direction. At the confluence of the Nádor and Malom Channels at 81 river km upstream of Sióagárd, the range of the simulated ammonia concentrations is 1.5 mg/l. BOD_5 simulation is affected by changes in K_r. The ranges of the BOD_5 concentration profiles of the Nádor and Malom Channels are shown in Figures 5.3 and 5.4, respectively. The effects of changes in K_a, K_n and K_r on DO simulation are shown in Figures 5.5 and 5.6 for the Nádor and Malom Channels. The parameters K_a, K_n, and K_r are varied one at a time while the other two are held constant. The parameter values in the base case to be held constant are 0.3 and 0.6 /day for K_n and K_r, respectively, and values of K_a are calculated using the O'Connor and Dobbins' formulation (1958). DO simulation of the Nádor Channel is more sensitive to parameter changes than that of the Malom Channel. The effect of changes in K_r on DO simulation of the Malom Channel is especially small, as shown in Figure 5.6(c).

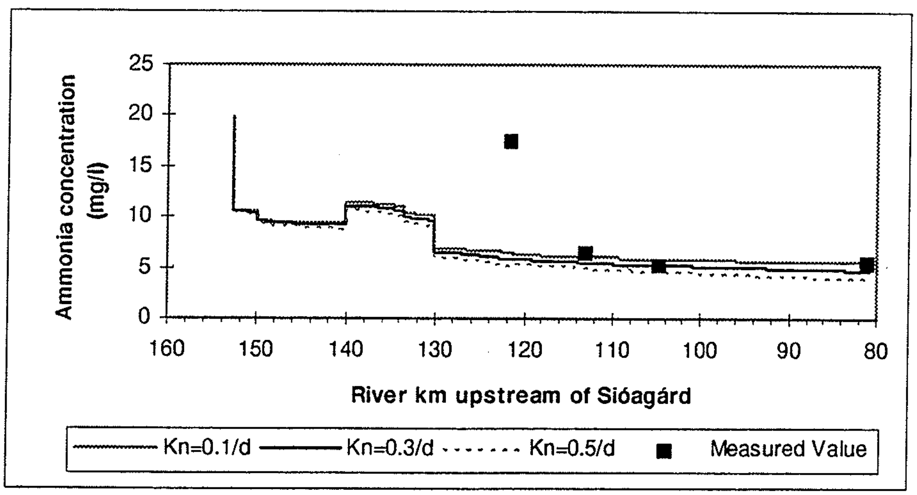

Figure 5.1 Ammonia concentration profiles of the Nádor Channel

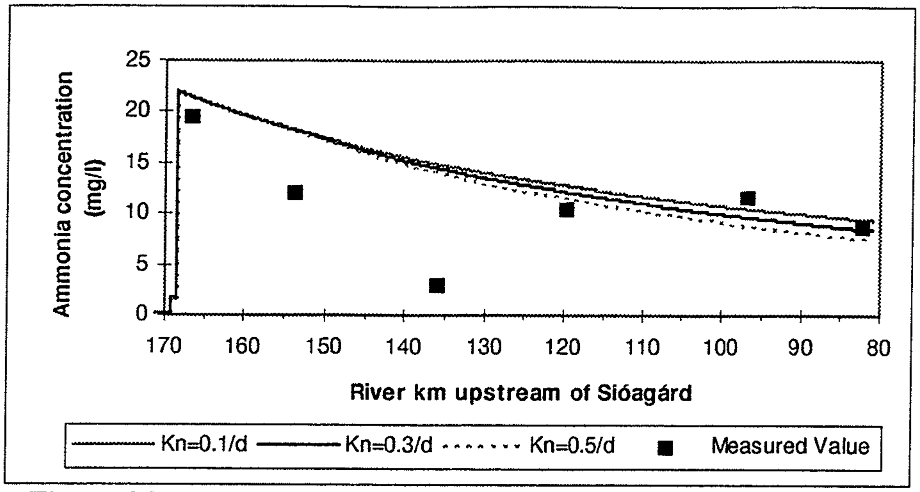

Figure 5.2 Ammonia concentration profiles of the Malom Channel

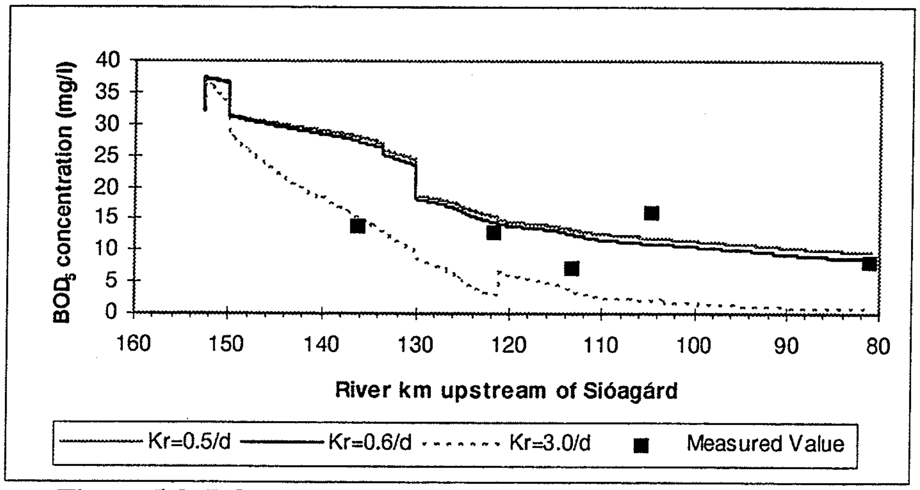

Figure 5.3 BOD_5 concentration profiles of the Nádor Channel

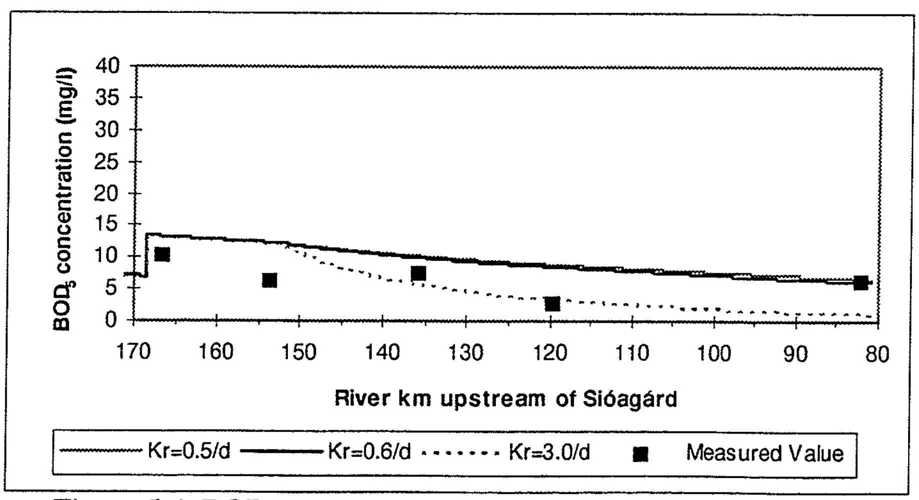

Figure 5.4 BOD_5 concentration profiles of the Malom Channel

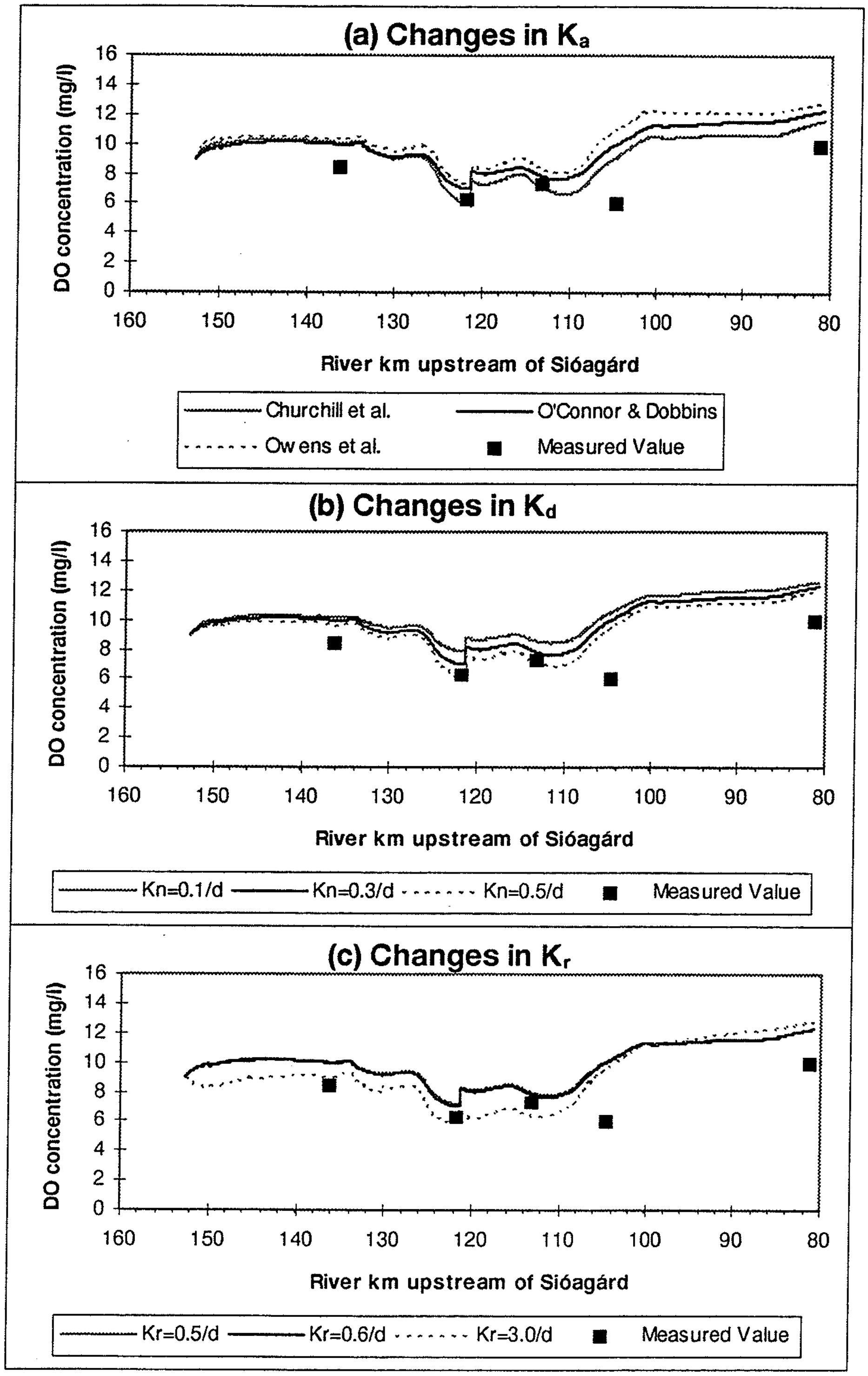

Figure 5.5 DO concentration profiles of the Nádor Channel

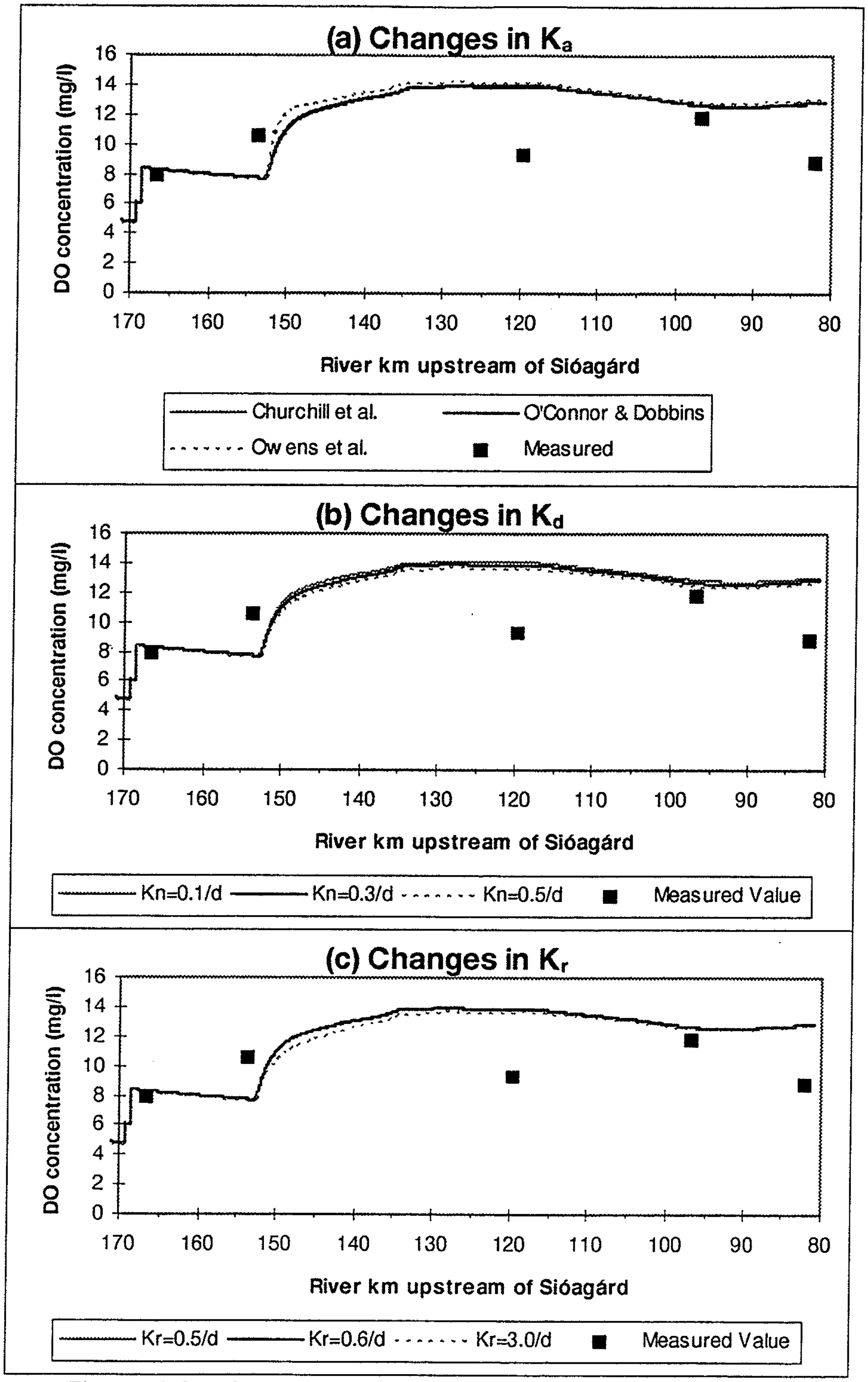

Figure 5.6 DO concentration profiles of the Malom Channel

The formulae for calculating K_a developed by O'Connor and Dobbins (1958), by Churchill *et al.* (1962), and by Owens *et al.* (1964) are first examined as alternatives for calculating the reaeration rates for simulating DO. K_n and K_r are then systematically varied until the simulated DO, ammonia, and BOD_5 concentration profiles match the measured values obtained in the field study. The calibrated model has a K_n of 0.3 /day and a K_r of 0.6 /day at 20°C, and values of K_a calculated using the formula developed by O'Connor and Dobbins (1958). The calibrated ammonia, BOD_5, and DO concentration profiles are shown by the dark curves in Figures 5.1 to 5.6. The calibration results are unsatisfactory as the discrepancies between the measured and simulated water quality profiles are numerous and large. The measured ammonia concentrations at 122 river km upstream of Sióagárd in the Nádor Channel and at 136 river km upstream of Sióagárd in the Malom Channel are different from the simulated values by nearly 12 mg/l. VITUKI, which compiled the data from the field sampling program, indicated that those measurements were single grab-samples and could be inaccurate (Masliev, 1995b). The lack of reliable water quality data significantly weakens the validity of the calibrated water quality model. A water quality management program determined based on an inadequate water quality model may result in violations of water quality standards or in overly conservative water quality management actions.

