APPENDIX B

WATER QUALITY ASSESSMENT

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FIGURES

TABLES

INTRODUCTION

This appendix provides detailed guidance for responding to water quality-related questions in the Application Questionnaire. Methods for predicting values of the following water quality variables are presented:

- Suspended solids deposition

- Dissolved oxygen concentration following initial dilution

- Farfield dissolved oxygen depression

- Sediment oxygen demand

- Suspended solids concentration following initial dilution

- Effluent pH after initial dilution

- Light transmittance

- Other water quality variables.

B-I. SUSPENDED SOLIDS DEPOSITION

The applicant must predict the seabed accumulation due to the discharge of suspended solids into the receiving water. Two prediction methods are described in this appendix. The first is a simplified approach for small dischargers only. If this method is applicable, then a small discharger need not perform dissolved oxygen calculations dependent on settled effluent suspended solids accumulations. The second prediction method is applicable for both small and large dischargers.

SMALL DISCHARGER APPROACH

Two types of problems (dissolved oxygen depletion and biological effects) and two types of receiving water environments (open coastal and semi-enclosed bays or estuaries) are considered in the following approach.

Figure B-1 is to be used for open coastal areas that are generally considered well flushed. The dashed line represents combinations of solids mass emission rates and plume heights-of-rise that would result in a steady-state sediment accumulation of 50 g/m^2. Review of data from several open coast discharges has indicated that biological effects are minimal when accumulation rates were estimated to be below this level. Consequently, if the applicant's mass emission rate and height-of-rise fall below this dashed line no further sediment accumulation analyses are needed. Applicants whose charge characteristics fall above the line should conduct a more detailed analysis of sediment accumulation discussed in the following section.

The solid line in Figure B-1 represents a combination of mass emission rates and plume heights-of-rise that were projected to result in sufficient sediment accumulation to cause a 0.2 mg/L oxygen depression. Applicants whose discharge falls below this solid line need not provide any further analysis of sediment accumulation as it relates to dissolved oxygen.
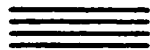
B-2

MASS EMISSION RATE OF SOLIDS, Kg/day

HEIGHT OF RISE, m

⧄⧄⧄⧄⧄ STEADY STATE SEDIMENT ACCUMULATION LESS THAN $50 g/m^2$

≡≡≡ DO DEPRESSION DUE TO STEADY-STATE SEDIMENT
DEMAND > 0.2 mg/l

Reference: Tetra Tech (1982).

Figure B-1. Projected relationships between suspended solid mass emission, plume height-of-rise, sediment accumulation, and dissolved oxygen depression for open coastal areas.

Figure B-2 should be used in a similar manner for discharges to semi-enclosed embayments or estuaries. Because estuaries and semi-enclosed embayments are potentially more sensitive than open coastal areas, the critical sediment accumulation was set at 25 g/m^2.

Methods described in Tetra Tech (1982) were used to determine the mass emission rates and heights-of-rise resulting in the sediment accumulation rates specified above. In order to use these methods, several assumptions were made. A current velocity of 5 cm/sec was assumed for the open coastal sites and a velocity of 2.5 cm/sec was assumed for the semi-enclosed embayments. These velocities are conservative estimates of average current velocities over a 1-yr period. The settling velocity (V_s) distribution used is considered typical of primary or advanced primary effluents and is shown below:

5 percent have $V_s \geq 0.1$ cm/sec
20 percent have $V_s \geq 0.01$ cm/sec
30 percent have $V_s \geq 0.006$ cm/sec
50 percent have $V_s \geq 0.001$ cm/sec

The remaining solids settle so slowly that they are assumed to remain suspended in the water column indefinitely. The effluent is considered to be 80 percent organic and 20 percent inorganic. The above distribution is based on the review of data in Section 301(h) applications and other published data (Myers 1974; Herring and Abati 1978).

The annual suspended solids mass emission rate should be calculated using the average flow rate and an average suspended solids concentration. The plume height-of-rise, determined previously in the initial dilution calculation, or 0.6 times the water depth, whichever is larger, should be used to enter the appropriate figure (Figure B-1 or B-2).

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MASS EMISSION RATE OF SOLIDS, Kg/day (y-axis: 0, 1000, 2000, 3000, 4000)

HEIGHT OF RISE, m (x-axis: 0, 2, 4, 6, 8, 10, 12, 14, 16, 18, 20)

⧄⧄⧄ STEADY STATE SEDIMENT ACCUMULATION LESS THAN $25g/m^2$

≡≡≡ DO DEPRESSION DUE TO STEADY-STATE SEDIMENT DEMAND > 0.2 mg/l
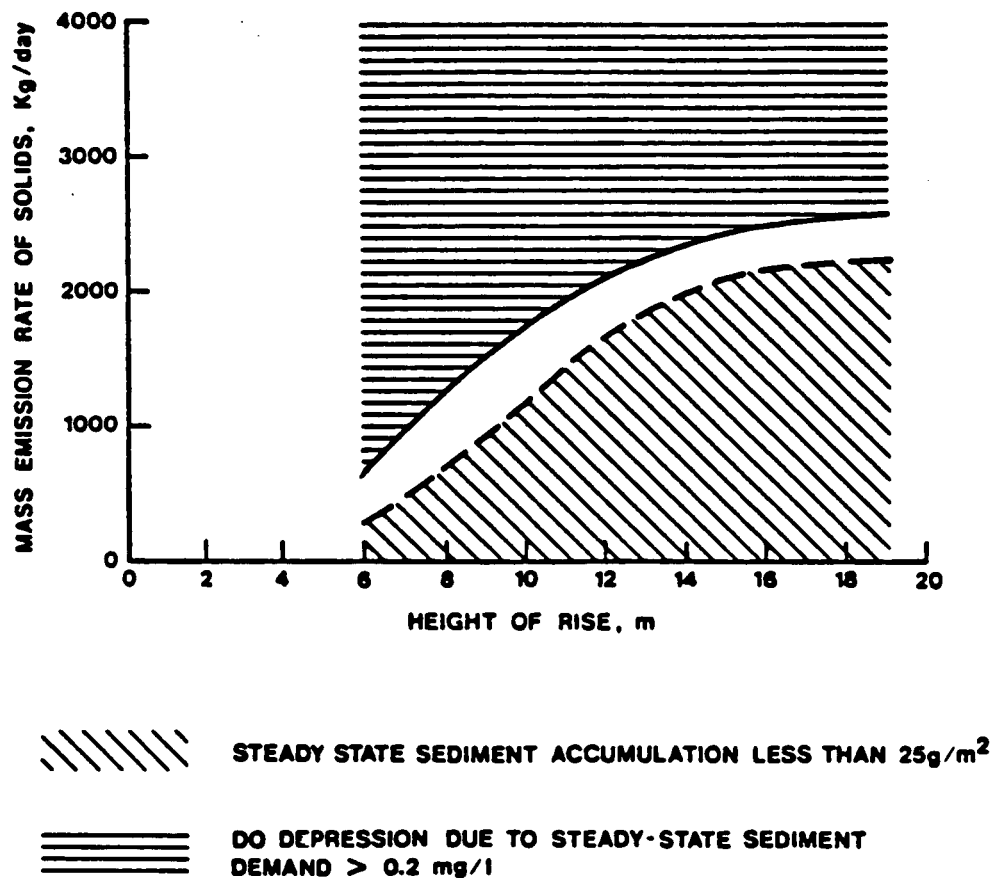
Reference: Tetra Tech (1982).

Figure B-2. Projected relationships between solid mass emission, plume height-of-rise, sediment accumulation, and dissolved oxygen depression for semi-enclosed embayments and estuaries.

LARGE DISCHARGER APPROACH

The approach described here considers the processes of sediment deposition, decay of organic materials, and resuspension. However, the strictly quantitative prediction of seabed accumulation is based only on the processes of deposition and decay. Because resuspension is not evaluated easily using simplified approaches, the analyses described in this chapter consider resuspension separately and in a more qualitative manner that is based on measured near-bottom current speeds in the vicinity of the diffuser.

Data Requirements

To predict seabed deposition rates of suspended solids, the following information is required:

- Suspended solids mass emission rate

- Current speed and direction

- Height-of-rise of the plume

- Suspended solids settling velocity distribution.

The mass emission rate, M (kg/day), is:

$$M = 86.4(S)(Q)$$ B-1

where:

S = Suspended solids concentration, mg/L

Q = Volumetric flow rate, m^3/sec.

It is suggested that the applicant develop estimates of the suspended solids mass emission rate for the season (90-day period) critical for seabed

deposition and for a yearly period. If the applicant anticipates the mass emission rate will increase during the permit term, the mass emission rate at the end of the permit term should be used.

Current-speed data are needed to determine the distance from the outfall that the sediments will travel before accumulating on the bottom. Consequently, depth-averaged values are best, if available. Otherwise, current speeds near mid-depth may be sufficient. The following current data are needed for the assessment:

- Average value upcoast, when the current is upcoast

- Average value downcoast, when the current is downcoast

- Average value onshore, when the current is onshore

- Average value offshore, when the current is offshore.

If no current data are available, values of 5 cm/sec for longshore transport and 3 cm/sec for onshore-offshore transport have been found to be reasonable values.

Plume trapping levels representative of the critical 90-day period and of the annual cycle are needed. The applicant should use density profiles, effluent volumetric flow rates, and ambient currents characteristic of these time periods. Extreme values should not be used. Usually the annual average and critical 90-day average flow rates and current speeds (in the predominant current direction) should be used. The expected average plume heights-of-rise above the seafloor should be determined using available receiving water density profiles. If large numbers of profiles exist for each month (or oceanographic season), then the applicant could compute the plume height-of-rise above the seafloor for each of the available profiles, and then average the heights. If relatively few profiles are available for each month, then the applicant could compute the plume height of risk for each profile and use the lowest height-of-rise as the average. The monthly

average heights of rise can then be used to compute the average height-of-rise for annual and critical 90-day periods. If so few profiles exist that it is not possible to determine whether differences exist between months (or oceanographic seasons), then the applicant should use the lowest plume height-of-rise (based on calculations using the average effluent flow and current speed) as the average height-of-rise for both the annual and critical 90-day periods.

If the applicant has not determined a suspended solids settling velocity distribution, the following can be used based on data from other Section 301(h) applications:

Primary or Advanced Primary Effluent	Raw Sewage
5 percent have $V_s \geq 0.1$ cm/sec	5 percent have $V_s \geq 1.0$ cm/sec
20 percent have $V_s \geq 0.01$ cm/sec	20 percent have $V_s \geq 0.5$ cm/sec
30 percent have $V_s \geq 0.006$ cm/sec	40 percent have $V_s \geq 0.1$ cm/sec
50 percent have $V_s \geq 0.001$ cm/sec	60 percent have $V_s \geq 0.01$ cm/sec
	85 percent have $V_s \geq 0.001$ cm/sec.

The remaining solids settle so slowly that they are assumed to remain suspended in the water column indefinitely (i.e., they act as colloids). Consequently, 50 percent of the suspended solids in a treated effluent and 85 percent of those in a raw sewage discharge are assumed to be settleable in the ambient environment.

Prediction of Deposition

Although a portion of the settled solids is inert, primary concern is with the organic fraction of the settled solids. For purposes of this evaluation, composition of the waste discharge can be assumed to be as follows:

■   80 percent organic and 20 percent inorganic, for primary or
     advanced primary effluent

■   50 percent organic and 50 percent inorganic, for raw sewage.

Accumulation should be predicted for the critical 90-day period when
seabed deposition is likely to be highest and for steady-state conditions
where average annual values are used. The results should be presented in
graphical form, as shown in Figure B-3. Supporting tables should be
submitted with the application. The applicant must exercise judgment when
developing the contours, especially when accounting for rapid depth changes
offshore. Sediment contours should be expressed in units of g/m^2, not as an
accumulation depth.

An applicant may use a proprietary or publicly available sedimentation
model. Two widely known models are those of Hendricks (1987), which has
been used extensively offshore of Palos Verdes Peninsula in the Southern
California Bight, and Farley (Tetra Tech 1987), which describes the Ocean
Data Evaluation System (ODES) model DECAL. The model DECAL is publicly
available through the U.S. EPA. A simple model is described herein. It can
be used to obtain acceptable estimates of sediment accumulation in a variety
of environments. If its use results in sediment accumulations that lead to
violations of state standards or federal criteria for receiving water
quality, an applicant may use a more sophisticated effluent sediment
accumulation model that better simulates the marine environment.

The method described below assumes that effluent sediment particles
having a specific particle fall velocity settle uniformly within an
elliptical area. This area depends on the plume height-of-rise relative to
the seafloor (not the port depth), the particle fall velocity, and the
average currents speeds in four directions (upcoast, downcoast, onshore, and '
offshore) appropriate for an effluent wastefield at the plume height-of-rise.
For the following sample calculations, the diffuser was assumed to be a
point source. Use of this assumption may not produce reasonable estimates of
sediment accumulation if the diffuser is long. If the diffuser is long,

120   **60**

180

5 g/m^2

10 g/m^2

50 g/m^2

100 g/m^2

DIFFUSER

0   1   **nautical miles**

0   2   **kilometers**
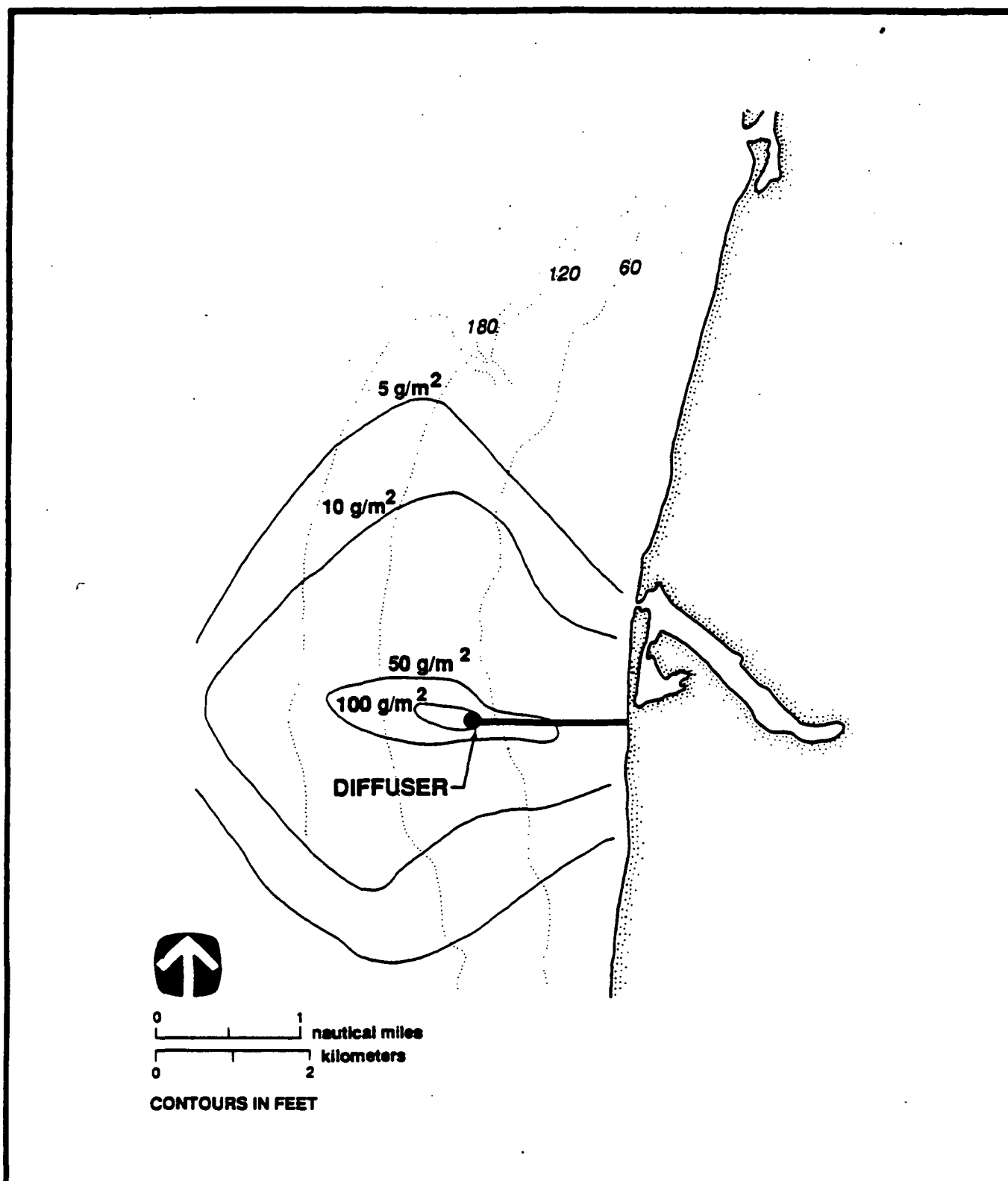
CONTOURS IN FEET

Figure B-3.   Examples of predicted steady-state sediment accumulation around a marine outfall.

then estimates of the sediment accumulation from each diffuser port can be summed to obtain an estimate for the entire diffuser. This sum is approximately the same as that obtained from assuming that the sediment accumulation area is a ZID-like area (with ends the same as the similar elliptical halves computed for a single point discharge) and that the effluent suspended solids having the specific particle fall velocity uniformly settle in this area. The sediment accumulation due to the entire discharge is the sum of the accumulations for each particle fall velocity modeled.

To begin computations for a discharge at a point location, the applicant can create a table similar to Table B-1, showing the amount of organic solids that settle within each settling velocity group, and the maximum distance from the outfall at which each group settles. If the applicant has current data for more than four quadrants, those data can be used. The maximum settling distances for each group in each direction are calculated using the formula shown in the footnote of Table B-1.

With a sufficiently detailed map (e.g., a NOAA bathymetric chart), each point D_1 through D_{16}, or R_1 through R_{20}, can be plotted with the center of the diffuser as the reference point. Depositional contours are formed by the four points that define the perimeter of a depositional field (e.g., $D_1D_2D_3D_4$). The applicant should join these points by smooth lines, so that the contours are elliptically shaped. If the applicant has current data at 60° or 30° intervals, instead of the 90° intervals used here, then the contours could be created more accurately.

The deposition rates corresponding to each contour are determined as follows. First, predict the deposition rate within each contour due to each individual settling velocity group, as shown in Table B-2. This quantity is M_i/A_i, or the group deposition rate divided by the area within the contour. The area within any contour is a function of the four points (e.g., D_1, D_2, D_3, and D_4), and is denoted in the table by $f(D_1D_2D_3D_4)$. A planimeter is probably the most accurate method of finding the area. Once the deposition rates by group have been found, then the total deposition rate can be calculated by summing all contributing deposition rates. For

Mass Emission Rate = M_T

Organic Component = Mo = $\begin{cases} 0.8\ M_T, \text{ for primary effluent} \\ \\ 0.5\ M_T, \text{ for raw effluent} \end{cases}$

Percent by Settling Velocity Group	Organic Component by Group	Maximum Settling Distance from Outfall[a]			
		Upcoast	Downcoast	Onshore	Offshore
Primary Effluent					
5 (V_s = 0.1 cm/sec)	0.04 M_T	D_1	D_2	D_3	D_4
15 (V_s = 0.01 cm/sec)	0.12 M_T	D_5	D_6	D_7	D_8
10 (V_s = 0.006 cm/sec)	0.08 M_T	D_9	D_{10}	D_{11}	D_{12}
20 (V_s = 0.001 cm/sec)	0.16 M_T	D_{13}	D_{14}	D_{15}	D_{16}
	Sum = $\overline{0.40\ M_T}$				
Raw Sewage					
10 (V_s = 1.0 cm/sec)	0.05 M_T	R_1	R_2	R_3	R_4
10 (V_s = 0.5 cm/sec)	0.05 M_T	R_5	R_6	R_7	R_8
20 (V_s = 0.1 cm/sec)	0.10 M_T	R_9	R_{10}	R_{11}	R_{12}
20 (V_s = 0.01 cm/sec)	0.10 M_T	R_{13}	R_{14}	R_{15}	R_{16}
25 (V_s = 0.001 cm/sec)	0.125 M_T	R_{17}	R_{18}	R_{19}	R_{20}
	Sum = $\overline{0.425\ M_T}$				

[a] The distance D (or R) is calculated as: $D\ (\text{or } R) = \dfrac{V_a H_T}{V_s}$

where:

V_a = Ambient velocity = 5 cm/sec upcoast and downcoast (default) and 3 cm/sec onshore and offshore (default)

H_T = Average trapping level of plume, measured above bottom

V_s = Appropriate settling velocity by group for primary or raw discharges

If the bottom slope is 5 percent or greater, D should be calculated as follows:

$$D = \frac{H_T}{S + \dfrac{V_s}{V_a}}$$

where:

S = Slope, m/m, positive if onshore, negative if offshore.

B-12

TABLE B-2. EXAMPLE TABULATIONS OF DEPOSITION RATES AND ACCUMULATION RATES BY CONTOUR

Organic Mass Component by Group	Bottom Area	Mass Deposition Rate, by Group	Total Organic Deposition Rate within Area ($g/m^2/yr$)	Accumulation (g/m^2) Steady-State	90 Day
Primary Effluent					
$0.04\ M_T = M_1$	$A_1 = f(D_1 D_2 D_3 D_4)$	M_1/A_1	$M_1/A_1 + M_2/A_2 + M_3/A_3 + M_4/A_4 = f_1$		
$0.12\ M_T = M_2$	$A_2 = f(D_5 D_6 D_7 D_8)$	M_2/A_2	$M_2/A_2 + M_3/A_3 + M_4/A_4 = f_2$	$\dfrac{f_i}{k_d}$	$\dfrac{f_i}{k_d}[1-\exp(-90k_d)]$
$0.08\ M_T = M_3$	$A_3 = f(D_9 D_{10} D_{11} D_{12})$	M_3/A_3	$M_3/A_3 + M_4/A_4 = f_3$		
$0.16\ M_T = M_4$	$A_4 = f(D_{13} D_{14} D_{15} D_{16})$	M_4/A_4	$M_4/A_4 = f_4$		
Raw Sewage					
$0.05\ M_T = M_1$	$A_1 = f(R_1 R_2 R_3 R_4)$	M_1/A_1	$M_1/A_1 + M_2/A_2 + M_3/A_3 + M_4/A_4 + M_5/A_5 = f_1$		
$0.05\ M_T = M_2$	$A_2 = f(R_5 R_6 R_7 R_8)$	M_2/A_2	$M_2/A_2 + M_3/A_3 + M_4/A_4 + M_5/A_5 = f_2$	$\dfrac{f_i}{k_d}$	$\dfrac{f_i}{k_d}[1-\exp(-90k_d)]$
$0.10\ M_T = M_3$	$A_3 = f(R_9 R_{10} R_{11} R_{12})$	M_3/A_3	$M_3/A_3 + M_4/A_4 + M_5/A_5 = f_3$		
$0.10\ M_T = M_4$	$A_4 = f(R_{13} R_{14} R_{15} R_{16})$	M_4/A_4	$M_4/A_4 + M_5/A_5 = f_4$		
$0.125\ M_T = M_5$	$A_5 = f(R_{17} R_{18} R_{19} R_{20})$	M_5/A_5	$M_5/A_5 = f_5$		

Note: Units of f_i are $g/m^2/day$.

example, the innermost contour receives contributions from all groups, while the outermost contour receives a contribution only from one group.

So far, only the rates of organic deposition (in units of $g/m^2/yr$) have been predicted. The accumulation of the organic material (S_i) can be predicted by including decay as follows:

$$S_i \ (g/m^2) = \frac{f_i}{k_d} \text{, at steady state}$$

$$S_i \ (g/m^2) = \frac{f_i}{k_d} \ [1 - \exp \ (-90 \ k_d)] \text{, for 90 days.}$$

B-2

The f_i are the deposition rates in units of $g/m^2/day$, as contrasted to the units of $g/m^2/yr$ in Table B-2. The decay rate constant, k_d, has a typical value of 0.01/day. For example, if the organic deposition rate for annual conditions is 100 $g/m^2/yr$, the steady-state accumulation is:

$$100 \ g/m^2/yr \ x \ \frac{1 \ yr}{365 \ days} \ x \ \frac{1}{0.01/day} = 27 \ g/m^2. \qquad \text{B-3}$$

If the organic deposition rate for the critical 90-day period is 300 $g/m^2/yr$, the 90-day accumulation is:

$$300 \ g/m^2/yr \ x \ \frac{1 \ yr}{365 \ days} \ x \ \frac{1}{0.01/day} \ x \ [1-\exp \ (-90 \ x \ 0.01)] = 49 \ g/m^2. \qquad \text{B-4}$$

This example shows that input data for the 90-day and steady-state accumulations are different. Consequently, Tables B-1 and B-2 should each be completed twice. Also the accumulation over a critical 90-day period can exceed the steady-state accumulation. This is caused by short-term deposition rates that are considerably higher than the long-term average. In the example, the maximum 90-day deposition rate of 300 $g/m^2/yr$ would eventually decrease to values below 100 $g/m^2/yr$, so that on a yearly basis the deposition rate is 100 $g/m^2/yr$.

B-II. DISSOLVED OXYGEN CONCENTRATION FOLLOWING INITIAL DILUTION

When wastewater is discharged through a single port or a diffuser, the effluent forms a buoyant plume that entrains ambient water as it rises. Because the initial dilution process occurs rapidly (i.e., on the order of minutes), BOD exertion (a relatively slow process) is negligible during this period. However, an immediate dissolved oxygen demand (IDOD), which represents the oxygen demand of reduced substances that are rapidly oxidized (e.g., sulfides to sulfates), might not be negligible. The dissolved oxygen concentration following initial dilution can be predicted using the following expression:

$$DO_f = DO_a + (DO_e - IDOD - DO_a)/S_a \qquad \text{B-5}$$

where:

DO_f = Final dissolved oxygen concentration of receiving water at the plume trapping level, mg/L

DO_a = Affected ambient dissolved oxygen concentration immediately upcurrent of the diffuser averaged over the tidal period (12.5 h) and from the diffuser port depth to the trapping level, mg/L

DO_e = Dissolved oxygen of effluent, mg/L

$IDOD$ = Immediate dissolved oxygen demand, mg/L

S_a = Initial dilution (flux-averaged).

The applicant should use the least favorable combination of values for effluent dissolved oxygen, IDOD, affected ambient dissolved oxygen, and initial dilution. The effluent dissolved oxygen concentration at the point

of discharge from the treatment plant is often 0.0 mg/L. Because the critical case is desired, a concentration of 0.0 mg/L is a reasonable value. However, if data show that dissolved oxygen concentrations in the effluent are greater than 0.0 mg/L during the critical periods, then these data may be used.

The IDOD values typically vary from 0 to 10 mg/L, but can be higher depending on the level of treatment and presence of industrial flows. Table B-3 can be used to select reasonable IDOD values. Alternatively, the IDOD can be measured as discussed below. The influence of the effluent IDOD on ambient dissolved oxygen can be estimated from the following table (calculated as $-IDOD/S_a$):

Contribution of IDOD to Lowering of DO_f (mg/L)

IDOD (mg/L)	Initial Dilution			
	10	30	50	100
1	-0.1	-0.03	-0.02	-0.01
2	-0.2	-0.07	-0.04	-0.02
5	-0.5	-0.17	-0.1	-0.05
10	-1.0	-0.33	-0.2	-0.10
20	-2.0	-0.67	-0.4	-0.20

At high initial dilutions, the IDOD contribution is small. Thus, the expense of laboratory tests may be unwarranted. If IDOD is to be determined experimentally, the procedures in *Standard Methods* (American Public Health Association 1985, p. 530) should be generally followed except that the dilution water should be seawater from the discharge site instead of distilled water, and the effluent sample should be incubated anaerobically for a length of time equal to the travel times from the plant through the diffuser for minimum, average, and maximum flow conditions. The effluent sample should be mixed with the dilution water after incubation. The dissolved oxygen concentration of the effluent and dilution water should be measured separately after incubation and before mixing the samples. The dissolved oxygen of the mixture should be measured 15 min after preparation.

TABLE B-3. TYPICAL IDOD VALUES

Treatment Level	Effluent BOD$_5$, mg/L	Travel Time, min[a]	IDOD, mg/L
Untreated or less than primary		<60	5
		60-200	10
		200-300	15
		>300	20
Primary	50-100	0-100	2
	50-100	100-300	3
	50-100	>300	4
	100-150	0-100	3
	100-150	100-300	4
	100-150	>300	5
	150-200	0-100	5
	150-200	100-300	7
	150-200	>300	8
Advanced primary	<50	0-60	0
	<50	>60	1

[a] Travel time should include the total travel time from the treatment plant through the diffuser, including any land portion of the outfall.

Note: Information compiled from 301(h) applications.

The IDOD is calculated using the following equation:

$$IDOD = \frac{(DO_D)(P_D) + (S)(P_S) - DO_M}{P_S} \qquad\qquad B-6$$

where:

IDOD = Immediate dissolved oxygen demand, mg/L

DO_D = Dissolved oxygen of dilut on water (seawater), mg/L

P_D = Decimal fraction of dilution water used

S = Dissolved oxygen of effluent after incubation, mg/L

P_S = Decimal fraction of effluent used

DO_M = Dissolved oxygen of mixture after 15 min, mg/L.

Several dilutions should be used, preferably close to the actual initial dilution, unless the difference between the initial and mixed concentrations is less than 0.1 mg/L. All data used in the above calculations, the incubation times, and the computed results for each test should be included in the application.

The lowest initial dilution (flux-averaged) should be used for the final dissolved oxygen calculation. Usually, this dilution will correspond to the maximum flow rate at the end of the permit term. Low initial dilutions can also occur at smaller effluent flow rates if stratification is sufficiently severe. Typically, dilutions during periods of maximum stratification should be used for the final dissolved oxygen calculation.

The affected ambient dissolved oxygen concentrations should also represent critical conditions. Usu y, critical conditions will occur

during the maximum stratification period in the late summer or in the spring when upwelling of deep ocean water occurs. For existing discharges, the affected ambient data should be collected at locations directly upcurrent of the diffuser, thereby incorporating the potential effects of recirculation. For proposed new or relocated discharges, affected ambient dissolved oxygen levels upcurrent of the diffuser should be estimated from mathematical models of the discharge or by extrapolation from similar situations. Dissolved oxygen data, as well as any ambient water quality constituent, may be averaged between the depth of the discharge ports and the plume trapping level, which corresponds to the lowest initial dilution that was used to predict the final dissolved oxygen concentrations. If applicants use a mathematical model that allows multiple vertical levels of input for ambient water quality instead of an average value, this should be noted.

The time period over which ambient data may be averaged may depend on specifications of intensity and duration factors in applicable water quality standards. For example, if certain numerical values shall not be compromised over a period of 4 h, a 4-h average of input data may be reasonable. Absent any more stringent specification in locally applicable standards, an average over a half tidal cycle (approximately 12.5 h) would provide a generally conservative estimate.

The affected ambient dissolved oxygen concentration can change substantially as a function of depth, depending on environmental character-istics and seasonal influences (e.g., upwelling). As the plume rises during initial dilution, water from deeper parts of the water column is entrained into the plume and advected to the plume trapping level. If the dissolved oxygen concentration is lower in the bottom of the water column than at the trapping level, the low dissolved oxygen water is advected to a region formerly occupied by water containing higher concentrations of dissolved oxygen. The result is an oxygen depression caused by entrainment.

This oxygen depression caused by the waste discharge and associated entrainment (ΔDO_1) should be computed as the difference between DO_f as

defined in Equation B-5 and the affected ambient dissolved oxygen concentration at the trapping depth (DO_t).

$$\Delta DO_1 = DO_f - DO_t = DO_a - DO_t + (DO_{e-} - IDOD - DO_a)/S_a \qquad \text{B-7}$$

The oxygen depression of the wastefield relative to the trapping depth expressed in percent is $(-\Delta DO_1/DO_t)100$.

