INTERNATIONAL ATOMIC ENERGY AGENCY
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FOURTH AUTUMN COURSE ON MATHEMATICAL ECOLOGY

(24 October - 11 November 1994)

"Systems Analysis in Water Quality Management - A 25 Year Retrospect"

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Reprinted from

SYSTEMS ANALYSIS IN WATER QUALITY MANAGEMENT

Edited by

M. B. BECK

PERGAMON PRESS

OXFORD · NEW YORK · BEIJING · FRANKFURT · SÃO PAULO · SYDNEY
TOKYO · TORONTO

1987

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SYSTEMS ANALYSIS IN WATER QUALITY MANAGEMENT— A 25 YEAR RETROSPECT

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ABSTRACT

A reflection is presented of the utility, credibility and application of water quality systems analysis techniques over the past several decades. The emphasis is on predictive water quality models and the U.S. experience. The complexity of the water quality questions and associated modeling has increased by orders of magnitude. Models of sediment interactions and effects of toxic substances are crucial to further development. Four criteria for judging performance and impact are discussed: usefulness, accuracy, serendipity and ownership. Models are widely used in decisions regarding alternative controls specifically to improve cost effectiveness. The results of systems techniques need to be detailed in a wide ranging effort of post audit analysis following implementation of environmental controls. Legislation and policies have incorporated, in a general way, the principles of water quality systems analysis, with the notable exceptions of a widespread reliance on technology based effluent programs and a general disregard of cost trade-offs using principles of optimization. It is concluded that the impact of systems techniques has been broad and significant. Increased quality assurance of model formulation and calculation is necessary to ensure frameworks that are rigorous and state of the art. A need exists for upgrading of understanding by users of water quality systems techniques and the time has arrived for a major world-wide effort to compile the economic advantages of using systems techniques for more informed and efficient decision making.

KEYWORDS

Water quality models; post audit; systems analysis; criteria for impact; model usefulness; model performance; dissolved oxygen; eutrophication.

INTRODUCTION

My dictionary (Webster's, 1963) defines a Retrospect as "a review of, or meditation upon past events" as opposed to a Retrospective which is "a generally comprehensive exhibition showing the work of an artist over a span of years." This paper is a reflection on our past, meditative perhaps at times, but not intended to be a comprehensive exhibition of our collective artistic works. That would be a task well beyond my own competence. So what I have to offer in this retrospect are my own thoughts, views, comprehensions and understanding of where we have been in the last 25 years in this (sometimes perceived) arcane practice of systems analysis applied to water quality management.

To review our past, to evaluate the impact, or lack thereof, of systems analysis in water quality management is essential, if we are to offer the decision making community a state-of-the-art understanding of comtemporary water quality issues. What effect has our work had on the decision making process? Does our work really matter or are we just talking and

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reporting to ourselves? What are our "successes"? Our "failures"? Indeed, what are the criteria that we can even suggest as useful for judging the significance of our work on the water pollution community at large? Finally, what does the future hold and what are the emerging issues?

NATURE OF SYSTEMS ANALYSIS IN WATER QUALITY MANAGEMENT

In the present context, we begin by offering a definition of systems analysis: "The engineering art of integrating and synthesizing the physical, chemical, biological and mathematical sciences with the social and economic sciences to construct frameworks that elucidate the consequences of alternative water quality and water use objectives."

The principal components of this definition are:

1. Engineering art of integration: implying (a), a focus that is practical in nature (the engineering), (b), a certain "flair" that tends to be personalized and less than totally scientifically rigorous (the art), and (c) a culling and rebuilding of key elements of diverse disciplines (the integration).
2. Synthesis of the natural and mathematical sciences with the social and economic sciences: implying that what we do is more than mathematical modeling of natural systems and incorporates policy, economic, social and cultural issues into the analysis.
3. Elucidation of consequences of alternatives: implying that water quality systems analysis has much to say in the process of decision-making including revelation of previously hidden behavior and formulation of new alternatives.

Within these broad components, the key steps are:

1. Evaluation of the Problem
    a. Residuals input determination
    b. Mathematical model construction
    c. Assessment of risk to human and ecosystem population without controls
    d. Specification of a range of feasible water quality/use objectives
2. Evaluation of Alternative Controls
    a. Determination of effectiveness of alternatives
    b. Optimal cost/benefit analysis
3. Decision and Promulgation of Control Program
    a. Water quality standard setting
    b. Determination of allowable risk
    c. Optimal control strategies
4. Implementation of Control Program
    a. Waste load allocation
    b. Negotiation and issuance of discharge permit
    c. Monitoring of Program
5. Post-Audit of Program
    a. Attainment of water quality standards
    b. Attainment of water use objectives
    c. Evaluation of costs and benefits
    d. Predictive capability of model framework

With no apologies for an obvious bias, at the heart of the entire sweep of these components and key steps is the construction of credible, defensible and predictively accurate mathematical modeling frameworks. Without such predictive capability, it is simply not possible to develop a firmly based water quality management program. It is for this reason that much of the effort in the past several decades has been in developing predictive mathematical models of water quality at a variety of different levels of complexity. All of these models are aimed first at calculating the expected concentrations of water quality variables. These concentrations then form the basis for risk assessment to the aquatic ecosystem and to the public health. Thus there have been intensive efforts in developing models that can be used with confidence in evaluation of alternative controls, cost/benefit analysis, risk assessment and optimal control strategies.

I would like to focus on this area of water quality models not to the exclusion of the socio-economic models (e.g. optimization of water quality) but simply to emphasize the central role that predictive models play in the decision-making process.

