United States
Environmental Protection
Agency

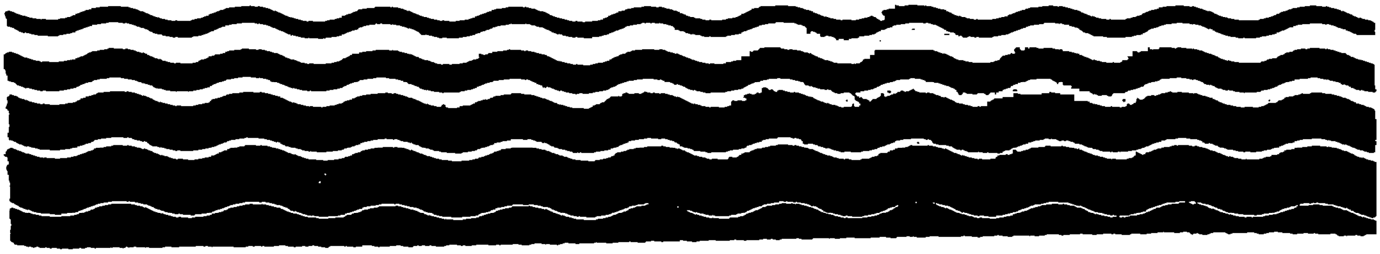
Office of Water Regulations
and Standards
Monitoring and Data Support
Division (WH-553)

Water

EPA

Technical Guidance Manual for Performing Waste Load Allocations

Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams



PREFACE

This document describes a simplified method for conducting waste load allocations (WLA) for relatively small municipal sewage treatment facilities discharging into low-flow streams, circumstances in which resources for analysis and data acquisition are relatively limited. The methodology, issued jointly by the Office of Water Regulations and Standards and the Office of Municipal Pollution Control, is subject to modifications as recommendations and/or more complete data become available. Thus, the method is recommended as an initial framework, subject to revision and site-specific considerations, for conducting simplified WLA's.

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Simplified Analytical Method
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NATIONAL GUIDANCE

Monitoring Branch

Monitoring and Data Support Division

Office of Water Regulations and Standards

September 26, 1980

I. INTRODUCTION

A. General

A simplified analytical method for determining effluent limitations for publicly-owned treatment works (POTWs) discharging into low-flow streams has been developed for nationwide use. This method should help ensure that proposed construction grant projects have adequate water quality justifications based on technically sound water quality analyses, and that construction grant funds are used in a cost-effective manner.

The analytical techniques which are used in water quality modeling should be the simplest possible that will still allow the water quality manager to make confident and defensible water pollution control decisions. In many cases, where similar and relatively simple conditions exist, simplified modeling efforts that have less extensive manpower and data requirements (than normal, more comprehensive efforts) are often adequate to make such decisions. Use of simplified efforts, when appropriate, can result in both substantial savings in State and EPA resources and cost-effective and technically sound effluent limitations that will protect designated water uses and allow water quality standards to be achieved.

This simplified analytical method has been developed because of the large number of relatively small municipal sewage treatment facilities discharging into low-flow streams, and the need for more cost-effective yet technically sound water quality analyses for these cases. For example, this simplified method may be applicable to over 50 percent of the existing construction grant projections in Region V. Additionally, this method will help ensure that similar dischargers in similar situations will receive consistent consideration.

It should be noted that the analytical techniques described below are intended to represent minimum levels of analysis acceptable to EPA as justification for treatment beyond secondary. Water quality analyses may, of course, employ more rigorous techniques incorporating detailed intensive surveys and more complex models. EPA, however, will not accept further simplification of the methods described below.

unless such simplifying assumptions are adequately supported by a technical justification.

B. Modifications to the Region V Technology

The simplified method discussed below is based on the approach originally proposed by Region V, and on comments and suggestions received in response to the Region V proposal. Though generally supportive of the overall approach, comments revealed some concern regarding some details on calculating rate constants, DO targets, and permit conditions. Consequently, several modifications have been made to the Region V proposal.

C. Needs for Additional Data and Future Modifications

The simplified method described below is based on the information and data available to date. The data bases for many of the recommendations are quite limited, and should be expanded. Users of this method (as well as others) are strongly encouraged to develop additional data and other information on reaction rates and other factors applicable to this method, and to submit this additional data and other information, along with suggested improvements for the method, to:

Chief, Wasteload Allocations Section
Monitoring Branch
MDSD/OWRS (WH-553)
U.S. Environmental Protection Agency
401 M Street, S.W.
Washington, D.C. 20460

(Telephone: (202) 382-7056)

This additional data and other information is needed so that appropriate additional improvements to the method can be made periodically. Users may also, when appropriate, make any modifications to this method that will allow the method to more accurately represent regional or local conditions. Any changes should be supported with an adequate technical justification, including sufficient applicable data.

Areas in the method that require additional research and/or data include:

- K-rates (reaeration, CBOD, NBOD), including relating them to waterbody characteristics, levels of treatment, etc.
- $CBOD_u/BOD_5$ ratio
- diurnal fluctuation factors
- methods for performing the sensitivity analysis
- sediment oxygen demand rates, including relating them to stream bottom characteristics, levels of treatment, etc.

Where the results of the water quality analysis indicates the need for treatment beyond secondary, State users of this method are encouraged to coordinate the modeling analysis with a review of the environmental benefits and costs of the receiving water's applicable water quality standards. Such assessments should be conducted in accordance with the revised water quality standards regulations and EPA guidance, when published.

II. APPLICATION AND CONSTRAINTS

This method may be applied only if <u>all</u> of the following conditions are met:

1) The discharger must be a publicly-owned treatment works (POTW) receiving predominantly sanitary wastewaters. Any nonsanitary wastewaters in the treatment plant's influent must exhibit essentially the same characteristics (e.g., reactions) as sanitary wastes.

2) The discharge must be to a free-flowing stream in which the design low flow (usually the 7-day, 10-year low flow) is approximately equal to or less than the design discharge flow from the POTW.

3) The design discharge flow from the treatment plant must be 10 MGD (15.5 cfs) or less.

4) There is no significant interaction between the discharger being analyzed and any other upstream or downstream discharger.

Recent experience and analyses indicate that this simplified method, when followed properly and with little or no site-specific data being employe should normally result in both technically sound water quality justifications being developed for nitrification levels of treatment and substantial savings in State and EPA resources. However, it has also been noted that this simplified analysis alone (i.e., without any site-specific data) usually cannot provide the confidence needed to adequately justify permit limits more stringent than about 10 mg/l $CBOD_5$ and 1.5 mg/l NH_3-N, including relatively costly filtration treatment after nitrification. Therefore, this simplified method cannot be used by itself to justify permit limits more stringent than 10 mg/l $CBOD_5$ and 1.5 mg/l NH_3-N (including filtration after nitrification).

Where treatment more stringent than 10 mg/l $CBOD_5$ and 1.5 mg/l NH_3-N (including filtration after nitrification) appears to be needed, appropriate supporting site-specific data should be collected and used in the analysis in order to increase confidence in the variables used in this method, in the modeling results that are obtained, and, most importantly, in the treatment decision itself. This additional level of analysis should also be accompanied by a rigorous sensitivity analysis (see Section III-C of the method). Based on past analyses and construction grant project reviews, it appears that this situation (e.g., the need for treatment beyond nitrification) will seldom be required except in certain cases where small streams with very low assimilative capacities are encountered.

Water quality in this type of system is highly dependent on effluent quality. Hence, upstream quality is less significant here than in systems where the upstream design flow is much greater than design effluent flows. This method can also be applied to simple systems where the upstream flow is greater than the POTW's discharge flow, provided the upstream water quality and reaction kinetics are well documented.

III. PROCEDURE

In order to determine the level of treatment required for a POTW, the following analytical steps are recommended:

(a) gather necessary data

(b) perform an ammonia toxicity analysis

(c) perform a dissolved oxygen analysis

(d) perform a sensitivity analysis

(e) interpret the results, and determine the final effluent limitations.

These five steps of the Simplified Method are discussed below.

A. Data Requirements

The data required for the Simplified Method, and some of their possible sources are listed below:

1) stream design flow-sources include USGS low-flow publications; drainage area yields; measurements during low-flow periods.

2) upstream water quality-including the necessary DO, BOD, ammonia, pH, alkalinity, temperature and other data needed for this Method. Sources include historical data (e.g., in STORET); State, EPA, or other water quality monitoring; sewage treatment plant monitoring; transferable data from similar streams.

3) stream physical characteristics-including stream slope, depth, etc. Sources include field measurements; USGS topographic maps; special Corps of Engineers or county project maps; stream gazetteers.

4) time of travel/velocity-sources include dye studies; direct velocity measurements; calculations based on field measurements of widths, depths, etc.; estimates based on slope/velocity relationships.

5) effluent design flow-sources include State or local agency population projections; Step 1 applications.

6) characteristics of design effluent-including the necessary pH, alkalinity, temperature, and other data needed for this Method. Sources include treatment capabilities for different levels of treatment, presented herein; other data can be obtained from State, EPA, or other water quality/effluent monitoring; sewage treatment plant monitoring; transferable data from similar treatment plants.

Direct field measurements of time-of-travel/velocity, upstream quality, stream physical characteristics (such as depth, type of bottom, benthic deposits, etc.), and other data should be employed for each segment studied, most notably for those where post-filtration of the sewage treatment plant effluent is considered. Since these data are readily obtainable by means of short duration, low resource surveys, efforts should be made to obtain the data through State agency monitoring programs or as part of the 201 grant process. When such data are not available, estimates can be made from some of the suggested sources listed above. The impact of less site-specific data should be considered in the sensitivity anlysis. Time-of-travel studies provide the most useful data when the upstream flow and existing sewage treatment plant flow are equivalent to the sum of the upstream $_7Q_{10}$ and the treatment plant design flow. If flows in the immediate range of the design flow are not encountered during the time-of-travel studies, a second study at a different flow will permit extrapolation of the data to the design flow.

As a minimum, all modeling efforts must include: (1) a search for all applicable historical data and information (e.g., in STORET, old modeling or water quality study reports treatment plant records, etc.) to support the current modeling work, and (2) a general on-site reconnaissance visit to visually observe the system to be modeled (to gain a better intuitive understanding of the system).