Besides the set of data obtained during the 1993 field sampling program, no other sets of data are available for model verification. Therefore, further sensitivity analyses are performed to investigate the sensitivity of the simulation model to changes in the design conditions. The design conditions analyzed are flow and temperature. The flow and temperature are varied by ±10%, ±33%, and ±67%. The results of the sensitivity analyses show that the water quality simulation for the Nádor Channel is more sensitive to changes in parameter values and design conditions than that for the Malom Channel. Therefore, only the graphs showing the fractional change in simulated water quality at the pumping site in the Nádor Channel are shown below.

Figure 5.7 shows the sensitivity of ammonia simulation to changes in K_n, flow, and temperature. The axes of the graph are normalized for comparison. The concentration of ammonia decreases as K_n, flow, and temperature increase, and vice versa. Increased flow increases dilution, and increased temperature increases the rate of nitrogen removal which results in reduced ammonia concentration. Ammonia concentration is most sensitive to changes in flow and least sensitive to changes in temperature. The graph also shows that a decrease in flow has more effect on ammonia concentration than an increase in flow.

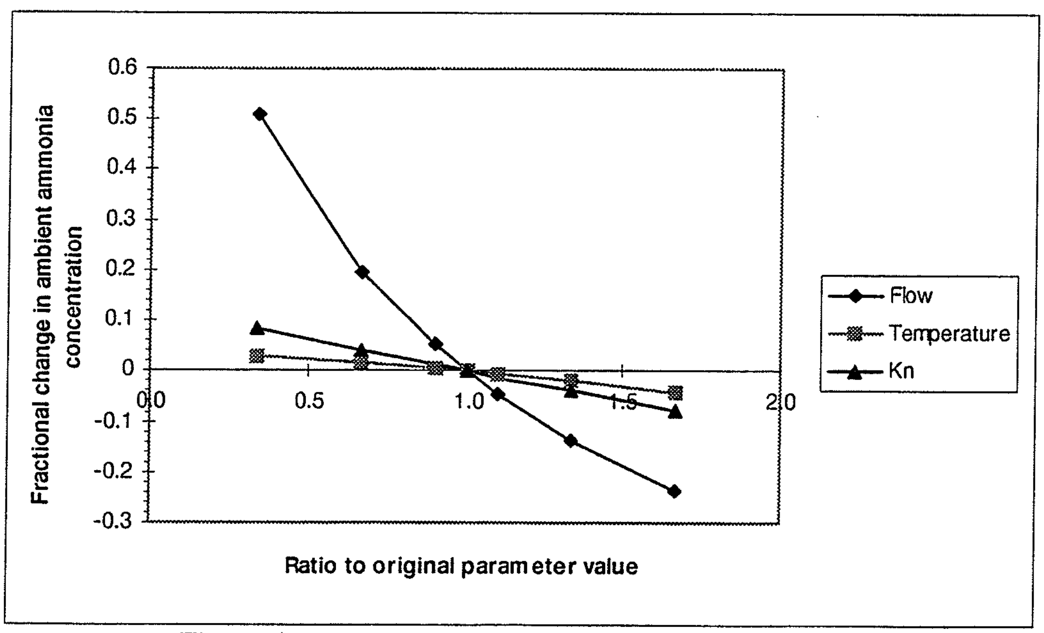

Figure 5.7 Sensitivity analysis of ammonia simulation

Figure 5.8 shows the sensitivity of unionized ammonia simulation to changes in K_n, flow, and temperature. The sensitivity of unionized ammonia simulation to changes in flow and K_n is the same as that of total ammonia. However, increased temperature causes the concentration of unionized ammonia to increase. The reason is that the effect of increased removal rate at increased temperature is offset by the increase in the fraction of unionized ammonia caused by increased temperature. Unionized ammonia concentration is most sensitive to changes in flow and least sensitive to changes in K_n.

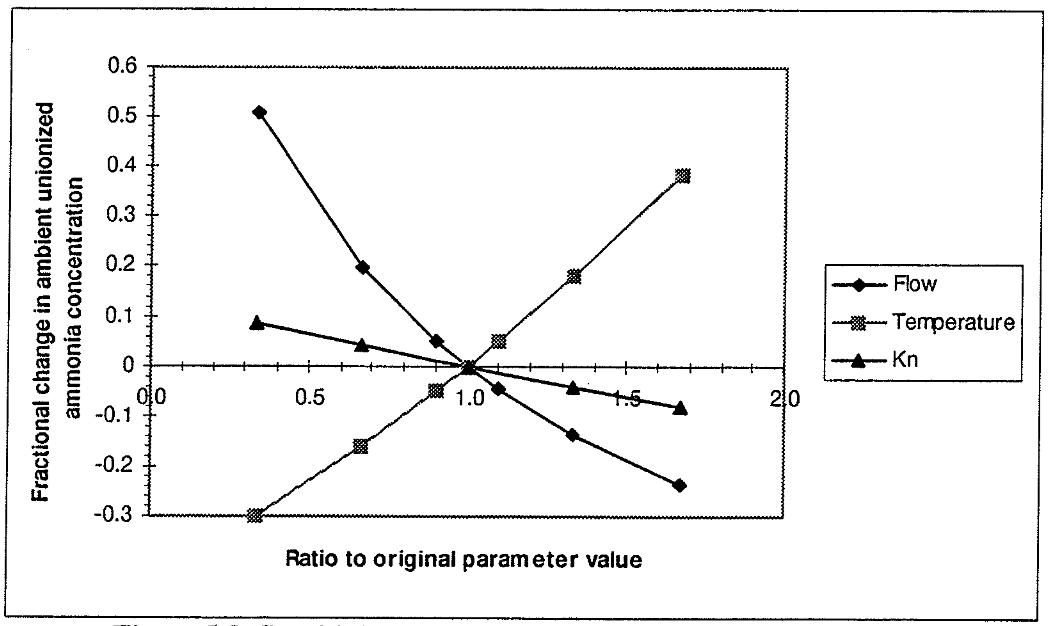

Figure 5.8 Sensitivity analysis of unionized ammonia simulation

Figure 5.9 shows the sensitivity of BOD_5 simulation to changes in K_r, flow, and temperature. Again, the concentration of BOD_5 decreases as K_r, flow, and temperature increase, and vice versa. BOD_5 concentration is most sensitive to changes in K_r and least sensitive to changes in temperature.

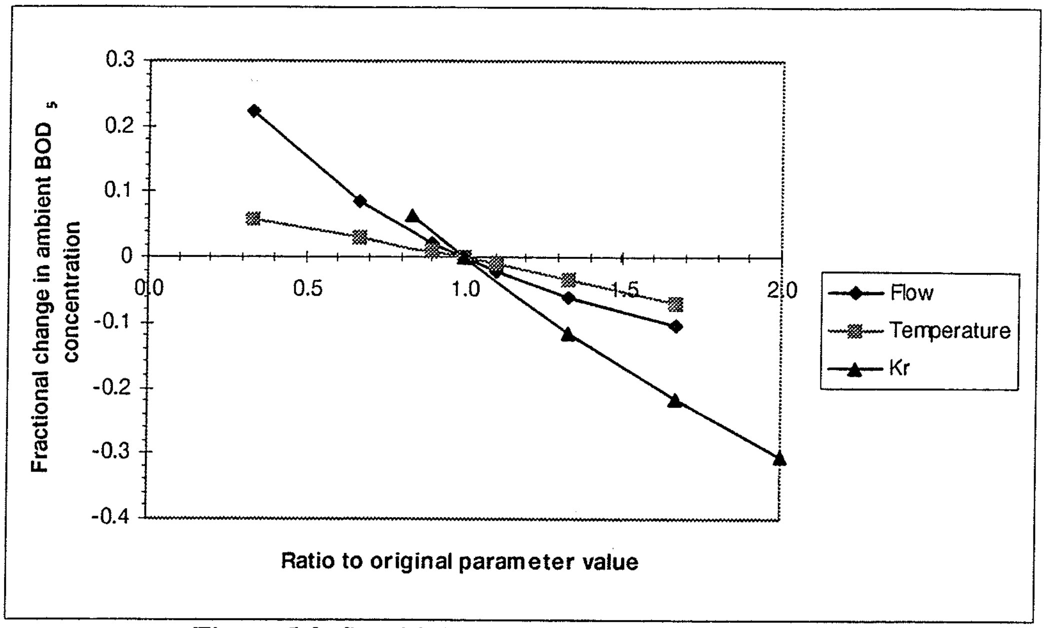

Figure 5.9 Sensitivity analysis of BOD_5 simulation

Figure 5.10 shows the sensitivity of DO simulation to changes in K_a, K_n, K_r, flow, and temperature. Increases in the pollutant removal rates cause the DO concentration to decrease because an increase in pollutant oxidation means an increased amount of oxygen needs to be consumed by the oxidizing bacteria. Increases in temperature also cause the DO concentration to decrease because the amount of oxygen that can be retained by water, or the saturation concentration of DO, decreases as temperature increases. Finally, increases in flow and K_a cause the concentration of DO to increase since an increase in flow causes the rate of atmospheric reaeration to increase, which in turns increases the concentration of DO.

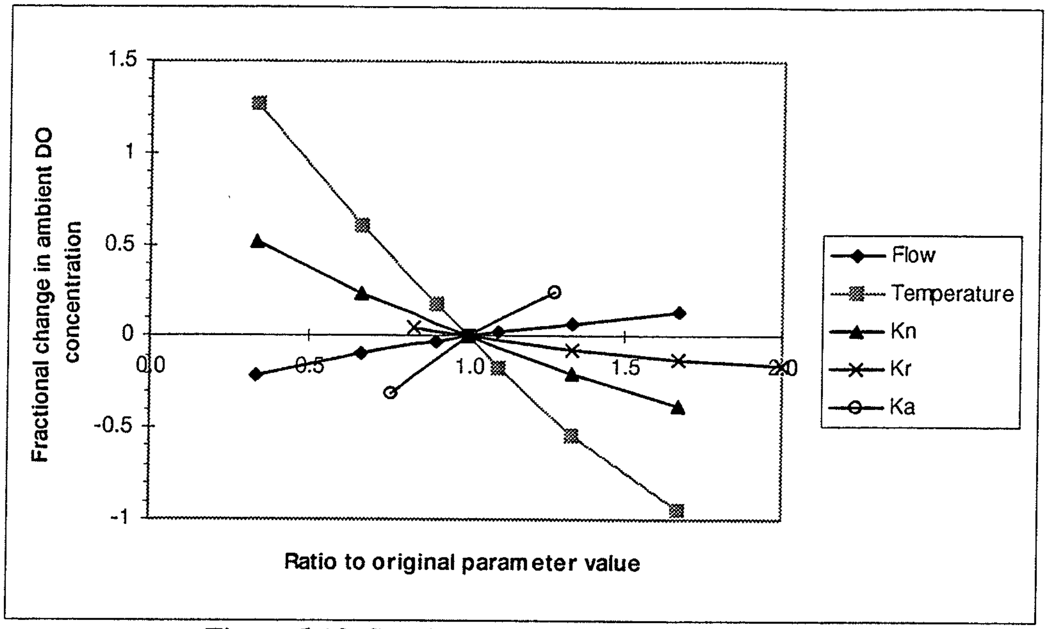

Figure 5.10 Sensitivity analysis of DO simulation

The calibrated water quality simulation model has a K_n of 0.3 /day and a K_r of 0.6 /day at 20°C, and values of K_a calculated using the formula developed by O'Connor and Dobbins (1958). The calibration is weak due to the lack of reliable water quality data. The results of the sensitivity analyses show that the simulation of ammonia and unionized ammonia is most sensitive to changes in stream flow, BOD_5 simulation is most sensitive to changes in the parameter K_r, and DO simulation is most sensitive to changes in temperature. The effects of changes in parameter values and stream design conditions on simulation is more pronounced for the Nádor Channel than for the Malom Channel.

Chapter 6

Water Quality Management of the Veszprémi-Séd River and Malom and Nádor Channel System

In order to identify water quality management strategies for the Veszprémi-Séd River and Malom and Nádor Channel System, wastewater treatment alternatives and costs are generated for the dischargers and the DSS DESERT is used in the process of evaluating WLAs. The three policies examined are the UEL, LC, and IEL policies. The efficacy of these three policies is examined on the bases of cost efficiency, equity, certainty of system outcome, and administrative ease.

6.1 Wastewater Treatment Alternatives and Costs

6.1.1 Available Information

The available information for the generation of wastewater treatment alternatives and costs include:

(1) descriptions of existing wastewater treatment schemes from individual dischargers,

(2) estimations of the annual OMC from individual dischargers,

(3) effluent flow and effluent water quality data collected by the Regional Environmental Inspectorate for the years 1991 and 1992 with a maximum of four measurements per year for all 14 dischargers except Nitrogén Müvek and Bakony Works, whose effluent BOD_5 concentrations are estimated, and

(4) effluent flow and effluent water quality data of the eight dischargers whose effluents were sampled during the field sampling program conducted in November, 1993.

6.1.2 Generation of Alternatives and Costs

Wastewater treatment alternatives and the corresponding costs are generated for nine dischargers in the Veszprémi-Séd River and Malom and Nádor Channel System. Treatment alternatives and costs are not generated for five dischargers in the system:

(1) Nitrokémia Rt. Plant V,

(2) Nitrokémia Rt. Plant VI,

(3) SZIM Köszörügépgyár Rt.,

(4) Bakony Works Emission I, and

(5) Bakony Works Emission II.

These five emissions are treated as background sources in the Veszprémi-Séd River and Malom and Nádor Channel System. Wastewater treatment alternatives are not generated for Nitrokémia Rt. Plants V and VI because these two dischargers emit cooling water with low mass loads of the pollutants of concern, ammonia and BOD_5. Table 6.1 lists the ammonia and BOD_5 mass loads of all the dischargers. SZIM Köszörügépgyár Rt. is an electroplating plant which performs industrial wastewater treatment and its effluent again has low mass loads of ammonia and BOD_5. Information on the current wastewater treatment scheme and costs of Bakony Works' two emissions are unavailable. Since the effluent of Bakony Works has low mass loads of ammonia and BOD_5, its two emissions are treated as background sources also.

The alternatives for the treatment of the two conventional pollutants of ammonia and BOD_5 range from primary mechanical treatment to tertiary treatment for nutrient removal. Ten wastewater treatment alternatives are generated for the dischargers in the Veszprémi-Séd River and Malom and Nádor Channel System:

(1) primary treatment, P,

(2) chemically enhanced primary treatment, CEPT,

(3) primary precipitation, PC,

(4) P and secondary treatment with high load activated sludge, B1,

(5) P and secondary treatment with low load activated sludge, B,

(6) CEPT and secondary treatment with low load activated sludge, BC1,

(7) PC and secondary treatment with low load activated sludge, BC2,

(8) BC1 and partial denitrification, BC1DN,

(9) BC2 and partial denitrification, BC2DN, and

(10)BC2 and denitrification, BCDN.

The design of these chosen treatment systems are well studied and the operation of such systems are relatively simple, precluding the need for unrealistic levels of training in CEE (Somlyódy *et al.*, 1994).