Historically, we have developed systems frameworks and more specifically water quality models for three broad classes of problem contexts: 1) Biochemical oxygen demand (BOD)/ dissolved oxygen (DO), 2) Aquatic plants and nutrients, and 3) Toxic substances. Within these contexts, attention has been variously placed on steady state and time variable deterministic frameworks to ensure credible inclusion of relevant mechanisms as well as incorporation of uncertainty and probabilistic concepts to insure consideration of stochastic elements in alternative evaluations. Models have grown from the two state variable BOD/DO models to multi-state variable (e.g. 20) models of phytoplankton/ nutrient models. Spatial detail has increased by orders of magnitude from simple stream calculations to finite difference models of 500 or more grid points. Time variable calculations have emerged extending from hour to hour calculations to long-term year to year calculations. Hydrodynamic circulation models are increasingly coupled to water quality models.

Reflection indicates some general observations:

1. The aquatic plant/nutrient problems are the most difficult models with which we have worked because of the complexity of the plant biology, the non-linear interactions between nutrients and aquatic plants and the interactions of the sediment.
2. The dissolved oxygen problems, connected intimately with primary productivity and sediment effects, in spite of the long history, tend to be considerably more complex than generally believed.
3. Sediment interactions are important to all water quality problem contexts and apparently credible interactive sediment models are only now appearing.
4. Toxic substances fate models, linear in nature, tend to be less complex than generally believed.
5. Past emphasis was on models of fate (i.e. concentration), future models must of necessity include prediction of effects on the aquatic ecosystem and to a degree on human health; toxic substances represent the most complex problem context experienced to date for prediction of effects of exposure concentrations.

With this background and observations, it is necessary to inquire to what degree water quality systems analysis has been "satisfactory" in some sense and the degree of impact on the larger decision-making process.

CRITERIA FOR JUDGING PERFORMANCE AND IMPACT OF SYSTEMS ANALYSIS IN WATER QUALITY MANAGEMENT

In this context, "systems techniques" are considered the entire process discussed earlier, within which are embedded predictive water quality modeling frameworks. The following criteria are offered for judging performance and impact of systems techniques:

1. The criterion of USEFULNESS, i.e. the degree of use of the frameworks in decision-making; does it really matter whether systems techniques are available?
2. The criterion of ACCURACY, i.e. the comparison of predicted water quality to actual water quality after a control program has been implemented; a post-audit analysis of the problem context.
3. The criterion of SERENDIPITY, i.e. whether systems techniques expose new, previously hidden interactions that are significant from a decision-making point of view.
4. The criterion of OWNERSHIP, i.e. the degree to which the community at large takes ownership of our principles through legislation, regulations and policies that reflect the insights of systems techniques.

Criterion #1 - Usefulness

There is little doubt that water quality modeling and system techniques are now used quite extensively in water quality management decision contexts. Negotiations for discharge permits, evaluation of varying alternatives, support for higher or lower degrees of treatment are all areas that now make extensive use of water quality modeling. On the other hand, the use of optimization planning models (e.g. cost minimization models) and optimal implementation programs (e.g. effluent charges) is apparently not as widespread. The relative extensive use of water quality modeling techniques has been justified on economic grounds, i.e. the belief on the part of regulatory agencies and dischargers that when properly applied, the application of the principles of predictive modeling is necessary but not sufficient to a rational decision. Tiemens (1986) reports that for the USEPA in Washington, D.C., about

100 projects have been reviewed over the past 8 years. In about one-half of these projects, the review, which included application of the principles of systems analysis in varying degrees, resulted in deferring the decision or a significant change in the proposed environmental control. The total capital costs of these projects impacted by the review was about $1 billion. Tiemens estimated that other smaller projects reviewed elsewhere at the state and regional levels may be an additional 200-300 in number but with a lesser overall total cost. So our techniques are useful and are being used in a variety of review contexts. A tributary to the Chesapeake Bay system serves as an illustration.

The Wicomico River. For this problem, (Salas and Thomann, 1975), data indicated a potential violation of a DO standard under low flow conditions due partly to a large diurnal variation of oxygen. Chlorophyll levels were high (e.g. 300 μg/l) in the vicinity of the input. The question was whether further removal of BOD was warranted. A detailed modeling analysis was conducted evaluating the various alternatives for control of the problem. Figure 1 shows the components of the DO deficit from this analysis. The maximum deficit

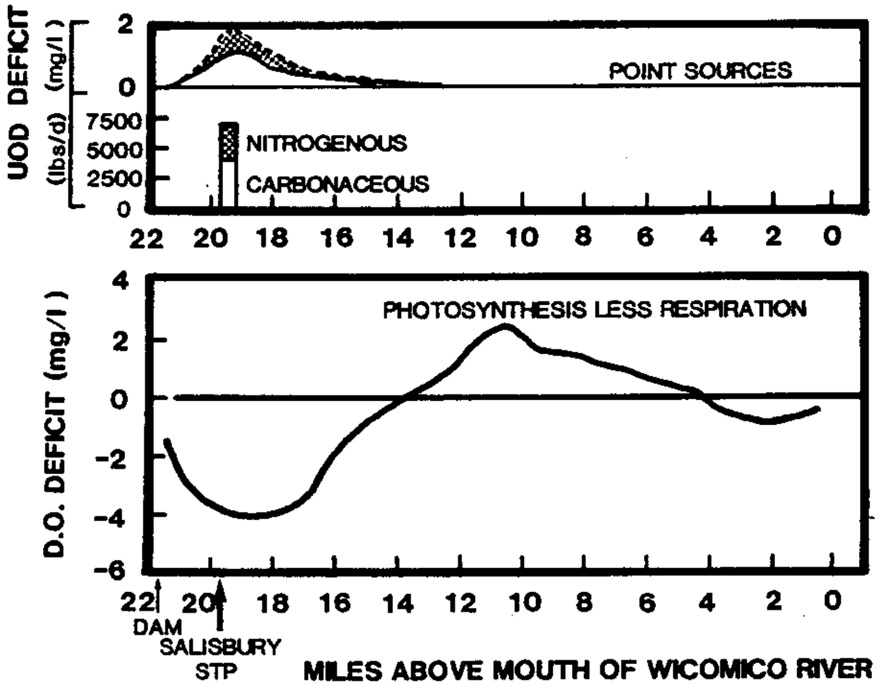

Fig. 1. Wicomico River, DO deficit components from water quality model indicating significance of phytoplankton respiration at mile 10 rather than point source carbon effects (Salas and Thomann, 1975).