B. Ammonia Toxicity Analysis

A mass balance analysis will be used to determine whether the nitrification unit process is required on the basis of instream ammonia toxicity. The total ammonia-N limitation for the proposed discharge will be determined by using the applicable water quality standards (WQS), upstream flow and background concentration, and design effluent flow as follows (2):

$$C_D = (C_{WQS}(Q_D+Q_U) - C_U Q_U)/Q_D \qquad \text{(Eq. 1)}$$

where C_D = allowable design discharge concentration of total ammonia-N for POTW,

C_{WQS} = water quality standard limit of total ammonia-N (usually based on un-ionized ammonia-N standard and selected pH and temperature),

C_U = upstream or background concentration of total ammonia-N,

Q_D = design POTW discharge flow rate,

Q_U = upstream design low-flow.

The allowable instream total ammonia-N concentration (C_{WQS}) will normally be based on the water quality standard for un-ionized ammonia-N and the expected values of pH and temperature downstream of the discharge (if an applicable total ammonia-N standard is specified, C_{WQS} will equal that standard). The value of C_{WQS} can be determined from a table or graph which relates the toxicity of un-ionized ammonia-N to pH and temperature (such a table and graph is presented in Exhibit 1). When selecting C_{WQS}, one should be sure to use appropriate values for the expected downstream pH and temperature conditions during the design season, after mixing of the discharge and the receiving stream. If no un-ionized or total ammonia-N standards are available for use, the following criteria are recommended:*

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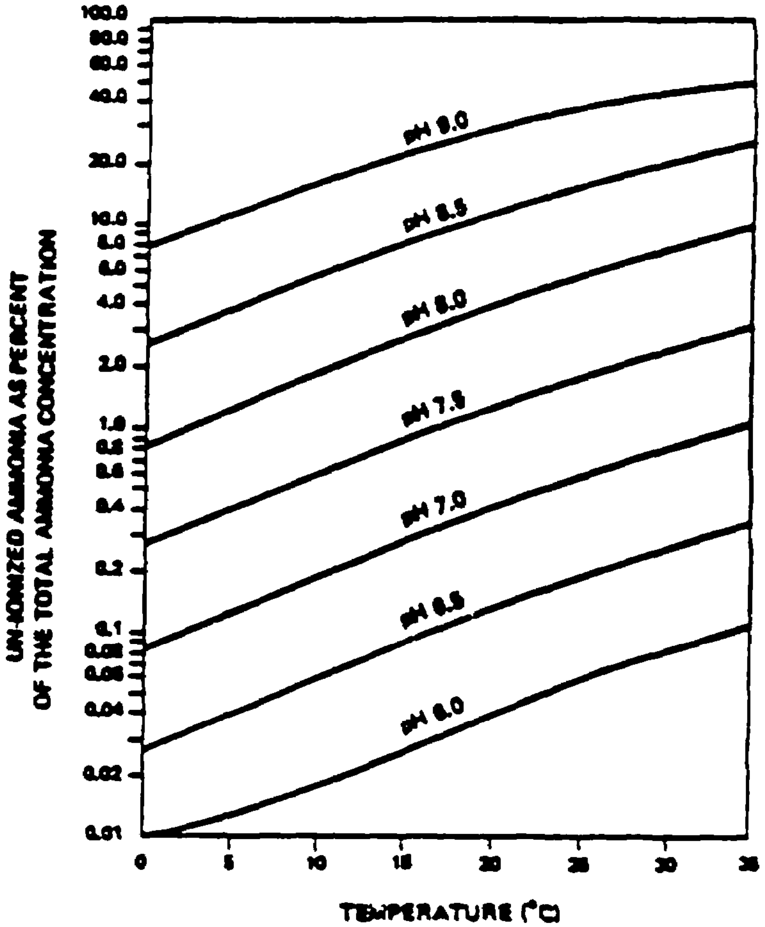
*Additional research indicates that these criteria may in many cases be more stringent than necessary to protect water quality. It is recommended that the latest EPA ammonia toxicity criteria, when promulgated, be used in conjunction with the supporting ammonia criteria implementation guidance document. Until the new EPA ammonia toxicity criteria are promulgated, AT facilities proposed solely to prevent ammonia toxicity may be approved only with supporting justifications based on either: (1) site-specific biological data showing that the designated uses cannot be restored without reducing ammonia toxicity, or (2) bioassay data (either from a laboratory or similar site) for indigenous species showing that existing or future ammonia toxicity levels will impair designated use attainment (exposure levels and durations for these tests should be similar to those occurring or anticipated to occur in the receiving water). Note: After publication of new ammonia toxicity criteria by EPA. Advanced Treatment processes proposed solely to prevent ammonia toxicity may be approved consistent with these criteria and this simplified method.)



Note: Redrawn from William T. Willingham, Ammonia Toxicity, USEPA 908/3-76-001, Feb. 1976

Percent un-ionized ammonia in aqueous ammonia solutions[a]

Temperature (°C)	pH Value								
	6.0	6.5	7.0	7.5	8.0	8.5	9.0	9.5	10.0
5 —	0.013	0.040	0.12	0.39	1.2	3.8	11.	28.	56.
10 —	0.019	0.059	0.19	0.59	1.8	5.6	16.	37.	65.
15 —	0.027	0.087	0.27	0.86	2.7	8.0	21.	46.	73.
20 —	0.040	0.13	0.40	1.2	3.8	11.	28.	56.	80.
25 —	0.057	0.18	0.57	1.8	5.4	15.	36.	64.	85.
30 —	0.080	0.25	0.80	2.5	7.5	20.	45.	72.	89.

[a] [Thurston, et al. (1974)]

Exhibit 1. Percentage of un-ionized ammonia in ammonia-water solution at various pH and temperature values.

- 0.02 mg/l un-ionized ammonia for freshwater cold water habitat
- 0.05 mg/l un-ionized ammonia for freshwater warm water habitat.

The expected values of pH and temperature downstream of the discharge should be based on the pH and temperature of the POTW effluent and of the upstream waters. If sufficient temperature data are available or can be estimated for the POTW and for the stream (upstream of the discharge point), the expected downstream temperature can be calculated as follows:

$$T_R = (Q_U T_U + Q_D T_D)/(Q_U + Q_D) \qquad \text{(Eq. 2)}$$

where T_R = resultant water temperature downstream of discharge after mixing.

Q_U = upstream design low-flow.

Q_D = design (POTW) discharge flow rate.

T_U = upstream water temperature.

T_D = temperature of POTW effluent.

Also, if sufficient pH data is available or can be estimated for the POTW and for upstream, the method outlined in Appendix A1 can be used to determine the expected downstream pH.

When sufficient data is available for pH and temperature, use of the maximum pH and temperature values ever recorded is generally not realistic. It is more appropriate to use pH and temperature values which are exceeded 25 percent of the time during the critical low-flow season. It appears that the likelihood of having simultaneous "worst case" occurrences of both pH and temperature significantly greater than their respective 25 percent exceedence values along with design low-flow conditions is not very great; however, additional research should be conducted in this area. When actual stream or POTW data are limited or not available, data from similar nearby streams and POTWs, or equilibrium water temperature data, may be used as design conditions and to help establish the range for a sensitivity analysis. In this analysis, the pH and temperature values which are used should be supported by one or more of the means described above. Any deviations must be supported by a sound technical justification.

This mass balance analysis should be used to determine whether the nitrification unit process is required solely on the basis of instream ammonia toxicity. Since nitrification in sewage treatment plants is generally an "all or nothing" process, the water quality analyst should be aware that, in terms of treatment capital costs, there may be no difference between an ammonia effluent limitation of, say, 2 mg/l, and 8 mg/l. For example, the cost of building a treatment plant to produce an effluent quality of 2 mg/l ammonia-N, may not be substantially different from that for a treatment plant designed to produce an effluent quality of 8 mg/l. For further details see the discussion below on treatment capabilities.

In light of the above discussion, this mass balance approach should be used to estimate instream un-ionized ammonia concentrations with and without nitrificaiton at the treatment plant. In cases where it appears that only marginal violations of the instream ammonia standard will result without nitrification at the plant, the decision to provide nitrification can be deferred until the dissolved oxygen

(DO) analysis (Step C, below) has been completed. The DO analysis may indicate the need for a reduction of ultimate oxygen demand (UOD), which would also support the need for the nitrification unit process at the plant (a process known to be more economical than filtration) for UOD removal. However, the latter conclusion is based on the implicit assumption that all of the instream ammonia will exert an oxygen demand in the stream segment under consideration, i.e., that nitrification occurs instream.

In situations where the DO analysis does not indicate a need for advanced treatment levels, but the ammonia toxicity analysis predicts toxicity problems, consideration shall be given to using pH adjustments (i.e., pH reductions) of the effluent during critical conditions to control ammonia toxicity in lieu of requiring nitrification. This consideration should include a determination of whether the temporary lowering of pH and increase in total dissolved solids (TDS) concentration would have any significant instream ecological or other effects.

In addition to this mass balance analysis, a qualitative if not quantitative assessment of nonpoint source contributions of ammonia must be made. This assessment may reveal that nonpoint source pollution may be of sufficient magnitude to preclude attainment of water quality objectives in terms of ammonia concentrations. An example which illustrates this point would be a treatment plant discharge located in a predominantly agricultural watershed that has significant nonpoint source problems. Because nitrification alone may not solve an ammonia toxicity problem, nonpoint source controls must also be considered before nitrification is chosen. National guidance for nonpoint source analysis related to facility planning is presently being developed and is expected in the near future.

C. Dissolved Oxygen Analysis

A simplified Streeter-Phelps (3) analysis will be used to determine the effluent dissolved oxygen (DO) and BOD limitations for the POTW. This approach incorporates both carbonaceous (CBOD) and nitrogenous (NBOD) oxygen demands in the analysis. The equation used to calculate the DO deficit downstream from the point source is shown below:

$$\begin{aligned} D = {} & D_o \exp(-K_2 t) \\ & + [K_1\ CBOD_o]/[K_2 - K_1][\exp(-K_1 t) - \exp(-K_2 t)] \\ & + [K_3\ NBOD_o]/[K_2 - K_3][\exp(-K_3 t) - \exp(-K_2 t)] \\ & + S/HK_2[1 - \exp(-K_2 t)] \end{aligned} \qquad \text{(Eq. 3)}$$

where D = the DO deficit (mg/l),

D_o = mixed initial DO deficit (at discharge point) (mg/l)

$CBOD_o$ = mixed ultimate CBOD concentration at discharge point (mg/l),

$NBOD_o$ = mixed NBOD concentration at dischage point (mg/l). The magnitude of the NBOD should be based on the total ammonia concentration, and can be estimated using the following stoichiometric relationship: NBOD = 4.57 (NH_3-N),

K_1 = CBOD reaction (decay) rate (base e) (1/day).

