CHAPTER 8

RECEIVING WATER MODELING

This chapter discusses the use of receiving water modeling to evaluate CSO impacts to receiving waters. It uses the term "modeling" broadly to refer to a range of receiving water simulation techniques. This chapter introduces simplified techniques, such as dilution and decay equations, and more complex computer models, such as QUAL2EU and WASP.

8.1 THE CSO CONTROL POLICY AND RECEIVING WATER MODELING

Under the CSO Control Policy a permittee should develop a long-term control plan (LTCP) that provides for attainment of water quality standards (WQS) using either the demonstration approach or presumption approach. Under the demonstration approach, the permittee documents that the selected CSO control measures will provide for the attainment of WQS, including designated uses in the receiving water. Receiving water modeling may be necessary to characterize the impact of CSOs on receiving water quality and to predict the improvements that would result from different CSO control measures. The presumption approach does not explicitly call for analysis of receiving water impacts.

In many cases, CSOs discharge to receiving waters that are water quality-limited and receive pollutant loadings from other sources, including nonpoint sources and other point sources. The CSO Control Policy states that the permittee should characterize the impacts of the CSOs and other pollution sources on the receiving waters and their designated uses (Section II.C.1). Under the demonstration approach, "[w]here WQS and designated uses are not met in part because of natural background conditions or pollution sources other than CSOs, a total maximum daily load, including a wasteload allocation and a load allocation, or other means should be used to apportion pollutant loads." (Section II.C.4.b)

Established under Section 303(d) of the CWA, the total maximum daily load (TMDL) process assesses point and nonpoint pollution sources that together may contribute to a water body's impairment. This process relies on receiving water models.

An important initial decision-which water quality parameters to model-should be based on data from receiving water monitoring. CSOs affect several receiving water quality parameters. Since the impact on one parameter is frequently much greater than on others, relieving this main impact will likely also relieve the others. For example, if a CSO causes exceedances of bacteria WQS by several hundredfold, as well as moderate dissolved oxygen (DO) depressions, solving the bacterial problem will likely solve the DO problem and so it may be sufficient to monitor bacteria only. Reducing the scope of modeling in this fashion may substantially reduce costs.

8.2 MODEL SELECTION STRATEGY

A receiving water model should be selected according to the following factors:

- The type and physical characteristics of the receiving water body. Rivers, estuaries, coastal areas, and lakes typically require different models.

- The water quality parameters to be modeled. These may include bacteria, DO, suspended solids, toxics, and nutrients. These parameters are affected by different processes (e.g., die-off for bacteria, settling for solids, biodegradation for DO, adsorption for metals) with different time scales (e.g., hours for bacterial die-off, days for biodegradation) and different kinetics. The time scale in turn affects the distance over which the receiving water is modeled (e.g., a few hundred feet for bacteria to a few- miles for DO).

- The number and geographical distribution of CSO outfalls and the need to simulate sources other than CSOs.

This section discusses some important considerations for hydrodynamic and water quality modeling of receiving waters, and how these considerations affect the selection and use of a model.

The purpose of receiving water modeling is primarily to predict receiving water quality under different CSO pollutant loadings and flow conditions in the receiving water. The flow conditions, or hydrodynamics, of the receiving water are an important factor in determining the effects of CSOs on receiving water quality. For simple cases, hydrodynamic conditions can be determined from the receiving water monitoring program; elsewhere a hydrodynamic model may be necessary.

Hydrodynamic and water quality models are either ***steady-state*** or ***transient.*** Steady-state models assume that conditions do not change over time, while transient models can simulate conditions that vary over time. Flexibility exists in the choice of model types; generally, either a steady-state or transient water quality simulation can be done regardless of whether flow conditions are steady-state or transient.

8.2.1 Hydrodynamic Models

A hydrodynamic model provides the flow conditions, characterized by the water depth and velocity, for which receiving water quality must be predicted. The following factors should be considered for different water body types:

- ***Rivers-*** Rivers generally flow in one direction (except for localized eddies or other flow features) and the stream velocity and depth are a function of the flow rate. The flow rate in relatively large rivers may not increase significantly due to wet weather discharges, and a constant flow can be used as a first approximation. This constant flow can be a specified low flow, the flow observed during model calibration surveys, or a flow typical of a season or month. When the increase of river flow is important, it can be estimated by adding together all upstream flow inputs or by doing a transient flow simulation. The degree of refinement required also depends on the time scale of the water quality parameters of interest. For example, assuming a constant river flow may suffice for bioaccumulative toxicants (e.g., pesticides) because long-term exposure is ofimportance. For DO, however, the time variations in river flow rate may be need to be considered.

