WATER QUALITY MODELING AND THE
WASTELOAD ALLOCATION PROCESS

By

BRENT EUGENE VANMETER

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Oklahoma State University

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Thesis Approved:

Don F Kincannon
Thesis Adviser

Marcia H Bates

John N. Veenstra

Enos L. Stover

Norman N. Durham
Dean of the Graduate College

To Trisha

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

Oxygen plays an important part in the various chemical and biological reactions which occur in nature. In fact, it is undoubtedly the single most important element in any life cycle. It is necessary to sustain life both on land and in water and since there is a finite amount of oxygen available to organisms in water due to the properties of saturation, (1), a depletion of the dissolved oxygen (D.O.) can result in the destruction of the life forms dependent on it.

Oxygen depletion can be the result of the introduction of an oxygen demanding substance into a natural water system. This may be in the form of domestic or industrial sewage (organic matter) which is being discharged into a stream or river. As a result of this discharge, biochemical reactions take place between the organic material, decomposing bacteria and the oxygen. As the volume of organic material increases so does the microbial population, and as a result of this interaction, the oxygen is used by the microbes until it is depleted, leaving none for the naturally occurring aquatic life.

The previous scenario is a simplified example of the complex problem of water quality management, a problem which may well be the number one concern of the 1980's. The topic of water quality management through the use of mathematical models and wasteload allocations to regulate sewage discharges into the nation's water systems has been one of increasing concern since Phelps benchmark work in 1909 and 1925 (2)(3). Modeling has more recently, through the passage in 1972 of Public Law 92-500 and its subsequent amendment to PL 95-217 in 1975, developed into a very sophisticated science in some respects, yet remains subject to much criticism due to the inherent use of assumed values and "engineering judgement" in lieu of hard and fast rules.

A prime example of the uncertainty of modeling lies in the use of the BOD test in the assay of organic loads and in the determination of the deoxygenation constant K_1. Gaudy and Gaudy (4) provide a great deal of insight on the "uses and misuses" of the BOD test and along with Hiser and Bush (5) evaluate the use of COD as an alternate test of pollutant load. The BOD test has been called probably the most misunderstood and most abused of all analytical tools used in stream system analysis (6).

The purpose of this report is not to belabor the BOD test or the many other complex and controversial components that make up a water quality model. There have been many books and papers written on nearly every aspect of this subject and many more are yet to come. This report will provide the basic concepts related to water quality modeling and wasteload allocations and show how they can be used as effective management and decision making tools.

CHAPTER II

LITERATURE REVIEW

The ability of a stream to degrade its organic load and regenerate its oxygen supply is commonly called "self-purification," (7). Velz correlates the streams assimilative phenomenon to a rational accounting system of oxygen credits and debits (8). Nemerow (6) concisely summarizes these creditors and benefactors with regard to stream oxygen balance; they are as follows:

Creditors depleting oxygen

1. Organic matter in continuously flowing water
2. Slime growth on attached rocks, debris, and other surfaces over which water flows
3. Primary organic bottom sludge deposits (benthal demand)
4. Secondary organic bottom deposits (dead algae)
5. Temperature rises causing oxygen vapor loss and increases in microbal metabolism
6. Fish and other aquatic respiration needs
7. Organic contamination in tributary streams
8. Stream salinity content

Benefactors contributing oxygen

1. Surface reaeration due to the physical reaction of the

air and water

2. Photosynthesis

3. Temperature decrease increasing oxygen saturation potential and decreasing microbiological metabolism

4. Dilution from uncontaminated streams.

It is obvious that if the totality of the creditors is greater than the contributions of the benefactors an oxygen deficit is incurred. The loading of the stream with organic matter (creditor #1) is definitely the most significant in terms of the quantity and ultimate effect upon the stream. Consequently, the normal controlling procedure has been directed toward improving industrial and municipal wastewater treatment facilities in an effort to reduce the demand from organic loadings at identifiable point sources.

A stream is a conglomerate of complex biological, chemical, physical, and hydraulic factors. To determine the combined effect of these various factors, mathematical models capable of representing some of the more important interrelationships between the variables have been developed (9). The primary objective of many studies is to select, develop, and implement such a model that simulates the variations of oxygen demanding parameters over time at points along selected stream segments.

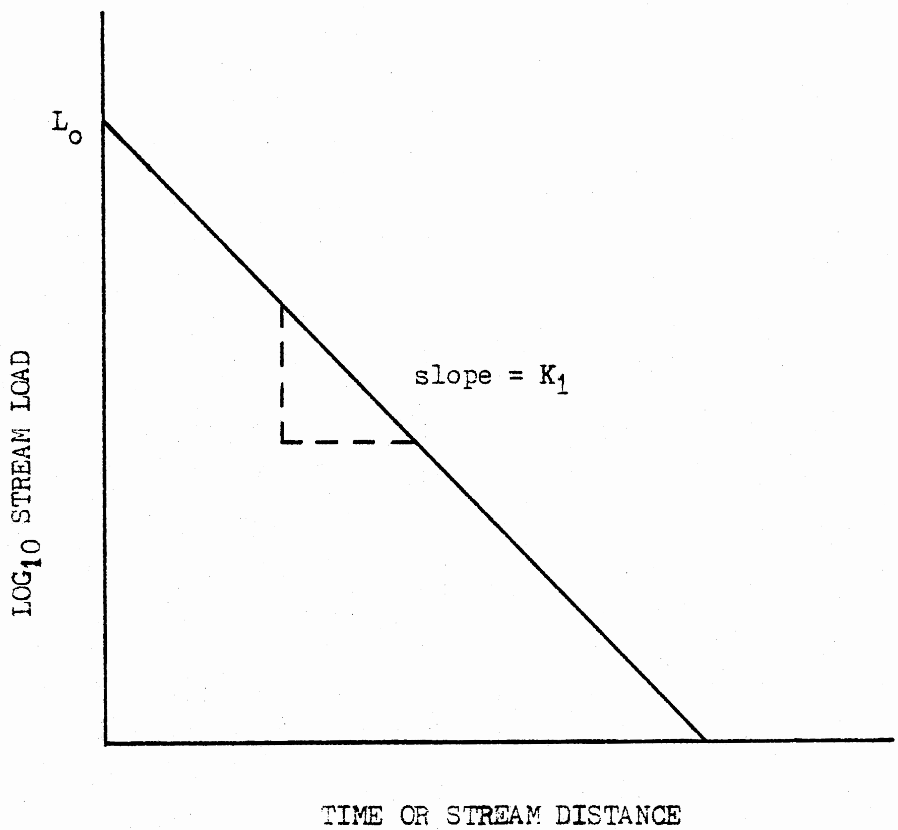
A basic set of established, yet simplified, mathematical formulas, based on pertinent stream data, can adequately define and predict the assimilative capacity of most streams. The basic Streeter-Phelps equations developed in 1925 and expanded by Phelps (10) in 1944 are the essential core of all major stream assimilative models. These equations or updated modifications are utilized in practicially all significant contemporary stream studies. The formulas have repeatedly been found to have the best theoretical mathematical rationale and validity. Numerous texts expound on the Streeter-Phelps equation and its derivations (6) (8) (11) (12) (13).

Development of Rate Coefficient

From information derived from the Ohio River Study in 1925, as well as many others, the Streeter-Phelps equation was developed. It allows a scientist to estimate the effect of a pollutant load to a stream with respect to dissolved oxygen. The rate of depletion of oxygen in a stream (K_1) can be developed mathematically from the first-stage BOD curve of the waste. The methods for determining the value for the deoxygenation constant (K_1) are discussed by Gaudy et al. (14). Gaudy et al. evaluated sixteen (16) methods for determining the constant (K_1) solely as a basis of reference. The most common methodology employed is one based on work by Streeter and Phelps and expanded upon by Fair, Geyer, and Okun (12).