For cases when the effect of entraining low dissolved oxygen water can be neglected, the oxygen depletion (ΔDO_2) should be computed as the difference between the average affected ambient dissolved oxygen concentration (DO_a) in the entrained water and DO_f as shown below.

$$\Delta DO_2 = DO_f - DO_a = (DO_e - IDOD - DO_a)/S_a \qquad \text{B-8}$$

The oxygen depletion of the wastefield relative to the average affected ambient dissolved oxygen concentration is $(-\Delta DO_2/DO_a)100$.

The equation of Baumgartner (1981) for the percentage depression is:

$$\frac{(DO_t - DO_e + IDOD)}{DO_t \times S_a} \times 100 \qquad \text{B-9}$$

This equation can be derived by assuming that $DO_a = DO_t$ in Equation B-7. Use of Equation B-9 has been allowed in the State of California.

These differences can be described as a percentage of the ambient concentration or as a numerical difference, depending on the requirements of the state. In some states, the final dissolved oxygen concentration must be above a specified limit or must be converted to percent saturation to determine whether the final concentration is above a prescribed limit. Dissolved oxygen saturation can be determined as a function of temperature and salinity using the method of Green and Carritt (1967) and Hyer et al. (1971) as tabulated in Table B-4. The applicant may want to consult with

TABLE B-4. DISSOLVED OXYGEN SATURATION VALUES

Temperature (° C)	Dissolved Oxygen Saturation, mg/L Salinity (ppt)								
	20	22	24	26	28	30	32	34	36
0	12.8	12.6	12.5	12.3	12.1	12.0	11.8	11.7	11.5
1	12.5	12.3	12.2	12.0	11.8	11.7	11.5	11.4	11.2
2	12.1	12.0	11.9	11.7	11.5	11.4	11.2	11.1	10.9
3	11.8	11.7	11.5	11.4	11.2	11.1	10.9	10.8	10.7
4	11.5	11.4	11.3	11.1	11.0	10.8	10.7	10.5	10.4
5	11.3	11.1	11.0	10.8	10.7	10.6	10.4	10.3	10.1
6	11.0	10.9	10.7	10.6	10.4	10.3	10.2	10.0	9.9
7	10.7	10.6	10.5	10.4	10.2	10.1	9.9	9.8	9.7
8	10.5	10.3	10.2	10.1	9.9	9.8	9.7	9.6	9.4
9	10.2	10.1	10.0	9.8	9.7	9.6	9.5	9.3	9.2
10	10.0	9.9	9.7	9.6	9.5	9.4	9.2	9.1	9.0
11	9.6	9.6	9.5	9.4	9.3	9.2	9.0	8.9	8.8
12	9.5	9.4	9.3	9.2	9.1	9.0	8.8	8.7	8.6
13	9.3	9.2	9.1	9.0	8.9	8.8	8.7	8.5	8.4
14	9.1	9.0	8.9	8.8	8.7	8.6	8.5	8.4	8.3
15	8.9	8.8	8.7	8.6	8.5	8.4	8.3	8.2	8.1
16	8.7	8.6	8.5	8.4	8.3	8.2	8.1	8.0	8.0
17	8.6	8.5	8.4	8.3	8.2	8.1	8.0	7.9	7.8
18	8.4	8.3	8.2	8.1	8.0	7.9	7.8	7.7	7.6
19	8.2	8.1	8.0	8.0	7.9	7.8	7.6	7.6	7.5
20	8.1	8.0	7.9	7.7	7.7	7.6	7.6	7.5	7.4
21	7.9	7.9	7.7	7.7	7.6	7.5	7.5	7.4	7.3
22	7.8	7.7	7.6	7.6	7.5	7.4	7.4	7.3	7.2
23	7.7	7.6	7.5	7.5	7.4	7.3	7.3	7.2	7.1
24	7.6	7.5	7.4	7.4	7.3	7.2	7.2	7.1	7.1
25	7.5	7.4	7.3	7.3	7.2	7.1	7.1	7.0	7.0
26	7.4	7.3	7.2	7.2	7.1	7.1	7.0	7.0	7.0
27	7.2	7.2	7.2	7.1	7.0	7.0	6.9	6.9	6.9
28	7.2	7.1	7.1	7.0	7.0	6.9	6.9	6.9	6.8
29	7.1	7.1	7.0	7.0	6.9	6.9	6.8	6.8	6.8
30	7.1	7.1	7.0	6.9	6.9	6.8	6.8	6.8	6.7

the state water quality agency to determine whether any other methods are used to determine compliance with the dissolved oxygen standards.

B-III. FARFIELD DISSOLVED OXYGEN DEPRESSION

Subsequent to initial dilution, dissolved oxygen in the water column is consumed by the BOD in the wastefield. The effluent BOD_5 after initial dilution is needed to estimate farfield dissolved oxygen depletion. The final BOD_5 concentration can be estimated using the following expression:

$$BOD_f = BOD_a + (BOD_e - BOD_a)/S_a \qquad \text{B-10}$$

where:

BOD_f = Final BOD_5 concentration, mg/L

BOD_a = Affected ambient BOD_5 concentration immediately updrift of the diffuser averaged over one-half the tidal period (12.5 h) and from the diffuser port depth to the trapping depth, mg/L

BOD_e = Effluent BOD_5 concentration, mg/L

S_a = Initial dilution (flux-averaged).

This equation provides an estimate of the total BOD_5 concentration in the receiving water. The maximum contribution due to the effluent alone can be determined by dividing the effluent BOD_5 concentration by the initial dilution. This value is used later to estimate farfield effects of the effluent. As a critical case, the maximum monthly average effluent BOD_5 concentration should be used with the (monthly) critical initial dilution. For existing plants, the previous 12 mo of effluent BOD_5 data is used to support the selection of a BOD_5 concentration. For proposed or modified treatment plants where effluent data are not available, monthly average influent BOD_5 data should be provided along with the range of daily values.

The average removal efficiency for the new or modified plant is also needed to compute estimated effluent BOD_5 concentrations.

Three approaches to assessing farfield dissolved oxygen demand are described below:

- Simplified mathematical models predicting dissolved oxygen depletions, using calculation techniques that do not require computer support

- Numerical models predicting dissolved oxygen depletions, using a computer

- Evaluation of field data, using a data-intensive approach where dissolved oxygen concentrations are measured in the water column and compared to ambient concentrations.

Before undertaking any analysis to determine whether farfield BOD exertion causes a violation of the dissolved oxygen standard, the applicant should first check to see whether:

$$DO_{STD} \leq DO_f - BOD_{fu}, \text{ for critical conditions} \qquad \text{B-11}$$

where:

DO_{STD} = Dissolved oxygen standard

DO_f = Dissolved oxygen concentration at the completion of initial dilution

BOD_{fu} = Ultimate BOD at the completion of initial dilution ($= BOD_f \times 1.46$).

If the above inequality is true, then the discharge will not violate the dissolved oxygen standard due to BOD exertion and no further analysis of

farfield BOD exertion is required. If the inequality is not true, then further analysis is required.

SIMPLIFIED MATHEMATICAL MODELS

Oxygen depletion due to coastal or estuarine wastewater discharges is primarily caused by exertion of BOD, although increased nutrient levels can affect oxygen concentrations indirectly by altering algal photosynthesis and respiration rates. BOD consists of a carbonaceous component (CBOD) and nitrogenous component (NBOD). Both components can contribute to oxygen depletion. CBOD is often reported as BOD_5, the 5-day BOD. Before using BOD to predict oxygen depletion, the applicant should convert it to BOD_L, the ultimate BOD, by the following relationship:

$$BOD_L = 1.46 \; BOD_5 \qquad\qquad B-12$$

A typical decay rate for CBOD is 0.23/day (base e) at 20° C. A temperature correction should be made as follows:

$$k_T = 0.23 \times 1.047^{T-20}/day \qquad\qquad B-13$$

where:

k_T = BOD decay rate at temperature T ($^\circ$ C).

NBOD might not always contribute to oxygen depletion. If the applicant discharges into open coastal waters where there are no other major discharges in the vicinity, the background population of nitrifying bacteria might be negligible. Under these circumstances, the NBOD will not be exerted immediately. In more enclosed estuarine waters, nitrification in the water column has been documented from numerous water quality studies. Applicants should analyze the potential impact of NBOD, if they discharge into estuarine waters.

NBOD can be estimated based on data for total Kjeldahl nitrogen concentration (the sum of organic nitrogen and ammonia nitrogen) in the waste discharge using the following relationships:

$$NBOD_L = 4.57 \ (TKN) \qquad\qquad B\text{-}14$$

$$NBOD_5 = NBOD_L/2.54$$

where:

   TKN = Total Kjeldahl nitrogen

   $NBOD_L$ = Ultimate NBOD

   $NBOD_5$ = 5-day NBOD.

The decay rate of NBOD can be taken as:

$$k_T = 0.10 \times 1.047^{T-20}/day \qquad\qquad B\text{-}15$$

where:

   k_T = The decay rate at temperature T (^{o} C)

   0.10 = The decay rate at 20° C (base e).

Simplified mathematical models are an acceptable alternative to the more complex numerical models. In the simplest model of oxygen depletion, the following are generally assumed:

■   The wastewater plume is submerged at the completion of initial dilution for critical conditions (so that direct reaeration of atmospheric oxygen into the wastefield does not occur).

■ Oxygen depletion is a function of distance from the discharge and is caused by carbonaceous oxygen demand and nitrogenous oxygen demand.

■ The wastefield entrains ambient water as a function of travel time. Lateral dilution is the predominant mechanism of entrainment.

If the applicant demonstrates that the plume will always surface, then the effects of atmospheric reaeration can be included; otherwise they should not be included.

When applying a model that predicts farfield oxygen depletion, it is suggested that the applicant plot dissolved oxygen depletion as a function of travel time so that the behavior of dissolved oxygen concentrations in the wastefield can be examined to locate minimum values.

Example oxygen depletion curves as a function of travel time are shown in Figure B-4. The depletion indicated at time, t=0, denotes the depletion immediately following initial dilution. The dissolved oxygen deficits plotted in the figure are relative to the ambient concentration, and tend to approach zero at travel times longer than those shown in the figure.

For the three cases, the maximum deficits occur at the following travel times:

■ 0.0 days for Curve A

■ Approximately 0.2 days for Curve B

■ Approximately 4.0 days for Curve C.

The primary reason for the difference in magnitude and time of occurrence of the maximum deficits is the IDOD, which varies from a high of 66 mg/L for Curve A to 0.0 mg/L for Curve C. When the IDOD is 66 mg/L (a high value,

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Figure B-4.   Dissolved oxygen deficit vs. travel time for a submerged wastefield.
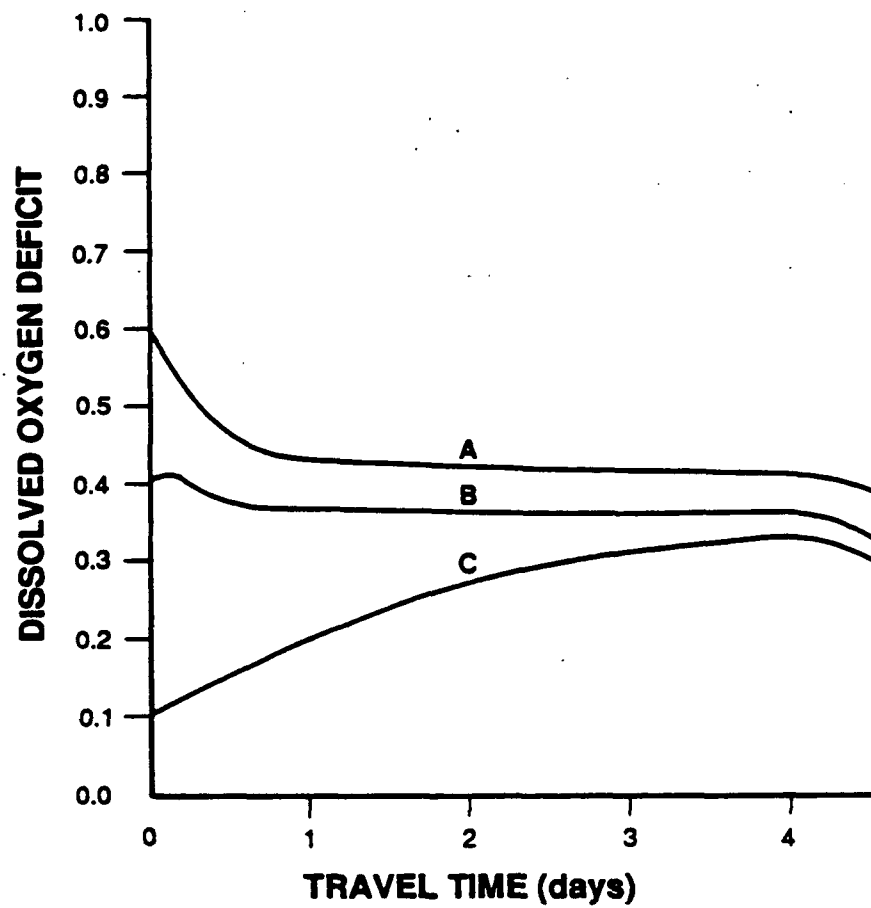
CURVE	BOD$_f$ (ultimate) (mg/L)	INITIAL DO DEMAND (mg/L)
A	3.5	66.
B	3.5	44.
C	3.5	0.

but one that could be associated with an unusual discharge), the maximum depletion is caused by initial mixing processes, and not by farfield BOD exertion. Conversely, when IDOD is 0.0 mg/L, the maximum depletion is caused by BOD exertion, and occurs at some distance from the discharge.

The simplified farfield oxygen depletion model for coastal waters suggested herein is based on an approach developed by Brooks (1960) for predicting wastefield dilution subsequent to initial dilution. The dissolved oxygen concentration in the receiving waters can be expressed as a function of travel time as follows:

$$DO(t) = DO_a + \frac{DO_f - DO_a}{D_s} - \frac{L_{fc}}{D_s} \left[1 - \exp(-k_c t) \right] - \frac{L_{fn}}{D_s} \left[1 - \exp(k_n t) \right] \qquad \text{B-16}$$

where:

$DO(t)$ = Dissolved oxygen concentration in a submerged wastefield as a function of travel time t, mg/L

DO_a = Affected ambient dissolved oxygen concentration immediately updrift of the diffuser, mg/L

DO_f = Dissolved oxygen concentration at the completion of initial dilution calculated using Equation B-5, mg/L

k_c = CBOD decay rate constant

k_n = NBOD decay rate constant

L_{fc} = Ultimate CBOD concentration above ambient at completion of initial dilution, mg/L

L_{fn} = Ultimate NBOD concentration above ambient at completion of initial dilution, mg/L

D_s = Dilution attained subsequent to initial dilution as a function of travel time.

The above equation expresses the dissolved oxygen deficit that arises because of an initial deficit at the completion of initial dilution (DO_a-DO_f) plus that caused by exertion of BOD in the water column. The last term in the above equation estimates the exertion due to NBOD. The dissolved oxygen deficit tends to decrease at longer travel times as a result of subsequent dilution and to increase as a result of BOD exertion. Depending on the particular case being analyzed, one influence can dominate the other over a range of travel times so that a minimum dissolved oxygen level can occur either immediately following initial dilution or at a subsequent travel time, as previously shown in Figure B-4.

To predict farfield oxygen distribution, one must determine the dilution attained within the wastefield as a function of time following discharge. For open coastal areas, dilution is often predicted using the 4/3 law (Brooks 1960), which states that the lateral diffusion coefficient increases as the 4/3 power of the wastefield width. In mathematical form:

$$\epsilon = \epsilon_0 \left(\frac{L}{b} \right)^{4/3}$$

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where:

ϵ = Lateral diffusion coefficient, ft^2/sec

ϵ_0 = Diffusion coefficient when L = b

L = Width of sewage field at any distance from the ZID, ft

b = Initial width of sewage field (approximately as the longest dimension of the ZID), ft.

The initial diffusion coefficient can be predicted from:

$$\epsilon_0 = 0.001 \, b^{4/3} \, ft^2/sec \qquad\qquad \text{B-18}$$

Based on the 4/3 law, the centerline dilution, D_S, is given by:

$$D_s = 1/erf\left(\left[\frac{1.5}{\left(1 + \frac{8 \, \epsilon_0 \, t}{b^2}\right)^3 - 1}\right]^{1/2}\right) \qquad\qquad \text{B-19}$$

where:

t = Travel time, sec

erf = The error function.

The 4/3 law is not always applicable, especially in coastal areas or estuaries. In coastal areas, Grace (1978) suggests that the diffusion coefficient vary linearly with L. The subsequent dilution can be expressed as:

$$D = 1/erf\left(\left[\frac{1.5}{\left(1 + \frac{12\epsilon_0 t}{B^2}\right)^2 - 1}\right]^{1/2}\right) \qquad\qquad \text{B-20}$$

A more conservative choice is to assume the diffusion coefficient is a constant. The subsequent dilution can then be expressed as:

$$D_s = 1/erf\left[\left(\frac{b^2}{16 \, \epsilon_0 \, t}\right)^{1/2}\right] \qquad\qquad \text{B-21}$$

These three equations are cumbersome to use, especially if repeated applications are needed. To facilitate predicting subsequent dilutions, values of D_S as a function of $12\epsilon_0 t/b^2$ are shown in Figure B-5 for values of Brooks' n equal to 0, 1, and 4/3. For example, if b = 100 ft, and t = 9,000 sec (2.5 h), then ϵ_0 = 0.464 ft^2/sec and $12\epsilon_0 t/b^2$ = 5.0. Assuming that Brooks' n = 1, then use of Figure B-5 shows that D_S = 4.3 approximately.

The figure also reveals that the predicted dilutions are substantially different, depending on the relationship obeyed by the lateral diffusion coefficient. In some instances, the Brooks' n = 1 law might overestimate subsequent dilution, even if the outfall is in coastal waters. To attain the subsequent dilutions predicted at large travel times, a large amount of dilution water must be available. Because many outfalls, particularly small ones, are not far from shore, the entrainment rate of dilution water can be restricted by the presence of the shoreline and the depth of the water. As the wastefield widens substantially, the rate of entrainment could decrease, and neither the Brooks' n = 4/3 nor the Brooks' n = 1 law may be obeyed. It is suggested that applicants be conservative and base subsequent dilution on a constant lateral diffusion coefficient (i.e., Brooks' n = 0), rather than the Brooks' n = 1 or Brooks' n = 4/3 laws. However, if the applicant can show that the 4/3 law (or some other relationship) is applicable to the discharge site, then that relationship should be used.

If the applicant's discharge is near the mouth of a wide estuary, the approach just discussed can be used directly to predict oxygen depletion. If, however, the applicant discharges into a long narrow estuary, then it is likely that the sides of the estuary will limit the lateral dilution that is attainable. Under these conditions, the maximum dissolved oxygen deficit with respect to saturation can be predicted as:

$$D = \frac{kW}{A(k_2-k)}\left[\frac{1}{\sqrt{4kE_L}} - \frac{1}{\sqrt{4k_2E_L}}\right] \qquad \text{B-22}$$
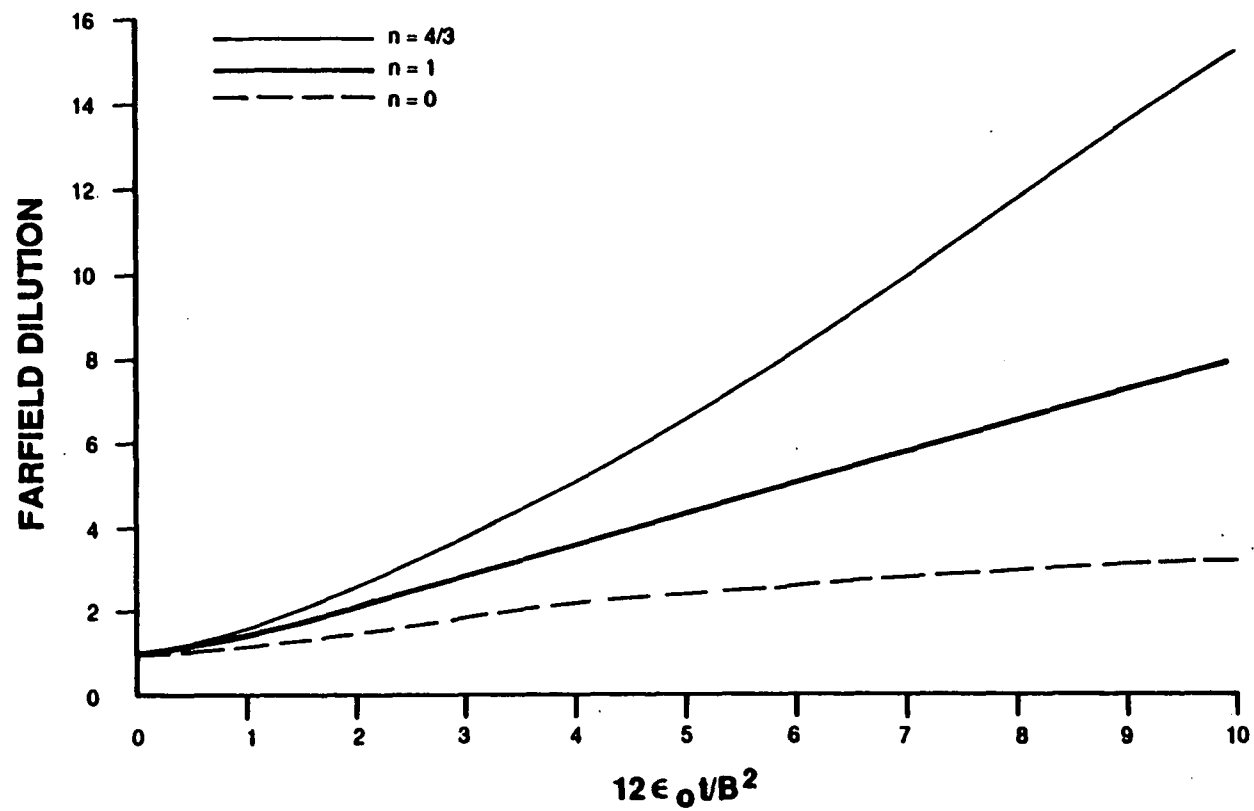
Figure B-5.   Farfield dilution as a function of $12\epsilon_0 t/B^2$.

where:

    D = Dissolved oxygen deficit

    A = Cross-sectional area of the estuary near the discharge site

    k = CBOD decay rate constant

    k_2 = Reaeration rate constant

    E_L = Longitudinal dispersion coefficient

    W = Mass loading rate of CBOD.

The applicant can predict the deficits due to NBOD by using the appropriate k and W values and adding the two deficits to get the total. With reasonable values for the constants, the total dissolved oxygen deficit for discharge to narrow estuaries becomes:

$$D = (3.14 \, W_c + 2.55 \, W_n) \, 10^{-4}/A \qquad \text{B-23}$$

where:

    A = Cross-sectional area in m^2

    W_c = Mass emission rate of CBOD, g/day

    W_n = Mass emission rate of NBOD, g/day

    D = Dissolved oxygen deficit, mg/L.

The NBOD term can be added when data are available.

NUMERICAL MODELS

Numerical models are an acceptable method of predicting oxygen depletion caused by a discharge. Numerical models may consider the combined effect of farfield demand in the water column, as discussed above, and the oxygen demand associated with organic sediments. If not, the applicant may have to augment the numerical modeling analysis to address unanswered questions associated with sediment oxygen demand.

The applicant should try to isolate the impact of the outfall on dissolved oxygen concentrations by considering that the applicant's discharge is the sole source of oxygen depletion in the system being modeled. The applicant can then predict the dissolved oxygen depletion caused by the discharge by subtracting the background dissolved oxygen level from those predicted by the model. This approach also simplifies the applicant's analysis because data from other wastewater sources are not required.

Specific guidelines can be offered to applicants who choose to use numerical models. Typically, the most severe dissolved oxygen depletion due to BOD exertion occurs when the water column is density stratified in the presence of tidally reversing currents and low nontidal currents, and the wastefield remains submerged following initial dilution. If such conditions occur at the applicant's outfall site, then the numerical model should be layered vertically, with a minimum of two layers. The plume should be discharged into the bottom layer to simulate the submerged discharge with the consequence that direct atmospheric reaeration is not present in this layer.

The applicant should set up the grid system for the numerical model such that the smallest segments are located in the vicinity of the diffuser and gradually increase in size with distance from the diffuser. The volume of the segments in the immediate vicinity of the diffuser should approximate the volume of the ZID in order to prevent an initial dilution that is artificially high and that would cause the model to underestimate dissolved oxygen depletion. The applicant might choose to experiment with grid

configuration by starting with a coarse grid and then decreasing grid size until the model results do not significantly change.

A steady-state numerical model will be acceptable for the dissolved oxygen analysis because dynamic or unsteady analyses are generally more costly, more difficult to implement, and require more data. The applicant should consider, however, whether intratidal variations can cause more severe depletions than are predicted by a steady-state model that calculates average oxygen depletions over a tidal cycle. Slack tide, for example, might be critical because oxygen-demanding materials can accumulate in the vicinity of the discharge. For existing discharges, the applicant might want to augment the steady-state modeling analysis by an abbreviated sampling program to determine dissolved oxygen depletions during slack-tide periods within a tidal cycle. Intratidal variations are likely to be more important in enclosed estuaries than along open coastal areas.

EVALUATION OF FIELD DATA

Extensive field data collection and analysis are required to fully implement this third approach. Limited samples of water column dissolved oxygen may be inadequate to demonstrate compliance with standards under critical conditions. Limited information should be supplemented with analyses based on numerical or simplified mathematical modeling.

These statements should not discourage applicants from collecting and submitting dissolved oxygen data from the vicinity of an existing discharge. To the contrary, such data, if available, should be submitted, particularly if the Section 301(h) application is for a current discharge or for an improved or altered discharge at the same location. However, the data might reveal only a portion of the impact of the wastefield, for the following reasons:

- The location of the maximum oxygen depletion might not be sampled.

- The sampling program could have been conducted during a period that was not critical with respect to the discharge or receiving water conditions. Critical discharge conditions are generally associated with high effluent BOD and high volumetric flow rates. Critical receiving water conditions are usually associated with minimum initial dilutions (maximum density stratification), maximum water temperatures, and possibly slack-tide or low nontidal current conditions.

- Ambient dissolved oxygen concentrations can vary spatially and temporally for conditions unrelated to the discharge (e.g., upwelling effects). Consequently, dissolved oxygen depletions associated with the discharge can be masked by background variability.

Some applicants might have access to dissolved oxygen demand data collected adjacent to another outfall at a nearby coastal area and attempt to use those data to show that their own discharge will not violate dissolved oxygen standards. This approach can be, but is not always, reliable. The applicants should include in the application sufficient information such that the data collection program for the nearby area can be reviewed, and then show that the predicted dissolved oxygen depletions are the maximum likely to be produced at the nearby discharge site. The applicant should also demonstrate that the results of the nearby discharge can be extrapolated to the applicant's discharge. Essentially, the dissolved oxygen depletion at the adjacent discharge (due to both BOD utilization and sediment oxygen demand) will need to be at least as severe as that at the applicant's discharge.

B-IV. SEDIMENT OXYGEN DEMAND

The oxygen depletion due to a steady sediment oxygen demand can be predicted by:

$$\Delta DO = \frac{\overline{S}_B \ X_M}{86,400 \ UHD} = \frac{a \ \overline{S} \ k_d \ X_M}{86,400 \ UHD} \qquad \text{B-24}$$

where:

ΔDO = Oxygen depletion, mg/L

$\overline{S}_B$ = Average benthic oxygen demand over the deposition area, $\cdot g \ O_2/m^2/day$

X_M = Length of deposition area (generally measured in longshore direction), m

H = Average depth of water column influenced by sediment oxygen demand, measured above bottom, m

U = Minimum sustained current speed over deposition area, m/sec

k_d = Sediment decay rate constant, 0.01/day

a = Oxygen:sediment stoichiometric ratio, 1.07 mg O_2/mg sediment

$\overline{S}$ = Average concentration of deposited organic sediments over the deposition area, g/m^2

D = Dilution caused by horizontal entrainment of ambient water as it passes over the deposition area (always ≥ 1).

Both $\bar{S}$ and X_M can be determined from the analysis performed in the Chapter B-I on "Suspended Solids Deposition." Figure B-4 in that chapter shows an example plot of seabed deposition. For that example, an appropriate estimate of $\bar{S}$ is the average of the maximum and minimum values, or

$$\frac{100 + 5}{2} = 52 \text{ g/m}^2 \qquad \text{B-25}$$

The distance X_M, measured parallel to the coast and within the 5 g/m^2 contour, is 8,000 m.

The depth of water affected by the sediment oxygen demand is not really a constant value (as suggested by the previous formula) but varies as a function of the travel time across the zone of deposition. The affected depth H (in meters) is chosen to represent the average depth influenced by the sediment oxygen demand and can be estimated as:

$$H = 0.8 \left(\frac{\epsilon_z X_M}{U} \right)^{1/2} \qquad \text{B-26}$$

where:

ϵ_z = Vertical diffusion coefficient (cm^2/sec).

For the example case where U = 3 cm/sec, X_M = 8,000 m, and ϵ_z = 1 cm^2/sec,

$$H = 0.8 \times \left(\frac{1 \times 8,000 \times 100}{3} \right)^{1/2} \times \frac{1}{100} \text{ m} = 4.1 \text{ m} \qquad \text{B-27}$$

If the applicant desires to compute a value of vertical diffusivity, the following empirical expression can be used:

$$\epsilon_z = \frac{10^{-4}}{\frac{1}{\rho}\frac{d\rho}{dz}}$$

where:

ϵ_z = Vertical diffusion coefficient, cm^2/sec

ρ = Ambient water density, kg/m^3 (1,024)

$\frac{d\rho}{dz}$ = Ambient density gradient, kg/m^4.

-The density gradient used should reflect the most severe stratification condition that is likely to occur during the critical period.

The dilution D that is used in Equation B-24 can be found from Table B-5 where the field width is the width of the deposition area. For the appropriate travel time and field width, the smaller of the two estimates shown in the table should be used.

In Chapter B-I (Suspended Solids Deposition), the applicant is asked to compute the long-term accumulation and the critical 90-day accumulation. Because the critical 90-day accumulation might exceed the long-term average, the applicant should use the more critical case when predicting sediment oxygen demand.

Oxygen Demand Due to Resuspension of Sediments

It is more difficult to accurately predict oxygen demand due to resuspension than due to either farfield BOD decay or a steady sediment oxygen demand. To simplify the analysis, the approach here considers a worst-case situation. The amount of sediment to be resuspended is equal to the critical 90-day accumulation, which is found using the methods discussed in the above guidance on "Suspended Solids Deposition."

TABLE B-5. SUBSEQUENT DILUTIONS[a] FOR VARIOUS INITIAL FIELD WIDTHS AND TRAVEL TIMES

Travel Time (h)	Initial Field Width (ft)					
	10	50	100	500	1,000	5,000
0.5	2.3/5.5	1.5/2.0	1.3/1.6	1.0/1.1	1.0/1.0	1.0/1.0
1.0	3.1/13	2.0/3.9	1.6/2.6	1.2/1.3	1.1/1.1	1.0/1.0
2.0	4.3/32	2.7/8.5	2.2/5.1	1.4/1.9	1.2/1.5	1.0/1.0
4.0	6.1/85	3.7/21	3.0/11	1.9/3.5	1.5/2.3	1.1/1.2
8.0	8.5/>100	5.2/53	4.1/29	2.5/7.3	2.0/4.4	1.4/1.7
12	10/>100	6.3/95	5.1/50	3.0/12	2.4/6.8	1.6/2.3
24	15/>100	8.9/>100	7.1/100	4.2/30	3.4/16	2.1/4.4
48	21/>100	13/>100	10/>100	5.9/80	4.7/41	2.8/10
72	26/>100	15/>100	12/>100	7.3/>100	5.8/73	3.4/17
96	29/>100	18/>100	14/>100	8.4/>100	6.6/100	3.9/24

[a] The dilutions are entered in the table as N_1/N_2, where N_1 is the dilution assuming a constant diffusion coefficient, and N_2 is the dilution assuming the 4/3 law.

For the material to remain suspended, the ambient current speed has to be sufficiently great that the volume of water containing the resuspended material increases over time as ambient water is entrained. It is assumed that this process continues for up to 24 h.