K_2 = reaeration rate (base e) (1/day)

K_3 = NBOD reaction (decay) rate (base e) (1/day).

t = travel time below discharge (days).

S = sediment (benthic) oxygen demand ($gm/m^2/day$).

H = stream depth (meters).

The instream average DO concentration at a given point downstream of the discharge point is calculated by using the following equation:

$$DO_{AVG} = DO_{SAT} - D \qquad \text{(Eq. 4)}$$

where DO_{AVG} = instream average DO concentration (mg/l).

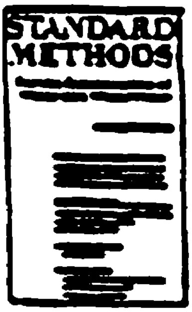
DO_{SAT} = saturation DO concentration at specified water temperature (mg/l). This can be determined from Exhibit 2.

D = as defined above.

Calculations using the above equations and a specified set of coefficients and assumptions apply for a given uniform stream reach. It is important to subdivide the stream into individual uniform reaches wherever any significant system changes occur (e.g., changes in channel geometry, significant tributary inflows, etc.) so that appropriate coefficients and assumptions that adequately represent each reach can be applied to the respective stream reaches when the model calculations are made.

Water-Saturated Air (mg/l)**

Temperature	Chloride Concentration in Water mg/L				
C	0	5,000	10,000	15,000	20,000
0	14.60	13.72	12.90	12.13	11.41
1	14.19	13.35	12.56	11.81	11.11
2	13.81	12.99	12.23	11.51	10.83
3	13.46	12.65	11.91	11.22	10.56
4	13.09	12.33	11.61	10.94	10.30
5	12.75	12.02	11.32	10.67	10.05
6	12.43	11.72	11.05	10.41	9.82
7	12.12	11.43	10.78	10.17	9.59
8	11.83	11.16	10.53	9.93	9.37
9	11.55	10.90	10.29	9.71	9.16
10	11.27	10.65	10.05	9.49	8.96
11	11.01	10.40	9.83	9.28	8.77
12	10.76	10.17	9.61	9.08	8.58
13	10.52	9.95	9.41	8.89	8.41
14	10.29	9.73	9.21	8.71	8.24
15	10.07	9.53	9.01	8.53	8.07
16	9.85	9.33	8.83	8.36	7.91
17	9.65	9.14	8.65	8.19	7.76
18	9.45	8.95	8.48	8.04	7.61
19	9.26	8.77	8.32	7.88	7.47
20	9.07	8.60	8.16	7.73	7.33
21	8.90	8.44	8.00	7.59	7.20
22	8.72	8.28	7.85	7.45	7.07
23	8.56	8.12	7.71	7.32	6.95
24	8.40	7.97	7.57	7.19	6.83
25	8.24	7.83	7.44	7.06	6.71
26	8.09	7.69	7.31	6.94	6.60
27	7.95	7.55	7.18	6.82	6.49
28	7.81	7.42	7.06	6.71	6.39
29	7.67	7.30	6.94	6.60	6.29
30	7.54	7.17	6.83	6.49	6.19
31	7.41	7.05	6.72	6.39	6.09
32	7.28	6.94	6.61	6.29	5.99
33	7.16	6.82	6.50	6.19	5.90
34	7.05	6.71	6.40	6.10	5.81
35	6.93	6.61	6.30	6.00	5.72
36	6.82	6.51	6.20	5.92	5.64
37	6.71	6.41	6.11	5.83	5.56
38	6.61	6.31	6.02	5.74	5.48
39	6.51	6.21	5.93	5.66	5.40
40	6.41	6.12	5.84	5.58	5.33
41	6.31	6.03	5.76	5.50	5.25
42	6.22	5.94	5.68	5.42	5.18
43	6.13	5.85	5.60	5.35	5.11
44	6.04	5.77	5.52	5.27	5.04
45	5.95	5.69	5.44	5.20	4.98
46	5.86	5.61	5.37	5.13	4.91
47	5.78	5.53	5.30	5.06	4.85
48	5.70	5.45	5.22	5.00	4.78
49	5.62	5.38	5.15	4.93	4.72
50	5.54	5.31	5.08	4.87	4.66



** At a total pressure of 101.3 kPa. Under any other barometric pressure, *P*, obtain the solubility, *S'* (mg/L) from the corresponding value in the table by the equation

$$S' = S\frac{P-p}{760-p}$$

in which *S* is the solubility at 101.3 kPa and *p* is the pressure (mm) of saturated water vapor at the water temperature. For elevations less than 1,000 m and temperatures below 25 C, ignore *p*. The equation then becomes

$$S' = S\frac{P}{760} = S\frac{P'}{29.92}$$

Dry air is assumed to contain 20.90% oxygen. (Calculations made by Whipple and Whipple, 1911. J. Amer. Chem. Soc. 33:362.)

The universe of possible coefficients and assumptions that can be used in the model may vary over a wide range, and can result in significantly different predictions of instream DO. It is extremely important that the best possible estimate of these parameters be obtained for each case to ensure that they adequately represent the stream system to be modeled. Site-specific data should be used whenever possible. When a comprehensive field survey is not feasible, appropriate data described in the literature or available transfer data may be used. However, if literature data or transfer data are used, it is strongly encouraged that a general reconnaissance visit of the site to be modeled be made in order to qualitatively evaluate the applicability and reasonableness of the data being used. The coefficients and assumptions to be used and minimum site-specific data requirements for this Method are discussed further, in the next section. It is important to ensure that the coefficients and assumptions which are used in the model do adequately represent the conditions in each uniform stream reach, and that they are changed in the model, as appropriate, between the stream reaches being modeled to reflect any system changes between the reaches (e.g., make appropriate changes in coefficients, travel time, etc., to reflect significant changes in slope, channel geometry, benthic characteristics, etc.)

After the appropriate assumptions are made and coefficients selected, the location of minimum DO concentration (i.e., the sag point) is to be determined. This can be accomplished by applying incremental time periods in Equations 3 and 4 and noting where the minimum DO concentration occurs. The number of trial-and-error iterations can be minimized by first using the following equation to determine the approximate location of the sag point:

(Eq. 5)

$$t_c = [1/(K_2 - K_{BOD})] \ \ln \ [(K_2/K_{BOD}) \ (1-[D_o(K_2-K_{BOD})/(K_{BOD} \ UBOD_o)])]$$

where t_c = approximate time (and, hence, distance) to the sag (critical) point.

K_2 = reaeration rate.

K_{BOD} = BOD reaction rate (for CBOD, NBOD, or average of two rates; see text, below).

D_o = mixed initial DO deficit (at discharge point).

$UBOD_o$ = initial ultimate BOD concentration (at discharge point).

Where CBOD and NBOD have different rates, the sag point can be bounded by assigning first the lower and then the higher rate to the total ultimate BOD, or approximated by averaging the two rates.

The next overall step in this Method is to establish the allowable loading rates needed in order to meet the water quality standards at the critical sag point. This can be accomplished by applying, in an iterative manner, successively lower CBOD and NBOD values until the DO standards are met at the sag point. An alternative to this "trial and error" method for dissolved oxygen is to separately calculate the dissolved oxygen deficit due to each BOD source (e.g., upstream BOD, sediment demand, plant carbonaceous BOD, and plant nitrogenous BOD).

By knowing the relative contribution of each BOD source to the total deficit when using this alternative approach, and since the various oxygen deficits are additive (see Equation 3), it is an easy task to identify the combinations of wastewater BOD reductions that will achieve water quality standards and to select the combination that will be the most cost-effective.

As stated above, it is extremely important that the best possible estimate of the parameters used in the model be obtained for each case, and that they adequately represent the system being modeled. The next several subsections discuss recommended parameter values and assumptions to be used with the Simplified Method. The following points are addressed:

- flow regime
- target DO
- treatment capabilities
- rate constants
- inital deficit determination
- conversion from $CBOD_u$ to BOD_5
- post-aeration
- nonpoint sources.

The recommendations made below are intended to help the Simplified Method user to develop modeling results that are consistent with the applicable water quality standards.

1) Flow Regime. Both low-flow and high-flow conditions should be assessed to determine the critical conditions. In some cases, severe non-point source pollutant contributions during high-flows might

preclude attainment of desired water quality objectives. If the high-flow condition is found to be critical, further analyses of nonpoint sources should be conducted.

The flow regime to be used in the Simplified Method is the design low-flow specified in the applicable water quality standards. This flow will, in most cases, be the 7-day, 10 year low-flow ($_7Q_{10}$). If no design flow is specified, the $_7Q_{10}$ should be used for design purposes. If any flow value other than those specified above is used in the analysis, a sound justification must be provided to support the use of this value.

2) *Target Dissolved Oxygen*. DO standards are often presented as a minimum *at all times*; some States include an average value along with the minimum. Outputs (the DO simulation) from steady-state models are based on the averaging period for input loadings; they, therefore, represent the average DO conditions likely to prevail at the flow condition being simulated. The two major factors that can cause the actual minimum DO to be considerably lower than the average predicted by steady-state model computations are: (1) diurnal variations in loadings to and from the contributing waste treatment plant, and (2) diurnal variations of instream DO caused by algal photosynthesis and respiration. The magnitude of these variations is likely to differ from plant to plant and from stream to stream. The problem is further complicated by the fact that prevailing fluctuations in a

stream may be radically altered under construction of a larger treatment plant with higher levels of treatment, and that quantification of these fluctuations through intensive field investigations may not accurately define these future conditions. For this reason, users of this Method are urged to obtain additional field data so that instream responses can be better correlated with different levels of treatment, and that better estimates under projected conditions can be made.

When modeling, the following DO targets should be used:

(a) If the DO standard is expressed as an average and a minimum requirement (e.g., an average of 5 mg/l and a minimum of 4 mg/l), the average number (e.g., 5 mg/l) should be used as the target.

(b) If the DO standard is expressed only as a minimum (e.g., a minimum of 5 mg/l at all times), the target DO may be obtained by adding one-half of the diurnal variation to the DO standard (e.g., for a total diurnal variation of 1 mg/l, then the target is 5 mg/l + 0.5 mg/l = 5.5 mg/l). In the absence of adequate site-specific or transferable data, 0.5 mg/l should be added to the DO standard. If any other values (either lesser or greater than 0.5) are used, they must be supported by adequate data.

There is not a strong quantitative basis for using the recommended value of 0.5 mg/l to compensate for diurnal variations; the choice is based in part on the acknowledgement of the existence of such variations and on the need to allow some reasonable compensation in the absence of an adequate data base. However, Thomann (19) feels that there is a basis for using the 0.5 mg/l value, but this value is associated more with random, rather than just photosythesis/respiration, fluctuations. Additional field studies should be conducted to either support or modify this value.