The rate of deoxygenation is directly related to the organic matter remaining to be oxidized at a given time. This load may be made up solely of carbonaceous BOD, oxidizable organic nitrogen that exerts a nitrogeneous oxygen demand (NOD) or a combination of both. The value of K_1 can be expressed as the slope of the straight line obtained by plotting the log of the organic loads (BOD, NOD) against time (days) or distance (miles) on semilog paper. It is extremely notable that while this traditional plot (Figure 1) of data usually results in a straight line, often two lines may be dictated (Figure 2). For example, when a municipal effluent is discharged into a stream, that effluent may contain relatively high settleable solids. As a result, in the vicinity of

Figure 1. K_1 Plot of LOAD vs. TIME


L_o
slope = K_1
LOG_{10} STREAM LOAD
TIME OR STREAM DISTANCE

the outfall, the removal of ultimate oxygen demand is accomplished by the physical settling and the oxidation of the organic matter simultaneously. Oxidation of the ultimate oxygen demand requires dissolved oxygen, while removal of the ultimate oxygen demand (organic matter) by settling does not directly use oxygen. As one proceeds downstream, the physical removal is completed and only oxidative removal remains (15).

The deoxygenation constant based on the carbonaceous load of the stream is commonly designated as K_1, while the deoxygenation constant for the nitrogenous load is referred to as K_n (ultimate NOD vs. time or distance).

The rate of the deoxygenation reaction is a function of temperature as biochemical reactions are increased with increases in temperature. The best mean conversion formula, from a comparison of several independent studies, as adopted by Streeter and Phelps (6), is :

$$K_1 \text{ at } T_1 = D_1 \text{ at } T_2 \, (1.047)^{T_1 - T_2} \qquad (1)$$

where:

K_{T1} = reaction rate at temperature 1

K_{T2} = reaction rate at temperature 2

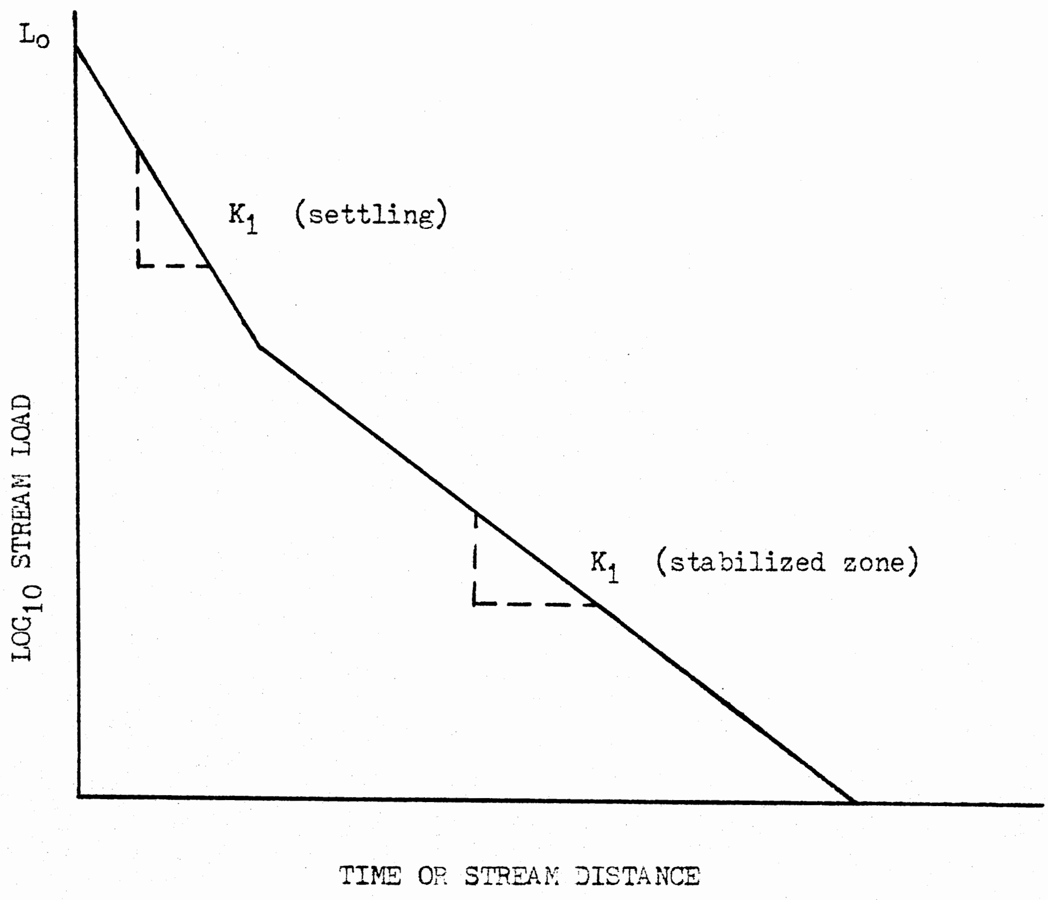
T_1 = temperature 1, ^oC

T_2 = temperature 2, ^oC

Stream Sage Curve and Reaeration Rate, K_2

Nature provides a physical mechanism for counteracting the depletion of oxygen in a stream. This mechanism is reaeration, a means by which atmospheric oxygen enters the surface water. Reaeration is a rate phenomenon directly proportional to the oxygen deficit in the water (the greater the deficit the greater the rate of solution of oxygen).

Figure 2. K_1 Plot with Allowance for Settling


Lo
K1 (settling)
K1 (stabilized zone)
LOG10 STREAM LOAD
TIME OR STREAM DISTANCE

Thus, the reaeration rate (K_2) is a function of both the deoxygenation rate (K_1) and the oxygen deficit. Streeter and Phelps originally stated this relationship as follows (6):

$$\frac{\Delta d}{\Delta t} = \overbrace{K_1 L}^{\text{deoxygenation}} - \underbrace{K_2 D}_{\text{reaeration}} ==== K_2 = K_1 \left(\frac{L}{D} - \frac{d}{2.3\ tD} \right) \qquad (2)$$

where:

K_1 = deoxygenation rate

K_2 = reaeration rate

L = average of the upstream and downstream organic load

D = average of the upstream and downstream oxygen deficit

t = change in time

d = change in deficit

By differentiating this basic equation, Streeter and Phelps developed the famous "Sag Curve Equation" :

$$D_t = \frac{K_1 L_o}{K_2 - K_1} (10^{-K_1 t} - 10^{-K_2 t}) + D_o (10^{-K_2 t}) \qquad (3)$$

where:

D_t = the oxygen deficit at a given time

L_o = the ultimate organic load

D_o = the initial oxygen deficit

t = a given time

This equation has been used in practically all important studies of stream assimilative capacity up to now . The equation defines the basic sag curve emboding the effects of deoxygenation and reaeration along a given stream segment. Figure 3. illustrates this basic relationship. It is apparent from the sag curve that as the two reactions proceed, a minimum D.O. results at a point along the stream. This is typically called the "critical point or the critical deficit (D_c)" and is said to occur at a "critical time (t_c)." The time to the critical oxygen sag point may be obtained by differentiating the sag equation and solving for t_c :

$$t_c = \frac{1}{K_2 - K_1} \log \left\{ \frac{K_2}{K_1} \left[1 - \frac{D_o(K_2 - K_1)}{L_o(K_1)} \right] \right\} \tag{4}$$

Numerous modifications of the original sag equation (16) (17) have been implemented in contemporary stream models. A typical example is illustrated by the following sophisticated equation:

$$\begin{aligned} D_t = \; & D_o(10^{-K_2t}) + \frac{K_1L_o}{K_2-K_1}(10^{-K_1t} - 10^{-K_2t}) \\ & + \frac{K_nN_o}{K_2-K_n}(10^{-K_nt} - 10^{-K_2t}) + \frac{R-P}{K_2}(1-10^{K_2t}) \\ & + \frac{S}{K_2}(1-10^{K_2t}) \end{aligned} \tag{5}$$

where:

D_t = oxygen deficit at a given time; mg/l

t = a given time; days

K_1 = deoxygenation rate of ultimate carbonaceous load; day^{-1}

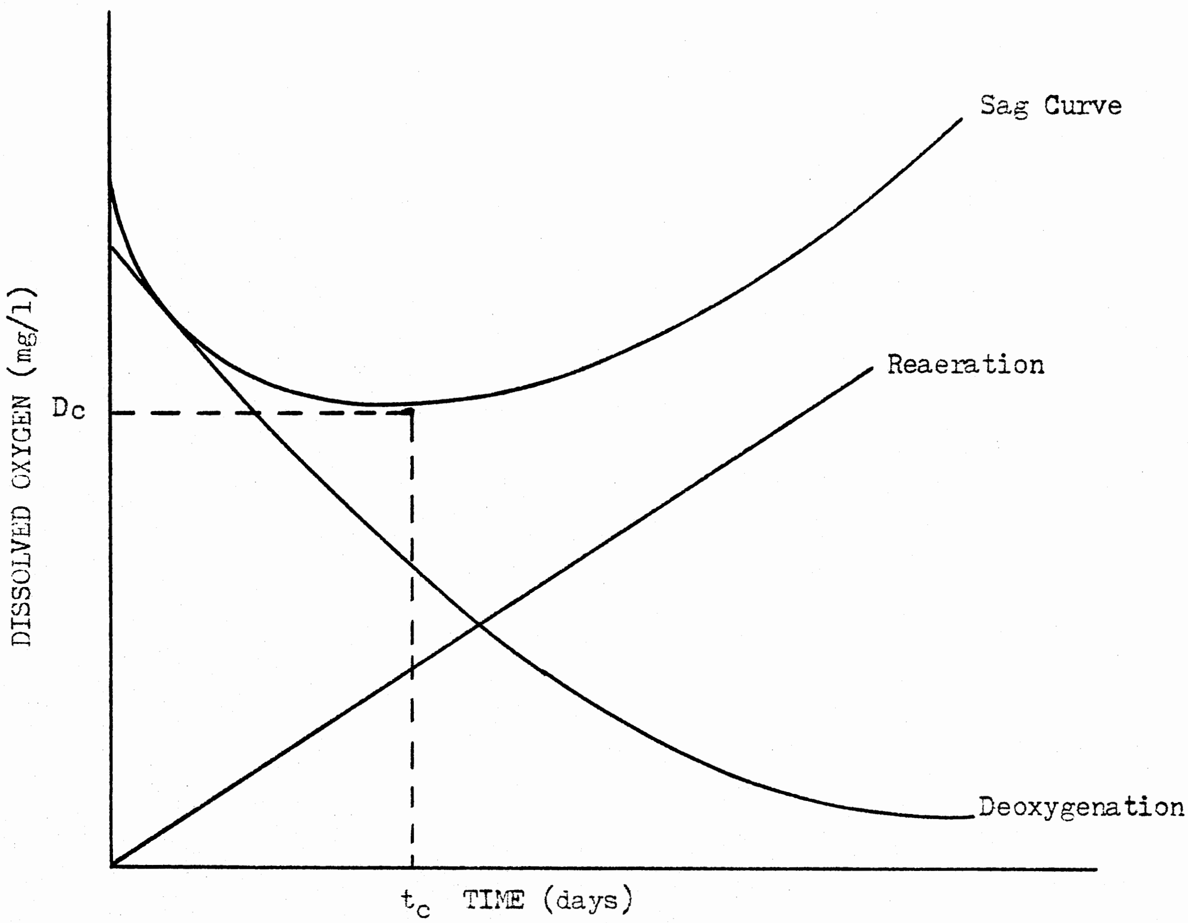
K_n = deoxygenation rate of ultimate nitrogenous load; day^{-1}

K_2 = reaeration rate; day^{-1}

L_o = ultimate carbonaceous load; mg/l

N_o = ultimate nitrogenous load; mg/l

Figure 3. Relationship Between Deoxygenation and Reaeration


Sag Curve
Reaeration
Deoxygenation
DISSOLVED OXYGEN (mg/1)
Dc
t_c TIME (days)

R = deoxygenation factor due to respiration in stream; mg/l/day

P = Oxygen produced by photosynthesis in stream; mg/l/day

S = deoxygenation factor due to benthic deposits in stream; mg/l/day

Although the actual stream reaeration coefficient (K_2) can be readily calculated by means of stream D.O. data and utilization of the sag curve equation, numerous other methods have been proposed over the years. As early as the Streeter-Phelps' work in 1925, it was proposed that K_2 depended upon physical factors of streams such as velocity, depth, slope, and channel irregularity. The majority of these 'empirical' methods propose a generalized equaiton of the form:

$$K_2 = \frac{C(U^x)}{H^y} \qquad (6)$$

where:

U = mean stream velocity

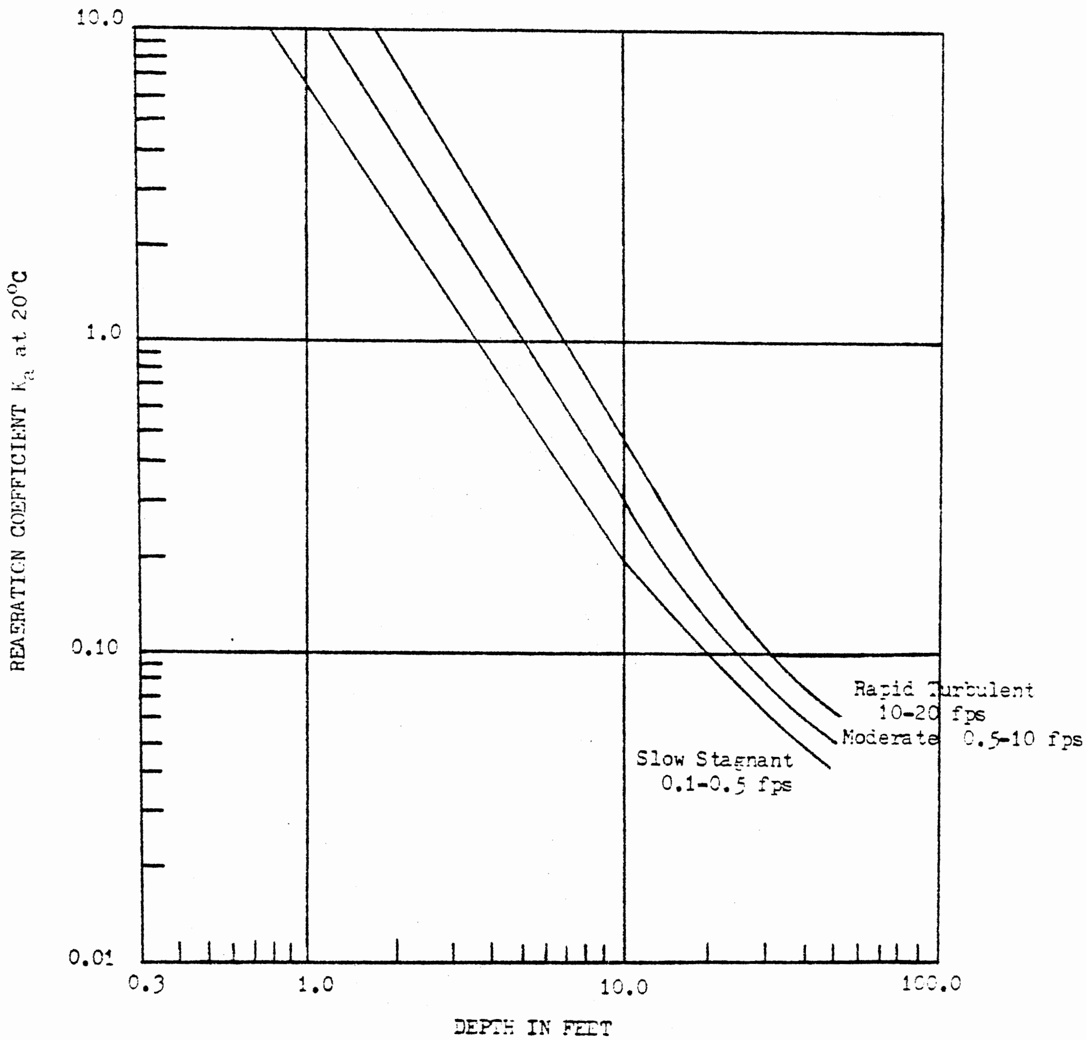
H = mean stream hydraulic depth

C, x and y = constants

The obvious differences between the various specific equations lie in the assigning of values to the constants. Excellent reviews of the most widely used empirical K_2 equations are presented by Covar (18) and Nemerow (6). Figure 4 shows a range of the reaeration coefficient (shown here as Ka) as a function of stream depth and average velocity. The K_2 of a stream can be approximated based on stream depth and velocity using the suggested method of Covar using Figure 5.