(minimum DO) at site 10 is calculated to be due to phytoplankton respiration exceeding photosynthesis (together with the sediment oxygen demand) and not due to the point source of oxidizable carbon and nitrogen. It was therefore concluded that further reduction in these inputs would be only marginally effective and that emphasis should be placed on reducing the phytoplankton productivity through nutrient control. In the course of the decision making process, the conclusion was accepted by the regulatory authorities and a recommendation was made for construction of phosphorus removal facilities rather than additional carbon removal.

Criterion #2 - Accuracy

Since a predictive framework employing theoretical principles and past experience is at the heart of the water quality management system, it is crucial that our models be credible from an engineering point of view, but, equally important trustworthy and reliable from a management point of view. The forecasting ability of water quality models and uncertainties associated with predictions have been examined in detail elsewhere (see, for example, Beck and van Straten, 1983). Here a few simple examples are presented for illustration. Since DO analyses have such a long history, it is reasonable to evaluate how accurate our past analyses have been by examining the performance of DO models.

Post audit of DO models. Post audit is the evaluation of system performance following actual implementation of environmental control facilities. Three questions are addressed:

1. Do the actual DO data after a treatment upgrade is installed generally reflect the basic principles of DO models, i.e. does the DO go up when the BOD goes down?
2. To what degree are the DO models successful in predicting quantitatively the observed DO?
3. Does the accuracy of the DO models really matter in the decision regarding the treatment facilities to be installed?

In the work summarized here, an evaluation was made of 52 water bodies where some data were available on water quality conditions before and after treatment (HydroQual, 1983). Thirty seven states, five USEPA regional offices and six regional planning agencies were contacted, but in no case was there a complete compilation of water quality, biology, water use, cost or benefit data to perform a detailed post audit analysis. However, 13 water bodies did have some information for a review. The treatment changes included increases from primary to secondary and secondary to nitrification and advanced waste treatment.

Regarding the first post audit question, the data for the 13 cases indicated that the increase in DO normalized by the reduction in ultimate oxygen demanding (UOD) load was inversely proportional to river flow (Fig. 2).

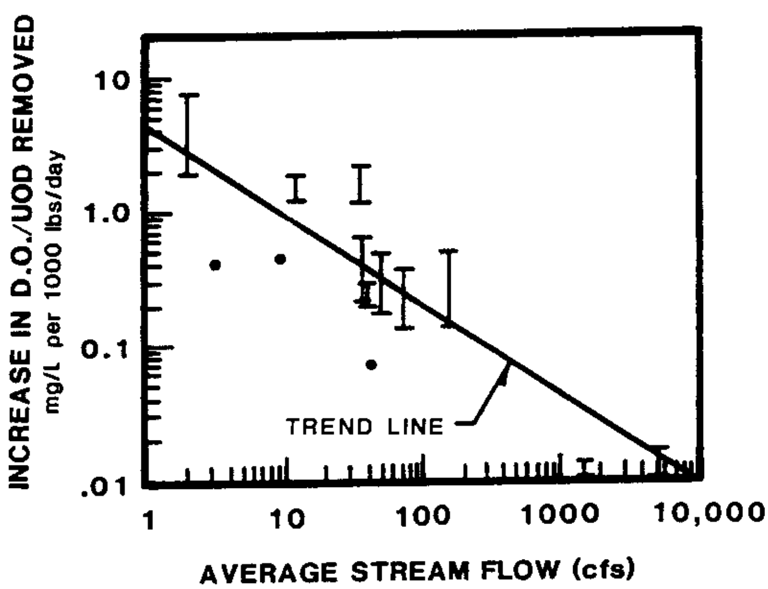

Fig. 2. Relation between actual DO increase (mg/ℓ) per 1000 lbs UOD/day removed and spatial average stream flow (cfs) - 13 water bodies. (From HydroQual, 1983)

At the very least, this is the simplest confirmation of the classical DO model framework. That is, from the basic DO sag equation we know that $\Delta DO/\Delta UOD$ reduction should be approximately inverse to the river flow.

To first approximation, then, our basic theory holds together and supports a fundamental tenet in DO systems analysis: the greatest DO improvement will result from facilities that provide the largest amount of UOD removal located on the smallest water bodies. The difficulty is that Figure 2 is a log-log plot, so the first approximation may not be all that satisfactory in a decision making context. Therefore, we need to take a closer look at the quantitative performance of DO models and address the second question.

Testing of six river models was performed by setting the conditions (i.e. river flow, temperature and effluent) for the appropriate "after treatment change". All model reaction rates were identical to those rates used in the original waste load allocation analysis. Root mean square (RMS) errors served as one quantitative measure of model accuracy in reproducing the data collected after a change in treatment. In post-improvement testing, RMS errors range from 0.0 mg/l to about 2.0 mg/l. The average error of 0.9 mg/l was somewhat larger than the RMS error of 0.7 mg/l associated with calibration of the six models.

Fig. 3 shows the correlation of observed to calculated mean DO concentrations for the calibration and post-improvement evalutions. The Figure clearly indicates that we do a good job in calibration partly because we have the data in front of us during this model calibration phase. On the other hand, the post-improvement comparisons, when we did not have the data a priori, indicate that the DO models tend to overestimate actual DO concentrations at levels less than 7 mg/l.