3) Treatment Capabilities. For typical domestic wastewaters, the following effluent concentrations should ordinarily be assumed for modeling purposes. These values are 30-day averages that would be expected during warm summer months (i.e., during conditions similar to those being modeled).

Treatment	Effluent Concentration (mg/l)	
	BOD_5	NH_3-N
Influent	100 - 300	12 - 35
Secondary	30 (or 85% removal)	10% less than raw concentration
Nitrification (single stage or two stage)	5 - 8**	1.0 - 1.5
Oxidation Ditch	10 - 15	1.0 - 1.5
Nitrification plus Tertiary Filtration	3 - 5**	1.0 - 1.5

**(See Appendix A2 for clarification.)

The values selected from the ranges given above should depend on the influent concentrations (e.g., lower values should be used for lower influent concentrations).

4) Rate Constants

(a) Reaeration Rate(K_2). The critical values in DO analyses of small streams are the reaeration rate, and to a lesser extent, the CBOD and NBOD decay rates and effluent DO levels. Many formulations have been developed for predicting stream reaeration rates based on physical characteristics such as stream width, depth, velocity, and slope (4,5). Recent work by Rathbun (6) and by Grant and Skavroneck (8) indicates that the Tsivoglou formula (7,20), in which K_2 is calculated by Equation 6, tends to most accurately predict stream reaeration. Presently, the data base on which most reaeration equations and recommendations are based is quite limited. Additional data collection efforts and research are needed in this area.

The Tsivoglou formula, presented below, should be used for computing reaeration rates on small, shallow streams:

$$K_2 = CVS \quad \text{at } 20^{\circ}C \qquad (\text{Eq. } 6)$$

where K_2 = reaeration rate (1/day)

V = stream velocity (ft/sec)

S = stream slope (ft/mi)

C = proportionality constant with the values shown below:

C = 1.8 for $1 \leq Q \leq 10$ cfs
= 1.3 for $10 < Q < 25$ cfs
= 0.88 for $25 \leq Q \leq 300$ cfs.

According to Tsivoglou (9), there is no evidence to support restricting the use of Equation 6 based on minimum slope. It should be pointed out that the values of the proportionality constant (C) given above for flows between 1-10 cfs and 25 - 300 cfs were determined by Tsivoglou (20) using data in these flow ranges. The value of C for the flow range between 10 and 25 cfs appears to be basically an estimated value only, since a paucity of data exists in this flow range. However, this C value has been recommended by Neal (22). When the streamflow (Q) is near either 10 or 25 cfs (the points at which the value of C changes), it is recommended that a sensitivity analysis be performed by doing separate DO calculations using both the higher and lower C values so that lower and upper limits of predicted DO can be established. If this range of predicted DO is found to be relatively large, it is recommended that additional work, including field measurements, be performed to help reduce the uncertainty in the reaeration rate to be used.

O'Connor's reaeration formula, presented below, may be used for larger, deeper streams with more uniform channel geometry or those with significant pooling (4):

$$K_2 = 12.9\ V^{0.5}/H^{1.5} \quad \text{at } 20^\circ C \qquad \text{(Eq. 7)}$$

where V = stream velocity (ft/sec),
H = average stream depth (ft).

The values of V and H used in the above equations should be based on actual field measurements so that the uncertainty in the rate can be reduced. The value of S can be determined from field measurements or from appropriate maps.

Any other applicable reaeration prediction methods may be used in lieu of the above methods only if these alternative methods are supported by an adequate technical justification that includes sufficient field data collected from the area.

(b) <u>CBOD Decay Rate(K_1)</u>. A review of reaction rates measured on low flow streams with similar characteristics showed that CBOD rates generally range from about 0.2 to around 3.0 or more (10, 11, 12, 13, 21.), depending in part on depth and degree of treatment (see

Exhibit 3). The minimal data available for small, low flow streams with treatment greater than secondary suggest that the CBOD rates under these conditions typically fall between about 0.2 and 0.5 (at 20°C). Adjusting CBOD rates by depth as proposed by Hydroscience (13) suggests 0.3 to be a representative value for these low flow streams with a mostly stable fairly rocky bottom, and about 0.2 for streams with a primarily unstable sediment bottom.

Using this approach, CBOD then becomes (14):

$$K_1 = C\,(H/8)^{-0.434} \qquad \text{for } H < 8 \text{ ft.} \qquad \text{(Eq. 8)}$$

$$= C \qquad \text{for } H \geq 8 \text{ ft.}$$

where K_1 = CBOD decay rate (1/day),

H = average stream depth (ft),

C = 0.3 for streams with mostly stable fairly rocky bottoms,

C = 0.2 for streams with primarily unstable sediment bottoms.

It should be noted that Equation 8 generally represents an average of a range of possible values at any given depth. Based on limited data presented in the literature (11, 13, 21) and elsewhere (10, 12; also see Exhibit 3), the following ranges of the CBOD rate are presently being suggested:

Exhibit 3. REACTION RATES MEASURED ON LOW FLOW STREAMS**

River	CBOD Rate		NBOD* Rate	Treatment Level	Flow (cfs)		Depth (ft
	Measured	Depth Adjusted			River	STP	
Upper Olentangy, Ohio (7)	1.24	0.43	—	AST - No nitrification	2.3	3.0	0.7
Patuxent River, (8) Maryland	0.30	0.3	.27-.50	AWT - Nitrification with microscreens	31	6.8	—
West Fork of Blue River, Indiana (9)	0.5-0.79	0.2-.32	0.5	AST - Secondary with rapid sand filters	4	1.4	0.9-1.2
Hydroscience (10)	.37-.96	.19-.42		Highly treated effluent nitrification			1-3
Recommended Values		0.3	.42				
Depth Adjustment $K = K (\text{Measured}) (D/8)^{.434}$			$D < 8$ ft				

*Reaction rates are at 20°C.
**Submitted by EPA Region V.

Exhibit 6. CBOD/BOD$_5$ DATA*

State	Plant	Type	Flow (MGD)	Percent Industrial Flow	Number of Samples	Ultimate CBOD BOD$_5$
Ohio	Lakewood	Activated Sludge	11.7	0%	1	3.27
Ohio	Mansfield	Activated Sludge	10.6	32%	1	3.43
Ohio	Shelby	Activated Sludge	1.2	0%	3	3.21
Ohio	Lorain	Activated Sludge	14.1	14%	4	3.13
Ohio	Coshocton	Activated Sludge	2.3	39%	1	4.34
Ohio	Conneaut	Activated Sludge	2.6	0%	1	2.61
Ohio	CRSD Easterly	Activated Sludge	136.0	12%	2	5.10
Minnesota	Minneapolis-St. Paul	Activated Sludge		27%	13	3.18
Wisconsin	Fall Creek	Trickling Filter			2	3.40
Wisconsin	Neenah-Menasha	Activated Sludge		40%	2	3.20
Wisconsin	Town Menasha East	Activated Sludge		22%	1	1.80
Wisconsin	Town Menasha West	Activated Sludge		42%	2	3.10
Wisconsin	Heart of the Valley	Act. and filters	5	< 10%	2	2.75
Wisconsin	Depere	Activated Sludge		20%	1	3.00
Average						3.2

*Submitted by EPA Region V.

Level of Treatment	CBOD Rate (K_1)**	
	Minimum	Maximum
Secondary treatment	0.3	1.5
Greater than secondary		
• streams with mostly stable fairly rocky bottoms.	0.3	0.5
• streams with primarily unstable sediment bottoms.	0.2	0.4

(Note: These ranges are based on a very limited data set; they are subject to modification, as necessary, as additional rate data are submitted to EPA Headquarters. Users are urged to collect site-specific rate data whenever possible.)

**-(1/day, base e, at 20°C)

Equation 8 may be used to estimate the CBOD rate within the ranges specified above. Users of this approach should note that, at best, the above equation is a crude, though rational, empiricism, based on a limited data set. A much larger data set consisting of accurate K_1 measurements for different levels of treatment and types of streams is needed before a more precise empirical correlation equation can be developed. Towards this end, States and EPA Regions are urged to expand relevant data bases, and to submit these data and suggestions to EPA Headquarters to assist in the refinement of the above approach and ranges. Post-construction intensive surveys to measure K_1 (and K_3) below AST and AWT plants would aid this effort significantly. Users are encouraged to collect site-specific data or to use transferable data to help reduce the uncertainty in the CBOD rate to be selected. Any deviations from the above approach must be supported by an adequate technical justification.

(c) NBOD Decay Rate (K_3). Several environmental factors have been shown to influence the rate at which nitrification occurs. Among them are pH, temperature, suspended particle concentration, hydraulic parameters, other pollutants that inhibit the nitrification process (e.g., some toxics), and the benthos of the receiving waters. While no attempt is made in this Method to quantify the effects of these factors on K_3, users are expected to determine qualitatively whether or not nitrification is likely to occur in the subject stream. If so, users must determine whether or not conditions are optimal for nitrification. For example, several researchers have shown that a pH in the range of 8.4 to 8.6 is optimal for nitrification, with a rapid decrease in nitrification outside the range of 7.0 to 9.0. Because most State water quality standards require a pH in the range of between 6.5 or 7.0 and 9.0, it is unlikely that the pH factor greatly influences the occurrence of nitrification. However, consistent pH observations in the range of 8.4 to 8.6 indicate that this factor is conducive to maximum nitrification. Nitrification is also a function of available benthic surface area for nitrifying organisms to attach themselves. For example, if a stream bottom is completely devoid of surfaces (such as rocks, etc.) on which nitrifiers can attach themselves, it is likely that nitrification will not be a significant factor in the DO analysis. Conversely, a shallow stream with a rocky bottom is likely to have a high nitrification rate.

A site inspection will indicate the likelihood of nitrification, and should be conducted. Based on the observations which are made during the inspection, the user must estimate the applicable NBOD rate value. In view of the tenuous nature of this rate selection procedure, particular care should be taken in evaluating the effects through sensitivity analyses. Another point to consider is the outcome of the ammonia toxicity analysis. If it was determined previously that ammonia removal is required on the basis of ammonia toxicity considerations, then the role of K_3 in the overall DO analysis becomes somewhat less critical. On the other hand, if the ammonia toxicity analysis does not clearly indicate the need for ammonia removal, then the decision to provide nitrification at the treatment plant will hinge solely on the DO analysis. This makes the determination of the K_3 value much more critical. The water quality analyst must then carefully assess stream conditions, and assign reasonable rate coefficients, accordingly.