The applicant should compute the oxygen depletion as a function of time during this period. This can be done using the following relationship:

$$\Delta DO = \frac{\overline{S}_r}{DH} \left[1 - \exp\left(\frac{-k_r t}{24}\right) \right] \qquad \text{B-29}$$

where:

ΔDO = Oxygen depletion, mg/L

$\overline{S}_r$ = Average concentration (in g/m^2) of resuspended organic sediment (based on 90-day accumulation)

H = Depth of water volume containing resuspended materials, m

k_r = Decay rate of resuspended sediments, 0.1/day

t = Elapsed time following resuspension, h (t varies from 0 to 24 h)

D = Dilution as defined previously (generally set equal to 1).

The variable H is a function of travel time and can be predicted from:

$$H = \frac{1.6}{100} (3,600 \, t \, \epsilon_z')^{1/2} \qquad \text{B-30}$$

where:

ϵ_z' = Vertical diffusion coefficient when resuspension is occurring (5 cm^2/sec)

t = Elapsed time following resuspension, h.

B-42

The applicant should check to be sure that H does not exceed the water depth. If it does, set H equal to the water depth.

The concentration of resuspended sediments $\bar{S}_r$ can be approximated as the average concentration over the width of the zone of deposition. This can be determined directly from the contour plots of sediment accumulation, developed in response to the guidance on "Suspended Solids Deposition" in Chapter B-I.

The applicant should calculate ΔDO for 3-h increments for a period of up to 24 h. The results can be tabulated as shown below. Data and calculations should be included in the application.

t (h)	DO (mg/L)
0	0
3	
6	
9	
12	
15	
18	
21	
24	predictions

Most often, a maximum depletion will occur somewhere in the 24-h period, with depletions decreasing for larger travel times.

B-V. SUSPENDED SOLIDS CONCENTRATION FOLLOWING INITIAL DILUTION

The concentration of suspended solids at the completion of initial dilution should be calculated using the following equation:

$$SS_f = SS_a + \frac{SS_e - SS_a}{S_a} \qquad\qquad \text{B-31}$$

where:

SS$_f$ = Suspended solids concentration at completion of initial dilution, mg/L

SS$_a$ = Affected ambient suspended solids concentration immediately upcurrent of the diffuser averaged over one-half the tidal period (12.5 h) and from the diffuser port depth to the trapping level, mg/L

SS$_e$ = Effluent suspended solids concentration, mg/L

S$_a$ = Initial dilution (flux-averaged).

The maximum change, ΔS, due to the effluent can be computed as follows:

$$\Delta S = SS_e/S_a \qquad\qquad \text{B-32}$$

where the terms are as defined above. Equation B-32 is appropriate as long as the effluent suspended solids concentration is much greater than the background concentration. During spring runoff in some estuaries, the background suspended solids concentration may exceed the effluent concentration. In these cases, the final suspended solids concentration will be below the background concentration.

U.S. EPA requires data for periods of maximum stratification and for other periods when discharge characteristics, oceanographic conditions, water quality, or biological seasons indicate more critical situations exist. The critical period generally occurs when water quality standards are most likely to be violated. If the standard is expressed as a maximum numerical limit, the critical period would be when the background concentrations are highest and the initial dilution is low. If the standard is expressed as a numerical difference from background, the critical period would be when effluent concentrations are high and initial dilution low. When the standard is expressed as a percent difference from background, the critical period could occur when background concentrations are low.

Because effluent suspended solids concentrations can vary with discharge flow rate, the concentration at the completion of initial dilution should be computed for the minimum, average dry- and wet-weather, and maximum flow rates, using the associated suspended solids concentration. The range and average effluent concentrations should be provided in the application by month, unless locally applicable standards require compliance over shorter durations. This information should be available from operating records.

The selection of an appropriate background suspended solids concentration may be difficult due to a general lack of data. A common problem for coastal sites is that measurements may be available only at the mouths of large rivers. Concentrations are often higher at such locations than farther offshore because of the solids contribution from runoff. Selected values of background suspended solids concentrations are shown in Table B-6. Suspended solids background data should be obtained at control stations, at the ZID boundary of the existing discharges, and at stations between the ZID-boundary and control stations. Data should be collected over the tidal cycle and at several depths so the average concentration over the height-of-rise of the plume over the tidal cycle can be calculated. This value should be used in Equation B-31.

TABLE B-6. SELECTED BACKGROUND SUSPENDED SOLIDS CONCENTRATIONS

Water Body	Suspended Solids Concentration, mg/L
Cook Inlet, AK	250-1,280
Southern California Bight	0.7-60
Pacific Ocean near San Francisco, CA	1-33
Broad Sound, MA	18.6-25.2
Massachusetts Bay near South Essex	1.2-30.5
New Bedford Harbor, MA	0.4-6.1
East River, NY	6.0-25.6
Ponce, PR (near shore)	13.5
Puget Sound, WA	0.5-2.0
Outer Commencement Bay, Tacoma, WA	33-51
Commencement Bay near Puyallup River, WA	23-136
Tacoma Narrows, WA	33-63

Note: Data are from 301(h) applications.

Compliance with the water quality standard can be determined directly if the standard is expressed in the form of suspended solids concentrations. If only a general standard exists, the maximum increase due to the effluent should be computed. If the increase is less than 10 percent, then no substantial effect in the water column is likely. However, seabed deposition could still be substantial depending on the mass emission rate of suspended solids and ambient currents at the discharge site, and thus should be evaluated.

The water quality standards may also specify limitations on the level of suspended solids removal. For example, California has a requirement that 75 percent of the solids entering POTWs must be removed. Compliance with this standard can be determined by estimating the average removal efficiency for each month based on the average monthly influent and effluent suspended solids concentrations. The removal efficiency should be equal to or greater than the required percentage in all months. The applicant should include the monthly average influent and effluent suspended solids concentrations along with the computed removal efficiencies.

B-VI. EFFLUENT pH AFTER INITIAL DILUTION

The calculation of effluent pH following initial dilution is chemically more sophisticated than other chemical calculations in this document. This appendix details the basis for Table 1 in the main text showing the ranges of probable effluent pH following initial dilution. The method for calculating effluent pH following initial dilution is described herein, assuming that all of the required variables are known. These variables include initial dilution and the temperature, salinity, pH, and alkalinity of the effluent and the receiving water. Effluent and receiving water temperature, salinity, and pH are normally measured. The (usually critical) initial dilution is routinely calculated as part of either the Section 301(h) waiver application process or the Section 301(h) permit renewal process. However, neither the alkalinity of the receiving water nor that of the effluent is usually measured. The alkalinity of seawater is relatively constant, however, at a value of 2.3 meq/L (Stumm and Morgan 1981). The alkalinity of effluent varies from 0.1 to 6.0 meq/L.

The method described herein predicts pH at the completion of initial dilution of an effluent-receiving water mixture. Because the initial dilution process occurs over a short time period, mixing is considered to occur in a closed system. Also, in stratified receiving waters, the wastewater plume is often trapped below the surface. Thus, the plume does not equilibrate with the atmosphere, and carbon dioxide exchange between the atmosphere and mixture is considered negligible. This method is useful for the calculation of pH, alkalinity, and total inorganic carbon concentration in the plume after initial dilution.

The pH of the effluent receiving water mixture is calculated using the equations for aqueous carbonate equilibrium in a closed system (Stumm and Morgan 1981). For this condition, the five equations that describe the relationships between pH, the carbonate species, and alkalinity are:

B-48

$$[H^+] \, [HCO_3^-]/[H_2CO_3^*] = K_1 \qquad\qquad \text{B-33}$$

$$[H^+] \, [CO_3^{2-}]/[HCO_3^-] = K_2 \qquad\qquad \text{B-34}$$

$$[H^+] \, [OH^-] = K_w \qquad\qquad \text{B-35}$$

$$C_T = [H_2CO_3^*] + [HCO_3^-] + [CO_3^{2-}] \qquad\qquad \text{B-36}$$

$$\text{Alkalinity} = [HCO_3^-] + 2[CO_3^{2-}] + [OH^-] - [H^+] \qquad\qquad \text{B-37}$$

where:

$[H_2CO_3^*]$ = The sum of aqueous CO_2 and true H_2CO_3 concentrations

C_T = Total carbonate concentration.

The carbonate species can also be expressed in terms of ionization fractions α_0, α_1, and α_2:

$$[H_2CO_3^*] = C_T \, \alpha_0 \qquad\qquad \text{B-38}$$

$$[HCO_3^-] = C_T \, \alpha_1 \qquad\qquad \text{B-39}$$

$$[CO_3^{2-}] = C_T \, \alpha_2 \qquad\qquad \text{B-40}$$

where:

$$\alpha_0 = \left[1 + \frac{K_1}{[H^+]} + \frac{K_1 K_2}{[H^+]^2} \right]^{-1} \qquad\qquad \text{B-41}$$

$$\alpha_1 = \left[\frac{[H^+]}{K_1} + 1 + \frac{K_2}{[H^+]} \right]^{-1} \qquad\qquad \text{B-42}$$

$$\alpha_2 = \left[\frac{[H^+]^2}{K_1 K_2} + \frac{[H^+]}{K_2} + 1 \right]^{-1} \qquad \text{B-43}$$

Substituting the hydroxide-hydrogen ion relationship and ionization fractions into the alkalinity equation yields:

$$\text{Alkalinity} = C_T (\alpha_1 + 2\alpha_2) + \frac{K_W}{[H^+]} - [H^+] \qquad \text{B-44}$$

Because total carbonate is conserved and α_1 and α_2 are functions solely of pH, the above equation has only one variable: hydrogen ion concentration. The model solves the equation to determine the pH of the effluent-receiving water mixture. The steps involved in the calculations are listed below:

- Determine input data

- Calculate ion product of water, K_W, and carbonate dissociation constants, K_1 and K_2, of the effluent and receiving water based on temperature and salinity data

- Check consistency between alkalinity and pH of both effluent and receiving water

- Calculate total carbonate in effluent and receiving water separately

- Calculate total carbonate, alkalinity, salinity, and temperature of the effluent-receiving water mixture following initial dilution (based on proportions of effluent and receiving water)

- Calculate K_W, K_1, and K_2 for the effluent-receiving water mixture following initial dilution

- Use a stepping procedure to find pH based on the computed values for total carbonate and alkalinity of the effluent-receiving water mixture

- Record results.

The ion product and dissociation constants are calculated for the appropriate temperature and salinity based on the equations given below. The equations for the receiving water have been revised so that salinity (in ppt) can be used.

For effluent:

$$pK_1 = \frac{3,407.7}{T} + 0.03279T - 14.8435 \text{ (Kelts and Hsu 1978, p. 300)} \qquad \text{B-45}$$

$$pK_2 = \frac{2,902.4}{T} + 0.02379T - 6.498 \text{ (Kelts and Hsu 1978, p. 300)} \qquad \text{B-46}$$

$$pK_w = \frac{4,471.0}{T} + 0.01706T - 6.0875 \text{ (Stumm and Morgan 1981, p. 127)} \qquad \text{B-47}$$

For receiving water and the effluent-receiving water mixture:

$$pK_1 = \frac{3,404.7}{T} + 0.03279T - 14.712 - 9.1575S^{1/3} \qquad \text{B-48}$$
$$\text{(Stumm and Morgan 1981, p. 205)}$$

$$pK_2 = \frac{2,902.4}{T} + 0.02379T - 6.471 - 0.3855S^{1/3} \qquad \text{B-49}$$
$$\text{(Stumm and Morgan 1981, p. 206)}$$

$$pK_w = \frac{3,441.0}{T} + 2.241 - 0.0925S^{1/2} \qquad \text{B-50}$$
$$\text{(Dickson and Riley 1979, p. 97)}$$

where:

T = Temperature in degrees Kelvin

S = Salinity in ppt.

The receiving water equations are valid for salinities down to about 10 ppt.

B-VII. LIGHT TRANSMITTANCE

Increased suspended solids concentrations associated with municipal discharges can cause a decrease in light penetration within the water column. Reductions in light penetration can result in a decrease in phytoplankton productivity as well as a reduction in the areal distribution of attached macroalgae such as kelp. Therefore, several states have enacted regulations governing the allowable levels of interference with light transmittance.

The evaluation of light transmittance may require the measurement of one or more water clarity variables and a comparison of values recorded in the vicinity of the outfall with those recorded in control areas. Variables that are widely measured to assess light transmittance include turbidity, Secchi disc depth, beam transmittance, and downward irradiance. While many of the state requirements are very specific in terms of the light transmittance measurements, others leave the selection of the sampling methods to the discretion of the applicant.

Turbidity is a measure of the optical clarity of water, and many standards are written in terms of Nephelometric Turbidity Units (NTU). Measurements are made with a nephelometer, which provides a comparison of the light-scattering characteristics of the sample with a standard reference. Differences in the optical design of nephelometers can cause differences in measured values even when calibrated against the same turbidity standard. For this reason, caution must be exercised when comparing measurements of turbidity made from different field sampling programs.

A Secchi disc is used to make visual observations of water clarity. Records of the depth at which the Secchi disc is just barely visible can be used to make comparisons of light transmittance among sampling sites. Measurements of Secchi disc depth are probably the most widely used means of estimating light penetration. The Secchi disc is easy to use, is accurate

over a wide range of conditions, and can be used to estimate the attenuation coefficients for collimated and diffuse light and, therefore, to estimate the depth of the euphotic zone. However, since a wastewater plume may be held below the upper regions of this zone during periods of stratification, Secchi disc measurements may not be appropriate under all conditions.

Beam transmittance is measured with a transmissometer and is a measure of the attenuation of a collimated beam of artificial light along a fixed path length (usually 1 m). The attenuation is caused by suspended and dissolved material as well as the water itself. These measurements, therefore, provide information about both the absorption and scattering properties of the water. The attenuation of a collimated beam of light in a water path is described by the Beer-Lambert law:

$$T_d = e^{-\alpha d} \qquad \qquad \text{B-51}$$

where:

T_d = The proportion of light transmitted along a path of length d, m

α = Light attenuation coefficient, m^{-1}.

Measurements of beam transmittance are made *in situ* at any depth.

The intensity and attenuation of daylight penetration are measured with an irradiance meter, which utilizes a photovoltaic cell to record incident light levels. Measurements are made just below the surface and at selected depth intervals throughout the water column so that light attenuation over specific depths can be determined. Unlike beam transmittance measurements, irradiance measurements are influenced by sunlight as well as surface conditions.

Empirical relationships can be derived among the light transmittance variables measured by these methods, which permits the estimation of one based on recorded values of another. These values can also be predicted

from suspended solids concentrations. The derivation of these relationships from existing data, in some instances, may be sufficient to allow for the demonstration of compliance with state standards. Existing data can also be used to predict the transparency characteristics in the vicinity of an improved discharge. Alternatively, a sampling program can be designed to assess compliance with light transmittance standards based on such empirical relationships.

Where standards are written in terms of maximum allowable turbidity or turbidity increase, predicted turbidity in the receiving water at the completion of initial dilution can be used to demonstrate compliance. By treating turbidity as a conservative variable, the turbidity in the receiving water at the completion of initial dilution can be predicted as:

$$T_f = T_a + \frac{T_e - T_a}{S_a} \qquad \text{B-52}$$

where:

T_f = Turbidity in receiving water at the completion of initial dilution, typically NTU or Jackson Turbidity Units (JTU)

T_a = Ambient or background turbidity

T_e = Effluent turbidity

S_a = Initial dilution.

Initial dilution can be predicted based on the methods presented in Appendix A. Equation B-52 can be used, then, to directly evaluate compliance with standards written in terms of maximum allowable turbidity or a turbidity increase.

Laboratory experimental work can also be used in lieu of field sampling to demonstrate compliance with standards written in terms of an allowable

turbidity increase. These analyses consist of determining the turbidity of a seawater-effluent mixture prepared in the same proportions corresponding to the predicted concentrations following initial dilution. Experiments should be conducted to simulate worst-case conditions. Simulations of expected receiving water turbidity should be made for periods of highest effluent turbidity (greatest suspended solids concentrations) as well as lowest initial dilutions. Values of the initial turbidity of the seawater, the effluent mixture, and the simulated dilution should accompany all test results.

By deriving a relationship between turbidity and Secchi depth and utilizing the method of prediction for turbidity in the receiving water following initial dilution (Equation B-52), compliance with state standards written in terms of Secchi depth can be evaluated. Secchi disc and turbidity can be related in the following manner. Assume that the extinction coefficient of visible light (α) is directly proportional to turbidity (T) and inversely proportional to Secchi disc (SD), or:

$$\alpha = k_1 \, T \qquad\qquad B\text{-}53$$

and

$$\alpha = \frac{k_2}{SD} \qquad\qquad B\text{-}54$$

where k_1 and k_2 are constants which need not be specified since they cancel out in further calculations. These two relationships have theoretical bases, as discussed in Austin (1974) and Graham (1966). Combining those two expressions, the relationship between Secchi disc and turbidity becomes:

$$T = \left(\frac{k_2}{k_1}\right)\left(\frac{1}{SD}\right) \qquad\qquad B\text{-}55$$

When state standards are written in terms of Secchi disc, it is convenient to combine Equations B-52 and B-55 to yield:

B-56

$$\frac{1}{SD_f} = \frac{1}{SD_a} + \frac{\frac{1}{SD_e} - \frac{1}{SD_a}}{S_a} \qquad \text{B-56}$$

or

$$SD_e = \left[\left(\frac{1}{SD_f} - \frac{1}{SD_a}\right) S_a + \frac{1}{SD_a}\right]^{-1} \qquad \text{B-57}$$

where:

SD_f = Minimum allowable Secchi disc reading in receiving water such that the water quality standard is not violated

SD_a = Ambient Secchi disc reading

S_a = Minimum initial dilution that occurs when the plume surfaces

SD_e = Critical Secchi disc depth of effluent.

In this manner, the critical effluent Secchi depth (SD_e) can be calculated. An effluent reading higher than this value indicates that standards will not be violated. This method of predicting the final Secchi depth in the receiving water can be utilized to provide an estimate of the effect of the wastewater discharge on the receiving water. This method should only be used where the standard is exclusively in terms of the acceptable decrease in the Secchi depth.

Values of the critical effluent Secchi depth (SD_e) calculated using Equation B-57 are presented in Table B-7. In this example, the water quality standard for the minimum Secchi visibility is 1 m (3.3 ft). Effluent having a Secchi depth greater than those presented for the selected ambient conditions and initial dilution will not violate the clarity standard of the example receiving water. Primary effluents typically have

TABLE B-7.   CALCULATED VALUES FOR THE CRITICAL EFFLUENT SECCHI DEPTH (cm)
FOR SELECTED AMBIENT SECCHI DEPTHS, INITIAL DILUTIONS, AND A WATER
QUALITY STANDARD FOR MINIMUM SECCHI DISC VISIBILITY OF 1 m

Initial Dilution	Ambient Secchi Depth (m)				
	2	3	4	5	10
10	18	14	13	12	11
20	10	7	7	6	6
40	5	4	3	3	3
60	3	2	2	2	2
100	2	1	1	1	1

Secchi disc values of 5-30 cm (2-12 in). For this case, with an initial dilution greater than 40 and an ambient Secchi depth of 2 m (6.6 ft) or greater, these calculations indicate that the standard would not be violated.

Since Secchi disc measurements are made from the water surface downward, critical conditions (in terms of the Secchi disc standard) will occur when the initial dilution is just sufficient to allow the plume to surface. It is notable that maximum turbidity or light transmittance impacts of a wastewater plume will occur when the water column is stratified, the plume remains submerged, and initial dilution is a minimum. Under these same conditions, however, Secchi disc readings might not be altered at all, if the plume is trapped below the water's surface at a depth exceeding the ambient Secchi disc depth.

The ability to relate measurements of turbidity to the attenuation coefficient (α) for collimated light has been demonstrated by Austin (1974). The attenuation coefficient can be expressed in terms of turbidity as:

$$\alpha = k \times JTU \qquad \text{B-58}$$

where:

JTU = Turbidity, JTU

k = Coefficient of proportionality.

Combining Equations B-51 and B-58, turbidity can be expressed as:

$$JTU = \frac{-\ln T_d}{kd} \qquad \text{B-59}$$

where:

T_d = Fraction of beam transmittance over distance d.

B-59

The coefficient of proportionality (k) takes on values 0.5-1.0. Therefore, to utilize these relationships for demonstrating compliance with a turbidity standard based on existing light transmittance data, the value of k must be determined empirically. This requires simultaneous measurements of beam transmittance and determination of turbidity covering the complete range of existing light transmittance records. If data are not available, the "k" value can be set equal to 1 as a conservative estimate.

Where a relationship between suspended solids concentration and beam transmittance data at a particular site can be derived, the suspended solids concentration at the completion of initial dilution from Equation B-31 can be used to predict compliance with standards written in terms of light transmittance.

B-VIII. OTHER WATER QUALITY VARIABLES

Other variables for which water quality standards may exist include total dissolved gases, coliform bacteria, chlorine residual, temperature, salinity, radioactivity, and nutrients. Variables concerned with aesthetic effects that also may be included are color, floating material, taste and odor, and hydrocarbons (i.e., grease and oil). For most dischargers, temperature, salinity, and radioactivity standards are unlikely to be violated. Aesthetic effects are more likely to occur when the plume surfaces and the dilution is low. Compliance with aesthetic standards can best be checked by field observations at the discharge site and along the shore.

TOTAL DISSOLVED GASES

Several states have a limit for total dissolved gases of 110 percent of saturation. Supersaturation of dissolved gases is not considered to be a likely problem for municipal wastewater discharges to the marine environment and is not discussed further.

CHLORINE RESIDUAL

Chlorine residual standards may be expressed as a concentration limit in the effluent or as a maximum concentration in the receiving water at the completion of initial dilution. If the effluent is not chlorinated, no further information is required. If the standard is expressed as an effluent limit, chlorine residual data from treatment plant operating reports, or other sources, should be presented in the application. If no data are available, then the procedure for chlorination, including the compound used, quantity, and occurrence of any operational problems, should be described. If the standard is expressed as a maximum limit at the

completion of initial dilution, the concentration in the receiving water, assuming the ambient concentration is 0.0 mg/L, can be estimated as follows:

$$Cl_f = Cl_e/S_a \qquad\qquad B-60$$

where:

Cl_f = Chlorine residual at completion of initial dilution, mg/L

Cl_e = Chlorine residual in effluent, mg/L

S_a = Lowest flux-averaged initial dilution.

As a worst-case approach, the maximum observed chlorine residual in the effluent should be used with the lowest dilution. If violations are predicted, the applicable water quality standard may require information on the frequency of occurrence.

NUTRIENTS

Standards can be expressed as maximum receiving water concentrations of total nitrogen or total phosphorus or as a general prohibition on amounts that would cause objectionable aquatic life. In general, for small discharges when the initial dilution is large, nutrients are not likely to cause problems. Appropriate state agencies should be contacted to ascertain if algal blooms, red tides, or other unusual biological activity have occurred near the discharge site in the past.

Receiving water and effluent nutrient data can be used to estimate concentrations at the completion of initial dilution. For screening purposes, the nutrients can be treated as conservative variables. The concentration is estimated as follows in a similar manner to suspended solids:

$$C_f = C_a + \frac{C_e - C_a}{S_a} \qquad \text{B-61}$$

where:

C_a = Affected ambient concentration immediately upcurrent of diffuser, mg/L

C_e = Effluent concentration, mg/L

S_a = Initial dilution (flux-averaged)

C_f = Concentration at the completion of initial dilution, mg/L.

The predicted concentration can then be compared to the state standard.

Because water quality criteria are often prescribed as maximum values not to be exceeded following initial dilution, it is useful to rearrange the above equation to express the maximum allowable effluent concentration as follows:

$$(C_e)_{max} = C_a + (S_a)_{min} (C_c - C_a) \qquad \text{B-62}$$

where:

$(C_e)_{max}$ = Maximum allowable effluent concentration such that water quality criteria are not exceeded

C_c = Applicable water quality criterion

$(S_a)_{min}$ = Minimum expected initial dilution.

The maximum observed effluent concentration can then be compared to the predicted allowable concentration. This approach can be used for any conservative constituent. Thus, if other specific limits exist such as

for color, effects due to the discharge can be determined as shown in Equations B-61 and B-62.

COLIFORM BACTERIA

Standards may exist for total or fecal coliform bacteria or enterococci and are usually expressed as a mean or median bacterial count and a maximum limit that cannot be exceeded by more than 10 percent of the samples. If the effluent is continuously disinfected using chlorination or an equivalent process, analyses for coliform bacteria may be needed only to verify the effectiveness of disinfection. If disinfection is done part of the year, analyses should be representative of conditions when the effluent is not so treated. The chemicals used, quantities, and frequency of use should be provided along with a discussion of the reliability of the system.

The coliform bacteria count at the completion of initial dilution due to the discharge can be estimated as follows:

$$B_f = B_e/S_a \qquad\qquad \text{B-63}$$

where:

B_e = Effluent coliform bacteria count, MPN/100 mL

S_a = Initial dilution.

As a conservative approach, the maximum effluent count and the lowest initial dilution should be used. If onshore currents occur only during a particular season, the coliform count at the completion of initial dilution can be estimated using the lowest initial dilution appropriate for that season. Effluent coliform data should be submitted to support the applicant's values. The predicted value can be compared with the appropriate standard at the ZID boundary. This value can also be used to estimate the bacterial concentration at specific locations away from the ZID.

Because different limits may apply to specific areas (e.g., shellfish harvesting areas, beaches, diving areas), the maximum bacterial count at a specified distance from the discharge may be of concern. This bacterial count can be estimated in a manner analogous to the estimation of the BOD exerted as the wastefield spreads out from the ZID. The maximum bacterial count at the centerline of the wastefield can be estimated as a function of distance from the discharge as follows:

$$B_x = B_a + \frac{B_f - B_a}{D_s D_b} \qquad \text{B-64}$$

where:

B_x = Bacteria count at distance x from ZID, #/100 mL

B_a = Affected ambient bacteria count immediately upcurrent of diffuser, #/100 mL

B_f = Bacteria count at completion of initial dilution, #/100 mL

D_s = Dilution attained subsequent to initial dilution at distance x

D_b = "Dilution" due to dieoff of bacteria caused by the combined effects of exposure to seawater and sunlight.

when x = 0, B_x = B_f. In cases where the background bacterial count is negligible or the effect of the discharge alone is desired, the terms for the ambient bacterial count can be dropped, simplifying Equation B-64 to:

$$B_x = \frac{B_f}{D_s D_b} \qquad \text{B-65}$$

Values for subsequent dilution as a function of $12\epsilon_0 t/B^2$ in Figure B-5. Guidance is included in Chapter B-III ("Farfield Dissolved Oxygen Demand")

on methods for estimating subsequent dilution for sites located in narrow estuaries or bays.

The decay rate of bacteria in the ocean is influenced by water temperature, incident light, salinity, and other factors. As a conservative estimate, the minimum decay rate should be used. If no violations would occur, then further calculations are not needed. Flocculation and sedimentation can cause an apparent decrease in coliform count in the water column, but the bacteria are retained in the sediment. Thus, this process is not included in the above approach. If the applicant has information indicating that the decay rate at the discharge site should be a different value, the revised decay rate may be used. The evidence for the revised decay rate, including any data or results of laboratory tests, should be included in the application.

In this report, dieoff due to the combined effects of exposure to saltwater and exposure to sunlight only are considered. The dieoff due to exposure to saltwater, D_{sw}, and the dieoff due to exposure to sunlight, D_{sl}, are (Gameson and Gould 1975):

$$D_{sw} = \exp(k_{sw}t) \qquad\qquad \text{B-66}$$

$$D_{sl} = \exp[\alpha I(t)] \qquad\qquad \text{B-67}$$

where:

k_{sw} = Bacteria decay rate due to exposure to saltwater, 1/h

α = Constant, m^2/MJ

$I(t)$ = Total intensity of sunlight received by bacteria during the travel time, MJ/m^2

t = Travel time, h.

The bacteria dieoff due to the combined effects of saltwater and sunlight is $D_b = D_{sw}D_{sl}$. Gameson and Gould (1975) indicate that $\alpha = 1.24 \ m^2/MJ$ *in situ* for Dorset, England seawater. The total intensity of sunlight received at the water surface can be measured, or estimated using site-specific data or general methods (Wallace and Hobbs 1977). If the wastefield is submerged, then the calculation of the total sunlight received should reflect the effect of turbidity on light transmission from the sea surface to the top of the wastefield.

The bacteria decay rate due to the exposure to saltwater is known for both coliform bacteria and enterococcus bacteria. For coliform bacteria,

$$k_{sw} = 2.303 \ exp[(0.0295T - 2.292)2.303] \ / \ h \qquad \text{B-68}$$

where T = water temperature (^{o} C), based on field measurements at Bridport (Dorset, England) (Gameson and Gould 1975). The enterococcus bacteria dieoff rate due to exposure to saltwater is:

$$k_{sw} = 0.5262 \ / \ (24 \ h) \qquad \text{B-69}$$

at a temperature of 20° C (Hanes and Fragala 1967). [It should be noted that Hanes and Fragala (1967) determined that k_{sw} for coliform bacteria is 0.0424/h at 20° C, a value slightly smaller than the value of 0.0457/h at 20° C based on the formula from Gameson and Gould (1975).]

The estimated coliform count at the location of interest should be compared to the applicable standard. If a violation is predicted, the water quality standards may require that the approximate frequency should be discussed based on the percentage or likelihood of currents transporting the wastefield in the direction of interest.

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APPENDIX C

BIOLOGICAL ASSESSMENT

APPENDIX C
BIOLOGICAL ASSESSMENT

Because benthic infauna are sedentary and must adapt to pollutant stresses or perish, this assemblage is often used to define the spatial extent and magnitude of biological impacts in the vicinity of sewage discharges. The general changes in benthic community structure and function that occur under conditions of organic enrichment of the sediments (e.g., due to municipal sewage effluent) have been well documented (Pearson and Rosenberg 1978). Slight to moderate enrichment results in slight increases in numbers of species, abundances, and biomass of benthic communities (see Figure 3 in main text), while species composition remains unchanged. As enrichment increases, numbers of species decline because less tolerant species are eliminated. The total abundance of organisms increases as a few species adapted to disturbed environments or organically enriched sediments become very abundant. When enrichment levels are optimal for those few species, they become extremely abundant and overwhelmingly dominate the benthic community (corresponding to the "peak of opportunists" shown in Figure 3). Biomass generally decreases, however, because many of those opportunistic species are small. Further organic enrichment of the sediments drastically reduces the number of species and abundances of benthic organisms, as conditions become intolerable for most taxa.

Because the model developed by Pearson and Rosenberg (1978) has been shown to be valid in many benthic environments, it is often instructive to examine the abundances of species that the authors identify as opportunistic or pollution-tolerant. Those data, in conjunction with the applicant's data on numbers of species, total abundances, and biomass at stations in the vicinity of the outfall, are often sufficient to determine the relative degree of impact within and beyond the ZID.

Comparable models that describe changes in the structure and function of plankton and demersal fish communities in organically enriched receiving environments have not yet been developed. However, it may be instructive to examine the scientific literature that is available for the biogeographic region in which the outfall is located. That literature often contains information describing the responses of the local fauna and flora to organic materials and other pollutants, and identifying opportunistic and pollution-tolerant species. Such information is extremely useful for interpreting data collected in the vicinity of the outfall.

A variety of analytical tools may be used to conduct biological comparisons for Section 301(h) applications. Applicants may analyze the data graphically or statistically, or may use other mathematical tools such as multivariate analyses (e.g., classification and ordination procedures). Graphical analyses can be especially useful for presenting data in an easily understood format. In Figure C-1, data on numbers of species in each replicate sample at stations in the vicinity of an outfall have been plotted to show the range of reference values in comparison with values at within-ZID, ZID-boundary, nearfield, and farfield stations. These data may be tested statistically to determine those test stations at which mean values differ from mean values at either or both reference stations. But even without such tests, the data in Figure C-1 clearly indicate that a gradient of effects occurs near the outfall. Relative to reference conditions, numbers of species are depressed at the within-ZID and downcurrent ZID-boundary stations, and may be depressed at the nearfield and upcurrent ZID-boundary stations.