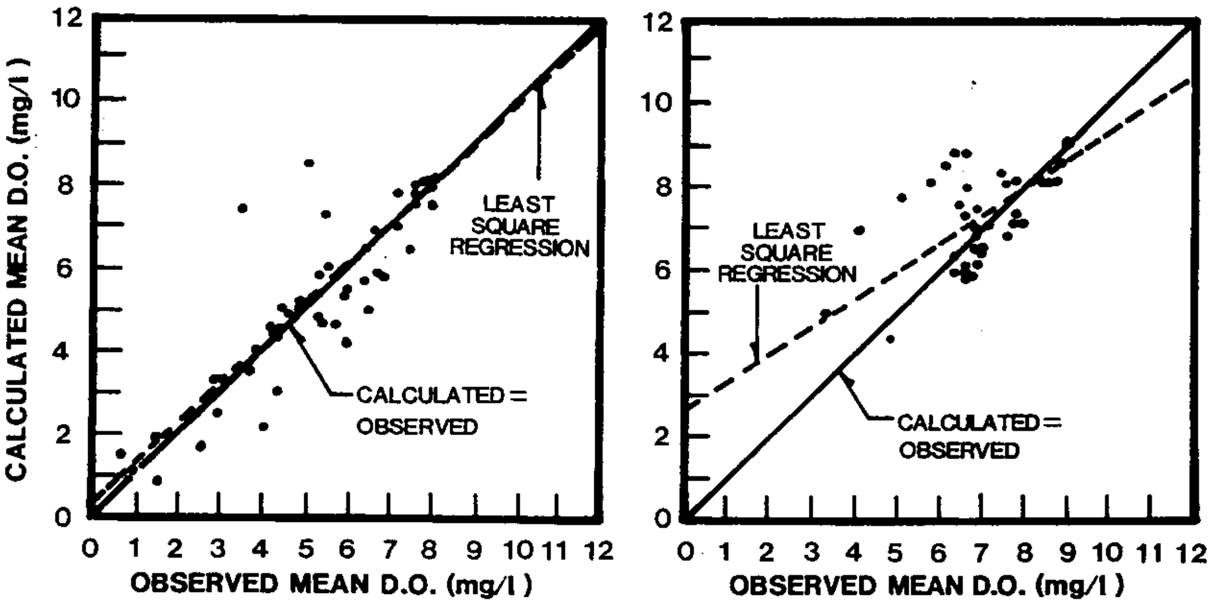

Fig. 3. Comparison of calculated and observed DO concentrations - 13 water bodies (a) from calibration stage of model, (b) from post-audit stage, after treatment upgrade. (From HydroQual, 1983)

Recognizing that many DO analyses are conducted without recourse to any data at all, it is important to also determine the credibility of the DO models where "simplified desk top" studies are conducted. Indeed, at least in the U.S., many more permits are probably issued by analysts who have never been within 100 miles of the river. The scenario therefore was as follows: an experienced water quality engineer was asked to analyze the DO in ten streams without looking at the DO data and only having available data on the river characteristics, e.g., flow and depth. Following the analysis, a comparison could then be made between the simplified analysis and the actual data. Quantitatively, the "simplified" models resulted in RMS errors that were 50% to 200% higher than the RMS errors developed from more complex data-available analyses. The average RMS error for the ten river analyses was about 2.0 mg/l.

The answer to the second question above is therefore somewhat sobering. With a detailed model construction and using reasonably extensive (and expensive) data sets, the RMS error in the actual subsequent comparison to DO levels after treatment upgrade is about 0.9 mg/l. Simplified, desk top analyses double that error. With these kinds of errors, one wonders about all the discussion that sometimes ensues in permit negotiations over a few tenths of a mg/l DO.

Now, to the third question, i.e. do these errors really make any difference in the decision-making phase? In the preceding discussion, there are two types of errors that may occur in the comparisons: the first error is overestimation of the water quality improvement for a given level of treatment. Therefore, water quality will be less than actually thought after treatment upgrade and a water use interference may occur that was not predicted. The second error is underestimation of the water quality improvement resulting in overdesigned treatment facilities and an overexpenditure of funds. The first error can be termed a water quality error (i.e. quality (use) will be less than projected). The second error can be thought of as a facilities error (i.e. the facility is overbuilt to meet target water quality.)

Comparisons for 10 rivers were made between the decisions reached using simplified desk-top techniques as compared to detailed modeling approaches. Simplified modeling could have potentially resulted in four water quality errors and two facilities errors. In four cases, the decision was identical. The comparison, of course, assumes that the more complex model analyses with available data results in "correct" decisions, which in fact is not always the case. With respect to an upgrade to nitrification facilities, the comparison indicated that the simplified models reached the same decision in nine of the ten cases. This is due principally to the step increases in UOD reduction with the installation of nitrification facilities.

One concludes from this post audit analysis that simplified DO models and to a lesser extent, more sophisticated DO models do not do very well in predicting actual values of DO after a treatment upgrade. RMS errors of 1-2 mg/l DO are the bad news. The good news is that from a decision-making point of view, it doesn't seem to make all that much difference especially for an upgrade to nitrification.

The Potomac Estuary Case. This estuary has been the subject of water quality management and modeling efforts for several decades. Freudberg (1985) has reviewed the history and the implications of the modeling work. In the late 1960's, extensive algal blooms developed in addition to a depressed oxygen condition in the Washington, D.C. vicinity. As a result of modeling efforts by people such as Jaworski et al. (1971), significant reductions in incoming carbonaceous and nitrogenous BOD loading was accomplished along with significant reductions in point source phosphorus loading. Considerable controversy surrounded the phosphorus reduction strategy since it was argued that nitrogen was the limiting nutrient and that nitrogen should be controlled. Concern was also expressed over the release of phosphorus from the sediment. The phosphorus removal strategy was founded on the notion that with sufficient reductions of phosphorus, that chemical could be made the limiting nutrient. Since it was considered cheaper to remove phosphorus than nitrogen, the phosphorus removal program was instituted. Fig. 4 shows the reductions in phosphorus during the late 1970's and early 1980's.

