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# WATER QUALITY ASSESSMENT MODEL-OXYGEN DYNAMICS MODEL FOR LOW-FLOW STREAMS

DET 1 (CEEDO), HQ ADTC
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TYNDALL AFB, 32403

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This investigation was undertaken to develop a low-flow stream waste assimilation model to determine the effect of point source discharges at selected Air Force installations. The impetus for this study was the lack of available low-flow stream models and the AF need to evaluate low-flow streams that traverse their installations.

From data acquired by this and previous investigations, an analysis of oxygen sinks and sources, unique in their level of significance for low-flow (cont'd)

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streams, was undertaken. Oxygen sources and sinks considered were carbonaceous BOD, nitrogenous BOD, stream reoxygenation, benthic oxygen demand, and photosynthesis/ respiration. In addition to these, the effects of toxic pollutants upon biological reaction rates and waste assimilative capacity were also explored. Techniques employed in this investigation may be considered as one approach to determine waste assimilation capacity and to simulate variations in the oxygen profile caused by point source wastewater discharges into a low flow stream. The validity and limitations of the overall approach, the ability to extrapolate oxygen profiles, and the ability to transpose conditions to facilitate prediction of future conditions is, at this time, not fully substantiated; however, positive indications have been obtained in model verification that lend credence to the procedures adopted. Future studies should be directed toward validation and improvement of the proposed model and sensitivity analysis to define confidence intervals.

# PREFACE

This report documents work performed during the period 1 July 1975 through 1 March 1977 by the Water and Solid Resources Division of the Environics Directorate. The in-house job order number was 19005W27 at the Air Force Civil Engineering Center, Air Force Systems Command, Tyndall AFB FL 32403. Captain Stephen P. Shelton, Water and Solid Resources Division, was the principal investigator for the Center.

This report has been reviewed by the Information Office (OI) and is releasable to the National Technical Information Service (NTIS). At NTIS it will be available to the general public, including foreign nations.

This technical report has been reviewed and is approved for publication.

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# SECTION I

# INTRODUCTION

## BACKGROUND

Current trends indicate continuing interest in improving water quality modeling techniques available for use in environmental impact and in planning for wastewater treatment facility needs. Historical modeling activities have largely been directed toward water bodies having significant flows with water quality changes predominantly related to only a few factors. As Air Force (AF) facilities are expanded along small streams, the water quality models necessary for planning mission impacts upon both point and area source discharges must include a wider range of environmental factors.

This technical report presents the logic and methodology used in facilitating design of an oxygen profile model for AF use. The model developed herein evaluates only two pollutants theoretically; however, other pollutants are considered empirically.

## OBJECTIVES

More than 50 percent of the AF installations in the United States are adjoined or traversed by small streams. Because the AF is required by regulation (AFR 19-2) and public law to assess the environmental impact of mission, facility, and operational modifications, it is apparent that some methodology to assess water related environmental impacts is required. The objective of this technical report is to describe a basic first order theoretical dissolved oxygen model that considers Biochemical Oxygen Demand (BOD), Nitrogenous Oxygen Demand (NOD), and instream reoxygenation as a function of time.

# SECTION II

# MODEL DEVELOPMENT

The initial element in developing the oxygen profile model involved use of a standard first-order oxygen sag model to describe carbonaceous oxygen demand. This model was re-derived, using the Laplace transforms, to consider a second first-order reaction for nitrogenous oxygen demand. It is designed for stepwise application to accommodate stream changes from free-flow to pools and to allow variation of the reoxygenation coefficient as a function of stream turbulence. Semi-empirical relationships were developed to correct this basic model for benthic demand, area source demand, photosynthesis, and respiration. Methods for applying these corrections are described in subsequent sections of this report. In addition, since industrial discharges are often a problem on AF installations, an empirical correction technique is developed to explore the significance of toxic discharges on the biological reaction rates and thereby evaluate their impact.

The subsequent sections of this technical report detail the elements involved in this approach and describe the verification of the model. Based upon substantive verification of the model, future projections are described for alternative mission, facility, and operational modifications and their associated pollution abatement strategies.

## FIRST ORDER OXYGEN SAG MODEL

The general form of the first-order oxygen sag model classically used to describe the simultaneous interaction of carbonaceous oxygen demand and surface reoxygenation is (Reference 1):

$$D_t = \frac{K_1 L_a}{K_2 - K_1} \left[\exp(-K_1 t) - \exp(-K_2 t)\right] + D_a \exp(-K_2 t) \qquad (1)$$

for $K_1 \neq K_2$ where

$D_t$ = oxygen deficit in mg/l at any time, t in days

$K_1$ = deoxygenation coefficient for carbonaceous BOD in days$^{-1}$

$K_2$ = reoxygenation coefficient in the stream in days$^{-1}$

$L_a$ = ultimate carbonaceous BOD in mg/l at time zero

$D_a$ = oxygen deficit in mg/l at time zero

## REOXYGENATION COEFFICIENT

Because of the physical system being modeled, a low-flow stream, generally involves a series of freeflow reaches interspersed with more slowly flowing pools, it was necessary to employ a relationship to evaluate $K_2$ as a function of the variable turbulence in the system. Models for $K_2$ developed by O'Connor and Dobbins (Reference 2), Dobbins (Reference 3), Krenkel and Orlob (Reference 4), Churchill (References 5 and 6), and Thackston and Krenkel (Reference 7) were considered for this purpose. The Thackston and Krenkel model was selected for use in this investigation since the range of conditions employed for its verification encompassed the conditions existing in the low-flow streams of concern to the AF. The general form of this model is:

$$K_2 = 24.87 \left[ 1 + \left( \frac{\bar{u}}{(gh)^{\frac{1}{2}}} \right)^{\frac{1}{2}} \right] \left[ \frac{S_e g}{h} \right]^{\frac{1}{2}} \qquad (2)$$

where

$K_2$ = reoxygenation coefficient (base e) for the streams in days$^{-1}$

$\bar{u}$ = average stream longitudinal velocity in feet per second (ft/sec)

$g$ = 32.2 ft/sec$^2$

$S_e$ = slope of the energy grade line in ft/ft

The stream hydraulics for a constant cross-section and slope can be described by Mannings' equation:

$$Q = \bar{u}A = \frac{1.49}{n} R^{2/3} S^{1/2} \tag{3}$$

where

Q = stream flow in cubic feet per second (cfs)

A = cross-sectional area of the stream in $ft^2$

R = hydraulic radius for the stream in ft

n = Mannings' roughness coefficient

For a given river reach, with semi-rectangular cross-section and constant width, R can be expressed for the reach:

$$R = \frac{A}{P} = \frac{hW}{W+2h} \tag{4}$$

where

P = the wetted perimeter in ft

h = mean river depth in ft

W = mean river width in ft

If h is much smaller than W, the 2h term in the expression becomes insignificant with respect to W, and R becomes essentially equal to h. With these approximations, the Manning equation may be reexpressed

$$Q = \bar{u}A = \bar{u}hW = \frac{1.49}{n} h^{5/3} W S^{1/2} \tag{5}$$

or solving for average depth and velocity

$$h = \left[0.667 \frac{QnS^{1/2}}{W}\right]^{3/5} \tag{6}$$

and

$$\bar{u} = \frac{Q}{hW} \quad (7)$$

Therefore, the velocity and depth can both be expressed as a function of flow for a given stream with constant width, slope, and roughness coefficient.

In reaches of a low-flow stream considered suitable for hydraulic modeling by Manning's equation, equation 2, can be simplified by substituting equations 6 and 7 for h and $\bar{u}$ variables in equation 2. If this is done, and the actual values of n, w, and S are included, $K_2$ for a specific stream can be expressed as a function of flow:

$$K_2 = C_1Q^{-0.3} + C_2Q^{-0.25} \quad (8)$$

When: $K_2$ = Reoxygenation coefficient in $days^{-1}$

Q = Stream flow in cfs

$C_1$ and $C_2$ = Constants dependent upon n, w, and s

For river reaches that contain pools, $K_2$ is computed directly from equation 2. Values used for depth, slope of the energy grade line, and mean velocity must be determined from field data.

## CARBONACEOUS OXYGEN DEMAND

While background stream organic generation and area source runoff sometimes contribute significantly to the organic load in a low-flow stream, this load can normally be expressed as a percentage of the stream's waste assimilation allocation. A model to evaluate the area source pollution components, compatible with this model algorithm, is currently being developed for AF applications. Generally, however, the point source discharges in a stream have the greatest pollution potential to determine the overall carbonaceous waste assimilation potential for the stream. Description of carbonaceous BOD is accomplished by use of field data to determine the range of values for $L_a$, $D_a$, and $K_1$ to be used in equation 2. When a model is validated, use of these field data to evaluate model sensitivity is a necessity to determine confidence intervals.

## NITROGENOUS OXYGEN DEMAND

Ammonia nitrogen can produce an additional oxygen demand upon a stream. The oxidation of ammonia to nitrite to nitrate is a twofold sequential biological process: (1) *Nitrosomonas* bacteria oxidizes ammonia to nitrite, then (2) *Nitrobacter* bacteria oxidizes nitrite to nitrate. The rate of these reactions is proportional to the concentrations of ammonia and bacteria present (*Nitrosomonas* and *Nitrobacter*) and the quality of the stream environment for biological activities (References 8, 9, 10, 11, and 12). The oxidation of ammonia to nitrite, however, is the rate-controlling reaction, since the nitrite to nitrate reaction proceeds at a much higher rate. Since both reactions are aerobic processes, oxygen is consumed in the reactions; approximately 4.47 mg/l of oxygen per mg/l of ammonia as N oxidized to nitrate as N (References 8, 9, 10, and 13). All subsequent references for concentrations of ammonia, nitrite, and nitrate are expressed in mg/l as nitrogen (N).

Most low-flow streams, particularly those that traverse populated or agricultural areas, leave a background ammonia concentration sufficient to maintain a seed culture of *Nitrosomonas* and *Nitrobacter*. Agricultural practices have a great potential for contributing ammonia to surface waters through seepage and runoff and/or accidental spills of chemical fertilizers and through livestock feeding and cleaning practices. Moreover, if the watershed of concern is a naturally productive area, it should be expected that nutrient levels in the runoff waters will be significant.