Table 6.1 Pollutant mass loads of dischargers

Discharger	Discharge	Ammonia		BOD_5	
	(m^3/s)	Concentration (mg/l)	Mass Load (g/s)	Concentration (mg/l)	Mass Load (g/s)
Bakony Works Emission I	0.0120	9.76	0.117	18.9	0.23
Veszprém MWWTP	0.1690	23.66	3.999	24.8	4.19
Nitrokémia Plants I-IV	0.2000	27.40	5.480	55.3	11.06
Nitrogén Müvek Rt.	0.2100	5.68	1.193	20.0	4.20
Nitrokémia Rt. Plant V	0.0370	4.39	0.162	17.8	0.66
Balatonfüred MWWTP	0.0370 [1] 0.0270 [2]	23.26	0.861 [1] 0.628 [2]	17.2	0.64 [1] 0.46 [2]
Balatonfüzfö MWWTP	0.0840 [1] 0.0430 [2]	12.75	1.071 [1] 0.548 [2]	19.6	1.65 [1] 0.84 [2]
Nitrokémia Rt. Plant VI	0.0100	4.39	0.044	17.8	0.18
Peremarton Chemical Works	0.0200	33.25	0.665	112.5	2.25
Várpalota MWWTP	0.0440	2.19	0.096	13.7	0.60
Bakony Works Emission II	0.0004	14.94	0.006	21.5	0.01
SZIM Köszörügépgyár	0.0030	0.96	0.003	45.0	0.14
ALBA Textil Kft.	0.0220	0.66	0.015	73.0	1.61
Székesfehérvár MWWTP	0.3510	22.23	7.803	16.5	5.79

[1] Effluent discharge for the August critical period
[2] Effluent discharge for the March critical period

6.1.3 Assumptions

The generated wastewater treatment costs are based on the current wastewater flows and characteristics. Currently, the primary level of wastewater treatment, sedimentation, is performed by all nine dischargers for whom wastewater treatment alternatives are generated. The existing wastewater treatment technologies employed by the dischargers are shown in Table 6.2. Besides ALBA Textil Kft. who performs primary treatment only, the other eight dischargers use biological wastewater treatment processes to treat their effluents beyond the primary level. The activated sludge process, a commonly used type of biological wastewater treatment, is used currently by five dischargers in the Veszprémi-Séd River and Malom and Nádor Channel System and it is the biological wastewater treatment technology that is assumed to be used for the upgrade of wastewater treatment plants. The dischargers that do not use the activated sludge process currently are assumed to switch to the use of activated sludge systems in their upgrades.

Two of the four industrial dischargers currently perform industrial wastewater pretreatment prior to biological wastewater treatment. The design of industrial pretreatment alternatives for the individual industries are beyond the scope of this study; therefore the pretreatment currently performed by the industrial dischargers is assumed to be adequate for rendering the wastewater suitable for the biological wastewater treatment alternatives that are generated for all dischargers.

The project life and the discount rate, both of which cannot be known with certainty, must be assumed in order to amortize the investment cost. The World Bank currently uses an interest rate of 10% to evaluate loan applications for environmental projects in CEE (Somlyódy, 1995). Using the producer price index for Hungary of 2% to account for inflation, a discount rate of 8% is used (Smith, 1995). A typical project life of 20 years is assumed for the analysis.

Table 6.2 Existing wastewater treatment technologies used by dischargers

Discharger	Wastewater Treatment
Veszprém MWWTP	Primary treatment, P Secondary treatment using activated sludge, B1
Balatonfüred MWWTP	Primary treatment, P Secondary treatment using activated sludge, B1
Balatonfüzfö MWWTP	Primary treatment, P Secondary treatment using activated sludge, B1
Várpalota MWWTP	Primary treatment, P Facultative ponds
Székesfehérvár MWWTP	Primary treatment, P Secondary treatment using activated sludge, B1
Nitrokémia Plants I-IV	Industrial pretreatment including neutralization and settling Primary treatment, P Secondary treatment using activated sludge, B1
Nitrogén Müvek Rt.	Stabilization ponds
Peremarton Chemical Works	Industrial pretreatment including neutralization Primary treatment, P Secondary treatment using trickling filters
ALBA Textil Kft.	Primary treatment, P

6.1.4 Estimated Alternatives and Costs

While it is possible to generate the costs of all 10 wastewater treatment technologies discussed in Section 6.1.2 for each discharger, only the non-inferior alternatives are used. That is, if the alternatives for each discharger are ranked from least expensive to most expensive, each alternative must have at least one effluent pollutant concentration, among ammonia and BOD_5, that is lower than that of the next less expensive alternative. Furthermore, it is assumed that dischargers are not allowed to increase their emissions from the current level under any new water quality management policies. Therefore the set of wastewater treatment alternatives for each discharger includes, besides the existing facility, only the alternatives that have lower effluent pollutant concentrations than the existing facility for at least one of the pollutants. The existing wastewater treatment technology and costs, and the non-inferior sets of the generated wastewater treatment alternatives and costs assuming a discount rate of 8% and a project life of 20 years are listed in Tables 6.3 and 6.4 for the MWWTPs and the industrial dischargers, respectively. The investment costs (IC) of facility upgrades, OMC, change in OMC from the current level (ΔOMC), and change in the total annual costs (ΔTAC) are given in 1992 U.S. dollars. Detailed information on the generation of wastewater treatment alternations and costs for individual dischargers are given in Appendix B.

Table 6.3 Wastewater treatment alternatives and costs of municipal dischargers

Discharger	Alternative[1]	IC[2]	OMC[3]	ΔOMC[4]	ΔTAC[5]	Effluent Discharge	Effluent Water Quality	
							BOD_5	Ammonia-N
		10^6 $	10^6 $/year	10^6 $/year	10^6 $/year	m^3/s	mg/l	mg/l
Veszprém MWWTP	Existing	0.00	0.58	0.00	0.00	0.169	25	23.7
	B	3.87	0.58	0.00	0.39		25	1.3
	BC2	2.99	0.90	0.32	0.63		13	1.3
	BC1DN	5.16	0.92	0.34	0.86		13	0.5
	BCDN	7.13	1.16	0.57	1.30		8	0.3
Balatonfüred MWWTP	Existing	0.00	0.10	0.00	0.00	0.037 [6]	17	23.3
	B	0.50	0.10	0.00	0.05	0.027 [7]	25	1.3
	BC2	0.64	0.16	0.06	0.12		13	1.3
	BC1DN	1.01	0.16	0.06	0.16		13	0.5
	BCDN	1.19	0.20	0.10	0.22		8	0.3
Balatonfüzfö MWWTP	Existing	0.00	0.18	0.00	0.00	0.084 [6]	20	12.8
	B	1.62	0.18	0.00	0.16	0.043 [7]	25	1.3
	BC2	1.69	0.28	0.10	0.27		13	1.3
	BC1DN	1.89	0.29	0.11	0.30		13	0.5
	BCDN	2.66	0.36	0.18	0.45		8	0.3
Várpalota MWWTP	Existing	0.00	0.09	0.00	0.00	0.044	14	2.2
	B	1.80	0.15	0.06	0.24		25	1.3
	BC2	1.97	0.23	0.14	0.34		13	1.3
	BC1DN	2.37	0.23	0.14	0.39		13	0.5
	BCDN	3.16	0.29	0.20	0.53		8	0.3
Székesfehérvár MWWTP	Existing	0.00	1.16	0.00	0.00	0.351	17	22.2
	B	12.98	1.16	0.00	1.32		52	1.3
	BC2	9.78	1.81	0.65	1.64		26	1.3
	BC1DN	19.14	1.84	0.68	2.63		26	0.5
	BCDN	25.50	2.31	1.15	3.75		16	0.3

[1] P: primary treatment
CEPT: chemically enhanced primary treatment
PC: primary precipitation
B1: P and secondary treatment with high load activated sludge
B: P and secondary treatment with low load activated sludge
BC1: CEPT and secondary treatment with low load activated sludge
BC2: PC and secondary treatment with low load activated sludge
BC1DN: BC1 and partial denitrification
BCDN: BC2 and denitrification

[2] Investment costs

[3] Operation and maintenance costs

[4] Change in operations and maintenance costs from existing level

[5] Change in total annual costs from existing level assuming a discount rate of 8% and an amortization period of 20 years

[6] Effluent discharge for the August critical period

[7] Effluent discharge for the March critical period

Table 6.4 Wastewater treatment alternatives and costs of industrial dischargers

Discharger	Alternative	IC	OMC	ΔOMC	ΔTAC	Effluent Discharge	Effluent Water Quality	
							BOD_5	Ammonia-N
		10^6 $	10^6 $/year	10^6 $/year	10^6 $/year	m^3/s	mg/l	mg/l
Nitrokémia Plants I-IV	Existing	0.00	1.52	0.00	0.00	0.200	55	27.4
	B	5.80	1.52	0.00	0.59		55	1.3
	BC2	6.21	1.90	0.37	1.00		28	1.3
	BC1DN	8.52	1.92	0.39	1.26		28	0.5
	BCDN	10.48	2.19	0.66	1.73		17	0.3
Nitrogén Müvek Rt.	Existing	0.00	0.03	0.00	0.00	0.210	21	5.7
	B1	2.59	0.74	0.70	0.97		15	35.1
	B	2.94	0.74	0.70	1.00		5	1.3
	BC2	2.85	1.14	1.11	1.40		3	1.3
	BC1DN	2.75	1.17	1.13	1.41		3	0.5
	BCDN	3.68	1.46	1.43	1.81		2	0.3
Peremarton Chemical Works	Existing	0.00	0.05	0.00	0.00	0.020	113	33.3
	PC	0.50	0.05	0.00	0.05		75	36.5
	BC1	0.91	0.05	0.01	0.10		25	1.3
	BC2	1.18	0.06	0.01	0.13		13	1.3
	BC1DN	1.47	0.06	0.01	0.16		13	0.5
	BCDN	1.98	0.07	0.02	0.22		8	0.3
ALBA Textil Kft.	Existing	0.00	0.05	0.00	0.00	0.022	73	0.7
	B	0.99	0.08	0.03	0.13		37	1.3
	BC2	1.28	0.12	0.07	0.20		18	1.3
	BC1DN	1.81	0.12	0.08	0.26		18	0.5
	BCDN	2.28	0.15	0.11	0.34		11	0.3

6.2 Water Quality Management Models

The three water quality management policies that are investigated are the UEL, LC, and IEL policies. The WLAs for the LC and IEL policies are determined using DESERT's optimization unit in addition to the simulation unit. However, the DP formulation in DESERT's optimization unit allows only water quality constraints, so the additional constraint required by the UEL policy to equate the effluent limit for all dischargers cannot be accommodated. Therefore, only simulation is used for the UEL policy and the minimum UEL is determined through a direct-search approach, described in Section 6.2.1.

6.2.1 UEL Formulation

The minimum UELs cannot be evaluated directly using DESERT since DESERT's optimization unit does not accommodate the constraint needed to equate the effluent limit for all dischargers. The process for determining the minimum UEL is shown in Figure 6.1. The minimum UEL is determined by repetitively decreasing the uniform effluent limit and running the simulation model in DESERT until the ambient water quality criteria are met at all water quality receptor locations.

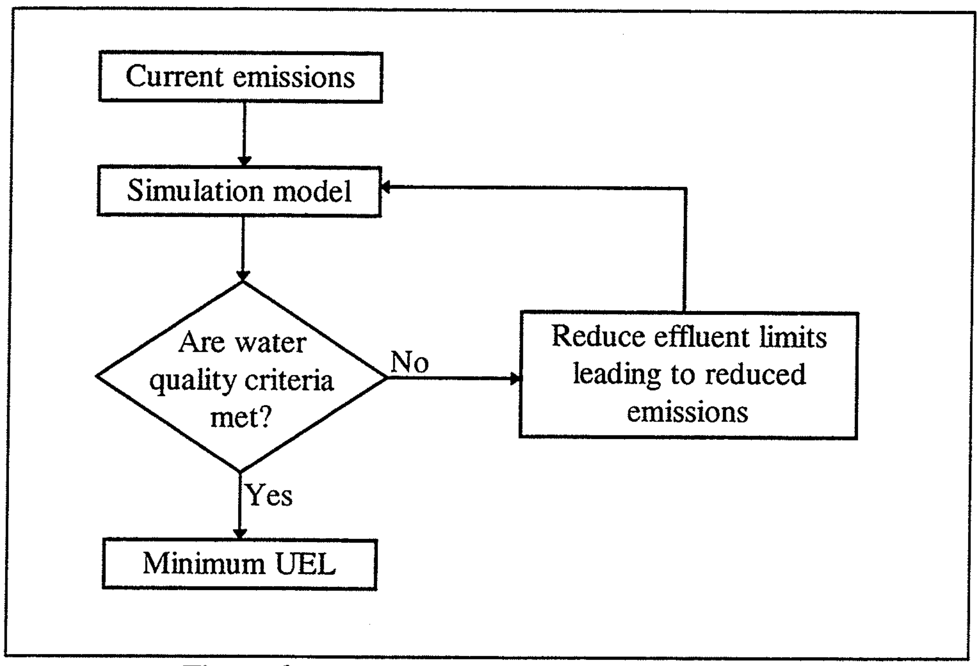

Figure 6.1 Determination of the minimum UEL

6.2.2 LC Formulation

Under the LC policy, ambient water quality criteria are maintained while minimizing the sum of the wastewater treatment costs of all dischargers. Using DP, the decisions are the wastewater treatment alternatives to be used by the dischargers, the stage corresponds to a river reach, and the states are the instream water quality conditions. The transformation that relates the states at one stage to the decisions of the previous stage is carried out by the water quality simulation model. The recursion equation that relates the costs incurred from one stage to those in the next stage is the cost minimization objective. Water quality constraints are evaluated at the water quality receptor locations of concern. Therefore, at each reach:

$$f_n(Q_n) = \min_{x_n \in X_n} [C_n(x_n) + f_{n-1}(Q_{n-1})] \tag{6.1}$$

$$Q_n = M_n(x_n, Q_{n-1}) \tag{6.2}$$

$$Q_n \geq S_n \tag{6.3}$$

where n = the stage or reach number;

$f_n(Q_n)$ = the recursion equation, or the value of the objective function at the n^{th} reach;

Q_n = the water quality at the downstream end of the n^{th} reach;

x_n = the decision, or the wastewater treatment alternative at the n^{th} reach;

X_n = the set of possible wastewater treatment alternatives at the n^{th} reach;

$C_n(x_n)$ = the cost function associated with the wastewater treatment alternatives at the n^{th} reach;

$M_n(x_n, Q_{n-1})$ = the transformation of water quality states using the water quality simulation model given the wastewater treatment alternative x_n and the water quality states of stage n-1; and

S_n = the water quality constraint, or the minimum water quality to be achieved at the downstream end of the n^{th} reach.

Function 6.1 is the recursion equation for the LC formulation. At a stage where no wastewater treatment alternatives are present, the cost function $C_n(x_n)$ is set to zero. Equation 6.2 represents the transformation of water quality between states. Equation 6.3 represents the water quality constraint that applies only at a stage where a water quality receptor location is present at the top of the immediately downstream reach.

6.2.3 IEL Formulation

Under the IEL policy, the total pollutant mass load in the river basin is maximized while maintaining the desired level of water quality. The only difference between the IEL and LC formulations appears in the recursion equation. In the optimization model in DESERT, the cost data of the dischargers are replaced by the mass loads that are emitted. Therefore, at each stage:

$$f_n(Q_n) = \max_{x_n \in X_n} [P_n(x_n) + f_{n-1}(Q_{n-1})] \tag{6.4}$$

$$Q_n = M_n(x_n, Q_{n-1}) \tag{6.5}$$

$$Q_n \geq S_n \tag{6.6}$$

where $P_n(x_n)$ represents the mass load of the pollutant discharged as a result of the decision at stage n, and all other variables are as described above. Again, at a stage where no dischargers are present, the function $P_n(x_n)$ is set to zero.

6.3 Comparison of the Policies

WLAs for the UEL, LC, and IEL policies are determined for the management of ammonia, BOD, and DO. The water quality constraints used correspond to the Hungarian and EU ambient ammonia, unionized ammonia, BOD_5, and DO criteria for fishery needs, and the USEPA ambient unionized ammonia criterion. The following sections compares the UEL, LC, and IEL policies on the bases of cost efficiency, equity, certainty of system outcome, and administrative ease.