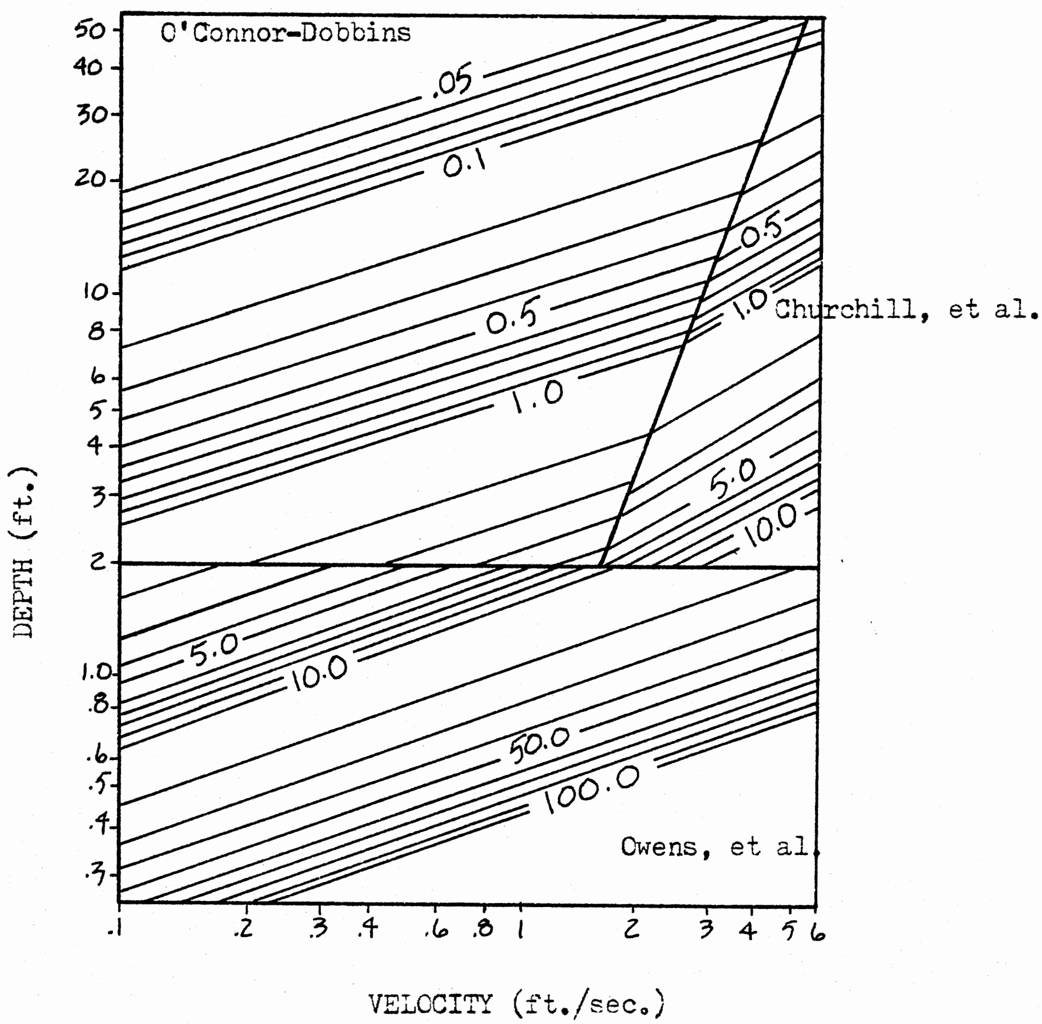
The reaeration phenomenon is directly proportional to temperature. The following temperature conversion formulated by Churchill (1962) was

Figure 4. Reaeration Coefficient (K_a) as a Function of Depth



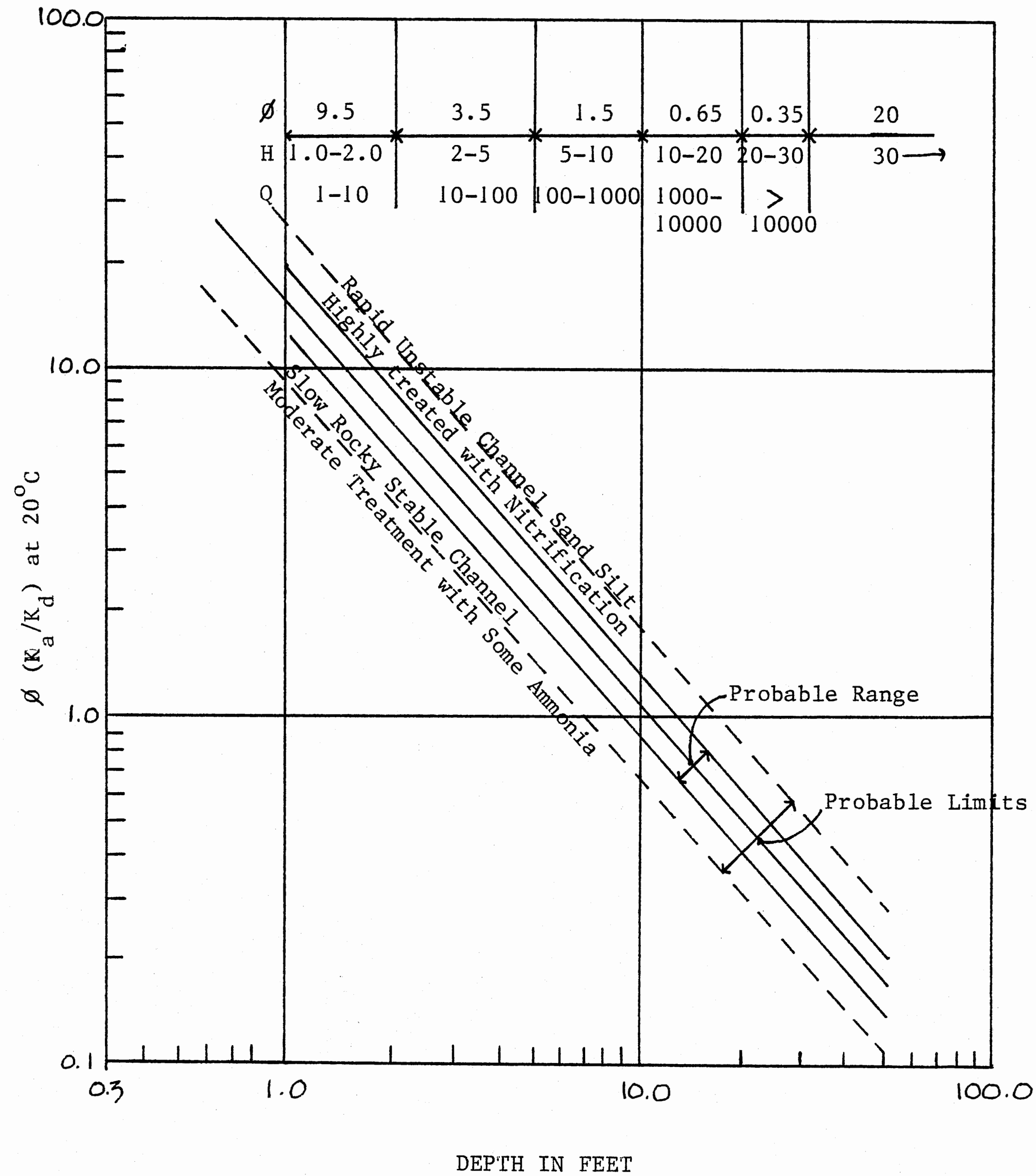
10.0
1.0
0.10
0.01
REAERATION COEFFICIENT K_a at 20°C
0.3
1.0
10.0
100.0
DEPTH IN FEET
Rapid Turbulent 10-20 fps
Moderate 0.5-10 fps
Slow Stagnant 0.1-0.5 fps

Figure 5. K_2 vs. Depth and Velocity Using the Suggested Method of Covar (1976)



k_2 vs. depth and velocity using the suggested method of Covar (1976).