From the foregoing discussion, it is apparent that the normal low-flow stream will require evaluation of the nitrogenous oxygen demand contribution to its oxygen dynamics. Since most domestic wastewater treatment plants and many industrial wastewater treatment plants discharge large quantities of ammonia, stream data must be collected both above and below any wastewater treatment facility. When the stream data are evaluated, oxidation rates can be determined. Four generalizations can normally be made concerning the impact of the wastewater discharge upon the stream (assuming it contains significant levels of ammonia and BOD): (1) Ammonia oxidation to nitrate is occurring at some rate in all reaches of the stream; (2) the area source component is a significant contributer of ammonia to the system; (3) the wastewater treatment facility significantly increases the stream's background concentration of ammonia; and (4) the apparent rate of ammonia oxidation, below the wastewater treatment plant, is lower than the background rate.

These statements can be expressed quantitatively and the rate of ammonia oxidation can be calculated. Stratton (References 10 and 11) proposed a general equation to describe the rate of oxygen utilization in the process of ammonia oxidation to nitrate as follows:

$$\frac{dO}{dt} = 3.43 \left(\frac{KMC}{K_s+C}\right)_{NH_3} + 1.14 \left(\frac{KMC}{K_s+C}\right)_{NO_2} \qquad (9)$$

Where $\frac{dO}{dt}$ = rate of oxygen utilization in mg/l·day

K = Substrate utilization rate constant for $NH_3$ or $NO_2$ in days$^{-1}$

M = Concentration of viable nitrifying organisms (for *Nitrosomonas* and *Nitrobacter*)

C = Substrate concentration for $NH_3$ or $NO_2$ in mg/l

$K_s$ = "Michaelis" half-velocity constant mg/l

If the overall reaction rate is assumed to be controlled by the ammonia to nitrite reaction (References 10 and 11), equation 9 can be approximated:

$$\frac{dO}{dt} = 4.47 \left(\frac{KMC}{K_s + C}\right)_{NH_3} \qquad (10)$$

This reaction assumes that nitrite is oxidized immediately to nitrate. If this reaction is written in terms of the rate of substrate utilization, the first order equation can be written:

$$\frac{-dNH_3}{dt} = \frac{dNO_3}{dt} = K_3 C_{NH_3} \qquad (11)$$

Where: $\frac{dNH_3}{dt}$ = Rate of ammonia utilization

$\frac{dNO_3}{dt}$ = Rate of nitrate production

$K_3$ = Effective rate constant for the oxidation reaction

Note that $K_3$ is the K in Stratton's equation (equation 9) for the rate of conversion from ammonia to nitrite.

If equation 11 is integrated between time t = 0 and some time, t, it can be reexpressed:

$$\frac{(NH_3)_t}{(NH_3)_i} = \exp(-K_3 t) \qquad (12)$$

Where: $NH_3$ = ammonia concentration in mg/l (as N)

Subscripts t and i = Concentrations at time t and time 0, respectively.

Values for $K_3$ can be calculated from these equations using field data from above and below the wastewater point source discharge of concern. These constants, along with other upstream data including stream flow, velocity, reach length, temperature, and flow times are then used to evaluate the background oxygen depletion attributable to ammonia oxidation. Because the background oxygen depletion is usually small by comparison to the point source ammonia depletion component, the value can normally be used for initializing the model; however, if the area source was large enough to contribute substantially to the upstream background oxygen balance, the model could be segmented to evaluate the upstream situation.

If the stream data indicate a large drop in $K_3$ below the wastewater treatment plant ($>10^{-1}$) the discharge is probably contributing a toxic component or enzyme inhibiter to the stream. This would be almost assured if there is a significant industrial component in the wastewater. If this occurs, sufficient field data should be acquired to (1) determine the toxic component and evaluate methods to eliminate its presence, and (2) determine the recovery rate for $K_3$ in the stream in sufficient detail that an empirical equation can be written to describe it.

If, in the first order reaction equation for ammonia oxidation (equation 11) the ammonia concentration, $NH_3$ as N is reexpressed as the ultimate nitrogenous oxygen demand ($NOD_u = 4.47 \times NH_3$ as N), the expression can be added to the differential equation form of the first order oxygen sag model, equation 1. When this is done, a modified form of the first order oxygen sag model that considers ammonia oxidation, results:

$$\frac{dD}{dt} = K_1 L_a - K_3 N_a + K_2 D_a \qquad (13)$$

Where: $\frac{dD}{dt}$ = Rate of oxygen deficit change as a function of time in mg/l·day

$K_1$ = Rate of carbonaceous BOD oxidation in $\text{days}^{-1}$

$K_2$ = Rate of stream reoxygenation in $\text{days}^{-1}$

$K_3$ = Rate of nitrogenous NOD oxidation in $\text{days}^{-1}$

$L_a$ = Ultimate carbonaceous BOD in mg/l

$D_a$ = Background oxygen deficit for the stream in mg/l

$N_a$ = Ultimate nitrogenous NOD in mg/l

This equation assumes that there are no intermediate ammonia nitrogen sources such as system inflows or nitrogen fixation by algae or organisms in the system. This assumption can be eliminated subsequently, however, by finite difference analysis.