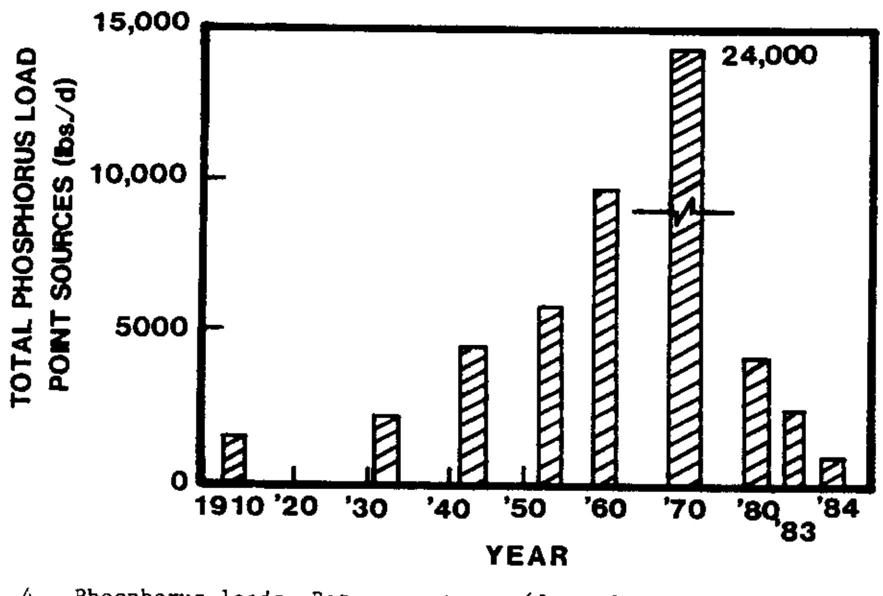

Fig. 4. Phosphorus loads, Potomac estuary (Jaworski et al., 1971; Thomann et al., 1985; Metro. Wash. Council of Govt's, 1985).

A major algal bloom occurred in 1977 following the first stage of the phosphorus reduction program. An intensive effort was then undertaken to update the modeling framework for eutrophication and the Potamac Eutrophication Model (PEM) was constructed (Thomann and Fitzpatrick, 1982). This model was calibrated and verified using 7 years of data. The PEM included sediment interactions and a fully developed time variable phytoplankton-nutrient species framework. The model was subsequently utilized for analysis of alternatives.

Then in 1983 with the phosphorus reductions now almost fully in place (2100 lbs Total Phosphorus (TP)/day versus 24,000 lbs TP/day in 1970), another major bloom occurred of the blue green alga, Microcystis aeruginosa. The embayments reached levels of almost 800 ug chl/l with the main channel reaching concentrations of about 300 ug chl/l. With the estuary a brilliant green, the treatment plants for Washington, D.C. removing phosphorus to levels of about 0.4 mg TP/l (compared to the permit levels of 0.2 mg/l), there was some consternation, to say the least. The questions abounded. Was this a major setback? Was the wrong nutrient being removed? Was the PEM faulty? Would the situation be relieved when treatment plant discharge levels actually reached the target effluent concentrations of 0.2 mg TP/l? What actually was the cause of the bloom?

The Metropolitan Washington Council of Governments, the user agency, applied the unchanged PEM with the 1983 meteorology and hydrology. The model tracked the onset of the bloom up to about 100 µg/l by the end of July but then failed to reproduce the further intensification of the bloom. The principal reason for the failure of PEM to capture the full bloom was that the model ran out of phosphorus (see Fig. 5). Sensitivity analyses indicated that there was an apparent additional phosphorus source of about 4000-8000 lbs/day not included in PEM. Fig. 5 from Freudberg (1985) shows the effect of including this source and it is seen that it partially explains the observed bloom.

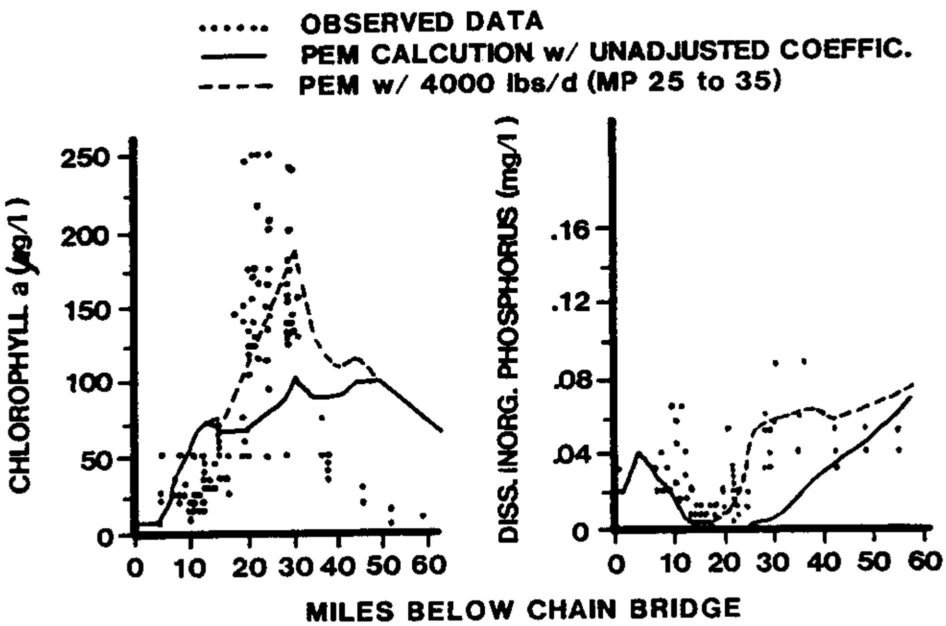

Fig. 5. August 28, 1983 bloom, Potomac estuary (Freudberg, 1985)

A series of hypotheses were investigated (Thomann et al., 1985). It was concluded that the most likely mechanism for the phosphorus input was an enhanced aerobic sediment phosphorus release resulting from the high pH in the overlying water (Di Toro and Fitzpatrick, 1984). Contributory mechanisms included the effect of two-dimensional vertical circulation trapping nutrients and elevating the available phosphorus concentration.