Figure 6. Assimilation Ratio as a Function of Depth



$$\phi = \frac{K_a}{K_d}$$

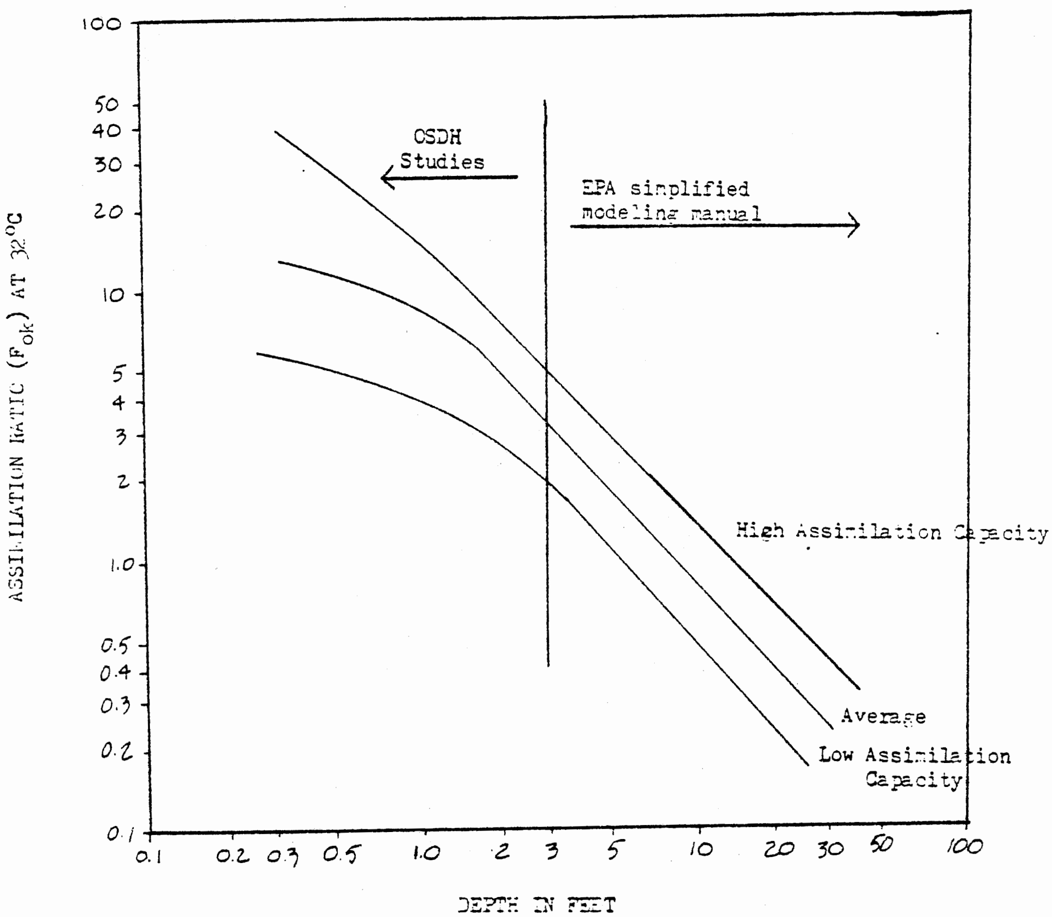
used:

$$K_2 \text{ at } T_1 = K_2 \text{ at } T_2(1.0238)^{T_1-T_2} \tag{7}$$

Stream Assimilative Ratio, F

Fair (20) originally defined the 'purification constant' or assimilative ratio as $F = K_2/K_1$. Correspondingly, all of the Streeter-Phelps equations can be expressed in terms of F. With reference to this basic concept, Fair published numerous F-values for specific types of receiving waters (e.g., slow streams, swift streams, lakes, etc.). The F-value is considered to be directly proportional to the water's assimilative capacity, the higher the F-value, the greater the stream's purification potential. Figure 6 (13) shows the relationship between the purification constant F (or $\emptyset$) and various stream characteristics while Figure 7 utilizes localized data from the Oklahoma State Department of Health to 'fine tune' the relationship between stream depth and assimilation. Table I is an expansion of this information shown in Figure 7 (21).

Figure 7. Assimilation Ratio (F_{ok}) as a Function of Depth



$$F_{ok} = \frac{K_a}{K_d}$$

Reproduced from "Modeling Analysis of Water Quality for the INCOG Planning Area," prepared by Hydroscience, March, 1978.

TABLE I

SUGGESTED ASSIMILATION RATIOS
FOR TULSA STREAMS

	FLOW RANGE (cfs)	DEPTH RANGE (feet)	ASSIMILATION RATIO, F_{ok} 32°C range	ASSIMILATION RATIO, F_{ok} 32°C average	F_{ok} 20°C average
Samll Tributary Streams	0.5-4	0.3-1	4-40	11.0	14.4
Intermediate Streams	4-10	1-2	3-18	6.1	8.0
Major Streams	10-100	2-5	1-8	2.7	3.5
Intermediate Rivers	100-1000	5-10	0.5-3	1.1	1.5
Major Rivers	1000-10000	10-20	0.2-1.4	0.5	0.65

[1]Reprinted from "Modeling Analysis of Water Quality for the INCOG Planning Area," prepared by Hydroscience, March 1978. These values are not applicable to the impounded sections of rivers.

Values of F_{ok} at other temperatures may be obtained from the following relationship:

$$\left(F_{ok}\right)_T = \left(F_{ok}\right)_{20^\circ C} \left(\frac{1.024}{1.047}\right)^{T-20} = F_{ok\ 20^\circ C}\ (0.978)^{T-20}$$

CHAPTER III

APPLICATION

With the growing evidence that pollution control is expensive, especially at levels of treatment greater than conventional secondary, and the necessity for water quality programs to compete for funds with other equally important programs, a careful balance between the determination of the consequences of proposed environmental control actions and the costs and benefits associated with such actions must be met. The effective attainment of desirable water uses predicates the use of a process which results in the equitable distribution of the permissible level of waste discharge to achieve a designated water use along with an evaluation of the costs, benefits and other implications of both the allowable discharge level and the appropriateness of the use and the water quality standards. This process is called the wasteload allocation (WLA) process.

The effective attainment of desirable water uses depends on a careful balance between the determination of the consequences of proposed environmental control actions and the costs and benefits associated with such actions. A rigorous and well-founded analysis of the estimation of the consequences of environmental controls is one of the central components of the process. Such an analysis, when coupled with an equally rigorous evaluation of the costs of control, the resulting benefits, and socio-political interactions permits the determination of the amount of wastes that may be discharged into a given body of water.

Principles of Wasteload Allocation

The wasteload allocation process is therefore the equitable distribution of the permissible level of waste discharge to achieve a designated water use with suitable recognition of the costs, benefits and socio-political implication of the allowable discharge level.

Note that the concept of a WLA is fundamentally not restricted to only a single class of water quality problems, such as dissolved oxygen. Rather, if the concept is viable, it should apply to the whole range of water quality situations including eutrophication, thermal pollution, and chemical discharges. This is not to say that for some situation, the WLA may not be zero, as for example, in the case of a highly toxic substance where the risk of a ecosystem of public health catastrope is substantial.

The entire concept of WLA is irrelevant if:

1. the cost of waste reduction or elimination is cheap in some sense or
2. regardless of cost or resulting water quality and water use, waste reduction to "high" levels or complete elimination is still worth doing on the general grounds of protecting the public health and/or the health of the aquatic ecosystem.

If either of these assumptions prevail, there is little to be gained from an analysis of water quality response and the determination of a WLA. However, there is growing evidence that neither of the assumptions are true in totality and the range of costs, effectiveness, and protection of water uses is wide indeed. With the growing evidence that pollution control is expensive, especially at higher levels of removal and the need for water quality programs to compete for funds with other equally important social programs, the desirability and necessity, of a sound WLA procedure becomes more apparent. A useful perspective on WLA can be obtained by a brief review of the use of such procedures in water quality management.