If the Laplace transforms of equations 1, 11 and 13 are solved simultaneously, and boundary conditions are defined:

L(o) = initial ultimate carbonaceous BOD in mg/l = $L_a$

N(o) = initial ultimate nitrogenous NOD in mg/l = $N_a$

D(o) = oxygen deficit in the stream at time t = 0, in mg/l = $D_a$, then the integrated form of the general equation may be written:

$$D_t = \frac{K_1 L_a}{K_2 - K_1} \left[\exp(-K_1 t) - \exp(-K_2 t)\right]$$

$$+ \frac{K_3 N_a}{K_2 - K_3} \left[\exp(-K_3 t) - \exp(-K_2 t)\right]$$

$$+ D_a \exp(-K_2 t) \qquad (14)$$

$$K_1 \neq K_2 \neq K_3$$

Where: $D_t$ = Oxygen deficit in the stream at any time due to the combined effects of BOD, NOD, and stream reoxygenation.

Equation 14 was developed to make the first estimate of oxygen deficit in a stream. This first estimate is then corrected to consider benthic demand, photosynthesis, and respiration.

## TOXIC INHIBITION OF CARBONACEOUS DECOMPOSITION

Sublethal concentrations of toxic compounds, periodically found in point source discharges on AF installations, may have an impact upon the oxygen dynamics of a stream. Because waste assimilation is primarily a biological process, sublethal toxic effects or inhibition of enzymatic reactions of the bacteria will modify the waste assimilative capacity. This modification may produce an apparent increase in the waste assimilative capacity of the stream with respect to its oxygen dynamics. This apparent increase is caused by the reduction in the carbonaceous and nitrogenous reaction rates in the sublethal system. If an AF treatment facility is found to have a sublethal toxic component in its discharge, this component must be considered in the evaluation of the environmental baseline for the system.

The general first order expression used to approximate carbonaceous BOD is:

$$L_t = L_a \exp(-K_1 t) \qquad (15)$$

Where: $L_t$ = BOD used at time t (in days), in mg/l

$L_a$ = Ultimate BOD of the sample in mg/l

$K_1$ = Carbonaceous BOD deoxygenation rate constant days$^{-1}$

The impact of toxic materials on a stream must be evaluated if they are contained in the wastewater discharge. This is true both for the projection of future impact or establishment of the current environmental baseline. It has been previously shown (References 14 and 15) that the oxygen profile impact of sublethal toxic components can be evaluated using wastewater treatment plant records. This is accomplished by evaluating the effect of the toxic substance upon the biological reaction rate constant, $K_1$. This is accomplished by comparing the apparent reaction rate constants across the wastewater treatment facility (secondary portion) when the defined toxic component is either present or not present. Ideally, the sublethal effects can be treated as a step function (on/off); however, if the toxicity decreases activity in a continuous function, this too can be evaluated using a curve fitting technique. The suggested approach for application of this technique is:

1. Determine, from wastewater treatment plant records, the value of $L_a$ (ultimate carbonaceous BOD) for the effluent from the primary clarifier when toxic components are absent.

2. Determine, from wastewater treatment plant records, the apparent value of $L_a'$, ($L_a' \equiv$ the apparent ultimate carbonaceous BOD with toxic influences) for the primary effluent when toxic components are present. Note, if the toxic effect is not a step function, this would require a number of evaluations to determine the equation of the function. Furthermore, the subsequent description would require consideration of $L_a'$ throughout as a function.

3. Express the effect of the toxic compound on the BOD test as a ratio:

$$\frac{L_a'}{L_a} = \eta \qquad (16)$$

Where: $\eta$ = the test efficiency when the toxic component is present

4. If it is assumed that the efficiency of the BOD test is the same for the effluent as for the primary effluent (assume that the toxic compound is conservative across the secondary treatment system), then the uninhibited ultimate

carbonaceous BOD of the wastewater treatment plant effluent may be expressed:

$$L_e = \frac{L'_e}{n} \tag{17}$$

Where: $L_e$ = the ultimate carbonaceous BOD of the effluent without toxic interferences

$L'_e$ = the apparent ultimate carbonaceous BOD of the effluent when the toxic material is present

5. If wastewater treatment plant records are used to establish the values of $L_a$, $L'_a$, $L_e$, and $L'_e$, the true values of $L_a$ and $L_e$ can be determined for times when the toxic component was present.

6. Since organic reduction in secondary wastewater treatment is a combination of sorption and oxidation, the apparent rate constant for the BOD is greater than an oxidation only process. However, an equation analogous to equation 15 can be adopted to describe this process:

$$R = \frac{\ln(L_a/L_e)}{DT} \tag{18}$$

Where: R = overall rate of reaction across the secondary treatment unit in days$^{-1}$

DT = average detention time across the secondary units in days

If the values obtained in step 5 are used in equation 18, two reaction rate constants result; R, the reaction rate constant when toxic material is absent and R′, the reaction rate when toxic material is present.

