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Kularathna, M. and Somlyody, L.

IIASA Working Paper

WP-94-003

January 1994

Working Paper

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Maddumage Kularathna
László Somlyódy

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

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Preface

Water quality management of highly polluted rivers in Central and Eastern Europe countries is one of the major concerns of IIASA's water resources project (WAT). A comprehensive decision support system (DSS) is being developed to aid the associated decision process.

This paper provides a state-of-the-art review of management models which is lacking despite the many literature available on individual studies. It also aims at selecting an appropriate generic methodology which can be used for the WAT's DSS on degraded river basin management.

Abstract

With the increasing human activities within river basins, the problem of water quality management is becoming increasingly important. Quality management can be achieved through control/prevention measures that have various economic and water quality implications. To facilitate the analysis of available management options, decision models are needed which represent the many facets of the problem. Such models must be capable of adequately depicting the hydrological, chemical and biological processes occurring in the river; while incorporating social, economic and political considerations within the decision framework.

Management analyses can be performed using simulation, optimization, or both, depending on the management goal and the size and type of the problem. The critical issues in a management model are the nonlinearities, uncertainties, multiple pollutant nature of waste discharges, multiple objectives, and the spatial and temporal distribution of management actions.

Literature on various management models were reviewed under the headings of linear, nonlinear and dynamic programming approaches; their stochastic counterparts, and combined or miscellaneous approaches. Dynamic programming was found to be an attractive methodology which can exploit the sequential decision problem pertaining to river basin water quality problems (downstream control actions do not influence water quality upstream). DP handles discrete decision variables which represent discrete management alternatives, and it is generic in the sense that both linear and non-linear water quality models expressing the relation between emissions and ambient quality levels can be incorporated. An example problem is presented which demonstrates the application of a DP-based management model to formulate least-cost strategies for the Nitra River basin in Slovakia.

However, it is hardly possible for a single model to represent all the aspects of a complex decision problem. Different types of management models (eg. deterministic vs stochastic models) have different capabilities and limitations. The only way to compensate for the deficiencies is to perform the analysis in a sensitivity style. The necessity for sensitivity analyses is further implied due to the fact that water quality problems are rather loosely formulated with respect to the quality and economic goals.

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RIVER BASIN WATER QUALITY MANAGEMENT MODELS: A STATE-OF-THE-ART REVIEW

M. Kularathna and L. Somlyódy

1 INTRODUCTION

Water Quality Control and Legislation

Rivers are polluted by various human interactions, which can be broadly categorized into agriculture, industry, and settlements. Polluted water affect the ecosystems and various water uses, and risk human health. Industries and public would incur additional purification expenditures if they are dependent on the polluted river water. However those who pollute are usually not affected by that waste, and therefore have little or no incentive for controlling their discharge. Due to this reason, river water quality control is usually achieved through legislation, by specifying water quality standards which have to be maintained. Important elements of the legislation and its enforcement are fees, fines, taxes which provide incentives to control (prevent) pollution or penalize if its level exceeds the norms set.

Main Goals of Water Quality Control

The goals of water quality control are twofold. The first is to maintain water quality at desired levels (expressed by quality standards) corresponding to water uses, while the second is to achieve the first goal with a minimum cost or maximum benefit to the society concerned. The "concerned" society, and the regulatory legislation, depends on whether the river basin covers a region (in one country), several regions, or several countries. Whichever the case is, the concern of quality control should be to achieve an efficient and equitable allocation of resources considering the society as a whole. Most benefits of water quality management actions, and damages caused by poor quality water, are rather intangible, which is the reason for often expressing the economic goal of quality control in terms of an efficient allocation of resources (usually by minimizing the costs) only.

Quality Standards

Water quality standards are usually expressed in two ways: standards for effluent quality or those for rivers, lakes etc (ambient quality standards). Mixed standards which specify both effluent and ambient standards are also used. Effluent standards indicate the allowable water quality for each discharge, thus specifying the management action required for each of them. This makes the task of a controlling agency relatively simpler, although the corresponding management actions, as a whole, might be economically unjustifiable. Ambient standards, on the other hand, do not set limits on individual discharges, but on the collective effect of many upstream discharges on the standard locations. In this case, the controlling authority must estimate the effect of various charges, and determine ways to maintain quality standards with a minimum expenditure.

Multiple Objectives

Economic implications of setting water quality goals are substantial, because the cost of management alternatives increases exponentially with increasing water quality standards. However, the implications are not only economic, but also can be political, when considering the large expenditures and their spatial and/or temporal distribution. This imply the multiobjective character of a water quality management issue: higher quality standards, lowest and equitable distribution of costs, and political goals, forms the basic multiobjective problem. Even within the cost objective, trade-offs between investment cost (IC) and operation, maintenance and replacement cost (OMRC) have to be considered. Moreover, "water quality" itself is defined by a vector of quality indicators. Therefore, selection of desired quality indicator(s) needs a comparative assessment of different quality indicators and their importance, thus giving rise to an embedded multiobjective character.

Need for Management Models

Water quality management is possible through actions that prevent pollution, or, change the level of pollution. The latter include wastewater treatment, storing of effluents, wastewater disposal on land, artificial stream aeration, low flow augmentation, or various combinations of them. Controlling of non-point source pollutants is another aspect which is usually achieved indirectly. The alternative appropriate for a particular situation depends on many factors, among which the type of discharge is an important one. Selection of the alternative, however, is only one initial step of the management decision process. More decisions are subsequently needed with regard to efficiency, capacity, location, and scheduling of implementation etc. of the selected option. This decision process becomes complex, especially for river basins which have many discharge locations and a large number of feasible alternatives to be compared.

If only effluent standards are specified, the purpose of a management model would be to select the most economical management action for each discharge independently. Subsequent checking of the combined effect of such actions, specified for various discharges, may be necessary to verify that the river quality is at acceptable levels. Influent standards, on the other hand, require more complex models which allow river water quality computations within the decision process.

An example for a management problem can be given with regard to municipal wastewater treatment of a river basin with only 10 municipal discharge locations, each having 5 feasible treatment alternatives from which to select. This indicates the existence of 5^{10} feasible schemes (if the standards are specified for ambient water quality), considering a deterministic case where all the parameters and inputs are known with certainty. Many other scenarios will have to be examined to represent uncertainties in parameters and hydrologic events. Estimation of water quality resulting from each of them is impractical, thus expressing the need to employ a management model for screening the alternatives efficiently. However, evaluation of all feasible actions might be applicable for a problem with only a few discharge locations and alternatives.

The Role of Water Quality Models

River water quality at a particular location is dependent on the joint effect of upstream discharges, among other factors. A water quality model is needed to estimate this effect, thus it is an essential component of a management model (especially if ambient quality standards are specified). There are different types of water quality models which are associated with different levels of complexities and input requirements (see Section 3). They operate using various quality parameters and inputs which are inherently uncertain due to hydrologic and meteorologic events. In order to make a reliable management decision, such uncertainties have to be considered appropriately within the modelling framework.

The selection of an appropriate water quality model depends, among other factors, on the quality indicators that are relevant for a particular river system. In cases where the organic pollutants are the cause of river water quality problems, Streeter-Phelps (1925) equations have been widely used. They model the dissolved oxygen (DO), and, the concentration of organic pollutants expressed by biochemical oxygen demand (BOD). Streeter-Phepls (S-P) equations describe the self-purification occurring in the river as a first order chemical reaction between pollutants and DO. In reality, this self-purification has a more complex relationship. As an example, nutrients such as nitrogen or phosphorus that cause eutrophication cause such complexities.

Organization of the Paper

The objective of this paper is to review the state-of-the-art river water quality management models, in order to select a generic methodology that can be used for decision support purposes of IIASA's ongoing research related to degraded river basin management in the CEE region. The management alternative which is of main concern for this paper is wastewater treatment, which is widely used for municipal and industrial discharges alike. Section Two describes the main elements of a management model. A description of water quality models is provided in Section Three, while wastewater treatment technologies are covered briefly in Section Four. Section Five contains the state-of-the-art review of management models, while an example problem is given in Section Six, followed by a discussion on scheduling of implementation.

2 MAIN ELEMENTS OF A WATER QUALITY MANAGEMENT MODEL

A management model aids the decision making process by providing decision makers with alternatives which satisfy various goals and restrictions. The structure and the solution procedure of such a model depends on its purpose and the extent to which various processes and relationships are represented in the model. This chapter gives a description of the main concerns of a management model. Type of problem is the governing issue, which is discussed first.

Types of Water Quality Management Problems

Management issues belong to the categories of planning, design, or operation. In the planning stage, the aim is a regional or basinwide management strategy, which is the focus of the present paper. Decisions regarding scheduling of capacity expansion, upgrading, and implementation of new treatment units also should be made in the planning stage, because the quantity and the concentration of pollutant discharges can significantly vary over the years. Consequently, management strategies must be sufficiently flexible to allow changes that might be required at a later stage.

"Design" in the present context implies the detailed design of individual treatment plants, and it usually follows the planning stage. However, operational and design aspects must be considered during the planning stage as well. Nevertheless, these categories are interdependent, and a systematic analysis of various aspects is needed to obtain a solution to the management problem.

Decisions Associated

Two types of interrelated decisions are applicable for wastewater treatment: quality and quantity of treated wastewater. Decisions about quality should correspond to those achievable by feasible treatment technologies, which comprises a discrete set of alternatives. These alternatives differ in costs and efficiency of treating different pollutants (see Section 4). Quantity related decisions stem from a continuos decision space. Very often, both types need consideration simultaneously, leading to mixed (discrete and continuous) decisions.

Cost Functions

Cost of an appropriate management alternative increases nonlinearly with the increase of treatment efficiency (selected from a discrete set of alternatives), thus forming discrete, nonlinear, cost functions. These functions can be incorporated in a management model in three forms: (1) as nonlinear functions, (2) by piece-wise linearization, and (3) assuming a linear variation. For most management models, these are in the order of increasing complexity, and decreasing order of proper representation. However, the formulation of "cost functions" for this relationship would be complicated and probably infeasible, when considering the variety of possible treatment alternatives. From a practicality point of view, it would be more suitable to determine a feasible set of treatment alternatives for each pollutant discharge. Such sets made available in advance can subsequently be considered in selecting the optimum set of treatment alternatives for the whole river basin. This implies a integer-decision problem, although it can be also reformulated as a continuous problem solvable by continuous mathematical programming approaches. Such reformulations, however, lead to model inaccuracies which produce suboptimal solutions.

Objectives and Constraints

A common objective of water quality management is the minimization of costs, while prespecified quality standards are examples for constraints which limit the feasible decision set. They can be incorporated into the decision making process in many ways. One option is to evaluate each feasible alternative which has an affordable cost, with the aid of a water quality simulation model. Alternatives which does not provide desired quality levels can be eliminated, and, the others can be compared with regard to various objectives. Although this type of enumeration approaches would be feasible for a small scale problem with few management options, most practical problems would render such methods inapplicable.

The other option is to use an optimization model which identify a solution by a more efficient screening process, by explicitly incorporating the objectives and constraints within the model. In this case also, water quality simulation models which provide necessary quality estimates must be accessible to the optimization model. Optimization techniques require simplified forms of simulation models for this purpose, however an exception is dynamic programming which allows usage of complex sub-models within the optimization process.

Under limited budget conditions, a constraint on the expenditure also must be included in the optimization model. If the aim is to satisfy preset quality standards, however, this might lead to an infeasible solution. An alternative formulation is to optimize a water quality indicator in order to minimize the deviation from predefined quality goals (see below).