6.3.1 Cost Efficiency

6.3.1.1 Ammonia Management

The WLAs for the management of ammonia are based on the design conditions of the March critical period. Table 6.5 shows the incremental costs required for meeting the ambient ammonia and unionized ammonia criteria in the Veszprémi-Séd River and Malom and Nádor Channel System. For the UEL policy, the minimum UEL required for meeting the ambient ammonia and unionized ammonia criteria are given in addition to the incremental costs. The crosses in the table indicate that the EU Guide and the Hungarian

Objective for ambient ammonia and unionized ammonia cannot be achieved under the stream conditions of the March critical period. The most stringent criterion that can be achieved given the generated alternatives is the Hungarian ambient ammonia criterion at the tolerable level in which case all the dischargers employ the highest level of waste removal possible regardless of the policy choice. The criterion that can be met with the least incremental costs is the USEPA unionized ammonia criterion. Except for the USEPA unionized ammonia criterion, the incremental costs required to achieve any criteria is lowest under the LC policy and highest under the UEL policy.

Table 6.5 Incremental costs for ammonia management

Ambient Criteria (mg/l as nitrogen)		ΔTAC in 10^6 US$/year		
		UEL (Minimum UEL in mg/l of Ammonia)	LC	IEL
Ammonia Criteria	EU, Mandatory: 0.8 mg/l	7.2 (1.0)	4.8	5.6
	Hungarian, Tolerable: 0.5 mg/l	10.3 (0.3)	10.3	10.3
	EU, Guide: 0.2 mg/l	×	×	×
	Hungarian, Objective: 0.05 mg/l	×	×	×
Unionized Ammonia Criteria	Hungarian, Tolerable; EU, Mandatory: 0.021 mg/l	7.4 (0.5)	5.9	6.5
	Hungarian, Objective; EU, Guide: 0.004 mg/l	×	×	×
	USEPA: 0.089 mg/l [1]	2.6 (8.0)	2.3	5.3

[1] The USEPA criterion value varies depending on the pH and the temperature. The value given is for the water quality receptor location at the pumping location in the Nádor Channel.

Figure 6.2 compares the costs of maintaining ambient ammonia criteria under the three policies. In order to compare the costs of achieving the different ambient criteria for ammonia management, the unionized ammonia criteria are converted to equivalent ammonia criteria in Figure 6.2 based on the temperature and the pH at the critical water quality receptor location, the pumping location in the Nádor Channel. The ambient unionized ammonia criteria of 0.021 and 0.089 mg/l are converted to equivalent ammonia criteria of 0.72 and 3.05 mg/l, respectively. The incremental costs of all three policies increase sharply when the ambient ammonia criteria are more stringent than 0.8 mg/l, the Hungarian ammonia criterion at the tolerable level. For the ambient ammonia criterion of 0.5 mg/l, the WLAs of all three policies have the same outcome as the BAT policy, where all dischargers upgrade their wastewater treatment facilities to the highest possible level. For all the ambient ammonia criteria less stringent than 0.5 mg/l, the UEL policy is more expensive than the LC policy. For ambient ammonia criteria more stringent than 0.8 mg/l, the costs of the IEL policy approach those of the LC policies. However, for the USEPA unionized ammonia criterion, or the equivalent ammonia criterion of 3.05 mg/l, the IEL policy is more than two times as expensive as the UEL policy.

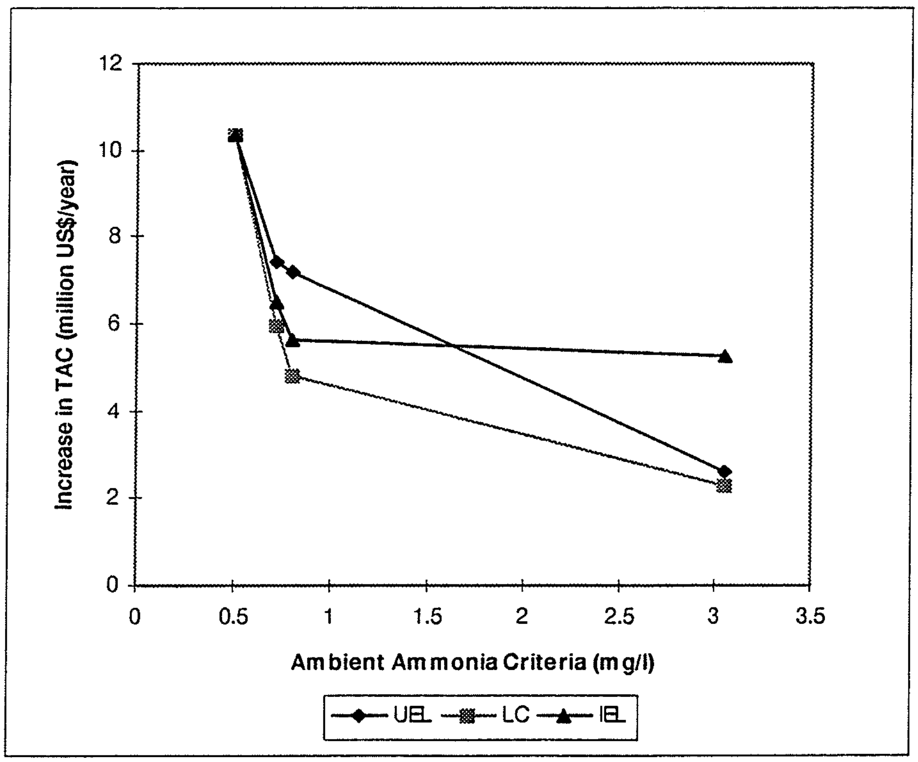

Figure 6.2 Cost curves for the UEL, LC, and IEL policies

The IEL policy can be costly because unlike the LC formulation, the IEL formulation takes into consideration the locational effects but not the relative costs of the dischargers. Table 6.6 shows the WLAs required for meeting the USEPA ambient unionized ammonia criterion, or the equivalent ammonia criterion of 3.05 mg/l, under the UEL, LC, and IEL policies. Under the LC policy, only two dischargers are required to perform additional waste removal in order for the USEPA ambient unionized ammonia criterion to be met as shown in Table 6.6. One of these two dischargers is Nitrokémia Rt. Plants I-IV, the second largest discharger of ammonia and the discharger that is furthest away from the receptor locations where the criterion is to be met. Under the IEL policy, however, Nitrokémia Rt. Plants I-IV emits at its current level while three smaller dischargers (Nitrogén Müvek Rt., Balatonfüzfö MWWTP, and Peremarton Chemical Works) that are closer to the receptor locations are required to perform additional waste removal. The WLAs of the three policies for achieving the different ambient ammonia and unionized ammonia criteria are contained in Appendix C.

Table 6.6 WLAs for meeting the USEPA ambient unionized ammonia criterion

Discharger	WLA		
	UEL	LC	IEL
Veszprém MWWTP	B	-	B
Nitrokémia Rt. Plants I-IV	B	B	-
Nitrogén Müvek Rt.	-	-	B
Balatonfüred MWWTP	B	-	-
Balatonfüzfö MWWTP	B	-	BC1DN
Peremarton Chemical Works	BC1	-	BCDN
Várpalota MWWTP	-	-	-
Székesfehérvár MWWTP	B	B	BCDN
ALBA Textil Kft.	-	-	-
ΔTAC in 10^6 US$/year	2.6	2.3	5.3

The achievement of ambient ammonia or unionized ammonia criteria does not necessarily mean that ambient BOD_5 criteria are met. For example, the WLAs for meeting the USEPA unionized ammonia criterion result in ambient BOD_5 concentrations that violate the Hungarian BOD_5 criterion at the tolerable level.

6.3.1.2 Management of Ammonia and BOD

Management of ammonia and BOD is considered in the March critical period. The Hungarian ambient BOD_5 criteria are 12 and 6 mg/l for the tolerable and the objective levels, respectively. EU specifies only a higher level of ambient criterion for BOD_5 of 6 mg/l. Even if a BAT policy is used in the Veszprémi-Séd River and Malom and Nádor Channel System, the 6 mg/l criterion can only be met at the water quality receptor locations in the Malom Channel. In the Nádor Channel, the lowest ambient concentration of BOD_5 that can be achieved using a BAT policy is 7 mg/l. Therefore only the Hungarian ambient BOD_5 criterion of 12 mg/l is used in combination with the ambient ammonia and unionized ammonia criteria for the joint management of ammonia and BOD. The WLAs for the joint management of ammonia and BOD are contained in Appendix C.

Table 6.7 lists six different combinations of the ambient ammonia, unionized ammonia, and BOD_5 criteria and the associated incremental costs under the three policies. The most stringent combination of criteria that is achievable is the combination of Hungarian Tolerable criteria as shown in Table 6.7. For this combination of criteria, the costs are the same as those incurred when only ammonia is managed for all three policies, since the high level of wastewater treatment required due to the stringent ammonia and unionized ammonia criteria can reduce the emission of BOD to such a level that the ambient concentration of BOD_5 is lower than 12 mg/l at the water quality receptor locations. When the BOD_5 criterion is combined with the less stringent ammonia or unionized ammonia criteria, the costs increase from the amounts required for the management of ammonia alone.

Table 6.7 Incremental costs for the joint management of Ammonia and BOD

Category	Ambient Criteria (mg/l)	ΔTAC in 10^6 US$/year		
		UEL	LC	IEL
Least stringent combination	USEPA unionized ammonia, 0.089 mg/l Hungarian BOD_5, 12 mg/l	5.9	4.7	6.0
EU Mandatory, ammonia only	Ammonia, 0.8 mg/l (Hungarian) BOD_5 , 12 mg/l	8.5	6.8	8.0
EU Mandatory	Ammonia, 0.8 mg/l Unionized ammonia, 0.021 mg/l (Hungarian) BOD_5 , 12 mg/l	8.5	7.6	7.7
Hungarian Tolerable	Ammonia, 0.5 mg/l Unionized ammonia, 0.021 mg/l BOD_5 , 12 mg/l	10.3	10.3	10.3
EC Guide	Ammonia, 0.2 mg/l Unionized ammonia, 0.004 mg/l BOD_5 , 6 mg/l	×	×	×
Hungarian Objective	Ammonia, 0.05 mg/l Unionized ammonia, 0.004 mg/l BOD_5 , 6 mg/l	×	×	×

Again, the LC policy is the least expensive policy. The IEL policy is most expensive for the two least stringent categories of criteria, but it costs the same as the UEL policy for the category of Hungarian tolerable criteria and less than the UEL policy for the category of EU mandatory criteria.

6.3.1.3 DO Management

DO is not a concern in the March critical period. Given the current level of pollutant discharge, the DO concentration in March at the critical water quality receptor location is still higher than 7 mg/l as shown in Figures 5.5 and 5.6 on pp. 53 and 54, meeting both levels of the Hungarian ambient DO criteria and the EU ambient DO criterion at the mandatory level. In the summer, however, even the least stringent ambient DO criterion is violated. The two levels of the Hungarian and EU ambient DO criteria are 4 and 6 mg/l, and 7 and 8 mg/l, respectively.

Figure 6.3 shows the incremental costs of the UEL, LC, and IEL policy for meeting ambient DO criteria. The EC Guide of 8 mg/l is not achievable even if the BAT policy is used. Again, the LC policy is least expensive for meeting any ambient DO criteria. The UEL policy is most expensive except for the ambient DO criterion of 4 mg/l, where the UEL and IEL policies are equal in costs. The WLAs for the management of DO are included in Appendix C.

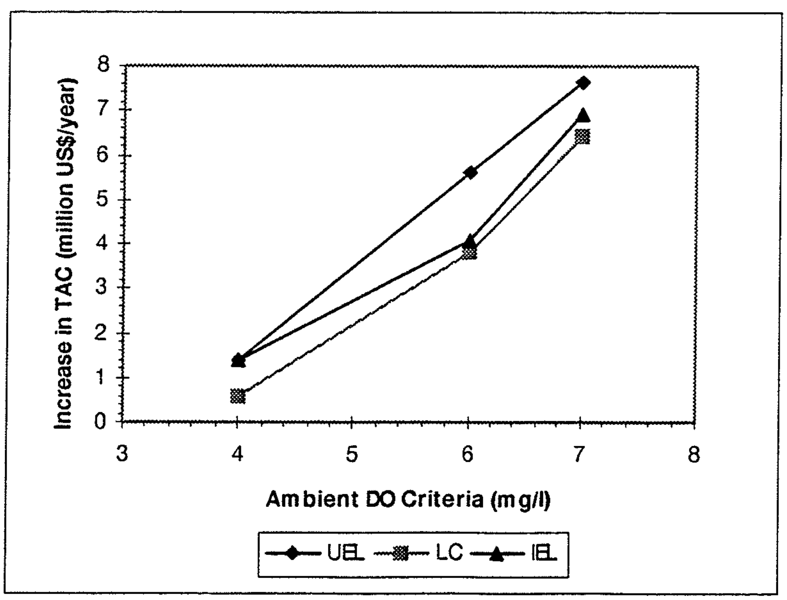

Figure 6.3 Incremental costs for meeting ambient DO criteria

6.3.2 Equity

Although LC policies are well noted for cost efficiency, conventional practice in water quality management favours UT programs. One of the attractive characteristics of UT programs is their equitable treatment of polluters by requiring all polluters to perform the same percentage of waste removal or in the case of the UEL program, by requiring all polluters to achieve the same effluent pollutant concentration.

Figure 6.4 compares the levels of waste removal of the dischargers under the UEL, LC, and IEL policy for achieving the ambient ammonia criterion of 0.8 mg/l. Figure 6.4(a) shows that eight of the nine dischargers perform ammonia removal between 98 and 99% under the UEL policy. ALBA Textil Kft. does not perform any additional wastewater treatment since it currently emits effluent with an ammonia concentration of less than 1 mg/l, the required UEL for meeting the ambient ammonia criterion of 0.8 mg/l. Although LC policies are generally inequitable, in the Veszprémi-Séd River and Malom and Nádor Channel System eight of the nine dischargers perform ammonia removal between 92 and 99% under the LC policy as shown in Figure 6.4(b). The full range of the percentage removal of ammonia is 45 to 99% under the LC policy. The WLAs of the LC and IEL policies differ only for three dischargers: Nitrogén Müvek Rt. and Balatonfüzfö MWWTP performs more ammonia removal under the IEL policy than under the LC policy while Balatonfüred MWWTP performs less ammonia removal under the IEL policy than under the LC policy. The IEL policy may be viewed as more equitable than the LC policy since the range of the percentage removal of ammonia is smaller for the IEL policy, 62 to 99%, than for the LC policy.

