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Somlyody, L. and Paulsen, C.M.

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C.M. Paulsen

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International Institute for Applied Systems Analysis □ A-2361 Laxenburg Austria

Telephone: +43 2236 715210 □ Telex: 079137 iiasa a □ Telefax: +43 2236 71313

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PREFACE

Among the many challenges facing Central and Eastern Europe is the problem of improving the quality of the region's rivers, reservoirs and lakes to acceptable levels. While industrial waste-water discharges and loads of agricultural origin can be expected to decline as a result of economic re-structuring, municipal emissions will probably increase over time as more urban areas are added to sewerage collection systems. This paper addresses in detail how research can be designed to meet ambient water quality standards cost-effectively, through the use of alternative treatment technologies, water quality models and optimization techniques.

ABSTRACT

Many countries in Central and Eastern Europe will be formulating new environmental regulations within the next few years. Among the many topics which these are likely to address is the development of control policies for waste-water dischargers, including municipal sewage treatment plants. In Western Europe and North America, standards have relied heavily upon so-called "best available technology" control policies, which require dischargers to use treatment processes that reduce emissions of BOD, phosphorus, and nitrogen as much as is technically feasible. However, these technologies are often very expensive. Given the state of Central and Eastern European economies, less expensive methods to improve water quality should be seriously considered.

In this paper, we investigate control policies, alternative sewage treatment possibilities, water quality models, and optimization methods required to identify least-cost strategies to improve the region's ambient water quality. We survey the costs and technical capacities of a variety of treatment techniques, ranging from simple primary or mechanical treatment to advanced technology to remove nutrients. We also survey existing water quality models and show how they can be adapted to the policy analysis problem. Finally, we characterize a number of potential policies in terms that are amenable to analysis of their costs and ambient quality impacts. Focussing on municipal waste-water treatment plants and water quality in rivers and streams, we show how these techniques can be integrated and applied. We conclude with an empirical example based on the Nitra, a small, heavily contaminated river in Slovakia.

COST-EFFECTIVE WATER QUALITY MANAGEMENT STRATEGIES IN CENTRAL AND EASTERN EUROPE

L. Somlyódy[1]
C.M. Paulsen[2]

1. INTRODUCTION

The poor quality of surface and groundwater resources in Central and Eastern Europe (CEE) has been documented extensively in the technical and popular press since the political changes which occurred in the region during the past three years (Golitsyn, 1992, Hughes, 1992, Somlyódy, 1991 and 1992). In addition, many analyses have shown that the cost of cleaning up the region's water quality problems is likely to be enormous, especially when viewed relative to the size of CEE national economies.

Some very rough calculations serve to suggest the magnitude of costs of handling the problem of municipal wastewater discharges. Poland, the Czech and Slovak Federal Republic (CSFR), and Hungary have a total population of approximately 65 million, of which about 40 million live in urban areas. Capital or investment costs to place a large portion of the urban population on public water supply and biological wastewater treatment (say 95% and 70%, respectively) are on the order of USD $20-50 billion, depending on a number of factors, such as the level of the removal of biological oxygen demand (BOD) and nutrients, pre-treatment of industrial discharges, and reconstruction of aged infrastructure. We note that effluent standards recommended by the European Community (EC) would result in costs at the upper end of the range. These figures should be viewed with some skepticism in light of incomplete, poor quality data on emissions, water consumption, and existing infrastructure, and the very preliminary nature of control cost estimates. However, it is clear that control costs are likely to be beyond the reach of many countries in the region for a decade or more.

In addition to the high costs of treatment facilities in CEE[3] countries, the resources potentially available to pay for them have declined markedly in recent years. For example, industrial output in Hungary declined by one percent in 1989, ten percent in 1990, and 19 percent in 1991 (ECE, 1992 and International Financial Statistics, 1992). There is also an understandable reluctance on the part of many CEE governments to take on additional long-term foreign debt to pay for environmental improvement, given the many other pressing

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[1]On leave of absence from the Water Resources Research Center, VITUKI, Budapest (also professor of the Budapest University of Technology).

[2]Research fellow, Resources for the Future, RFF, Washington.

[3]Solely for reasons of data availability, our focus in the course of the current research will be on Poland, the CSFR, Hungary, and Bulgaria. We expect that methods developed for these countries should be broadly applicable to other CEE (and CIS, Commonwealth of Independent States) countries, depending upon the extent of economic changes in each country.

needs in the region, including industrial re-structuring, modernizing transportation and communication infrastructure, and improvements in agricultural practices. Despite these problems, there is strong pressure within CEE governments from international organizations, and from some professionals to adopt European Community (EC) technology-based standards.[4]

The movement toward expensive pollution-control policies seems to us to have at least four broad causes. The first is the obvious failure of past policies. Those policies were based in part on substantial central-government subsidization of many industries, including artificially low prices for inputs (e.g., capital, energy and raw materials), guaranteed markets for products, and very low fees and fines for exceeding emission standards, such that the fines had essentially no effect on plant-level discharges. Second, many governments in the region are under substantial pressure to put an environmental policy of some kind into place quickly. This is perfectly understandable, given the atrocious conditions which prevail in some regions. It has led to a move toward EC standards, since they are already well-defined and understood, rather than to preparation of environmental strategies consistent with the region's lack of financial resources. In addition, if EC standards could in fact be imposed successfully it is obvious, without the need for any analysis whatsoever, that regional environmental quality would be substantially improved. Finally, to date there has been almost no analysis of the trade-offs between capital investment, treatment costs, and ambient water quality. Therefore, it is impossible to say at present what the probable consequences of any planning or investment strategy is likely to be, beyond making the obvious point that imposing Western European standards on CEE dischargers will be immensely expensive.

Unfortunately, however, the region cannot afford the price of meeting Western-style standards in the short run (i.e., the next decade or so) without imposing appalling costs on other sectors of the economy. For example, in Hungary the capital cost of bringing public water to smaller urban areas, extending the collection network and installing secondary sewage treatment is estimated at USD 5-10 billion, depending on the level of development. However, the amount budgeted for 1993 is about USD 200 million, or 2-4 percent of the required sum. Absent a better way of doing things, it seems to us that the end result will be wasted investments in environmental projects unlikely to produce appreciable improvements in environmental quality.[5]

Unique changes occurring in the region call for a unique planning process. If existing tools and techniques can be brought to bear on this process in a meaningful way, we believe that there are real opportunities for substantial improvements in CEE environmental quality at prices which are affordable for the region's economies.

On the environmental policy-making side, this will require several things. The first is a willingness to treat the restructuring of the region's economic and political base as an

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[4]The standards proposed are 25 mg/l for BOD, 10 mg/l for total nitrogen, and 1 mg/l of total phosphorus for urban discharges above 100,000 population equivalent, far below current effluent concentrations at most treatment plants in the region, especially for nutrients.

[5]This result, should it occur, would not of course be unique to CEE. One example of sizable investments producing little environmental quality improvement includes the U.S. Superfund program.

opportunity to simultaneously enhance the region's environment. The second is a recognition that, because of severe constraints on available resources, **innovative policies** will be required in the short run that will likely be very different than those that have been applied in the EC and elsewhere. These policies will probably include the clever application of cost-effectiveness, meaningful economic incentives to encourage efficient behavior by dischargers, and the use of flexible, innovative, low-cost treatment technologies. The third requirement is the acknowledgement that, absent a rapid and sustained improvement in the region's economy (which seems unlikely at present) it will probably require several decades for CEE environmental quality to approach that of the EC. While perhaps disappointing in some respects, taken together these points suggest that CEE governments should utilize this opportunity to learn from the mistakes made in the West during the past several decades, rather than trying to imitate those errors.[6] Clearly, these all entail some risks for part of policy-makers. However, we believe that there will be **substantial benefits, in terms of real improvements in environmental quality in the near-term at costs the region can afford, and matching EC ambient quality levels at reasonable costs in the long run.** Indeed, to the extent that CEE countries can eventually meet "internationally accepted" ambient quality standards more cost-effectively than Western European nations, this could give them a long-run competitive advantage in international markets. Section 5 and Appendix A suggest the type of policies which may be useful, including economic incentives and **flexible, cost-effective technology-based standards.**

On the research side, several innovations will also be required. First, techniques of systems analysis will be needed to integrate methods and results from several different disciplines (see Section 2 for more details). Second, substantial efforts must be devoted to making methods, data, and results available to analysts and policy-makers within CEE, in the form of workshops, training, and installation of and extensive support for computer models. Finally, the methods, data, and so forth must be adapted to each country's institutions and traditions to be useful to those who will apply the tools in planning and (eventually) implementation. Ideally, of course, the customization would be performed by in-country "graduates" of the training programs; how this would work in practice remains to be seen.

In this paper, **we focus on municipal point-sources**, for several reasons. First, industrial activity, and hence industrial discharges, are almost impossible to predict given the upheaval in the region's economy. For example, industrial output in Hungary and the CSFR has declined by more than 25 percent since 1989. Meanwhile, as an example, biological oxygen demand in the Sajó (which is located in a heavily industrialized region on the border between the two countries) has improved by more than 50 percent since the late 1980's, as a direct result of the closure of industrial plants.[7] It is entirely possible that whole industrial sectors

[6]Examples (again from the U.S.) include the applying so-called best available technology to all point sources (even though regional ambient environmental quality may be perfectly acceptable), requiring expensive emission controls for new cars (when older cars cause far more pollution per kilometer driven) and placing much stricter controls on new point sources of airborne emissions than on existing sources (which has probably resulted in poorer air quality, since old plants are not replaced as quickly).

[7]Obviously, as privatization of industrial plants continues in the region, there will be problems with newly privatized plants being run for short-term profits and no regard for existing environmental laws under re-formulation. However, recent overall trends in water quality for heavily industrialized regions clearly point to water quality improvements, and we suspect the industrial discharges in general will continue to decline as privatization proceeds.

may vanish within the next decade (e.g., weapons and munitions). Given the probable magnitude of changes in the basic structure of the region's industrial economy, we believe that detailed prescriptions to address treatment and reduction of water-borne residuals from specific industries are at best a poor investment of scarce research time and at worst a substantial waste of investment capital. The latter will obviously occur if industries in the region invested substantial funds in reducing wastewater discharges and subsequently went out of business. Therefore, despite the fact that industrial discharges are an important influence on water quality in many subbasins throughout the region, we will not consider them in detail in the research plan, the background of which we outline here. Second, although agricultural point and area sources are also important in their water quality impacts, we will not address them in detail either. Again using Hungary as an example, total fertilizer application declined by over 50 percent between 1981-85 and 1990, while nutrient concentrations declined by 20-40 percent. This is primarily a result of removing subsidies for fertilizer prices. As with industrial sources of pollution, it is clear that the agricultural sector is experiencing major, unpredictable changes in its basic structure and in its environmental effects. As noted earlier, this provides opportunities to reshape control policies to take advantage of these changes, provided that policies are planned and implemented creatively.