Graphical analyses are especially useful for presenting data on the physical characteristics of the habitat. For example, it is often instructive to plot water column or substrate characteristics in relation to distance from the outfall (see Figures C-2 and C-3). Gradients of effects (as in Figure C-3) are often revealed in such simple presentations. An especially useful method for presenting data on sediment grain size distributions that has proven useful in analyses of 301(h) data was developed by Shepard (1954). Sediments are classified by the proportions of
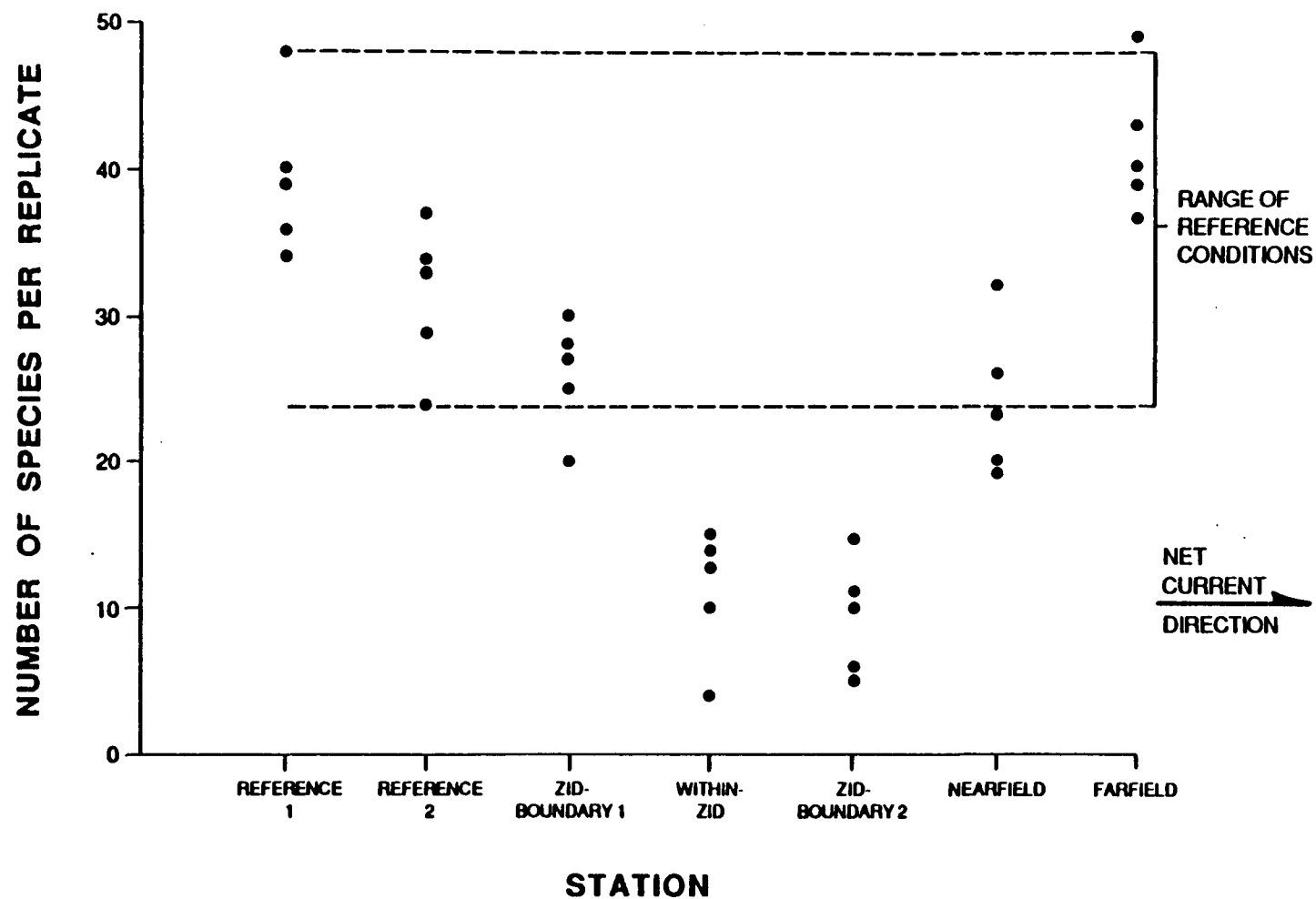
Figure C-1. Numbers of species collected in replicate benthic grab samples at stations in the vicinity of the outfall.
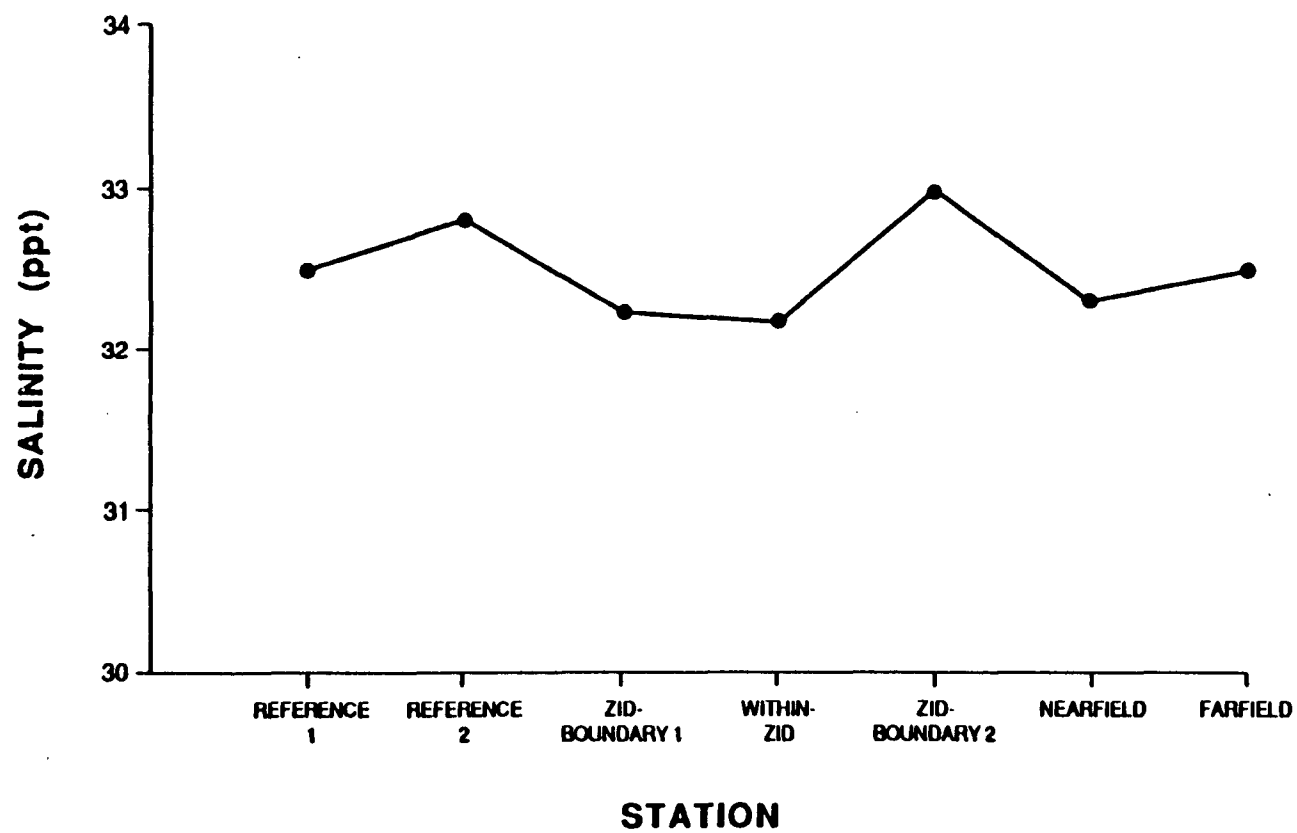
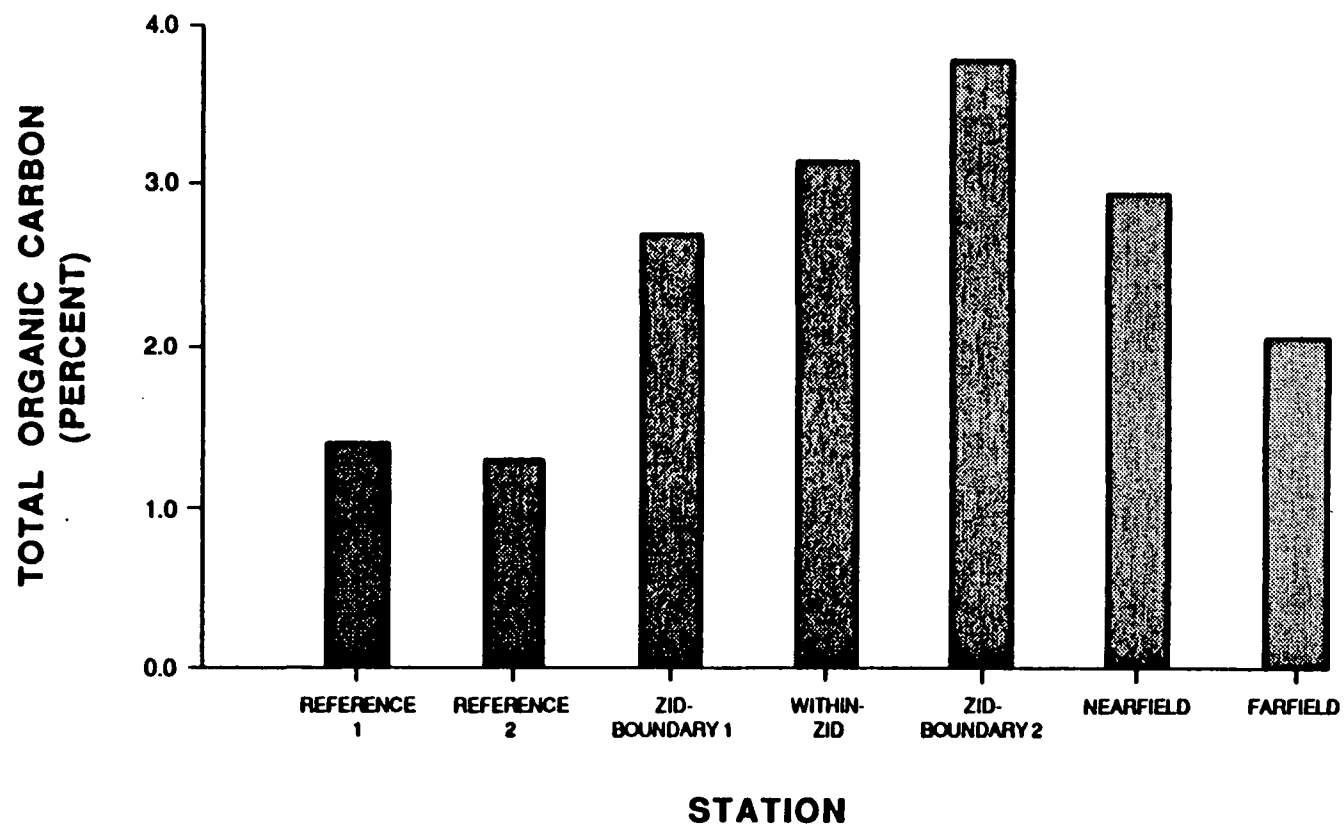Figure C-2. Salinity at stations in the vicinity of the outfall.

Figure C-3. Total organic carbon content of the sediments at stations in the vicinity of the outfall.

their three major grain-size categories (Figure C-4)
. Sand, silt, and clay are often the most useful categories. However, the gravel, sand, and mud (silt plus clay) categories are useful where sediments are relatively coarse. [See Shepard (1963) for information on sediment grain size scales.]

Statistical tests are among the most effective tools for comparing biological communities among stations. A variety of statistical tests are available, the most widely used of which is one way analysis of variance (ANOVA). ANOVA and other statistical tests have been used extensively for biological comparisons in the 301(h) program, but they have often been used improperly. For this reason, procedures for conducting statistical com- parisons using biological data are discussed briefly below. Applicants are encouraged to consult references on biostatistics (e.g., Zar 1974; Sokal and Rohlf 1981) for more specific guidance on the application of these pro- cedures.

The use of one way ANOVA for biological comparisons is preferred because ANOVA is an efficient and robust test. ANOVA compares the mean values of a given variable among stations (or groups of stations) for the purpose of detecting significant differences at a predetermined probability level. ANOVA requires a minimum of three replicate values at each station to estimate the mean value and associated variance.

ANOVA is a parametric test based on three assumptions: the error of an estimate is a random normal variate, the data are normally distributed, and the data exhibit homogeneous variances. Corrections for the first are not easily achieved, and an erroneous assumption can greatly affect the results of the test. Fortunately, error estimates in survey data are usually independent.

ANOVA is relatively robust with respect to the assumption that the data are normally distributed. Substantial departures from normality can occur before the value of the F-statistic is affected greatly (Green 1979). For
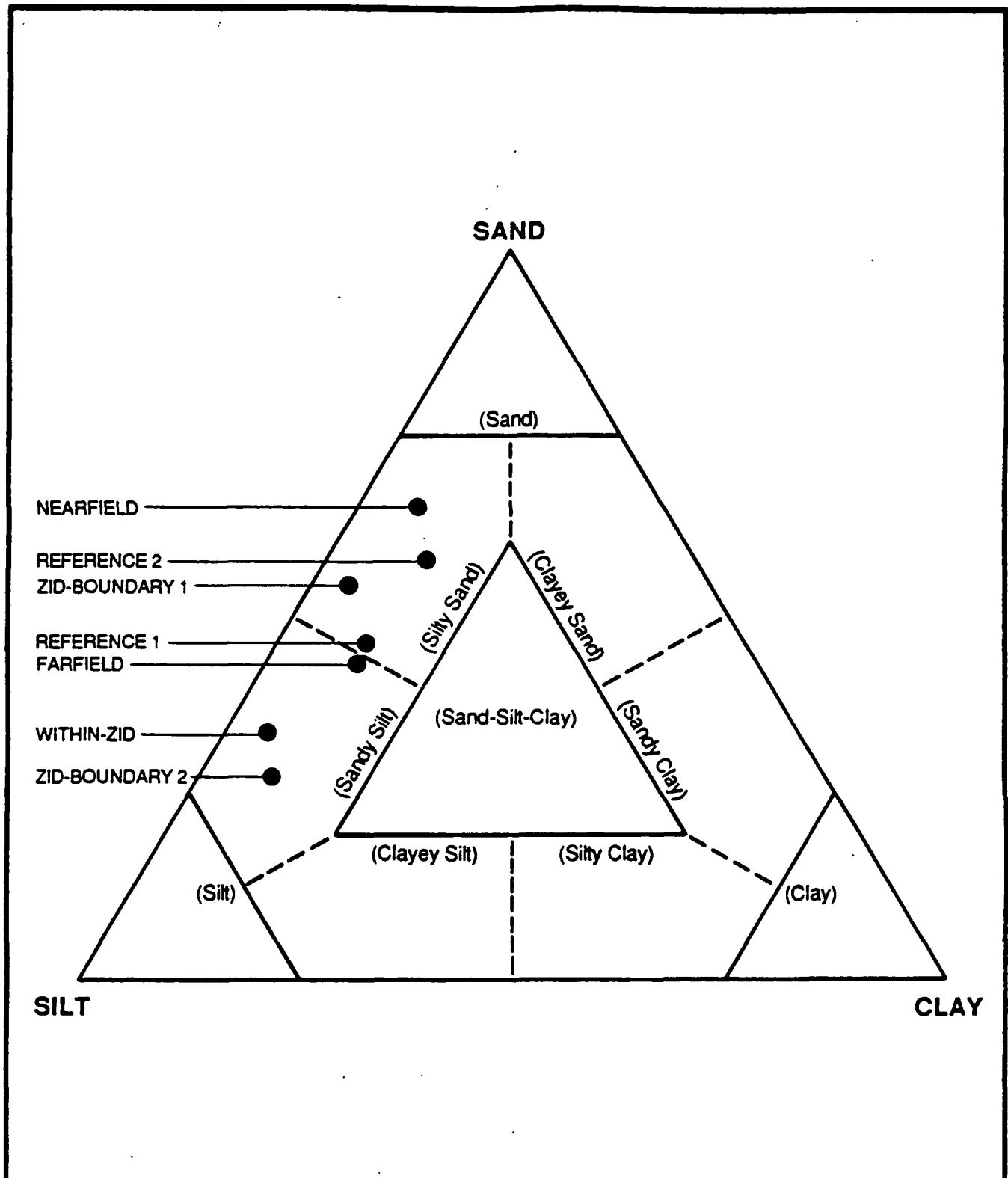
Figure C-4. Sediment grain size characteristics at stations in the vicinity of the outfall.

this reason, tests for normality are not usually conducted before data are analyzed using ANOVA.

The third assumption, that variances are homogeneous, is critical to execution of ANOVA. Heterogeneous variances can greatly affect the value of the F-statistic, especially in cases where the statistical design is unbalanced (i.e., where numbers of replicate values vary among the stations or station groups being tested).

Several tests are available to determine whether variances are homogeneous. The Fmax test (see Zar 1974; Sokal and Rohlf 1981) and Cochran's C test (Winer 1971) are both appropriate, although the latter is preferred because it uses more of the information in the data set. Bartlett's test is not recommended because it is overly sensitive to departures from normality (Sokal and Rohlf 1981).

When sample variances are found to differ significantly ($P<0.01$), a transformation should be applied to the data. [A more conservative probability level (e.g., $P<0.05$) should be used when the statistical design is unbalanced. ANOVA is sensitive to unbalanced statistical designs.] Sokal and Rohlf (1981) describe several transformations that may be used. Because ANOVA on transformed data is usually a more efficient test for detecting departures from the null hypothesis than is the Kruskal-Wallis test (the nonparametric analog of ANOVA), the Kruskal-Wallis test should only be used when the appropriate transformation fails to correct for heterogeneous variances (Sokal and Rohlf 1981). The Kruskal-Wallis test requires a minimum of five replicate values per station because it is a test of ranks.

When ANOVA or a Kruskal-Wallis tests are performed, significant differences ($P<0.05$) among individual stations or groups of stations may be determined using the appropriate *a posteriori* comparison. Of most importance in 301(h) demonstrations are differences among reference stations and stations within the ZID, at the ZID boundary, and beyond the ZID. It is primarily these comparisons upon which determination of the presence or absence of a balanced indigenous population is based.

Classification analyses (e.g., cluster analyses) have also been used extensively in the 301(h) program. In the normal classification mode, stations are grouped by the attributes of the assemblages that occur there (e.g., species composition and abundance). This type of analysis is very useful for identifying the stations that are the most similar and least similar to one another in fauna and/or flora. Because biological communities respond to organic materials and other pollutants, stations at which pollutant impacts are occurring typically cluster together in interpretable groups. Inverse classification analysis, in which taxa are grouped by the stations at which they co-occur, is also helpful because it defines assemblages that are characteristic of different levels and types of pollutant impacts.

Classification analysis involves two analytical steps: calculation of a matrix of similarity values for all possible station pairs, and grouping of stations based on those between-station similarity values. Many similarity indices and clustering strategies are available to perform these two tasks (see Boesch 1977; Green 1979; Gauch 1982; Pielou 1984; Romesburg 1984). However, only the Bray-Curtis similarity index and either the group average clustering strategy (i.e., the unweighted pair-group method using arithmetic averages) or the flexible sorting strategy have been used commonly in 301(h) demonstrations. Their continued use is recommended. The Bray-Curtis index is easily understood, and has been used widely in ecological studies. Moreover, two comparisons of similarity indices (i.e., Bloom 1981; Hruby 1987) have shown it to be superior to many of the other commonly used resemblance measures. Both the group average clustering strategy and the flexible sorting strategy are recommended because they produce little distortion of the original similarity matrix. [See Tetra Tech (1985) for additional rationale on the use of these three indices.]

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APPENDIX D

NAVIGATIONAL REQUIREMENTS AND METHODS

CONTENTS

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APPENDIX D
NAVIGATIONAL REQUIREMENTS AND METHODS

Information presented below addresses navigational requirements and methods for Section 301(h) dischargers. It is summarizes more detailed discussions in Tetra Tech (1987, 1988).

MONITORING STATION LOCATIONS

Compliance with conditions of a secondary treatment variance requires monitoring at a site-specific array of sampling locations. The types of stations usually specified in 301(h) monitoring programs are depicted in Figure D-1. Positioning accuracy is most critical for the within-ZID and ZID-boundary stations (Stations Z_0, Z_1, Z_2 in Figure D-1). Applicants must be able to sample at a specific boundary location on any given occasion, and to return to nearly the same location on subsequent trips. At gradient (G_1, G_2, G_3, G_4) and control or reference (C_1) stations, initial accurate location is not as critical. However, it is important to relocate these stations accurately during subsequent surveys to enable quantification of temporal changes in the variables sampled (e.g., benthic community characteristics). This requirement for high repeatable accuracy also applies to stations in or near special habitats (H_1, H_2). The ability to conduct sampling at the appropriate depth contour is also very important. Sampling programs for 301(h) typically include requirements that a bottom sampling station can be relocated to within 10 m (32.8 ft).

ACCURACY LIMITATIONS

Both the procedures and equipment used to establish a navigational position contribute errors that affect the overall accuracy of a fix. Absolute or predictable accuracy is a measure of nearness to which a system can define a position by latitude and longitude (Bowditch 1984). Repeatable
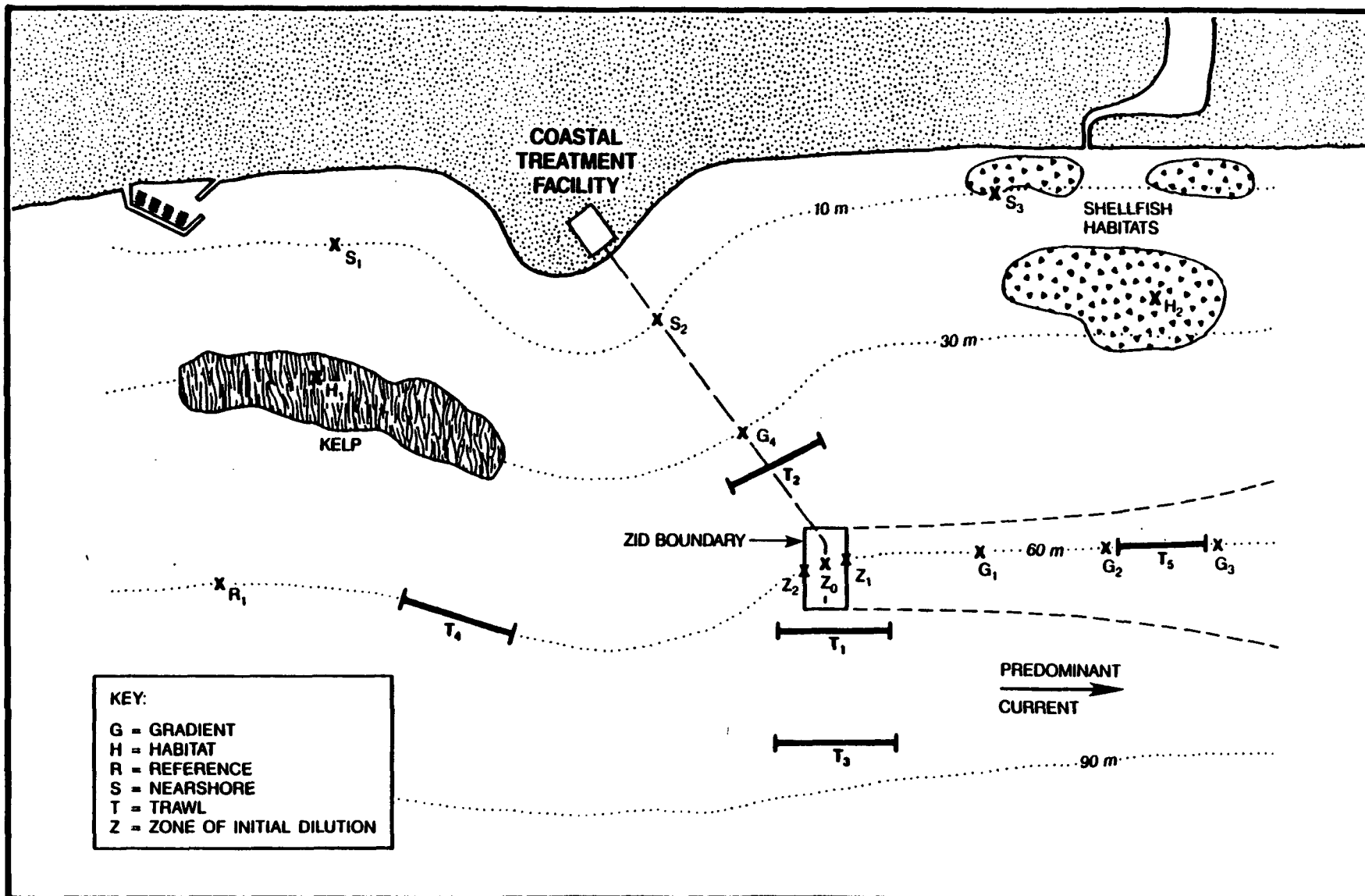
Figure D-1. Examples of some key 301(h) monitoring station locations for a medium-large marine municipal discharge.

or relative accuracy is a measure of a system's ability to return the user to a given position with coordinates that were previously measured with the same system. The difference between these two accuracies can be substantial. For example, depending on one's location in the coverage area, Loran-C has a repeatable accuracy in offshore areas of 15-90 m (49-295 ft), but an absolute accuracy of 185-463 m (607-1,519 ft) (Dungan 1979). In many instances, repeatable accuracy is more important than absolute accuracy (e.g., retrieval of crab pots, return to desirable fishing grounds, avoidance of underwater obstructions, and reoccupation of reference stations).

For coastal outfall monitoring, both repeatable and absolute accuracy can be important, depending on the type of sampling site. For within-ZID and ZID-boundary stations, both accuracies are important because sampling stations must be located within or very near the boundary and be repeatedly occupied during the program. For gradient, special habitat, and reference stations, repeatable accuracy is more important than absolute geographic location. Once such a station is established within a special habitat, it is often necessary to return to the same site to identify temporal variations in the previously sampled biological community. Thus, it is important to select navigational procedures and equipment with both the absolute and repeatable accuracies needed to meet the monitoring program objectives.

Because repeatable accuracy of navigational equipment is usually at least 1 order of magnitude better than absolute accuracy, the latter frequently limits the overall positioning accuracy of a sampling vessel during coastal monitoring programs. Therefore, the following discussion focuses on absolute accuracies that can be achieved by various procedures and associated equipment.

Practical considerations also limit the accuracy of an offshore positional fix. Resolution of a position to better than 1-2 m (3.3-6.6 ft) becomes meaningless when measuring the location of a moving vessel (e.g., during trawling) or a vessel that is on station but pitching and rolling.

Antenna movement alone usually precludes higher resolution in position coordinates. Exceptions to this rule can occur when conditions are unusually calm.

POSITIONING ERROR

Many factors contribute to the total error in position of the water column or benthic sampling point. These factors include movement or drift of the "on-station" vessel, offsets between the deployment point of sampling equipment and the navigational system antenna, and offsets between the deployment point and the subsurface location of the sampling or profiling equipment, and error in the ship's initial location. Most of these factors are site- or operationally specific, and can be estimated with varying degrees of confidence. Because the accuracy to which the actual sampling point is known is highly dependent on all these factors, they should be carefully considered in both the design and conduct of monitoring programs.

A ZID-boundary error proportional to some percentage of the ZID dimension has been selected as the controlling parameter for 301(h) navigational requirements. Because ZID size is proportional to water depth, the allowable error in position is thus also proportional to depth. For example, ZID-boundary stations can be located at a distance from the diffuser axis equal to one-half the ZID width plus 20 percent of the water depth at mean tide level. The allowable maximum error in the location of these stations can then be ±20 percent of the water depth. As a result, the closest to the diffuser that sampling would occur is at the ZID boundary, and the farthest from the diffuser that sampling would occur is 40 percent of the water depth beyond this boundary. Nominally, however, sampling would be performed within a distance from the ZID boundary equal to 20 percent of the water depth. Example ZID-boundary station locations using this approach for a variety of ZID sizes are listed in Table D-1. The ZID-boundary and sampling station locations for discharges at the 100-, 60-, and 20-m (328-, 197-, and 66-ft) depths are shown in Figure D-2.

TABLE D-1. EXAMPLE ZID-BOUNDARY STATION LOCATIONS

Average Diffuser Depth (m)	Average Diffuser Diameter (m)	ZID Width (m)	Recommended Station Location[a] (m)	Recommended Allowable Error[b] (m)
100	4.0	204.0	122.0	±20
90	3.6	183.6	109.8	±18
80	3.4	163.4	87.7	±16
70	3.2	143.2	85.6	±14
60	3.0	123.0	73.5	±12
50	2.5	102.5	61.3	±10
40	2.2	82.2	49.1	±8
30	2.0	62.0	37.0	±6
20	1.8	41.8	24.9	±4
15	1.5	31.5	18.8	±3
10	1.5	21.5	13.8	±3
5	1.0	11.0	8.5	±3
3	0.5	6.5	6.3	±3

[a] Distance from the zone of initial dilution centerline to the station, based on 0.5 times the ZID width plus 20 percent of the average water depth of the diffuser when over 15 m (49 ft).

[b] Error magnitude is equal to ±20 percent of the average diffuser depth, when over 15 m (49 ft).
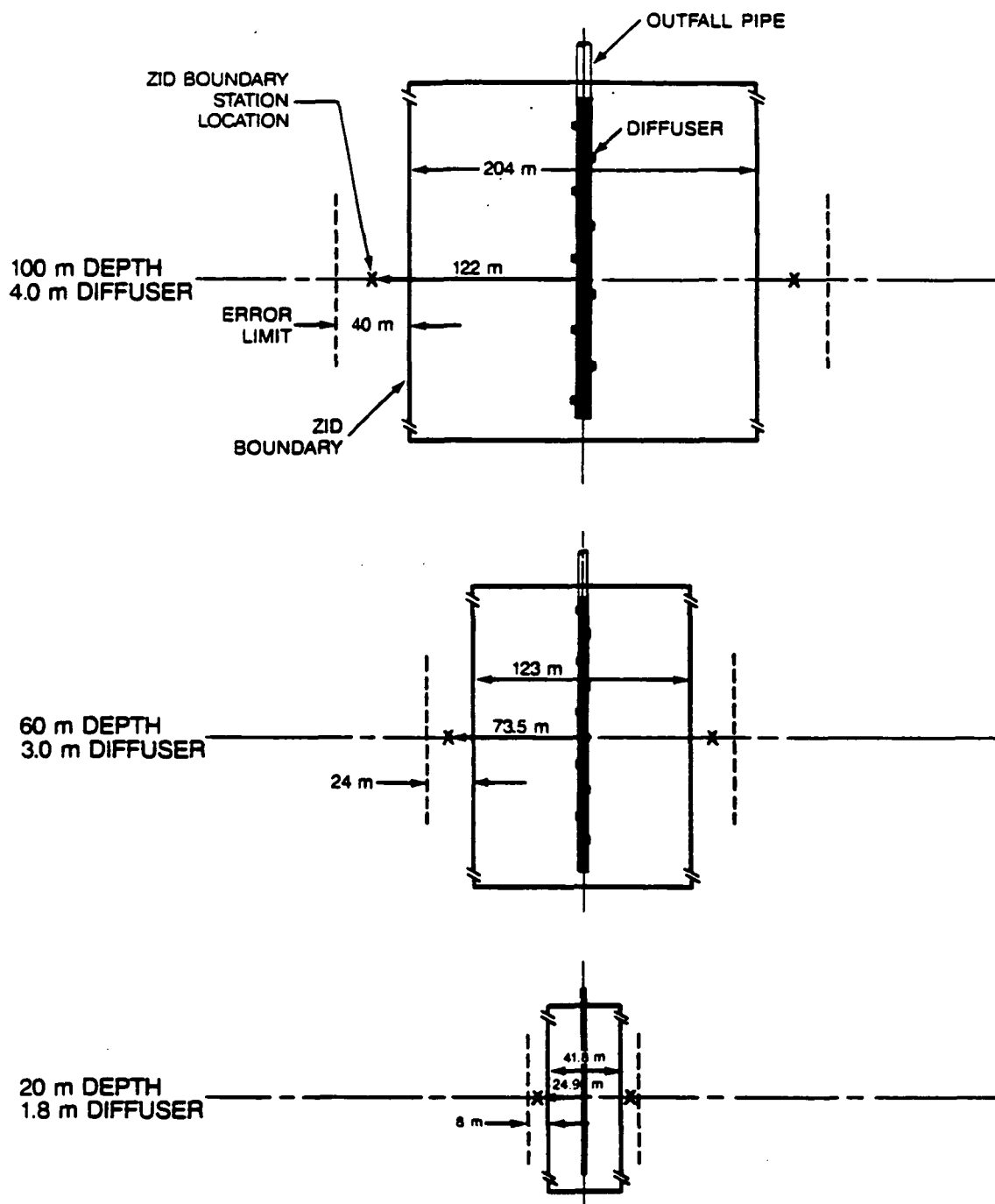
Figure D-2. Locations of ZID-boundary stations for selected ZID sizes.