- ***Estuaries-*** CSO impacts in estuaries are affected by tidal variations of velocity and depth (including reversal of current direction) and by possible salinity stratification. Tidal fluctuations can be assessed by measuring velocity and depth variations over a tide cycle or by using a one- or two-dimensional model. Toxics with relatively small mixing zones can be analyzed using steady currents corresponding to different times during the tidal cycle, but this may require using a computed circulation pattern from a model.

- ***Coastal Areas-*** CSO impacts in coastal areas are also affected by tidal fluctuations. The discussion on estuaries generally applies to coastal areas, but, because the areas are not channelized, two-dimensional or even three-dimensional models may be necessary.

- ***Lakes-*** CSO impacts in lakes are affected by wind and thermal stratification. Wind-driven currents can be monitored directly or simulated using a hydrodynamic model (which may need to cover the entire lake to simulate wind-driven currents properly). Thermal stratification can generally be measured directly.

Because the same basic hydrodynamic equations apply,[1] some of the major models for receiving waters can be used to simulate more than one type of receiving water body. Ultimately, three factors dictate whether a model can be used for a particular hydraulic regime. One factor is whether it provides a one-, two-, or three-dimensional simulation. A second is its ability to handle specific boundary conditions, such as tidal boundaries.

A third factor is whether the model assumes steady-state conditions or allows for time-varying pollutant loading. In general, models that assume steady-state conditions cannot accurately model CSO problems that require analysis of far-field effects. However, in some instances a steady-load model can estimate the maximum potential effect, particularly in systems where the transport of constituents is dominated by the main flow of the water body, rather than local velocity gradients. For example, by assuming a constant source and following the peak discharge plug of water downstream, the steady-load model QUAL2EU can determine the maximum downstream effects of conventional pollutants. The result is a compromise that approximates the expected impact but neglects the moderating effects of longitudinal dispersion. However, QUAL2EU cannot give an accurate estimate of the duration of excursions above WQS.

8.2.2 Receiving Water Quality Models

The frequency and duration of CSOs are important determinants of receiving water impacts and need to be considered in determining appropriate time scales for modeling. CSO loads are

[1] The basic hydrodynamic equations are for momentum and continuity. The momentum equation describes the motion of the receiving water, while the continuity equation is a flow balance relationship (i.e., total inflows to the receiving water less total outflows is equal to the change in receiving water volume).

typically delivered in pulses during storm events. Selection of appropriate time scales for modeling receiving water impacts resulting from a pulsed CSO loading depends upon the time and space scales necessary to evaluate the WQS. If analysis requires determining the concentration of a toxic at the edge of a relatively small mixing zone, a steady-state mixing zone model may be satisfactory. When using a steady-state mixing zone model in this way, the modeler should apply appropriately conservative but characteristic assumptions about instream flows during CSO events. For pollutants such as oxygen demand, which can have impacts lasting several days and extending several miles downstream of the discharge point, it may be warranted to incorporate the pulsed nature of the loading. Assuming a constant loading is much simpler (and less costly) to model; however, it is conservative (i.e., leads to impacts larger than expected). For pollutants such as nutrients where the response time of the receiving water body may be slow, simulating only the average loading rate, usually over a period of days (e.g., 21 days) depending on the nutrient, may suffice.

Receiving water models vary from simple estimations to complex software packages. The choice of model should reflect site conditions. If the pulsed load and receiving water characteristics are adequately represented, simple estimations may be appropriate for the analysis of CSO impacts. To demonstrate compliance with the CWA, the permittee may not need to know precisely where in the receiving water excursions above WQS will occur. Rather, the permittee needs to know the maximum pollutant concentrations and the likelihood that excursions above the WQS can occur at any point within the water body. However, since CSOs to sensitive areas are given a higher priority under the CSO Policy, simulation models for receiving waters with sensitive areas may need to use short time scales (e.g., hourly pollutant loads), and have high resolution (e.g., several hundred yards or less) to specifically assess impacts to sensitive areas.

8.3 AVAILABLE MODELS

Receiving water models cover a wide variety of physical and chemical situations and, like combined sewer system (CSS) models, vary in complexity. EPA has produced guidance on receiving water modeling as part of the Waste Load Allocation (WLA) guidance series. These models, however, tend to concentrate on continuous sources and thus may not be the most suitable for CSOs. Ambrose et al. (1988a) summarizes EPA-supported models, including receiving water models.

This guidance does not provide a complete catalogue of available receiving water models. Rather, it describes simplified techniques and provides a brief overview of relevant receiving water models supported by EPA or other government agencies. In many cases, detailed receiving water simulation may not be necessary. Use of dilution and mixing zone calculations or simulation with simple spreadsheet models may be sufficient to assess the magnitude of potential impacts or evaluate the relative merits of various control options.

Types of Simulation

Water quality parameters can be simulated using either single-event, steady-state modeling or continuous, dynamic modeling. Many systems may find it beneficial to use both types of modeling.