In contrast to the industrial and agricultural sectors, influent municipal sewage loads will almost certainly increase over time, and as already noted, the cost for treating those loads is enormous. This is true whether they are analyzed in terms of their relative contribution to total loading or as absolute amounts. The increase in the relative importance of municipal loads is, of course, due to the probable declines in loads from industry and agriculture. The reason for the absolute increase is the push within many CEE countries to place most of their urban populations on the public water supply system (especially in smaller cities and villages of low development), and subsequently add both new and existing public water users to the collection network (sewerage rates are less than 50 percent at present in Hungary and Slovakia, and 70-80 percent in Poland and the Czech portion of the CSFR). Although recent increases in water prices have lowered hydraulic loads on many urban wastewater treatment plants, BOD and nutrient loading can be expected to increase as the proportion of urban populations on public water and sewerage systems increases over time. In addition, industrial dischargers to municipal sewage systems (20-40 percent of total flows) presently have little or no pre-treatment of effluents, while lax enforcement of discharge regulations, very low fees and fines relative to discharge reduction costs, and haphazard privatization leaves industrial users of municipal treatment systems with few incentives to reduce emissions.

In addition to their importance in terms of emissions (and subsequently on ambient water quality), control of municipal sources will be very costly, especially if EC technology-based standards are employed. However, EC treatment technology is but one of many possible environmental policies which CEE governments could implement. Other policies and technologies may be substantially less expensive in the short run, while producing environmental benefits that are comparable to those resulting from imposing EC standards. (note that this does not exclude imposing EC standards in the long run, as resources become available).

To reiterate, we believe that this presents CEE countries with an opportunity to take advantage of the ongoing economic re-organization to formulate policies that are clever, environmentally effective, economically efficient, and flexible, gradually increasing standards over time as the region's economies improve.

The remainder of this paper lays out a research plan to investigate the costs and environmental effects of a variety of environmental policies for CEE as they apply to municipal waste-water treatment. Section 2 addresses the goals of the research in more detail, while Sections 3 and 4 outline the data collection and water quality model developments, respectively, which will be needed for the effort. Section 5 describes the control policies we intend to investigate, while Section 6 delineates the tangible products (i.e., computer models, training, etc.) that we expect to produce in the course of the research.

2. GOALS OF THE RESEARCH

The research which we propose is intended to support development of cost-effective policies for waste-water treatment and ambient water quality improvements in CEE. The intent is to develop data, analytical tools, training, and computer models to assist policy-makers in exploring trade-offs among changes in:

(1) Total economic costs, including both investment or capital costs and operating expenses;

(2) Distribution of costs among economic sectors, countries, and within-country regions;

(3) Total water-borne emissions of BOD, nitrogen, and phosphorus, and sectoral and geographic distributions of emissions;

(4) Ambient water quality changes, at spatial resolutions ranging from the tributary or subbasin level (e.g., the Nitra in Slovakia or the Sajó in Hungary) to large river basins (e.g., the Danube) and nutrient loadings from larger rivers to seas (e.g., the Adriatic and Baltic);

(5) Emission control technologies, including conventional and innovative waste-water treatment;

(6) Emission control policies, including traditional technology-based standards, economic incentives, and other innovative policies;

(7) Timing of policy implementation, investment in treatment facilities, and changes in ambient quality, for both short-term and long-run strategic planning.

It is clear from the this list that CEE countries face many challenges in planning municipal waste-water treatment facilities. First and foremost, they must develop policies to prioritize investment in municipal treatment infrastructure, including both collection systems and treatment facilities. Second, these investments will probably be spread over several decades, due to the limited resources available to address water quality problems, and the research results must address this directly. Third, it is obvious that investments should be robust with respect to uncertainties in the region's economic development, and suited to each country's political culture and governmental system. **The requirement for robustness, in combination with resource limitations, clearly suggest policies which invest first in technologies and treatment facilities which would be required under almost any conceivable economic scenario and allow for flexibility in subsequent investment, such as nitrogen removal from wastewater.** As a corollary, the data, models, and supporting software must be easy to update and modify, as circumstances change over time.

At the risk of belaboring an obvious point, the planning process and hence any research designed to support it will involve a tight integration between many disciplines and issues, including, though not necessarily limited to:

(1) Regional planning and decision-making;

(2) Micro-economics, to analyze the sectoral and regional costs of different policies;

(3) Wastewater treatment design and operation, to calculate the costs and effectiveness of alternative treatment plants;

(4) Ambient water quality assessment and modeling, to project the water quality effects of different policies;

(5) Public finance, to address the financial (or cash-flow, as distinct from economic) implications of various policies;

(6) Macro-economics, to develop reasonable scenarios for changes in the industrial and agricultural sectors, which will not be addressed in detail in the work planned;

(7) Innovative research in treatment plant design, water quality modeling, optimization techniques and control policy design.

The next three sections explore how these goals might be accomplished.

3. ALTERNATIVE WASTEWATER TREATMENT TECHNOLOGIES AND RELATED DATA

In this section, we address policies designed strictly to reduce discharges, as opposed to policies designed primarily to improve ambient quality. Both types of policy require data on base-case loads and alternative treatment technologies (i.e., removal rates and costs). However, policies targeted directly at ambient quality improvements also need information on hydrology, hydraulics, water quality and associated ambient quality models, a topic discussed in Section 4[8].

The obvious place to begin is with base-case data describing current discharges from municipal waste-water treatment plants. As noted earlier, we will focus on so-called "conventional" discharges (i.e., BOD, nitrogen, and phosphorus), and exclude heavy metals, persistent organic compounds, et cetera. Our working assumption is that non-conventional discharges, primarily originating in industrial plants discharging into municipal collection and treatment systems, eventually will be reduced to acceptable levels by industrial re-structuring and pre-treatment processes. Data on existing plants in Poland, the CSFR, Hungary and Bulgaria are currently being gathered and processed. This data, in combination with "engineering" estimates for municipal treatment plant discharges where actual monitoring data is of poor quality or unavailable altogether, will serve as the emissions database for the present situation. More specifically, the database will incorporate, among others, annual average data on:

(1) Public water supply;

(2) Municipal sewage collection systems;

(3) Municipal sewage treatment facilities, including type(s) of treatment, capacities, current loads, and efficiencies;

(4) Industrial loading on municipal systems;

(5) Designs or plans to upgrade or extend collection and treatment systems, including projected removal efficiencies, capital costs, and operating costs.

Industrial point sources can often achieve reductions in emissions either by changing process technology (e.g., more efficient solvent recovery), altering their product mix (e.g., producing products which generate less wastewater per unit) or by end-of-pipe treatment (e.g., chemical treatment to remove BOD or heavy metals). In contrast, reductions in municipal emissions to receiving waters are usually achieved exclusively by end-of-pipe treatment[9]. Therefore, an effort is underway to develop estimates on the **removal efficiencies and costs** (capital and operating) of a wide variety of **alternative municipal wastewater treatment facilities.**

[8]To re-emphasize a point made in the previous section, "clever" policies designed to improve ambient quality cost-effectively will need both discharge data, cost data for different types of waste-water treatment, and ambient quality models. In other words, they will need all of the tools and techniques of integrated river basin management. However, environmental policies often concentrate on discharges, and deal with ambient quality as a "side calculation." Therefore, we have decided to preserve the traditional emission/ambient quality distinction in the presentation of the present paper.

[9]There are three significant exceptions to this generalization. If wastewater and stormwater collection systems are linked in a city, reductions in emissions during storm events can often be reduced substantially by installing separate drainage systems. In addition, if a wastewater drainage system receives substantial infiltration from groundwater, hydraulic loads on the treatment plant can be reduced by sealing and upgrading the collection system. Finally, phosphate loads can be reduced by banning the sale of cleaning agents containing phosphorus. We will not deal with any of these issues, however.

These include mechanical, chemical, biological, biological-chemical, and various advanced (biological-chemical) treatment techniques (see e.g., Ødegaard and Henze, 1992).

The simplest of the alternatives is mechanical, or primary, treatment. It is based on sedimentation of particulate matter, subsequent to screening and grit removal. It results in removal rates of approximately 30%, 60%, 15%, and 15% for BOD, suspended solids (SS), total phosphorus (TP), and total nitrogen (TN), respectively. Biological, or secondary treatment (BT) is obtained by adding an aeration tank where organic material is metabolized by aerobic bacteria. BT increases the removal rate of BOD and SS to about 80%-90%, depending on the composition of the wastewater, while TN and TP removal are only slightly higher. Adding a variety of coagulants, a flocculation basin, and a post-settling tank to primary treatment (known as secondary chemical treatment, or SCT) results in removal rates of approximately 80%, 90%, 95%, and 25%-30% for BOD, SS, TP, and TN, respectively. Obviously, the important difference between BT and SCT is the improved removal of phosphorus; other rates are essentially similar (Ødegaard and Henze, 1992).

If one adds coagulants and flocculants to simple primary treatment, the technique is known as chemically enhanced mechanical treatment (CEMT). This is slightly less effective than SCT, with removal rates are about 60%, 80%, 80%, and 25% for BOD, SS, TP, and TN, respectively (Morissey and Harleman, 1990). However, existing mechanical treatment plants can be significantly upgraded with virtually no capital investment, since CEMT requires only the addition of chemicals to the existing settling basin.

For denitrification, or removal of nitrogen from the effluent, the only widely applied technique is conversion of nitrogen oxides to nitrogen gas by bacteria under carefully controlled combinations of oxic and anoxic conditions. If an anaerobic tank is added, biological phosphorus removal can also be achieved. The process thus obtained is known as tertiary or advanced biological treatment, and many plant specific configurations are possible (see Degremont, 1991, for details). In combination with chemical treatment, removal rates can be as high as 95% for BOD, SS, and TP, and 85% for TN.

Obviously, the capital cost of the plant depends upon the treatment technology. The capital cost of an advanced biological plant is approximately twice that of a mechanical treatment plant. For a large plant serving a town of about 100,000 inhabitants, the capital cost of mechanical treatment is about USD 1.0/m^3, BT is about USD 1.50/m^3, and advanced biological treatment is above USD 2/m^3. Operation and maintenance (O&M) costs for the three configurations vary by a ratio of about 1:3 (with annual costs of 5–17 USD/cap assuming 100 m^3/cap/y water consumption) although the O&M costs of BT, SCT, and CEMT are essentially identical. Sludge production can increase slightly for SCT and CEMT relative to "no-chemical" alternatives, and sludge disposal costs can increase as a result. Sludge treatment costs are very site-specific, but in general, dewatering alone adds 10%-25% to the capital costs of wastewater treatment, while incineration can triple to required investment. Operation and maintenance (O&M) costs follow roughly the same pattern.