When discharge depths are less than approximately 15 m (49 ft), the 20 percent error allowance results in an overly restrictive positional error [i.e., less than ±3 m (9.8 ft)]. Therefore, a positioning error of ±3 m (9.8 ft) is considered to be more appropriate when sampling station depths are less than 15 m (49 ft). Although the percent error as a function of water depth increases at shallower depths, this minimum error is considered reasonable given available navigating techniques for small sampling vessels in other than extremely calm waters. Stations beyond the ZID may be similarly located using the 20 percent of depth rule beyond the 15-m (49-ft) contour and the ±3-m (9.8 ft) error limitation for shallower locations. As indicated earlier, it is recognized that the ability to reoccupy a given site can be as important as knowing its exact geographical location. However, relocation beyond the ZID probably will not be a problem if the same navigational equipment used to locate ZID-boundary stations is also used elsewhere.

SUMMARY OF RECOMMENDED PROCEDURES AND EQUIPMENT

Based on Tetra Tech's evaluation of optional positioning methods, the systems recommended for coastal positioning include theodolites, sextants, electronic distance measuring instruments (EDMIs), total stations, and microwave and range-azimuth systems. Although satellite systems offer adequate accuracy (when used in a differential mode), their use may be limited because a sufficient number of satellites may not always be available.

CANDIDATE SYSTEM SELECTION

The details of positioning techniques and associated equipment are described in Tetra Tech (1987). No single system is best for all coastal monitoring purposes. Needs vary according to the size and complexity of the planned monitoring program, the nature of the immediate and surrounding areas, and other navigational or surveying requirements of a municipality.

Positioning techniques fall into three principal measurement categories:

■   Multiple horizontal angles

        Theodolite intersection

        Sextant angle resection

■   Multiple electronic ranges

        Distance-measuring instruments

        Range-range mode

        Hyperbolic mode

        Satellite ranging

■   Range and angle

        Theodolite and EDMI

        Total station

        Range-azimuth navigation systems

Systems within these categories that will meet or exceed the positional accuracy recommended herein are summarized in Table D-2. Additional information on the recommended categories is provided below.

Multiple Horizontal Angles

    In the multiple horizontal angles category, theodolites were found to have the angular accuracies required for the maximum ranges anticipated. Their costs range from $1,000 to $4,000 (30-sec vs. 10-sec accuracy), and

TABLE D-2. SUMMARY OF RECOMMENDED SYSTEMS

Category	Representative[a] Equipment	Accuracy	Cost	Advantages	Disadvantages
Theodolite	Table B-1 Table B-2	10-30 sec ±1 m (3.3 ft)	$1,000-$4,000	Traditional method. Inexpensive. High accuracy. Successfully applied. Restricted areas.	Line-of-sight. Two manned shore stations. Simultaneous measurements. Limits on intersection angles. Area coverage; station movement.
Sextant	Table B-3	±10 sec ±2 m (6.6 ft)	$1,000-$2,000	Rapid. Easy to implement. Most widely used. Low cost. No shore party. High accuracy.	Simultaneous measurement of two angles. Target visibilities, location, maintenance. Line-of-sight. Best in calm conditions. Limits on acceptable angles.
EDMI	Table B-4	1.5-3.0 cm	$3,500-$15,000	Extremely accurate. Usable for other surveying projects. Cost. Compact, portable, rugged.	Motion and directionality of reflectors. Visibility, unless microwave. Two shore stations. Ground wave reflection.
Total stations	Table B-5	5-7 cm	$8,000-$30,000	Single onshore station. Other uses. Minimum logistics.	Reflector movement and directionality. Prism costs.
Microwave navigation systems	Table B-6	±1-3 m	$40,000-$90,000	No visibility restrictions. Multiple users. Highly accurate. Radio line-of-sight.	Cost. Multiple onshore stations. Logistics. Security.
Range-azimuth systems	Table B-10	0.01° and 0.5 m	$65,000-$100,000	High accuracy. Single station. Circular coverage.	Single user. Cost.
Satellite systems	Table B-9	1-10 m	$150,000-$300,000 (initial units)	High accuracy. Minimum logistics. Use in restricted/congested areas. Future cost. No shore stations.	Current coverage. Initial development cost.

[a] Table references refer to Tetra Tech (1987).

D-9

they are readily available because they are widely used as a surveying instrument. At least two theodolites, two operators, a vessel siting target, and a three-way communications link to coordinate fixes are required. Visibility can be a limiting factor.

By comparison, sextant angle resection can be performed using one instrument if the vessel is stationary, or using two instruments simultaneously if the vessel is moving. Achievable angular accuracy of $\pm$10 sec is adequate, and relatively inexpensive sextants ($1,000-$2,000) are readily available. Again, visible range can be limiting. Shooting an accurate fix from a non-stationary platform in any significant sea or swell could be more difficult than shooting with theodolites from shore. A distinct advantage of sextant angle resection is location of the navigators on the survey vessel. The method generally requires highly visible shore targets and a three-arm protractor for plotting positions.

Multiple Electronic Ranges

Positioning using multiple ranges can be accomplished with two staffed EDMI stations. Accuracies were found to be more than adequate but ranges were found to be marginal [if needed beyond 3 km (1.9 mi)] unless multiple prisms are used. Because such prisms are directional, procurement of multiple clusters for more than one direction could result in substantial additional costs. The initial investment (i.e., without multiple prisms) is $3,500-$5,000 each for two shorter-range units, or $8,000-$15,000 each for two longer-range units. Several microwave navigation systems with more than adequate range and sufficient accuracy are available in the $40,000-$90,000 range. Limitations include geometry of shore stations; position of the vessel in the coverage area (i.e., crossing angle limitations); and possible interferences due to line-of-sight obstructions, sea-surface reflective nulls, and land-sea boundaries. The hyperbolic mode provides multiple user capability, but at the cost of an additional shore station.

Satellite ranging holds promise because required accuracies should be achievable in the near future. Transit satellite-based systems do not offer

sufficient accuracy, except with multiple passes, and multiple passes are impractical when a given sampling station is occupied only briefly. Accuracies needed will undoubtedly be achievable in the future using differential global positioning system (GPS) techniques ($10,000-$40,000 for first units; as low as $1,000 for subsequent production models). Commercial geosynchronous satellite networks, such as GEOSTAR, may become available at a proposed system interrogator cost of $450 plus a monthly fee. However, this system is in the very early stages of planning, having only recently received FCC approval of requested frequencies. Finally, the codeless GPS systems (SERIES or Aero Services Marine GPS System) currently under development could be used, but at a current cost of over $250,000.

Range and Angle

Systems in the range-azimuth category show great promise. Required angular and range accuracies are available, only one shore station is needed, and costs depend on system refinements. At the low end of the scale, an EDMI and theodolite could be paired with a communication link for approximately $10,000-$12,000. Total stations developed specifically for this requirement range in cost from $8,000 for a manual station to $15,000-$30,000 for a fully automatic station. Optical and infrared range limitations apply to these systems. The three range-azimuth navigational systems examined provide sufficient positional accuracy with a single station at costs ranging from $65,000 for manual tracking to $70,000-$100,000 for fully automatic tracking.

SHALLOW-WATER POSITIONING METHODS

When sampling stations are located in relatively shallow water, they can be identified by relatively inexpensive methods (in addition to those discussed earlier in this report). Provided the center of the ZID over the outfall can be located (e.g., by diver-positioned surface float), an optical range finder may be used to establish the required distances to nearby water quality or biological sampling stations. An optical range finder is used by simply focusing a split-image on the target float, enabling the slant

distances to the target to be read from the instrument scale. When combined with a careful compass reading, this distance reading allows positioning of the sampling vessel.

A survey of accuracies claimed for commercially available instruments suggests that the ±3 m (9.8 ft) recommended minimum accuracy can be achieved for ranges up to approximately 100 m (328 ft) from the surface target. The Lietz Model 1200, for example, provides an accuracy of ±1 m (3.3 ft) at 100 m (328 ft). Beyond this distance, instrumental errors increase rapidly. For the instrument cited, a ±9 m (29.5 ft) accuracy is quoted at 300 m (984 ft). The suggested U.S. list prices of optical range finders vary from $35 to $120 (Folk, L., 21 March 1985, personal communication).

An acceptable alternative method for collecting bottom samples from desired locations in shallow water is to use divers. Provided visibility is adequate, divers may measure radial distances to desired locations by holding a tape at the outfall and traversing the appropriate distance over the bottom in the proper direction.

Visual ranges have sometimes been used to establish a station position. This method requires that a minimum of two objects are in alignment, enabling the vessel to be placed on a common axis extending to the vessel's position. Simultaneous siting on a second set of at least two objects places the vessel at the intersection of the two common axes. The accuracy of each visual range is highly dependent on the quality of the visual range (e.g., alignment), the distance from the alignment objects to the vessel, and the angle between each range. Also, the number of visual ranges used affects the magnitude of the positional error. Although this technique is frequently used for positioning single sampling stations in bays, harbors, and other areas in which two or more conveniently alignable targets can be selected, the method is not considered acceptable for coastal monitoring at ZID-boundary stations. Also, it is not likely that a sufficient number of alignment target-pairs will be present for all desired locations. In addition, the unpredictability of repeatable position error detracts from the value of this method.

Permanent installation of a marker buoy at the outfall terminus or midpoint of the diffuser allows easy return to this point on subsequent sampling trips. Using the previously discussed range-finder technique or a line of desired length enables positioning at desired distances from the marker buoy. However, it is not uncommon to lose such a buoy due to vandalism, impact, or severe weather conditions. Therefore, it is necessary that the sampling party be prepared to relocate the outfall (e.g., by diver, sonar, or pinger mounted on the outfall itself), if location of stations is dependent on knowledge of the outfall location.

Because the techniques described here are inexpensive to implement (as are use of the sextant resection or theodolite intersection methods), they are attractive to small coastal municipalities. However, use of more sophisticated and less labor-dependent techniques may be achievable at moderate costs by renting or leasing, rather than buying such equipment.

USE OF LORAN-C

In their evaluation of positioning methods, Tetra Tech (1987) concluded that Loran-C did not provide the absolute and repeatable accuracies needed for the 301(h) program. However, because Loran-C is in such wide use and is relatively inexpensive, use of Loran-C in a special operating mode was re-examined in Tetra Tech (1988). The special operating mode is called differential Loran-C, which requires an additional Loran-C receiver onshore at a known geographic location. At this location, the Loran-C signals are received, and a correction is generated and transmitted to the survey vessel, allowing the correction to be applied to signals received by the ship's Loran-C unit.

Differential Loran-C was found to significantly improve the positional accuracies achievable compared to Loran-C in the normal mode. During a simulated monitoring program near Newport Beach, California, normal Loran-C positional errors of 40-50 m (131-164 ft) were reduced to 7-15 m (23-49 ft) using differential Loran-C in conjunction with special vessel operating

procedures, a video display, and data averaging techniques. Higher accuracies are expected in other coastal areas where improved lattice line crossing angles occur. Acceptability may depend on relative orientations of the diffuser and the error ellipse axes (Figure D-3, Table D-3). For those considering use of differential Loran-C, a procedure for determining the error in a ZID-boundary station location is provided in Tetra Tech (1988).

SYSTEM SELECTION PROCEDURE

A procedure for selecting an appropriate navigation system is described in detail in Tetra Tech (1987). The procedure involves definition of positioning requirements, establishment of screening criteria (e.g., range, accuracy, availability, and costs), review of candidate systems, and evaluation of purchase/lease options. As indicated in Figure D-4, a stepwise screening technique is recommended to identify an optimal system for a given monitoring program. At each step in the screening process, systems that cannot achieve the required criterion are removed from further consideration.

OUTFALL PIPE

DIFFUSER

ZID
BOUNDARY

95% PROBABILITY
ELLIPSE

(X,Y)

ACROSS-ZID
ERROR VARIATION

ELLIPSE ROUGHLY
PARALLEL TO DIFFUSER

ACROSS ZID
ERROR VARIATION

ELLIPSE ROUGHLY
PERPENDICULAR TO
DIFFUSER

X,Y   Coordinates of ZID-
        Boundary Sampling
        Station.

        95% Probability of
        Actual Sampling
        Station Position being
        in this Area.
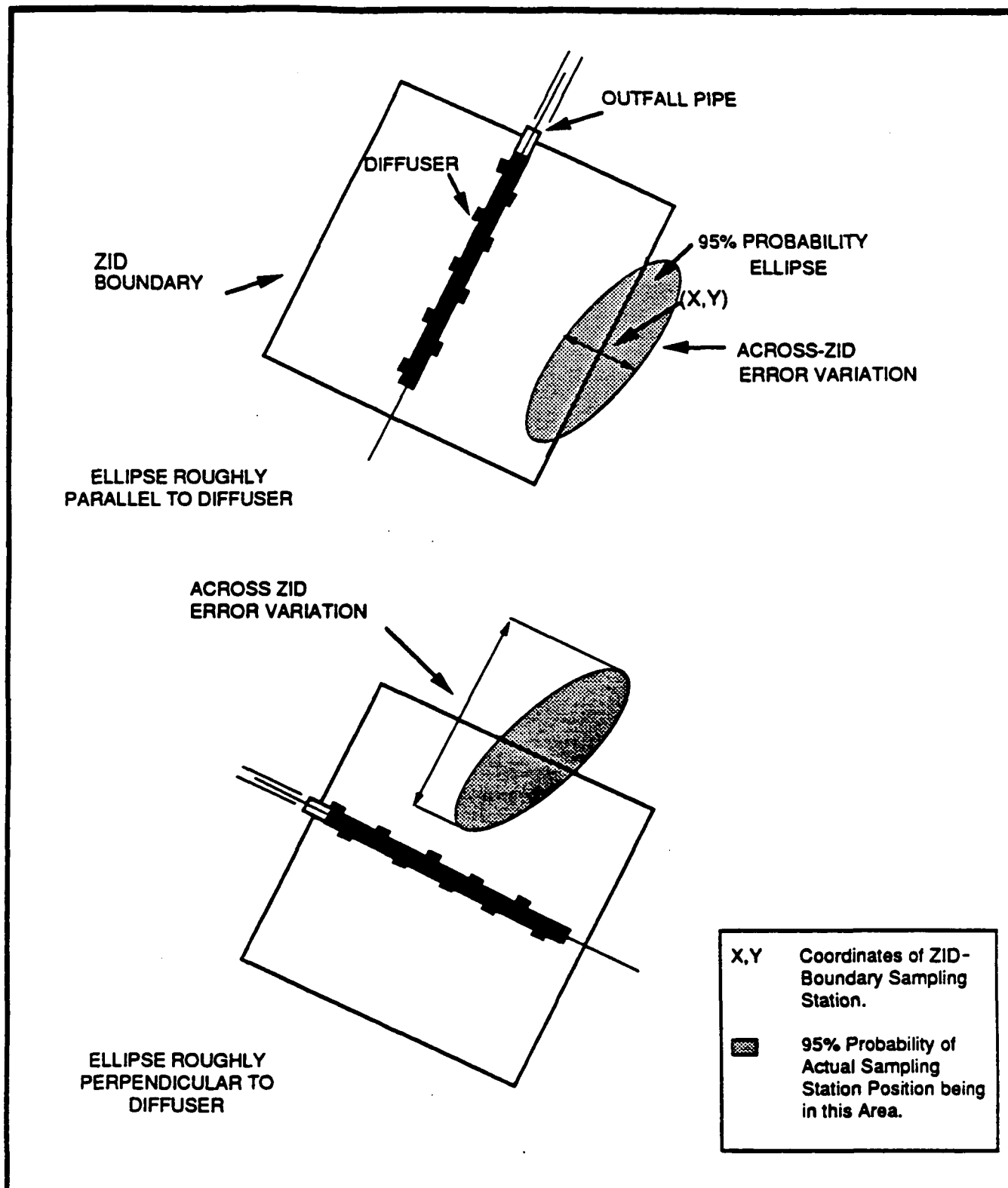
Figure D-3. Examples of differential Loran-C error ellipse orientation
at a ZID-boundary sampling station.

TABLE D-3. THEORETICAL ERROR ELLIPSES OF DIFFERENTIAL LORAN-C
FOR VARIOUS U.S. LOCATIONS

Location	Approximate Direction of Major Axis	Length of Major Axis[a]	Length of Minor Axis[a]
Anchorage, AK	NW/SE	70	20
Puget Sound, WA	NW/SE	180	40
San Francisco, CA	NE/SW	60	30
Los Angeles, CA	NE/SW	90	30
San Diego, CA	N/S	90	20
Mississippi Delta, LA	NW/SE	50	20
Panama City, FL	N/S	30	20
Chesapeake Bay, VA	W/E	40	20
Boston, MA	N/S	30	20

[a] Lengths are given to the nearest 10 m based on 95 percent confidence level error ellipses. Standard deviation of time differences is 25 nsecs (achievable with differential Loran-C).
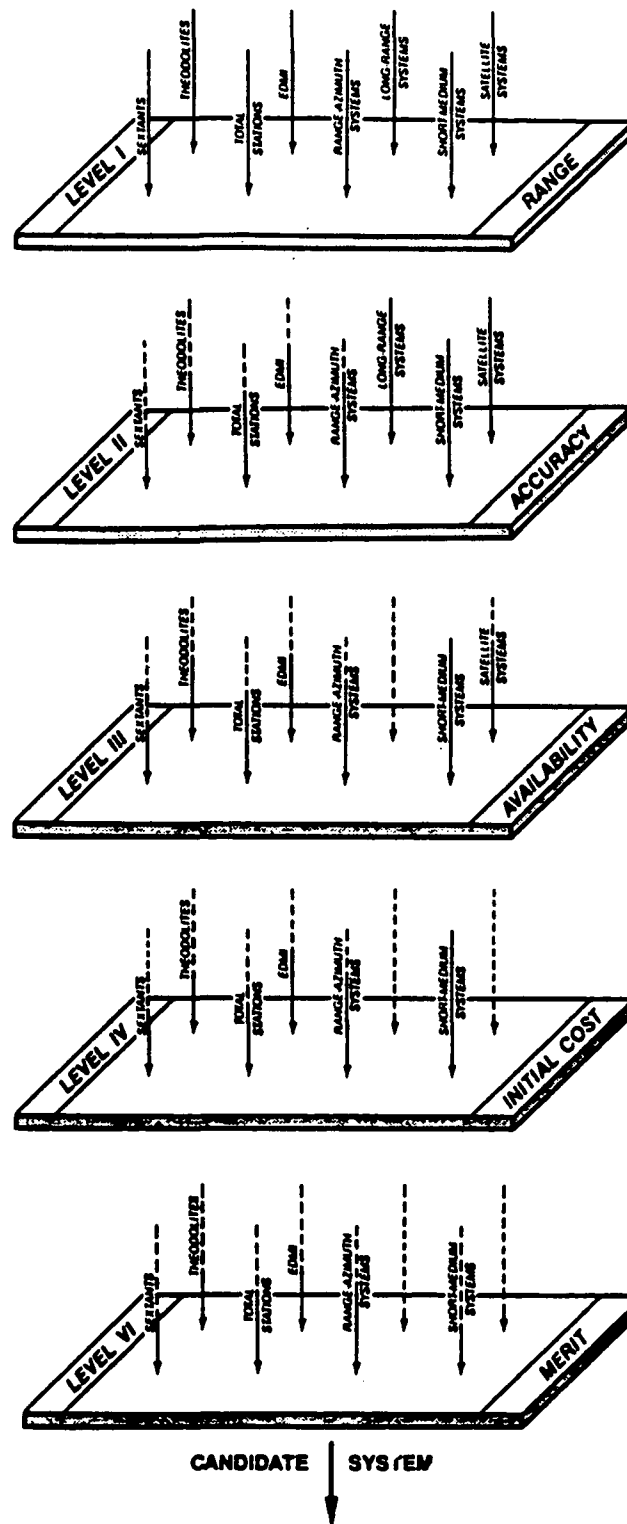
Figure D-4. Navigation system preliminary screening criteria.

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APPENDIX E

URBAN AREA PRETREATMENT REQUIREMENTS

CONTENTS

FIGURES

TABLES

INTRODUCTION

Section 303(c) of the Water Quality Act of 1987 amended Section 301(h) of the 1977 Clean Water Act by adding the "urban area pretreatment requirement." This requirement applies only to POTWs serving a population of at least 50,000 and only to toxic pollutants introduced by industrial dischargers. For each toxic pollutant introduced by an industrial discharger in affected POTWs, the applicant must demonstrate that it meets one of the following two conditions:

- Has an "applicable pretreatment requirement in effect"

- Achieves "secondary removal equivalency."

This new statutory requirement complements the toxics control program requirements in the existing Section 301(h) regulations (40 CFR 125.66) and other pretreatment requirements in 40 CFR 403.

The intent of this appendix is to help POTWs interpret and comply with the new requirement. For site-specific reasons, concepts and procedures recommended herein may not necessarily apply to all 301(h) applicants. Issues that are not addressed by this appendix should be directed to U.S. EPA Regional offices. Applicants should also check with appropriate state and local agencies for any explicit requirements (e.g., water quality standards) that apply to them. The procedures to demonstrate compliance with this urban area requirement must be formulated and implemented by each POTW with concurrence from the appropriate U.S. EPA Regional office. Compliance with the urban area pretreatment requirement is required before a 301(h) permit may be issued by U.S. EPA, although tentative approval may be granted on demonstration of the applicant's good faith effort.

When a review of the 301(h) application indicates that noncompliance with pretreatment requirements is substantial and that the applicant is not taking effective steps to assure compliance, then U.S. EPA may deny the permit. Factors relevant to such a decision include the number of non-complying industrial sources, the nature of their toxic pollutant contribution to the POTW, and potential or actual POTW interference of pass-through.

APPLICABLE TREATMENT PROGRAM APPROACH

Applicable pretreatment requirements for each toxic pollutant may take one of two forms:

- Categorical standards

- Local limits.

Categorical standards are nationally uniform, technology-based pretreatment limitations developed for specific industrial categories under Section 307 of the Clean Water Act. All categorical industries must comply with categorical standards, even if they discharge to a POTW without a federally approved local pretreatment program. By contrast, local limits are developed by the POTW to prevent interference with the treatment works or pass-through of toxic pollutants, as required by 40 CFR 403.5(b).

A given industrial discharger may be subject to categorical standards for some pollutants and local limits for other pollutants, or to both types of limitations for the same pollutant. In the latter case, the stricter standard applies. The urban area pretreatment requirement for all toxic pollutants entering a POTW will probably require a combination of both forms of pretreatment standards.

U.S. EPA GUIDANCE

The U.S. EPA Office of Water Enforcement and Permits (OWEP) and the U.S. EPA Office of Water Regulations and Standards (OWRS) have issued the following guidance manuals to assist POTWs in implementing pretreatment regulations and developing technically based local limits:

■ *Guidance Manual for POTW Pretreatment Program Development* (U.S. EPA 1983a)

■ *Procedures Manual for Reviewing a POTW Pretreatment Program Submission* (U.S. EPA 1983b)

■ *NPDES Compliance Inspection Manual* (U.S. EPA 1984)

■ *Guidance Manual for Implementing Total Toxic Organics (TTO) Pretreatment Standards* (U.S. EPA 1985a)

■ *Guidance Manual for the Use of Production-Based Pretreatment Standards and the Combined Wastestream Formula* (U.S. EPA 1985b)

■ *Pretreatment Compliance Monitoring and Enforcement Guidance* (U.S. EPA 1986a)

■ *Guidance Manual for Preventing Interference at POTWs* (U.S. EPA 1987a)

■ *Guidance for Reporting and Evaluating POTW Noncompliance with Pretreatment Implementation Requirements* (U.S. EPA 1987b)

■ *Guidance Manual on the Development and Implementation of Local Discharge Limitations Under the Pretreatment Program* (U.S. EPA 1987c) (enclosed as Attachment 2 to this appendix).

The implementation and enforcement guidelines in these manuals require the POTW to undertake the following:

■ In the POTW industrial waste survey (which must be updated on a regular basis), identify and locate all industries that discharge pollutants into the POTW

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- Demonstrate that the sampling procedures and analysis program undertaken were adequate to characterize industrial and nonindustrial pollutant loading to the POTW, and pollutant concentrations in the POTW influent, effluent, and sludge

- Compare measured pollutant concentrations to applicable sludge criteria or guidelines, water quality criteria or standards, and POTW process inhibition thresholds

- Demonstrate that the existing pretreatment program is adequate to control industrial user discharges, and that it contains specific numerical limits for industrial pollutants

- Demonstrate that local limits are technically based, adequate to protect the POTW, and allow compliance with its NPDES permit

- Demonstrate that steps have been taken to identify the causes of past POTW operating problems (e.g., industrial discharges, equipment failures, plant upsets, NPDES permit violations, sludge contamination) and correct them

- Demonstrate that POTW inspection and compliance monitoring procedures exist and are being implemented

- Demonstrate that the needed resources (e.g., funds, staff, equipment) are available to carry out program requirements.

LOCAL LIMITS

The technical approach used by a POTW to develop local limits is primarily a local decision, provided that the local limits are enforceable and scientifically based. Most POTWs use the headworks loading method in the U.S. EPA (1987c) local limits guidance manual. OWEP-recommended procedures for developing local limits appear as Attachment 1 to this

appendix. Best professional judgment can be used to establish pretreatment requirements when data or criteria are insufficient to perform a pollutant loading analysis for a specific pollutant of concern. The applicant may implement the local limits via uniform maximum allowable concentrations or discharger-specific maximum allowable mass emissions.

Local limits should be reviewed and revised periodically in response to changes in federal or state regulations, environmental protection criteria, plant design and operational criteria, or the nature of industrial contributions to POTW influent. For example, the following specific changes could affect criteria used to derive local limits:

- Changes in NPDES permit limits to include additional or more restrictive toxic pollutant limits

- Changes in water quality limits including toxicity requirements

- Changes in sludge disposal standards or POTW disposal methods

- Availability of additional site-specific data pertaining to pollutant removal efficiencies and/or process inhibition.

OWEP is presently developing guidance to determine the technical adequacy of local limits and to ensure their enforcement. This guidance will also clarify the use of best professional judgment for establishing local discharge limits or technology-based limits when the data are insufficient.

SECONDARY REMOVAL EQUIVALENCY APPROACH

One approach that 301(h) applicants may use to satisfy the new urban area pretreatment requirement is to demonstrate secondary removal equivalency. As noted in 40 CFR 125.65(d):

> An applicant shall demonstrate that it achieves secondary removal equivalency through the use of a secondary treatment pilot plant at the applicant's facility which provides an empirical determination of the amount of a toxic pollutant removed by the application of secondary treatment to the applicant's discharge, where the applicant's influent has not been pretreated. Alternatively, an applicant may make this determination using influent that has been pretreated, notwithstanding section 125.58(w).

In effect, the applicant's existing treatment processes and industrial pretreatment program must remove at least as much of a toxic pollutant as would be removed if the applicant applied secondary treatment and did not have an industrial pretreatment requirement for that pollutant. This approach can be represented as follows:

```
POTW existing  +  industrial    =  POTW existing  +  no industrial
   treatment       pretreatment        treatment        pretreatment
                                     upgraded to
                                   secondary treatment
```

U.S. EPA recognizes, however, that it would be simpler for applicants to perform this demonstration by using a secondary treatment pilot plant on the actual pretreated influent. This approach is shown below:

```
POTW existing  +  industrial    =  POTW existing  +  industrial
   treatment       pretreatment        treatment        pretreatment
                                     upgraded to
                                   secondary treatment
```

Although U.S. EPA will consider them, demonstrations to account for the effects of industrial pretreatment will probably be difficult. The secondary treatment pilot plant approach is conservative (i.e., protective) where it uses influent that has received industrial pretreatment, because the calculated required removals will be greater than those resulting from a demonstration using influent that has not been pretreated.

Secondary treatment at POTWs typically involves biological processes that remove organic matter through biochemical oxidation, usually variations of the activated sludge process. Other physical-chemical secondary treatment processes (e.g., coagulation, filtration, carbon adsorption) may also be used, particularly for nonbiodegradable wastewaters. The specific secondary treatment process used by a POTW is dependent on numerous factors such as wastewater quantity, waste biodegradability, and available space at the POTW site. Each POTW must determine the best strategy and the most applicable treatment process for complying with the secondary removal equivalency requirements.

The level of effluent quality attainable through the application of secondary treatment is defined in 40 CFR 133 (Table E-1). Treatment processes that are considered equivalent to secondary treatment (e.g., trickling filter, waste stabilization pond) are discussed in 40 CFR 133.105. Minimum levels of effluent quality attainable from these equivalent treatment processes differ from those shown in Table E-1.

Because secondary treatment levels were defined only for BOD, suspended solids, and pH, POTWs were usually not required to institute technology specifically to control toxic pollutants. Under the 1977 Clean Water Act, toxic pollutants in the POTW effluents were controlled predominantly through pretreatment programs, categorical standards, and local POTW limits required by the issuance of NPDES permits.

TABLE E-1. EFFLUENT WATER QUALITY VALUES THAT SHALL NOT BE EXCEEDED UNDER SECONDARY TREATMENT

Variable[a]	30-Day Average	7-Day Average	30-Day Average (Percent Removal)
BOD_5	30 mg/L	45 mg/L	>85
$CBOD_5$[b]	25 mg/L	40 mg/L	>85
SS	30 mg/L	45 mg/L	>85
pH	6.0 to 9.0		

[a] BOD_5 = 5-day measure of biochemical oxygen demand; $CBOD_5$ = 5-day measure of carbonaceous biochemical oxygen demand; SS = suspended solids.

[b] At the option of the NPDES-permitting authority, $CBOD_5$ may be substituted for BOD_5.

SECONDARY TREATMENT PILOT PLANT DESIGN CRITERIA

A secondary treatment pilot plant should be designed for an average flow of approximately 150 GPD. The flow rate should remain constant over a 24-h period. The pilot plant should require minimum operation and maintenance time, and must be able to operate unattended for 16-24 h. The organic loading will vary with the diurnal and seasonal fluctuations in the BOD_5 concentration in the existing POTW effluent. Design criteria for the secondary treatment pilot plant are shown in Table E-2.

A conventional activated sludge system (Figure E-1) for a POTW includes the following related components:

■ Single or multiple reactor basins (i.e., aeration tanks) where microorganisms consume the organic wastes. These basins are designed to allow for complete mixing of its contents, which are defined as mixed liquor suspended solids (MLSS). Each basin must provide typical hydraulic retention times of 2-24 h.

■ Pressurized or atmospheric oxygen-containing gases that are dispersed into the reactor basin.

■ Settling basin (i.e., final clarifier) to separate the MLSS from the treated wastewater.

■ Equipment to collect the solids in the settling basin, and to recycle the active biological solids (i.e., activated sludge) to the reactor basin.

■ Equipment to remove excess active biological solids from the system.