Based on limited observations of NBOD decay rate (2; also see Exhibit 3), it appears that, where instream nitrification is found to occur, most K_3 values range between about 0.1 and 0.6. <u>In the</u>

absence of applicable site-specific data, a K_3 value within the range of 0 to 0.6 is to be selected. This selection should be based in part on a site inspection (as stated above), and on any appropriate available transfer data. Generally, a K_3 value of around zero should be selected only if strong evidence suggests a lack of instream nitrification occurring under the projected conditions. Otherwise, the K_3 values which are selected might be roughly 0.2-0.3 for deeper streams with a primarily sediment bottom, around 0.4 for shallower streams with a moderately rocky bottom, and about 0.6 for shallow, rocky streams. The user is strongly encouraged to collect site-specific data, and data from similar sites, whenever possible to support the selection of the NBOD decay rate.

(d) Sediment (Benthic) Oxygen Demand (S). Sediment oxygen is a factor that is often significant in the DO analysis. For the types of situations applicable to this Method, Thomann (19) suggests that the following benthic demand rates be used when simulating stream DO response to various treatment levels:

	S(gm/m^2/day of O_2 at 20°C)	
Treatment Level	Vicinity of Outfall	Downstream of Outfall
Poor Secondary Trt	2 - 4	0.5 - 1
Secondary Trt	1 - 2	0.3 - 0.7
Greater than Secondary	0.2 - 0.5	0.1 - 0.2*

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*(Even with high levels of treatment at the point source, there will usually be at least a minimal benthic demand present, e.g., due to "background" or other sources).

For the purpose of this analysis, the benthic demands should be considered to be at a minimum under future conditions, unless site-specific circumstances indicate a continued presence of substantial benthic deposits in the future (e.g., from nonpoint sources). The applicable rates suggested above should be used in the analysis, unless site inspections indicate that higher or lower values should be used.

When selecting the benthic demand rate for future conditions, consideration should be given to the fact that there might be continued other sources of benthic demand, such as from nonpoint sources. Therefore, a site inspection should be conducted to determine the characteristics of the stream bottom and the areal extent and magnitude of the benthic demand, and to reveal any possible continuing benthic demand problems.

(e) Temperature Corrections of Reaction Rates. Temperature effects on the various reaction rates can be approximated by the following equations:

- for the Sediment Oxygen Demand Rate:

$$(SB)_T = (SB)_{20}(1.065)^{T-20} \qquad \text{(Eq. 9)}$$

where $(SB)_T$ = adjusted benthic demand rate for specified stream temperature,

$(SB)_{20}$ = selected benthic demand rate (for stream temperature of 20°C),

T = specified stream temperature (°C).

- <u>for K_1, K_2, and K_3</u>: $K_T = K_{20}\ \theta^{T-20}$ (Eq. 10)

where K_T = adjusted K-rate for specified stream temperature.

K_{20} = selected K-rate (for stream temperature of 20°C).

T = specified stream temperature (°C).

θ = 1.047 for K_1.

= 1.024 for K_2.

= 1.08 for K_3.

(5) <u>Initial Deficit (D_o) Determination</u>. When performing a DO analysis, the initial dissolved oxygen deficit (D_o) must be known (see Equation 3). In order to calculate D_o, the resultant mixed DO concentration at the discharge point must first be calculated. This can be accomplished by performing the following mass balance:

$$DO_o = [DO_u Q_u + DO_D Q_D]/[Q_u + Q_D] \qquad \text{(Eq. 11)}$$

where DO_o = mixed instream dissolved oxygen concentration at discharge point (mg/l).

DO_u = upstream dissolved oxygen concentration (mg/l).

DO_D = POTW discharge dissolved oxygen concentration (mg/l).

Q_u = upstream design low flow.

Q_D = POTW design discharge flow rate.

Then the saturation DO (DO_{SAT}) concentration must be determined using the resultant water temperature downstream of the discharge

(T_R) which was calculated using Equation 2 (in the "Ammonia Toxicity Analysis" section). This can be determined from Exhibit 2. By using DO_o and DO_{SAT}, the initial mixed dissolved oxygen deficit (D_o) can be calculated as follows:

$$D_o = DO_{SAT} - DO_o \qquad \text{(Eq. 12)}$$

(6) Conversion from $CBOD_u$ to $CBOD_5$. Ratios of $CBOD_u/CBOD_5$ discussed herein are based on a nitrification-inhibited test. The ratio is a function of the level of treatment and the associated degradability of the waste. Thus higher ratios are expected and have been observed for higher levels of treatment since the CBOD remaining in more highly treated effluents degrades more slowly than that in less treated wastewaters. It is recommended that the permit BOD effluent limits which are finally selected after the water quality analysis be written as a carbonaceous and not a total BOD_5; the use of $CBOD_5$ effluent limits and, correspondingly, a carbonaceous (inhibited) BOD test when monitoring the effluent can help avoid potential data inaccuracies that can result from nitrification occuring in the bottle during an uninhibited BOD test due to the presence of sufficient nitrifiers in the test bottle. Data submitted by Region V (see Exhibit 4) indicate that the ratio of $BCOD_u$ to $CBOD_5$ should be about 3.0. Other limited data presented in the literature suggest that this ratio is about 1.5 to 2.0. It is suspected that many of the lower ratio values were established using data from older, less efficient treatment plants, and that the Region V data is generally from newer, more efficient plants.

When evaluating secondary treatment discharges, water quality analysts should use a ratio in the range of 1.5 to 2.0, unless applicable long-term CBOD tests indicate some other value. The value or 1.5 should generally be applied to "poorer" secondary plants, and 2.0 should probably be applied to the "better", more efficient secondary treatment plants. Analyses of very limited sewage treatment plant effluent BOD (total and carbonaceous) 5-day, long-term (ultimate), and time series data indicate that a $CBOD_u$ to $CBOD_5$ conversion ratio of about 2.3 should be used for nitrification facilities. Until additional treatment plant effluent data can be collected and analyzed to further refine this ratio, a factor of 2.3 should be used for nitrification and higher-level treatment facilities. It must be emphasized that this value is presently based on a very limited set of data, and that additional treatment plant effluent data is needed to gain greater confidence in the suggested value. All EPA Regions, the States, and others are strongly urged to voluntarily participate in a nationwide data gathering effort so that more accurate ratios can be developed.

Such an effort would not be very resource intensive, and the results would be extremely useful. Care should be taken to ensure that only the carbonaceous demand is measured. This data collection effort should include information on the type of treatment and type of influent, and the sampling should, to the extent possible, only be performed on sanitary wastewaters that are unchlorinated. This data should be submitted to EPA Headquarters for compilation and analysis.

It is recommended that, whenever possible, existing plant effluent data and/or pilot plant data should be collected to assist in the selection of an appropriate conversion ratio. Caution should be exercised, however, when using data from an existing plant that has a level of treatment that is significantly lower than that which is proposed. Such data should not be blindly applied when selecting the appropriate conversion ratio; it should merely be used as a guide. A sensitivity analysis of the co[redacted]ersion ratio and its implications on the final treatment decision to be ma[redacted]can help the water quality analyst determine the relative importance of [redacted]gathering such additional data.

It is also recommended that the BOD_5 effluent limits in the tr[redacted] plant's permit be reviewed after the new facility is on-line to hel[redacted] that the correct $CBOD_u$ to $CBOD_5$ ratio was applied to the model outp[redacted]is can be accomplished by collecting and analyzing appropriate plant e[redacted]t CBOD data after the new treatment facility is on-line.

(7) Post-Aeration. Post-aeration of the effluent to a DO concentration of 7 mg/l should always be evaluated as an alternative[redacted] to higher levels of treatment, unless there is a site-specific constraint that precludes the use of post-aeration equipment. This [redacted] technique can be particularly useful in cases where dilution is low [redacted] and reaeration rates are also low.

(8) Nonpoint Sources. In some cases, nonpoint sources may preclude attainment of dissolved oxygen water quality objectives even with stringent advanced treatment. For example, streams with extensive wetlands may contribute low DO water in sufficient quantities to cause standards violations, and agricultural or urban runoff in the vicinity of a plant discharge may also nullify the benefits of advanced treatment. In cases such as these, nonpoint

source control tradeoffs must be considered before advanced treatment is chosen. Site-specific evaluations should be made to identify possible nonpoint source problems.

D. Sensitivity Analysis

The sensitivity of computed stream responses to changes in estimated input variables must be determined before a final decision of treatment levels is made. A sensitivity analysis combined with judgement is essential to help establish greater confidence in the results that are obtained.

The sensitivity of computed (predicted) instream responses to the various input values should be determined by repeating the analyses described above with changes (increases and decreases) in the input variables. The following steps should be followed:

1) Initially, three sets of calculations should be made to reflect a "worst," "best," and "average" case for each alternative treatment level. This can be accomplished by using model input values that represent, respectively, the "worst" and "best" ends of their sensitivity ranges and the values actually selected for the model. The outputs of these computations should be plotted as DO profiles. If all three cases indicate a violation of the water quality standard with the given level of treatment, then the next level of treatment is needed, and no further justification is necessary. If all three cases do not indicate a violation, then the next step must be taken.

2) Next, each input variable to each equation above should be individually increased and decreased, so that the magnitude of the differences in predicted instream responses can be assessed. Each input variable (including rate coefficients, travel time, physical characteristics, etc.) should be varied over a range of values that reflects the uncertainty in the particular variable. If direct measurements of certain input variables are made, then the uncertainty in the variable would tend to be relatively small and, therefore, the range to be used in the sensitivity analysis would generally be relatively small. Very close scrutiny should be given to those input variables which have no site-specific or transferable data to support their having been selected. If rates (or rate formulations) other than those suggested in the above analyses are used, then the sensitivity analysis should be used as part of the justification for the alternative rates (or formulations).

The results of these sensitivity analyses (in step 2) should be reviewed within the context of the effluent quality expected for various treatment levels. Therefore, if the effluent requirements determined using the range of inputs for each variable fall within the expected effluent quality from a single treatment level (e.g., AST or AWT), then additional analyses would generally not be required for that variable since the need for that level of treatment is obvious. However, if the required treatment level is heavily sensitive to, and dependent on, the selection of an input value(s) <u>especially</u> where existing data are inadequate to characterize the variable(s), <u>then a</u>

sufficient amount of additional data shall be obtained to more accurately define that model variable(s) (thus increasing the confidence in that variable) so that the selection of the treatment alternative can be clarified. For even further confirmation of the selected effluent limitations, the sensitivity analysis can also be rerun for the individual input variables at a less stringent level of treatment and the results analyzed to determine if the desired water quality objectives could possibly be met at that lesser treatment level within the range of individual inputs being utilized. It must be emphasized that the use of sound professional judgment is essential when evaluating the confidence in the model input variables used and in the modeling results obtained.