A further step in the analytical process is handling of multiple objectives inherent in water quality management problems. The decision expected under such situations is a "satisfactory" or "compromise" solution, rather than the "optimum" with respect to a single objective. An appropriate multiobjective decision making technique is needed to aid the associated decision process.

Uncertainty Issues

Water quality problems are dominated by various forms of uncertainties, caused primarily due to uncertain hydrological and meteorological conditions. Uncertainties in water quality model parameters and quality models themselves tend to complicate the problem further. Planning is sometimes done under simplified deterministic assumptions, using selected values for inputs and water quality model parameters. Usually, the "worst case" or "design" scenario is considered for such deterministic analyses. Although this appears to lead to a conservative management decision, even an insufficient decision may result depending on the parameters employed.

A more comprehensive analysis is needed to incorporate uncertainties, and estimate the probability/magnitude of possible failures. Uncertainty can be taken into account by two approaches: explicitly or implicitly. Explicit approaches usually require complex models, and, blows up problem size, often beyond practical limits. This is because they should explicitly consider decisions for a large number of alternative scenarios. Implicit approaches, on the other hand, require generation of several scenarios and determination of the "optimal" solution for each of them. Such an analysis, which can be called a "scenario analysis", opens up possibilities to trade-off among different options that are associated with different probabilities of occurrence. If the generation of scenarios and parameters is done by Monte Carlo simulation, probabilistic conclusions on the performance of various management alternatives can be obtained, at the expense of a lengthy computation.

Parameter uncertainty is perhaps the most important issue in a water quality model. Different methods for estimating uncertain parameters have been reported in the literature. The methodology proposed by Hornberger and Spear (1981) based on prespecified behavior patterns is an example. Model uncertainty is usually accounted for by considering different model formulations to compare the results.

Common Formulations

A common objective function of a management model is the minimization of costs. Quality standards which must be satisfied are considered as constraints. Such a formulation identifies the least-cost management strategy under given constraints, although the least-cost solution may correspond to an unaffordable cost. In cases with limited funds and the possibility of lowering quality standards, analysis can be done considering different quality standards, in order to determine the best utilization of funds. Mathematical formulation of a least-cost management model can be expressed as:

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

subject to $\quad Q_{j,k} \geq S_{j,k} \quad \forall\ j,k$

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

where $C_i(\eta_i)$ is the treatment cost (total annual cost, TAC; or investment cost, IC) required to achieve a treatment efficiency η_i at the i^{th} wastewater discharge. T_i denotes the feasible set of treatment alternatives for the i^{th} discharge. $Q_{j,k}$ is the water quality at the standard location j, expressed by the quality indicator k. The corresponding quality standard is denoted by $S_{j,k}$. $E_{i,k}$ and $E_{standard,k}$ stand for the effluent quality (expressed by indicator k) at the i^{th} discharge, and the effluent standard for k^{th} indicator respectively. It should be noted that if the ambient standards are neglected, no optimization would be required to identify the resulting management action.

If available funds are limited, it is also appropriate to aim at the best quality achievable under given cost constraints. Minimization of an indicator which represents deviations from quality standards is a suitable objective function, and it can be presented mathematically as given below.

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

subject to $\quad$ TAC $\leq$ Maximum allowable TAC

$\quad$ IC $\leq$ Maximum allowable IC

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

Various combinations of the above formulations are employed in management studies, mostly through sensitivity analyses or multiobjective decision making techniques. Formulation of a combined objective function, which consist of the sum of different objectives weighted with predefined weight factors, is another approach. However, determination of the weight factors is the most critical and impractical element of such an approach.

Miscellaneous Management Strategies

A brief description of various other management strategies, some of which need the use of above model formulations (or modified forms of them), are described below. (see Somlyódy and Paulsen, 1992; for more details).

(1) The best-available technology (BAT) which is commonly applied in North America and Western Europe, is associated with very high costs, although the task of controlling authority becomes much easier. This is a uniform management policy which requires no

optimizations, and it is safe from an environmental view point. However, there exists the possibility to choose much cheaper technologies which provide nearly the same water quality levels, because the cost has an exponential relationship to water quality in the higher quality range.

(2) Uniform percentage reduction, requires each discharger to reduce emissions by the same percentage in order to meet a set of regional ambient standards. This policy can be formulated using an ambient water quality model, and no optimization is required. The appropriate percentage can be determined by estimating, by trial-and-error, the river water quality resulting from different reduction levels.

(3) Least-cost strategy that meets limitations on total emissions, while achieving ambient quality goals approximately, requires quality simulation and simple optimization models for the policy formulation. In this case, the optimization model identifies the least-cost strategy which satisfies a particular constraint for total emissions. Subsequently, the effect of the resulting control strategy is estimated by the water quality model. If the ambient quality goals are not achieved adequately, or, if the strategy is too expensive, the constraint (on total emissions) is adjusted accordingly, and the procedure is repeated.

Economic Instruments

Water quality management cannot be achieved without the use of proper economic instruments such as fees and fines. They are needed to encourage polluters to reduce their discharges by using efficient treatment technologies. However, incorrectly established economic instruments might cause a greater loss to the economy, than when they are not imposed. In order to formulate management actions which are beneficial to the whole society, economic instruments must be properly identified together with management alternatives during the policy analysis stage.

Reliability of Management Actions

Any management alternative, especially in water quality management, is associated with a particular reliability. This is mainly due to uncertainties in hydrological events, water quality model parameters, and water quality models themselves. Alternatives with very high reliability levels would need heavy expenditures. As a result, "reliability" virtually becomes an additional objective of the existing multiobjective problem. There are methods which incorporate reliability as an input parameter in a management model formulation. However they may lead to models which are either overly complicated and prohibits incorporation of detailed water quality models, or provide coarse (and often conservative) results. On the other hand, it may be more appropriate to evaluate the reliability of management actions by posteriori simulations.

Solution Methodology - Simulation vs Optimization

Most of the management strategies are formulated using simulation models, optimization ones, or both, although some strategies do not need any of them (eg. BAT strategy). As already noted, small scale problems involving only a few decisions can be solved with simulation only, by evaluating each feasible decision in order to select the preferred one. Optimization-only problems are not common, since a water quality simulation is needed for most cases. However, an example for such a problem is the minimization of cost needed to maintain total effluent loads below a basin-wide maximum.

A common approach is to use an optimization model for preliminary screening only, and perform solution refinements by subsequent detailed simulations. This method is required for many optimization techniques, since they require simplified forms of the problem: simplified water quality models, cost functions etc. In many cases, water quality estimations within the optimization procedure are obtained using a set of "transfer coefficients" which relate pollutant discharges to ambient quality at various locations. Alternatively, simple linear water quality models can be incorporated within the optimization.

As an example, we refer to the eutrophication management model developed for Lake Balaton in Hungary (Somlyódy, 1986). The method, for different lake basins, incorporated stochastic, linear load-response functions (or transfer coefficients) which were derived from a dynamic phosphorus (P) cycle model (used in a Monte Carlo fashion under systematically changed external P load conditions where climatic factors and loads were generated in a stochastic fashion), control variables characterizing P removal at existing wastewater treatment plants, the export of sewage (out of the watershed) and the construction of pre-reservoirs. Nonlinear cost functions were piece-wise linearized. The stochastic problem was approximated by an expectation-variance approach solved by linear programming (see Section 6.2 as well). The objective function was formulated such that the weighted sum of the improvement in the expectation and variance of the water quality indicator (the annual peak chlorophyll-a concentration characterizing algal biomass) be minimized. A budgetary constraint was set to the total annual cost.

An accurate representation of a water quality problem often requires the use of nonlinear models, nonlinear cost functions, and, incorporation of uncertainty issues in the analysis. Most of the commonly used optimization techniques are inappropriate in this respect (see Section 4 for details), except dynamic programming (DP) which imposes no such restrictions. However, DP has other limitations with regard to the number of state variables (water quality indicators) that could be used to represent the system state with sufficient accuracy.

Whichever the methodology used, it would rarely represent the reality with sufficient accuracy. Solutions obtained can differ depending on various factors, the most crucial ones being uncertainty in parameters and models. Furthermore, quality management problems are rather loosely defined in many countries. There can be many aspects of the problem which are not well defined, including quality standards, economic instruments, type of management strategy needed (short-term, long-term, stage-wise development), economic objectives etc. Thus, a sensitivity-style analysis is unavoidable to obtain a satisfactory solution. Such an analysis opens up the possibilities to use various approximate methods (e.g. linearized problem formulations; see Somlyódy and Wets, 1988) satisfactorily.

3 WATER QUALITY MODELS

The original BOD and DO model proposed by Streeter and Phelps (1925) has subsequently been improved in various ways. Many of these improvements include fairly simple first-order exponential decay, dilution and sedimentation models for other nonconservative and conservative substances (Loucks et al. 1981). One basic improvement is to model BOD as comprising of two components: carbonaceous and nitrogenous BODs (CBOD and NBOD). In this case, two different decay rate coefficients for the two components are considered. The extended S-P model (Thomann and Mueller, 1987) includes the sediment oxygen demand (SOD) in addition to DO, CBOD and NBOD.

In river systems where nitrogen is a predominant constituent, additional forms of nitrogen must be considered. O'Connor et al (1976) proposed sequential reaction models for four nitrogen components: organic nitrogen (ON), ammonia nitrogen (NH_3-N), nitrite nitrogen (NO_2-N) and nitrate nitrogen (NO_3-N).

The underlying assumption of the above steady state models is that the flow and temperature do not change over time. More complex non-steady-state models are needed to analyze the time-varying flows and pollutant discharges. They are important especially for controlling the short-term operation. Simpler steady-state models are more relevant to long-term planning.

In 1970, the first version of the well known QUAL model family of the US Environmental Protection Agency simulated stream temperature in addition to BOD and DO. The possibility to simulate both steady and unsteady flow as well as the impacts of four nitrogen components and their reactions was included in the second version. Over the past two decades there has been an increasing emphasis on the effect of nutrients on aquatic ecosystems. One example is the QUAL2 model (Water Resources Engineers, Inc., 1973). It incorporated CBOD, DO, NH_4-N, NO_2-N, NO_3-N, phosphorous, algal biomass (chlorophil-a), coliforms and radionuclides. Consideration of eutrophication (chlorophil-a) results in model non-linearities, which occur also due to nutrient exchanges between sediment and water. This is an important issue in the integrated use of water quality models and optimization techniques for management purposes. Model non-linearities cause significant complications in such situations. The consequences of model non-linearities in the optimization approaches are discussed in Section 4. (see Somlyódy and Varis (1992) for a recent review of state-of-the-art water quality models).

QUALIIe (Brown and Barnwell, 1987) is a dynamic water quality model, designed solely to calculate diurnal variations. It has a large number of parameters (about 50) and a more comprehensive definition of reactions. Nutrient and light limitation are incorporated in detail. The large number of parameters complicates the calibration of the model.

All these models, irrespective of how comprehensive they are, can only provide rough approximations of the many interactions taking place in a river. Multiconstituent models generally provide a comparatively more detailed representation, but they also require more data and careful calibrations. Spatial dimensionality considered in the models is another simplifying factor. The assumption, very often, is that the pollutants are completely mixed over the river cross section. This enables modelling the system as a one dimensional one (along the river). A multidimensional river model of a river would imitate the reality more closely, but the data requirements, complexity and the subsequent computational difficulties render such approaches impractical. Selection of steady-state or dynamic water quality models also need the evaluation of such trade-offs. Another choice has to be made between stochastic or deterministic models. A stochastic model takes into account the uncertainty of various processes, and demands more data than the deterministic counterpart. The calibration procedure of a stochastic model is also difficult as the comparisons are to be done statistically.