Typical design variables for the conventional activated sludge process are shown in Table E-3. Additional information on activated sludge systems is

TABLE E-2. SECONDARY TREATMENT PILOT PLANT DESIGN CRITERIA

Reactor Basin (Aeration Tank)

Volume	50 gal (189 L)
Detention time	8 h
Organic loading	25-60 lb BOD/1,000 ft^3/day (0.4-1.0 kg/m^3/day)
Air requirement	0.20-0.44 ft^3/min (0.33-0.75 m^3/h)

Settling Basin (Final Clarifier)

Volume	20 gal (76 L)
Surface Area	0.375 ft^2 (0.035 m^2)
Overflow Area	400 gal/ft^2/day (16.3 m^3/m^2/day)
Solids Loading	14 lb/ft^2/day (68.4 kg/m^2/day)
Weir Length	0.5 ft (0.152 m)
Detention Time	3 h

Influent Feed Pump

Capacity	0-290 gal/day (0-12.7 L/sec)
Type	Peristaltic

Return Activated Sludge Pump

Capacity	0-130 gal/day (0-5.7 L/sec)
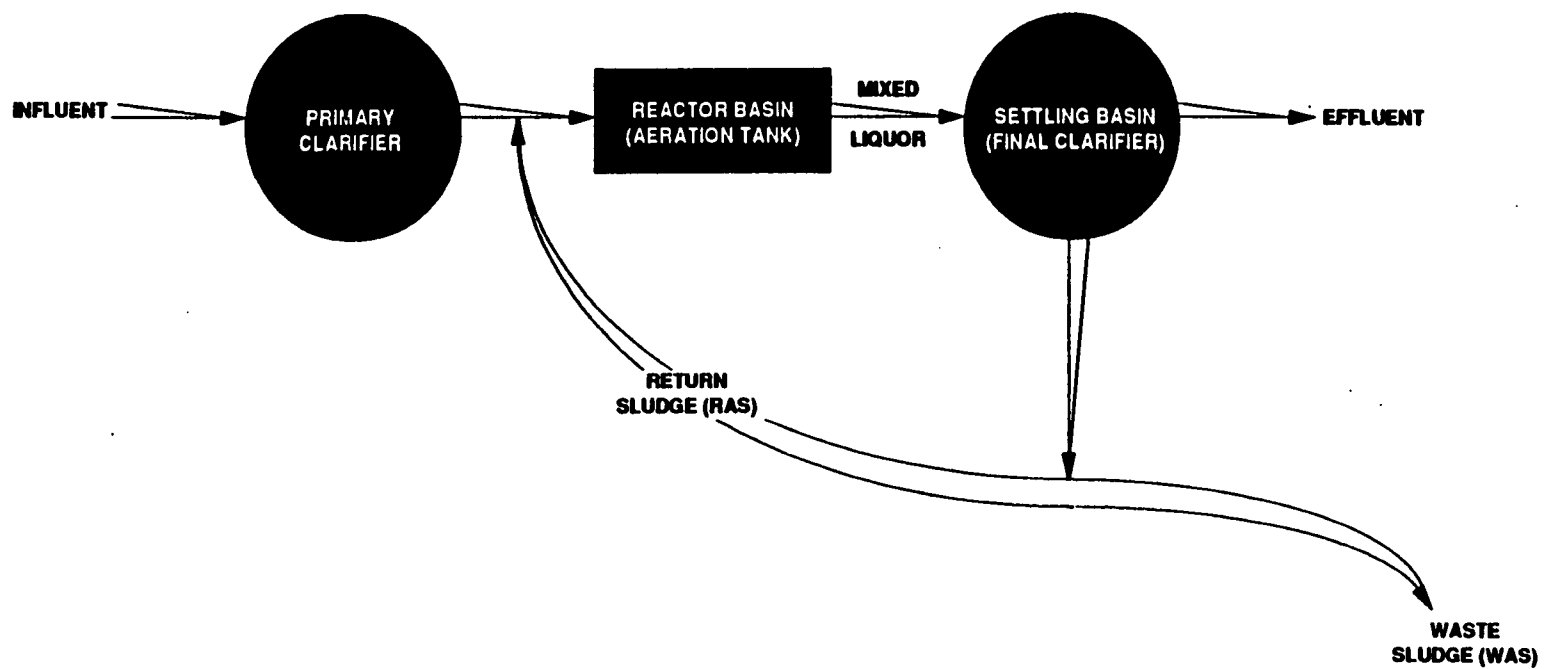
Figure E-1. Components of a conventional activated sludge system.

TABLE E-3.   CONVENTIONAL ACTIVATED SLUDGE DESIGN PARAMETERS

Food to microorganism ratio	0.15-0.4 lb BOD_5/lb MLSS/day
Mean cell residence time	5-15 days
Aeration detention time	4-8 h
Oxygen requirements	0.8-1.1 lb (kg) O_2/lb (kg) BOD_5 removed
Return activated sludge flow rate	30-100 percent influent flow
Mixed liquor suspended solids (MLSS)	1,500-4,000 mg/L
Organic loading at 3,000 mg/L MLSS	20-60 lb BOD/1,000 ft^3 (0.3-1.0 kg BOD/m^3)
Respiration (oxygen uptake) rate at 3,000 mg/L MLSS	15-45 mg oxygen/L/h
Sludge volume index	90-150
Waste activated sludge	0.4-0.6 lb (kg)/lb (kg) BOD removed

provided by the Water Pollution Control Federation [(WPCF) 1976, 1987] and WPCF/American Society for Civil Engineers (1977).

SECONDARY TREATMENT PILOT PLANT STARTUP

In the activated sludge process, microorganisms metabolize nearly all soluble organic matter in the influent. The microorganisms (i.e., active biological solids) must be removed from the settling basin to produce an acceptable effluent, and the proper operation of the settling basin is critical. The following process control parameters should be monitored to ensure proper operation of the activated sludge system:

- MLSS

- Mixed liquor volatile suspended solids (MLVSS)

- Dissolved oxygen

- Sludge volume index (SVI)

- Sludge density index (SDI)

- Organic loading

- Return activated sludge (RAS) flow rate

- Waste activated sludge (WAS) flow rate

- Mean cell residence time (MCRT)/solids retention time (SRT)

- Food/microorganism ratio (F/M)

- Temperature

■ Hydrogen ion concentration (pH)

■ Respiration rate (RR).

In addition to these process control parameters, microscopic examination of the MLSS should be performed.

An initial F/M ratio of 0.2 should be achieved. Field operators should adjust the F/M ratio by changing the MLSS concentration in the reactor basin if the required 30-day average effluent quality (i.e., 30 mg/L BOD, 30 mg/L suspended solids) cannot be achieved. If temperature varies substantially between summer and winter, the F/M ratio will probably need to be lowered during winter to achieve the required effluent quality.

The pilot plant should be seeded with MLSS from a local domestic wastewater treatment facility. Acclimation of the pilot plant will require about 4-6 wk. If there is no local source of MLSS, the pilot plant may be started using the POTW's effluent. An additional 4-6 wk may be needed to ensure that the MLSS meets the desired design concentration.

The MLSS should be fed with domestic wastewater for the first 2-3 days. The volumetric proportion of the effluent should then be adjusted to 10 percent of the total feed for 4-5 days. After the initial week of operation, the volumetric proportion of the regular POTW effluent in the pilot plant feed can be increased approximately 5 percent per day until the system is receiving 100 percent POTW effluent.

Sampling for BOD_5 and suspended solids should be conducted daily during and after the acclimation period. Sampling for toxic pollutants should not be started until 2 wk after the pilot plant is receiving 100 percent POTW effluent.

SECONDARY TREATMENT PILOT PLANT OPERATING CRITERIA

The process control parameters identified in the preceding section should be monitored to provide information for process control and to determine treatment efficiency. A monitoring schedule is shown in Table E-4. The frequency of sample collection and analysis may vary for each POTW, however, depending on the size of the POTW, available laboratory facilities, available staff, and technical skills of personnel. Additional sampling and analysis may be required for abnormal conditions or during periods of process upsets. Implementation of the monitoring program, data interpretation, and pilot plant operation and maintenance is estimated to require about 5 labor hours per day. Each process control parameter is discussed below.

Mixed Liquor Suspended Solids

Samples of MLSS should be collected from the effluent end of the reactor basin twice daily and analyzed for suspended solids. This analysis will measure the total amount of solids in the aeration system. The concentration of the MLSS, which depends on the influent BOD_5 concentration, should be adjusted based on the daily average.

Mixed Liquor Volatile Suspended Solids (MLVSS)

Each MLSS sample should be analyzed for MLVSS. This analysis indirectly measures the living biological percentage of the MLSS. The concentration of MLVSS is normally 70 to 80 percent of the concentration of the MLSS.

Dissolved Oxygen

The concentration of dissolved oxygen in the reactor basins should be measured twice daily to ensure that a concentration of 1-3 mg/L is maintained. Samples should be collected about 2 ft below the surface of the reactor basin, near the effluent weir. The plant operator should adjust the air supply to provide more air if the dissolved oxygen concentration is less than 1 mg/L and less air if it is greater than 3 mg/L.

TABLE E-4. PILOT PLANT MONITORING SCHEDULE

Sampling Point	Parameters[a]	Frequency
Primary Effluent	Temperature	1 grab daily
	pH	1 grab daily
	SS	4 grabs weekly and 3 24-h composites weekly
	BOD_5	3 24-h composites weekly
	Overflow rate	1 grab daily
	$CBOD_5$	1 24-h composite weekly
MLSS	Temperature	1 grab daily
	pH	1 grab daily
	Dissolved oxygen	2 grabs daily
	Respiration rate	2 grabs daily
	Sludge volume index	2 grabs daily
	SS	1 grab daily
	VSS	1 grab daily
	Microscopic examination	1 grab daily
WAS/RAS	SS	1 grab daily
Secondary Clarifier	Sludge blanket depth	2 grabs daily
Final Effluent	Temperature	1 grab daily
	pH	1 grab daily
	Settleable solids	1 grab daily
	SS	4 grabs weekly and 3 24-h composites weekly
	BOD_5	3 24-h composites weekly
	$CBOD_5$	1 24-h composite weekly

[a] SS = Suspended solids; BOD_5 = 5-day biochemical oxygen demand; $CBOD_5$ = 5-day carbonaceous biochemical oxygen demand; VSS = volatile suspended solids.

Sludge Volume Index (SVI)

The rate at which the MLSS settles in the settling basin depends on the sludge characteristics. These characteristics are described by a simple settling test: 1,000 mL of the MLSS effluent is collected and allowed to settle for 30 min in a Mallory settleometer. At the end of 30 min, the volume of the settled sludge is measured. The SVI is calculated as follows:

$$SVI = \frac{\text{volume of settled sludge (mL) x 1,000}}{\text{MLSS (mg/L)}}$$

The lower the SVI, the more dense the sludge. An SVI of 150 or less is usually considered good.

Sludge Density Index (SDI)

The SDI test is also used to indicate the settling characteristics of the sludge, and it is arithmetically related to the SVI:

$$SDI = \frac{100}{SVI}$$

SDI of a "good settling sludge" is about 1.0. A value of less than 1.0 indicates light sludge that settles slowly. An index greater than 1.5 indicates dense sludge that settles rapidly.

Organic Loading

From routine laboratory BOD_5 analysis, the plant operator can determine organic loading in the reactor basin.

$$\text{Organic loading} = (\text{lb BOD/1,000 ft}^3/\text{day})$$

$$\text{POTW Effluent BOD (mg/L) x } \frac{\text{POTW Effluent Flow (MGD) x 0.0624}}{\text{Reactor Basin Volume (MG)}}$$

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Return Activated Sludge (RAS) Flow Rate

To properly operate the activated sludge process, an MLSS that settles adequately must be achieved and maintained. The MLSS are settled in the settling basin and then returned to the reactor basin as RAS. The RAS allows the microorganisms to remain in the treatment system longer than the flowing wastewater. Changes in the activated sludge quality and settling characteristics will require different RAS flow rates.

Two basic approaches can be used to control the RAS flow rate. One approach establishes a constant RAS flow rate, independent of the influent flow. This approach is simple (i.e., maximum solids loading in the settling basin occurs at the start of the peak flow periods) and less operator attention is needed. A disadvantage of this approach is that the F/M ratio is constantly changing. However, because of short-term variation in the MLSS due to hydraulic loading, the range of fluctuation in the F/M ratio is generally small enough to ensure that no significant problems arise.

A second approach establishes the RAS flow rate as a constant percentage of the influent flow. This approach reduces variations in the MLSS concentration and the F/M ratio, and the MLSS remain in the settling basin for shorter time periods (which may reduce the possibility of denitrification in the basin). The most significant disadvantage of this approach is that the settling basin is subjected to maximum solids loading when the basin contains the maximum amount of sludge, which produces excessive solids in the effluent.

Two methods are commonly used to determine the RAS flow rate. The settleability method uses the settled sludge volume from the SVI test to calculate the RAS flow rate:

$$\text{RAS Flow Rate (MGD)} =$$

$$\text{Volume of Settled Sludge (mL)} \times \frac{[\text{POTW Effluent Flow (MGD)}]}{1,000 \text{ mL}}$$

The second and more direct method is to monitor the depth of the sludge blanket in the settling basin. The depth of the sludge blanket should be less than one-fourth of the water depth of the settling basin sidewall. The operator must check the sludge blanket depth twice daily, adjusting the RAS flow to control the blanket depth. If the depth of the sludge blanket is increasing, increasing the RAS flow is only a short-term solution. Increases in sludge blanket depth may result from too much activated sludge in the treatment system, a poorly settling sludge, or both. If the sludge is settling poorly, increasing the RAS flow may cause even more problems by further increasing the flow through the settling basin. Long-term corrections noted below must be made to improve the settling characteristics of the sludge or remove the excess solids from the treatment system:

- If the sludge is settling poorly because of bulking, the environmental conditions for the microorganisms must be improved

- If there is too much activated sludge in the treatment system, the excess sludge must be wasted.

The best time to measure RAS flow is during the period of maximum daily flow, because the clarifier is operating under the highest solids loading rate. Adjustments in the RAS flow rate should be needed only occasionally if the activated sludge process is operating properly.

Waste Activated Sludge (WAS) Flow Rate

The increase of activated sludge is a cumulative process that eventually produces surplus WAS. This surplus has to be permanently removed from the treatment process and collected for ultimate disposal. The WAS flow rate should be determined and adjusted daily to maintain the desired mean cell

residence time (MCRT), based on the MLSS in the entire secondary system, and RAS suspended solids concentration:

$$\text{WAS Flow Rate (MGD)} = \text{MLSS (mg/L)} \times \frac{[\text{Aeration Tank Volume (MG)} + \text{Clarifier Volume (MG)}]}{[\text{desired MCRT(days)}] \times [\text{RAS Suspended Solids (mg/L)}]}$$

Mean Cell Residence Time (MCRT)/Solids Retention Time (SRT)

The MCRT, which is also called the SRT, is a measure of the age of sludge. Under normal conditions, the MCRT is 5-15 days. MCRT is defined as:

$$\frac{\text{Suspended solids in total secondary system (lb)}}{\text{Solids wasted (lb/day)} + \text{effluent solids (lb/day)}} =$$

$$\frac{[\text{MLSS (mg/L)}] \times [\text{Aeration Tank Volume(MG)} + \text{Secondary Clarifier Volume(MG)}]}{[\text{WAS Suspended solids (mg/L)} \times \text{WAS Flow (MGD)}] + [\text{Effluent Suspended solids (mg/L)} \times \text{Effluent Flow (MGD)}]}$$

MCRT is the best process control technique available to the plant operator. By using the MCRT, the operator can control the quantity of food available to the microorganisms and calculate the amount of activated sludge that should be wasted.

Food/Microorganism Ratio (F/M)

The F/M ratio is the ratio of BOD in the POTW effluent to the MLVSS. An F/M ratio of 0.15 to 0.4 is desirable. F/M is defined as:

$$\frac{\text{POTW Effluent BOD (mg/L)}}{\text{MLVSS (mg/L)}}$$

To control the F/M ratio, the operator must adjust the MLSS by wasting more or less sludge.

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Temperature

In process control, accurate temperature measurements are required to predict and evaluate process performance, thereby enhancing microbial growth. Typically, the rate of microbial growth doubles for every 10° C increase in temperature within the specific temperature range of the microbe.

Hydrogen Ion Concentration (pH)

The activity and health of microorganisms is affected by pH. Sudden changes or abnormal pH values may indicate an adverse industrial discharge. A pH drop will also result when nitrification is occurring in a biological process; alkalinity is destroyed and carbon dioxide is produced during the nitrification process.

Respiration Rate (RR)

The efficiency of the activated sludge process depends primarily on the activity of bacteria that use organic compounds in sewage for energy and reproduction. When in contact with an adequate food supply, viable bacteria will have a respiration rate (i.e., oxygen uptake rate) of 5-15 mg oxygen/h/g MLSS. Respiration rate data provide immediate information concerning viability, nitrification, organic loading, nutrient levels, and toxicity in the activated sludge.

The respiration rate, or oxygen uptake rate, is monitored with a dissolved oxygen meter over a time interval (t) (e.g., 6-10 min). The respiration rate is a measure of the decrease in dissolved oxygen concentration:

$$RR \text{ (mg oxygen/h/g MLSS)} = \frac{[DO \text{ change over } t \text{ (mg/L)}] \times [60,000]}{[MLSS \text{ (mg/L)}] \times [t \text{ (min)}]}$$

Microscopic Examination

Microscopic examination of the MLSS can be used to evaluate the effectiveness of the activated sludge process. The most important microorganisms are the protozoa, heterotrophic bacteria, and autotrophic bacteria responsible for purifying the wastewater. Both protozoa (e.g., ciliates) and rotifers are indicators of treatment performance, and large numbers of these organisms in the MLSS indicate good quality sludge. Large numbers of filamentous organisms and certain ciliates indicate poor sludge quality, a condition commonly associated with a sludge that settles poorly (i.e, the sludge floc is usually light and fluffy because it has a low density). Many other organisms in the sludge (e.g., nematodes, waterborne insect larvae) may be found in the sludge. However, these organisms are not significant to the activated sludge process.

TOXIC POLLUTANT MONITORING PROGRAM, TESTING PROCEDURES, AND QUALITY ASSURANCE/QUALITY CONTROL (QA/QC)

A sampling strategy must be developed to estimate the difference between toxic pollutant concentrations in the existing discharge and those in the secondary treatment pilot plant discharge. Samples must be collected using proper techniques and analyzed using appropriate analytical methods. Both field and laboratory methods must be performed under defined QA/QC procedures.

Applicants are referred to the following documents for guidance on specific topics relevant to the design and execution of 301(h) monitoring programs:

■ Sampling/Monitoring Program

- *NPDES Compliance Sampling Manual* (U.S. EPA 1979a)

- *Design of 301(h) Monitoring Programs for Municipal Wastewater Discharges to Marine Waters* (U.S. EPA 1982a)

- *Handbook for Sampling and Sample Preservation of Water and Wastewater* (U.S. EPA 1982c)

■ Chemical Analytical Methods

- *Methods for Chemical Analysis of Water and Wastes* (U.S. EPA 1979b, revised 1983)

- *Guidelines Establishing Test Procedures for the Analysis of Pollutants* [40 CFR Part 136 (1984)]

- *Standard Methods for the Examination of Water and Wastewater (16th ed.)* (American Public Health Association 1985)

- *Analytical Methods for EPA Priority Pollutants and 301(h) Pesticides in Estuarine and Marine Sediments* (Tetra Tech 1986a)

- *Analytical Methods for EPA Priority Pollutants and 301(h) Pesticides in Tissues from Estuarine and Marine Organisms* (Tetra Tech 1986b)

■ Quality Assurance/Quality Control (QA/QC)

- *Handbook for Analytical Quality Control in Water and Wastewater Laboratories* (U.S. EPA 1979c)

- *Quality Assurance/Quality Control (QA/QC) for 301(h) Monitoring Programs: Guidance on Field and Laboratory Methods* (Tetra Tech 1987).

Information from these documents is summarized below.

SAMPLING FREQUENCY

The frequency of sampling is dependent on the characteristics of the discharge (e.g., influent and effluent toxic pollutant variability, flow, size and location of the discharge). For example, large applicants with substantial quantities of toxic pollutants should conduct more frequent sampling than small dischargers with fewer toxic pollutants. Also, if existing toxic pollutant data are minimal, and estimates of periods of maximum pollutant loadings and peak concentrations are not known, then more frequent monitoring is needed.

The concentrations of toxic pollutants in the discharge may vary in response to periodic peak inflows. If a fixed periodic trend is observed (e.g., a sine curve) the sampling plan could be designed to collect samples during the peak period.

If a fixed sampling interval is chosen that is equal to or a multiple of the period, every sample would be taken at the same inflow condition and the estimate of the mean difference in toxic pollutant concentrations between samples would not take into effect all possible inflows. The most favorable sampling situation occurs when the fixed sampling interval is an odd multiple of the half-period (i.e., successive deviations above and below the mean inflow would mathematically cancel one another, and the mean difference in concentration between samples would take into effect the mean inflow). However, toxic pollutant effluent data from the applicant may not be sufficient to identify the odd multiple of the half-period. In this case, a fixed sampling interval would not be recommended.

Assuming that the toxic pollutant limits for the POTW will be based on the pollutant concentrations measured in the secondary treatment effluents, a flexible sampling scheme for secondary treatment pilot plant effluents may involve sample collection for 1 day/wk (over 24 h) on different days of the week over a 1-yr period of pilot plant operation. This flexible sampling frequency would generate a data set that represents an acclimated biological treatment system. It would also address the day/night, weekday/weekend, and seasonal variations in domestic, industrial/commercial, and wet- and dry-weather discharges.

SAMPLE COLLECTION AND ANALYSIS

Representative samples must be collected to ensure that data are reliable. Care must be taken to select appropriate sampling devices and procedures. Depending upon the toxic pollutant to be analyzed, three types of samples may be collected:

- Grab sample - a discrete sample volume is collected. (This type of sample will not always provide an accurate measure of wastewater characteristics, particularly when the flow or pollutants are heterogeneous or vary with time.)

- Simple composite sample - equal sample volumes are collected sequentially overtime and combined in a single reservoir. (This type of sample does not measure the mass of pollutants discharged, because pollutant loading is a flow-related value.)

- Flow-proportioned composite sample - incremental samples are collected over time and sample volumes are proportional to flow. Incremental samples are combined in a single reservoir. (This type of sample provides the most accurate measurement of wastewater quality and pollutant loading.)

The methods to be used for the analysis of toxic pollutants are summarized in Tables E-5, E-6, and E-7. Grab samples for volatile organic compounds, total recoverable phenolic compounds, and cyanide should be collected manually at least four times during the discharging period of the POTW during a 24-h period (e.g., at least every 6 h within a 24-h period, assuming continuous discharge). Samples for all other variables should be collected using an automatic sampler. The automatic sampler should collect a selected number of sample aliquots (minimum of 100 mL each) during the discharging period of the POTW. Recommended sample sizes, containers, preservation techniques, and holding times are shown in Table E-8. Sample analyses will generally be completed by the analytical laboratory within 4-6 wk; data analyses will generally require an additional week. Interpretation of all data collected at the pilot plant during 1 yr will require about 2 wk.

TABLE E-5. LIST OF TEST PROCEDURES APPROVED BY U.S. EPA
FOR INORGANIC COMPOUNDS IN EFFLUENT

Note: This table is an exact reproduction of Table 1B in 40 CFR 136.3.

Parameter, units, and method	Reference (method No. or page)				
	EPA 1979	Standard methods 16th Ed.	ASTM	USGS [1]	Other
1 Acidity, as CaCO₃, mg/L: Electrometric end point or phenolphthalein end point.	305.1	402(4.a)	1067–82(E)		
2. Alkalinity, as CaCO₃, mg/L:					
Electrometric or colorimetric titration to pH 4.5, manual, or.	310.1	403	D1067–82(B)	I–1030–84	33.014 [2]
Automated	310.2			I–2030–84	
3. Aluminum—Total [3] mg/L: Digestion [3] followed by:					
AA direct aspiration	202.1	303C		I–3051–84	
AA furnace	202.2	304			
Inductively coupled plasma, or					200.7 [4]
Colorimetric (Erichrome cyanine R)		306B			
4. Ammonia (as N), mg/L: Manual distillation (at pH 9.5) [5] followed by:	350.2	417A			33.057 [2]
Nesslerization	350.2	417B	D1426–79(A)	I–3520–84	33.057 [2]
Titration	350.2	417D			
Electrode	350.3	417 E or F	D1426–79(D)		
Automated phenate or	350.1	417G	D1426–79(C)	I–4523–84	
Automated electrode					Note 6.
5. Antimony—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	204.1	303A			
AA furnace, or	204.2	304			
Inductively coupled plasma					200.7 [4]
6. Arsenic—Total [3], mg/L: Digestion [3] followed by	206.5				
AA gaseous hydride	206.3	303E	D2972–84(B)	I–3062–84	
AA furnace	206.2	304			
Inductively coupled plasma, or					200.7 [4]
Colorimetric (SDDC)	206.4	307B	D2972–84(A)	I–3060–84	
7 Barium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	208.1	303C		I–3084–84	
AA furnace, or	208.2	304			
Inductively coupled plasma					200.7 [4]
8. Beryllium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	210.1	303C	D3645–84(A)	I–3095–84	
AA furnace,	210.2	304			
Inductively coupled plasma, or					200.7 [4]
Colorimetric (aluminon)		309B			
9. Biochemical oxygen demand (BOD₅), mg/L:					
Dissolved Oxygen Depletion	405.1	507		I–1578–78 [7]	33.019 [2], p. 17. [8]
10. Boron—Total, mg/L:					
Colorimetric (curcumin)	212.3	404A		I–3112–84	
or Inductively Coupled plasma					200.7 [4]
11. Bromide, mg/L: Titrimetric	320.1		D1246–82(C)	I–1125–84	p. S44. [9]
12. Cadmium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	213.1	303 A or B	D3557–84 (A or B).	I–3135–84 or I–3136–84.	33.089 [2], p. 37 [a]
AA furnace	213.2	304			
Inductively coupled plasma					200.7 [4]
Voltametry [10], or			D3557–84(C)		
Colorimetric (Dithizone)		310B			
13 Calcium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	215.1	303A	D511–84(B)	I–3152–84	

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Parameter units. and method	EPA 1979	Standard methods 16th Ed.	ASTM	USGS [1]	Other
Inductively coupled plasma. or					200.7 [4]
Titrimetric (EDTA)	215.2.	311C...	D511-84(A)		
14 Carbonaceous biochemical oxygen demand (CBOD). mg/L [11] Dissolved Oxygen Depletion with nitrification inhibitor		507(5.6.6)			
15 Chemical oxygen demand (COD). mg/L.					
Titrimetric. or	410.1. 410.2. or 410.3.	508A	D1252-83	I-3560-84 or I-3562-84.	33.034 [2]. p. 17 [5]
Spectrophotometric. manual or automated.	410.4			I-3561-84	Notes 12 or 13.
16. Chloride. mg/L.					
Titrimetric (silver nitrate)		407A	D512-81(B)	I-1183-84	
or (Mercuric nitrate). or	325.3	407B	D512-81(A)	I-1184-84	33.067 [2]
Colorimetric. manual or			D512-81(C)	I-1187-84	
Automated (Ferricyanide)	325.1. or 325.2	407D		I-2187-84	
17 Chlorine—Total residual. mg/L. Titrimetric:					
Amperometric direct	330.1	408C	D1253-76(A)		
Starch end point direct	330.3.	408A	D1253-76(B) Part 18.3		
Back titration either end point [14]. or	330.2.	408B			
DPD-FAS	330.4.	408D			
Spectrophotometric. DPD	330.5.	408E			
Or Electrode					Note 15
18 Chromium VI dissolved. mg/L: 0.45 micron filtration followed by:					
AA chelation-extraction. or	218.4.	303B		I-1232-84	
Colorimetric (Diphenylcarbazide)				I-1230-84	307B. [16]
19 Chromium—Total [3]. mg/L. Digestion [4] followed by:					
AA direct aspiration	218.1	303A	D1687-84(D)	I-3236-84	33.089 [2]
AA chelation extraction	218.3.	303B			
AA furnace	218.2.	304			
Inductively coupled plasma or					200.7 [4]
Colorimetric (Diphenylcarbazide)		312B	D1687-84(A)		
20. Cobalt—Total [3]. mg/L. Digestion [3] followed by:					
AA direct aspiration	219.1	303 A or B	D3558-84 (A or B).	I-3239-84 or. I-3240-84	P 37. [5]
AA furnace. or	219.2.	304			
Inductively coupled plasma					200.7 [4]
21 Color platinum cobalt units or dominant wavelength. hue. luminance purity:					
Colorimetric (ADMI). or	110.1	204D			Note 17
(Platinum cobalt). or	110.2	204A		I-1250-84	
Spectrophotometric	110.3	204B			
22. Copper—Total [3] mg/L. Digestion [3] followed by:					
AA direct aspiration	220.1	303 A or B	D1688-84 (D or E).	I-3270-84 or I-3271-84.	33.089 [2]. p. 37 [5]
AA furance.	220.2.	304			
Inductively coupled plasma.					200.7 [4]
Colorimetric (Neocuproine). or		313B	D1688-84(A)		
(Bicinchoninate).					Note 18.
23. Cyanide—Total. mg.L. Manual distillation with MgCl; followed by		412B			
Titrimetric. or		412C			p. 22 [5]
Spectrophotometric. manual or	335.2	412D	D2036-82(A)	I-3300-84	
Automated. [19]	335.3.		D2036-82(A)		

Parameter units. and method	Reference (method No. or page)				
	EPA 1979	Standard methods 16th Ed	ASTM	USGS [1]	Other
24 Cyandie amendable to chlonnation. mg/L Manual distillation with MgCl; followed by titrimetric or spectrophotometric.	335.1	412F	D2036-82(B)		
25 Fluonde—Total. mg/L. Manual distillation [3] followed by:		413A			
Electrode. manual or	340.2	413B	D1179-80(B)		
Automated.				I-4327-84	
Colorimetric (SPADNS).	340.1	413C	D1179-80(A)		
Or Automated complexone	340.3	413E			
26. Gold—Total[3]. mg/L. Digestion[3] followed by:					
AA direct asprration, or	231.1	303A			
AA furnace	231.2	304			
27. Hardness—Total. as CaCO₃ mg/L					
Automated colorimetric.	130.1				
Titrimetric (EDTA), or Ca plus Mg as their carbonates. by inductively coupled plasma or AA direct aspiration. (See Parameters 13 and 33.)	130.2	314B	D1126-80	I-1338-84	33.082.[2]
28. Hydrogen ion (pH). pH units:					
Electrometric. measurement, or	150.1	423	D1293-84 (A or B).	I-1586-84	33.006.[2]
Automated electrode					Note 20.
29. Indium—Total[3]. mg/L. Digestion[3] followed by:				–	
AA direct aspiration, or	235.1	303A			
AA furnace	235.2	304			
30. Iron—Total[3]. mg/L. Digestion[3] followed by:					
AA direct aspiration.	236.1	303 A or B.	D1068-84 (C or D).	I-3381-84	33.089.[2]
AA furnace.	236.2	304			
Inductively coupled plasma. or					200.7.[4]
Colorimetric (Phenanthroline).		315B	D1068-84(A)		Note 21.
31 Kjeldahl nitrogen—Total. (as N). mg/L. Digestion and distillation followed by:	351.3	420 A or B.	D3590-84(A)		
Titration.	351.3	417D	D3590-84(A)		33.051.[2]
Nesslerization..	351.3	417B	D3590-84(A)		
Electrode	351.3	417 E or F			
Automated phenate	351.1			I-4551-78 [7]	
Semi-automated block digestor. or	351.2		D3590-84(A)		
Potentiometric	351.4		D3590-84(A)		
32 Lead—Total [3]. mg/L. Digestion [3] followed by:					
AA direct aspiration	239.1	303 A or B.	D3559-85 (A or B).	I-3399-84	33.089.[2]
AA furnace	239.2	304			
Inductively coupled plasma.					200.7 [4]
Voltametry [10]. or			D3559-85(C)		
Colorimetric (Dithizone)		316B			
33 Magnesium—Total [3]. mg/L. Digestion [3] followed by:					
AA direct aspiration	242.1	303A	D511-84(B)	I-3447-84	33.089.[2]
Inductively coupled plasma, or					200.7.[4]
Gravimetric		318B	D511-77(A)		
34 Manganese—Total [3]. mg/L. Digestion [3] followed by:					
AA direct aspiration	243.1	303 A or B.	D858-84 (B or C)	I-3454-84	33.089.[2]
AA furnace	243.2	304			
Inductively coupled plasma. or					200 7 [4]
Colorimetric (Persulfate). or		319B.	D858-84(A)		33 126 [2]
(Periodate)					Note 22.