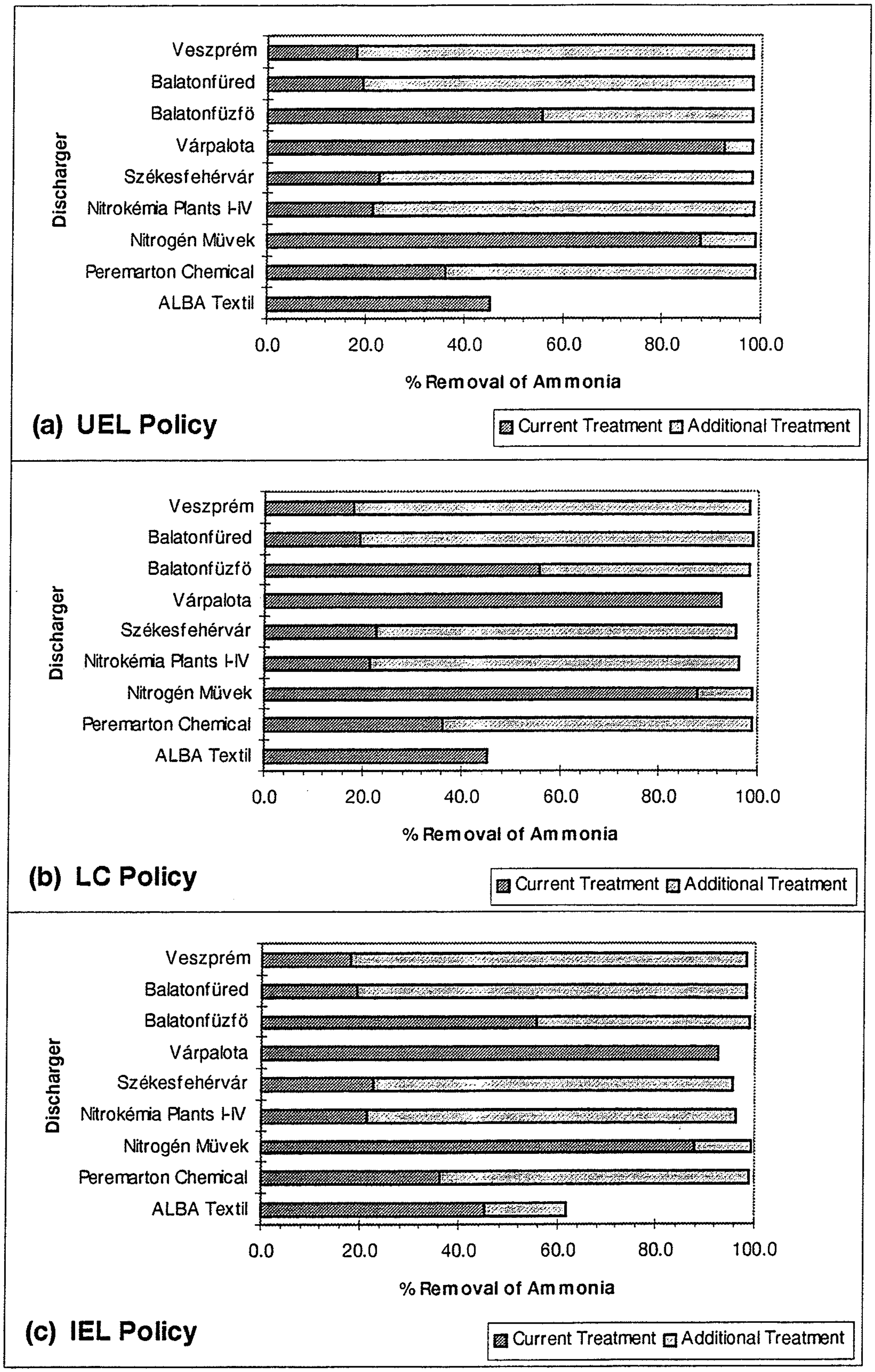

Figure 6.4 Percentage removal of ammonia for the ambient ammonia criterion of 0.8 mg/l

The costs of the UT and LC policies usually serve as the benchmarks for comparing the costs of water quality management programs. For the nine dischargers in the Veszprémi-Séd River and Malom and Nádor Channel System, the sums of their TAC under the UEL and LC policies for achieving the ambient ammonia criterion of 0.8 mg/l are 11.2 and 8.6 million U.S. dollars per year, respectively. Figure 6.5 compares the TAC of the individual dischargers under the UEL and LC policies for achieving the ambient ammonia criterion of 0.8 mg/l. Five of the nine dischargers incur the same TAC under the UEL and LC policies. Only Balatonfüred MWWTP has higher costs under the LC policy than the UEL policy. The three remaining dischargers experience substantial cost savings under the LC policy compared to the UEL policy. In Figure 6.5, the difference between the TAC of the LC and UEL policies for the dischargers can be considered to be the costs of equity.

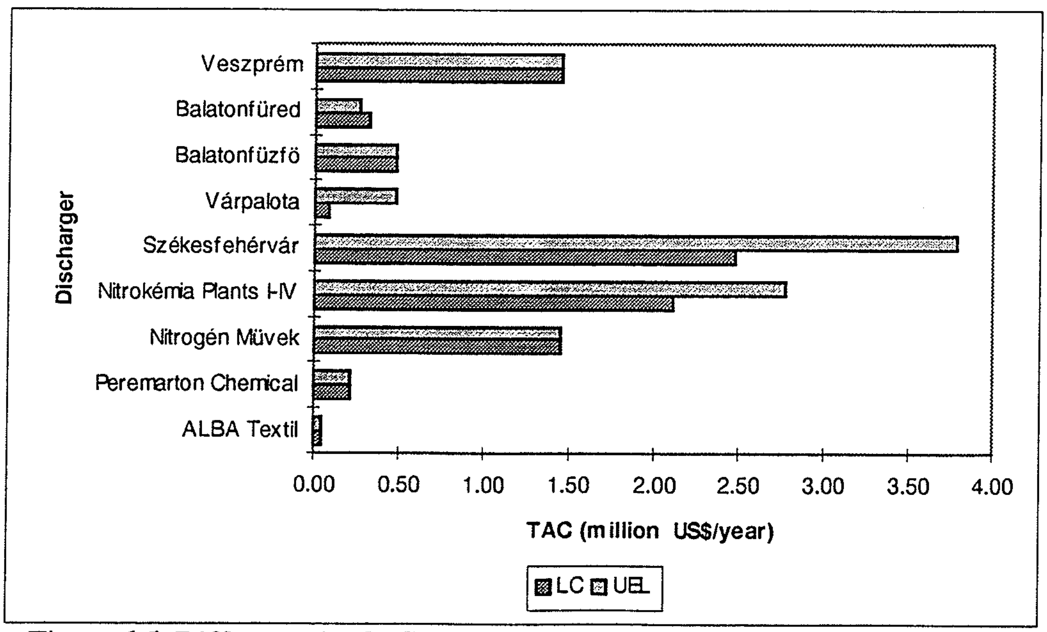

Figure 6.5 Difference in TAC between the UEL and LC policies for the ambient ammonia criterion of 0.8 mg/l

6.3.3 Certainty of System Outcome

The WLAs and costs of different management policies may change when conditions different from those assumed for design arise. The water available in the Veszprémi-Séd River and Malom and Nádor Channel System may be reduced in five to seven years due to the termination of mining activity that currently supplies 20 000 to 30 000 m^3/day of water to the Gaja Creek. The effect of this flow reduction and the sensitivity of the management optimization model to changes in design assumptions are the subjects of this section.

6.3.3.1 Effect of Reduced Flow from the Gaja Creek

The Gaja Creek is a tributary to the Nádor Channel which supplements the water supply in the Veszprémi-Séd River and Malom and Nádor Channel System. The mean flows of the Gaja Creek in March and August are 0.9 and 0.6 m^3/s, respectively. Since the Regional Environmental Inspectorate expects the flow in the Gaja Creek to reduce by 20 000 to 30 000 m^3/day due the termination of mining activities near the Gaja Creek in five to seven years, WLAs of the UEL, LC, and IEL policies are determined for two scenarios of reduced flow from the Gaja Creek. Assuming the same flow reduction throughout the year, the first scenario is a flow reduction of 30 000 m^3/day in the Gaja Creek resulting in flows of 0.5 and 0.2 m^3/s in March and August, respectively. The second scenario is a flow of 0.1 m^3/s in the Gaja Creek in March and August, representing a situation of severe water shortage in the Gaja Creek (Garáné, 1995).

The reduced dilution caused by flow reduction results in higher levels of waste removal and higher costs under all three policies. Figure 6.6 shows the increases in incremental costs of the three policies under the three flow scenarios for achieving the ambient ammonia criterion of 0.8 mg/l. Of the three policies, the UEL policy is least affected by flow reduction in the Gaja Creek since WLAs of the UEL policy result in reserved stream assimilative capacity that can buffer the effect of flow reduction. Under the UEL policy, the minimum UEL on ammonia required to meet the ambient ammonia criterion of 0.8 mg/l in the current flow scenario is 1.0 mg/l, where all dischargers upgrade their wastewater treatment facilities to BC1DN (partial denitrification) except ALBA Textil Kft. which requires no upgrade. Reduced flow in the Gaja Creek results in a minimum UEL of 0.5 mg/l of ammonia, where all dischargers upgrade their wastewater treatment facilities to BC1DN—only ALBA Textil Kft. is affected.

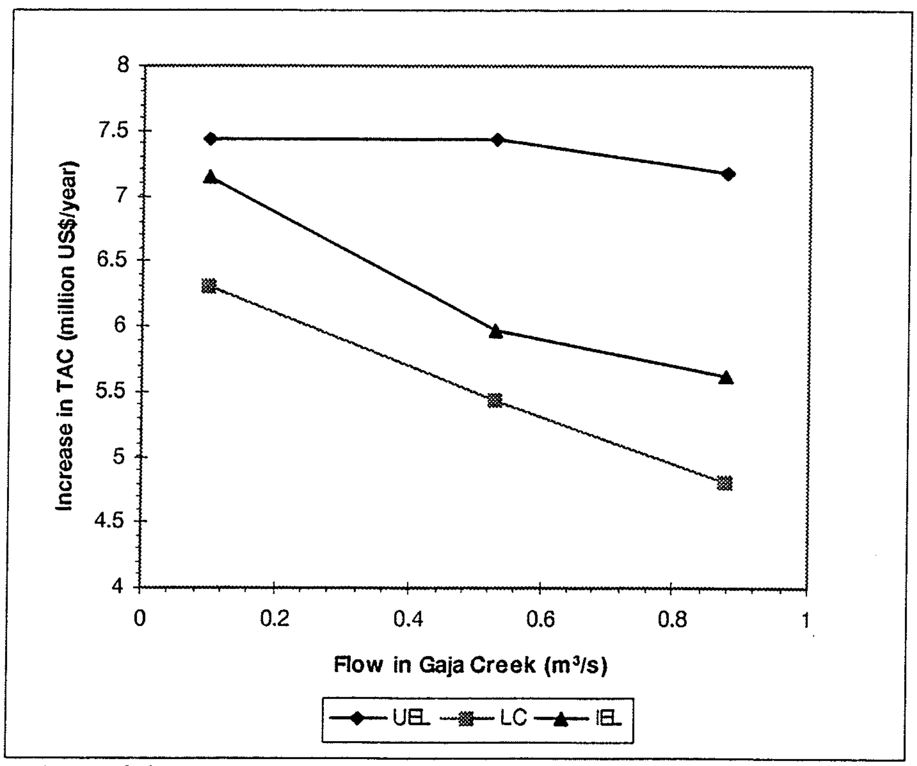

Figure 6.6 Effect of flow reduction on incremental costs for the ambient ammonia criterion of 0.8 mg/l

Contrary to the UEL policy, the optimal allocation of waste loads under the LC and IEL policies allows little or no reserved assimilative capacity; therefore, the WLAs and costs of the LC and IEL policies are more sensitive than those of the UEL policy to flow reduction. Table 6.8 shows the WLAs required to meet the ambient ammonia criterion of 0.8 mg/l under the LC and IEL policies for the three flow scenarios. Flow reduction does not result in gradual decreases in the allowed ammonia concentrations of the dischargers' effluents. Instead, some dischargers are allocated lower ammonia removal levels while others are required to perform more ammonia removal as the flow in the Gaja Creek is reduced. Under the LC policy, a flow reduction from the current amount of 0.9 m^3/s to 0.5 m^3/s in the Gaja Creek requires Nitrokémia Rt. Plants I-IV and Peremarton Chemical Works to increase ammonia removal, but Balatonfüred MWWTP can perform less ammonia removal, as shown in Table 6.8. A further flow reduction from 0.5 to 0.1 m^3/s in the Gaja Creek requires Várpalota and Székesfehérvár MWWTPs to increase ammonia removal, but Nitrokémia Rt. Plants I-IV, which increases its ammonia removal due to the first flow reduction, can perform less ammonia removal. Such "reversals" in the allocated waste loads occur under the IEL policy as well, as can be seen in Table 6.8. The allocated waste loads remain the same in the three flow scenarios for four and three of the nine dischargers under the LC and IEL policies, respectively.

Table 6.8 WLAs for meeting the ambient ammonia criterion of 0.8 mg/l under flow reduction

Discharger	WLA					
	LC Policy			IEL Policy		
	Q=0.9 [1]	Q=0.5 [1]	Q=0.1 [1]	Q=0.9 [1]	Q=0.5 [1]	Q=0.1 [1]
Veszprém MWWTP	BCIDN	BCIDN	BCIDN	BCIDN	BCIDN	BCIDN
Nitrokémia Rt. Plants I-IV	B	BCIDN	B	B	BCIDN	BCDN
Nitrogén Müvek Rt.	BCIDN	BCIDN	BCIDN	BCDN	BCDN	B
Balatonfüred MWWTP	BCDN	B	B	BCIDN	B	B
Balatonfüzfö MWWTP	BCIDN	BCIDN	BCIDN	BCDN	B	BCDN
Peremarton Chemical Works	BCIDN	BCDN	BCDN	BCIDN	BCIDN	BCIDN
Várpalota MWWTP	-	-	B	-	-	-
Székesfehérvár MWWTP	B	B	BCIDN	B	B	BCIDN
ALBA Textil Kft.	-	-	-	BCIDN	BCDN	BCIDN
ΔTAC in 10^6 US$/year	4.8	5.4	6.3	5.6	6.0	7.2

[1] Flow in the Gaja Creek

6.3.3.2 Sensitivity Analysis of Optimization Results

Sensitivity analyses are performed to investigate the results of the water quality management optimization model as input parameters are varied. The parameters that are analyzed are the discount rate, amortizing period, and stream flow. Flow is the stream design condition to which ammonia and unionized ammonia simulation of the Veszprémi-Séd River and Malom and Nádor Channel System are most sensitive. Moreover, the flow of the Gaja Creek may change drastically in five to seven years due to the termination of mining activity that currently supplies water to the system. The costs of meeting a criterion under the UEL, LC, and IEL policies are affected by the discount rate and amortizing period, and the allocated waste loads are affected by the discount rate and amortizing period under the LC policy. Sensitivity analyses are conducted using the LC policy only to illustrate the variability in the results of optimization, the WLAs and costs, that occur due to changes in the discount rate, amortizing period, and stream flow. The discount rate is varied from the original value of 8% within the range of 2 to 12%. The amortizing period, or the project life, is varied from the original length of 20 years within the range of 10 to 40 years (Somlyódy *et al.*, 1994). Stream flow is varied from the original value to ±10%, ±33%, and ±67%.

Figure 6.7 shows the sensitivity of the policy cost to changes in the discount rate, the amortizing period, and flow for achieving the ambient ammonia criterion of 0.8 mg/l. Changes in the discount rate, project life, and flow have greater effects on the policy costs during the March critical period than the August critical period; therefore the sensitivity of the costs of the LC policy to changes in the discount rate, the project life, and flow are only shown for the March critical period. The axes of the graph are normalized for comparison. As the flow increases, the costs for meeting the ambient ammonia criterion of 0.8 mg/l decrease due to increased dilution. Increases in the discount rate increase the TAC which in turn increase the policy costs. As the amortizing period is shortened, the TAC increases and again increases the policy costs.