As this brief review implies, there are many different types of water quality models. The appropriate model-type for a particular situation mainly depends on the purpose of the study, the characteristics of the river system and the various objectives, constraints, and goals of a particular planning agency. State-of-the-art water quality models suitable for river basin studies are summarized in Table 1 (see Somlyódy and Paulsen, 1992; for details).

A detailed description of different models (including those dealing with heavy metals) is beyond the scope of this paper. In addition to the literature cited previously, the following can be recommended for detailed descriptions of water quality modelling aspects: Rinaldi et al. (1979), Orlob (1982), and Somlyódy and van Straten (1986).

TABLE 1 State-of-the-art water quality models

	Model	State variables	Parameters	Receiving Waters	Use	References
(1)	Streeter-Phelps (S-P)	2 (DO, BOD)	2	Rivers	Static	Streeter and Phelps (1925)
(2)	Extended S-P	3(DO,CBOD,NBOD, SOD)	3-5	Rivers	Static	Thomann and Mueller (1987)
(3)	S-P and N cycle	6 (DO, CBOD, ON, NH3-N, NO2-N, NO3-N)	8-11	Rivers	Static	Orlob(1982), Thomann and Mueller (1987)
(4)	QUAL2e	10 (as for (3) and SOD, OP, DIP, CHLA, T)	about 50	Rivers	Dynamic	Brown and Barnwell (1987), Orlob(1982), Thomann and Mueller (1987)
(5)	TP	1 (TP)	1	Lakes, reservoirs, rivers	Dynamic, Static	Vollenweider (1968), Thomann and Mueller (1987)
(6)	Simple P cycle	3 (DP, DIP, AP)	>14	Lakes, reservoirs, rivers	Dynamic, Static	Thomann and Mueller (1987), Somlyódy and van Straten (1986)

CBOD=carbonaceous BOD, NBOD=nitrogeneous BOD, SOD=sediment oxygen demand, ON=organic N, OP=organic P, DP=detrius P, DIP=dissolved inorganic P, CHLA=chlorophyll-a, AP=algae(or phytoplankton) P, T=temperature

4 WASTEWATER TREATMENT ALTERNATIVES

Wastewater treatment technologies include mechanical, mechanical-chemical, mechanical-biological, and various advanced combinations of them. The simplest alternative is mechanical, or primary, treatment, which usually refers to the removal of suspended solids by settling or floating. The most widely used primary treatment operation is sedimentation. Removal rates of primary treatment corresponding to BOD, suspended solids (SS), total phosphorus (TP), and total nitrogen (TN) are approximately 30%, 60%, 15%, and 15% respectively.

Chemically enhanced mechanical treatment (of low dosage) refers to addition of coagulants and flocculants to simple primary treatment. The associated removal rates are 60%, 80%, 80%, and 25% for BOD, SS, TP, and TN respectively (Morissey and Harleman, 1990).

Secondary chemical treatment (of high dosage) is obtained by adding a variety of coagulants, a flocculation basin, and a post settling tank to primary treatment. The removal rates are approximately 80%, 90%, 95%, 25%-30% for BOD, SS, TP and TN respectively.

Biological, or secondary, treatment generally involves a biological process to remove organic matter through biochemical oxidation. Activated sludge reactor is a commonly used biological process in which wastewater is fed to an aeration tank where organic material is metabolized by aerobic bacteria. The resulting activated sludge is settled in a sedimentation vessel. With biological treatment, the removal rates of BOD and SS increase to about 90%, depending on the composition of the wastewater. Removal rates of TN and TP are only slightly higher than the primary treatment.

Chemical removal of TP in addition to biological treatment and/or upgrading of highly overloaded biological plants by adding chemicals is termed biological/chemical treatment. Additional TN removal is possible through oxic and anoxic regimes, while TP can be also removed biologically through anaerobic processes. Tertiary, or advanced biological treatment commonly removes nitrogen and phosphorus under carefully controlled combinations of oxic and anoxic conditions. Thus, a large number of process combinations exist which are made up of aerobic, anoxic, and anaerobic (biological phosphorus removal) processes; together with chemical addition (pre-, simultaneous or post-precipitation). High removal rates: 95%, 95%, 85%, and 85%, for BOD, SS, TP, and TN respectively, can be achieved.

The cost of a treatment plant depends on the treatment technology (effluent requirements) and the capacity. Capital cost of an activated sludge biological plant is approximately twice that of a mechanical treatment plant, while an advanced plant costs about three times. For a large plant serving a town of about 100,000, mechanical treatment requires a capital expenditure of about USD 1/m^3. USD 0.1/m^3 is needed for operation, maintenance and replacement, which, for a advanced biological plant, is about USD 0.3/m^3.

5 SOLUTION METHODS FOR THE WATER QUALITY MANAGEMENT PROBLEM

Optimization, simulation, or both are employed in formulating "optimal" wastewater treatment strategies for river basins. Optimization models perform a screening to select an optimal decision, while simulation models evaluate the effect of various processes and management alternatives on the river water quality. Various physical, chemical and biological processes that characterize a river system are described in a simulation model in significant detail. This is one advantage of a simulation approach over an optimization approach, which require considerable simplifications of the processes occurring in the system. However, the large number of simulation runs required to identify the optimal solution is the main disadvantage in using a simulation-alone management model.

Due partly to the above reason, many of the water quality management models in the literature are optimization models. They have been commonly used in combination with simple simulation models as well. Linear programming and dynamic programming have been the most widely used techniques. Despite the complexity of nonlinear programming approaches, their use in the water quality management has also been demonstrated. The systematic heuristic search approaches (e.g. simulated annealing, and taboo search; see Pirlot, 1993 for details) that differ significantly from mere simulations have not received much attention in this field.

Management models can be also categorized into centralized or decentralized models, depending on how the river basin is actually managed. In a centralized management, there is a single agency which owns and operates all control facilities. This permits the construction of few large treatment plants that are preferred due to economies of scale. However, this cost-saving may be accompanied by a greater distance to which wastewater must be pumped. In a decentralized system, the polluter has the freedom to treat the waste, or to pay a charge imposed by the regional authority. For modelling purposes, the centralized approach can be termed as a generalized method. The decentralized approach becomes a specific case of the centralized approach which consider wastewater transfers from one location to the other. A brief survey of the problems and future prospects for the development and application of models for water quality management can be found in Beck (1984).

The applicability of the optimization techniques of linear programming, nonlinear programming and dynamic programming; to water quality management problems; is discussed below. A discussion on multiobjective decision making techniques, which form an essential component of management modelling, is also included.

5.1 Linear Programming

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

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

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

Several different methods are available to handle uncertainty in a LP formulation. Stochastic linear programming is one such method, which results in a large increase in the number of variables and constraints. This is a result of incorporating various alternative scenarios into the model formulation explicitly. An alternative approach is to use chance constraints. It reformulates the stochastic problem into a corresponding deterministic one by defining constraints based on certain reliability levels. This is considered as a pessimistic approach. Also, the selection of the reliability values (which can be even be a part of the optimization problem) is not a straightforward task.

5.2 Nonlinear Programming (NLP)

Wastewater treatment cost functions, as well as "transformation functions" which relate waste discharge to river water quality, are often nonlinear. Therefore, a nonlinear optimization model is a more accurate representation of the physical system than the linear model. Nevertheless, NLP approaches have been less popular, due to their complexity and large computational requirement. Furthermore they cannot easily incorporate uncertainty in the model formulation.

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

5.3 Dynamic Programming

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

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

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

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

5.4 Multiobjective Decision Making Techniques

A management decision taken by considering only one or a few of various objectives may be undesirable in light of the other objectives. Multiobjective decision making techniques aids the process of trading-off among various goal-achievements, so that a compromising decision can be made.

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

Multiobjective models for regional water resources investment programs (but not for water quality management) are presented by Cohon and Marks (1973). Literature on MCDM techniques include those presented by Zeleny (1982) and Goicoechea et al. (1982), among many others.

6 STATE-OF-THE-ART REVIEW OF MANAGEMENT MODELS

6.1 Linear Programming Models

According to Loucks et al (1967), the first attempt to apply mathematical programming techniques to the river water quality problem was by Deininger (1965). In that work, a linear programming model was structured using various approximations of the differential equations that describe the dissolved oxygen profile of streams. Loucks et al. (1967) presented two linear programming models for determining the amount of wastewater treatment required to achieve, at minimum cost, a set of DO standards within a river basin. The two methods differ in the way they imposed the minimum DO standards. One method was to set several DO constraints within each river reach which was assumed homogeneous. In this case, a single constraint is sufficient for the reaches for which the critical time is longer than the travel time from the beginning to the end of the reach. The second method was to ensure that the BOD concentration at the beginning of the reach is less than the critical BOD level. This critical BOD level is that which would result in a DO deficit equal to the maximum allowable deficit for the reach. It was emphasized that any comparison of quality standards should be based on an assessment of the resulting DO profile in each reach, and not on the changes in the minimum allowable DO concentrations. This is due to the following reason. There can be reaches of which a change in the minimum allowable DO level does not affect the DP profile of any other reaches, while such a change in some other reach may affect the DO profile in many reaches.

Bundgaard-Nielsen et al (1975) discussed the interaction between the level of effluent charge or taxation and the choice of treatment technology. A linear programming model was used to estimate the least-cost treatment alternatives. It is shown that there is a risk of overtaxation, i.e., simply increasing taxation may fail to improve water quality but only increase production costs and thus consumer prices. The possible inefficiency of a surcharge to abate pollution is similar to that of taxation, and it is also discussed by these authors.

Loucks et al. (1981) presented an example model which can be solved by mixed integer programming. The purpose of the model is to determine the degrees of treatment for carbonaceous and nitrogenous BOD components at each waste outfall in a river basin. Effluent standards were expressed by maximum BOD levels, and, BOD and DO limits were included as ambient standards. The objective of the analysis was to identify the treatment efficiencies that minimize the sum of the wastewater treatment costs. The possibility to start the solution procedure with a small number of constraint points (thereby reducing the computations) was pointed out. This leads to a trial-and-error solution procedure, as the solution must be checked to ensure that the concentrations within the entire river are acceptable. If the solution is unacceptable, the solution procedure should be repeated after setting constraints for the concentrations at the critical locations. This example was formulated using analytical equations to predict BOD and DO. Alternatively, a two-stage approach can be used to analyze the river system using finite-section models. In such a model, the river reach is divided into a number of reaches within which the quality parameters are constant.

Burn (1987) examined three model formulations for water quality management. The model formulations consisted of: a linear cost minimization, a chance constrained optimization (considering the pollutant loadings as random variables) and, minimization of the variance of the quality response. These were applied to a problem involving five pollution sources at which treatment plants are located, and twelve receptor locations at which the water quality is of concern. The transfer coefficients describing the response at each receptor location for a unit release from each source have been prespecified. The results showed, as expected, that the greatest economic efficiency was obtained by the use of minimum cost formulation. The minimum variance model resulted in the poorest expected quality, but it guaranteed a smaller variability of the water quality than the other two models. Both the chance constrained formulation and the variance formulation indicated reductions of the mean water quality. The value of different model interpretations to the decision maker was emphasized, as the decision maker would then be better able to choose a solution considering the implementation which incorporates the pertinent aspects of the problem.