Parameter, units, and method	Reference (method No. or page)				
	EPA 1979	Standard methods 16th Ed.	ASTM	USGS [1]	Other
35. Mercury—Total [3], mg/L:					
Cold vapor, manual or	245.1	303F	D3223—80	I—3462—84	33.095.[3]
Automated	245.2				
36. Molybdenum—Total [3], mg/L: Digestion [5] followed by:					
AA direct aspiration	246.1	303C		I—3490—84	
AA furnace, or	246.2	304			
Inductively coupled plasma					200.7.[4]
37. Nickel—Total [3], mg/L: Digestion [5] followed by:					
AA direct aspiration	249.1	303 A or B	D1886—84 (C or D).	I—3499—84	
AA furnace	249.2	304			
Inductively coupled plasma, or					200.7.[4]
Colorimetric (Heptoxime)		321B			
38. Nitrate (as N), mg/L: Colorimetric (Brucine sulfate), or Nitrate-nitrite N minus Nitrite N (See parameters 39 and 40).	352.1		D992—71		33.063 [3], 419D [16], p. 28.[8]
39. Nitrate-nitrite (as N), mg/L: Cadmium reduction, Manual or	353.3	418C	D3867—85(B)		
Automated, or	353.2	418F	D3867—85(A)	I—4545—84	
Automated hydrazine	353.1				
40. Nitrite (as N), mg/L: Spectrophotometric:					
Manual or	354.1	419	D1254—67		Note 24.
Automated (Diazotization)				I—4540—84	
41. Oil and grease—Total recoverable, mg/L: Gravimetric (extraction).	413.1	503A			
42. Organic carbon—Total (TOC), mg/L: Combustion or oxidation.	415.1	505	D2579.85 (A or B).		33.044 [3], p. 4.[33]
43. Organic nitrogen (as N) mg/L: Total Kjeldahl N (Parameter 31) minus ammonia N (Parameter 4.).					
44. Orthophosphate (as P), mg/L: Ascorbic acid method:					
Automated or	365.1	424G		I—4601—84	33.116. [3]
Manual single reagent	365.2	424F	D515—82(A)		33.111. [3]
or Manual two reagent	365.3				
45. Osmium—Total [3], mg/L: Digestion [5] followed by:					
AA direct aspiration, or	252.1	303C			
AA furnace	252.2	304			
46. Oxygen dissolved, mg/L: Winkler (Azide modification), or	360.2	421B	D888—81(C)	I—1575—78 [7]	33.026. [8]
Electrode	360.1	421F		I—1576—78 [7]	
47. Palladium—Total [3], mg/L: Digestion [5] followed by:					
AA direct aspiration, or	253.1				P. S27.[9]
AA furnace	253.2				P. S28.[9]
48. Phenols, mg/L:					
Manual distillation [35]	420.1		D1783—80 (A or B).		Note 26.
Followed by:					
Colorimetric (4AAP) manual, or	420.1				Note 26.
Automated [19]	420.2				
49. Phosphorus (elemental) mg/L: Gas-liquid chromatography.					Note 27.
50. Phosphorus—Total, mg/L: Persulfate digestion followed by	365.2	424C(III)			33.111.[3]
Manual or	365.2 or 365.3.	424F	D515—82(A)		
Automated ascorbic acid reduction, or.	365.1	424G		I—4600—84	33.116.[3]
Semi-automated block digestor	365.4				

Parameter, units, and method	Reference (method No. or page)				
	EPA 1979	Standard methods 16th Ed.	ASTM	USGS [1]	Other
51. Platinum—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration, or	255.1	303A			
AA furnace	255.2	304			
52. Potassium—total [3], mg/L: Digestion followed by:					
AA direct aspiration	258.1	303A		I-3630-84	33.103.[3]
Inductively coupled plasma					200.7.[4]
Flame photometric, or		322B	D1428-82(A)		
Colorimetric (Cobaltinitrate)					317B.[16]
53. Residue—Total, mg/L: Gravimetric, 103-105°C.	160.3	209A		I-3750-84	
54. Residue—filterable, mg/L: Gravimetric, 180°C.	160.1	209B		I-1750-84	
55. Residue—nonfilterable, (TSS), mg/L: Gravimetric, 103-105°C post washing of residue.	160.2	209C		I-3765-84	
56. Residue—settleable, mg/L: Volumetric, (Imhoff cone) or gravimetric.	160.5	209E		1	
57. Residue—Volatile, mg/L: Gravimetric, 550°C.	160.4	209D		I-3753-84	
58. Rhodium—Total [3], mg/L: Digestion [3] followed by:.					
AA direct aspiration, or	265.1	303A			
AA furnace	265.2	304			
59. Ruthenium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration, or	267.1	303A			
AA furnace	267.2	304			
60. Selenium—Total [3], mg/L: Digestion [3] followed by:					
AA furnace	270.2	304			
Inductively coupled plasma, or					200.7[4]
AA gaseous hydride	270.3	303E	D3859-84(A)	I-3667-84	
61. Silica—Dissolved, mg/L: 0.45 micron filtration followed by:					
Colorimetric, Manual or	370.1	425C	D859-80(B)	I-1700-84	
Automated (Molybdosilicate), or				I-2700-84	
Inductively coupled plasma					200.7.[4]
62. Silver—Total [30], mg/L: Digestion [3] followed by:					
AA direct aspiration	272.1	303 A or B		I-3720-84	33.089 [3], p. 37.[9]
AA furnace.	272.2	304			
Colorimetric (Dithizone), or					319B.[16]
Inductively coupled plasma					200.7.[4]
63. Sodium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	273.1	303A		I-3735-84	33.107.[3]
Inductively coupled plasma, or					200.7.[4]
Flame photometric		325B	D1428-82(A)		
64. Specific conductance, micromhos/cm at 25°C: Wheatstone bridge	120.1	205	D1125-82(A)	I-1780-84	33.002.[3]
65. Sulfate (as SO4), mg/L:					
Automated colorimetric, (barium chloranilate).	375.1				
Gravimetric, or	375.3	426 A or B	D516-82(A)		33.124.[3]
Turbidimetric.	375.4		D516-82(B)		426C.[19]
66. Sulfide (as S), mg/L:					
Titrimetric (iodine) or	376.1	427D		I-3840-84	228A.[30]
Colorimetric (methylene blue)	376.2	427C			
67. Sulfite (as SO3), mg/L: Titrimetric (iodine-iodate)	377.1	428A	D1339-84(C)		
68. Surfactants, mg/L: Colorimetric (methylene blue).	425.1	512B	D2330-82(A).		
69. Temperature, °C., Thermometric	170.1	212			Note 31

Parameter, units, and method	Reference (method No. or page)				
	EPA 1979	Standard methods 16th Ed.	ASTM	USGS [1]	Other
70. Thallium—Total [3], mg/L. Digestion [3] followed by:					
AA direct aspiration	279.1	303A			
AA furnace, or	279.2	304			
Inductively coupled plasma					200.7 [4]
71. Tin—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration, or	282.1	303A		I-3850-78 [1]	
AA furnace	282.2	304			
72. Titanium—Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration, or	283.1	303C			
AA furnace	283.2	304			
73. Turbidity, NTU: Nephelometric	180.1	214A	D1889-81	I-3860-84	
74. Vanadium, Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	286.1	303C			
AA furnace	286.2	304			
Inductively coupled plasma, or					200.7 [4]
Colormetric (Gallic acid)		327B	D3373-84(A)		
75. Zinc-Total [3], mg/L: Digestion [3] followed by:					
AA direct aspiration	289.1	303A or B	D1691-84 (C or D)	I-3900-84	33.089 [2], p. 37. [5]
AA furnace	289.2	304			
Inductively coupled plasma, or					200.7 [4]
Colormetric (Dithizone) or		328C			
(Zincon)					Note 32.

[1] "Methods for Analysis of Inorganic Substances in Water and Fluvial Sediments," U.S. Department of the Interior, U.S. Geological Survey, Open-File Report 85-495, 1986, unless otherwise stated.

[2] "Official Methods of Analysis of the Association of Official Analytical Chemists," methods manual, 14th ed. (1985).

[3] For the determination of total metals the sample is not filtered before processing. A digestion procedure is required to solubilize suspended material and to destroy possible organic-metal complexes. Two digestion procedures are given in "Methods for Chemical Analysis of Water and Wastes, 1979." One (section 4.1.3), is a vigorous digestion using nitric acid. A less vigorous digestion using nitric and hydrochloric acids (Section 4.1.4) is preferred; however, the analyst should be cautioned that this mild digestion may not suffice for all sample types. Particularly, if a colormetric procedure is to be employed, it is necessary to ensure that all organo-metallic bonds be broken so that the metal is in a reactive state. In those situations, the vigorous digestion is to be preferred making certain that at no time does the sample go to dryness. Samples containing large amounts of organic materials would also benefit by this vigorous digestion. Use of the graphite furnace technique, inductively coupled plasma, as well as determinations for certain elements such as arsenic, the noble metals, mercury, selenium, and titanium require a modified digestion and in all cases the method write-up should be consulted for specific instruction and/or cautions.

NOTE: If the digestion included in one of the other approved references is different than the above, the EPA procedure must be used.

Dissolved metals are defined as those constituents which will pass through a 0.45 micron membrane filter. Following filtration of the sample, the referenced procedure for total metals must be followed. Sample digestion for dissolved metals may be omitted for AA (direct aspiration or graphite furnace) and ICP analyses provided the sample solution to be analyzed meets the following criteria:
a. has a low COD (<20)
b. is visibly transparent with a turbidity measurement of 1 NTU or less.
c. is colorless with no perceptible odor, and
d. is of one liquid phase and free of particulate or suspended matter following acidification.

[4] The full text of Method 200.7, "Inductively Coupled Plasma Atomic Emission Spectrometric Method for Trace Element Analysis of Water and Wastes," is given at Appendix C of this Part 136.

[5] Manual distillation is not required if comparability data on representative effluent samples are on company file to show that this preliminary distillation step is not necessary; however, manual distillation will be required to resolve any controversies.

[6] Ammonia, Automated Electrode Method, Industrial Method Number 379-75 WE, dated February 19, 1976, Technicon AutoAnalyzer II, Technicon Industrial Systems, Tarrytown, NY, 10591.

[7] The approved method is that cited in "Methods for Determination of Inorganic Substances in Water and Fluvial Sediments", USGS TWRI, Book 5, Chapter A1 (1979).

[8] American National Standard on Photographic Processing Effluents, Apr. 2, 1975. Available from ANSI, 1430 Broadway, New York, NY 10018.

[9] "Selected Analytical Methods Approved and Cited by the United States Environmental Protection Agency," Supplement to the Fifteenth Edition of Standard Methods for the Examination of Water and Wastewater (1981).

[10] The use of normal and differential pulse voltage ramps to increase sensitivity and resolution is acceptable.

[11] Carbonaceous biochemical oxygen demand (CBOD$_5$) must not be confused with the traditional BOD$_5$ test which measures "total BOD." The addition of the nitrification inhibitor is not a procedural option, but must be included to report the CBOD$_5$ parameter. A discharger whose permit requires reporting the traditional BOD$_5$ may not use a nitrification inhibitor in the procedure for reporting the results. Only when a discharger's permit specifically states CBOD$_5$ is required, can the permittee report data using the nitrification inhibitor.

TABLE E-5. (Continued)

[12] OIC Chemical Oxygen Demand Method. Oceanography International Corporation. 512 West Loop. P.O. Box 2980. College Station. TX 77840

[13] Chemical Oxygen Demand, Method 8000, Hach Handbook of Water Analysis. 1979. Hach chemical Company. P.O. Box 389. Loveland. CO 80537

[14] The back titration method will be used to resolve controversy.

[15] Orion Research Instruction Manual. Residual Chlorine Electrode Model 97-70. 1977. Orion Research Incorporated. 840 Memorial Drive. Cambridge. MA 02138.

[16] The approved method is that cited in Standard Methods for the Examination of Water and Wastewater. 14th Edition. 1976.

[17] National Council of the Paper Industry for Air and Stream Improvement. (Inc.) Technical Bulletin 253. December 1971.

[18] Copper. Biocinchoinate Method, Method 8506, Hach Handbook of Water Analysis. 1979. Hach Chemical Company. P.O. Box 389. Loveland. CO 80537.

[19] After the manual distillation is completed. the autoanalyzer manifolds in EPA Methods 335.3 (cyanide) or 420.2 (phenols) are simplified by connecting the re-sample line directly to the sampler. When using the manifold setup shown in Method 335.3. the buffer 6.2 should be replaced with the buffer 7.6 found in Method 335.2.

[20] Hydrogen Ion (pH) Automated Electrode Method, Industrial Method Number 378-75WA, October 1976. Technicon Auto-Analyzer II, Technicon Industrial Systems. Tarrytown. NY 10591.

[21] Iron. 1,10-Phenanthroline Method, Method 8008, 1980, Hach Chemical Company. P.O. Box 389. Loveland. CO 80537

[22] Manganese. Periodate Oxidation Method. Method 8034. Hach Handbook of Wastewater Analysis. 1979. pages 2-113 and 2-117. Hach Chemical Company. Loveland. CO 80537.

[23] Goerlitz. D.. Brown. E.. "Methods for Analysis of Organic Substances in Water." U.S. Geological Survey Techniques of Water-Resources Inv.. book 5, ch. A3, page 4 (1972).

[24] Nitrogen. Nitrite. Method 8507. Hach Chemical Company. P.O. Box 389. Loveland. CO 80537.

[25] Just prior to distillation. adjust the sulfuric-acid-preserved sample to pH 4 with 1 + 9 NaOH.

[26] The approved method is that cited in Standard Methods for the Examination of Water and Wastewater. 14th Edition. The colorimetric reaction is conducted at a pH of 10.0±0.2. The approved methods are given on pp. 576-81 of the 14th Edition: Method 510A for distillation. Method 510B for the manual colorimetric procedure. or Method 510C for the manual spectrophotometric procedure.

[27] R. F. Addison and R. G. Ackman. "Direct Determination of Elemental Phosphorus by Gas-Liquid Chromatography." Journal of Chromatography. vol. 47, No. 3. pp. 421-426, 1970.

[28] Approved methods for the analysis of silver in industrial wastewaters at concentrations of 1 mg/L and above are inadequate where silver exists as an inorganic halide. Silver halides such as the bromide and chloride are relatively insoluble in reagents such as nitric acid but are readily soluble in an aqueous buffer of sodium thiosulfate and sodium hydroxide to a pH of 12. Therefore. for levels of silver above 1 mg/L, 20 mL of sample should be diluted to 100 mL by adding 40 mL each of 2 M Na₂S₂O₃ and 2M NaOH. Standards should be prepared in the same manner. For levels of silver below 1 mg/L the approved method is satisfactory.

[29] The approved method is that cited in Standard Methods for the Examination of Water and Wastewater. 15th Edition.

[30] The approved method is that cited in Standard Methods for the Examination of Water and Wastewater. 13th Edition.

[31] Stevens. H. H.. Ficke. J. F.. and Smoot. G. F.. "Water Temperature—Influential Factors, Field Measurement and Data Presentation." U.S. Geological Survey. Techniques of Water Resources Investigations. Book 1, Chapter D1, 1975.

[32] Zinc. Zincon Method. Method 8009. Hach Handbook of Water Analysis. 1979. pages 2-231 and 2-333. Hach Chemical Company. Loveland. CO 80537.

TABLE E-6. LIST OF TEST PROCEDURES APPROVED BY U.S. EPA FOR NON-PESTICIDE ORGANIC COMPOUNDS

Note: This table is an exact reproduction of Table 1C in 40 CFR 136.3.

Parameter [1]	EPA Method Number [2] [3]			Other
	GC	GC/MS	HPLC	
1. Acenaphthene	610	625, 1625	610	
2. Acenaphthylene	610	625, 1625	610	
3. Acrolein	603	*624, 1624		
4. Acrylonitrile	603	*624, 1624		
5. Anthracene	610	625, 1625	610	
6. Benzene	602	624, 1624		
7. Benzidine		‵625, 1625	605	Note 3, p. 1:
8. Benzo(a)anthracene	610	625, 1625	610	
9. Benzo(a)pyrene	610	625, 1625	610	
10. Benzo(b)fluoranthene	610	625, 1625	610	
11. Benzo(g,h,i)perylene	610	625, 1625	610	
12. Benzo(k)fluoranthene	610	625, 1625	610	
13. Benzyl chloride				Note 3. p. 130: Note 6. p. S102.
14. Benzyl butyl phthalate	606	625, 1625		
15. Bis(2-chloroethoxy) methane	611	625, 1625		
16. Bis(2-chloroethyl) ether	611	625, 1625		
17. Bis(2-ethylhexyl) phthalate	606	625, 1625		
18. Bromodichloromethane	601	624, 1624		
19. Bromoform	601	624, 1624		
20. Bromomethane	601	624, 1624		
21. 4-Bromophenylphenyl ether	611	625, 1625		
22. Carbon tetrachloride	601	624, 1624		Note 3. p 130
23. 4-Chloro-3-methylphenol	604	625, 1625		
24. Chlorobenzene	601, 602	624, 1624		Note 3. p. 130.
25. Chloroethane	601	624, 1624		
26. 2-Chloroethylvinyl ether	601	624, 1624		

Parameter	EPA Method Number			Other
	GC	GC,MS	HPLC	
27 Chloroform	601	624. 1624		Note 3. p 130
28 Chloromethane	601	624. 1624		
29 2-Chloronaphthalene	612	625. 1625		
30 2-Chlorophenol	604	625. 1625		
31 4-Chlorophenylphenyl ether	611	625. 1625		
32 Chrysene	610	625. 1625	610	
33 Dibenzo(a,h)anthracene	610	625. 1625	610	
34 Dibromochloromethane	601	624. 1624		
35 1,2-Dichlorobenzene	601 602. 612	624. 625. 1625		
36 1,3-Dichlorobenzene	601. 602. 612	624. 625. 1625		
37 1,4-Dichlorobenzene	601. 602. 612	625. 1624. 1625		
38 3,3'-Dichlorobenzidine		625. 1625	605	
39 Dichlorodifluoromethane	601			
40 1,1-Dichloroethane	601	624. 1624		
41 1,2-Dichloroethane	601	624. 1624		
42 1,1-Dichloroethene	601	624. 1624		
43 trans-1,2-Dichloroethene	601	624. 1624		
44 2,4-Dichlorophenol	604	625. 1625		
45 1,2-Dichloropropane	601	624. 1624		
46 cis-1,3-Dichloropropene	601	624. 1624		
47 trans-1,3-Dichloropropene	601	624. 1624		
48 Diethyl phthalate	606	625. 1625		
49 2,4-Dimethylphenol	604	625. 1625		
50 Dimethyl phthalate	606	625. 1625		
51 Di-n-butyl phthalate	606	625. 1625		
52 Di-n-octyl phthalate	606	625. 1625		
53 2,4-Dinitrophenol	604	625. 1625		
54 2,4-Dinitrotoluene	609	625. 1625		
55 2,6-Dinitrotoluene	609	625. 1625		
56 Epichlorohydrin				Note 3. p. 130; Note 6. p. S102.
57 Ethylbenzene	602	624. 1624		
58 Fluoranthene	610	625. 1625	610	
59 Fluorene	610	625. 1625	610	
60 Hexachlorobenzene	612	625. 1625		
61 Hexachlorobutadiene	612	625. 1625		
62 Hexachlorocyclopentadiene	612	625. 1625		
63 Hexachloroethane	612	625. 1625		
64 Ideno(1,2,3-cd)pyrene	610	625. 1625	610	
65 Isophorone	609	625. 1625		
66 Methylene chloride	601	624. 1624		Note 3. p. 130;
67 2-Methyl-4,6-dinitrophenol	604	625. 1625		
68 Naphthalene	610	625. 1625	610	
69 Nitrobenzene	609	625. 1625		
70 2-Nitrophenol	604	625. 1625		
71 4-Nitrophenol	604	625. 1625		
72 N-Nitrosodimethylamine	607	625. 1625		
73 N-Nitrosodi-n-propylamine	607	625. 1625		
74 N-Nitrosodiphenylamine	607	625. 1625		
75 2,2'-Oxybis(1-chloropropane)	611	625. 1625		
76 PCB-1016	608	625		Note 3. p. 43;
77 PCB-1221	608	625		Note 3. p. 43;
78 PCB-1232	608	625		Note 3. p. 43;
79 PCB-1242	608	625		Note 3. p. 43;
80 PCB-1248	608	625		Note 3. p. 43;
81 PCB-1254	608	625		Note 3. p. 43;
82 PCB-1260	608	625		Note 3. p. 43;
83 Pentachlorophenol	604	625. 1625		Note 3. p. 140;
84 Phenanthrene	610	625. 1625	610	
85 Phenol	604	625. 1625		
86 Pyrene	610	625. 1625	610	
87 2,3,7,8-Tetrachlorodibenzo-p-dioxin		613		
88 1,1,2,2-Tetrachloroethane	601	624. 1624		Note 3. p. 130;
89 Tetrachloroethene	601	624. 1624		Note 3. p. 130;
90 Toluene	602	624. 1624		
91 1,2,4-Trichlorobenzene	612	625. 1625		Note 3. p. 130;

Parameter		EPA Method Number -			Other
		GC	GC/MS	HPLC	
92 1,1,1-Trichloroethane		601	624, 1624		
93 1,1,2-Trichloroethane		601	624, 1624		Note 3 p 130
94 Trichloroethene		601	624, 1624		
95 Trichlorofluoromethane		601	624		
96 2,4,6-Trichlorophenol		604	625, 1625		
97 Vinyl chloride		601	624, 1624		

Table IC Notes

' All parameters are expressed in micrograms per liter (µg/L).

' The full text of Methods 601-613, 624, 625, 1624, and 1625, are given at Appendix A. "Test Procedures for Analysis of Organic Pollutants," of this Part 136. The standardized test procedure to be used to determine the method detection limit (MDL) for these test procedures is given at Appendix B. "Definition and Procedure for the Determination of the Method Detection Limit," of this Part 136.

' "Methods for Benzidine; Chlorinated Organic Compounds, Pentachlorophenol and Pesticides in Water and Wastewater." U.S. Environmental Protection Agency, September, 1978.

' Method 624 may be extended to screen samples for Acrolein and Acrylonitrile. However, when they are known to be present, the preferred method for these two compounds is Method 603 or Method 1624.

' Method 625 may be extended to include benzidine, hexachlorocyclopentadiene, N-nitrosodimethyamine, and N-nitrosodiphenylamine. However, when they are known to be present, Methods 605, 607, and 612, or Method 1625, are preferred methods for these compounds.

* 625, Screening only.

' "Selected Analytical Methods Approved and Cited by the United States Environmental Protection Agency." Supplement to the Fifteenth Edition of Standard Methods for the Examination of Water and Wastewater (1981).

' Each analyst must make an initial, one-time, demonstration of their ability to generate acceptable precision and accuracy with Methods 601-613, 624, 625, 1624, and 1625 (See Appendix A of this Part 136) in accordance with procedures each in section 8.2 of each of these Methods. Additionally, each laboratory, on an on-going bases must spike and analyze 10% (5% for Methods 624 and 625 and 100% for methods 1624, and 1625) of all samples to monitor and evaluate laboratory data quality in accordance with sections 8.3 and 8.4 of these Methods. When the recovery of any parameter falls outside the warning limits, the analytical results for that parameter in the unspiked sample are suspect and cannot be reported to demonstrate regulatory compliance.

NOTE: These warning limits are promulgated as an "interim final action with a request for comments."

TABLE E-7. LIST OF TEST PROCEDURES APPROVED BY U.S. EPA FOR PESTICIDES[1]

Note: This table is an exact reproduction of Table 1D in 40 CFR 136.3.

Parameter µg. L)	Method	EPA [3,7]	Stand-ard Meth-ods :5th Ed	ASTM	Other
1. Aldrin	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
2. Ametryn	GC				Note 3. p. 83; Note 6. p. S58.
3. Aminocarb	TLC				Note 3. p. 94; Note 6. p. S16.
4. Atraton	GC				Note 3. p. 83; Note 6. p. S68.
5. Atrazine	GC				Note 3. p. 83; Note 6. p. S68.
6. Azinphos methyl	GC				Note 3. p. 25; Note 6. p. S51.
7. Barban	TLC				Note 3. p. 104; Note 6. p. S64.
8. α-BHC	GC	608	509A	D3086	Note 3. p. 7.
	GC/MS	⁴ 625			
9. β-BHC	GC	608		D3086	
	GC/MS	625			
10. δ-BHC	GC	608		D3086	
	GC/MS	⁴ 625			
11. γ-BHC (Lindane)	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
12. Captan	GC		509A		Note 3. p. 7.
13. Carbaryl	TLC				Note 3. p. 94; Note 6. p. S60.
14. Carbophenothion	GC				Note 4. p. 30; Note 6. p. S73.
15. Chlordane	GC	608	509A	D3086	Note 3. p. 7.
	GC-MS	625			
16. Chloropropham	TLC				Note 3. p. 104; Note 6. p. S64.
17. 2.4-D	GC		509B		Note 3. p. 115; Note 4. p. 35.
18. 4.4'-DDD	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC-MS	625			
19. 4.4'-DDE	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
20. 4.4'-DDT	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			

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Parameter µg/L	Method	EPA[2]	Standard Methods 15th Ed	ASTM	Other
21. Demeton-O	GC				Note 3. p 25; Note 6. p. S51
22. Dementon-S	GC				Note 3. p. 25; Note 6. p. S51
23. Diazinon					Note 3. p. 25; Note 4. p. 30; Note 6. p. S51
24. Dicamba	GC				Note 3. p. 115.
25. Dichlofenthion	GC				Note 4. p. 30; Note 6. p. S73.
26. Dichloran	GC		509A		Note 3. p. 7
27. Dicofol	GC			D3086	
28. Dieldrin	GC	608	509A		Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
29. Dioxathion	GC				Note 4. p. 30; Note 6. p. S73.
30. Disulfoton	GC				Note 3. p. ; Note 6. p. S51.
31. Diuron	TLC				Note 3. p. 104 Note 6. p. S64.
32. Endosulfan I	GC	608	509A	D3086	Note 3. p. 7.
	GC/MS	625			
33. Endosulfan II	GC	608	509A	D3086	Note 3. p. 7.
	GC/MS	625			
34. Endosulfan sulfate	GC	608			
	GC/MS	625			
35. Endrin	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
36. Endrin aldehyde	GC	608			
	GC/MS	625			
37. Ethion	GC				Note 4. p. 30; Note 6. p. S73.
38. Fenuron	TLC				Note 3. p. 104; Note 6. p. S64.
39. Fenuron-TCA	TLC				Note 3. p. 104; Note 6. p. S64.
40. Heptachlor	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
41. Heptachlor epoxide	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30; Note 6. p. S73.
	GC/MS	625			
42. Isodrin	GC				Note 4. p. 30; Note 6. p. S73.
43. Linuron	TLC				Note 3. p. 104; Note 6. p. S64.
44. Malathion	GC		509A		Note 3. p. 25; Note 4. p. 30; Note 6. p. S51.
45. Methiocarb	TLC				Note 3. p. 94; Note 6. p. S60.
46. Methoxychlor	GC		509A	D3086	Note 3. p. 7; Note 4. p. 30.
47. Mexacarbate	TLC				Note 3. p. 94; Note 6. p. S60.
48. Mirex	GC		509A		Note 3. p. 7.
49. Monuron	TLC				Note 3. p. 104; Note 6. p. S64.
50. Monuron-TCA	TLC				Note 3. p. 104; Note 6. p. S64.
51. Neburon	TLC				Note 3. p. 104; Note 8. p. S64.
52. Parathion methyl	GC		509A		Note 3. p. 25; Note 4. p. 30.
53. Parathion ethyl	GC		509A		Note 3. p. 25.
54. PCNB	GC		509A		Note 3. p. 7.
55. Perthane	GC			D3086	
56. Prometron	GC				Note 3. p. 83; Note 6. p. S68.
57. Prometryn	GC				Note 3. p. 83; Note 6. p. S68.
58. Propazine	GC				Note 3. p. 83; Note 6. p. S68.
59. Propham	TLC				Note 3. p. 104; Note 6. p. S64.
60. Propoxur	TLC				Note 3. p. 94; Note 6. p. S60.
61. Secbumeton	TLC				Note 3. p. 83; Note 6. p. S68.
62. Siduron	TLC				Note 3. p. 104; Note 6. p. S64.
63. Simazine	GC				Note 3. p. 83; Note 6. p. S68.
64. Strobane	GC		509A		Note 3. p. 7.
65. Swep	TLC				Note 3. p. 104; Note 6. p. S64.
66. 2.4.5-T	GC		509B		Note 3. p. 115; Note 4. p. 35.
67. 2.4.5-TP (Silvex)	GC		509B		Note 3. p. 115.
68. Terbuthylazine	GC				Note 3. p. 83; Note 6. p. S68.
69. Toxaphene	GC	608	509A	D3086	Note 3. p. 7; Note 4. p. 30.
	GC/MS	625			
70. Trifluralin	GC		509A		Note 3. p. 7

Table ID Notes

[1] Pesticides are listed in this table by common name for the convenience of the reader. Additional pesticides may be found under Table IC. where entries are listed by chemical name.

TABLE E-7. (Continued)

² The full text of methods 608 and 625 are given at Appendix A. "Test Procedures for Analysis of Organic Pollutants" of this Part 136. The standardized test procedure to be used to determine the method detection limit (MDL) for these test procedures is given at Appendix B. "Definition and Procedure for the Determination of the Method Detection Limit", of this Part 136

³ "Methods for Benzidine, Chlorinated Organic Compounds, Pentachlorophenol and Pesticides in Water and Wastewater," U.S. Environmental Protection Agency, September, 1978. This EPA publication includes thin-layer chromatography (TLC) methods.

⁴ "Methods for Analysis of Organic Substances in Water." U.S. Geological Survey. Techniques of Water-Resources Investigations, Book 5, Chapter A3 (1972).

⁵ The method may be extended to include a-BHC, δ-BHC, endosulfan I, endosulfan II, and endrin. However, when they are known to exist, Method 608 is the preferred method.

⁶ "Selected Analytical Methods Approved and Cited by the United States Environmental Protection Agency." Supplement to the Fifteenth Edition of *Standard Methods for the Examination of Water and Wastewater* (1981).

⁷ Each analyst must make an initial, one-time, demonstration of their ability to generate acceptable precision and accuracy with Methods 608 and 625 (See Appendix A of this Part 136) in accordance with procedures given in section 8.2 of each of these methods. Additionally, each laboratory, on an on-going basis, must spike and analyze 10% of all samples analyzed with Method 608 or 5% of all samples analyzed with Method 625 to monitor and evaluate laboratory data quality in accordance with Sections 8.3 and 8.4 of these methods. When the recovery of any parameter falls outside the warning limits, the analytical results for that parameter in the unspiked sample are suspect and cannot be reported to demonstrate regulatory compliance.

NOTE: These warning limits are promulgated as an "interim final action with a request for comments."