7. The reaction rate constant, $K_1$, for the wastewater treatment plant effluent with the calculation of R and R′, can be approximated. This rate constant will be used in the oxygen sag model to simulate the oxygen dynamics of the stream. The expression for this approximation is:

$$K_1' = \left(\frac{R'}{R}\right) K_1 \tag{19}$$

Where: $K_1'$ = BOD rate constant for the stream when toxic components are present in days$^{-1}$

$K_1$ = BOD rate constant when toxic components are not present in days$^{-1}$; the value of $K_1$ is determined from ultimate BOD tests of the plant effluent when toxic compounds are not present

$R'$ = the apparent rate constant across the secondary treatment system when toxic material is present in days$^{-1}$

$R$ = the rate constant across the secondary treatment units when toxic materials are absent in days$^{-1}$

It is believed that this procedure provides a good approximation of the reaction rate existing in the stream when toxic conditions occur. This is supported by the rationale that, with low flow systems, effluent dilution is small and toxic materials in sufficient quantities to cause biological retardation in the wastewater treatment plant are subsequently deposited into the stream. Therefore, in some instances, a toxic material may reduce the rate of oxygen use by organic decomposition and cause an apparent increase in the waste assimilative capacity relative to organic loadings.

## BENTHIC DEMAND

Part of the naturally occurring or man-made organic material in a stream exists in the form of suspended solids material. Under high flow conditions when this material remains in suspension, it is decomposed aerobically in a manner similar to the soluble BOD which is the major fraction of the $L_a$ term discussed previously. When the stream velocity is reduced, suspended materials will settle out of the flowing water and form deposits on the stream bottom. The depth, location, and behavior of these deposits may vary with the type and amount of organic materials in the stream; the stream hydrographic conditions; and the rate and characteristics of the decomposition processes occurring in the deposit during various times of the year.

The most accurate approach to the evaluation of benthic demand is an extensive field sampling program coupled with a laboratory testing procedure. The laboratory procedure is designed to estimate the oxygen uptake rate for the sludge blanket under a number of ambient dissolved oxygen conditions. The field sampling estimates benthic deposit volume and surface area. This procedure is very expensive and extremely time consuming. A more realistic and less expensive method relies on probabilistic techniques, good engineering judgment, and relatively little field data. For most low-flow streams, a qualitative evaluation will yield the conclusion that the bulk of deposited material is located in pooled reaches. Furthermore, unless the river is biologically dead, the benthic blanket decays at a slow rate without significant gasification to stir the deposits. Using this rationale, the benthic associated oxygen demand can be attributed to the pooled reaches of the stream. Furthermore, an averaging technique can be adapted for estimating the average influence of the deposits. Since oxygen transfer into this type of sludge deposit is a function of the oxygen residual above the blanket (References 15, 16, and 17), estimation of benthic influence can be related to oxygen residual in the stream.

The data used to evaluate this process should include field data for interim measurements of dissolved oxygen, temperature, and $BOD_{5,20}$ (subscripts 5 and 20 refer to the standard 5-day BOD test at 20°C) for two stations, one upstream from the pooled reach and one downstream from the pooled reach. For many low-flow streams sufficient data for this evaluation can be obtained from the United States Geological Survey (USGS) streamflow and water quality records.

Most low-flow streams are subject to extreme flow variations during the year. Generally, streamflow will be a minimum during the dry season and when the wet season arrives, an increase in flow of several hundred to several thousand percent is not uncommon. Usually with the first large storm of the wet season a flushing action occurs and benthic sediments are resuspended. This occurrence reduces the benthic oxygen requirement to that of the bottom muds. If the rate of oxygen depletion across the stagnant reaches of the stream, below point source discharges, are evaluated and rates are corrected for flow, temperature and waste load fluctuations, the rate difference between pre- and post-washout can be considered to be caused by benthic demand.

There are two basic stages involved in determination of benthic oxygen demand from the raw field data; these are (1) pre-processing the data to ensure comparable conditions of flow, temperature, and waste load, and (2) determination of benthic oxygen demand from differential oxygen concentrations. Point source discharge data, from treatment plant records is required. These data should be chosen to reflect similar point source discharge conditions. In many systems, midweek data (Tuesday through Thursday) is the most recursive with regard to flow and $BOD_{5,20}$ from week to week. Using the most recursive weekly data; preprocessing using stocastic techniques (References 18 and 19) of these data is undertaken to select the appropriate combinations of system parameters for comparison. Normally, the flow and $BOD_{5,20}$ data will fall into a skewed normal distribution. If data beyond one standard deviation about the mean is disregarded, reasonable projections can be made from the remainder of the data. Streamflow decision factors should be evaluated using the mean dry season flow rate. The range for stream flow, from the mean, should be chosen so that stagnation in dead zones (under very low flows), or scour (during higher flows), is not a significant consideration. A reasonable value for this range is ± 10 percent of the mean dry season stream flow; this choice also permits prediction of flow times without the influence of surges from increased flow.