Stochastic LP Approaches

An algorithm that produces an optimal phosphorus management strategy for stochastic phosphorus loading conditions was developed by Fontaine and Lesht (1987). An optimal strategy was defined as that which could achieve desired phosphorus concentrations for the least basin-wide cost. The optimization model was encapsulated in a Monte Carlo framework. Loading rates and settling rates were picked randomly from pre-specified truncated normal distributions. Three thousand optimal strategies were identified. These strategies were used to form statistical distributions for the load reduction capacities at each treatment location. Treatment capacities at different locations that correspond to the same percentile of their respective distributions were identified, considering the cost limitations as well.

A stochastic programming model for regional water quality management was developed by Lohani and Saleemi (1982). The model incorporated; as stochastic entities; water quality parameters (deoxygenation rate and reaeration rates), streamflow, waste flow rate, and effluent BOD concentration. Probability distributions were fitted for the streamflow and the waste water flow. Effluent BOD concentration was assumed to be uniformly distributed. The two rate coefficients were considered to be distributed normally. Model constraints were expressed in terms of random observations, and they were transformed to deterministic equivalents, for obtaining the solution using a deterministic technique. An application of the model was made to the Hsintien River in Taiwan. The results were compared with the results of an earlier model which had the streamflow as only stochastic variable. More economical strategies than those identified by the earlier model have been obtained.

Burn and Lence (1992) proposed an approach for including uncertainty by considering multiple scenarios. A scenario is comprised of a flow value, a water temperature, and a pollutant loading impact from nonpoint source contributions (which will be unaffected by the selected treatment levels). Four optimization model formulations called: (1) minimize maximum violation, (2) minimize maximum regret, (3) minimize total violation, and (4) minimize total regret were applied to obtain alternative solutions. The optimization problems were solved by LP. The first optimization model formulation minimizes the worst water quality within the system. In the second formulation, the maximum regret was defined as the maximum difference between: the water quality violation that occurs when all the scenarios are equally likely and the violation that occurs if the correct scenario is known with certainty. Third formulation minimizes the sum of water quality violations at each of the check points for all scenarios. The final formulation is analogous to (3) for the basic model (2). This study has employed the analytical DO model described by Thomann and Mueller (1987) to estimate the transfer coefficients. The results suggest that the model formulations based on total violations and total regret are preferable over the other two, for the Willamette river basin that was analyzed. However, the assumption that all the scenarios are equally likely will not generally be true, and may consequently result in solutions that are too conservative.

6.2 Nonlinear Programming (NLP) Models

A nonlinear water quality management model has been presented by Hwang et al. (1973). In that model thermal as well as organic wastes have been considered. The water quality standards considered were: minimum DO, maximum BOD, maximum stream temperature and the allowable rise in the stream temperature. The optimization problem was formulated to find a set of treatment alternatives that minimum the total treatment cost, yet satisfying the preset water quality standards. This problem has been solved by the generalized reduced gradient method (GRG).

Bayer (1974) used a nonlinear optimization model to examine the least cost combination of wastewater treatment facilities along a stream required to meet specified BOD and DO standards. Results were compared to those obtained by some other researchers (ReVelle et al., 1968; Liebman and Lynn, 1966) using linear and dynamic mathematical programming techniques. Bayer (1974) concluded that the nonlinear programming is a better solution method for these problems, than LP or DP. The results however show that the DP formulation provided a better solution in terms of the total cost. It is stated that the reason for this "discrepancy" is not known. A similar problem was examined by Pingry and Whintson (1974) using a nonlinear multigoal optimization model. A simulation model of the river basin served as the constraint set for the optimization model that searched for the least cost set of treatment capacities required to meet water quality goals. Special attention was given to the possibility of relaxing the quality goals downstream in order to improve water quality upstream.

Water quality management under time varying river flow and discharger control was studied by Herbay et al. (1983). The hydrology of the river was presented by a set of steady state flow regimes. For each of those regimes, a river water quality model was used to give a constraint set to be imposed on the dischargers in order to achieve the stream standards during that regime. An objective function consisting of investment costs, fixed operating costs, and variable operating costs was considered. A treatment strategy was sought that minimizes the sum of these costs, taking account of the possibility of operating the treatment system at various different levels during the year. The S-P type water quality model used in the analysis modelled BOD and DO. This problem formulation had a major assumption that an activated sludge treatment plant followed by an activated carbon process is present at each treatment plant. The reason for such a simplifying assumption has been the lack of information about operating costs of wastewater treatment plants. With this assumption, the system was supposed to be dimensioned for the maximum removal capacity, a fraction of that being exploited for a particular hydrologic regime. A nonlinear optimization system (MINOS, Murtagh and Saunders, 1977) was used to solve the resulting nonlinear problem.

An interactive multi-objective decision support system was developed by Rathke et al. (1987). There were no special requirements with regard to the type of process equations and goal functions. Nonlinear models with time delay of any order, non-convex or non-concave goal functions, and discrete control variables could be incorporated into the computation. Time dependant upper and lower bounds for the control variables could also be specified. A river basin in Germany was analyzed. A one-dimensional river water quality model (a modified S-P model) was employed, considering a segmentation of the river. DO and BOD was modelled. Mean flow conditions and mean low flow conditions were considered to develop management alternatives. The objective functions considered were the maximization of the sum of DO concentrations (in space and time), minimization of BOD levels, and minimization of total costs. Based on these objectives, Pareto-optimal solutions were computed, from which a decision maker could select a satisfying decision.

Rossman (1989) presented a method to design seasonal discharge programs that limits the risk of one or more water quality standard violations in any year. A case study of controlling ammonia toxicity was presented as well as a comparison of the potential savings available from seasonalization for several pollutants on two rivers with different seasonal regimes. This analysis considered a single point source discharge. The effluent concentration limits for different months of the year were the decision variables. A function of these decision variables was used to compute the risk of annual water quality violations. The receiving water's seasonal load capacity (which is a random variable) was defined as the maximum discharge concentration that it can accept without incurring water quality violations. The statistical properties of these values were estimated from the historical observations of water quality. A Markov-like assumption was made for the behavior of these random variables. Based on these, an expression was obtained for a discharge limit that corresponds to a particular risk. The resulting nonlinear optimization problem was to determine the minimum cost strategy for the quality management. It was solved by a version of the Hestenes-Powell-Fletcher multiplier penalty algorithm as described by

Cuthbert (1987). It was concluded that, a proper evaluation of the effectiveness of seasonal discharge programs need a comparison on a risk equivalent basis. It was pointed out that seasonal discharge limits can reduce treatment efforts without increasing the frequency of violations. However, the total cumulative mass discharged to the receiving water will be greater than for a constant year-round limit.

Stochastic NLP Models

A chance-constrained stochastic programming model was presented by Ellis (1987), based on the water quality management model of Fujiwara et al. (1986). It determines the least-cost allocation of wastewater treatment efficiencies, considering BOD removal rates. Probabilistic restrictions were imposed on maximum allowable ambient DO deficit. Random variables included in the model are ambient BOD, ambient DO deficit; initial values of them, streamflow, waste flow rates, rate coefficients of the S-P DO sag model (deoxygenation, reaeration, and sedimentation-scour rates), photosynthetic input-benthic depletion rates and nonpoint source BOD input rate. These random variables appeared in highly aggregated forms which constitute part of the probabilistic constraints of the water quality optimization model. A new chance-constrained programming variant, imbedded chance constraints, is presented along with an example application. This method imbeds a chance-constraint within a chance constraint. It permits the selection of nonexpected value realizations of the mean and variance estimates employed in the deterministic equivalents of traditional chance-constained models. A joint chance-constained formulation is also presented which illustrates the possibility for prescription of an overall system reliability level, rather than reach-by-reach reliability assignment. The approach has been demonstrated with a hypothetical example containing only one model constraint. Being a highly mathematically oriented approach, the drawbacks inherent in chance constrained formulations (see Section 4.1) might make it unjustifiable.

A fully stochastic optimization procedure was applied by Somlyódy and Wets (1988) for the Lake Balaton problem referred to earlier (in Section 2). As contrasted to the expectation-variance approach, the objective function minimizes the sum of deviations of the water quality indicator (chlorophyll-a) from pre-specified goals for all the lake basins by using a piecewise linear-quadratic-linear penalty function. Detailed analyses have shown the similarity between the two approaches, and the inability of deterministic (linear or quadratic) model versions to capture major features of the problem.

6.3 Dynamic Programming Models

Dysart (1969) described the use of dynamic programming in regional water quality planning, using a simplified example. A river basin with one major stream, several industries, and municipalities, was considered. The waste discharged to the river is assumed to be characterized by its BOD. The transfer function used for BOD was the simple first-order decay function, while the basic Streeter-Phelps model is used for DO. To meet specified DO standards with a minimum cost, decisions were made regarding the levels of treatment at each of the outfalls. The solution process was described to begin downstream and to move upstream, which is computationally inefficient in comparison to an upstream-to-downstream computation (see below).

Futagami (1970) applied dynamic programming for optimal sewage system planning problem in the Yodo River System in Japan. Consideration was given to maintaining allowable levels for ambient BOD levels only. The objective function was to minimize the cost (construction + maintenance). It was expressed as a quadratic function of the decision variable which is the treatment efficiency for each treatment plant. This indicates a major simplification of the actual discrete, nonlinear, relationship between the efficiency and the associated costs.

Shih (1970) developed a river basin water quality management model based on DP. The objective was to find the least-cost solution to a wastewater treatment problem, considering the costs of wastewater and water treatment. Although water quality goals (BOD and DO were modelled) were not included as constraints in the model formulation, they were indirectly incorporated in the objective function, through the costs of water treatment necessary to produce potable water. A set of simplified cost functions was used in the model. The cost of a wastewater treatment alternative was taken to be dependant on the plant capacity and the removal efficiency. Water-treatment cost functions were expressed in terms of raw water quality and plant size. A hypothetical river was analyzed, in which water intakes and waste discharges were assumed to be located along the river.

The need to optimize the combined costs of wastewater and water treatment (which reveal direct benefits of waste treatment) was emphasized. If the wastewater treatment strategies were developed using prespecified water quality levels, and by minimizing wastewater treatment costs only, the resulting cost savings may be overridden by water treatment costs or vice versa.

Newsome (1972) described the Trent river model. The river system was divided into reaches which were identifiable geographically. Each of the reaches was sub-divided into "stages". A stage was defined as a length of river from a point immediately upstream of an installation to a point immediately upstream of the next installation downstream. The locations of "installations" were those of the measurement points and the locations which affect, or are affected by the river water quality. At the measurement points, the river water quality in the model is constrained to conform to minimum value(s). The model has normally dealt with each individual user's installation separately. But some installations were aggregated and treated as a single installation.

The performance of the model was checked by using data from existing installations and comparing the results in terms of both quality and quantity with actual observations. Subsequently, new treatment plant locations were introduced.

Only a broad classification of river water quality as described by the pollutants was considered in the model. This classification was based on the "uses" that were defined by the needs of the abstractors (or for the survival of fish) and the decisions available to them, when confronted by different qualitites of river water. Each decision is discrete, and produces a discrete "quality state" in which a unique set of uses is possible.

The DP model progresses through the river system, stage by stage, from upstream to downstream. As it continues, it carries forward a proper chemical analysis of the selected pollutants. The results of the chemical analyses are compared with the table of quality states. From the available decisions, the model selects those which are possible, records them and their associated costs and proceeds to the next stage. When it reaches the downstream boundary, the model selects the decision sequence which results in the least cost, and retraces the decisions taken at the previous stages.