While these **cost ratios** (see also Appendix A) will probably apply to most sites, it is obvious that local economic conditions and cultural practices will strongly influence both absolute costs and the choice of treatment technology. Costs for land, labor and locally produced materials vary widely within CEE countries, and traditions (especially regarding biological versus chemical treatment, as well as sludge treatment and disposal) differ substantially within the region. Nevertheless, given the shortage of investment capital in the region, we can draw some tentative conclusions from the cost data presented above. First of all, **it will probably be preferable to invest in treatment plants in a stepwise fashion**, and the obvious starting point is with sewage collection systems (where they do not exist at present) and mechanical treatment. Then, assuming that investment capital is indeed limiting and that the sludge from chemical treatment is locally acceptable, the next step should probably be chemically enhanced mechanical treatment, which seems to offer the best results (i.e., removal rates) per unit cost (including sludge handling). Finally, except where nitrate discharge is of particular local concern (e.g., due to nitrogen contamination of drinking water or eutrophication of downstream lakes and seas) advanced biological treatment should probably be postponed until capital is more abundant.

4. METHODS AND MODELS FOR PROJECTING WATER QUALITY IMPROVEMENTS

We argue that the major element in developing cost-effective water quality control policies for CEE is the definition and application of professionally well justified, publicly acceptable ambient water quality standards. This means that both short-term and long-term management goals should be set in terms of the quality of receiving waters, in preference to using uniform emission standards. **Such a policy cannot be implemented without the use of water quality models relating emissions at various sites to water quality at monitoring locations.** We believe that this type of policy is reasonable for traditional emissions such as BOD, nitrogen, and phosphorus, for the following reasons:

(1) Well-established technologies are available to remove these pollutants, and their costs and effects are thoroughly understood.

(2) Researchers have extensive experience in predicting the dissolved oxygen, nitrogen, and phosphorus households in rivers, reservoirs, and lakes (see e.g. Orlob, 1982, Thomann and Mueller, 1987). At least by implication, our understanding of the processes which govern the behavior of conventional pollutants is fairly good.

(3) Analytical techniques for monitoring ambient concentrations of these substances are readily available.

In contrast, for toxic organics, heavy metals, and other micro-pollutants (excluded from the present study), the understanding of their behavior in the environment is substantially worse. Therefore, we would suggest a policy based on effluent standards.

We next list some of the **major features of the problem**. These will frame our discussion on the type of water quality models we intend to apply.

(1) The geographic scale can range from a few hundred km^2 to several hundred thousand km^2, with 10-100's of sources.

(2) The water quality problems of interest are caused by BOD, nitrogen, and phosphorus.

(3) Local problems, such as dissolved oxygen (DO) depletion, and regional problems, such as eutrophication of seas fed by large river systems, may occur simultaneously and have closely related causes.

(4) Water quality goals or standards can be expressed either in terms of local concentrations (e.g., mg/l of BOD) or loads (e.g., tons of nitrogen delivered to a river mouth each year).

(5) Goals should be formulated for both the short term (within the next 2-10 years) and long-term (one-two decades out).

(6) The quality problems we will address are strategic investment considerations rather than short-term operational concerns.

(7) Our interest is in deriving policies which meet ambient quality goals at least-cost.[10] These features have many implications for our choice of water quality models.

Firstly, from (1), (6), and (7), it follows that we will select relatively simple water quality models. Secondly, (6) suggests that models can be steady-state, and be employed for carefully chosen design or critical-flow conditions. For a DO problem, this is often characterized by Q_{355} (the annual 10-day low flow), or by the lowest seven-day flow occurring in ten years. In contrast, for nutrients, the low-flow condition may not capture the worst-case regional impacts (even for point sources). The result is that different flow regimes may be used as design conditions for different components.

Thirdly, the consequence of (2) is that we will model oxygen, nitrogen, and phosphorus households and associated indicators, and relate these to appropriate ambient quality goals or standards. The indicators could include DO, chlorophyll-a (Chl-a), BOD, and numerous forms of nitrogen and phosphorus.

Fourthly, the need for an optimization framework suggests models which are linear (or linearizable) with respect to emissions or loads. However, the state of the aqueous environment combined with the behavior of natural systems as loads are reduced over time may lead to important non-linearities. While we take up this issue later on, we note here that the **incorporation of water quality, as opposed to water quantity into optimization models is not well-established.** Few examples can be found in the literature (see Spofford et al., 1979, Somlyódy, 1986, or Somlyódy and Wets, 1988, for examples). Therefore, it follows that the initial focus will be on simple, steady-state water quality models. However, after a solution with simple models is obtained (see Section 5), we plan to check the results against more complex, more realistic models. This is necessitated due to the simplifications and assumptions required to formulate the problem in a manner amenable to optimization. The more complex water quality models will be employed to evaluate compliance with water quality standards. They can also be used to derive improved, piece-wise linear aggregated

[10] See Section 5 for details on optimization procedures to accomplish this.

models for the optimization framework (see Somlyódy, 1986, for an example). We can visualize the entire procedure such that "behind" a simplified, linear model, there is a more detailed, possibly dynamic, version. The two communicate with each other as needed, and the more complex model can correct the simpler one (e.g., when the predictions of the two differ substantially). With progress in computational software and faster hardware, simple simulation and optimization models could gradually be replaced by more complex systems, depending on real needs. This raises many research questions which are outside the scope of the present paper.

We turn next to **state-of-the-art water quality models** which could be used for our research (see Table 4.1). We will not discuss these models in depth, as this can be found in the literature referenced in the table (for a recent review see Somlyódy and Varis, 1992). From the table, one can see the development of the QUAL model family (by USEPA) over the past two decades, starting with the Streeter-Phelps equations, next incorporating nitrification/denitrification, and subsequently simple phosphorus cycling. The growing number of state variables and parameters show how the models' complexity has increased over time.

Model structure depends on the treatment of flow and physical transport (convection and dispersion). For Models (1)-(4) in Table 4.1, the underlying assumption is constant flows and emissions leading to a steady-state version. Concentrations are computed along the river length or over travel time with constant inputs.

QUAL2e is dynamic, but it is designed solely to calculate diurnal variations. A dynamic river model simulating daily changes is obtained for any of the first six models by adding reaction terms to the right-hand side of the longitudinal dispersion equations and if daily data are used for forcing functions (emissions, solar radiation, etc., see Somlyódy and Varis (1992) for details).

The basis for the total phosphorus model is the Vollenweider (1968) type of empirical approach (developed originally for lakes) assuming that material is removed by net sedimentation characterized by an apparent settling velocity (the approach can also be applied for suspended solids and nitrogen). The simplest phosphorus cycling model, (6) in Table 4.1, is essentially the same as the corresponding block of QUAL2e (OP and DP are practically equivalent, while Chl-a can be converted to AP on the basis of stoichiometry), assuming that the impact on DO is negligible. The usage of complex ecosystem and nutrient cycling models is beyond the scope of this paper. They are listed for the sake of completeness. They have been applied primarily in a research framework in the study of large water bodies.

As noted earlier in this section (and explained in detail in Section 5) we will apply water quality models in an optimization framework, minimizing costs while meeting ambient standards. For this reason, we wish to specify **transmission coefficients**, $TC_{i,j}$'s, expressing the impact of emissions from discharger **i** on ambient quality at monitoring point **j**. If, as this implies, the response function of ambient quality with respect to emissions is linear, water quality is then related to emissions by a system of linear algebraic equations. We now analyze the nature and derivation of transmission coefficients from Models (1)-(6) in Table 4.1.

Table 4.1. Summary of "state-of-the-art" water quality models (state variables and parameters refer to reaction equations of models, only)

Model	State Variables	Parameters	Receiving Waters	Use	References
(1) Streeter–Phelps (S–P)	2 (DO, BOD)	2	Rivers	Static ("travel time" approach)	Streeter & Phelps (1925)
(2) Extended S–P	3 (DO, CBOD, NBOD, SOD)	3–5	Rivers	As for (1)	Thomann & Mueller (1987)
(3) S–P and N cycle	6 (DO, CBOD, ON, NH3–N, NO2–N, NO3–N)	8–11	Rivers	As for (1)	Orlob (1982) Thomann & Mueller (1987)
(4) QUAL 2e	10 (as for (3) plus SOD, OP, DIP, CHLA, T)	~50	Rivers	Dynamic (diurnal fluctuations only in the EPA software version)	Brown & Barnwell (1987) Orlob (1982) Thomann & Mueller (1987)
(5) TP	1 (TP)	1	Lakes, reservoirs, rivers	Static, dynamic	Vollenweider (1968) Thomann & Mueller (1987)
(6) Simple P cycle	3 (DP, DIP, AP)	>14	Lakes, reservoirs, rivers	Dynamic, or as for (1)	Thomann & Mueller (1987) Somlyódy & van Straten (1986)
(7) Complex nutrient cycle	>10	>50	Lakes, reservoirs	Dynamic	Orlob (1982)
(8) Ecosystem	>50	Several hundred	Lakes, reservoirs, seas	Dynamic	Scavia & Robertson (1979)

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

For ease of exposition, we start with the traditional Streeter-Phelps model ((1) in Table 4.1). Its solution for L = BOD (in mg/l) is:

$$L = L_0 \exp(-K_1 t^*) = \frac{E^B}{Q + q} * \exp(-K_1 t^*) \quad , \tag{1}$$

where

L_0 = BOD in the river just below the discharge point, assuming complete mixing (mg/l);
E^B = BOD emission (kg/d);
Q = river flow (m^3/d);
q = wastewater flow (m^3/d);
K_1 = decay rate (1/d);
t^* = travel time in days (expressed as x/U, where U = "average" velocity in km/d, and x=distance in km).

From (1), we can now derive the transmission coefficient for BOD($TC^B_{i,j}$):

$$L_j = EB_i * \frac{1}{Q + q} * \exp\left(-K_1 t^*\right) = EB_i * TC^B_{i,j} \quad . \tag{2}$$

This obviously assumes that the reaction rate, flow rates, and travel time are constant.

The expression for oxygen deficit is $D=DO_s$-DO, where DO_s is the saturation concentration (Thomann and Mueller, 1987). This can also be expressed in a similar fashion as:

$$D_j = E^B_i * TC^{BD}_{i,j} - E^D_i * TC^D_{i,j} \quad , \tag{3}$$

where E^D_i is the DO "load" (in terms of deficit), $TC^D_{i,J}$ is the transmission coeffcient for dissolved oxygen while $TC^{BD}_{i,j}$ expresses the impact of BOD emission reduction on oxygen deficit (D).