To further assist in evaluating the results of the sensitivity analyses, the incremental present worth cost (construction and O&M) of the proposed improved treatment process may also be considered when deciding the necessary level of treatment. For example, if oversizing the clarifiers, providing additional aeration and clarifiers for nitrification, or seasonal chemical addition could provide the level of treatment in question, such treatment could be partly justified based on best judgment due to the relatively low cost of such additional treatment. On the other hand, filters (as an add-on to nitrification), due to their high incremental cost, could be justified by this Simplified Method only if the results of these sensitivity analyses indicate sufficient confidence in the results. Otherwise, additional data (including for calibrating and verifying the model) would be required.

Generally, the variables to be analyzed in the sensitivity analysis should include those listed below, and should generally be varied by sensitivity ranges in the order of those which are suggested below (especially if little or no data is available to support the variable's selection):

- *CBOD rate* - vary by about ±50 to 100% (and appropriate increments in between), depending on the uncertainty in the estimated value.
- *NBOD rate* - vary by about ±25 to 75% or more (and appropriate increments in between), depending on the uncertainty in the estimated value.
- *Reaeration rate* - vary by about ±25% to ±100% or more (depending on the uncertainty in the estimated value), and by intermediate increments. An appropriate sensitivity analysis should also be performed on the variables used in the respective reaeration equations (i.e., velocity, and slope or depth).
- *Benthic demand* - generally should use the suggested ranges presented in Section III(C)(4)(d).
- *Temperature, pH* - use ranges appropriate for the situation.

E. *Results*

1) *Permit Conditions*. After determining the final effluent limitations necessary for the maintenance of water quality standards, these limitations should be entered into the NPDES permit. Municipal effluent limitations are often specified as 30-day and 7-day average values for BOD_5, ammonia-N, and suspended solids. For streams with zero flow at the critical conditions, the results of the DO analysis shall be used as *7-day* average effluent limits rather than 30-day averages, since these small streams are often very reactive to variable waste inputs. For streams with nonzero flow at the critical

conditions, the currently adopted and applicable State or EPA Region procedures for applying modeling results to POTW discharge permit effluent limitations shall be used.

Technical analyses are being conducted which study the effects o effluent concentration and streamflow variability, different dilution ratios, and the use of alternative averaging period schemes on receiv water quality. Preliminary results indicate that effluent and stream variability and, to a lesser extent, differing dilutions are critical factors which often greatly affect the frequency of severe water qual violations.

A site-specific analysis that considers the effects of the indiv stream's flow variability, available dilution, and the selected treat process' effluent concentration variability on the stream's water qual (including the frequency and severity of water quality violations) sho be performed in each case to determine the appropriate averaging perio Technical guidance on performing such analyses is being developed by EPA Office of Water Regulations and Standards to aid in the selection appropriate averaging periods.

Based on currently available data for treatment plant performance, full nitrification treatment is a relatively stable process during the summer months. When considered together with streamflow variability a dilution, this treatment process should normally preclude frequent hig levels of water quality violations when the stream flow is at low flow conditions and the stream's flow characteristics are not highly variabl

It appears that in most cases, especially where the stream's flow variability is not extremely high, fluctuations in the effluent quality of full nitrification facilities designed to achieve 30-day average permit limitations are not likely to have a significant impact on the aquatic habitat due to increased loadings or decreased dilution. Actual impacts on the aquatic habitat or designated uses will be determined when site-specific analyses are conducted.

The CBOD and NBOD outputs from the DO analysis should be converted to BOD_5 and ammonia-N NPDES permit limitations using the following relationships (established earlier):

(a) $BOD_5 = CBOD/r$

(where r = the UCBOD:BOD_5 ratio, for the appropriate level of treatment, selected in section III(C)(6).)

(b) NH_3-N = NBOD/4.57

2) Seasonal Effluent Limitations. The effects of variations in temperature and flow should be evaluated to determine whether or not operating costs can be reduced through seasonal relaxation of effluent limitations. For example, it is conceivable that in winter, higher flows and lower temperatures would allow for a relaxation of BOD and ammonia limitations from a toxicity and DO standpoint.

EXAMPLE PROBLEM

(An example problem will be prepared and provided to the users of this Method in the near future.)

APPENDIX A1

METHOD FOR DETERMINING EXPECTED DOWNSTREAM pH

Calculating the pH of the stream after mixing of upstream flow with wastewater discharge is straightforward - provided there is some minimal water quality information available on each.

It can be done with information on different combinations of alkalinity, acidity and pH. The most direct method, and simplest to present for a simplified methodology, requires information on pH and alkalinity. If no pH or alkalinity data is presently available for the POTW and/or for upstream, then a short-term program of collecting the needed pH and/or alkalinity data should be initiated. Assuming generally average conditions exist at the POTW and upstream during the data collection program, then a fairly accurate determination of their average pH and alkalinity conditions can be made in several days or a few weeks. The tests for pH and alkalinity are easy and inexpensive:

- pH - is easily (and commonly) measured by pH meter.
- Alkalinity - is also commonly measured. Standard Methods specifies titration with strong acid (0.02 N H_2SO_4) - in which case alkalinity is reported as mg/l as Ca CO_3.

  The ionic forms comprising alkalinity (HCO_3^-, $CO_3^=$ and OH^- be calculated from the relative amounts of acid required to reach the two end points (Phenolphthalein at pH 8.3, and Methyl Orange at about pH 4.5). For natural waters between pH 4.5 and 8.3 (essentially all we need be concerned with for this exercise), there will be no phenolphthalein end point -- and <u>all</u> of the alkalinity will be present as bicarbonate ion (HCO_3^-).

Note that the procedure presented, and the chart used, will apply for any situation. The instructions for its use are much simpler to present if it is assumed that the only waters being dealt with will be in the pH 4.5 to 8.3 range (which will usually be the case).

There are a number of techniques, methods, etc., for calculating pH. The one presented here seems to be best suited for these purposes. The chart is taken from "Aquatic Chemistry" by Stumm & Morgan (Wiley Interscience, 1970), and the approach is a portion of the overall approach they describe - which for simplicity is limited to the usual case which was selected.

The calculation is illustrated by the following example:

- In a natural water system (or wastewater) with a pH in the range of 4.5 to 8.3, the alkalinity measured is all bicarbonate ion (HCO_3^-) reported as mg/l as $CaCO_3$. The Total Inorganic Carbon (C_T) will consist of both HCO_3^- and soluble CO_2 which coexist in water in this pH range. (NOTE: The Standard Methods test for acidity would measure soluble CO_2.)

- The chart requires that alkalinity be reported in terms of milliequivalents/liter (meq/l). The conversion is as follows:

  Alkalinity-mg/l as $CaCO_3$ X 1/50 = Alk.-meq/l

- Total Inorganic Carbon (C_T) must be in terms of millimoles/liter (mM/l). However, if we are working with pH and alkalinity, we need not worry about this conversion, nor about determining the acidity or CO_2 concentration.

- In a situation like the one being addressed, both alkalinity and C_T are conservative (pH is not). Thus, when two waters with different concentrations of alkalinity and C_T are mixed - the final concentration in the blend can be determined by simple mass balance.

- The steps are:

  1. From alkalinity and pH, determine C_T for stream and waste.

  2. Calculate concentration of alkalinity and C_T in blend.

  3. Determine pH of blend from blend alkalinity and C_T.

- An example problem and a calculation sheet are given in Exhibit 5. A blank calculation sheet, with chart, is also attached (Exhibit 6).

EXHIBIT 6. EXAMPLE CALCULATION.

1. GIVEN

	POTW Discharge	Upstream Flow
Flow =	2.49 MGD = 3.86 CFS	3.0 CFS
pH =	7.2	7.6
ALK =	150 mg/l = 3.0 meq/l as $CaCo_3$	50 mg/l = 1.0 mg/l as $CaCo_3$

2. DETERMINE C_T FROM CHART

POTW Discharge	Upstream Flow
ALK = 3.0 pH = 7.2 → C_T = 3.56	ALK = 1.0 pH = 7.6 → C_T = 1.1

3. CALCULATE BLENDED ALK AND C_T

B = Blend S = Stream D = Discharge

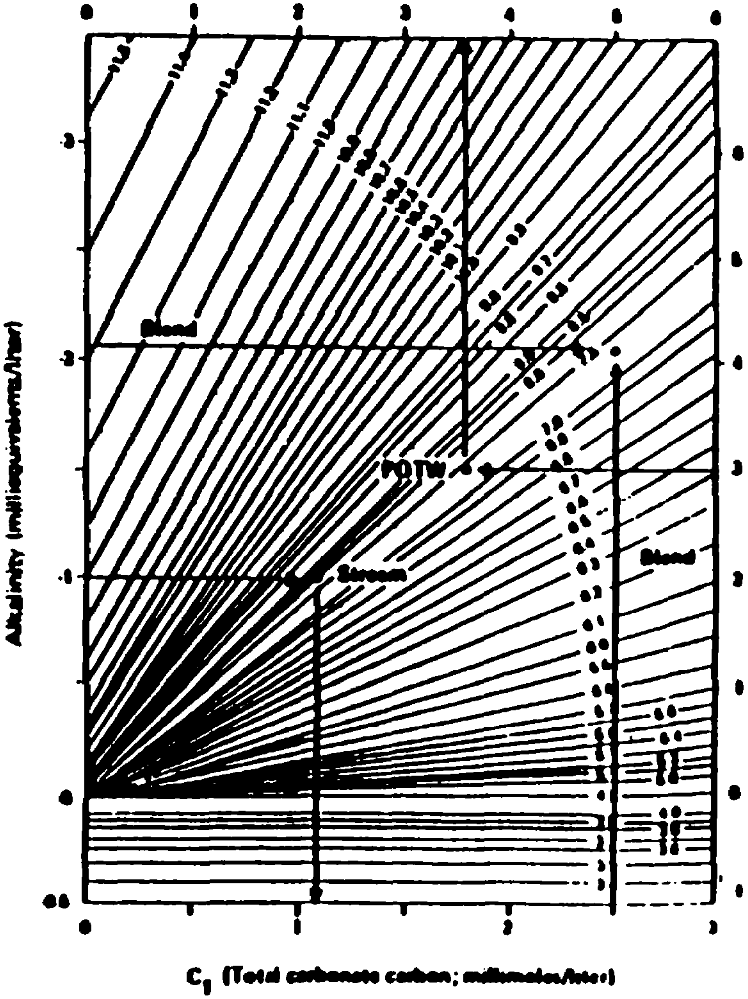
$$C_{T} \text{ or } ALK_B = \frac{(\overset{C_{T_S}}{ALK_S} \cdot Q_S) + (\overset{C_{T_D}}{ALK_D} \cdot Q_D)}{(Q_S + Q_D)}$$

$$ALK_B = \frac{(1.0 \cdot 3.0) + (3.0 \cdot 3.86)}{(3.0 + 3.86)} = \frac{14.58}{6.86} = 2.12$$

$$C_{T_B} = \frac{(1.1 \cdot 3.0) + (3.56 \cdot 3.86)}{(3.0 + 3.86)} = \frac{17.04}{6.86} = 2.48$$

4. DETERMINE pH OF BLEND FROM CHART

ALK = 2.12
C_T = 2.48 → pH = 7.2



Note: Doubled scales apply if pH lines goes through origin

EXHIBIT 8. BLANK CALCULATION SHEET.