The concept of allocating a permissible discharge load to a given discharger has had a torturous history. From the earliest beginnings of water quality analyses, the basic concepts of WLA have been operative. The roots of sound water quality modeling as part of a determination of required degrees of water treatment are found in the work of the 1920's and 1930's in the Ohio River by Streeter and Phelps and Crohurst (22). A comprehensive report on the status of water pollution in the U.S. in 1939 is based extensively on the concepts of WLA, and it is assumed that optimum use would be made of the natural purification processes in streams, and that, because of differences in stream flow, in polluting substances and in water use, much waste would be discharged untreated or with only minor treatment. Complete treatment of all waste is not attainable. It would cost several times more than the programs outlined and would not be necessary even though possible. Moreover, the desirable standard of water quality may vary greatly from one part of a drainage area to another (23). (At this time, approximately 25% of the total urban population received primary treatment and about 25% received secondary treatment). This general view prevailed for more than a quarter century.

One school of thought that developed in the late 1960's, during the rapidly increasing public awareness of environmental problems, advocated a position that largely discarded the concept of WLA. The argument rested primarily on the grounds that the process of designating water uses, water quality standards, evalation and study of wasteload inputs and resulting water quality, and the establishment of allowable waste discharge levels from a cost-benefit viewpoint was too cumbersome, time consuming and inefficient. The alternate approach was to simply establish effluent requirements at various levels of technology on the presumption that if such levels were technologically available, their use should be mandated. If the application of such levels failed to achieve a water quality standard, a higher level would be required. Only to this degree was there a recognition of the relationship between waste discharge and resulting water quality. The emphasis in this "end of pipe" school is on establishing effluent levels that are then written into subsequent legislation and/or implementing regulations. The school was so successful that in the Federal Water Pollution Control Act Amendments of 1972, the first goal of the Act is that the discharge of pollutants into the navigable waters be eliminated by 1985 (24). Thus, the "end of pipe" school became known as the "zero discharge" advocates. Categories of treatment were defined.

1. Best practical treatment (BPT) for industries by 1977, i.e. available and practicable control treatment
2. Best available treatment (BAT) for industries by 1983, i.e. economically available treatment
3. Secondary treatment for all publicly owned treatment waters (POTW)

The resulting effect on national water quality planning was profound, but relatively short lived. The 1972 Act did continue to include language and approaches based on water quality standards. Many allocation studies were rapidly done under Section 303 of the Act which called for Basin Plan Studies. However, the latitude for evaluating alternative wasteload allocations was severely constrained in practice.

Over the next half dozen years, secondary treatment was promulgated as the minimum level for all POTWs on the assumption that the expected water quality responses were worth the expenditure. Further thrusts were made to press on to advanced waste treatment (AWT), often with only minimal justification of expected water quality benefits. Wasteload allocation approaches, however, will be continued to be used as part of the National Pollutant Discharge Elimination System (NPDES) of permits for all discharges. The allocation was used specifically for those instances where there was some doubt that the achievement of water quality standards could be achieved by secondary treatment alone.

The pendulum continued to swing between wasteload allocation and effluent requirements. Various reports were issued by the Government Accounting Office (GAO) chastising the EPA for failing to adequately assess water quality costs and benefits, but in turn GAO often failed itself to recognize fully the pressures under which EPA was placed to achieve virtually impossible goals under a clearly impossible time table. The National Commission on Water Quality (NCWQ) created as part of PL 92-500, evaluated the situation and reported that the 1977 and 1983 deadlines could not be met and reviewed in great detail, the results that might be expected from the implementation of the effluent criteria.

Reports and legislation have continued, including the Clean Water Act of 1977 and numerous implementing regulations. Water quality criteria for toxic substances has been prepared and, in 1979, EPA indicated that any requests for construction funds for AWT must be rigorously justified from a cost-benefit viewpoint. It is clear, that following this history, a much clearer recognition of the importance of a rational approach to water quality management has emerged. The close correlation to external political economic forces cannot, however, be ignored. Nevertheless, contemporary water quality management programs are now an integrated system of basic effluent requirements, supplemented by specific analyses of individual situations to arrive at a meaningful allowable discharge. Wasteload allocation, therefore, continues to be an important part of this overall process.

Wasteload Allocation Procedure

The principle steps in the WLA process are summarized as:

1. A designation of a desirable water use or uses, e.g. recreation, water supply, or agriculture
2. An evaluation or water quality criteria that will permit such uses
3. The synthesis of the desirable water use and water quality criteria to a water quality standard promulgated by local, State, Interstate, or Federal agency
4. An analysis of the cause-effect relationship between present and projected wasteload inputs and water quality response through use of
   a. available field data or data from related areas and
   b. a calibrated and verified mathematical model
   c. a simplified modeling analysis based on literature, other studies and engineering judgment

5. A sensitivity analysis and projection analysis of achieving the water quality standard through various levels of wasteload input
6. Determination of the "factor of safety" to be employed through, for example, a set aside of reserve wasteload capacity
7. For the residual load, an evaluation of the
   a. the individual costs to the discharges
   b. the regional cost to achieve the load and the concommitant benefits of the improved water quality standard
8. Given all of the above, a complete review of the feasibility of the designated water use and water quality standard
9. If both are satisfactory, a promulgation of the wasteload allocation permitted for each discharger.

Additional Considerations

As noted, an integral part of the WLA process is the analysis of cause-effect relationships via a mathematical model of waste input and resulting water quality response. Figure 8. shows the principal components of mathematical modeling framework and shows the need to carefully integrate general theory, field and laboratory data with the process of model calibration and verifications. The WLA rests heavily on the credibility and predictive capability of the mathematical modeling framework.

However, the adequacy of the modeling framework is only one of many issues that must be considered in a WLA process. Table II lists some of