Equation (3) is interesting in comparison to (2). As seen, linearity with respect to discharge (seemingly) holds, but a cross-impact also appears (i.e. $TC^{BD}_{i,j}$). E^B_i is a function of the treatment methods or alternative, $X_{i,k}$, and since treatment also affects the DO concentration of the effluent water, Eq. (3) can be written as follows:

$$D_j\left(X_{i,k}\right) = E^B_i\left(X_{i,k}\right) * TC^{BD}_{i,j} - E^D_i\left(X_{i,k}\right) TC^D_{i,j} \quad . \tag{4}$$

Equation (4) suggests that for a multi-component problem, **the transfer coefficients**[11] **are themselves a function of the treatment technology**. This occurs because treatment to remove BOD affects the dissolved oxygen concentration of the effluent.[12] Note that the second term in the right-hand-side of Equation (4) may lead to **non-linearities** in terms of emissions of BOD if the treatment technology, $X_{i,k}$ affects E_i^D at a different rate than it affects BOD. However, the second term is usually small so long as discharge, q, is small relative to total flow, Q.

While D_j is linear with respect to BOD discharge, this is not true for the so-called critical distance, the location where the largest DO deficit occurs (Thomann and Mueller, 1987). Consequently, if we wanted to impose standards at the critical distance, Equation (4) should be solved simultaneously with a second equation non-linear in BOD discharge. However, this problem can also be resolved piece-wise linearization or simple iteration. As we noted previously, more complex, non-linear models will be required to check the validity of the assumptions in the simpler linear models.

Models (3) and (5) of Table 4.1 introduce no additional difficulties not already addressed (for our purposes Model (5) is equivalent to Equation (2)). In Model (4) the joint impact of BOD, nitrogen, and phosphorus control on DO and chlorophyll-a levels appears to lead to an extension of Equation (4). The inclusion of eutrophication (i.e., chlorophyll-a) results in at least two non-linearities, both related to the principle of limitation. First, the removal of BOD and associated suspended solids (SS) leading to increases in ambient light intensity; this can then induce higher algal biomass (Chl-a). Eutrophication in turn can depress DO levels depending on the relationship of respiration, photosynthesis and mortality (over-saturation is also possible).

The second non-linearity is emission-dependent. At ambient concentrations below those where nutrients limit algal growth, the response of algal concentrations to nutrient loads is practically linear (Somlyódy, 1986). However, when nutrients are in excess supply, light is the limiting factor leading to a Monod type of saturation character for chlorophyll-A (see Somlyódy and Varis, 1992, for an example). It follows that for hypertrophic systems there is a range of initial load reductions which have no effect on algal concentrations, until nutrients become limiting. This can be handled by piece-wise linearization or in the checking phase (see Figure 5.1).

Sediment-water nutrient exchange and internal loads cause similar difficulties for Models (2), (4), and (6). These can also lead to significant time delays between external load reductions and improvements in ambient quality, a factor to consider in setting short-term and long-term emission and ambient quality goals. From an analytical viewpoint, this also means that we may switch from one model to another as we move from short-term to long-term policies (see Section 5). This may mean nothing more than updating transfer coefficients in the simple models on the basis of detailed ones, which we refer to as aggregation.

[11] Actually forming a matrix, even for a single monitoring point.

[12] Decay rate, settling velocity etc. in receiving waters depend on the composition of effluent sewage water, i.e., also the treatment technology.

Finally, given the regional goals of this analysis, we cannot perform detailed modeling of lakes, reservoirs, and seas (which could be considered part of the "checking phase"). What we can do, however, is to set emission reductions for rivers such that the quality of these large water bodies will meet regional standards (see Policy (3) in Section 5). Lakes and reservoirs are generally sinks for nutrients, at least viewed from the perspective of the river basins of which they are a component. Their transmission coefficients can be obtained for instance from Model (5) for components such as TP, TN, and SS.

5. MODELS TO ANALYZE THE COST-EFFECTIVENESS OF SOURCE-CONTROL POLICIES

In this section, we integrate the data and models described in Sections 3 and 4 into a set of relatively simple combined economic-water quality control policy models. The integration is required to investigate the cost-effectiveness of a variety of source-control policies, and to demonstrate how these models can be used to help design policies which lead to cost-effectiveness. We address five specific types of control policies, as follows:

(1) Best-available technology. This would require each discharger to install the most advanced, practicable treatment system available (e.g., USEPA Water Pollution Control Act of 1970 and amendments).

(2) Uniform percentage reduction by all dischargers to meet a set of ambient standards (e.g., Spofford and Paulsen, 1988);

(3) Cost-effective reductions in **total emissions** in a river or region to approximately meet a set of ambient standards (see later);

(4) Cost-effective, **source-specific reductions in emissions** to meet set of ambient standards;

(5) Policies using emission charges, ambient quality impact charges, and other economic instruments.

Since we address only municipal treatment plants in the proposed research, when we say "all dischargers" this should be read as "all municipal sewage treatment plants" since all other sources are treated as background. However, the methods described can be applied to any set of dischargers whose base-case emissions (BOD, phosphorus, and nitrogen), control costs and potential discharge reductions are known, at least roughly, to environmental regulators. We deal first with **static policies** (i.e., those where ambient standards and control technologies are planned and implemented only once) and then turn to **long-run changes in policies**, where ambient standards are gradually tightened over time.

Before proceeding to the details of the models needed to evaluate these policies, we define some terms for subsequent use in the discussion in Table 5.1. Although the variables are explicitly defined for only one pollutant, both the definitions and the discussion and equations that follow generalize to multi-pollutant situations, unless otherwise noted in the text. The equations are defined for a pollutant whose fate in the ambient environment is of interest locally or regionally (details on the models needed to calculate transfer coefficients are contained in Section 4.)

Table 5.1. Variable names and definitions for the discussion of control policies.

Variable	*Definition*
$C_{i,k}(X_{i,k})$	Cost of control tehology, source i, control technology k (annualized USD/y).[a] The X represents the "decision variable" for the control technology (its value is zero if no action is taken, while the best available technology defines X_{max}). Note that in general any given control technology will affect BOD, nitrogen, and phosphorus from source i, albeit differentially.
$CMAX_i$	Cost of the most expensive control technology, source i (annualized USD/y).
$CTOT_m$	Total regional cost of control policy m (annualized USD/y).
$E_i(X_{i,k})$	Emission of pollutant for source i (kg/d).
$EMIN_i$	Minimum technically feasible emission, source i (kg/d). Corresponds to the control technology with maximum cost, $CMAX_i$.
$ETOT_m$	Total regional emissions, policy m (kg/d).
$AQBASE_j$	Base case ambient quality, monitoring point j (mg/l), with no new control technology for any source.
AQS_j	Ambient quality standard, monitoring point j (mg/l).
AQB_j	Ambient quality background level, due to sources other than municipal treatment plants (mg/l).
$TC_{i,j}$	Transfer coefficient, from emission source i to monitoring point j((mg * d)/(kg * l)).
AQ_j	Ambient quality, monitoring point j (mg/l).[b]

[a] We define costs here in annualized terms, as annual O&M cost plus capital cost * a capital recovery factor. However, in countries where local investment capital is limited and governments are unwilling or unable to borrow capital on world markets to finance environmental projects, capital or investment costs may be a more important scarcity measure than annualized costs.

[b] Calculated as

$$AQ_j = AQB_j + \sum_i E_i(X_{i,k}) * TC_{i,j} \ .$$

Figure 5.1 illustrates how the various models are to be integrated. Models are shown in boxes, while arrows indicate exchanges of information among models. Item 1 shows the sources of raw loads to sewage treatment plants (note that because of the large number of unsewered urban areas in some countries in CEE, the "fixed" costs of placing most establishments on the sewer network will be high). This provides loads to sewage treatment plants, represented by Item 2, "treatment technologies," containing the array of possible treatment types, including data on costs and removal efficiencies, for each plant. Item 3, "control policies," takes information on treatment costs, emissions, ambient water quality, and ambient standards, and decides on the "best" array of treatment technologies, producing summary information on ambient quality, technology choices, emissions, and costs.[13] The simple linear ambient quality model (Item 4) uses information on flows, background loads, reaction rates, and emissions, to calculate receiving water quality. Taken together, Items 1-4 can be placed into an optimization framework to select the array of control technologies which meets ambient standards at least cost (see discussion on each policy below). The optimization framework is shown as the dashed-line box, Item 5, in the figure. Items 6 and 7 refer detailed checking of the ambient quality predicted by the optimization model, and the usage of complex, dynamic water quality models for updating transfer coefficients. These are discussed in Section 4 and later in this section.

(1) **The best-available technology policy** is one which has been applied frequently in Western Europe and North America. Its analysis is certainly the simplest of the five. Since it does not make any projections or assumptions regarding ambient quality, no water quality model is required. Analysis of the costs is a matter of straight-forward engineering calculations. One starts with an inventory of dischargers[14] and calculates the cost for each source to install the most advanced treatment available. Although the policy has no explicit ambient quality goals, the obvious assumption behind it is that water quality will improve to acceptable levels. In the context of Figure 5.1, the control policy model simply specifies the "best" and most expensive treatment for each source, using only Item 2, and no others.

In term of the variables defined in Table 5.1, the policy can be described as follows:

$$AQ_j = AQB_j + \sum_i EMIN_i\,(X_{i,k}) \; * \; TC_{i,j}$$

$$ETOT_1 = \sum_i \; EMIN_i$$

$$CTOT_1 = \sum_i \; CMAX_i \quad .$$

Thus, although the ambient quality is guaranteed to be the best possible with any policy (from the definition of $EMIN_i$), the costs are also guaranteed to be the highest of any policy (from the definition of $CMAX_i$).

[13]Note that the definition of "best" will of course vary with the control policy.

[14]An emissions inventory, listing the types and amounts of dischargers and current levels of sewage treatment in the region of interest is required to analyze the costs of all five policy types.