1. GIVEN

	POTW Discharge	Upstream Flow
Flow =		
pH =		
ALK =		

2. DETERMINE C_T FROM CHART

POTW Discharge		Upstream Flow	
ALK =	C_T =	ALK =	
pH =		pH =	

3. CALCULATE BLENDED ALK AND C_T

B = Blend S = Stream D = Discharge

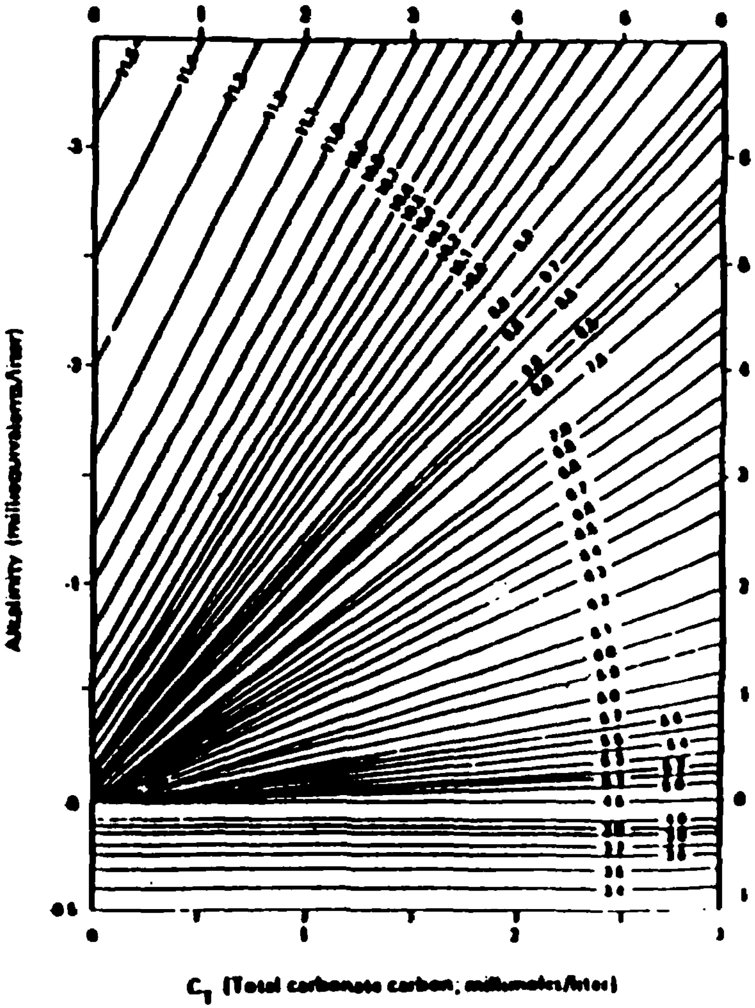
$$C_T \text{ or } ALK_B = \frac{\overset{C_{T_S}}{(ALK_S \cdot Q_S)} + \overset{C_{T_D}}{(ALK_D \cdot Q_D)}}{(Q_S + Q_D)}$$

$$ALK_B = \frac{(\qquad) + (\qquad)}{} = \underline{\qquad} =$$

$$C_{T_B} = \frac{(\qquad) + (\qquad)}{} = \underline{\qquad} =$$

4. DETERMINE pH OF BLEND FROM CHART

ALK =
C_T =
pH =



APPENDIX A2

The results of new data supplied by EPA EMSL-Cincinnati show that BOD_5 levels in nitrification system effluents range from BOD_5 5 - 8 mg/L. Tests run on typical nitrified effluents using the Standard Methods BOD_5 test consistently exhibited higher than actual carbonaceous BOD_5 due to nitrification occurring during the CBOD test. The reason for the high BOD_5 result is that aqueous solutions containing ammonium salts are used in the standard BOD_5 test. These ammonium salts in the presence of nitrifying organisms in the nitrified effluent create an additional apparent BOD_5 demand in the effluent. For this reason, data from plants with nitrification cannot be used to predict treatment capabilities unless nitrification inhibitors are used.

The Agency has proposed (December 3, 1979 Federal Register) an inhibitory BOD_5 test for nitrified effluents. Early work at the Washington, D.C.—Pilot Plant used allyl thiourea as the inhibitory chemical. These tests show that the BOD_5 measured for uninhibited nitrified effluents is two to three times greater than for inhibited effluents (See Exhibit 7).

It is advised that all treatment plants with nitrification use the inhibiting chemical and report the BOD_5 values that are determined from this method to expand the data base of plant operational capabilities.

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UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

JUN 25 1982

MEMORANDUM

SUBJECT: Addendum to Simplified Mathematical Modeling Methodology

FROM: Steven Schatzow, Director
Office of Water Regulations and Standards (WH-551)
William A Whittington
Henry L. Longest II, Director
for Office of Water Program Operations (WH-546)

TO: Regional Water Division Directors
Regional Environmental Services Division Directors
Regional Wasteload Allocation Coordinators

Attached, for national use, is the final version of the addendum to the "Simplified Analytical Method" which provides several technical revisions or clarifications to the national guidance document dated September 26, 1980. The guidance provided in this addendum will supersede that in the respective sections of the original document.

As you will note, the addendum provides modifications related to the following issues:

- application of the guidance.
- performing ammonia toxicity analyses.
- calculation of stream travel time to reach the point of critical dissolved oxygen deficit.
- temperature correction factor for K_3.
- dissolved oxygen saturation concentration determination.
- conversion ratio of $CBOD_u$ to $CBOD_5$.
- permit conditions.

These changes are based on additional information and analyses which have become available.

If you have any questions or comments or desire additional information, please contact Tim J. Stuart, Chief, Monitoring Branch, Monitoring and Data Support Division (WH-553) on (FTS) 426-7766.

Attachment

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

ADDENDUM
TO
"Simplified Analytical Method for Determining NPDES Effluent Limitations for POTW's Discharging into Low-Flow Streams: National Guidance"

I. INTRODUCTION

This addendum provides several technical revisions to the "Simplified Analytical Method" that clarify, correct, or modify (based on additional information and analyses which have become available) certain sections of the original national guidance document dated September 26, 1980. The guidance provided herein supercedes that in the respective sections of the original document.

Users of this method (as well as others) are strongly encouraged to develop additional data and other information on reaction rates and other factors applicable to this method, and to submit this additional data and other information, along with suggested improvements for the method, to:

Chief, Wasteload Allocations Section
Monitoring Branch
MOSD/OWRS (WH-553)
U.S. Environmental Protection Agency
401 M Street, S.W.
Washington, D.C. 20460

(Telephone: (202)-426-7778)

This additional data and other information is needed so that appropriate additional improvements to the method can be made periodically. Users may also, when appropriate, make any modifications to this method that will allow the method to more accurately represent regional or local conditions. Any changes should be supported with an adequate technical justification, including sufficient applicable data.

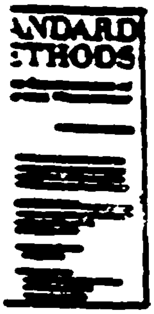
Where the results of the water quality analysis indicates the need for treatment beyond secondary, State users of this method are encouraged to coordinate the modeling analysis with a review of the environmental benefits and costs of the receiving water's applicable water quality standards. Such assessments should be conducted in accordance with the revised water quality standards regulations and EPA guidance, when published.

A site-specific analysis that considers the effects of the individual stream's flow variability, available dilution, and the selected treatment process' effluent concentration variability on the stream's water quality (including the frequency and severity of water quality violations) should be performed in each case to determine the appropriate averaging period. Technical guidance on performing such analyses is being developed and will soon be released (anticipated date is about July 1982) by the EPA Office of Water Regulations and Standards to aid in the selection of appropriate averaging periods.

Based on currently available data for treatment plant performance, full nitrification treatment is a relatively stable process during the summer months. When considered together with streamflow variability and dilution, this treatment process should normally preclude frequent high levels of water quality violations when the stream flow is at low flow conditions and the stream's flow characteristics are not highly variable. It appears that in most cases, especially where the stream's flow variability is not extremely high, fluctuations in the effluent quality of full nitrification facilities designed to achieve 30-day average permit limitations are not likely to have a significant impact on the aquatic habitat due to increased loadings or decreased dilution. Actual impacts on the aquatic habitat or designated uses will be determined when site-specific analyses are conducted.