The model selects only the minimum-cost alternative within a quality state to be the representative state to proceed with. This has violated the basic principles of DP (by selecting a local optimum), and the solution can be far from the global optimum, unless there was a fine discretization of quality states. However on the other hand, it is not practical to consider all the alternatives for the subsequent computations, as this may lead to an enormous computation load. One alternative approach will be to use the concept of Incremental Dynamic Programming, which searches iteratively over imaginary "state-space corridors" through the stages.

It was found that the river quality in the model usually trapped into one or two quality states. Attempts to improve the situation by sub-dividing the quality states has proved impracticable. (due to the computer storage problems). Subsequently, "combined quality states" (the product of ammoniacal nitrogen and suspended solids concentrations) were introduced with a limited success. This has led to a decrease of the number of quality states as the model progresses. Attempts to maintain a sufficient number of quality states also is reported.

Klemetson and Grenney (1978) developed a wastewater treatment optimization model to evaluate the timing and capacity expansion decisions for wastewater treatment systems. The procedure consisted of selecting the combination of treatment plants and treatment levels over the planning period that best meets the desired objectives at the lowest total discounted future costs. The following were among the factors considered in the model. (1) quality and quantity of wastewater and its change with respect to time, (2) rate of interest and the rate of inflation, (3) capital, operation, and maintenance costs, (4) treatment efficiencies, (5) economies of scale, (6) excess capacity, and (7) service life.

The entire optimization problem was decomposed into a sub-optimization of each of the alternative treatment schemes for each year. This produces a single cost parameter for each alternative that could be used as the state variable to optimize the entire system. The term 'cost' refers to the present value of the future costs of building, expanding or upgrading, and the costs of the operation and maintenance.

All of the alternative treatment schemes to be considered by the model were specifically designated by the user. The designation does not change the fact that the existing system of the previous year still has its capacity and debt.

The costs were determined by simplified cost equations. The model adds the costs generated to the capital debt, and the annual capital repayment was calculated on the basis of the previous years debt. The capital debt remaining was reduced by this amount. The annual costs were determined for each plant and sewer, and these were converted to present worth values at the base of the study. This value was available as a state in the DP-based optimization process. An attempt has been made to reduce the computer storage requirements by having an additional "forward pass" in the DP computation, in which the detailed computations along the optimal path were performed. The model was applied for the Lower Jordan river in Utah, USA.

Hahn and Cembrowicz (1981) described the Neckar River model for which DP was employed following the applications, with limited success, of linear and mixed-integer programming techniques. The river system was approximated by a sequence of sections whose morphologic, hydraulic, chemical and biological properties were assumed to be homogeneous. Points of inflow, outflow, delivery etc were located at the beginning of the sections. The concentrations of quality indicators were computed from the beginning to the end of any section with one of the two quality model formulations. A simple two-parameter model which consider BOD and DO was the first; while a more detailed biochemical model describing organic carbon and organic nitrogen oxidation, mineralization through sessile and suspended organisms, photosynthesis, sedimentation, and to limited extent erosion; was the second. The simpler first model has been used more often, due to computational reasons.

Discrete quality states were employed for the analysis, and the DP computations were started from the most downstream section. This, however, has the disadvantage that all possible state transformations at each section (stage) of the problem must be evaluated. After continuing the computation upto the uppermost section, the optimal solution was retraced downstream.

In defining the discrete quality states, feasible ranges were divided into equal intervals. For example, the size of the intervals for DO-deficit and BOD were selected to be 0.1 mg/l and 1.0 mg/l respectively. Four treatment options were considered for each discharge location. As in the model of Newsome (1972), if several solutions have quality states which fall into the same discrete quality interval, the solution which had the best objective function value only was selected to represent that quality state.

An objective function which comprise net costs (costs - benefits) was minimized by the DP approach. It was found that the monetary evaluation of benefits has only a weak impact on the solution. Considering the weaknesses and difficulties in evaluating benefits, an alternative objective function was formulated, which gave a measure of the intangible benefits, by a combined objective function comprising two objectives. The first objective was the net costs, while the second one represented intangible environmental benefits in terms of the deficit of the DO concentration. Scalarized values of the two objectives; weighed individually with optional factors, raised to a power p ($p \geq 1$) for scaling the deviations, and summed-up; constituted the objective function.

Although the results presented based on different objective formulations do not favor one formulation, authors concluded the suitability of the combined objective to develop economic bases for setting of quality standards. However, the solution obtained with a combined objective may be highly sensitive to individual weight factors. It is also possible that only a few critical discharges govern the results, while treatment decisions obtained for other locations would need further improvements (see Section 2 for a similar single-objective formulation and its consequences).

Joshi and Modak (1987) developed a reliability based discrete differential dynamic programming (DDDP) model for river water quality management under financial constraint. Their paper includes some critical remarks on the heuristic selection of the best decision of Newsome (1972). The model accommodated uncertainty in the system by Monte Carlo simulation. Water quality models based on EDI-QUAL-1 (Willis et al.,1975) was used to generate the river water quality profile. At the most seven water quality parameters namely, DO, BOD, Total Coliforms, Ammonia, Nitrates, Nitrites and Organic Nitrogen could be considered.

The solution procedure in DDDP (or Incremental Dynamic Programming (IDP), which is the generalized version of DDDP) is based on successively examining comparatively small regions of the state space. These smaller regions in the state space are termed as "corridors" in the DP terminology. In order to reduce the chance of obtaining a local optimum solution, "compartments" within corridors were introduced. Two policies were considered to select the states with which the subsequent computations downstream should proceed. In policy (1), the states that yield the best set of objective function values were selected. Policy (2) selects the best state within each compartment, to form the state space with which to proceed. At the end of each iteration, the solutions obtained by these two policies were compared. This can be described as a substantial improvement over the approach of Newsome (1972). The uncertainty of the system was incorporated by performing a large number of Monte Carlo simulations within each stage. However, such an approach would take a considerable time even in today's computers.

Stochastic DP Approaches

Lohani and Hee (1983) applied chance-constrained dynamic programming (CCDP) with the objective of minimizing operation costs of a regional water quality management system. The streamflow in each month of the year was considered as a random variable having a known probability density function. A zero-order decision rule that permits the conversion of the chance constraints into equivalent deterministic constraints was used. Thereby the problem was reformulated as a straightforward deterministic optimization problem. The model was applied to the Hsintien River in Taiwan for determining the optimal solutions. Different combinations of minimum DO levels and degrees of reliability were considered to obtain corresponding optimal treatment levels.

Stochastic dynamic programming (SDP) models for water quality management were developed by Cardwell and Ellis (1993). Both parameter (Type II) and model (Type I) uncertainty were taken into account. S-P equations as well as sophisticated water quality models (QUAL2E and WASP4) were employed to account for the model uncertainty. The models were applied to a waste load allocation problem for the Schuylkill River in Pennsylvania. The goal was to identify least-cost management policies. Model formulations that permit violations of water quality standards were developed. Control strategies that allow operational variability were devised and aggregate measures of the violations were traded off against total cost. In a deterministic DP problem, a water quality state and control option at one stage strictly maps to a single state at the next stage. In contrast to this, the SDP model was formulated by considering a Markov transition matrix at each stage. This allows various feasible states (associated with various probabilities) at the next stage. Monte Carlo simulation was used to generate these probability matrices, thereby incorporating the parameter uncertainty (this was done by considering different water quality models mentioned above).

In the work of Cardwell and Ellis (1993), the variable treatment strategies developed using Markov transitional matrices gives rise to some inconsistencies, unless certain preconditions are met. This is due to the steady state assumptions involved in using Markov probability matrices in a SDP model. This is however described as a minor issue, if the travel time in a reach is considerably smaller than the time required for treatment process modification. It is also to be noted that the discretization used for the state variables DO and BOD were 9 and 24 intervals respectively. This gave rise to 216 possible water quality states. Thus, for each water quality state and a control action at one stage, there can be upto 216 feasible states at the next stage. Most of these transitions will occur very seldom, or never, but there is a probability that it can happen. The main problem with the present approach is the number of Monte Carlo simulations that would be needed to estimate these probability matrices accurately. Even if the probability matrices were determined accurately enough, subsequent computational load in the SDP model will be huge. This is mainly due to the fact that all the state transitions, with the associated transitional probabilities, have to be considered in the SDP optimization. The reported application included six river reaches and five point sources only.

6.4 Simulation Models

Warren and Bewtra (1974) developed a deterministic simulation model to analyze the effects of discharges from combined sewer drainage areas on receiving water bodies. The model examined the BOD, total suspended solids, total phosphate, and coliform concentration changes resulting from alterations in the capacities of treatment facilities and interceptor sewers in three drainage area types (residential, industrial and agricultural) in a typical community in Ontario, Canada. Results indicated that combined wastewater retention tanks are economically feasible solutions to combined sewer outflow problems. This was due to the sedimentation effects in storage and subsequent treatment, which produces a superior final effluent.

Simulation models were developed by Tapp (1976) to predict the instream effects of storing wastewater during the periods of low streamflow and releasing the treated effluent during periods of high streamflows. Allowable wastewater flows were expressed as functions of streamflow, temperature, and effluent quality for use by the operators of wastewater storage ponds.

A preliminary analysis of the problem of designing an adequate, and possibly automatic, control system for maintaining pollution levels within reasonable bounds, was presented by Young et al. (1976). A discussion on the derivation of a simple dynamic DO-BOD model for rivers was also included. This study raised the following two important points, among some others. (1) Simple lumped parameter dynamic models appear to provide an adequate description of the DO-BOD balance in a non-tidal river system, and (2) The implementation of local control of effluent treatment facilities requires the development of flexible methods of regulating the effluent water quality. This is also possible through the approaches such as detention and controlled discharge of effluent.

Boner and Furland (1982) presented a study that addresses the issue of seasonally variable effluent quality based on assimilative capacity. The Chattahoochee River in at Atlanta was considered for the analysis. The necessity for a variable operation arose from the high cost required to perform year-round nitrification treatment, which had been based on the analyses done on critical conditions. The major objective of the study was to determine the cost-effective treatments for two treatment plants. By considering various levels of treatment at each plant, a matrix of possible combinations were analyzed. Seasonal water quality modelling was performed on a monthly basis.

A state-of-the-art review of waste load allocation simulation models has been presented by Ambrose et al. (1988). Steady-state as well as dynamic modelling approaches are described. The additional levels of detail needed in modelling toxic chemicals are examined, in contrast to the conventional pollutants such as BOD. It is pointed out that the near-field mixing zone and the far-field transport problems must be addressed to cover the range of pollutant impacts, both acute and chronic.

6.5 Combined or Miscellaneous Approaches

An investigation of regional management of organic pollution was presented by Anderson and Day (1968). The study employed linear programming and simulation techniques to determine the minimal regional operating costs of conventional waste treatment (BOD removal), without violating river quality standards (DO levels). Five different regional treatment policies have been proposed and evaluated. The operation according to each policy have been simulated for 400 years using synthetically generated hydrologic data. The results of these simulations were used to obtain monthly operating costs, average annual cost and, the percentage of time the quality standards were violated. It is to be noted that several assumptions with regard to both engineering and economic aspects have been made in the study. A modified form of the S-P equation has been employed. The major economic assumption has been that the operating cost of a conventional waste treatment plant is a linear function of the degree of treatment involved.