In summary, the PEM failed to capture the full 1983 bloom because of a combination of a "missed mechanism" and a simplification of the estuarine transport in the lower reaches of the estuary. However, one should not conclude, I believe, that the Potomac situation is an example of where water quality modeling is of little value. Rather, without PEM, the evaluation of what happened in the 1983 bloom would have been reduced to qualitative discussion of the data. The availability of PEM immediately pinpointed the principal problem: more phosphorus in the estuary than could be accounted for from previous observations. The

Potomac illustration does indicate clearly the need for extensive monitoring and post-auditing of our work, both to demonstrate credibility, or lack thereof, and to uncover previously hidden or unsuspected mechanisms that become important after treatment programs are in place.

Model Quality Assurance. In recent years, a wide variety of individuals with widely varying educational and experience backgrounds are "practicing" systems analysis in water quality management. These are people representing local, state, or regional governments, or individual dischargers. They have been placed with the responsibility of doing calculations to support the decision-making process.

Recently, Gallagher (1986) completed the results of a variety of water quality modeling analyses prepared to support applications for AWT systems. The compilation reveals some surprising insights into where we are in the actual usage of water quality models on the more local day to day decision making level. A review of Gallagher's compilation indicates that quality assurance of model formulation and application is essential for the maturation of the discipline.

The following problem areas in model formulation arise:

1. Input data are not always carefully checked.
2. Software coding errors occur.
3. Variables are incorrectly used in formulae.
4. Parameters not always checked for reasonablenss of range.

In the calibration and verification stage, the most difficult areas appear to be:

1. Parameter estimates may vary significantly enough between analysts to result in significantly different decisions; reaeration rate determination is an example.
2. Sediment-water column interaction uncertainty persists and often cannot properly address the issue at hand; the calculation of nutrient release, toxic substance release and sediment oxygen demand within the model framework are examples.
3. A lack of fuller understanding of the variety of mechanisms that may "explain" data hampers the credibility of models; improper assignment of deoxygenation and nitrification coefficients sediment interactions, and algal effects on BOD are examples.

In the projection and evaluation of alternative phase, the most difficult areas are:

1. Projection of parameter changes under different environmental controls, e.g. microbial degradation of toxic substances.
2. Projection of sediment water column interactions under environmental conditions that have not yet been observed.

The difficulty with these concerns is that errors in coding or input, unreasonableness of parameters or failure to recognize mechanisms are not always easily uncovered in the analysis. Improper overestimation of one parameter may be compensated for by the analyst by underestimating another parameter. In the evaluation of alternatives, erroneous projections of effectiveness may then be made.

Criterion #3 - Serendipity

This criterion addresses the issue of whether systems techniques reveal previously undetected mechanisms or interactions that have a bearing on the degree and kind of environmental control. The preceding case of the pH mediated release of phosphorus from the sediments of the Potomac estuary is an example. If systems techniques on occasion indicate such serendipitous results then indeed it is worthwhile to use such techniques in the decision-making process.

Cost minimization frameworks often display such behavior where optimum solutions indicate cost trade-offs that were not previously apparent. The degree to which such solutions have been included in actual program implementation is not clear. However, in the area of water quality modeling serendiptious results may be directly incorporated into the decision on environmental control.

The Back River Case. For the Back River, a tributary of the Chesapeake Bay (Thomann et al., 1981), phytoplankton levels were of the order of 200 μg chl/l and reached maximum levels of greater than 500 μg/l. Simple model analyses indicated that a 2 mg/l orthophosphorus effluent limit would not be sufficient to reduce phytoplankton. Subsequent interest then focused on AWT to total phosphorus levels of 0.2 mg/l versus an effluent relocation to Baltimore Harbor. More detailed modeling indicated that there was virtually no difference in the orthophosphorus concentration in the Back River between the two alternatives (see Fig. 6). This result was not anticipated since relocating the discharge out of Back

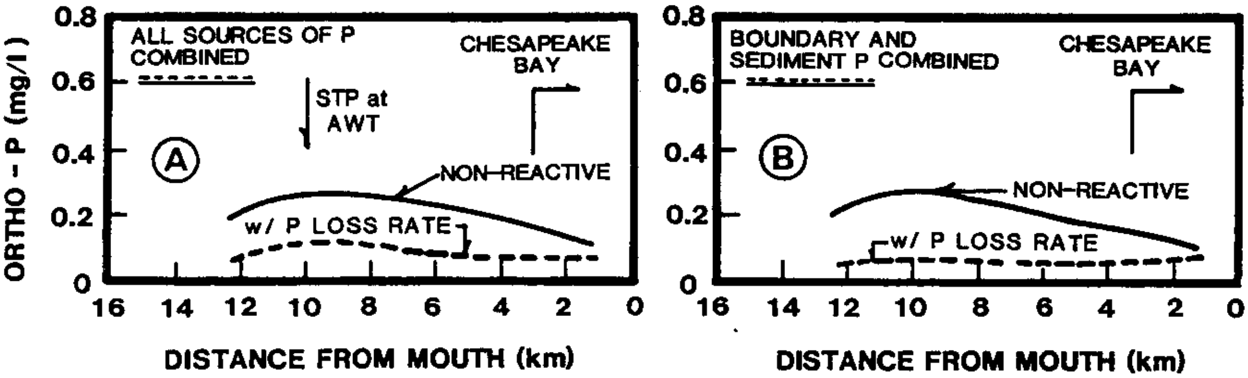

Fig. 6. Calculated phosphorus levels in Back River with (a) AWT and (b) effluent diversion

River was thought to be more effective. But decreasing the input load by treatment was offset by diversion because the loss of flow through the Back River enhanced the effects of sediment release and allowed increased uptstream migration of the Bay boundary waters. As a partial consequence of this result, effluent relocation was not implemented. Rather, a staged program of treatment upgrade to high levels of phosphorus removal has been implemented. The first phase is under construction with phosphorus removal scheduled for a later phase. I believe that many more such examples exist, but are often buried in the archives of individual studies. It is important to document such results and especially to document the economic consequences of these types of unexpected behavior.