Solubility of Oxygen in Water Exposed to Water-Saturated Air (mg/l) **

Temperature C	Chloride Concentration in Water mg/L 0	5,000	10,000	15,000	20,000
0	14.60	13.72	12.90	12.13	11.41
1	14.19	13.35	12.56	11.81	11.11
2	13.81	12.99	12.23	11.51	10.83
3	13.44	12.65	11.91	11.22	10.56
4	13.09	12.33	11.61	10.94	10.30
5	12.75	12.02	11.32	10.67	10.05
6	12.43	11.72	11.05	10.41	9.82
7	12.12	11.43	10.78	10.17	9.59
8	11.83	11.16	10.53	9.93	9.37
9	11.55	10.90	10.29	9.71	9.16
10	11.27	10.65	10.05	9.49	8.96
11	11.01	10.40	9.83	9.28	8.77
12	10.76	10.17	9.61	9.08	8.58
13	10.52	9.95	9.41	8.89	8.41
14	10.29	9.73	9.21	8.71	8.24
15	10.07	9.53	9.01	8.53	8.07
16	9.85	9.33	8.83	8.36	7.91
17	9.65	9.14	8.65	8.19	7.76
18	9.46	8.95	8.48	8.03	7.61
19	9.26	8.77	8.32	7.88	7.47
20	9.07	8.60	8.16	7.73	7.33
21	8.90	8.44	8.00	7.59	7.20
22	8.72	8.28	7.85	7.45	7.07
23	8.56	8.12	7.71	7.32	6.95
24	8.40	7.97	7.57	7.19	6.83
25	8.24	7.83	7.44	7.06	6.71
26	8.09	7.69	7.31	6.94	6.60
27	7.95	7.55	7.18	6.83	6.49
28	7.81	7.42	7.06	6.71	6.38
29	7.67	7.30	6.94	6.60	6.28
30	7.54	7.17	6.83	6.49	6.18
31	7.41	7.05	6.71	6.39	6.08
32	7.28	6.94	6.61	6.29	5.99
33	7.16	6.82	6.50	6.19	5.90
34	7.05	6.71	6.40	6.10	5.81
35	6.93	6.61	6.30	6.01	5.72
36	6.82	6.51	6.20	5.92	5.64
37	6.71	6.40	6.11	5.83	5.56
38	6.61	6.31	6.02	5.74	5.48
39	6.51	6.21	5.93	5.66	5.40
40	6.41	6.12	5.84	5.58	5.33
41	6.31	6.03	5.76	5.50	5.25
42	6.22	5.94	5.68	5.42	5.18
43	6.13	5.85	5.60	5.35	5.11
44	6.04	5.77	5.52	5.27	5.04
45	5.95	5.69	5.44	5.20	4.98
46	5.86	5.61	5.37	5.13	4.91
47	5.78	5.53	5.29	5.06	4.85
48	5.70	5.45	5.22	5.00	4.78
49	5.62	5.38	5.15	4.93	4.72
50	5.54	5.31	5.08	4.87	4.66



** At a total pressure of 101.3 kPa. Under any other barometric pressure, P, obta
S′ (mg/L) from the corresponding value in the table by the equation:

$$S' = S\frac{P - p}{760 - p}$$

in which S is the solubility at 101.3 kPa and p is the pressure (mm) of saturated wa
water temperature. For elevations less than 1,000 m and temperatures below 25 C
equation then becomes:

$$S' = S\frac{P}{760} = S\frac{P}{29.92}$$

Dry air is assumed to contain 20.90% oxygen. (Calculations made by Whipple and
J. Amer. Chem. Soc. 33:362.)

E. Initial Deficit Determination (page 18 and Exhibit 3).

The dissolved oxygen (DO) saturation concentrations table presently included as Exhibit 3 in the method is based on Streeter. Additional evidence which has been accumulated indicates that these saturation DO values are not entirely accurate. The new 15th edition of "Standard Methods" contains an updated table of saturation DO concentrations which represents the most up-to-date and accurate saturation DO concentration data currently available. This table is reproduced in Attachment A of this addendum. It should be noted that the tabulated values for DO saturation are for distilled water at standard pressure (sea level). These values should be corrected for altitude and dissolved solids levels using the formula at the bottom of the table.

It is important for all wasteload allocation and other water quality modeling efforts to be consistent in the use of saturation DO values. Use of the attached table is recommended for this method (and other modeling efforts) because it represents the best information currently available.

F. Conversion from CBOD$_u$ to BOD$_5$ (page 18).

(a) The method presently discusses conversion ratios of CBOD$_u$ to BOD$_5$ for various levels of treatment; however, it is unclear whether the method is referencing carbonaceous (based on a nitrification-inhibited test) or total (based on an uninhibited test) BOD$_5$. Therefore, it should be clarified that the ratios being discussed in the guidance are for CBOD$_u$ to CBOD$_5$ (based on a nitrification-inhibited test). (It is recommended that the permit BOD effluent limits which are finally selected after the water quality analysis be written as a carbonaceous and not a total BOD$_5$; the use of CBOD$_5$ effluent limits and, correspondingly, a carbonaceous (inhibited) BOD test when monitoring the effluent can help avoid potential data inaccuracies that can result from nitrification occuring in the bottle during an uninhibited BOD test due to the presence of sufficient nitrifiers in the test bottle.)

(b) The CBOD$_u$ to CBOD$_5$ ratio is a function of the level of treatment and the associated degradability of the waste. Thus higher ratios are expected and have been observed for higher levels of treatment since the CBOD remaining in more highly treated effluents degrades more slowly than that in less treated wastewaters.

The method presently suggests that, for plants with treatment levels greater than secondary, a ratio of 2.5 to 3.0 should be used for determining permit limitations. The guidance implied, though did not specifically state, that 2.5 should be used for nitrification levels of treatment and a ratio of approximately 3.0 should be used for treatment levels greater than nitrification.

Additional analyses of very limited sewage treatment plant effluent BOD (total and carbonaceous) 5-day, long-term (ultimate), and time series data indicated that a $CBOD_u$ to $CBOD_5$ conversion ratio of about 2.3 should be used for nitrification facilities. Until additional treatment plant effluent data can be collected and analyzed to further refine this ratio, a factor of 2.3 should be used for nitrification and higher-level treatment facilities. It must be emphasized that this value is presently based on a very limited set of data, and that additional treatment plant effluent data is needed to gain greater confidence in the suggested value. All EPA Regions, the States, and others are again strongly urged to voluntarily participate in a nationwide data gathering effort so that more accurate ratios can be developed.

It is recommended that, whenever possible, existing plant effluent data and/or pilot plant data should be collected to assist in the selection of an appropriate conversion ratio. Caution should be exercised, however, when using data from an existing plant that has a level of treatment that is significantly lower than that which is proposed. Such data should not be blindly applied when selecting the appropriate conversion ratio; it should merely be used as a guide. A sensitivity analysis of the conversion ratio and its implications on the final treatment decision to be made can help the water quality analyst determine the relative importance of gathering such additional data.

It is also recommended that the BOD_5 effluent limits in the treatment plant's permit be reviewed after the new facility is on-line to help ensure that the correct $CBOD_u$ to $CBOD_5$ ratio was applied to the model output. This can be accomplished by collecting and analyzing appropriate plant effluent CBOD data after the new treatment facility is on-line.

6. Permit Conditions (page 22).

The guidance presently requires that, for streams with zero flow at the critical conditions, the results of the modeling analysis should be used as 7-day average effluent limits, and, for streams with nonzero flow at the critical conditions, the currently adopted and applicable State or EPA Regional procedures for applying modeling results should be used. The original guidance also indicates that this issue of applying modeling results to effluent limitations is being analyzed in support of the development of forthcoming policy guidance on wasteload allocations/total maximum daily loads (WLA's/TMDL's).

Technical analyses are being conducted which study the effects of effluent concentration and streamflow variability, different dilution ratios, and the use of alternative averaging period schemes on receiving water quality. Preliminary results indicate that effluent and streamflow variability and, to a lesser extent, differing dilutions are critical factors which often greatly affect the frequency of severe water quality violations.

II. TECHNICAL REVISIONS

A. Applications and Constraints (page 3).

As stated in the original guidance document, the analytical techniques which are used in water quality modeling should be the simplest possible that will still allow the water quality manager to make confident and defensible water pollution control decisions. In many cases, where relatively simple conditions exist, simplified modeling efforts that have minimal manpower and data requirements are often adequate to make such decisions. Use of simplified efforts, when appropriate, can result in both substantial savings in State and EPA resources and cost-effective and technically sound effluent limitations to be achieved.

Recent experience and analyses indicate that this simplified method, when followed properly and with little or no site-specific data being employed, should normally result in both technically sound water quality justifications being developed for nitrification levels of treatment and substantial savings in State and EPA resources. However, it has also been noted that this simplified analysis alone (i.e., without any site-specific data) usually cannot provide the confidence needed to adequately justify permit limits more stringent than about 10 mg/l $CBOD_5$ and 1.5 mg/l NH_3-N, including relatively costly filtration treatment after nitrification. Therefore, this simplified method cannot be used by itself to justify permit limits more stringent than 10 mg/l $CBOD_5$ and 1.5 mg/l NH_3-N (including filtration after nitrification).

Where treatment more stringent than 10 mg/l $CBOD_5$ and 1.5 mg/l NH_3-N (including filtration after nitrification) appears to be needed, appropriate supporting site-specific data should be collected and used in the analysis in order to increase confidence in the variables used in this model, in the the modeling results that are obtained, and, most importantly, in the treatment decision itself. This additional level of analysis should also be accompanied by a rigorous sensitivity analysis (see page 20 of the method). Based on past analyses and construction grant project reviews, it appears that this situation (e.g., the need for treatment beyond nitrification) will seldom be required except in certain cases where small streams with very low assimilative capacities are encountered.

B. Ammonia Toxicity Analysis (page 5).

The original guidance document recommends that, if no un-ionized or total ammonia-N standards are available for use, a criterion of 0.02 mg/l un-ionized ammonia be used for freshwater cold water habitats, or 0.05 mg/l un-ionized ammonia be used for freshwater warm water habitats. Additional research, however, indicates that these criteria may in many cases be more stringent than necessary to protect water quality. It is now recommended that the latest EPA ammonia toxicity criteria, when promulgated, be used in conjunction with the supporting ammonia criteria implementation guidance document. Until the new EPA ammonia toxicity criteria are promulgated, AT facilities proposed solely to prevent ammonia toxicity may be approved only with supporting justifications based on either: (1) site-specific biological data showing that the designated uses cannot be restored without reducing ammonia toxicity, or (2) bioassay data (either from a laboratory or similar site) for indigenous species showing that existing or future ammonia toxicity levels will impair designated use attainment (exposure levels and durations for these tests should be similar to those occurring or anticipated to occur in the receiving water). (Note: After publication of new ammonia toxicity criteria by EPA, Advanced Treatment processes proposed solely to prevent ammonia toxicity may be approved consistant with these criteria and this simplified method.)

C. Dissolved Oxygen Analysis (page 9).

Equation 5 on page 9 of the method is presently written incorrectly. The correct form, which should be used, is:

$$t_c = [1/(K_2-K_{BOD})] \ \ln [(K_2/K_{BOD}) \ (1- [D_o(K_2-K_{BOD})/(K_{BOD}UBOD_o)]])] \qquad \text{(Eq. 5)}$$

D. Temperature Corrections of Reaction Rates (pages 17-18).

The method presently states that the temperature correction coefficient (θ) for K_3 to be used in Equation 10 is 1.10. A more reasonable correction coefficient for K_3, which should be used, is 1.08. This latter value represents an average of the range of correction factors found by different researchers ("Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling", EPA-600/3-78-105).