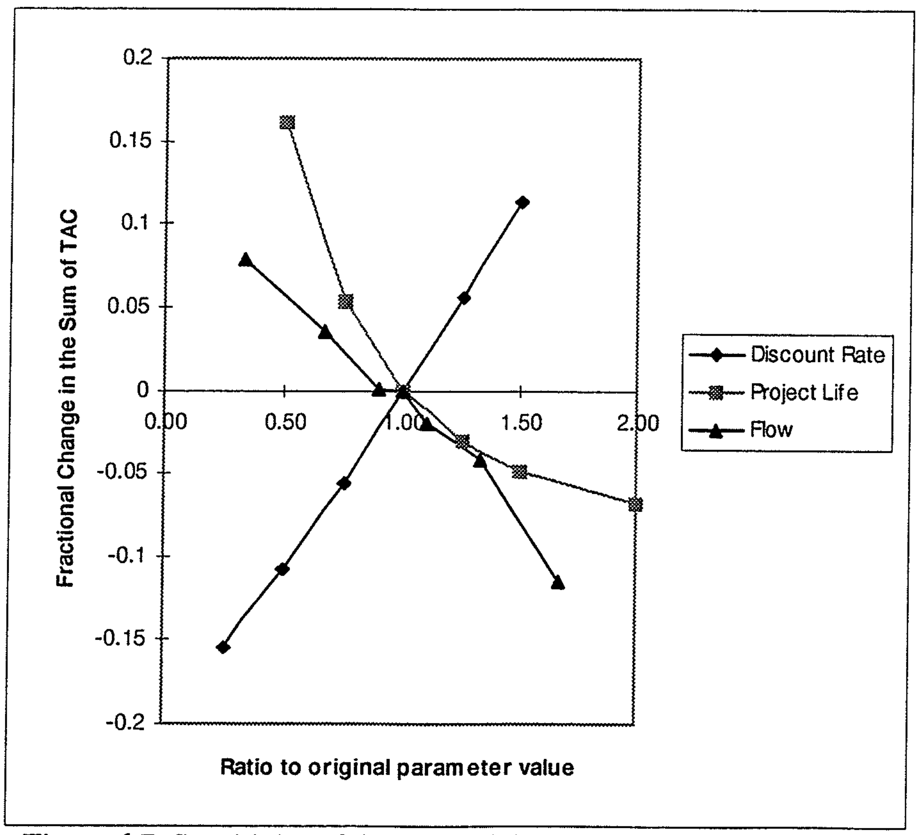

Figure 6.7 Sensitivity of the costs of the LC policy for meeting EU ambient ammonia and DO criteria at the mandatory level

Changes in the discount rate and the project life affect the WLA as well as the TAC. High discount rates or short project life favour wastewater treatment alternatives that have low investment costs, and vice versa. The treatment alternatives B and BC1 produce the same effluent quality for some of the dischargers. Treatment alternative B has high IC and low OMC while alternative BC1 has low IC and high OMC. With a discount rate below 12% or a project life longer than 10 years, the Székesfehérvár MWWTP upgrades to the treatment alternative B with high IC and low OMC for achieving the ambient ammonia criterion of 0.8 mg/l. With a discount rate of 12% or greater or a project life 10 years or shorter, the Székesfehérvár MWWTP upgrades instead to treatment alternative BC1, which has a lower IC, for meeting the same criterion.

6.3.4 Administrative Ease

Administrative ease is an important consideration for managers of water quality. A water quality management program that is difficult to implement or enforce may be unacceptable to the administrative body, regardless of its other attractive characteristics. Some measures of administrative ease are the information requirements, monitoring and enforcement efforts, and political acceptability.

The information requirements of the UEL and IEL policies are low compared to those of the LC policy since cost information is not needed to evaluate the UEL and IEL policies. The administrative body needs only to relate the emissions of pollutants to the desired ambient water quality to determine the effluent limits for the UEL and IEL programs. The need for cost information under the LC policy is a significant administrative burden because not only does the administrative body need to gather cost data, it must also guard against misrevelation of cost information from the dischargers. Appropriate discount rates and amortizing periods also need to be selected if project investment costs are to be amortized for comparison. The need for extensive data for the initiation of a program also means that extensive data are necessary in the future as adjustments of standards become necessary.

The need for monitoring and enforcement is common to all water quality management programs. The effort required to monitor dischargers is similar for all command and control policies requiring the installation of end-of-pipe wastewater treatment systems. However, the enforcement of emission limits may be complicated by the possibility of litigation by the dischargers. The potential for litigation is higher for the LC and IEL policies than for the UEL policies since the dischargers may not agree with the administrative body on the allocated waste loads, the information on which the WLAs are based, or the formula for WLA itself. Frequent and lengthy litigation are not only costly, they also reflect poorly on the efforts of water quality management and may compromise the effectiveness and authority of the administrative body.

Finally, the ease in implementing a water quality management policy depends in part on how acceptable the policy is to the public and to the dischargers. Equity is an important factor that affects the acceptability of different water quality management policies. Some groups may find the UEL policy most acceptable since it treats all polluters in the same way. Yet there is another view that polluters who are different should be treated differently. If efficiency in wastewater treatment is acceptable to the public and to the dischargers as a consideration in allocating waste loads, then the LC policy may be viewed as equitable since dischargers that are efficient in wastewater treatment engage in high levels of waste removal while those who are less efficient remove less. In CEE where financial resources are limited, the LC policy may be favoured since cost efficiency has a high priority. The IEL policy is difficult to justify to both the public and the dischargers because it is inequitable and may not be cost efficient; the IEL policy may be viewed as being arbitrary and having the worst characteristics of the UEL and LC policies. Acceptability of a policy may also be affected by the assurance of water quality standards being met. The UEL policy generally results in reserved stream assimilative capacity and may be viewed as a safer policy than the LC and IEL programs.

Chapter 7

Summary and Conclusions

Water quality management in CEE is a challenging problem because water quality is poor due to past environmentally incompatible practices and financial resources are limited in these transitional economies. The water quality management of the Veszprémi-Séd River and Malom and Nádor Channel System in Hungary is such a case. The Hungarian government is seeking to improve the water quality in this region but is concerned about adding to the financial burdens that the municipalities and industries already face.

The objective of this work is to identify water quality management strategies for the Veszprémi-Séd River and Malom and Nádor Channel System in Hungary. The principal water quality concerns in the system are the high levels of ammonia and BOD during spring and the low concentration of DO in the summer. Wastewater treatment alternatives and costs are generated for the dischargers in the system and the DSS DESERT is used to evaluate WLAs under three water quality management policies.

Only command and control policies are investigated in this work since incentive based policies tend to have high administrative costs compared to command and control policies and they may be unsuitable for unstable economies in transition. The characteristics of cost efficiency, equity, certainty of system outcome, and administrative ease of the UEL, LC, and IEL policies are examined.

The UEL policy is noted for its administrative simplicity and its equitable treatment of polluters. Since the WLAs of the UEL policy generally result in reserved stream assimilative capacity, the UEL policy is less sensitive to uncertainty and offers more assurance of meeting water quality criteria than the LC and IEL policies. However, the UEL policy is economically inefficient, a disadvantage that may be unacceptable in the transitional economies of CEE.

The main advantage of the LC policy is cost efficiency. However, the administrative burden of the LC policy is higher than those of the UEL and IEL policies due to the need for cost information and the LC policy is more sensitive to uncertainty than the UEL policy. The LC policy is often considered to be inequitable since dischargers are required to remove waste at different levels. However, another view of equity is that dischargers who are different should be treated differently. If cost efficiency is acceptable to the public and to the dischargers as a consideration in allocating waste loads, then the LC policy may be viewed as equitable. In CEE where financial resources are limited, the LC policy may be favoured since cost efficiency has a high priority.

The single advantage of the IEL policy is its low information requirement, similar to the UEL policy; however, unlike the UEL policy the IEL policy is not equitable.

Moreover the IEL policy may not be cost efficient since cost considerations are not included in the optimization as under the LC policy. Therefore the IEL policy may be viewed as being arbitrary and having the worst characteristics of the UEL and LC policies.

The determination of water quality goals and water quality management strategies may be aided by the use of mathematical models only if such models are adequately formulated based on sufficient and reliable information. For the water quality modelling of the Veszprémi-Séd River and Malom and Nádor Channel System, the lack of reliable data gives rise to difficulties in model calibration and in the selection of critical design conditions. Allocations of waste loads based on inadequate model parameters and design conditions may result in violations of water quality standards or in excessive commitments of financial resources to water quality improvement. In addition, the allocation of waste loads under the LC policy is affected by the accuracy of the cost data.

Besides illustrating the importance of reliable data, the analysis of the Veszprémi-Séd River and Malom and Nádor Channel System also accentuates the important distinction between nationally established and regionally achievable water quality criteria. The technological limits of wastewater treatment systems and the assimilative capacity of individual river basins need to be considered in establishing realistic water quality criteria.

Finally, the difficulty in improving the water quality of the Veszprémi-Séd River and Malom and Nádor Channel System points to the need for the investigation of water quality management alternatives other than those examined in this work. The LC policy, shown to be most cost efficient in the analysis presented in this work, is only least costly when compared with other policies that require the installation of end-of-pipe treatment systems by individual dischargers. Other management alternatives such as regional-based wastewater treatment and instream aeration may be more cost efficient than the LC policy. Further benefits may also be obtained by instituting incentive based policies, such as the ECH policy, which encourage dischargers to behave in a cost efficient manner and promote innovation in waste reduction. The collection of data on the dischargers and the estimation of wastewater treatment costs conducted for the evaluation of the LC policy may provide some of the groundwork necessary for the investigation of such alternatives.

The establishment of water quality criteria and the selection of water quality management approaches depend also on many other factors that are not examined in this work. Further work that may contribute to the effective water quality management of the Veszprémi-Séd River and Malom and Nádor Channel System includes:

(1) improving the collection of water quality data to provide reliable information for water quality modelling,

(2) examining the tradeoffs between the costs of wastewater treatment and the benefits of fish production in the region,

(3) including equity measures as constraints in the LC formulation to improve the equity characteristics of the LC policy,

(4) investigating the use of the unamortized capital cost instead of the TAC as the basis of comparison for the WLAs of the LC policy due to the limited financial resources in CEE countries,

(5) determining the phases and the standards to be used in the stepwise tightening of water quality standards, and

(6) investigating the use of incentive based policies as the transitional economies become more stable.

Since water quality management is a public-sector problem, the determining forces of water quality management programs may well be political and social considerations that cannot easily be incorporated into analyses similar to those presented or recommended in this work. The process of transition in CEE further adds to the complexity of the water quality management problem. The value of a systematic analysis of the water quality problems and alternatives is that it serves to clarify the problem and can provide insight for the managers of water quality as the transition proceeds.

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APPENDICES

APPENDIX A

Cost Estimates and Design Criteria
for Wastewater Treatment Alternatives

The design and cost estimation of wastewater treatment alternatives follow those of Somlyódy *et al.* (1994). The alternatives are:

(1) primary treatment, P,
(2) chemically enhanced primary treatment, CEPT,
(3) primary precipitation, PC,
(4) P and secondary treatment with high load activated sludge, B1,
(5) P and secondary treatment with low load activated sludge, B,
(6) CEPT and secondary treatment with low load activated sludge, BC1,
(7) PC and secondary treatment with low load activated sludge, BC2,
(8) BC1 and partial denitrification, BC1DN,
(9) BC2 and partial denitrification, BC2DN, and
(10)BC2 and denitrification, BCDN.

Table A.1 lists the alternatives, their efficiencies, and their unit cost estimations. Figure A.1 shows the effect of scale on the cost of conventional biological wastewater treatment. Table A.2 gives the design criteria for the alternatives.

Table A.1 Treatment efficiencies and costs

Technology	BOD % removal (conc., mg/l)	TSS % removal (conc., mg/l)	TP % removal (conc., mg/l)	TN % removal (conc., mg/l)	Unit IC in US$/$m^3$	Unit OMC in US$/$m^3$	Process Combination
P	30 (175)	60 (100)	15 (10)	15 (40)	0.90	0.064	Primary treatment
CEPT	55 (113)	80 (50)	75 (3)	25 (34)	0.98	0.103	Chemically enhanced primary treatment (low dosage)
PC	70 (75)	90 (25)	90 (1.2)	30 (34)	1.20	0.122	Primary precipitation (high dosage)
B	90 (25)	90 (25)	30 (8.4)	30 (34)	1.65	0.106	P + low load activated sludge, ASP
BC1	90 (25)	90 (25)	90 (1.2)	35 (31)	1.54	0.143	CEPT + B(ASP)
BC2	95 (12)	95 (12)	95 (0.6)	35 (31)	1.73	0.165	PC + B(ASP)
BC1DN	95 (12)	90 (25)	90 (1.2)	60 (19)	1.92	0.168	CEPT + B(ASP) + partial denitrification
BC2DN	95 (12)	95 (12)	95 (0.6)	60 (19)	2.11	0.200	PC + B(ASP) + partial denitrification
BCDN	97 (7)	95 (12)	95 (0.6)	85 (7)	2.37	0.210	PC + B(ASP) + denitrification

Note: Based on a plant capacity of 100 000 P.E. and wastewater production of 400 l/cap/day
Influent wastewater characteristics: BOD_5 = TSS = 250 mg/l, TP = 12 mg/l, TN = 48 mg/l

Source: Somlyódy *et al.* (1994)

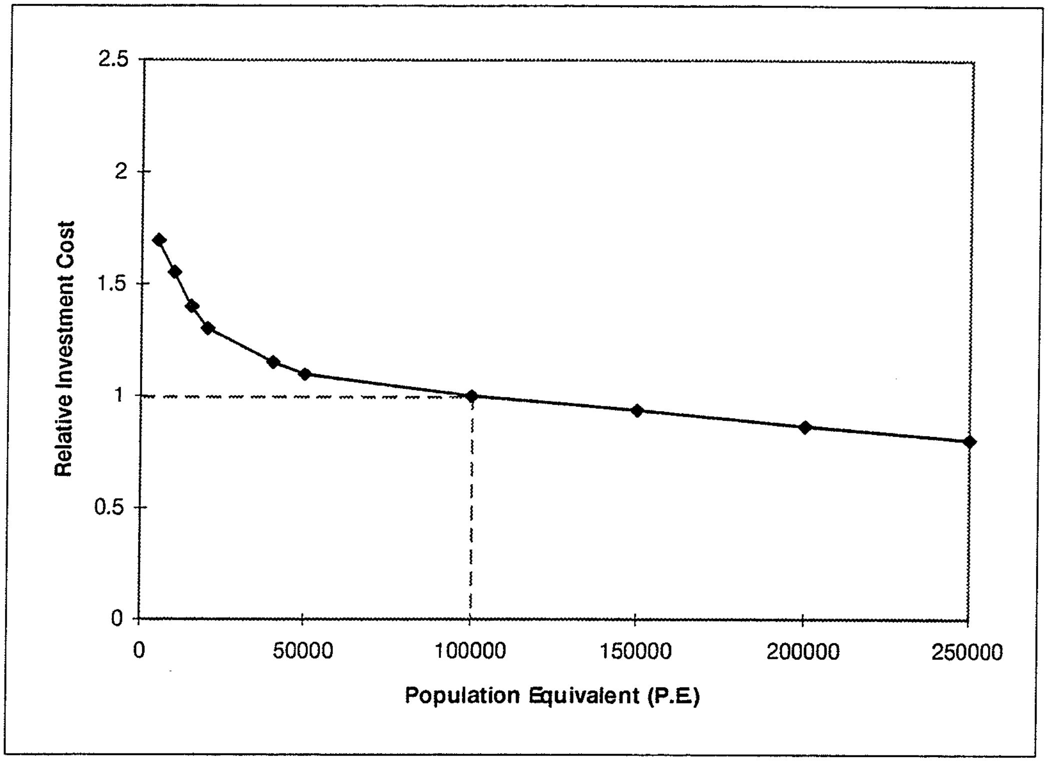

Figure A.1 Effect of scale on the cost of biological wastewater treatment
Redrawn after Somlyódy *et al.* (1994)