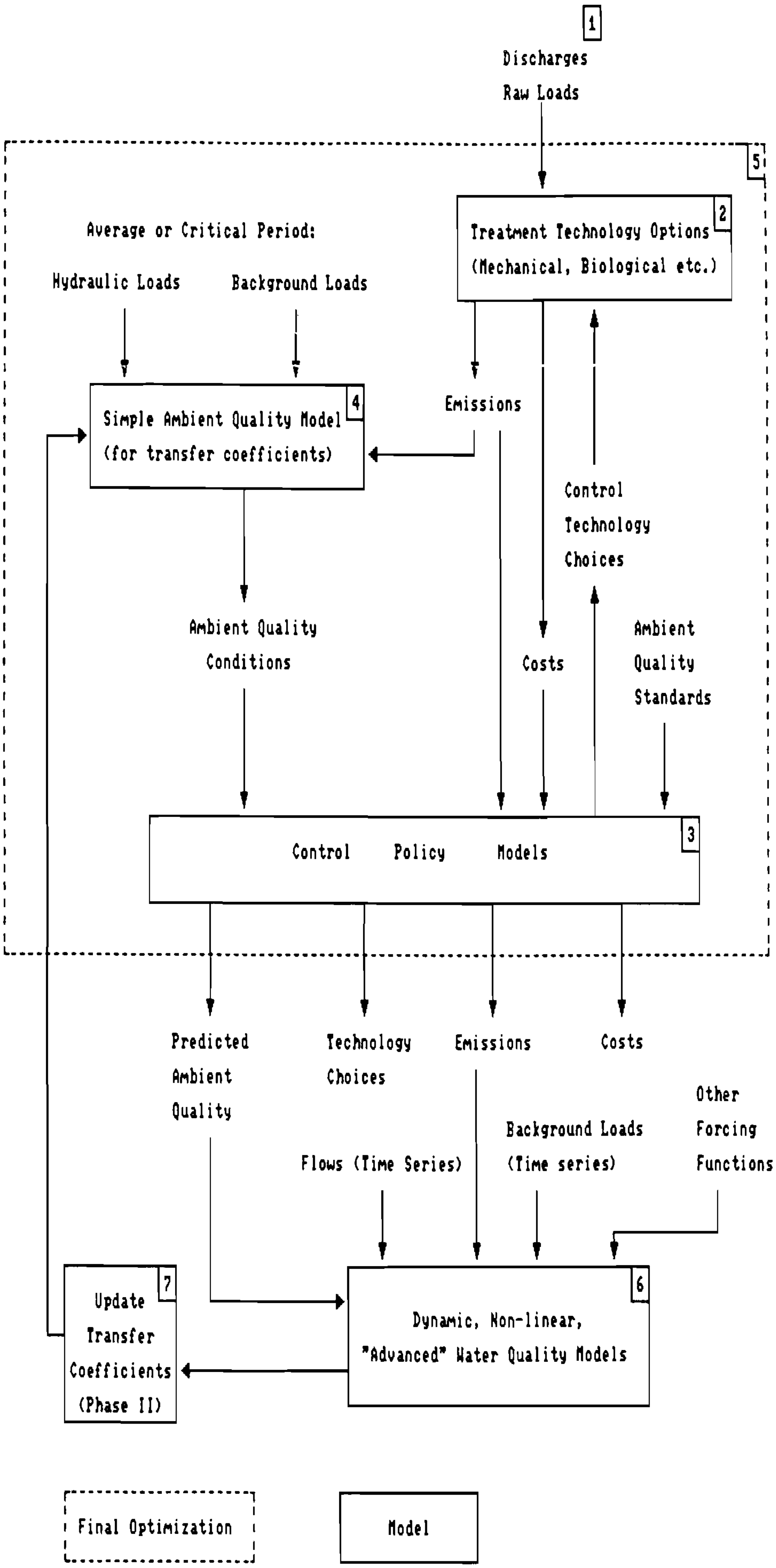

Figure 5.1: Control Policy Model Schematic

In addition to being easy to analyze, the policy is also simple to enforce, since all a regulator need do is check to see if a discharger has installed the correct type of treatment plant for his/her municipality or industrial plant. Nevertheless, it has at least two potentially serious problems. The first is that the "best" technology is generally substantially more expensive than other treatment methods, which may not be quite as effective in removing BOD or nutrients. The second is that there is no guarantee that this approach will result in acceptable levels of ambient quality; indeed, some examples in regions where it has been applied suggest that even with universal application of the approach to all point sources, water quality will often be worse than regulators and interest groups would like, due to diffuse loads and other problems.[15] This can either be interpreted such that regulators have set impossibly high standards, which must be revised downward, or that other emission sources should be included in control policies. Unfortunately, all that is guaranteed is that it will be the most expensive policy, albeit the one with the largest likely improvement in ambient conditions. As such, it sets an upper bound on expected costs and environmental improvements (recall that we are not addressing heavy metals, toxic organics, et cetera here). As with the other policies, its actual costs and environmental effects are, of course, determined by particular circumstances.

(2) **Uniform percentage reduction** is often believed to be one of the fairest policies, since all discharges are required to reduce emissions by the same percentage in order to meet a set of regional ambient standards. Unlike the first policy, it of course requires an ambient quality model to calculate the required percentage reduction, since a portion of environmental quality impacts are usually due to uncontrolled background sources.[16] In addition, in practice it is often the case that some sources affected by the policy cannot reduce their emissions by the required percentage, and so the other sources must reduce by some additional, empirically determined percentage in order to meet the ambient quality targets (this could well be the subject of negotiation between sources and regulators, another reason for having an ambient quality model). Therefore, in order to predict the policy's costs and environmental effects, planners would need: (1) a simplified ambient quality model, which would contain transfer coefficients for each discharger and each environmental monitoring point; and (2) a "menu" of possible control technologies for each source, containing each alternative's cost and removal rates. Referring again to Figure 5.1, no formal optimization is needed. Although the control policy does consider ambient quality explicitly, it does not use cost information in deciding upon control alternatives. In effect, it uses Items 2-4, but no others, in upon the policy. One cannot say in general what the costs from the policy will be, but again assuming that all dischargers can meet the required percentage reduction, then total discharges will be:

$$ETOT_2 = \sum_i EBASE_i * REDU_{max} ,$$

[15]For example, in the Chesapeake Bay in the U.S., although point-source discharges have been reduced substantially over the past 20 years, nutrient levels in the Bay are still quite high, due to the effects of non-point sources of water pollution and deposition of nitrogen from area sources of combustion (especially automobiles).

[16]If background load is negligible then no formal ambient quality model is required.

where $EBASE_i$ is the base-case discharges from point i, in kg/day, and $REDU_{max}$ is the required percentage reduction from each source. Obviously, if one or more dischargers cannot meet the reduction requirement, then the others will need to reduce their emissions beyond $REDU_{max}$ in order to meet the standard.

While we do not address the equity or fairness issue, it is obvious that the uniform percentage reduction policy is unlikely to be cost-effective. Since it ignores both the costs of discharge reduction and the relative values of transfer coefficients for different dischargers, it is extremely unlikely that it will produce a cost-effective strategy for meeting an ambient quality standard.

(3) **Total emission reduction**, unlike the first two policies, explicitly tries to minimize costs (though not in an especially clever way). The goal of the policy is to approximately meet ambient standards in a river (or some subset thereof) by placing a limit on total emissions in the river, and meet that criterion cost-effectively. As with the uniform percentage reduction policy, this requires an ambient quality model with transfer coefficients for each discharger, base-case discharges, and a menu of treatment alternatives for each source. In addition, it requires a simple cost-minimization or optimization model, that calculates the least-cost combination of emission reductions to meet a total discharge limit (imposed by the regulator or analyst). Cost-minimization models of this sort are widely used in many fields, but creating and verifying the optimization model admittedly adds a further complication to the planning process.

In practice, a single constraint is used for emission reduction (although ambient quality constraints could be used simultaneously). An analyst would first identify the pollutant(s) affecting an ambient quality parameter--for example, total BOD and dissolved oxygen (DO), respectively.[17] After examining base-case emissions and DO levels, the analyst then makes an educated guess as to what the level of total emissions should be in order to meet the standard. This level of emissions is then used as a constraint in the optimization model. The optimization looks at the possible control policies for each source on the river, and selects the least-cost way to meet the total emission constraint. This array of emissions the sources is then used as input to a water quality model, which calculates the resulting DO level.

It is unlikely that the ambient standard will be met at all monitoring points on the first iteration. Obviously, if the constraint results in DO levels lower than the standard, the analyst must select a lower total emission constraint and try again; the converse is true if the resultant DO level is higher than the standard. The process continues until the standard is met, plus or minus some acceptable margin of error.[18] In terms of the variables in Table 5.1, the procedure can be defined as follows:

[17]It is obvious that many other emissions (e.g., dissolved inorganic phosphorus, ammonia nitrogen, etc.) also affect dissolved oxygen levels (see Section 4 for more details). We simplify matters to make the example as concise as possible.

[18]The adjustment process could obviously be automated using a simple heuristic convergence algorithm.

Minimize

$$\sum_i \sum_k C_{i,k}(X_{i,k})$$

subject to

$$\sum_i E_i(X_{i,k}) \leq E_{\text{target}} \quad ,$$

where E_{target} is a level of total emissions chosen by the analyst to try and meet approximately the ambient quality standard at all monitoring points, and the minimization is performed using standard linear programming techniques. Note that Policies (3) and (4), while similar, are not identical unless all transfer coefficients are equal.

The total-emission constraint clearly offers some additional flexibility over the uniform percentage reduction policy, since the constraint is placed on total emissions rather than discharger-by-discharger. It also explicitly tries to minimize costs, which may lead to lower costs than for uniform percentage reduction (although there is no guarantee that it will be less costly). While Policy 3 will probably be more costly than Policy 4, there are some situations where it may be preferable. Assume, for example, that several countries have agreed upon both the total permissible nutrient loading to the Baltic Sea and have further agreed upon how the total loading should be allocated among countries. In addition, if transfer coefficients for the nutrient are not well-known, the simplest assumption may be that the coefficients are more or less equal, at least within countries (depending upon the judgement of experts). Then Policy 3 is clearly the method which will minimize costs for each country.[19] Alternatively, although there may be agreement within a river basin that emissions must be reduced, there may be substantial disagreement regarding the relative values of transfer coefficients for each source. In this case, total discharges might be reduced efficiently (i.e., using (3)), followed by a period of observation of actual environmental quality to see if standards are actually met. If the answer is yes, then the process is at an end; otherwise the total emission constraint may need to be adjusted as described above.

Both uniform percentage reduction and total emissions constraints share a potentially serious problem, given that we want to meet ambient standards for DO, phosphorus, and nitrogen simultaneously. It is well-known that different sewage treatment technologies reduce emissions of both BOD, phosphorus, and nitrogen, albeit by different percentages. Therefore, it is almost impossible to meet standards in a stepwise fashion using either uniform percentage reduction or total emission constraints. That is, one cannot **first** meet DO standards (by limiting BOD), **then** meet phosphorus standards (by limiting P emissions), **and finally** meet nitrogen standards (by limiting N emissions). If one attempts this sort of stepwise approach, the result will generally be that in controlling phosphorus one "over-controls" BOD, or vice versa, leading to increased costs.

[19]Note that in this case (3) is simply a special case of (4).

While there is nothing wrong in principle with having ambient quality that is better than a standard, it does result in wasted resources devoted to the water quality problem, since the system will probably be constrained more than is really necessary. In circumstances where there are abundant funds available to finance environmental quality improvements, this may be acceptable. In CEE, where resources are extremely scarce, it seems likely that this will create serious problems.