A probabilistic approach to stream quality management was presented by Burges and Lettenmaier (1975). A first-order uncertainty analysis was used to obtain an estimate of the uncertainty in a deterministic model due to uncertain parameters. The method is applied to the simplified S-P equations for estimating DO and BOD. A more descriptive Monte Carlo simulation was employed to check the accuracy of the results. The first-order analysis has given accurate estimates of means and variances of DO and BOD upto travel times exceeding the critical time. Uncertainty in travel time and the BOD decay constant were found to be the most important for small travel times. Uncertainty in the reaeration coefficient dominated near the critical time, while temperature uncertainty has been found to be negligible.

Whitlatch and ReVelle (1976) used a sequential procedure incorporating LP, DP and heuristic location techniques to estimate the least-cost number, location, and degree of treatment of regional domestic wastewater plants along an estuary or river. Illustrations were presented for the Delaware estuary having 44 domestic and industrial waste sources and 9 potential regional wastewater treatment plants.

Biswas (1981) presented an analysis based on LP and simulation, for the Saint John River in Canada. The LP model was formulated to minimize a linearized cost function or maximize a water quality indicator (weighted sum of DO levels at various locations). Only BOD and DO were considered for the analysis, and they were computed based on the S-P equations. The optimal solutions (treatment levels) identified with this model were input to the simulation model which was run through a series of consecutive low flow seasons of synthetic daily flows. Two-day and Three-day time intervals were available as optional time intervals. This simulation model provided a more accurate representation of the magnitude, frequency, and the persistence of various levels of water quality; than the optimization model. Only conventional pollution abatement methods have been considered for the analysis. Furthermore, relatively complex cost functions were derived from only three or four data points, and their accuracy is questionable.

A combined optimization and simulation approach is presented by deLucia and McBain (1981). The nonlinear optimization model (separable programming) minimized the present value of costs over a multiple period time horizon. Provision was made in the model for inclusion of various equity and budgetary constraints. BOD, DO, N, and P are modelled by the simulation model. In addition to these quality indicators, an additional indicator, biomass potential (estimated from conventional water quality criteria), was considered. It indicates the extent to which substances in the waste stream distort the biological activity of stream, beyond natural levels.

In the optimization model, the costs were expressed as a separable function of biomass potential removal and capacity. Determination of the degree of treatment and capacity expansion simultaneously (by the use of certain heuristics) was possible due to specific planning requirements which had to be satisfied.

Monte-carlo simulation together with an optimization (for each scenario generated with the simulation) can be employed to account for the probabilistic nature of a system. However there is little practical applicability of such an approach, when considering the myriad of possible scenarios. Nonetheless, Burn (1989) presented an approach for water quality management, based on monte-carlo simulation and optimization. In that approach, a monte-carlo simulation model was used to generate a series of water quality responses that lead to a set of constraints for allowable pollutant discharges. These constraints ensure that the prespecified water quality standards at certain locations of the river are satisfied. A repeated solution of an integer programming optimization was done in order to determine the optimal set of treatment alternatives corresponding to each constraint set generated. The resulting optimal treatment costs as well as the treatment levels at each site were interpreted as cumulative probability distributions from which a decision maker was supposed to identify the ultimate solution. These probability distributions represent the trade-off curve between total cost of treatment and the probability level that can be used to reflect the risk aversion of the decision maker.

Multiobjective Approaches

Monarchi et al (1973) presented the first research paper on multiobjective water quality management. The sequential multiobjective problem-solving technique (SEMOPS), which was developed for this purpose, was applied to a hypothetical case study of regional quality management of Bow River Valley. SEMOPS allows the decision maker to trade-off one objective versus another in an interactive manner. It cyclically uses a surrogate objective function based on goals and the decision makers aspirations about achieving these goals.

Transferable Discharge Permit (TDP) Programs

The applicability of transferable discharge permit (TDP) programs for river basin water quality management has been described by several authors. Eheart (1980) described a TDP program based on permits defined in terms of allowable BOD (kg/d) discharges. Such permits can be initially distributed to the polluters by auction or without charge, using an allocation formula. Subsequently the permits are allowed to be exchanged subject to various specifications. This allows the dischargers with high waste-treatment costs to purchase permits from those having low costs. Also the incentives for innovative treatment techniques increases, as the unneeded permits can be sold.

However, resulting changes of the locations of discharges not only change the location of impact, but also affects the degree of impact. Therefore an evaluation of the resulting impacts has to be done beforehand, considering the desired water quality standards at various locations. Certain restrictions on permit-exchanges are to be imposed, as unrestricted exchanges may lead to severe degradations of water quality.

Brill et al. (1984) described a framework for evaluating water quality impacts of TDP programs.

The trade-offs among cost efficiency, equity and uncertainty are illustrated with respect to meeting a given water quality standard. Examples were given for the Delaware River estuary and the Willamette River. It was shown that restrictions on the market could be used to control standard violations which may occur under a TDP program. Limits on the total discharge in sections of the basin, zone boundaries for markets, and revaluation factors for transferred permits were shown to be effective individually or in various combinations.

The large capital expenditures needed to construct waste-treatment units is one major factor that makes the applicability of TDP programs, which assume the possibility to shut them down, unjustifiable. It is also complicated by the multiple-pollutants character of discharges. Pollutants can be managed on an aggregate basis, if the marginal damage of each pollutant is the same at all concentration levels and at all locations, which is rarely the case. On the other hand, individual pollutant management programs require a large number of permit markets. However, they gives administrators greater control over each pollutant, and allows dischargers more flexibility than an aggregated permit program. Lence et al (1988) described an approach for evaluating the cost efficiency of an individual-pollutant permit program in relation to BOD, P, and N discharges.

7 Example Problem

As the review implies, dynamic programming is an attractive methodology which can exploit the sequential characteristics of river basins (downstream control actions do not influence water quality upstream). DP handles discrete decision variables which represent different treatment levels, and it is generic in the sense that both linear and non-linear water quality models expressing the relation between emissions and ambient water quality can be incorporated. Due to these favorable attributes of DP, it was selected to provide an illustrative example of its usage.

The example problem is to formulate least-cost water quality management strategies for the Nitra River Basin in Slovakia. Nitra River is a tributary of the Váh which enters the Danube downstream to Bratislava, where the Danube forms the border between Slovakia and Hungary. The length of the river is about 170 km. The mean streamflow near the mouth is 24 m^3/s, while a typical August low flow condition is characterized by not more than 3 m^3/s. The watershed area is 5100 square km and 650 000 inhabitants live there (about 40% of them live in rural areas). The region is highly industrialized. Metallurgical works, meat and sugar beat factories, tanneries, chemical plants etc. can be found at various locations. Level of water re-cycling and industrial wastewater treatment is low. Agricultural non-point source pollution is negligible. Sewage networks and wastewater treatment plants can be found in eleven larger municipalities (with a total population of 350 000). The level of public water supply as well as sewerage in these municipalities is close to or above 90%. (however in rural settlements, only a small portion of the population is supplied by potable water, and wastewater treatment is practically non-existant). The level of wastewater treatment is below 50%. Treatment plants, most of which based on the activated sludge process without nitrification, are about 20 years old. Several plants are overloaded by 100% or more, leading to BOD removal rates between 60% and 70%. The excess flow above the design capacity is generally given only primary treatment. A map of the watershed illustrating the levels of municipal wastewater treatment is presented in Fig.1.

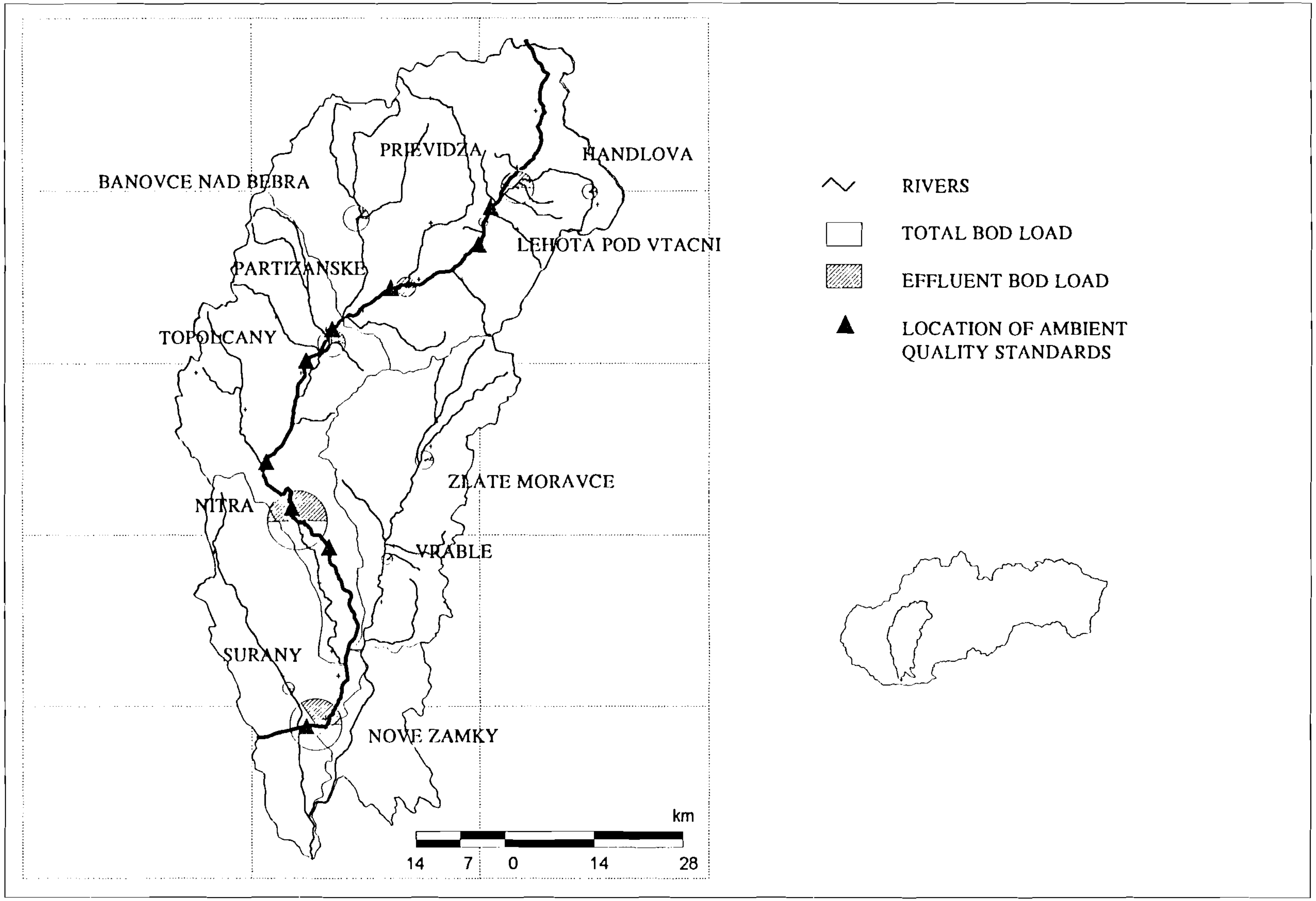

Fig. 1 Municipal wastewater treatment in the Nitra River Basin

The water quality of the river is one of the poorest in Slovakia. Parameters of the oxygen regime and other chemical components specify Class IV-V quality, according to the existing classification system (V indicates the worst quality). For instance, BOD values can exceed 30 mg/l, and dissolved oxygen (DO) is sometimes below 1-2mg/l. Total phosphorus (TP) can reach 2 mg/l, while ammonia nitrogen (NH4-N) can be around 7 mg/l. High algae biomass levels can be observed at the downstream stretch of the river (a-chlorophyll values of 150 mg/m^3 are not rare), where travel time is sufficiently long.