Table A.2 Design criteria

Technology	Processes and Design Criteria
P	Pretreatment (screening and grit removal) Settling: overflow rate = 2 m/hr
CEPT	Pretreatment (screening and grit removal) Chemical addition: low dosage of main coagulant ≤ 50 mg/l iron chloride; polymer dosage for coagulation/flocculation ≤ 1 mg/l Settling: overflow rate = 3 m/hr
PC	Pretreatment (screening and grit removal) Chemical addition: high dosage of main coagulant = 150 mg/l iron chloride; no polymer Mechanical flocculation Settling: overflow rate = 1.5 m/hr
B1	Pretreatment (screening and grit removal) Activated sludge process: F/M = 0.5 kg BOD_5/kg SS/day; organic sludge load in aeration tank = 0.5 kg BOD_7/kg SS/day Settling: overflow rate = 1 m/hr
B	Pretreatment (screening and grit removal); pre-settling Activated sludge process: F/M = 0.5 kg BOD_5/kg SS/day; organic sludge load in aeration tank = 0.2 kg BOD_7/kg SS/day Settling: overflow rate = 1 m/hr
BC1	Pretreatment (screening and grit removal) Activated sludge process with coagulant addition (simultaneous precipitation): organic sludge load in aeration tank = 0.2 kg BOD_7/kg SS/day Settling: overflow rate = 1 m/hr
BC2	Pretreatment (screening and grit removal) Coagulant addition; flocculation settling (pre-precipitation): overflow rate in pre-settling tank = 1.5 m/hr Activated sludge process: organic sludge load in aeration tank = 0.2 kg BOD_7/kg SS/day Post-settling: overflow rate in post-settling tank = 1 m/hr
BCDN1	Pretreatment (screening and grit removal) Activated sludge process designed for nitrification including predenitrification zone (recirculation system): total sludge age in activated sludge tank = 13 days; residence time in activated sludge tank = 15 hr Coagulant addition (simultaneous precipitation) Final settling: overflow rate = 1 m/hr

Source: Somlyódy *et al.* (1994)

APPENDIX B

Wastewater Treatment Facilities of Dischargers

Veszprém MWWTP

Information on the existing wastewater treatment plant:

Existing facility includes pretreatment, primary settling, activated sludge treatment, secondary settling, and disinfection:

Screens and sand trap

Primary settling tank: ∅ 22 m; volume = 800 m^3

Aeration tanks: 2×900 m^3

Secondary settling tank: ∅ 40 m; volume = 2200 m^3

This plant has a high load activated sludge system. Treatment efficiencies are 83% COD removal, 20% nitrogen removal, and 15% phosphorous removal.

Influent characteristics:

Flow = 15 000 m^3/d

Influent water quality data are unavailable; therefore typical influent water quality characteristics in CEE, given by Somlyódy *et al.* (1994), are assumed:

BOD_5 = 250 mg/l

Total nitrogen (TN) = 48 mg/l

Total phosphorous (TP) = 12 mg/l

TSS = 250 mg/l

Balatonfüred MWWTP

Information on the existing wastewater treatment plant:

Existing facility includes pretreatment, primary settling, activated sludge treatment, secondary settling, and disinfection:

Screens and sand trap

Primary settling tank: ∅ 26 m; volume = 600 m^3; surface area = 530 m^2 (estimated)

Aeration tanks: 2×200 m^3
2×500 m^3

Secondary settling tanks: 2 of ∅ 26 m; volume = 2×600 m^3; surface area = 2×530 m^2 (estimated)

This plant has a high load activated sludge system. Treatment efficiencies are not given. This plant has excess capacity during the non-tourist season and some units are not used. Since more units are used during the tourist season than during the non-tourist season to provide wastewater treatment, the effluent water quality is assumed to be the same for the tourist and non-tourist seasons.

Influent characteristics:

Flow: summer average = 3 200 m^3/d
yearly average = 2 900 m^3/d

Influent water quality data are unavailable; therefore typical influent water quality characteristics in CEE, given by Somlyódy *et al.* (1994), are assumed:

BOD_5 = 250 mg/l
Total nitrogen (TN) = 48 mg/l
Total phosphorous (TP) = 12 mg/l
TSS = 250 mg/l

Cost estimation:

The investment costs are estimated based on the mean wastewater inflow in the summer tourist season because the plant needs be able to handle the high wastewater inflow in the summer tourist season. Treatment units can be left idle during the non-tourist season, consistent with the current operation of Balatonfüred MWWTP.

The annual OMC is estimated as follows:

OMC = (OMC calculated based on the inflow of the tourist season)×(3/12)
+ (OMC calculated based on the inflow of the non-tourist season)×(9/12)

Balatonfüzfö MWWTP

Information on the existing wastewater treatment plant:

The existing facility provides pretreatment, activated sludge treatment, post-aeration settling, and disinfection. Secondary biological treatment, aeration (activated sludge) and settling, takes place in circular unified tanks. Aeration takes place in the outer ring and post-aeration settling occurs in the centre of the tank. Under the current practice, primary settling is only performed during the tourist season by using two of the unified tanks as primary settling tanks. The existing wastewater treatment units include:

Screens and sand trap

Unified tanks: aerating volume = 6×2000 m^3

settling volume = 6×400 m^3; (estimated settling surface = 6×400 m^2)

This plant operates as a high load activated sludge system. Treatment efficiencies are not given. This plant has excess capacity during the non-tourist season and some units are not used. Since more units are used during the tourist season than during the non-tourist season to provide wastewater treatment, the effluent water quality is assumed to be the same for the tourist and non-tourist seasons.

Influent characteristics:

Flow: summer average = 7 200 m^3/d

yearly average = 4 800 m^3/d

Influent water quality data are unavailable; therefore typical influent water quality characteristics in CEE, given by Somlyódy *et al.* (1994), are assumed:

BOD_5 = 250 mg/l

Total nitrogen (TN) = 48 mg/l

Total phosphorous (TP) = 12 mg/l

TSS = 250 mg/l

Cost estimation:

The investment costs are estimated based on the mean wastewater inflow in the summer tourist season because the plant needs be able to handle the high wastewater inflow in the summer tourist season. Treatment units can be left idle during the non-tourist season, consistent with the current operation of Balatonfüred MWWTP.

The annual OMC is estimated as follows:

OMC = (OMC calculated based on the inflow of the tourist season)×(3/12)
+ (OMC calculated based on the inflow of the non-tourist season)×(9/12)

Várpalota MWWTP

Information on the existing wastewater treatment plant:

The existing facility provides pretreatment and primary settling and uses facultative ponds for biological treatment:

Screens and sand trap

Primary settling tanks: settling volume = 3×70 m^3

Facultative pond: 2 cells separated by an earthen weir

Cell 1: area = 330 000 m^2
volume = 370 000 m^3
average depth = 1.1 m

Cell 2: area = 362 000 m^2
volume = 430 000 m^3
average depth = 1.2 m

Influent characteristics:

Flow = 3 800 m^3/d

Influent water quality data are unavailable; therefore typical influent water quality characteristics in CEE, given by Somlyódy *et al.* (1994), are assumed:

BOD_5 = 250 mg/l
Total nitrogen (TN) = 48 mg/l
Total phosphorous (TP) = 12 mg/l
TSS = 250 mg/l

Cost estimation:

The costs of facility upgrades are based on biological wastewater treatment using activated sludge systems. The use of the facultative pond is assumed to be discontinued.

Székesfehérvár MWWTP

Information on the existing wastewater treatment plant:

Existing facility includes pretreatment, primary settling, activated sludge treatment, and secondary settling:

Screens and aerated grit chambers
Primary settling tanks: ∅ 20 m; area = 310 m^2; volume = 460 m^3
∅ 22 m; area = 380 m^2; volume = 740 m^3
Aeration tanks: 5×300 m^3
4×1350 m^3
Secondary settling tanks: 1 of ∅ 34 m; area = 710 m^2; volume = 2000 m^3
2 of ∅ 32 m; area = 2×630 m^2;
volume = 2×2150 m^3

This plant has a high load activated sludge system. Treatment efficiencies are not given.

Influent characteristics:

Flow = 30 000 m^3/d

Influent water quality data are available for the water quality constituents of BOD_5 and TSS. The effluent concentration of total nitrogen assumes the value given by Somlyódy *et al.* (1994):

BOD_5 = 520 mg/l
Total nitrogen (TN) = 48 mg/l
TSS = 350 mg/l

Nitrokémia Rt. Plants I-IV

Information on the existing wastewater treatment facility:

The existing facility provides industrial pretreatment, primary settling, activated sludge treatment, and secondary settling. The wastewater is pretreated using the processes of neutralization (lime addition), flocculation, and settling. Primary settling, activated sludge treatment, and secondary settling are then performed. Secondary biological treatment, aeration and clarification, takes place in unified tanks. The effluent of secondary clarification passes through an emergency reservoir before it is discharged into the Veszprémi-Séd River. The purpose of the emergency reservoir is to contain the effluent of secondary clarification should there be a failure in the wastewater treatment system. Under normal operation of the plant, the emergency reservoir acts as a polishing pond. The existing wastewater treatment units include:

Neutralization tanks: 3
Flocculating basins: 2
Gypsum settling tanks: 4
Primary settling tanks: 2×300 m^3
Biological units: total volume = 4×2600 m^3
estimated aeration volume = 4×2200 m^3
estimated settling volume = 4×400 m^3
(assume the same ratio of aeration volume to settling volume as the unified units at Balatonfüzfö MWWTP)
Additional secondary settling pond: area = 19.6 ha
Emergency reservoir: net volume = 290 000 m^3 if the sludge is not dredged; area = 210 000 m^2

The biological treatment system is a high load activated sludge system. Treatment efficiencies are 92% COD removal and 37% ammonia removal.

Influent characteristics:

Flow = 17 300 m^3/d

Influent water quality data are unavailable. Using the effluent water quality data collected by the Regional Environmental Inspectorate and the treatment efficiencies given by Nitrokémia Rt., the influent water quality is calculated to be:

Influent ammonia concentration = 35 mg/l as nitrogen. Assuming that ammonia makes up 60% of TN (USEPA Technology Transfer, 1975, Process Design Manual for Nitrogen Control), TN = 58 mg/l

BOD_5 = 550 mg/l (assuming that the removal efficiency is the same for COD and BOD)

Cost estimation:

The current level of industrial wastewater pretreatment is assumed to be adequate without any upgrades. Only the costs of upgrades to the primary, secondary, and tertiary treatment facilities are estimated. The contribution to wastewater treatment of the emergency reservoir is not considered in the design of the upgrades.

Nitrogén Müvek Rt.

Information on the existing wastewater treatment facility:

The existing wastewater treatment facility is a stabilization pond for biological treatment of wastewater. The pond is divided into four cells and wastewater flows through these cells by gravity. The volume, surface area, and average depth of the pond are 400 000 m^3, 520 000 m^2, and 0.8 m, respectively. Ammonia removal is 60%.

Influent characteristics:

Flow = 18 200 m^3/d

Influent ammonia concentration = 47 mg/l as nitrogen. Assuming that ammonia makes up 60% of TN (USEPA Technology Transfer, 1975, Process Design Manual for Nitrogen Control), TN = 78 mg/l.

Neither influent nor effluent BOD_5 data are available. Since Nitrogén Müvek Rt. is a manufacturer of fertilizer products, and given the effluent COD concentration of 64 mg/l and the method of treatment, the influent BOD_5 concentration is estimated to be 50 mg/l (Fehér, 1995a; Masliev, 1995b).

Cost estimation:

The costs of facility upgrades are based on biological wastewater treatment using activated sludge systems. Since Nitrogén Müvek Rt. currently uses a stabilization pond for wastewater treatment, the costs of the its wastewater treatment alternatives are the costs of the construction of new plants as though there is no existing wastewater treatment facility.

Peremarton Chemical Works

Information on the existing wastewater treatment facility:

The existing wastewater treatment plant removes wastes from three waste streams: municipal wastewater, industrial wastewater, and run-off in the territory of Peremarton Chemical Works. Figure B.1 is a simplistic flow diagram of the wastewater treatment facility.

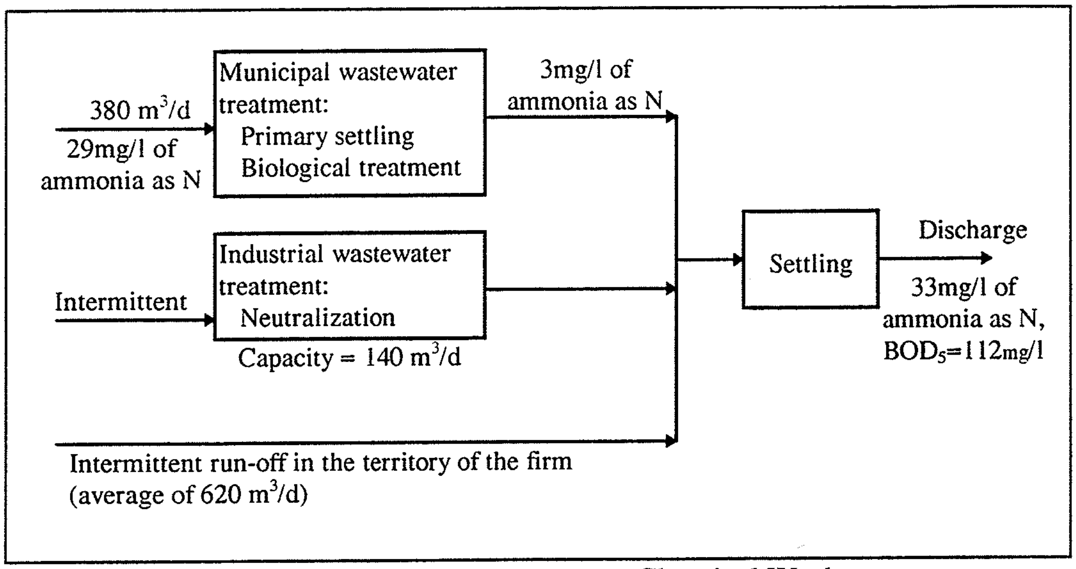

Figure B.1 Wastewater treatment at Peremarton Chemical Works

The existing wastewater treatment units include:

Screens
Primary settling tanks: 2, estimated total surface area = 10 m^2
Trickling filters (biological treatment): 2
Neutralization tanks: 2, total capacity of 140 m^3/d
Secondary settling tanks: 6, total volume = 2860 m^3

Generation of upgrade alternatives and costs

Ammonia concentration in the effluent of the firm is 33 mg/l while the treated municipal wastewater (before final settling) has an ammonia concentration of 3 mg/l. This suggests that the ammonia concentration of the industrial wastewater and/or the run-off is high. In this work, the industrial wastewater is assumed to contain high concentrations of ammonia; therefore the industrial wastewater is included as wastewater inflow for the generation of wastewater treatment alternatives and costs. The industrial pretreatment performed is assumed to be adequate without any upgrades to render the industrial wastewater suitable for biological wastewater treatment. The run-off collected is assumed to require settling only, consistent with the current wastewater treatment scheme.

The design inflow for the wastewater treatment facility is the sum of the municipal wastewater flow (380 m^3/d) and the capacity of the industrial wastewater treatment system (140 m^3/d). The influent flow is then 520 m^3/d.

From Figure B.1, the flow of municipal wastewater is 380 m^3/d, the ammonia concentration of the effluent of the trickling filter is 3 mg/l, and the effluent ammonia concentration of the firm = 33 mg/l. Assuming that the flow of industrial wastewater is 140 m^3/d, using a simple mass balance, the ammonia concentration of the industrial wastewater is calculated to be 114 mg/l. Given that the influent to the municipal wastewater treatment system has an ammonia concentration of 29 mg/l, the mixed inflow of the municipal and industrial wastewater has an ammonia concentration of 52 mg/l.

The effluent BOD_5 of the firm is 113 mg/l. Data of BOD_5 concentrations at other points of the treatment system are not available. An influent BOD_5 concentration of 250 mg/l is assumed.