Figure 8. Illustration of Allocation Procedure

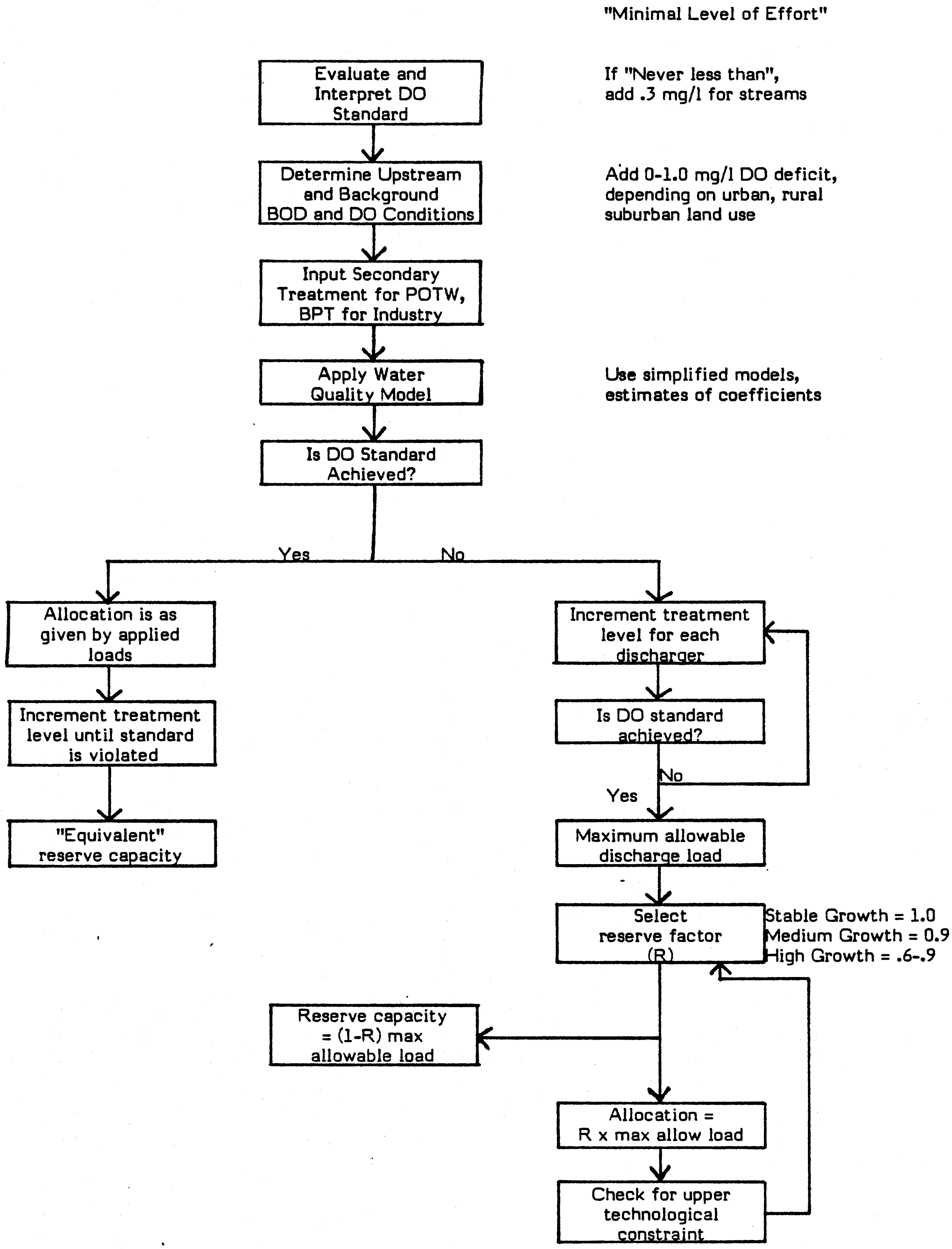

"Minimal Level of Effort"
Evaluate and Interpret DO Standard
If "Never less than", add .3 mg/l for streams
Determine Upstream and Background BOD and DO Conditions
Add 0-1.0 mg/l DO deficit, depending on urban, rural suburban land use
Input Secondary Treatment for POTW, BPT for Industry
Apply Water Quality Model
Use simplified models, estimates of coefficients
Is DO Standard Achieved?
Yes
No
Allocation is as given by applied loads
Increment treatment level until standard is violated
"Equivalent" reserve capacity
Increment treatment level for each discharger
Is DO standard achieved?
No
Yes
Maximum allowable discharge load
Select reserve factor (R)
Stable Growth = 1.0
Medium Growth = 0.9
High Growth = .6-.9
Reserve capacity = (1-R) max allowable load
Allocation = R x max allow load
Check for upper technological constraint

TABLE II

QUESTIONS RELATING TO THE WASTELOAD ALLOCATION PROCEDURE

1. How should the cost of waste removal and/or the benefits of water quality improvement be incorporated in the allocation procedure, if at all?

2. What are the dimensions of the notion of equity? What is a "fair" allocation procedure? What does it mean to treat all discharges equally?

3. What kind of water use and water quality problem contexts are amendable to allocation procedures today? Which kind of problems may "never" lend themselves to an allocation procedure?

4. Should other alternatives for improving water quality, e.g. flow augmentation, in-stream treatment, spatial distribution of effluents be included in the wasteload allocation process?

5. Is it possible to consider allocations of waste assimilation capacity on a time variable (e.g. seasonal) basis ? What might be some of the technical, economical and administrative implications of such a procedure?

6. Are the mathematical models available today sufficiently accurate to warrant their use in allocation computations?

7. What kind of sampling programs should be carried out by the regulatory agency to ensure that allowable allocations are being met, that water quality standards are achieved and that the designated water uses are obtained?

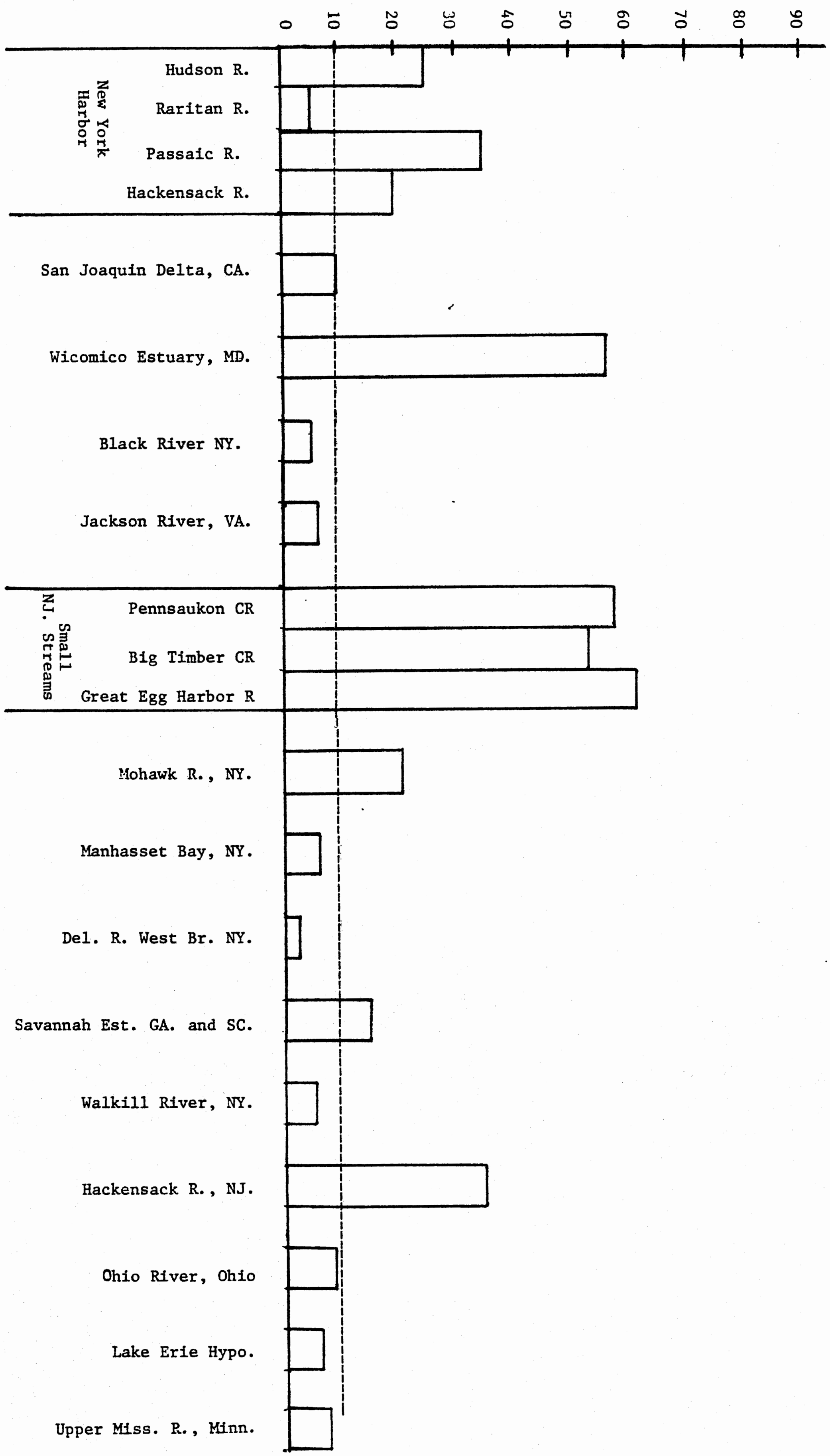
8. How does one incorporate new discharges or increases of existing discharges into the allocation procedure?

9. What kind of incentives can be built into the program to encourage dischargers to remove waste beyond their allocation? Are effluent charges feasible?

the questions associated with these issues. Not all issues arise in all cases and some relate specifically to cases where there are interactions between discharges. At the present time, there is no widely promulgated procedure that addresses each of the issues of Table II in an integrated readily usable analysis framework. For example, the issue of costs and benefits of waste reduction and water quality improvement is considered only in a qualitative manner. In spite of the fact that there are techniques available to analyze the cost trade-offs, and in spite of the fact that it is widely recognized that cost is an important factor, the rigorous inclusion of cost analyses in the WLA procats is not usually practical. The first five questions of Table II relate to this overall consideration of costs in the WLA.

The question of model accuracy is often a crucial question in situations where a given allocation is being negotiated or contested. Thomann (26) has discussed this question and compiled a distribution of relative errors between model calibration output and the observed data. (Relative error is the absolute value of the difference between observed and calculated value divided by the observed value). Figure 9. displays the median relative error in dissolved oxygen (DO) for water bodies of varying complexity. The models used generally represented state of the art DO models, applied by experienced practitioners using best judgement on loads, parameters and model structure. That is, the calibrations were conducted with defensible theoretical bases and not merely to go through the data points. The median relative error of 10% with maximum errors of 60% for small streams can be noted and is suggestive of present ability to reproduce the observed data with a credible model.