Point source discharge data and streamflow data that do not meet the selection criteria are eliminated. The remaining data are tabulated along with corresponding daily mean BOD, dissolved oxygen, and temperature data for stations above and below the pooled reach of concern. The data for the stream stations must reflect the flow time from the point source discharge such that water quality parameters considered at all locations are for the same slug of water as it flows downstream. Therefore, data selected from upstream and downstream sampling stations correspond to successively later times than the point source discharge data. If the temperature between stations varies by more than 2°C or, if the BOD projected by the rate equation varies by more than 50 percent, the data for that incident should be rejected.

At this point the remaining data is retabulated and classified according to pre- or post-washout conditions by date. The mean daily dissolved oxygen level (to eliminate diurnal fluctuations caused by photosynthesis and respiration) and mean temperature are used for both stations. When this is accomplished, oxygen deficit is calculated, in percent of saturation concentration, for each piece of data at its respective temperature. This calculation minimizes temperature effects between data and thus improves their comparability. The percent of change across the system for each data set is then calculated, the mean of these pre-and post-washout data is determined and the mean change in percent oxygen deficit caused by benthic demand is calculated. This value is then expressed in terms of the oxygen deficit caused by benthic demand for pre-washout conditions of temperature and flow.

Once the above sequence is accomplished, for low-flow pre-washout conditions, the benthic oxygen demand between the two stations can be expressed in terms of grams of oxygen used per square meter per day ($gm/m^2 \cdot day$). This value is normally decreased by 1.00 $gm/m^2 day$ to account for background; this value is typical of those reported by Fair, et al (Reference 20) for a relatively inert filler material. Thus, the net oxygen damand caused by benthic deposits becomes the difference between the measured value and the background value. Using known flow, volume, area, and depth relationships for the given subreach, between the two field sampling stations, the net benthic oxygen demand may be reexpressed in terms of milligrams per liter (of stream flow) per day.

As previously discussed, benthic oxygen demand may be a function of oxygen concentration above the sludge blanket. It was found by Burdick and Parker (Reference 16), Burdick (Reference 17), and Shelton, et al (References 14 and 15), that, for compacted deposits this influence was significant, when dissolved oxygen levels dropped below 4.0 mg/l, since oxygen diffusion into the deposit is reduced substantially. Unless the low-flow stream is grossly polluted with suspended organics, benthic deposits will not be subject to intense decomposition which releases gases and stirs the deposit. With this condition, the above-noted description of the diffusion effect should be applicable.

The net benthic demand, developed previously, is evaluated using data when the average stream oxygen concentration is greater than 4.0 mg/l. In order to determine the effect of reduced dissolved oxygen levels upon benthic demand, selected data from the previous analysis are used. These data are selected so that the mean dissolved oxygen level between the two stream sampling stations is approximately 2 mg/l. Using the previously described techniques, the benthic oxygen demand under these conditions is determined. If it is assumed that the benthic oxygen demand approaches zero as the dissolved oxygen concentration in the stream approaches zero, these points may be used to estimate the benthic oxygen demand for dissolved oxygen levels between zero and 4.0 mg/l. This relationship can be approximated by curve fitting an exponential function:

$$B' = B \exp(a\, C_t) + d \qquad (20)$$

Where: $B'$ = Benthic oxygen demand in mg/l day

$B$ = Average benthic oxygen demand for dissolved oxygen concentrations greater than 4.0 mg/l

$C_t$ = Dissolved oxygen concentration ($<$ 4.0 mg/l) in the stream

a,d = Empirical constants

The data points can be evaluated using a pattern recognition curve fitting technique (References 19 and 21) to determine the values of a and d. The resulting expression will yield an approximation of the benthic oxygen demand for stream dissolved oxygen concentrations between zero and 4 mg/l.

## PHOTOSYNTHESIS AND RESPIRATION

In systems which have only relatively small amounts of green plant materials, the diurnal sequence of oxygen production and demand may be safely omitted from consideration in an oxygen profile simulation model; however, in other systems, particularly small fertile streams, the photosynthesis/respiration influence may be a significant factor.

In some systems, the amount of algae present is sufficient to raise the dissolved oxygen concentration above the saturation concentration during daylight hours, and oxygen is lost to the atmosphere. This situation is most common in lakes and large streams and is rare in small low flow streams; however, the other extreme, development of critical oxygen deficits due to nocturnal respiration, occurs frequently in pooled reaches of low flow streams.

Evaluation of stream data should indicate the presence or absence of diurnal oxygen fluctuations caused by algae. If active, fluctuations in the oxygen concentration due to photosynthesis/respiration should be accentuated below the point source wastewater discharge of concern. This increase in diurnal activity is a manifestation of the organic material released at the wastewater treatment facility. The organics decompose in the stream and release carbon dioxide and other nutrients which stimulate growth in the algae population. Stagnant areas or pools in the stream also facilitate increased algae activity by allowing accumulation and thus greater growth times through storage effects.