The costs of facility upgrades are based on biological wastewater treatment using activated sludge systems. The use of trickling filters is assumed to be discontinued.

ALBA Textil Kft.

Information on the existing wastewater treatment plant:

> The existing wastewater treatment facility performs only primary treatment. The settling tank has two cells and the total net volume of the tank is 120 m^3. The dimensions of the tank are 13.5 × 13.5 × 0.75 m. The treatment efficiency is 80% for the removal of BOD and TSS.

Influent characteristics:

> Flow = 2 000 m^3/d
>
> Influent water quality data are unavailable. Using the effluent water quality data collected by the Regional Environmental Inspectorate and the treatment efficiencies given by ALBA Textil Kft., the influent water quality is calculated to be:
>
> BOD_5 = 365 mg/l
> TSS = 435 mg/l
>
> The treatment efficiency for ammonia removal is not available. The effluent of ALBA Textil Kft., which is a textile plant, contains only 0.7 mg/l of ammonia. Assuming a typical ammonia removal efficiency of 15% for primary treatment, the influent ammonia concentration is 1.2 mg/l (Somlyódy *et al.*, 1994). The wastewaters of textile industries generally contain low levels of ammonia (Sittig, M. 1975. *Environmental Sources and Emissions Handbook*, Noyes Data Corp., Park Ridge, New Jersey).

APPENDIX C

Waste Load Allocations for the Management of
Ammonia,
Ammonia and BOD, and
DO

Ammonia Management

WLAs[1] for **ammonia** management under the **UEL** Policy

Discharger	Ambient ammonia criteria[2] (mg/l as nitrogen)			
	3.05[3]	0.8[4]	0.72[5]	0.5[6]
Veszprém MWWTP	B	BC1DN	BC1DN	BCDN
Nitrokémia Rt. Plants I-IV	B	BC1DN	BC1DN	BCDN
Nitrogén Müvek Rt.	-	BC1DN	BC1DN	BCDN
Balatonfüred MWWTP	B	BC1DN	BC1DN	BCDN
Balatonfüzfö MWWTP	B	BC1DN	BC1DN	BCDN
Peremarton Chemical Works	BC1	BC1DN	BC1DN	BCDN
Várpalota MWWTP	-	BC1DN	BC1DN	BCDN
Székesfehérvár MWWTP	B	BC1DN	BC1DN	BCDN
ALBA Textil Kft.	-	-	BC1DN	BCDN
Minimum UEL in mg/l ammonia as N	8.0	1.0	0.5	0.3
ΔTAC in 10^6 US$	2.6	7.2	7.4	10.3

1 P: primary treatment
CEPT: chemically enhanced primary treatment
PC: primary precipitation
B1: P and secondary treatment with high load activated sludge
B: P and secondary treatment with low load activated sludge
BC1: CEPT and secondary treatment with low load activated sludge
BC2: PC and secondary treatment with low load activated sludge
BC1DN: BC1 and partial denitrification
BC2DN: BC2 and partial denitrification
BCDN: BC2 and denitrification

2 Only the ambient ammonia criteria that are achievable are listed

3 Equivalent ambient ammonia criterion of the USEPA ambient unionized ammonia criterion

4 EU ambient ammonia criterion at the mandatory level

5 Equivalent ambient ammonia criterion of the Hungarian and EU ambient unionized ammonia criteria at the lower level, 0.021 mg/l unionized ammonia as N

6 Hungarian ambient ammonia criterion at the tolerable level

WLAs for **ammonia** management under the **LC** Policy

Discharger	Ambient ammonia criteria (mg/l as nitrogen)			
	3.05	0.8	0.72	0.5
Veszprém MWWTP	-	BC1DN	BC1DN	BCDN
Nitrokémia Rt. Plants I-IV	B	B	B	BCDN
Nitrogén Müvek Rt.	-	BC1DN	BC1DN	BCDN
Balatonfüred MWWTP	-	BCDN	B	BCDN
Balatonfüzfö MWWTP	-	BC1DN	B	BCDN
Peremarton Chemical Works	-	BC1DN	BCDN	BCDN
Várpalota MWWTP	-	-	-	BCDN
Székesfehérvár MWWTP	B	B	BC1DN	BCDN
ALBA Textil Kft.	-	-	-	BCDN
ΔTAC in 10^6 US$	2.3	4.8	5.9	10.3

WLAs for **ammonia** management under the **IEL** Policy

Discharger	Ambient ammonia criteria (mg/l as nitrogen)			
	3.05	0.8	0.72	0.5
Veszprém MWWTP	B	BC1DN	BC1DN	BCDN
Nitrokémia Rt. Plants I-IV	-	B	B	BCDN
Nitrogén Müvek Rt.	B	BCDN	B	BCDN
Balatonfüred MWWTP	-	BC1DN	BCDN	BCDN
Balatonfüzfö MWWTP	BC1DN	BCDN	BCDN	BCDN
Peremarton Chemical Works	BCDN	BC1DN	BCDN	BCDN
Várpalota MWWTP	-	-	BCDN	BCDN
Székesfehérvár MWWTP	BCDN	B	BC1DN	BCDN
ALBA Textil Kft.	-	BC1DN	-	BCDN
ΔTAC in 10^6 US$	5.3	5.6	6.5	10.3

Ammonia and BOD Management

WLAs[1] for **ammonia** and **BOD** management under the **UEL** Policy

Discharger	Category of Ambient Criteria[2]			
	Least stringent combination[3]	EU Mandatory, ammonia only[4]	EU Mandatory[5]	Hungarian Tolerable[6]
Veszprém MWWTP	B	BC1DN	BC1DN	BCDN
Nitrokémia Rt. Plants I-IV	BC2	BC1DN	BC1DN	BCDN
Nitrogén Müvek Rt.	-	BC1DN	BC1DN	BCDN
Balatonfüred MWWTP	BC2	BC1DN	BC1DN	BCDN
Balatonfüzfö MWWTP	BC2	BC1DN	BC1DN	BCDN
Peremarton Chemical Works	BC1	BC1DN	BC1DN	BCDN
Várpalota MWWTP	-	BC1DN	BC1DN	BCDN
Székesfehérvár MWWTP	BCDN	BCDN	BCDN	BCDN
ALBA Textil Kft.	BC1DN	BC1DN	BC1DN	BCDN
Minimum UELs: mg/l ammonia as N (mg/l BOD_5)	8.0 (35)	1.0 (35)	0.5 (35)	0.3 (35)
ΔTAC in 10^6 US$	5.9	8.5	8.5	10.3

[1] P: primary treatment
CEPT: chemically enhanced primary treatment
PC: primary precipitation
B1: P and secondary treatment with high load activated sludge
B: P and secondary treatment with low load activated sludge
BC1: CEPT and secondary treatment with low load activated sludge
BC2: PC and secondary treatment with low load activated sludge
BC1DN: BC1 and partial denitrification
BC2DN: BC2 and partial denitrification
BCDN: BC2 and denitrification

[2] Only the combinations of ambient criteria that are achievable are listed

[3] USEPA ambient unionized ammonia criterion and Hungarian ambient BOD_5 criterion at the tolerable level (12 mg/l)

[4] EU ambient ammonia criterion at the mandatory level (0.8 mg/l as N) and the ambient BOD_5 criterion of 12 mg/l

[5] EU ambient ammonia criterion at the mandatory level (0.8 mg/l as N), EU ambient unionized ammonia criterion at the mandatory level (0.021 mg/l as N), and the ambient BOD_5 criterion of 12 mg/l

[6] Hungarian ambient ammonia criterion at the tolerable level (0.5 mg/l as N), Hungarian ambient unionized ammonia criterion at the tolerable level (0.021 mg/l as N), and Hungarian ambient BOD_5 criterion at the tolerable level (12 mg/l)

WLAs for **ammonia** and **BOD** management under the **LC** Policy

Discharger	Category of Ambient Criteria			
	Least stringent combination	EU Mandatory, ammonia only	EU Mandatory	Hungarian Tolerable
Veszprém MWWTP	B	BC1DN	BC1DN	BCDN
Nitrokémia Rt. Plants I-IV	B	B	BC2	BCDN
Nitrogén Müvek Rt.	-	B	BC1DN	BCDN
Balatonfüred MWWTP	-	BC2	BC2	BCDN
Balatonfüzfö MWWTP	-	BC1DN	BC1DN	BCDN
Peremarton Chemical Works	-	BC2	BC1DN	BCDN
Várpalota MWWTP	-	-	-	BCDN
Székesfehérvár MWWTP	BCDN	BCDN	BCDN	BCDN
ALBA Textil Kft.	-	-	-	BCDN
ΔTAC in 10^6 US$	4.7	6.8	7.6	10.3

WLAs for **ammonia** and **BOD** management under the **IEL** Policy

Discharger	Category of Ambient Criteria			
	Least stringent combination	EU Mandatory, ammonia only	EU Mandatory	Hungarian Tolerable
Veszprém MWWTP	B	BC1DN	BC1DN	BCDN
Nitrokémia Rt. Plants I-IV	-	BC1DN	BC1DN	BCDN
Nitrogén Müvek Rt.	BCDN	BCDN	B	BCDN
Balatonfüred MWWTP	BC1DN	-	BC2	BCDN
Balatonfüzfö MWWTP	BC2	BC1DN	BC1DN	BCDN
Peremarton Chemical Works	-	BC1DN	BC1	BCDN
Várpalota MWWTP	-	-	-	BCDN
Székesfehérvár MWWTP	BCDN	BCDN	BCDN	BCDN
ALBA Textil Kft.	-	BC1DN	BCDN	BCDN
ΔTAC in 10^6 US$	6.0	6.8	7.6	10.3

DO Management

WLAs[1] for **DO** management under the UEL, LC, and IEL policies

Discharger	UEL Policy			LC Policy			IEL Policy		
	Ambient DO Criteria[2] (mg/l)			Ambient DO Criteria[2] (mg/l)			Ambient DO Criteria[2] (mg/l)		
	4[3]	6[4]	7[5]	4[3]	6[4]	7[5]	4[3]	6[4]	7[5]
Veszprém MWWTP	-	-	-	-	-	-	-	-	-
Nitrokémia Rt. Plants I-IV	BC2	BC2	BCDN	-	BCDN	BC2	-	BCDN	BC2
Nitrogén Müvek Rt.	-	-	B	-	B	B	BC1DN	BC2	BC2
Balatonfüred MWWTP	-	BC2	BC2	BCDN	BC2	-	-	BCDN	BCDN
Balatonfüzfö MWWTP	-	-	BC2	-	BCDN	BC2	-	BC2	-
Peremarton Chemical Works	BC1	BC1	BC1	BC1	BC1DN	BC2	-	BC2	BCDN
Várpalota MWWTP	-	-	-	-	-	-	-	-	-
Székesfehérvár MWWTP	-	BCDN	BCDN	-	-	BCDN	-	-	BCDN
ALBA Textil Kft.	BC1DN	BC1DN	BC1DN	BC1DN	BCDN	BC1DN	-	BCDN	BCDN
Minimum UELs: mg/l ammonia as N (mg/l BOD_5)	30 (40)	13 (30)	5 (25)						
ΔTAC in 10^6 US$	1.4	5.6	7.6	0.6	3.8	6.4	1.4	4.1	6.9

[1] P: primary treatment
CEPT: chemically enhanced primary treatment
PC: primary precipitation
B1: P and secondary treatment with high load activated sludge
B: P and secondary treatment with low load activated sludge
BC1: CEPT and secondary treatment with low load activated sludge
BC2: PC and secondary treatment with low load activated sludge
BC1DN: BC1 and partial denitrification
BC2DN: BC2 and partial denitrification
BCDN: BC2 and denitrification

[2] Only the ambient DO criteria that are achievable are listed
[3] Hungarian ambient DO criterion at the tolerable level
[4] Hungarian ambient DO criterion at the objective level
[5] EU ambient DO criterion at the mandatory level

APPENDIX D

Notation

B = primary treatment and secondary treatment with low load activated sludge

B1 = primary treatment and secondary treatment with high load activated sludge

BAT = best available technology

BC1 = chemically enhanced primary treatment and secondary treatment with low load activated sludge

BC1DN= chemically enhanced primary treatment, secondary treatment with low load activated sludge, and partial denitrification

BC2DN= primary precipitation, secondary treatment with low load activated sludge, and partial denitrification

BC2 = primary precipitation and secondary treatment with low load activated sludge

BCDN = primary precipitation, secondary treatment with low load activated sludge, and denitrification

BOD = biochemical oxygen demand

BOD_5 = 5-day biochemical oxygen demand

c = concentration of dissolved oxygen (mg/l)

CBOD = carbonaceous biochemical oxygen demand

CEE = Central and Eastern Europe

CEPT = chemically enhanced primary treatment

$C_n(x_n)$ = cost function associated with the wastewater treatment alternatives at the n^{th} reach in the water quality management optimization model

COD = chemical oxygen demand

c_s = saturation concentration of dissolved oxygen (mg/l)

DO = dissolved oxygen

DP = dynamic programming

DSS = decision support system

ECH = effluent charge

EU = European Union

f' = fraction of unionized ammonia

$f_n(Q_n)$ = recursion equation representing the value of the objective function at the n^{th} reach in the water quality management optimization model

GDP = gross domestic product

H = average depth (ft)

IC = investment costs

IEL = individual effluent limit

IIASA = International Institute for Applied Systems Analysis

K_a = rate of atmospheric reaeration (/day)

K_d = decay rate of biochemical oxygen demand (/day)

K_n = removal rate of ammonia (/day)

K_r = removal rate of biochemical oxygen demand (/day)

K_s = loss rate of biochemical oxygen demand due to settling (/day)

K_T = rate at temperature T (/day)

L = concentration of 5-day biochemical oxygen demand (mg/l)

L_5 = concentration of 5-day carbonaceous biochemical oxygen demand (mg/l)

LC = least cost

LP = linear programming

L_u = ultimate carbonaceous biochemical oxygen demand (mg/l)

$M_n(x_n,Q_{n-1})$ = transformation of water quality states using the water quality simulation model given the wastewater treatment alternative x_n and the water quality states of stage n-1, in the water quality management optimization model

MWWTP= municipal wastewater treatment plant

N = concentration of ammonia (mg/l as nitrogen)

n = stage or reach number in the water quality management optimization model

NBOD = nitrogenous biochemical oxygen demand

NKTK = National Environment and Nature Conservation Concept (Hungary)

NLP = nonlinear programming

OMC = operation and maintenance costs

P = primary treatment

PC = primary precipitation

PHARE= Poland and Hungary: Assistance to Restructure the Economy

pka = equilibrium constant for the partitioning of ammonia into ionized and unionized fractions

$P_n(x_n)$ = mass load of the pollutant discharged as a result of the decision at stage n in the water quality management optimization model

Q_n = water quality at the downstream end of the n^{th} reach in the water quality management optimization model

S_n = water quality constraint to be met at the downstream end of the n^{th} reach in the water quality management optimization model

T = water temperature (°C)

t = time (time step in the water quality simulation model, day)

TAC = total annual costs

TDP = transferable discharge permit

TSS = total suspended solids

U = velocity (fps)

UEL = uniform effluent limit

USEPA= United States Environmental Protection Agency

UT = uniform treatment

v = velocity (m/day)

VITUKI= Water Resources Research Center (Hungary)

WLA = waste load allocation

x = distance (distance step in the water quality simulation model, m)

X_n = set of possible wastewater treatment alternatives at the n^{th} reach in the water quality management optimization model

x_n = the decision, or the wastewater treatment alternative at the n^{th} reach in the water quality management optimization model

ZUT = zoned uniform treatment

ΔOMC= change in the operation and maintenance costs from the current level

ΔTAC = change in the total annual costs from the current level

θ = temperature coefficient for the calculation of reaction rates