Municipalities contribute about 70% of the total emission of traditional pollutants (organic material, phosphorus and nitrogen) of which 30-50% is due to industrial discharges (see Figs. 1 and 2). Industrial contamination is characterized, among others, by high arsenic concentrations (the origin of which is yet unknown) and high conductivity.

At present, the primary utilization of the Nitra River is waste disposal (as apparent from what was outlined above) and water abstraction for industrial and irrigation purposes. For the latter purpose, small dams have been built, mostly at the downstream part of the river.

The economy of the country and the region is in a strong transition. In spite of increasing water prices, domestic water consumption has not yet been reduced significantly. The same statement also applies for industries connected to municipal wastewater treatment plants (WWTPs).

Thus, the flow and load of WWTPs have been reduced only slightly during the past three years. In contrary, larger industrial plants outside of urban areas went through significant changes. A large portion of them were fully shut down. Some others show about 50% discharge reduction and only a few operate in an unchanged fashion. In the short run, this trend is likely to be continued, while longer term alterations are hard to visualize.

A new standard system was set in 1992 by the earlier Czech and Slovak Republics (and adopted by the Slovak Republic in 1993) as a basis for future legislation. The system is based on effluent standards, but it leaves open the possibilities to also incorporate ambient water quality criteria. The procedure realistically distinguished two stages - before and after 2004. For a 100 000 population-equivalents or larger municipality the standards are as follows: BOD=30 mg/l, NH4-N=10 mg/l and TP=3 mg/l. The corresponding limits are slightly more stringent as of 2005.

Under the present economic conditions, financial resources are not available for major environmental investments. In spite of that, the consequences of specifying the above effluent criteria, which demand major investments, have not been analyzed. Thus, the issue of developing a least-cost policy (based on ambient quality criteria) on the short run was raised.

The development and discussion of this strategy (together with effluent standards based ones and mixed ones) is the major objective here, specifically for the Nitra watershed. The work to be outlined is an element of a comprehensive, policy oriented research program with the involvement of the Water Research Center (Bratislava) and the Váh River Basin Authority (see Somlyódy et al., 1994; for details). The other goals and components of the study include: a survey of various emissions, the development of wastewater treatment alternatives and upgrading strategies, longitudinal water quality profile measurements and their evaluation (see Masliev and Somlyódy, 1993), and the application of various water quality simulation models (from that of Streeter and Phelps to QUAL2e; see Breithaupt and Somlyódy (1994) for the latter) to be used for policy purposes. The results presented herein include different control strategies developed with a focus on municipal emissions and oxygen household. The role of industrial emissions is demonstrated in a sensitivity fashion.

DP Formulation

The river system was subdivided into a number of reaches that were further divided into "stages". The river network has been defined by the interconnection of different reaches. A "stage" was considered as a part of the river from a point immediately upstream of a "point of action" to a point immediately upstream of the next "point of action" downstream. A "point of action" can be: a wastewater discharge, an abstraction point, a measurement point, a point with a prespecified water quality standard, a weir or one of the artificial points introduced to maintain a generalized computational procedure.

The computations of DP were started from the most upstream point of the river system. Using Bellman (1957)'s principle of optimality, the DP calculations were performed stage by stage, proceeding towards the most downstream point of the system. A description of the DP computations is given below, for a simplified case of a river without any tributaries. An extended form of this procedure was employed to analyze the Nitra system comprising of several tributaries. The optimization problem for the most upstream stage can be expressed as:

$$f_1(Q_1) = \min_{\eta_1} C_1(\eta_1)$$
$$Q_1 = T_1(\eta_1, Q_0)$$

where, $f_n(Q_n)$ (n=1 for the most upstream stage) is the cumulative optimal cost required to achieve the allowable water quality state Q_n at the end point of stage n. $C_n(\eta_n)$ is the treatment cost required (which is zero if there is no wastewater discharge or if no treatment is to be made at stage n) to achieve an efficiency of η_n at stage n. This efficiency η_n (the removal rate) is the decision variable at stage n of the problem. For the most upstream stage, Q_0 indicates the headwater quality. $T_n(\ )$ represents the transfer function at stage n, and is a water quality simulation model. The treatment cost considered in this study was the total annual cost (TAC); which is comprised of the contributions of investment cost (IC) and the operation, maintenance, and replacement cost (OMRC).

The efficiency of a particular treatment alternative is characterized by the influent and effluent quality. For each WWTP, a set of feasible treatment alternatives were developed. Each alternative is represented by its IC, OMRC, and the effluent concentrations (see below). The optimization task is to make a (0,1) decision with regard to each feasible treatment alternative at each of the WWTPs. These decisions should sum-up to one at each WWTP location.

For the subsequent stages (from n=2 upto the most downstream stage), the recursive relation takes the following form:

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

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

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

Design Condition

Results of an experiment performed in August 1992 (Masliev et al., 1994) under critically low flow conditions served as the basis for the single design scenario (including the derivation of parameters of the DO model). Background pollution of the tributaries and the temperature profile (showing an increase downstream from 19°C to about 25°C) were obtained from historical observations. For the generation of design emissions for municipalities, annual average values available for the period 1990-1992 were used in addition to measured ones. Industrial emissions were kept at their 1990 level, mainly as a worst-case scenario and to demonstrate the role of changing industrial emissions (as observed recently) by a sensitivity study. The optimal policies to be developed will focus on municipal emissions. Industry will be considered without analyzing the costs, due to lack of reliable information on industrial discharge control measures.

Quality Indicators Selected for Setting of Standards

The present analysis relies mainly on ambient DO level which is the key element in maintaining the balance of the river ecosystem. Without proper DO concentrations, it is not possible to improve the water quality in terms of the other indicators. The pollutant levels measured by BOD and NH4-N also play an important role in this respect, and they are considered by the existing legislation in Slovakia. Therefore the policy decisions of this study will be based on these three quality indicators, out of which ambient DO is considered as the most important one. The analysis was not limited to a single set of quality standards. Instead, the economic impacts of setting various quality standards were estimated.

Location of Quality Standards

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

Economic Objectives

The benefits of water quality management are rather intangible. In many cases, the objectives of management exercises are expressed in terms of costs, which was the case for this study as well. From the point of view of the costs, two different types of objectives were considered separately. The first was to formulate a policy with a minimum total annual cost (TAC) of waste treatment. The total annual cost comprises the operation, maintenance and replacement costs (OMRC) and the annual repayment of the investment cost. The second goal was to find a policy which requires the minimal basin-wide investment cost (IC), which stem from the tight budgeting situation in CEE countries including Slovakia.

Industrial Discharge Reductions

As indicated above, the industrial discharge control was not an explicit part of this policy analysis. The effects of the possible discharge reductions were estimated in a sensitivity fashion, by considering three different industrial discharge scenarios (in harmony with structural changes observed nowadays). The three scenarios considered were the discharges in 1990 and reduced discharge conditions corresponding to 50% and 75% reductions of 1990 amounts.

Results

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

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

The implications on river water quality, of implementing these strategies, are summarized in Table 2 together with the associated costs and treatment configurations. The minimum and maximum values of water quality indicators of Table 2 display the corresponding worst water quality in the river. Treatment configurations are presented with the treatment level numbers selected at each discharge location. A "0" indicates that there is no treatment. The treatment level numbers corresponding to the BAT strategy indicate the highest treatment level available for a particular discharge. They are indicative of the number of treatment alternatives available at each discharge point. It is also to be noted that the treatment level numbers of different discharge points have no relation to each other. For example, the treatment level 8 at Topolcany (To) and the level 4 at Handlova (Ha) both refer to the same type of treatment technology. (see

Somlyódy et al., 1993; for details).

The following abbreviations for the names of municipal discharges have been used in Table 2 and in the subsequent tables. (see Fig. 1 for the locations).

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

Table 2 shows that the feasible ranges of water quality are relatively narrow. The maximum BOD levels corresponding to no treatment and the best available treatment are 35 (very poor quality) and 11 mg/l respectively. However the corresponding difference in cost is 95.5 million USD. The reason for the high BOD that occurs even with the BAT (violating the standards of legislation), is the industrial and non-controllable discharges which were assumed to be unchanged. The same reason applies for the minimum DO level of 5.7 mg/l observed with the BAT strategy.

Table 2 Performances of the Current, BAT and Slovakian Effluent Standard Based Treatment Strategies

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

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

Table 2 also indicates that the present and future effluent standards of Slovakia, in spite of their cost difference, are identical from the view point of ambient water quality.

The setting of ambient quality standards opens up the possibilities to formulate least-cost policies. Table 3 displays several strategies that were obtained by specifying DO standards between 3 mg/l and 6 mg/l (without setting limits to the minimum treatment level). The "cost" in this case was taken as the total annual cost (TAC). Table 3 shows that substantial improvements are attainable without incurring heavy investments (in contrast to Table 2). The IC shows an exponential increase (see Fig. 2) with tightening minimum DO level. The performance of the strategy corresponding to DO≥6 differs only slightly from the BAT strategy of Table 2. On the contrary, the difference in the investment costs of those two alternatives is 62 million USD. The quality improvements under least-cost policies is further illustrated in Fig. 3, in which longitudinal DO profiles along the Nitra River are displayed. These DO profiles correspond to those resulting from current treatment, Slovakian effluent standards (present) based strategy, BAT, and the least-cost policies obtained with DO standards of 3 and 5 respectively.

Table 3 Least-Cost Treatment Strategies Obtained by Specifying Ambient DO Standards

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

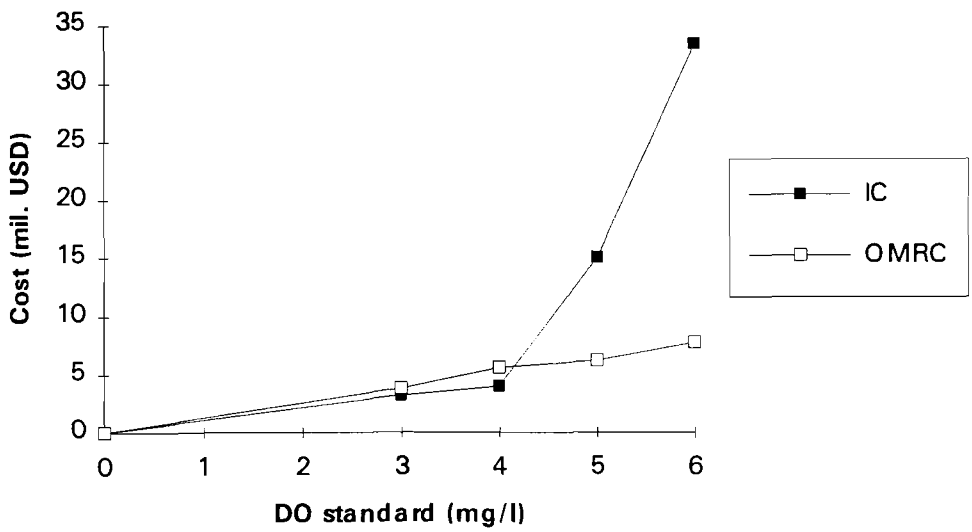

Fig. 2 Cost implications of setting different ambient quality standards

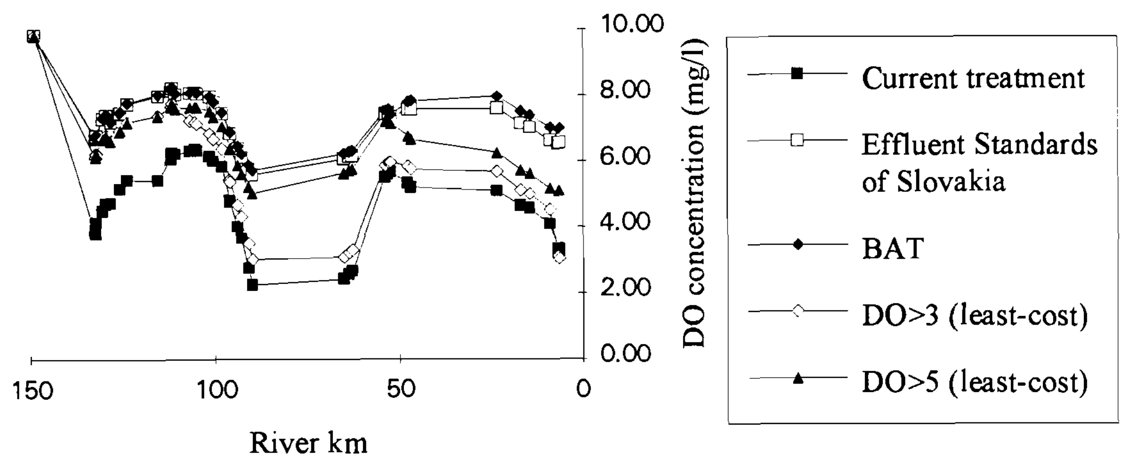

Fig. 3 Longitudinal profiles of DO concentration for five different treatment strategies