A technique used to project oxygen fluctuations, caused by algae in low flow systems, has been suggested by Shelton, et al (References 14 and 15) as modified from O'Connor (References 22 and 23). The procedure is empirical since it uses oxygen residual data taken at all times of the day to simulate a diurnal pattern about the mean value from the data. The two components that cause the diurnal cycle are net daily photosynthesis and respiration. Net daily photosynthesis is an oxygen source. The amount of oxygen acquired by the system from this source is dependent upon the amount of algae present, the amount of sunlight received by the algae mass, and influences of the oxygen deficit within the system. If, for a given short season of several weeks, the sunlight is considered to be constant and the algae mass is considered, for this period, to be in equilibrium at any given stream location, then the amount of photosynthetic oxygen produced can be expressed as a function of the daily oxygen deficit variation. Respiration is an oxygen sink caused by algae and is constant for a given algae mass (References 22 and 23). The total effect of Net Daylight Photosynthesis minus respiration (P-R) can, in most low-flow systems, be expressed as a function of the daily oxygen deficit variation. Daily cycles of net P-R effects can be expressed as a sinusoidal function when the fluctuation does not exceed saturation or zero dissolved oxygen. This curve is symmetric about the mean oxygen level for the day.

For application of this technique, stream sampling stations are required wherever significant stream bed or waste loading characteristics occur. A minimum of one background and two downstream stations are required. Oxygen data for the stream at the stations is required for representative times (a minimum of every 6 hours) between 0000 and 2400 hours. These data should be obtained over at least one low flow season. Once acquired, the data should be expressed in terms of percent of saturation concentration for the stream temperature. Each station should be fitted to the general P-R equation using a pattern recognition curve fitting technique (References 19 and 21):

$$D' = a \text{ sine} \left[\left(\frac{t' + b}{24}\right) 2\pi\right] + D \qquad (21)$$

where: $D'$ = dissolved oxygen deficit adjusted for P-R impacts in mg/l

$D$ = mean 24-hour dissolved oxygen deficit in mg/l

$a$ = amplitude function for P-R activity (a constant for each station)

$t'$ = time of day in hours (0000 to 2400)

$b$ = lag time between noon and maximum production of oxygen in hours (a constant for each station)

$2\pi/24$ = conversion factor from time of day to radians

For most systems the lag time, b, is a constant for all stations since it is related only to sunlight intensity.

If the algae mass is in dynamic equilibrim, P-R fluctuations can be predicted as a function of oxygen deficit. With this, the amplitude term in equation 21 can be expressed as a function of the deficit, thereby permitting projection of P-R effects for any location where the mean oxygen deficit, D, is known.

Using the least-square regression analysis (Reference 19) for the general equation:

$$a = gD^{h} \qquad (22)$$

Where $a$ = amplitude constant for any sampling station

$D$ = corresponding mean oxygen deficit mg/l

$g$ and $h$ = constants evaluated in the least-square analysis.

The relation between stream station and amplitude can be determined.

In summary, the P-R correction factor may be applied to oxygen predictions made by continuous models for the oxygen profile. Since the empirical relationships between amplitude and deficit is derived using data from stream sampling stations both upstream and downstream from the point source wastewater discharge, it may be considered to be reasonably independent of the location on the river and thus transposable to permit prediction of diurnal effects for all cases to be considered.

## SUMMARY - MODEL DEVELOPMENT

The oxygen profile simulation model resulting from an analysis of sinks and sources for low flow streams is expressed by equation 14 with corrections for benthic demand from equation 20. This deficit projection is then adjusted by equation 21 to correct for P-R effects. The equations in order used to project dissolved oxygen deficits are:

$$D_t = \frac{K_1 L_a}{K_2 - K_1}\left[\exp(-K_1 t) - \exp(-K_2 t)\right] + \frac{K_3 N_a}{K_2 - K_3}\left[\exp(-K_3 t) - \exp(-K_2 t)\right] + D_a \exp(-K_2 t) \qquad K_1 \neq K_2 \neq K_3 \tag{14}$$

This expression projects oxygen deficit changes caused by carbonaceous BOD, nitrogenous BOD, stream reoxygenation, and the effects caused by toxic compounds (reduced $K_1$ and $K_3$) when present. The rate constants were developed in previous sections.

Adjustments in the oxygen deficit, projected by equation 13, for benthic oxygen demand are made by modifications of equation 20:

$$(D_t)_B = D_t + t_B\left\{d - B \exp\left[-a(C_s - D_t)\right]\right\} \tag{23}$$

for $0 \leq (C_s - D_t) < 4.0$ mg/l

Where: $(D_t)_B$ = Adjusted dissolved oxygen deficit with benthic demand in mg/l

$D_t$ = Dissolved oxygen deficit as calculated by equation 13 in mg/l

$C_s$ = Saturation concentration for dissolved oxygen at stream temperature in mg/l

$t_B$ = Contact time in days of water volume with benthic deposit

B = Benthic demand when stream dissolved oxygen is greater than 4.0 mg/l

a and d = empirical constant determined for equation 20.