(4) **Cost-effective, source-specific reductions to meet ambient quality standards,** avoids many of the problems with the first three policy types, at the cost of a modest increase in analytical complexity. This type of policy guarantees a least-cost solution to the problem of meeting a set of ambient standards.[20] However, although the data and ambient quality models are the same as for Policies (2) and (3), the optimization model is slightly more complex. This is most easily seen by comparison with (3), cost reduction in total emissions. There, an optimization model was used to minimize the cost of meeting a constraint on total emissions. For (4), an optimization routine would be used to minimize the costs of meeting a constraint on ambient quality. In the case of a single type of discharge and a single ambient quality parameter, the optimization model works by minimizing the control costs to bring the total ambient quality impact of all sources down to acceptable limits, where the ambient standard is just met. In so doing, this type of policy takes advantage of the fact that the importance of different dischargers depends on the relative sizes of their transfer coefficients, in addition to their control costs. The equations for (4) are shown below.

Minimize

$$\sum_i \sum_k C_{i,k} (X_{i,k})$$

subject to

$$AQ_j \leq AQS_j \quad ,$$

where

$$AQ_j = AQB_j + \sum_i E_i (X_{i,k}) * TC_{i,j} \quad .$$

As can be seen from the equations, this is the only policy (of those discussed so far) which explicitly minimizes costs while meeting environmental standards strictly.[21]

[20]At an intuitive level it is obvious that policies of type (4) are least-cost "by definition". The formal proof that these policies are least-cost is beyond the bounds of this paper. See Spofford and Paulsen (1988) for a fuller explanation and proof.

[21]An alternative formulation would minimize the deviation of water quality from ambient standards under given budgetary constraint.

The result is that policies of Type (4) are guaranteed from theory to be the least costly means for meeting a set of ambient standards (assuming, of course, that they can be met at all within the bounds of the planning problem at hand).[22] In addition, unlike Policies (2) and (3), they can easily be extended to handle multiple emissions and multiple ambient standards (e.g., DO, nitrogen, and phosphorus concentrations). The difficulty, as noted above, is that the optimization methods required are slightly more complex[23], but techniques for constructing and implementing this type of model are well known and have many applications in environmental resource planning (see e.g., Spofford and Paulsen (1988), Klassen and Amman (1992), etc.). In practice, the most expensive portion of the research needed to support environmental planning is the data collection effort, and as noted data for models of this type are identical to that necessitated for (2) and (3). The important cost is not the financial investment required to establish the modelling framework, but rather the additional complexity introduced into the decision and planning process by the optimization models. However, the economic savings which could result from application of this approach may be substantial. Although application of this type of model to water quality management is rare, it has been used relatively often to analyze air quality problems. For example, Spofford and Paulsen (1988) report the following results in an analysis of regional control of airborne particulates for Philadelphia.

Table 5.2. Cost Ratios for Control of Particulates (Spofford and Paulsen, 1988).

Policy	*Type*[a]	*Cost Ratio*[b]
Best Available Technology	(1)	>22[c]
Uniform Percent Reduction	(2)	22
Total Emission Constraint	(3)	1.97
Regional Least-Cost	(4)	1.0

[a] According to the scheme presented here, where (1) = best available technology, (2) = uniform percentage reduction, (3) = total regional emission reduction, and (4) = cost-effective, source-specific reductions.
[b] Cost Ratio = Ratio of control costs to cost of regional cost-effective policy.
[c] Best available technology (BAT) was not analyzed by Spofford and Paulsen (1988) but the costs are guaranteed to be higher than uniform percentage reduction.

One of the few examples from the water quality area comes from Lake Balaton in Hungary. Somlyódy (1986) shows that an effluent standard of 2 mg/l for phosphorus (imposed to meet

[22]If it is impossible to meet one or more ambient standard, obviously some trade-offs will be required. This is discussed more fully later in this section.

[23]The supposed difficulty in modeling is often cited as one reason that approaches of Type (4) are rarely used in environmental planning (see for example Spofford and Paulsen (1988)). However, methods to solve problems of this type are well-understood, and as subsequent discussion suggests, the data requirements are no more complex than that for any method which tries explicitly to meet ambient standards.

an ambient standard in the lake) is twice as expensive as the least-cost strategy; this could double at least (i.e., be four times as expensive or more) if one tried to meet recommended EC standards of 1 mg/l. phosphorus per liter. Obviously, the actual costs and cost ratios will vary widely, depending on individual circumstances, but the results shown above suggest that policies can differ dramatically in their costs to achieve essentially the same objective. For instance, the river basin example of Appendix A shows that best available technology leads to a five times more expensive policy than the least-cost strategy.

(5) **The final set of policies concerns the use of economic incentives** in pollution control[24]. Economic incentives have attracted a great deal of attention in recent years, especially in the context of improving environmental quality in CEE. This seems to us to be due to at least three factors. Firstly, among economists, there is the wide-spread belief that proper application of economic incentives can substantially reduce the cost of meeting a set of ambient standards, especially when compared to technology-based standards. Secondly, as CEE governments move away from central planning toward free-market economies, economic incentives clearly have more appeal than the "command and control" policies of the past, since incentives are viewed by many as being much more in line with a free-market system. Finally, regulators and government agencies see effluent charges, fees, and fines as an important potential source of revenue for both environmental improvement projects and general revenues. We deal briefly with each of these matters in turn, in the specific context of improving ambient water quality in CEE countries.

With respect to reducing costs of meeting ambient standards, the key phrase is "properly designed." There is no guarantee that poorly conceived economic incentives will save any money at all; indeed, they may be more costly than policies which contain no incentives. For example, if one placed an extremely high charge on each kilogram of BOD discharged into a river, all dischargers would probably switch to the best available technology (assuming that financing was available to cover treatment plant costs). From the viewpoint of social costs, the high fees are simply a transfer payment from industrial and municipal dischargers to whatever government agency is collecting the fees, and so in principle society is no worse off than if the policy had been to require best available technology for all dischargers. On the other hand, best available technology may cost more than an alternative which meets ambient standards at a lower cost, and more than a region's economy can afford. In addition, from the viewpoint of the dischargers, they are clearly worse off, since in addition to having to upgrade sewage treatment plants, they must also pay a high fee for every unit of BOD discharged. We do not mean to imply that economic incentives lead to policies which are less cost-effective than other ones. However, economists should, from time to time, remind themselves that there is nothing inherent in their use that guarantees that a system will somehow become more efficient.[25] **We believe that incentives have a role to play in water quality control in CEE, but further research is required** to address potential cultural and methodological problems.

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[24]We define economic incentives broadly, to include charges on effluent concentrations, absolute amounts of effluents, and ambient quality impacts, as well as markets in tradable permits for these "commodities". Unless otherwise specified, our comments refer to all of these possibilities.

[25]Problems with the simultaneous control of BOD and nutrients also will arise in using economic instruments to control water quality.

The second point, that economic incentives are preferable to command and control policies, will in the end depend upon national and regional preferences for how to implement and enforce environmental standards. While an analysis of the type we propose here can provide a great deal of information on the costs, emissions, and ambient quality effects of various policies, it cannot substitute for policy-makers preferences for how policies should be chosen and implemented. Clearly, most improvements in water quality in CEE will be caused (directly or otherwise) by government policies.[26] To repeat a point made above, there is no guarantee that policies will be more efficient if they employ economic instruments. In addition, deciding to use economic instruments cannot, in its own right, substitute for having a realistic, detailed plan for improving ambient quality. They may, however, be an important component of such a plan if they are cleverly constructed to improve water quality cost-effectively.

Finally, the use of emission fees, fines, and so forth to raise revenue will obviously be extremely attractive to regional and national governments faced with enormous environmental expenses and no obvious means to pay for them. In addition, there is a long history of using emission fees and similar economic instruments to pay for regional sewage treatment plants, instream aeration installations, and other projects which are efficient on a regional basis but which are not cost-effective for any single discharger to undertake on their own (Bower et al, 1981). Their use to finance regional projects is reasonable, where it amounts to a means to charge users of a resource, such as a treatment plant, for the cost of providing the service. However, governments should avoid the temptation to impose user fees simply to raise general revenues, since this may lead to increasing fees and fines as a substitute for raising income or other taxes. If the fees are sufficiently high, the result may be that dischargers will reduce their emissions far more than is needed to meet ambient standards. It is impossible to say from theory alone what the "best" mix of regulations, fees, fines, and so forth will be. The answer will depend upon both the physical and economic circumstances for a region (e.g., flow patterns, discharge quantities, control costs, et cetera) and political and cultural traditions. As part of the planned research, we hope to be able to discover what features of workable strategies are particular to specific countries, river basins, and so forth, and what features those strategies have in common.

Subsequent to the discussion of control policies, we turn next to several issues that are more methodological. Specifically, we will show explicitly how the regional least-cost policy (4, above) can be extended to situations where a region does not have sufficient financing available to meet ambient standards immediately, but still wants to improve ambient quality subject to a limitation on total costs. Finally, we briefly discuss some methodological problems and the issue on how to place control policy models and associated data, computer algorithms, and so forth into a comparatively user-friendly environment.

In regions facing budget constraints, the simplest solution is to add a series of "penalties" to the objective function, as follows.

[26]Note that the recent improvements in water quality in Hungary (mentioned in Section 1) were caused indirectly by decisions to reduce or remove subsidies from various industries, such as fertilizer production. In addition, although fees and fines have been a formal part of water pollution control in CEE for decades, they were often set at levels far too low to influence behavior--it was cheaper to pay the fines than to reduce discharges.

Minimize

$$\sum_i \sum_k C_{i,k} (X_{i,k}) + \sum_j P_j * C_j$$

subject to

$$AQ_j \le AQS_j + P_j$$

and

$$\sum_i \sum_k C_{i,k} \le B \quad ,$$

where

$$AQ_j = AQB_j + \sum_i E_i (X_{i,k}) * TC_{i,j}$$

and

$$P_j \ge 0 \quad .$$

"B" is simply a budget constraint established by policy-makers. The interpretation of the P_j's is that they are a function of the difference between ambient quality standards and the actual ambient conditions resulting from a control policy, and the C_j's are the "costs" of the failure to meet the standards. Note that in the present context these "costs" are not actual financial or economic costs, but primarily a device used to persuade the optimization program "to do its' best" to meet ambient standards, knowing that it will be unable to do so for all monitoring points. Numerous variations are possible, including have a penalties increase in some faster-than-linear fashion for each reach, so that the marginal penalty increases with the deviation from the standard, and having different penalties for different monitoring points. The values for the C_j's will obviously need to be determined empirically, in accord with decision-makers preferences and model results. Note that this also introduces the possibility of varying the budget constraint to look at trade-offs between ambient quality and expenditures ("B" in the equations above).