Criterion #4 - Ownership

Aside from the more technical uses of our product, to what degree has our work been incorporated or adopted into legislation and policy related to water quality management and control? On the U.S. side, it is encouraging that some, but not all of the basic principles of water quality management have been recognized and are reflected in various laws, regulations and policies. It is discouraging that some principles have been largely ignored.

The most fundamental principle on which our entire work is based is that the aquatic environment has a certain capacity for absorbing and transforming residual inputs without creating undesirable impacts. Under this fundamental principle, there follow the key steps outlined above. The entire process is iterative and following an evaluation of the costs, benefits and risks avoided of a series of alternative objectives, a final determination is made as to the most desirable set of water uses and water quality standards. Treatment of residuals is based on the specific characteristics of the water body, and various economic, social and policy constraints. Technology-based effluent limits are therefore justified only where the ability to establish receiving water standards does not exist.

If the fundamental principle of environmental absorption of wastes were not recognized in legislation and policy, then the reason for our work would simply cease. The regulatory program would then be an effluent based program with effluents set at the highest technologically attainable level. The program is utter simplicity: either you treat to the highest level possible or else you are in violation. The process is then strictly a legal matter to be argued under adversary proceedings.

In the U.S., our national legislation has essentially embraced this latter concept with our zero-discharge goal and to a related degree our program of effluent based requirements for specific industries through Best Practical Treatment (BPT) and Best Available Treatment (BAT). On the surface then, one could argue that the most fundamental principle of systems analysis and water quality management has been ignored and that we have not only lost the battle, but the war as well. After all, there is really no rational engineering/scientific justification, in my opinion, for an across-the-board requirement of secondary treatment or technology-based industrial chemical control; such effluent requirements to be imposed regardless of the type of water body, or resulting impact on water quality or aquatic ecosystem. We arrived at that point, at least as far as I can see, from a mistaken perspective that the water quality standard route was not practical from a regulatory point of view. The lack of practicality centered around the belief that our ability to assess water quality impacts was imperfect and that it would not be possible to evaluate each site specific situation.

In actual implementation however, the situation is often quite different. Questions abound about the cost of effluent based programs and about the relative impact of effluent based programs on water quality and use. At least one result of this questioning has been the reevaluation of the requirement for secondary treatment in coastal waters with high effluent dilution, although even here the results have been spotty. Ease of regulation and political and social constraints appear to have been the major factors for not incorporating the principles of water quality management more fully.

On balance, however, the previous principles have been incorporated in a variety of ways albeit not totally. Examples include (1) the requirement by the USEPA that any request for funding for an AWT project costing more than $3 million be subject to a rigorous review of costs and effectiveness; (2) the Great Lakes Water Quality Agreement (International Joint Commission, 1985) and (3) the water quality-based procedures and policies for toxic substances control (USEPA, 1985).

For the first example, the Great Lakes Agreement explicitly calls for target loadings of phosphorus from Canada and the United States for Lakes Erie and Ontario to meet specified target objectives. The significant economic savings resulting from the second example have been noted above in this paper (see Criterion #1 - Usefulness). In the third example, due recognition is given of dilution effects, downstream decay and probabilistic interactions in establishing effluent levels for chemicals or whole effluent toxicity.

The Delaware Estuary case. Intensive water quality management studies began for this estuary in the early 1960's and resulted in waste load allocations for individual discharges to meet water quality standards promulgated by the Delaware River Basin Commission (DRBC)(See Thomann, 1972 for discussion of the full study). Emphasis was on the DO. Extensive use was made of water quality modeling and cost minimization techniques in arriving at trade-offs between various alternatives. Actual program implementation incorporated cause-effect relationships to establish waste load allocations, but did not formally include any optimal control strategy. In the ensuing two decades, this program has remained largely in place. Significant reductions in CBOD have been accomplished (about 70% reduction from 1964 to 1986)(DRBC, 1984, 1986a,b), but since NBOD was not allocated, NBOD loads have remained virtually constant. The DO response is shown in Fig. 7. The trend in improvement is close to the projection made 20 years ago. The water quality standard that was adopted as a result of the earlier efforts, has almost been achieved after two decades, although there is still some variation of DO below the 24 hour minimum average standard.

To what degree did the application of systems techniques in the 1960's make a difference in the water quality management program for the Delaware estuary? Since the average percentage CBOD removal is 86-89%, one could argue that such levels would have been mandated in any event by the across-the-board secondary treatment equivalent of U.S. Clean Water legislation. On the other hand, the implementation of a rationally based program provided a continuous defensible framework for regulatory purposes. By having an allocation system that included a reserve capacity, provision for reallocation and renegotiation, the Delaware program could proceed in a reasonably orderly fashion and aim toward a water quality objective that was realistic, cost effective and consistent with the goals of the Delaware community.

In summary, the record for this criterion of including the basic tenets of our work in legislation and policy is a mixed one at best. The objective seems to be laudable, i.e. a blend of a program that is easy to implement with a program that is scientifically well founded and cost effective. However, the widespread application of technology based effluent programs with little evaluation of the effectiveness and costs of such programs testifies to the dominance that the ease of regulation can assume when the public pressure to get something done is intense. I believe that the continued uncritical use of such effluent-based programs is a mistake and is counter-productive in the long run. A detailed study of this type of policy would seem to be most warranted. Nevertheless, the need for rational decision-making is often so compelling that the techniques of our work can no longer be avoided and are subsequently included in the decision-making process.