If $D_t$ from equation 13 is small, such that $(C_s - D_t)$ is 4 mg/l or greater, then equation 23 is simplified:

$$(D_t)_B = D_t + Bt_B \qquad (24)$$

for $(C_s - D_t) \geq 4.0$ mg/l

The benthic demand correction predictions made by this equation improve in accuracy as $t_B$ approaches zero; thus, if stream reaches where benthic demand is present are evaluated in small sections and the correction factor is carried forward cumulatively, the true continuous benthic demand can be approximated (analogous to approximating an integral by an infinite series expansion). This type of evaluation could be expressed:

$$(D_t)_B = D_t + J + (t_B)_i \left\{ 1\text{-}B \exp\left[-a(C_s\text{-}D_t)\right]\right\} \qquad (25)$$

Where: J = Benthic demand that has been exerted in the sum of previously evaluated contiqeous reaches

$(t_B)_i$ = Time increment for the current approximation

In this instance, J for the next increment becomes the sum of the old J and the $(t_R)_i\{$d-B exp $-a(C_s-D_t$ ) term. The same logic also applies when $(C_s-D_t) \leq 4.0$ mg/l, this may be expressed:

$$(D_t)_B = D_t + J + B(t_B)_i \qquad (26)$$

for $(C_s-D_t) \geq 4.0$ mg/l

Adjustments in the oxygen deficit for diurnal fluctuations caused by photosynthesis/respiration are made using equations 21 and 22. These equations project the P-R impact on dissolved oxygen deficit with respect to the mean dissolved oxygen deficit calculated in equations 13, 23 and 24 for a given time of day. This correction may be expressed:

$$D_t' = g(D_t)_B^h \text{sine}\left[\left(\frac{t_D+b}{24}\right)2\pi\right] + (D_t)_B \qquad (27)$$

Where: $t_D$ = Time of day (0000 to 2400 hours)

$D_t$ = Final dissolved oxygen deficit projection in mg/l

g, h, b = Constants

$(D_t)_B$ = $D_t$ from equation 13 when benthic demand is not present

The final predicted values of dissolved oxygen deficit at pertinent points are determined by a sequence of predictions which refine the initial calculated estimates. The first calculation is accomplished using the general model, equation 13, which considers carbonaceous BOD, ammonia oxidation, stream reoxygenation, and $K_1$ rate constant which may reflect toxic effects. This prediction is then corrected for benthic oxygen demand and photosynthesis/respiration.

The low-flow stream of concern should be divided into subreaches according to their hydraulic characteristics. This segmented approach is necessary since rate constants vary greatly with hydraulic characteristics. For each subreach, water quality parameters are evaluated independently; values calculated for the downstream end of each subreach are used as the initial conditions for the contigeous subreach and time reverts to zero. If $K_3$ is a function instead of a constant, further reduction in each subreach is required to approximate the integral of the function.

# SECTION III

## MODEL APPLICATION

The model developed in the foregoing sections predicts the oxygen dynamics of a low-flow stream in its existing condition with one or more point source discharges. The objectives of applying the model to the existing system are twofold: (1) it is necessary to simulate the range of background conditions and compare the oxygen profile predicted with field data for model verification; (2) the objective of the modeling effort is to project the range of conditions to be expected after proposed mission, operational, or facility modifications are accomplished. Therefore, model verification should be stressed on the critical physical conditions of concern.

# SECTION IV

# MODEL VERIFICATION

The oxygen profile simulation model should be applied to the stream over the reaches of impact from point source discharge. This should include downstream reaches, below the point source discharge, to a point where background conditions are restored and the system is again in equilibrium.

Oxygen profiles predicted by the model and observed oxygen concentrations measured during the field studies should be compared for agreement. A complete verification matrix should be designed so that stream flow, organic load, benthic influences, and photosynthesis/respiration are considered at both mean and maximum conditions.

While verification of the model is constrained by having data at only selected locations within the system, if uniformly good agreement can be attained between predictions and observations for a wide range of matrix conditions, this suggests that the overall oxygen dynamics of the system are adequately described by the model. Therefore, it may be assumed that the model can be successfully transposed, with appropriate input assumptions, to predict future conditions within the limits of statistical confidence gained during model verification.

## SECTION V

## SUMMARY AND CONCLUSIONS

For low-flow streams which receive wastewater discharges, the oxygen dynamics may be significantly influenced by a wide variety of factors. In this investigation, techniques were developed, using a rationale previously proposed (References 14 and 15) to consider the effects of carbonaceous BOD, nitrogenous BOD, stream reoxygenation, benthic oxygen demand, and photosynthesis/respiration. In addition, the effects of toxic compounds upon the waste assimilative capacity were discussed. Methods to verify the ability of the model were discussed. The ability of the proposed techniques to simulate water quality conditions caused by future mission, operation, or facility modifications were discussed.

Since very few previous waste assimilative capacity investigations (References 14 and 15) have attempted to investigate the many aspects involved in the oxygen dynamics of low flow streams receiving wastewater discharges, it was necessary to develop guidelines for conducting this type of investigation during the course of this study. The techniques adopted for use herein may be considered as one approach to evaluating the behavior of low flow systems. The validity and limitations of model extrapolation to project current and future conditions is not fully substantiated; however, some investigations (References 14 and 15) have reported excellent results. Therefore, it is believed that the potential value of this model and its rationale is significant.

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