The effect of setting standards on DO and NH4-N, and the impact of industrial emission control on cost of municipal treatment, is displayed graphically in Fig. 4. It summarizes the least-costs required to maintain DO and NH4-N ambient standards for the Nitra River basin under three different industrial discharge scenarios. The least-cost strategies formulated under reduced discharge conditions suggest cost "savings" where the actual saving obviously depends on expenditures of the industrial control. The issue clearly calls for the development of an integrated least-cost strategy covering all the discharges of different origins.

Conclusions and Recommendations on the Usage of DP

The DP approach was found to be a generic methodology for river basin management modelling purposes. Although it possesses all of the favorable attributes mentioned above, there are certain critical issues which require further attention to make a successful use of the technique. The first issue is the huge increase in the computational load, which results as the number of state variables (with a reasonable discretization) increases. A substantial reduction of computations is possible by manipulating the state variable (quality indicator) ranges and the number of discrete intervals considered for each of them.

In the present study, uniform ranges of water quality indicators were considered for the DP computations. However the feasible quality range at different locations along the river can be significantly different. A reduction of computations or an improvement in the accuracy of the results can be obtained by considering location-specific ranges instead of uniform ones. If the range is narrowed; an unchanged interval-size reduces computations, whereas maintaining the same number of intervals improve the accuracy.

An equal number of feasible "quality intervals" or "quality states" was considered at each stage of computations in the present study. Nonetheless, it is advantageous to allow the possibility to discretize the quality range into a different number of intervals at different places. This allows to use a rough discretization at the locations with few feasible quality states. Significant reductions in computational load can be realized if it is possible to have a non-uniform discretization, combined with non-uniform quality ranges.

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

8 Scheduling Problem - A General Description

The example problem, and most of the other management models discussed above, are for static planning situations, such as for a particular year. Waste discharges, quality standards, etc are considered fixed in deterministic models, or, their uncertainty (but not the future trends) is included in stochastic analyses. However, with population growth, and social, industrial as well

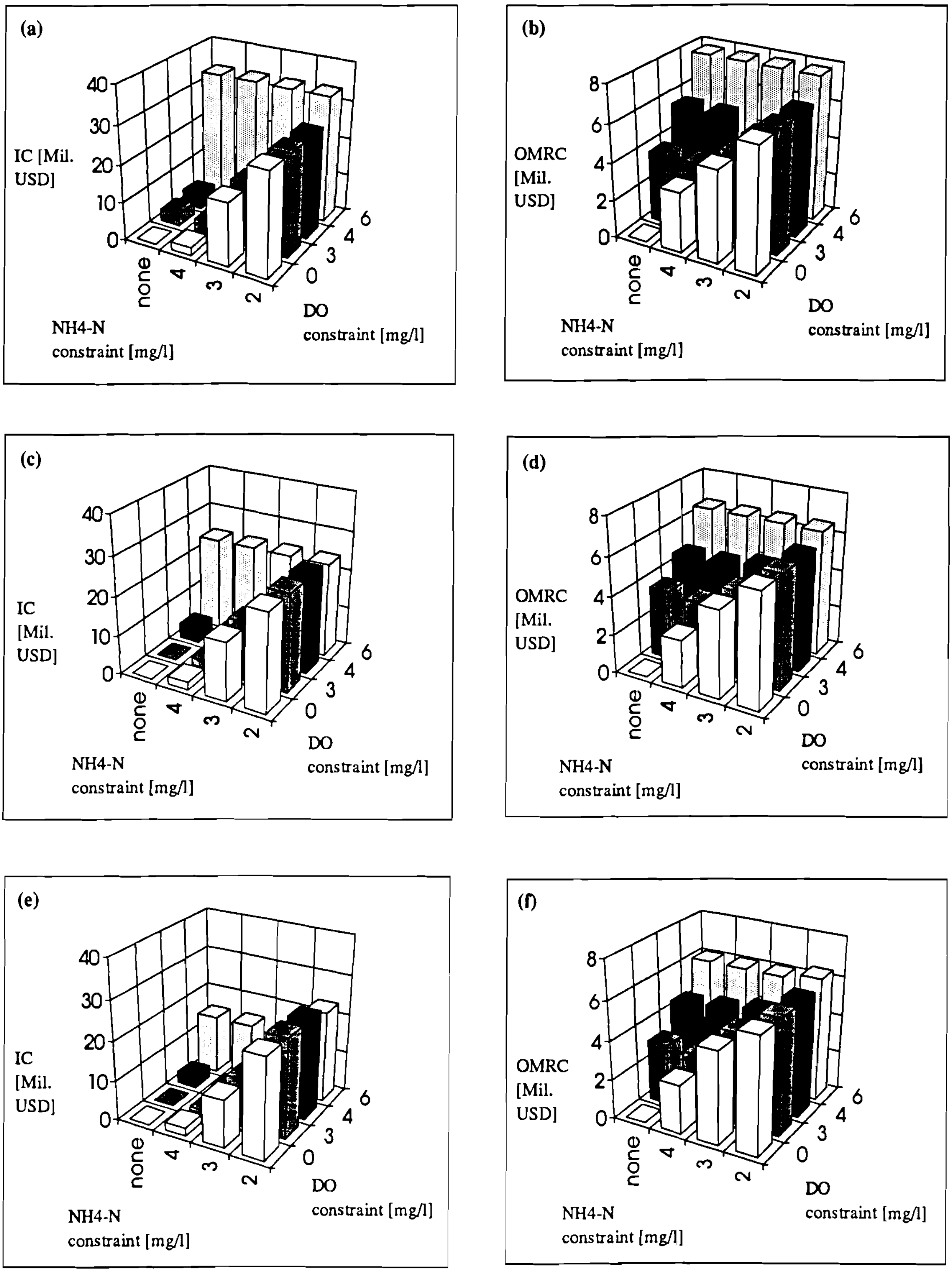

Fig. 4 Cost implications of setting DO and NH4-N standards under different industrial discharge scenarios. (a),(b) IC and OMRC respectively, corresponding to 1990 discharges; (c),(d) IC and OMRC respectively, for a 50% reduction of ind: discharges; (e),(f) IC and OMRC respectively, for a 75% reduction of ind: discharges

as climatic changes, many of the assumptions regarding fixed planning situations becomes invalid. Therefore, instead of static planning models, dynamic (scheduling) models are required to formulate long-term management policies which aim at stage-wise implementation of the plans. Long-run variations in the available financial resources further strengthen the need to use such models.

The amount of information needed for such models would be much larger than the static counterparts. Since even the static planning models are burdened with excessive computational loads, a scheduling approach would require relatively coarse descriptions of the system. This implies that a detailed posteriori sensitivity analysis is compulsory to justify the long-term management plans obtained with scheduling models.

Integer programming and dynamic programming are the two basic approaches used for solving of scheduling problems (Loucks et al., 1981). Assuming a hypothetical situation in which relevant information for each year during the planning horizon are known with certainty, a DP-based approach offers a feasible solution method (see Klemetson and Grenney, 1978; for details). The first step of such an approach involves the generation of a number of different strategies for each year separately, which is the standard approach described in the previous DP-based models. This optimization process is performed through decision stages in space. Subsequently, these strategies can be jointly considered within another step of optimization, which is done considering the entire planning horizon. The decision stages in this case are in time (years in the future). In this stage, information with regard to the possible improvements in treatment methods will have to be provided, presumably in the form of constraints.

The computational requirements of the above approach will be excessive, even under deterministic assumptions. Incorporation of uncertain future conditions with sufficient detail under such situations, is nearly impossible. This necessitates a more selective analysis than the consideration of each feasible situation, and a scenario-based analysis would be the most suitable in this sense. The essence of the scheduling problem would therefore be in selecting a limited number of appropriate scenarios and parameters.

9 Concluding Remarks

Water quality management of a river basin requires an evaluation of many facets of the problem, which arise due to various hydrological, chemical and biological processes, types and quantities of waste discharges, and social, economic and political considerations. Uncertain future situations add another dimension to the problem. Models are needed to facilitate the analysis of available management options, and, the complexity and completeness of the analysis depends on its purpose. There are two basic categories to which required models belong: water quality simulation models and management models. Unavailability of data often influence the selection of a water quality model, in which case a simple model which has a low input requirement is desirable. Complex models are usually associated with greater input data requirements.

Management analyses can be performed using simulation, optimization or both, depending on the management goal and the problem size and type. The critical issues in a management model formulation are the nonlinearities, uncertainties, multiple pollutant problems, multiple objective nature of problems, and the spatial and temporal distribution of management actions.

Whichever the technique used, it would rarely represent all aspects of the problem in a single model. As an example, a deterministic optimization model may include fairly detailed quality simulation models, while uncertainty issues are neglected. On the other hand, a stochastic model which incorporate uncertainty issues, would need rather simplified quality models. Such trade-offs are common in management models, and the only way to compensate for such deficiencies, is to perform the analysis within a sensitivity framework.

The review imply the suitability of dynamic programming as an optimization tool for water quality problems. Its applicability is basically due to the ability of incorporating nonlinear model formulations and discrete decisions efficiently. However its main drawback is the inability to consider a large number of water quality indicators. Although uncertainty issues can be handled with DP, such analyses will require a heavy computational load, which is the case with many other techniques as well. An appropriate method for including uncertainty would be to perform a scenario analysis using a selected set of scenarios. Sensitivity of the obtained solution to alternative scenarios is again an important aspect that must be evaluated. An alternative approach is to formulate a stochastic DP model based on the expectation and the variances of uncertain parameters. This, however, imposes severe restrictions on the number of uncertain parameters that could be included in the analysis.

The fact that water quality management problems are rather loosely formulated, further suggests the requirement for a sensitivity style analysis. There can be situations where water quality, and even the economic goals, are open to appropriate changes. Evaluation of such options require an integrated modelling framework which include water quality models, optimization ones as well as multiobjective decision making techniques. Interactive models, which can be categorized more specifically as decision support systems will be the future trend in this direction.

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