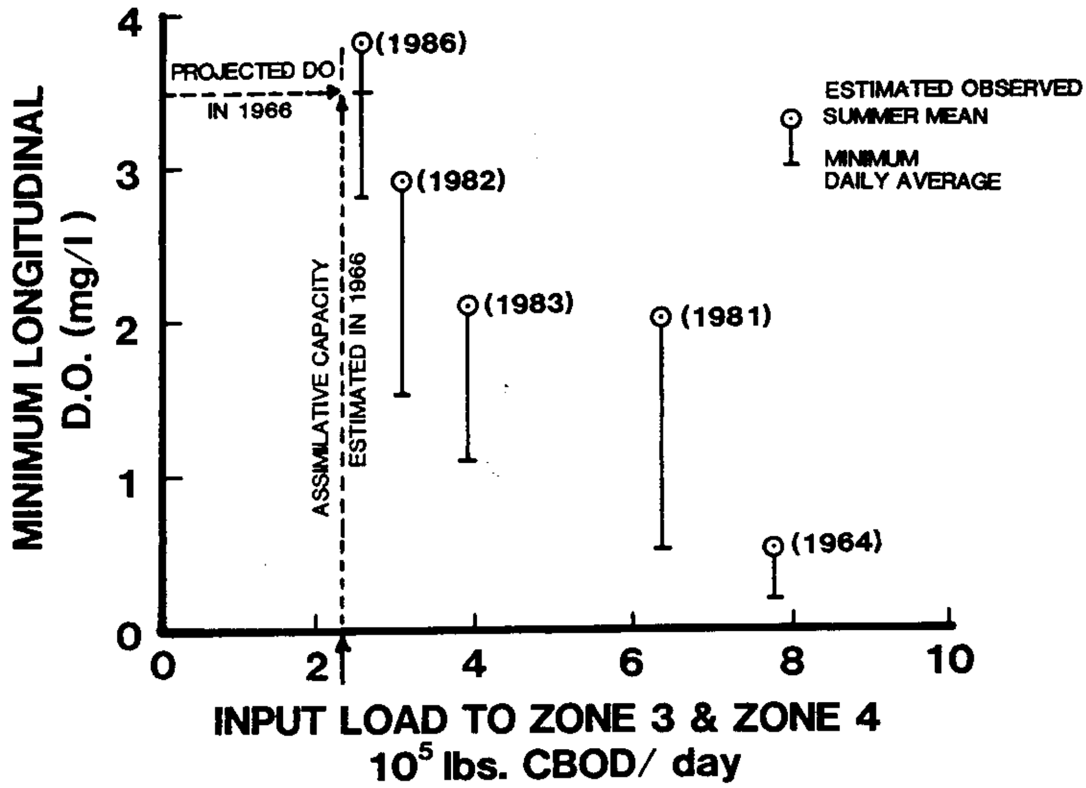

Fig. 7. Response of Delaware estuary DO to reductions in CBOD compared to to projection made in 1966. (Data from Thomann, 1972; DRBC, 1984, 1986a,b). Different years have different fresh water flow inputs.

CONCLUSIONS AND SUGGESTIONS

Over the past quarter century, systems analysis in water quality management has come of age. More specifically, water quality modeling techniques coupled loosely to cost analysis frameworks are now utilized in varying degrees at all levels of decision-making. From early screening of potentially toxic chemicals using generic water bodies to detailed analyses of alternative treatment strategies, systems techniques especially water quality modeling have been used extensively. Reflection on the preceding criteria and our few illustrative examples indicates that indeed systems analysis has had an impact, in fact a decided impact. The difficulty is that the impact is not widely documented or known among those responsible for key budgetary and regulatory decisions. It is now considered almost essential that evaluations of cost and effectiveness of alternative actions be made using state of the art modeling and systems techniques. Practical decisions are made using these frameworks. The overall supposition is that such decisions are better decisions because they are more informed, the rationale for the decision is more structured and has a firmer engineering/scientific foundation. As a result, there is the hope that decisions in which systems analysis played a role will be more universally acceptable. That this universal acceptance is not always achieved is obvious and the endless, sometimes contentious, arguments over water quality modeling give testimony to this lack of universal acceptance. Although there is still a lingering group who see systems techniques and attempts at a priori alternative assessment as modified "witchcraft", the voices are getting dimmer. There are exceptions, of course, where the clamor for maximum technological removal of wastes sometimes drowns out a plea for a more balanced approach. The present directions of control of chemical discharges may fall into this category unless there is a continual campaign for the use of systems techniques for more informed decision-making.

We can properly take credit at least to some degree for a general incorporation of some of the basic principles into policy and legislation. The use of effluent discharge permits, seasonal treatment and associated seasonal permits and writing the requirement for quantitative assessment directly into policy are examples. On the other hand, many key ideas of optimization apparently still lie dormant: cost minimization, effluent charges, optimum program implementation are examples.

Where we need to do more work is obvious. As noted previously, there is a need to document our work more extensively, especially those specific examples where decisions have been impacted, costs have been reduced, benefits realized and new interactions exposed. It is suggested then that a world-wide effort be devoted to compiling such work with the expressed purpose of displaying the considerable economic advantages from using systems techniques in water quality management against a strictly effluent based approach.

Technically, we lack defensible tested models of sediment interactions, effects of chemicals on aquatic ecosystems and stochastic frameworks that have been calibrated and verified. Increased quality assurance of model formulation and calculation is necessary and indicates a need for upgrading the training of operational uses of water quality systems techniques. State of the art computer technology, e.g. graphics, and use of expert systems should be fully exploited to meet this critical need for properly trained analysts.

We have filled an essential role admirably during this past quarter century. Our work has been good; most importantly, it has been truly beneficial in fully respecting the intricate balance that must be struck between a healthy environment and a healthy economy.

ACKNOWLEDGEMENTS

The thoughtful discussions and input of Thomas Gallagher of HydroQual, Inc., Charles App and Myron Tiemens of the USEPA, and the assistance of Bruce Lin and Dennis Scannell, graduate students at Manhattan College, are gratefully recognized. The readiness of the Delaware River Basin Commission to be of help and provide data is also appreciated. Special thanks are due to Mrs. Eileen Lutomski for her patient typing and retyping of the manuscript.

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