Figure 9. Relative Error in Water Quality Models


0
10
20
30
40
50
60
70
80
90
New York Harbor
Hudson R.
Raritan R.
Passaic R.
Hackensack R.
San Joaquin Delta, CA.
Wicomico Estuary, MD.
Black River NY.
Jackson River, VA.
Small NJ. Streams
Pennsaukon CR
Big Timber CR
Great Egg Harbor R
Mohawk R., NY.
Manhasset Bay, NY.
Del. R. West Br. NY.
Savannah Est. GA. and SC.
Walkill River, NY.
Hackensack R., NJ.
Ohio River, Ohio
Lake Erie Hypo.
Upper Miss. R., Minn.

In order to perform an actual WLA a variety of issues must be considered. A specific level of effort must be chosen for the analysis, a modeling framework must be determined, specific parameter values must be assigned and judgments made on backgound conditions, reserve capacity and model accuracy. As an illustration, Figure 8. indicates some of the considerations for the allocation of oxygen demanding substances (carbonaceous and nitrogeneous BOD) to meet a DO standard. The procedure does not address issues of costs/benefits or alternative control actions. The procedure also indicates a "minimal effort analysis", e.g. extensive collection of field data and model calibrations are not employed. Guidelines for a "minimal effort analysis" can be found in Hydroscience 1974 (27).

The determination of the DO standard as the first step includes an evaluation of the statistical requirements of the standard. Thus, if the standard indicates that the DO should "never be less than" 5 mg/l, then recognition should be given to random uncontrollable variations in DO. For streams and rivers, these fluctuations may be on the order of a standard deviation of 0.25 mg/l. Thus, if 0.5 mg/l is added to the standard, then the resulting level of 5.5 mg/l represents the target minimum level which if attained will meet the absolute minimum level of 5 mg/l with only a 2-1/2% chance of dropping below the standard. This does not imply that the short term fluctuations may or may not be damaging to the ecosystem. That determination is part of the interpretation of the standard. For other bodies of water, such as estuaries or harbors, other analyses may be necessary.

The selection of a background DO deficit is subject to a wide variation depending on the specifics of the area, such as urban, suburban or rural land use. The deficit may be determined from upstream BOD and DO conditions and calculated through the region of interest. This requires assignment of BOD deoxygenation coefficients. A minimal effort analysis would simply assign a constant DO deficit throughout the river reach of 0-1 mg/l depending on the problem conditions. This step is clearly a subject of potentially widely varying engineering judgment. It should be noted that the use of 1 mg/l DO deficit may result in a significantly higher degree of required treatment than if no background were assigned.

The inputs from each of the discharges are then estimated following general guidelines for expected effluent concentrations. The application of the water quality model may also vary widely depending on the level of effort; from simplified paper studies to full scale field and calibration studies. If the DO standard is achieved with presently mandated effluent levels, then the allocation is as given by those levels and an "equivalent" reserve capacity can be estimated. If the standard is not achieved by application of minimal levels of treatment, the procedure continues by incrementing treatment by discrete levels. The technological upper bound (e.g. BAT for industry) should be checked here. The maximum allowable discharge load is then the load needed to achieve the standard. However, this is not necessarily the load to be allocated. If a relatively rapid growth is forecasted for the area, then some fraction of the maximum allowable load should be placed in reserve as a type of "safety factor". Thus, if a reserve factor of 0.8 is chosen, then 20% of the allowable load is placed in reserve to be used for new discharges or, if necessary, to increase the allocation of existing discharges at a future date. The allocation is given by the reserve factor times the maximum allowable load. However, a final check should be made to insure that the required treatment level is technologically feasible. If an upper technological treatment bound has been exceeded, the reserve factor may have to be adjusted.

CHAPTER IV

CONCLUSION

Wasteload allocations or effluent limits are set to protect water quality standards. The water quality standards then become a critical factor in plant design and resultant costs based on effluents. For this reason, the standards should be evaluated whenever accelerated levels or treatments must be utilized to meet the standard.

The classical steps in developing water quality standards are:

1. A stream or stream reach is designated as having a desirable water use or uses, such as primary body contact recreation; fish and wildlife propagation; drinking water suppply.
2. In order to protect these uses, certain criteria must be determined necessary to be met. These criteria may be general or very specific and technically based.
3. The criteria, along with the beneficial use(s) are then adopted as a water quality standard (WQS) by local, state, and federal interests.

In Oklahoma, water quality standards are adopted by the Oklahoma Water Resources Board (OWRB) (28) and are subject to review at least every three (3) years. Although the review process has been undertaken as mandated, many aspects of our WQS have not changed. The main shortcoming in our WQS is the "wholesale" beneficial use designation. Virtually every stream in the State is assigned the same set of beneficial uses without regard to the applicability or to the desirability of the use. Each use is then, for the most part, given the same set of criteria as a basis for protection without considering background conditions, ambient water quality or attainability. The Oklahoma WQS make no provision for variances for whatever reason and do not address their economic ramifications.

The wasteload allocation (WLA) process is the equitable distribution of a permissible level of waste discharge to achieve a designated water use. In order to determine what is the maximum level of waste that can be discharged and still maintain the desired use, several interactions must be evaluated keeping the standards in mind. There should be:

a. An analysis of the cause-effect relationship between present and projected wasteload inputs and water quality response through the use of:
    1. available field data;
    2. a mathematical model(s); and
    3. a modeling analysis based on literature, other studies and engineering judgment.
b. A sensitivity analysis and projection analysis of achieving the WQS through the employment of various levels of wasteload input (different treatment levels).
c. An evaluation of the:
    1. costs associated with the required treatment level; and
    2. the benefits of the resultant water quality.
d. A complete review of the feasibility of the designated water use and WQS.

In Oklahoma, there is a 5.0 mg/l dissolved oxygen standard set for the protection of the fish and wildlife beneficial use. There is concern as to both the appropriateness of this use designation on many streams and on the attainability of the 5.0 mg/l DO where there are municipal sewage discharges, regardless of the level of treatment employed at the facility. There is also great concern regarding the cost of protecting this use. These basic questions now arise:

1. Do these concerns have any basis?
2. Can documentation be provided?
3. If so, what next?

To determine the applicability of the standard, an "instream" evaluation would be necessary. This would consist of a detailed sampling program, data analysis and water quality model application. The results of this modeling effort would then be used to indicate resultant levels of treatment, DO, and associated costs along with the potential effects on the designated use (in this case, fisheries). Several alternative scenarios could then be developed and presented for local input. A decision must then be made as to the appropriateness of the use by the effected citizenry and if applicable a recommendation made to the State that the use and/or standard be changed.

The main drawback to reevaluating the WQS is that no precedent has been set with State in modifying a use designation or standard. EPA policy has been that the only change allowed will be toward a more stringent standard and the State has paralleled this viewpoint. However, it is time for the recognition of the importance of a rational approach to water quality management and it has been assured that a concerted effort with adequate justification would be considered in the 1982 standards update by the State.

Until that time, municipal facilities should be instructed to develop alternative treatment processes where possible (land application, total retention, etc.) or to "phase" treatment to the secondary level while maintaining the capability of expansion to more stringent levels of treatment. At the same time, intensive stream analysis and modeling must be undertaken to document and technically justify all required levels of treatment that are greater then secondary. Resultant instream benefits at "marginal" levels of treatment could also be predicted by the modeling effort allowing a more complete look at what is to be realized by varying levels of treatment.

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VITA

Brent Eugene VanMeter

Candidate for the Degree of

Master of Science

Thesis: WATER QUALITY MODELING AND THE WASTELOAD ALLOCATION PROCESS

Major field: Bioenvironmental Engineering

Biographical:

Personnal Data: Born in Henryetta, Oklahoma, September 9, 1953, the son of Mr. and Mrs. Eugene A. VanMeter.

Education: Graduated from Henryetta High School, Henryetta, Oklahoma, in May, 1971; received Bachelor of Science degree in Biology from Oklahoma State University in May, 1975; completed requirements for Master of Science degree from Oklahoma State University in December, 1981.

Professional Experience: Environmental Engineer, Oklahoma State Department of Health 1977 - 1980; Director of Planning and Standards Division, Oklahoma State Department of Health, 1981.