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TABLE E-8. RECOMMENDED SAMPLE SIZES, CONTAINERS, PRESERVATION, AND HOLDING TIMES FOR EFFLUENT SAMPLES[a]

Measurement	Minimum Sample Size[b] (mL)	Container[c]	Preservative[d]	Maximum Holding Time
pH	25	P, G	None	Analyze immediately[e]
Temperature	1,000	P, G	None	Measure immediately[e]
Turbidity	100	P, G	Cool, 4° C	48 h
Total suspended solids	1,000	P, G	Cool, 4° C	7 days
Settleable solids	1,000	P, G	Cool, 4° C	48 h
Floating particulates	5,000	P, G	None	Analyze immediately[e,f]
Dissolved oxygen				
Probe	300	G bottle and top	None	Analyze immediately[e]
Winkler	300	G bottle and top	Fix onsite; store in dark	8 h
Biochemical oxygen demand	1,000	P, G	Cool, 4° C	48 h
Total chlorine residual	200	P, G	None	Analyze immediately[e]
Oil and grease	1,000	G only	Cool, 4° C H_2SO_4 to pH<2	28 days
Nitrogen				
Ammonia-N	400	P, G	Cool, 4° C H_2SO_4 to pH<2	28 days
Total Kjeldahl-N	500	P, G	Cool, 4° C H_2SO_4 to pH<2	28 days
Nitrate+Nitrite-N	100	P, G	Cool, 4° C H_2SO_4 to pH<2	28 days
Phosphorus (total)	50	P, G	Cool, 4° C H_2SO_4 to pH<2	28 days
Priority pollutant metals				
Metals, except mercury	100	P, G	HNO_3 to pH<2	6 mo
Mercury	100	P, G	HNO_3 to pH<2	28 days
Priority pollutant organic compounds				
Extractable compounds (includes phthalates, nitrosamines, organochlorine pesticides, PCBs, nitroaromatics, isophorone, polycyclic aromatic hydrocarbons, haloether, chlorinated hydrocarbons, phenols, and TCDD)	4,000	G only, TFE-lined cap	Cool, 4° C 0.008% $Na_2S_2O_3$[g] Store in dark	7 days until extraction 40 days after extraction
Purgeable compounds	40	G only, TFE-lined septum	Cool, 4° C 0.008% $Na_2S_2O_3$[g]	7 days[h]

TABLE E-8. (Continued

Measurement	Minimum Sample Size[b] (mL)	Container[c]	Preservative[d]	Maximum Holding Time
Total and fecal coliform bacteria	250-500	P, G	Cool, 4° C 0.008% $Na_2S_2O_3$[g]	6 h
Enterococcus bacteria	250-500	P, G	Cool, 4° C 0.008% $Na_2S_2O_3$[g]	6 h

[a] Reference: Adapted from U.S. EPA (1979b), 40 CFR Part 136.

[b] Recommended field sample sizes for one laboratory analysis. If additional laboratory analyses are required (e.g., replicates), the field sample size should be adjusted accordingly.

[c] P = Polyethylene; G = Glass.

[d] Sample preservation should be performed immediately upon sample collection. For composite samples, each aliquot should be preserved at the time of collection. When use of an automated sampler makes it impossible to preserve each aliquot, the samples should be maintained at 4° C until compositing.

[e] Immediately means as soon as possible after the sample is collected, generally within 15 min (U.S. EPA 1984).

[f] No recommended holding time is given by U.S. EPA for floating particulates. Analysis should therefore be made as soon as possible.

[g] Should only be used in the presence of chlorine residual.

[h] Holding time and preservation technique for purgeable compounds are based on the use of U.S. EPA Method 624 for screening all priority pollutant volatiles organic compounds, including acrolein and acrylonitrile. If analysis of acrolein and acrylonitrile is found to be of concern, a separate subsample should be preserved by adjusting the pH to 4-5 and the sample should then be analyzed by U.S. EPA Method 603.

QA/QC

QA/QC procedures should be detailed in the quality assurance project plan (QAPP) (U.S. EPA 1979c; Tetra Tech 1987). The following items should be discussed in the QAPP:

- Statement and prioritization of study objectives

- Responsibilities of personnel associated with sample collection and analysis

- Sampling locations, frequency, and procedures

- Variables to be measured, sample sizes, sample containers, preservatives, and sample holding times

- Equipment checklist

- Sample splits or performance samples to be submitted with the samples

- Sample handling, packaging, labeling, and shipping requirements

- Laboratories to which samples will be shipped.

Tetra Tech (1987) provides QA/QC guidance for the following activities:

- Preparation for sampling program

- Sample collection

- Sample processing

- Sample size

- Sample containers

- Sample preservation

- Sample holding times

- Sample shipping

- Recordkeeping

- Labeling

- Custody procedures

- Analytical methods

- Calibration and preventive maintenance

- Quality control checks

- Corrective action

- Data reporting requirements.

Field Sampling Procedures

For the field sampling effort, the following procedures are recommended:

- Establish and implement chain-of-custody protocols to track samples from the point of collection to final disposition

- Establish and implement protocols to prepare sample containers

- Prepare field "blank" samples to assess potential sample contamination by the sampling devices

- Prepare "trip blanks" to assess potential contamination by volatile organic analytes en route to the laboratory (1 trip blank per sample shipment)

- Collect replicate samples to assess sample precision and the homogeneity of samples collected

- Use appropriate sample collection procedures (see Table E-8).

Volatile organic samples and split composite samples should be collected carefully. Grab samples for volatile organic analyses should be collected in duplicate. Residual chlorine should be eliminated, and the volatile sample containers should be filled with a minimum of mixing and to capacity leaving no headspace. When splitting composite samples into discrete aliquots for analyses, the composite sample should be mixed to provide a homogeneous mixture. A representative portion of any solids in the container should be suspended in the composite sample. Composite samples may be homogenized by hand stirring with clean glass rods or by mechanical stirring with teflon-coated paddles. Metal mixing devices should not be used.

Laboratory Procedures

Laboratory analytical results must be accurate and reliable. Laboratory QA/QC procedures are generally specified for each different analytical method, and the level of QA/QC and associated deliverables vary among methods (Tables E-5 to E-7). The following documentation is required by the analytical laboratory for QA review of data on organic substances (see Tables E-6 and E-7):

- Initial multipoint calibration

- Demonstration of method proficiency

- Determination of method detection limit [usually 5-10 ppb for base, neutral, and acid organic compounds (U.S. EPA Method 625); 0.005-0.10 ppb for pesticide/PCB analysis (U.S. EPA Method 608); and 1-10 ppb for volatiles (U.S. EPA Method 624)]

- Daily checks of calibration and instrument tuning

- Daily analysis of method blanks (1 blank/20 samples)

- Analysis of duplicate samples (minimum of 5 percent of samples analyzed) and conduct of matrix spikes to determine organic recoveries.

The following documentation is required by the analytical laboratory for QA review of data on inorganic substances (see Table E-5):

- Multipoint calibration

- Analysis of reagent blanks

- Matrix spikes of 0.5-5 times the sample concentration

- Determination of method detection limits

- Analysis of full method blanks at a minimum frequency of every 20 samples, rather than reagent water blanks

- Verification of calibration by analysis of standards (daily or with every 10 sample batches)

■ Performance of duplicate analyses for a minimum of 5 percent of the total number of samples analyzed

■ Use of the method of standard additions for samples demonstrating interferences.

Data Evaluation

Data generated from the monitoring program should be evaluated using the step-wise approach discussed below.

1. Assemble the original raw data reports and the associated QA/QC data. The analytes and analytical methods used will determine the types of QA/QC data generated, and may include the following:

■ Sample results

■ Blank sample results

■ Instrument calibrations (initial and continuing)

■ Matrix spike/matrix spike duplicate results

■ Surrogate recovery data

■ Instrument tuning data

■ Chain-of-custody records

■ Analytical request forms

■ Gas chromatograms

■ Mass spectra

- Instrument detection limit determinations

- Serial dilution results

- Clean-water precision and accuracy studies

- Furnace atomic absorption quality control data

- Interference check data

- Laboratory control sample results

- Holding time documentation.

2. Because the resulting data will be used to determine regulatory compliance of the discharge, the following sequence is recommended to conduct a QA review of the data:

- Confirm the sample identifier, container, and preservation with chain-of-custody records

- Confirm the analytical (e.g., extraction or digestion) procedure used with the procedure requested

- Confirm that an acceptable instrument detection limit was achieved

- Confirm that the analysis proceeded in the manner specified

- Confirm that all quality control data deliverables specified by the analytical protocol have been submitted

- Confirm that the analysis was performed within the specified sample holding time

- Confirm that the instrumentation used was properly calibrated initially and that the method was validated

- Confirm detection limits, precision, and accuracy for each substance and review duplicate analysis results

- Confirm that blank samples were analyzed and that the field sampling and analytical procedures did not contaminate the data

- Evaluate the presence of matrix interferences through the use of surrogate recoveries and matrix spikes

- Annotate the data with appropriate qualifiers, and note deviations from prescribed methods

- Detail problems associated with the analyses.

UPGRADING TO A FULL-SCALE SECONDARY TREATMENT FACILITY

Data obtained from the monitoring program described above will be used to determine the mean and peak concentrations and site-specific toxic pollutant removal capabilities for secondary treatment. Performance of the secondary treatment pilot plant will be closely related to the attention and expertise of the operator controlling the plant. If the pilot plant is not properly operated, the data will not approximate the removals that could be achieved with a full-scale facility. Conventional pollutant data (e.g., suspended solids, BOD) can be used to determine when the pilot plant is operating within the expected design removal efficiencies. The most important factor in performing site-specific toxic pollutant removal investigations is to ensure that an acclimated biological seed exists prior to initiating sample collection for pollutant analyses.

Plant operators should be aware that activated sludge microorganisms are susceptible to biological and chemical effects that may kill the organisms or severely inhibit their effectiveness. Accumulations of toxic waste components (via gradual concentration from continuous discharges, or sudden slugs) could limit the ability of the activated sludge system to achieve design effluent quality (see Tables E-9 and E-10). Disruptions or changes could be found by reviewing operating records (e.g., settling characteristics of secondary sludge, species populations in the MLSS). If inhibition or upset conditions are found, the concentration and source of each pollutant of concern should be determined. Concentrations shown in Tables E-9 and E-10 are not absolute and should be used only for comparison purposes and preliminary investigations.

Toxic pollutant removal efficiencies at the secondary treatment pilot plant may be greater than those expected in a full-scale secondary treatment facility. The pilot plant will be operated at a constant flow rate and will not be subject to the diurnal and seasonal flow fluctuations normally

TABLE E-9. REPORTED VALUES FOR ACTIVATED SLUDGE BIOLOGICAL
PROCESS TOLERANCE LIMITS OF ORGANIC PRIORITY POLLUTANTS

Pollutant	Threshold of Inhibitory Effect (mg/L)[a]
Acenaphthene	NI[b] at 10
Acrolein	NI at 62
Acrylonitrile	NI at 152
Benzene	125
Benzidine	5
Carbon tetrachloride	NI at 10
Chlorobenzene	NI at 1
1,2,4-Trichlorobenzene	NI at 6
Hexachlorobenzene	5
1,2-Dichloroethane	NI at 258
1,1,1-Trichloroethane	NI at 10
Hexachloroethane	NI at 10
1,1-Dichloroethane	NI at 10
1,1,2-Trichloroethane	NI at 5
1,1,2,2-Tetrachloroethane	NI at 201
bis-(2-Chloroethyl) ether	NI at 10
2-Chloroethyl vinyl ether	NI at 10
2-Chloronaphthalene	NI at 10
2,4,6-Trichlorophenol	50
para-Chloro-meta-cresol	NI at 10
Chloroform	NI at 10
2-Chlorophenol	NI at 10
1,2-Dichlorobenzene	5
1,3-Dichlorobenzene	5
1,4-Dichlorobenzene	5
1,1-Dichloroethylene	NI at 10
1,2-trans-Dichloroethylene	NI at 10
2,4-Dichlorophenol	NI at 75
1,2-Dichloropropane	NI at 182
1,3-Dichloropropylene	NI at 10
2,4-Dimethylphenol	NI at 10
2,4-Dinitrotoluene	5
2,6-Dinitrotoluene	5
1,2-Diphenylhydrazine	5
Ethylbenzene	NI at 10
Fluoranthene	NI at 5

Pollutant	Threshold of Inhibitory Effect (mg/L)[a]
bis-(2-Chloroisopropyl) ether	NI[b] at 10
Chloromethane	NI at 180
Bromoform	NI at 10
Dichlorobromomethane	NI at 10
Trichlorofluoromethane	NI at 10
Chlorodibromomethane	NI at 10
Hexachlorobutadiene	NI at 10
Hexachlorocyclopentadiene	NI at 10
Isophorone	NI at 15.4
Naphthalene	500
Nitrobenzene	500
2-Nitrophenol	NI at 10
4-Nitrophenol	NI at 10
2,4-Dinitrophenol	1
N-Nitrosodiphenylamine	NI at 10
N-Nitroso-di-N-propylamine	NI at 10
Pentachlorophenol	0.95
Phenol	200
bis-(2-Ethyl Hexyl) phthalate	NI at 10
Butyl Benzyl phthalate	NI at 10
Di-n-butyl phthalate	NI at 10
Di-n-octyl phthalate	NI at 16.3
Diethyl phthalate	NI at 10
Dimethyl phthalate	NI at 10
Chrysane	NI at 5
Acenaphthylene	NI at 10
Anthracene	500
Fluorene	NI at 10
Phenanthrene	500
Pyrene	NI at 5
Tetrachloroethylene	NI at 10
Toluene	NI at 35
Trichloroethylene	NI at 10
Aroclor-1242	NI at 1
Aroclor-1254	NI at 1
Aroclor-1221	NI at 1
Aroclor-1232	NI at 10
Aroclor-1016	NI at 1

[a] Unless otherwise indicated.

[b] NI = no inhibition at tested concentrations. No concentration is listed if reference lacked concentration data.

Reference: U.S. EPA (1986c).

TABLE E-10. REPORTED VALUES FOR ACTIVATED SLUDGE BIOLOGICAL
PROCESS TOLERANCE LIMITS OF INORGANIC PRIORITY POLLUTANTS

Pollutant	Threshold of Inhibitory Effect (mg/L)
Arsenic	0.1
Cadmium	1
Chromium (VI)	1
Chromium (III)	10
Copper	1
Cyanide	0.1
Lead	0.1
Mercury	0.1
Nickel	1
Silver	5
Zinc	0.03

Reference: U.S. EPA (1986c).

experienced at treatment facilities, nor the slug loadings and batch discharges which POTWs can experience in daily operation. In addition, at the relatively higher aeration rates of the pilot plant system, higher degrees of volatile organics stripping may occur, thereby implying higher levels of removal of biodegradable material than might actually happen under full-scale situations.

DEMONSTRATING COMPLIANCE USING PILOT PLANT DATA

The purpose of operating a secondary treatment pilot plant is to determine the concentrations of toxic substances in the effluent that would be realized if the applicant were providing secondary treatment, rather than less-than-secondary treatment as requested in the 301(h) application. Effluent from the secondary treatment pilot plant is then analyzed to determine the concentration of each toxic substance in the effluent. These concentrations define the maximum allowable concentrations in the discharge of less-than-secondary treated effluent.

To demonstrate secondary equivalency, the applicant must demonstrate that the concentration of each toxic substance in the effluent of the Section 301(h) modified discharge is equal to, or less than, the concentration achieved using the secondary treatment pilot plant. For toxic substances whose concentration in the Section 301(h) modified discharge is greater than the concentration in the secondary treated effluent, the applicant must lower the concentration using either or both of two approaches. The first approach is to establish local limits for such substances, in accordance with the guidance given above. The second approach is to upgrade the treatment process within the POTW. Having implemented either or both of these approaches, the applicant must then provide results of additional effluent analyses to demonstrate that the maximum allowable concentrations of toxic substances are not being exceeded after the proposed controls have been implemented.

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ATTACHMENT 1 TO APPENDIX E

U.S. EPA OFFICE OF WATER ENFORCEMENT AND PERMITS
PROCEDURES FOR DEVELOPING TECHNICALLY BASED LOCAL LIMITS

U.S. EPA OFFICE OF WATER ENFORCEMENT AND PERMITS PROCEDURES FOR DEVELOPING TECHNICALLY BASED LOCAL LIMITS

INTRODUCTION

Publicly owned treatment works (POTWs) which discharge wastewater into marine waters may be granted a waiver under Section 301(h) of the Clean Water Act (CWA) from the requirement for secondary treatment [Section 301(b)(1)(B)]. The Water Quality Act (WQA) of 1987 added a new requirement, the urban area pretreatment program, to Section 301(h) of the CWA for POTWs serving a population of 50,000 or more with respect to toxic pollutants introduced by industrial dischargers. This provision now requires each applicant to demonstrate that it has a pretreatment program in effect for each toxic pollutant which, in combination with the applicant's own treatment of discharges, removes the same amount of a given toxic pollutant as would be removed if the applicant were to apply secondary treatment (as defined in 40 CFR Part 133) and if it had no pretreatment program for the toxic pollutant. This new "secondary removal equivalency" requirement applies only with respect to a toxic pollutant introduced into a POTW by an industrial discharger for which there is no "applicable pretreatment in effect."

Under this new provision, for each toxic pollutant introduced by an industrial user, the applicant must demonstrate either that there is an applicable pretreatment requirement in effect or that it has a secondary removal equivalency program for any toxic pollutant from industrial sources for which there is no applicable pretreatment requirement. Applicable pretreatment requirements may take the form of Federal categorical pretreatment standards, local limits developed in accordance with 40 CFR Part 403, or a combination thereof.

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POTWs must demonstrate that local limits developed are adequate and enforceable. This new CWA provision also requires POTWs to demonstrate that industrial sources are in compliance with all of their pretreatment requirements, including numerical standards set by local limits, and that those requirements will be enforced.

The following discussion provides a review of procedures for developing technically-based local limits. Further details on the various approaches are provided in U.S. EPA's Guidance Manual on the Development and Implementation of Local Discharge Limitations Under the Pretreatment Program (December 1987). Questions about this guidance should be directed to the U.S. EPA Regional Pretreatment Coordinators or to The Office of Water Enforcement and Permits in Washington, DC.

OVERVIEW OF LOCAL LIMITS

Local discharge limitations are requirements developed by a POTW based on local conditions and unique requirements at the POTW. These limits are primarily intended to protect the treatment plant from industrial discharges which could interfere with POTW treatment processes or pass through the treatment plant to receiving waters and adversely impact water quality. Local limits are also designed to prevent sludge contamination and protect workers at the treatment plant.

Local limits are usually developed on a chemical specific basis and are implemented as requirements that individual industrial dischargers must meet. Once adopted, local limits are deemed to be Federal standards for the purposes of the Clean Water Act Section 307(d) prohibition against violating pretreatment standards [40 CFR 403.5(d) and 40 CFR 403.3(j)].

LOCAL LIMITS DEVELOPMENT APPROACHES

U.S. EPA's Guidance on the Development and Implementation of Local Discharge Limitations Under the Pretreatment Program (1987) provides various methods for calculating local limits. The predominant approach

used by POTWs and advocated in the Guidance is a chemical specific approach known as the maximum allowable headworks loading method. This method involves back calculating from environmental and plant protection criteria to a maximum allowable headworks loadings. This is accomplished pollutant by pollutant for each environmental criteria or plant requirement and the lowest or most limiting value for each pollutant serves as the basis for allocation to industry and ultimate local limits. The steps of the maximum allowable headworks loading local limits development process are shown in Figure 1, and discussed below.

Maximum Allowable Headworks Loading Method

Determine Applicable Environmental Criteria--

The first step in developing local limits by the maximum allowable headworks loading method is to determine applicable environmental criteria. Environmental criteria generally include NPDES permit limits, water quality standards or criteria, sludge disposal requirements, and unit process inhibition values. The POTW should use all applicable environmental criteria when developing local limits. Other appropriate requirements may include worker health and safety criteria, collection system effects, incinerator emission requirements or other applicable federal, state, or local environmental protection requirements. Further information on how to incorporate applicable environmental criteria into the local limits development process is contained in the guidance manual.

Another less frequently used environmental criterion is biological toxicity. POTWs that have conducted biological toxicity testing indicating toxicity should develop local limits to correct the toxicity. Although there is no method in the guidance manual to calculate maximum allowable headworks loadings based on the results of toxicity testing, the manual provides guidance and additional references on the Toxicity Reduction Evaluation (TRE) process.

Characterize Existing Loadings--

Industrial Users--During the local limits development process, the POTW must characterize existing loadings to the treatment plant. Local limits should be based on site-specific monitoring data. This can be accomplished by conducting monitoring of all industrial users. Either POTW monitoring or self-monitoring data are acceptable, and information from the POTW's industrial waste survey may also be of use.

Hauled Waste--If hauled wastes are accepted at the POTW, they may be a significant source of toxic pollutant loadings. In such a case the POTW should consider them as a significant nondomestic source in the determination of local limits.

Domestic Loadings--The POTW must also characterize domestic loadings. Site-specific monitoring of a representative portion of the POTW's collection system should form the basis for loadings from domestic/background sources. Use of literature values must be justified by the POTW.

Treatment Plant Monitoring--The POTW must conduct sufficient monitoring at the treatment plant to characterize influent, effluent, and sludge loadings. Monitoring of the treatment plant influent, effluent, and sludge should represent a minimum of 5 consecutive days. Preferably, monitoring should include data for at least 1 day per month over at least 1 yr for metals and other inorganic pollutants, and 1 day of sampling per year for toxic pollutants [priority pollutants and Resource Conservation and Recovery Act (RCRA) Appendix 9 constituents].

Determine Pollutants of Concern--

As one approach for achieving compliance with Section 301(h) regulations, POTWs serving a population of 50,000 or more must demonstrate that applicable pretreatment requirements are in effect for any and all toxic pollutants contributed by an industrial user. Therefore, data should be collected for any toxic pollutants of concern that could reasonably be

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expected to be discharged to the POTW in quantities that could pass through or interfere with the POTW treatment process, contaminate the sludge, or jeopardize worker health and safety or the collection system.

The POTW should perform at least one priority pollutant scan and one RCRA Appendix 9 scan to identify potential pollutants of concern in the influent, effluent, and sludge. The POTW must then address all toxic pollutants (40 CFR 401.15) that are identified in any analysis above detection limits by developing a local limit for each pollutant.

Calculate Maximum Headworks Loadings--

The POTW must calculate the maximum amount (lb/day) of each toxic pollutant contributed by an industrial user or received at the headworks of the treatment plant that will allow the POTW to achieve all of the above applicable environmental criteria. If the POTW does not calculate the maximum allowable headworks loading to the POTW for each toxic pollutant, it must provide justification why it has not done so. The nonconservative pollutants (volatiles) require special consideration when conducting headworks analysis (e.g., alternative formulas and allocation methods). All calculations should be consistent with the approach outlined in the guidance manual.

During this step of the local limits development process, the POTW should demonstrate that an acceptable mass balance exists between the actual loadings of pollutants at the headworks and the estimated loadings of pollutants from specific source discharges. This mass balance can be accomplished by calculating the actual loading of each pollutant from influent monitoring data and comparing this value with the sum of the estimated loadings from all individual sources (e.g., domestic, industrial, hauled waste). The resulting calculated loadings from various sources should be within 80 to 120 percent of the actual total influent loading and flow.

Calculate Allowable Industrial Loadings--

Once the POTW has calculated the maximum allowable headworks loading, a safety factor must be applied and the value discounted for domestic/background loadings in order to determine the maximum allowable allocation available for industrial users. A safety factor is incorporated into the calculations to allow for future industrial growth and other discrepancies that may enter into the calculations because of the use of default data or variations in analytical procedures. The POTW should provide justification for the selected safety factor, which will usually range from 10 to 30 percent.

Allocate Allowable Industrial Loading--

After the POTW has calculated the allowable industrial loading, the method chosen to allocate this loading depends on the number and types of industrial users and the method of application (permits, contract, or sewer use ordinance) employed by the POTW. Where the current loading of a pollutant exceeds the maximum allowable headworks loading, the POTWs must establish a local limit to reduce loadings to within the range of the maximum allowable headworks loading. Where the current loading is far below or approaches the maximum allowable headworks loading, the POTW must set industrial discharge limits at current loadings to maintain the status quo.

The POTW should ensure that it has selected local limits that are reasonable. All local limits should be at or above detection limits and should not be so lenient as to provide industry additional opportunity to pollute or encourage discharge of hazardous waste to the POTW.

Revise Local Limits--

Many variables on which these local limits calculations are based may vary with time. Local limits must be revised on a periodic basis to reflect changes in conditions or assumptions. Conditions which might require that local limits be revised include but are not limited to changes in environ-

mental criteria, availability of additional monitoring data, changes in plant processes, and changes in POTW capacity or configuration.

Implement Local Limits--

Once local limits have been developed, they must be effectively implemented. Local limits should be incorporated into the sewer use ordinance or some form of individual control mechanisms.

OTHER LOCAL LIMITS APPROACHES

Other methods of local limits development have been used by POTWs. They include the collection system approach, industrial user management practice plans, and case-by-case discharge limits. These approaches are briefly described below. U.S. EPA has published extensive guidance on the development and implementation of local limits. Further information on each of these methods and the maximum allowable headworks loading method can be found in the *Guidance Manual on the Development and Implementation of Local Discharge Limitations Under the Pretreatment Program* (U.S. EPA 1987).

Collection System Approach

To apply this method, the POTW identifies pollutants that may cause fire and explosion hazards or other worker health and safety concerns. Pollutants found to be present are evaluated for their propensity to volatilize and are simplistically modeled to evaluate their expected concentration in air. Comparisons are made with worker health exposure criteria and lower explosive limits. Where values are of concern, the POTW may set limits or require development of management practices to control undesirable discharges. The collection system approach may also consider the prohibition of pollutants with specific flashpoints to prevent discharge of ignitable wastes.

Industrial User Management Practice Plans

This approach consists of POTWs requiring industrial users to develop management practices as enforceable pretreatment requirements for the handling of chemicals and wastes. Examples of management practice plans include chemical management practices, best management practices, and spill prevention plans. Management practice plans are usually narrative local limits.

Case-by-Case Discharge Limits

In this approach a POTW sets numeric local limits based on best professional judgment and on available technologies that are known to be economically feasible. This approach is most often used when insufficient data are available to employ the other methods noted above.

ATTACHMENT 2 TO APPENDIX E

U.S. EPA GUIDANCE MANUAL ON THE DEVELOPMENT AND
IMPLEMENTATION OF LOCAL DISCHARGE LIMITATIONS
UNDER THE PRETREATMENT PROGRAM

APPENDIX F

WATER QUALITY-BASED TOXICS CONTROL

WATER QUALITY-BASED TOXICS CONTROL

Most applicants for Section 301(h) modified NPDES permits must demonstrate satisfactorily to the U.S. EPA that discharge from the POTWs to the marine or estuarine waters is in compliance with Section 301(h) of the Clean Water Act (CWA). POTWs must enforce all applicable industrial pretreatment requirements and demonstrate the effectiveness of both industrial and nonindustrial source control programs. (Small dischargers, with service area populations of less than 50,000 people and average dry weather flows of less than 5.0 MGD, are exempt from effluent analysis and industrial pretreatment requirements if they can certify that there are no known or suspected sources of toxic pollutants or pesticides to the POTW.) Section 301(h) industrial source control programs must be consistent with pretreatment regulations and NPDES permit requirements. Under Sections 308 and 402 of the CWA, NPDES permit applicants [including 301(h) POTWs] are required to collect effluent chemical (and possibly toxicity) data and receiving water biological data to assure compliance with state water quality standards. [If no state standards have been developed for specific pollutants at the time of permit issuance, small and large dischargers must then meet U.S. EPA's marine water quality criteria at the boundary of the zone of initial dilution (ZID).]

In 1984, U.S. EPA (1984) recommended that whole-effluent toxicity testing be used as a complement to chemical-specific analyses to assess effluent discharges and determine NPDES permit limitations. [U.S. EPA developed this approach because of certain disadvantages of the chemical-specific techniques (i.e., the difficulty in identifying all potentially toxic pollutants; the antagonistic, synergistic, or additive effects of toxic pollutants; and the possibility of complex chemical interactions).] The integrated approach is recommended to assure the attainment of water

quality standards, to protect designated water uses, and to provide a tool to control pollutants beyond the CWA technology-based requirements [e.g., Best Available Technology Economically Achievable (BAT)].

The Water Quality Act (WQA) of 1987 also emphasized the need for an integrated approach of whole-effluent and chemical-specific analyses. Congress required U.S. EPA to report on methods for establishing and measuring water quality criteria for toxic pollutants through the use of biological monitoring and assessment methods, and pollutant-specific analyses. The WQA also signalled a shift in emphasis from discharge requirements that were based primarily on technology-based pollution controls to requirements that combined both technology-based and water quality-based pollution controls.

In 1985, U.S. EPA's Office of Water Enforcement and Permits (OWEP) and the Office of Water Regulations and Standards (OWRS) prepared the *Technical Support Document for Water Quality-based Toxics Control* (U.S. EPA 1985a). Guidance was provided on the implementation of a biomonitoring policy for the assessment and control of toxics using both the chemical-specific approach and the whole-effluent toxicity approach. The chemical-specific approach uses water quality criteria or state standards to limit specific pollutants directly. The whole-effluent toxicity approach, as described in the technical support document predominantly for non-marine waters, involves the use of test organisms [e.g., *Daphnia spp.* (water flea), *Pimephales Promelas* (fathead minnow)] that are exposed to serial dilutions of municipal or industrial effluent/receiving water to measure acute (rapid response) and/or chronic (long term response) toxicity. The document also provided guidance for each step in the water-quality based toxics control program, including the development of water quality standards and criteria, effluent characterization, health hazard assessment, wasteload allocation, permit requirements, and compliance monitoring.

In 1985, the U.S. EPA also issued a manual that established standardized methods for measuring the acute toxicity of effluents to freshwater and marine organisms (U.S. EPA 1985b) and the chronic toxicity of effluents to

freshwater organisms (U.S. EPA 1985c). In 1988, U.S. EPA released a document that established standardized methods for estimating the chronic toxicity of effluents to marine and estuarine organisms (U.S. EPA 1988). Chronic toxicity test methods were provided for five species: the sheepshead minnow *(Cyprinodon variegatus)*, the inland silverside *(Menidia beryllina)*, the mysid *(Mysidopsis bahia)*, the sea urchin *(Arbacia punctulata)*, and the red macroalga *(Champia parvula)*. However, because these tests use non-indigenous species to estimate the chronic toxicity of effluents and receiving waters to marine and estuarine organisms, test results may not necessarily reflect actual field conditions within or near the ZID. Moreover, test results may not accurately represent impacts of pollutant discharges on balanced indigenous populations (BIPs).

U.S. EPA developed a permit writer's guide (U.S. EPA 1987) to assist state and Federal NPDES permit writers in establishing water quality-based permit limits for toxic pollutants. To meet these water quality-based limits, the U.S. EPA is continuing to develop criteria that will assist states in establishing their water quality standards and effluent permit limitations. The U.S. EPA criteria under development include recommended magnitudes, durations, and allowable frequencies of exceedance of pollutant concentrations for both acute and chronic biological effects. POTW permit limits on effluent toxicity could be imposed, and the NPDES permittee would be required to conduct a toxicity reduction evaluation (TRE) and implement, if necessary, a toxics control program (TCP) (U.S. EPA 1985a, 1987).

The TRE, a critical component of the TCP, must be conducted to identify effluent toxicity sources, to determine (if possible) specific pollutants responsible for the toxicity, and to identify source control options. The TRE includes a review of the magnitude and extent of the toxicity problem, the discharge characteristics, the receiving water characteristics, the need for additional monitoring to determine water quality/toxicity effects, and the other potential point and nonpoint toxicity sources in the POTW service area.

Because all NPDES-permitted discharges are unique, no single effluent TRE procedure is applicable to every case. A TCP must be developed on an individual case-by-case basis, and must include an evaluation of the impact of 1) the existing POTW wastewater treatment process, 2) point and nonpoint contributors to the POTW influent, 3) types of industries in the POTW service area, 4) the variability, toxicity, and treatability of chemicals in the effluent, and 5) the variability in species sensitivity based on whole-effluent toxicity test results. Either technology-based or water quality-based) source control options also need to be evaluated to determine their effectiveness in reducing effluent toxicity and alleviating the water quality violations.

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