All of the technologies described in Section 3 and Appendix A can be characterized individually as (0,1) variables. The problem of non-integer solutions to the least-cost model can be addressed in a number of ways. Conceptually, the simplest solution is to convert the problem into a mixed-integer formulation, and force the model to use (0,1) integer variables for the treatment alternatives. However, this may increase computer solution times unacceptably (particularly if interactive usage is considered important). A second possibility is to solve the problem once as a continuous one, and note the dual values (call them D_j's) on ambient quality constraints at the monitoring point(s) (only those which are binding will have non-zero dual values, of course). Next, one can solve the problem a second time, removing the constraints on ambient quality and using the absolute values of the D_j's as penalties on ambient quality in the objective function. The resulting problem formulation for the second step would then be as follows.

Minimize

$$\sum_i \sum_k C_{i,k} + \sum_j AQ_j \; * \; |D_j|$$

where AQ_j is defined as before.

The result will be an integer solution (at least for the single-pollutant case), in that all treatment options will either be used to capacity or not be used at all. Ambient standards may not be met exactly, since treatment technologies are generally discrete choices (i.e., one cannot build only 10% of a treatment plant), but modest adjustments to the dual values should be able to handle the problem if this is a cause for concern.

Placing these methods in a user-friendly, decision-support system (DSS) is straight-forward in theory, but the details will obviously depend strongly on the actual user audience and on whether or not advanced features are supported in a DSS environment. For example, updating transfer coefficients in the simple ambient models included in the optimization framework (see Items 6 and 7 in Figure 5.1) could either be incorporated as a feature for experienced water quality modelers, or as something which should be directly accessible to "naive" users of the system. In the first case, little programming is required, while in the second use of expert system software would probably be needed.[27] At present, we cannot say exactly what the outcome of this portion of the research will be. The reader is referred to (Somlyódy and Varis 1992) for details on applications of DSS methods to water quality modeling.

In conclusion, we believe that **models to support cost-effective planning in water quality management are likely to result in considerable saving in CEE countries.** The models themselves have been in use for many years in other fields, and techniques to formulate and solve them are well-established. In addition, **their use in the policy process raises a number of interesting research issues.** Finally, methodological issues of the sort mentioned above (i.e., fast optimization techniques for discrete problems and inclusion of complex water quality models in an optimization framework) are interesting research issues in their own right.

6. PROSPECTIVE PRODUCTS

In this section, we summarize the expected products and results from the research described in Section 1 through Section 5 of this working paper. Despite the fact that much of the work will consist of data collection and processing, and the integration of well-established tools and techniques, there will obviously be surprises along the way, requiring variants on existing algorithms, application of methods from disciplines other than those we have already identified, and other unexpected problems. Since these cannot be characterized except in a

[27]Other possibilities on this front include automatic updating of coefficients, based on comparisons of water quality predictions from the simple and complex models, and direct incorporation of complex models into the optimization framework. These are clearly potential topics for future research.

very speculative way, we will confine this section to the tangible products we expect to produce.

The first set of products will be a collection of workable, albeit unfriendly, computer algorithms which integrate the waste-water treatment technologies discussed in Section 3, and water quality modeling techniques described in Section 4 with the policy analysis tools of Section 5. Initially, we plan to focus on tools which would enable professionals in the field to perform the types of control policy simulations and optimization described in Section 5, using fairly simple water quality models. From there, we foresee expanding the software development efforts in two directions simultaneously. The first is to make **relatively simple models more user-friendly. The second** would incorporate more complex water quality models, interactive updating of transfer coefficients in simple models based on the results of **complex models, and adapt the optimization models of Section 5 to handle non-linear discharge-ambient quality effects directly.**

In addition to software, we hope to be able to conduct a series of **studies and workshops** on methods and results obtained for **three distinct audiences**. The first would consist of **waste-water treatment professionals**. Presentations would center on the costs and emissions of alternative treatment technologies (on the basis of detailed treatment plant case studies) and how these can affect regional water quality. The second is **professionals in water quality management**, who have a general background in the quantitative techniques described in earlier sections of this paper. The focus of the workshops for this audience would be to familiarize them with the methods and their usefulness we discussed in this paper. For this purpose case studies performed on the new river basin level will be used (see Appendix A). We in turn would incorporate their suggested improvements into the tools we develop.

The third audience will be environmental planners and policy-makers in international lending institutions, CEE governments, and related establishments. For this group, the focus would be much more results-oriented to demonstrate how the costs and water quality effects of different policies compare in real-world examples. We would consider larger regions (e.g., a country or countries in CEE) and show how cost-effective water quality management policies should be developed from the international or national level down to subbasin levels (international considerations are of crucial importance for shared water resources, i.e., the Baltic Sea or the Danube). We may also develop software specifically for this audience, if there appears to be a demand for it.

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APPENDIX A

ILLUSTRATIVE RESULTS FOR THE NITRA RIVER BASIN

Introduction

In Section 5, we noted that the differences in outcomes among various water quality control policies are largely empirical. That is, although one can make a few generalizations from theory regarding the cost ranking of the control policies, one cannot say what the actual difference in costs will be except with regard to a given river basin or region. In addition, even the ranking of several of the policies cannot be predicted for a general case. In this appendix, we demonstrate the usage of some of the methods developed in the body of the paper. As an example, we use the Nitra River in Slovakia. It is of rather poor quality, especially under low-flow conditions, and its control is an important issue in Slovakia. However, it is not our intention to attempt to "solve" the river's water quality problems. Instead, we will use it primarily to illustrate the major features of some of the different policies, and to show how their costs and ambient quality effects can differ. On the technical side, the preparation of a "real" policy obviously requires substantial additional effort on data collection, development of a discharge inventory, and model calibration and testing. In addition, considerable effort will be required to learn what the goals and objectives of regional policy makers actually are, and what means are available to them for attaining those objectives. These are in fact the subject of an ongoing study at IIASA performed in cooperation with the Water Research Institute (VUVH, Bratislava) and the Váh River Basin Authority.

Regional Description

The Nitra is a tributary of the Vah river, which enters the Danube downstream of Bratislava. Its catchment area is slightly more than 5000 km^2, with about 600,000 inhabitants. Its length is about 171 km, and its mean flow at the mouth is 25 m^3/s. The overall BOD discharge to the river system is above 10,000 tons/y, approximately 70% of which is of municipal origin.

Major point sources and monitoring sites are illustrated in Figure A.1. They represent more than 90% of the municipal contributions and 70% of the industrial contributions of BOD point-source loads to the river. While the Nitra is of high quality upstream, below location (1) there is a gradual deterioration on dissolved oxygen (DO). Between points (6) and (7) abrupt changes occur, leading to possible DO depletion during the summer, when BOD may exceed 30 mg/l. Downstream of (7) the river slows and travel time increases substantially. Travel time from (7) to the river mouth may exceed one week during low flow conditions (3-5 m^3/s). As a result, "self-purification" is effective and DO at the mouth is generally above 4 mg/l even during the summer, when the temperature may reach 25 degrees C.

Current emissions of BOD from major municipalities and industries are shown in Table A.1. We frequently used rough estimates if emission information was uncertain or lacking. The type of treatment indicated in Table A.1 may also differ from the treatment actually used. For example, if there were a highly overloaded biological plant with a BOD removal rate of less than 50%, we considered it as a mechanical unit.

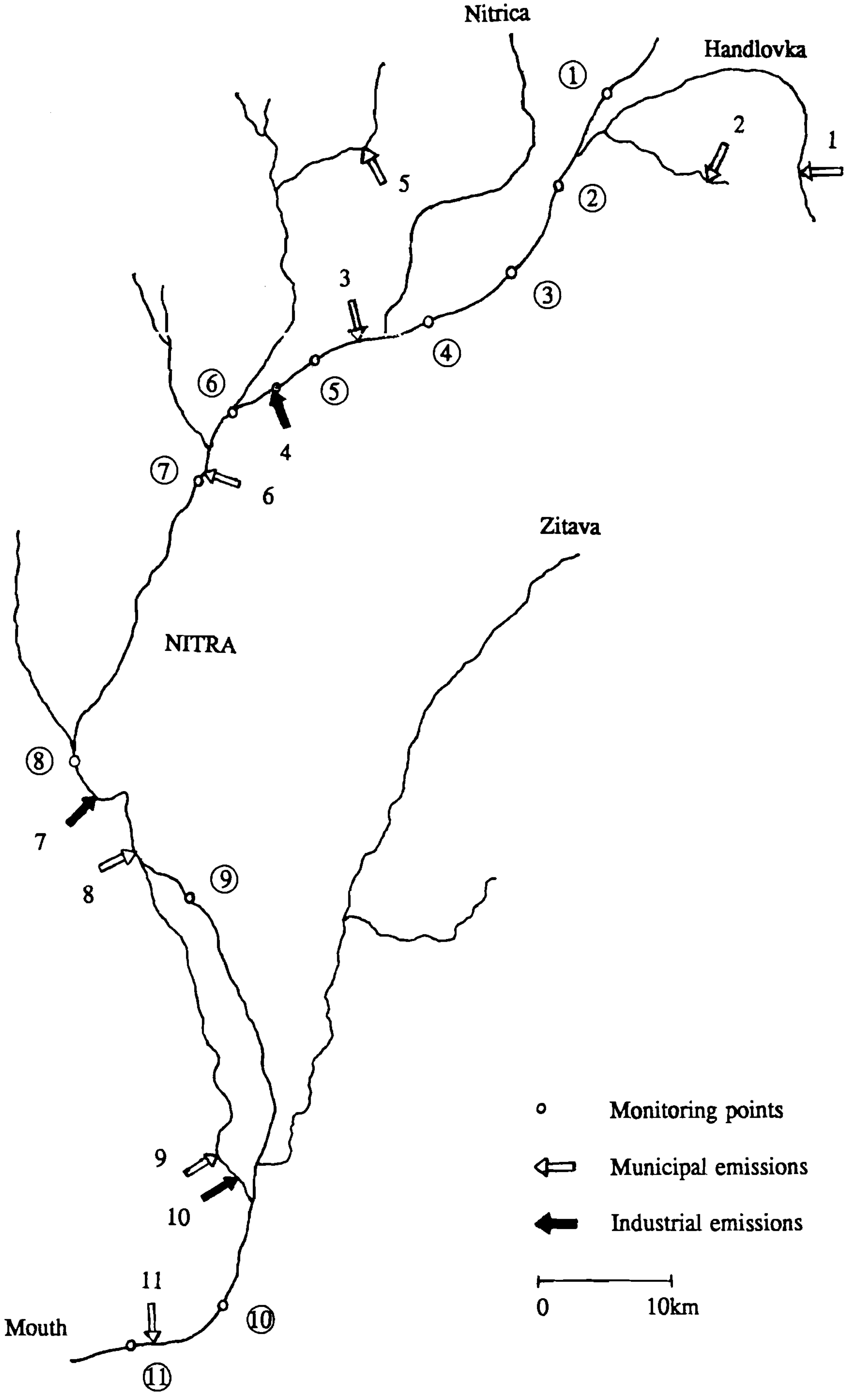

Fig. A.1 The Nitra catchment together with major discharges and the water quality monitoring points

Table A.1. Major parameters of emissions

Code	Name	Population	Effluent BOD [kg/d]	Treatment
1	Handlova	18000	100	B
2	Prievidza	60000	2400	M
3	Partizanske	27000	1800	0
4*	Bosany	–	1200	0
5	Banovce	20000	5500	0
6	Topolcany	38000	800	M
7*	Nitra	–	1700	M
8	Nitra	91000	4200	M
9	Surany	12000	100	B
10*	Surany	–	3000	B
11	N. Zamky	43000	2500	0

0 = no treatment; M = mechanical treatment; B = biological treatment (* indicates industrial sources)

For the example, we assumed 100 m^3/cap/year water consumption for municipalities. Raw BOD concentrations we adjusted on the basis of Table A.1; we used 250 mg/l for towns with little or no industrial load. For influent TN and TP, we utilized 48 mg/l and 12 mg/l (see Ødegaard and Henze (1992)), respectively, as data were absent. We also employed the same BOD:TN:TP ratio for industrial sources, since data were missing for these as well. Industrial sources were assumed to be uncontrollable, as hence were treated as fixed, "background" sources.

Design Conditions and Treatment Alternatives

For design conditions, we chose August 1990. Data on flow rates, travel times, temperature, BOD and DO concentrations were available for monitoring points (1) to (11) in Figure A.1 (for details see Koivusalo et al, 1992). Parameters of the Streeter-Phelps model (see Table 4.1) were selected by approximately matching the observed data; this was used to calculate transmission coefficients for BOD and DO. TN and TP concentrations were missing, but data were available for inorganic nutrients for another period. The latter served as a basis to produce rough estimates of transmission coefficients for TN and TP by using Model (5) in Table 4.1.

Five different municipal wastewater treatment alternatives were used (see Odergaard and Henze, 1992, for details):

M	mechanical only;
CM	chemically enhanced mechanical;
B	biological;
BC	biological with the addition of chemicals;
BCN	biological-chemical with de-nitrification.

Dewatering and anaerobic stabilization were assumed for sludge treatment for all alternatives. Generalized cost functions were obtained from Odegaard and Henze (1992). A summary for a 100,000-population equivalent (p.e.) plant is shown in Table A.2, including effluent quality (if influent water is as described above). Costs for treating unit m^3 wastewater were assumed to be constant down to 30 000 p.e., below which a 50% increase was assumed. Sewage collection development costs were excluded, as these would be a "fixed" cost regardless of treatment alternative, and existing sewerage infrastructure is quite acceptable in the towns considered.

Following the typology of Section 5, we analyzed four different types of control policies:

(1) Minimum discharges, roughly equivalent to imposing EC standards on all municipal treatment plants.

(2) Uniform percentage reductions in discharges to meet an ambient standard set uniformly along the river.

(3) Constraining total regional discharges to meet approximately a uniform ambient standard.

(4) Regional least-cost strategy to meet a uniform ambient standard.

Unfortunately, time did not permit us to investigate the use of economic incentives.

Results

The results of the example analysis are shown in Table A.3. As can be seen from the first line of the results, the base case conditions in the river are fairly poor. The minimum DO concentration is only 0.7 mg/l, maximum TP concentration is 1.8 mg/l, and maximum TN concentration is 7.8 mg/l. According to the model, however, very substantial improvements are possible through control of municipal wastewater treatment alone, even assuming no control of other point sources.[28] The second result shows the costs and water quality that would result from imposing EC emission standards on all municipal plants. This corresponds to the minimum-discharge policy, (1) in Section 5. While water quality obviously improves markedly (e.g., minimum DO concentration rises to almost 7 mg/l), this policy is also very expensive, with capital costs of near USD 65 million, and annualized costs over USD 14 million. As we noted in Section 5, this result provides an upper bound on costs and water quality improvements.

We investigated two different standards for dissolved oxygen: minimum concentrations of 4.0 mg/l, and of 6.0 mg/l.[29] The results clearly show that substantial savings are possible using a least-cost control policy. For the 4.0 mg/l standard, the least-cost policy has capital

[28]Industrial sources and those small dischargers excluded from the model for simplicity.

[29]The 4.0 and 6.0 mg/l standards correspond to the limits for second-class and first-class water, respectively, in many European countries.

Table A.2. Costs and effluent water quality for alternative treatment technologies (100000 p.e.)

Alternative	IC [10^6USD]	O&MC [10^6USD]	TAC [10^6USD]	BOD [mg/l]	TP [mg/l]	TN [mg/l]
M	10.3	0.64	2.00	175	10.0	40
CM	11.8	1.03	2.60	100	2.5	35
B	15.1	1.07	3.10	20	8.5	35
BC	17.1	1.73	4.00	10	0.5	31
BCN	24.6	1.91	5.20	10	1.0	15

IC = investment cost; O&MC = operation and maintenance cost; TAC = total annual cost (assuming 20 years of economic life and 12% interest rate); for M, CM, B, BC and BCN see the text.

costs that are about one-fifth of the BAT strategy. At the same time they are roughly 40% less than the uniform percentage reduction policy, and more than 50% less than a limit on total regional BOD emissions. In the example, at least, the region could save almost USD 10 million in capital costs by using the least-cost policy, as compared to the next most efficient policy. While the potential savings are slightly smaller for the 6.0 mg/l standard, the least-cost policy for the 6.0 mg/l standard still represents savings of almost USD 8 million in capital cost and USD 2 million in annualized costs over the next most efficient policy. Note that as ambient standards become stricter, all control policies will approach the minimum-discharge policy in their costs and ambient quality effects.

We also looked at the same three policies for two total phosphorus standards--0.5 mg/l and 0.7 mg/l--and a total nitrogen standard of 6.0 mg/l (to which 30 mg/l nitrate concentration belongs; an acceptable standard from the viewpoint of drinking water supply). The cost savings of the least-cost policy for these three standards are not as large as for dissolved oxygen. This is due to the relatively limited number of control technologies for the nutrients and the fact that more treatment plants must control emissions, since both TN and TP tend to have higher concentrations as one moves downstream. Nevertheless, the least-cost policies for each are still substantially less than the minimum-discharge policy, and the results in terms of ambient concentrations are not very much higher.

Finally, we examined a policy which called for meeting a DO standard of 6.0 mg/l, a TP standard of 0.5 mg/l, and a TN standard of 6.0 mg/l simultaneously.[30] The resulting costs are shown in the last line of Table A.3. While this is more expensive than meeting any one standard while ignoring the other two, it is still far less costly than the minimum discharge policy, with capital costs of USD 42 million versus USD 65 million for the minimum discharge policy, and annualized costs of USD 10 million versus USD 14 million.

Looking at the control technologies for each type of policy, one finds, as expected, that the minimum-discharge policy utilizes BCN techniques exclusively. There is considerable variation in the mix of treatment techniques for the least-cost policies. For DO > 4.0 mg/l,

[30]For the three-pollutant standard, we did not analyze total emission reductions or uniform percentage reductions, due to time constraints. The costs for the two policies not analyzed would be between the least-cost policy (USD 42 million capital cost) and the minimum-discharge cost of USD 65 million.

Table A.3. Results from Policy Analysis for DO, P, and N

Ambient Quality Standard	Control Policy	Capital Cost	Ann. Cost	Min. DO Conc (mg/l)	Max. TP Conc (mg/l)	Max. TN Conc (mg/l)
None	Base case	0	0	0.7	1.8	7.8
–	Minimum discharges (BAT)	64.7	14.4	6.9	0.4	4.1
DO $\geq$ 4	Uniform % reduction	23.4	5.7	4.3	0.6	6.5
DO $\geq$ 4	Limit on regional discharges	28.8	6.9	4.3	0.8	6.7
DO $\geq$ 4	Regional least-cost	13.2	2.8	4.0	1.3	7.3
DO $\geq$ 6	Uniform % reduction	40.7	10.4	6.9	0.4	6.0
DO $\geq$ 6	Limit on regional discharges	33.8	8.6	6.2	0.4	6.3
DO $\geq$ 6	Regional least-cost	25.6	6.6	6.0	0.7	6.7
TP $\leq$ 0.5	Uniform % reduction	31.5	7.9	5.6	0.5	6.1
TP $\leq$ 0.5	Limit on regional discharges	36.5	9.3	6.2	0.4	6.2
TP $\leq$ 0.5	Regional least-cost	27.2	6.5	5.1	0.5	6.4
TP $\leq$ 0.7	Uniform % reduction	23.4	5.7	4.3	0.6	6.5
TP $\leq$ 0.7	Limit on regional discharges	22.9	5.3	4.3	0.6	6.6
TP $\leq$ 0.7	Regional least-cost	22.6	5.3	4.0	0.6	6.6
TN $\leq$ 6	Uniform % reduction	42.4	10.5	6.9	0.4	5.9
TN $\leq$ 6	Limit on regional discharges	40.7	10.4	6.9	0.4	6.0
TN $\leq$ 6	Regional least-cost	35.3	8.2	4.8	1.2	6.0
DO > 6 TP < 0.5 TN < 6 (least-cost)		41.9	10.2	6.1	0.5	5.8

DO = Dissolved Oxygen; TP = Total Phosphorus; TN = Total Nitrogen. Capital and Annual Costs in Millions of USD.

a mix of CM, B, and no treatment is employed, depending on the dischargers' size and location. For DO > 6.0 mg/l, the mix shifts to B and BC, since the latter is more effective than CM in removing BOD. The least-cost policy for TP < 0.7 results in most sources moving to CM methods, while for TP < 0.5, one source adds biological-chemical treatment (BC). Constraining TN < 6.0 results in a least-cost solution which employs mostly BCN (although one source uses only mechanical treatment). The least-cost solution for DO > 6, TP < 0.5, and TN < 6.0 forces about half of the plants to BCN while the rest employ BC methods. Obviously, as with the cost differences, the choice of treatment technology varies considerably with the ambient standards and region-specific considerations.

Conclusions

Obviously one must interpret these results cautiously. The input data on emissions and ambient conditions is far from perfect, and time constraints did not permit us to develop robust estimates of model parameters or to apply formal optimization techniques to the analysis of the control policies.[31] Nonetheless, **the results strongly suggest that substantial cost savings are possible using a least-cost control policy, especially when one compares it to a minimum-discharge or best available technology approach. In addition, the ambient quality which results from the least-cost policy is very similar to that which results from the minimum-discharge approach.** Given CEE countries limited financial resources, we believe that the benefits of the approach we have outlined are likely to be substantial as the region formulates environmental policy in the coming years.

[31] Given the limited number of control policies, sources and treatment alternatives, we applied simple heuristic algorithms to handle the problem.