Many of the simpler approaches to receiving water evaluation assume steady flow and steady or gradually varying loading. These assumptions may be appropriate if an order-of-magnitude estimate or an upper bound of the impacts is required. The latter is obtained by using conservative parameters such as peak loading and low current speed. If WQS attainment is predicted under realistic worst-case assumptions, more complex simulations may not be needed.

Due to the random nature of CSOs, the use of dynamic simulation may be preferable to single-event, worst-case, steady-state modeling. Dynamic techniques allow the modeler to derive the fraction of time during which a concentration was exceeded and water quality was impaired. For instance, when using daily simulated results, specific concentrations are first ranked with the corresponding number of occurrences during the simulation period. Frequency distribution plots are then developed and used to determine how often the l-day-acute water quality criteria are likely to be exceeded. The same approach can be used to develop frequency distributions for longer periods such as 4-day or 30-day average concentrations. EPA (1991a) recommends three dynamic modeling techniques: continuous simulation, Monte Carlo simulation, and lognormal probability modeling.

Continuous simulation models solve time-dependent differential equations to simulate flow volume and water quality in receiving waters. These deterministic models incorporate the manner in which flow and toxic pollutant concentrations change over time in a continuous manner rather than relying on simplified terms for rates of change. They use daily effluent flow and concentration data with daily receiving water flow and concentration data to estimate downstream receiving water concentrations. If properly calibrated and verified, a continuous simulation model can predict variable flow and water quality accurately-although at a considerable time and resource expenditure, however.

Monte Carlo simulation is generally used for complex systems that have random components. Input variables are sampled at random from pre-determined probability distributions and used in a toxic fate and transport model. The distribution of output variables from repeated simulations is analyzed statistically to derive a frequency distribution. However, unlike continuous simulation models, the temporal frequency distribution of the output depends on the temporal frequency distribution of the input data. For instance, if the water quality criterion is based on a 4-day average, the input variables must use the probability distributions based on a 4-day average.

Lognormal probability modeling estimates the same output variable probability distributions as continuous and Monte Carlo simulations but with less effort. However, like Monte Carlo simulation, the input must be probability distributions based on input data for the specific temporal frequency distribution desired. The theoretical basis of the technique permits the stochastic nature of the CSO process to be explicitly considered. This method assumes that each of the four variables that affect downstream receiving water quality (rainfall, runoff, event mean concentration of contaminant in the runoff (EMC), and streamflow) can be adequately represented by a lognormal probability distribution. When the EMC is coupled with a lognormal distribution of runoff volume, the distribution of runoff loads can be derived. The storm water load frequency is then coupled with a lognormal distribution of streamflow to derive the probability distribution of in-stream concentrations. The main advantage of lognormal probability modeling is that the probability distributions can be derived using only the median and the coefficient of variation for each input variable.

8.3.1 Model Types

The following sections discuss techniques for simulating different water quality parameters in rivers, lakes and estuaries.

RIVERS

Bacteria and Toxics. Bacteria and toxic contaminants are primarily a concern in the immediate vicinity of CSO outfalls. They are controlled by lateral mixing, advection, and decay processes such as die-off (for bacteria), vaporization (for toxics), and settling and resuspension (for bacteria and toxics). When stream flow is small relative to CSO flow, lateral mixing may occur rapidly and a one-dimensional model may be appropriate. Initial estimates can be made using a steady-state approach that neglects the time-varying nature of the CSO. In this case, concentrations downstream of a CSO are given by:

$$C_x = \frac{Q_u C_u + Q_e C_e}{Q_s} e^{\frac{-KX}{u}}$$

where:[2]

- C_x = max pollutant concentration at distance X from the outfall (M/L^3)
- C_e = pollutant concentration in effluent (M/L^3)
- C_u = pollutant concentration upstream from discharge (M/L^3)
- Q_e = effluent flow (L^3/T)
- Q_u = stream flow upstream of discharge (L^3/T)
- Q_s = stream flow downstream of discharge, $Q_u + Q_e$ (L^3/T)
- X = distance from outfall (L)
- u = stream flow velocity (L/T)
- K = net decay rate (die-off rate for bacteria, settling velocity divided by stream depth for settling, resuspension velocity divided by stream depth for resuspension, vaporization rate divided by stream depth for vaporization) (1/T)
- e = 2.71828...

Since bacteria and toxics can settle out of the water column and attach to sediments, sediments can contain significant amounts of these pollutants. Resuspension of sediments and subsequent desorption of bacteria and toxics into the water column can be an important source of receiving water contaminants. Modeling of sediment resuspension requires estimation of

[2] M=unit of mass, L=unit of length, and T=unit of time.

resuspension velocities and knowledge of sediment transport processes. Thomann and Mueller (1987) discusses how to determine the solids balance in a river and estimate sediment resuspension velocities. Modeling of sediment transport is complex and is often done using computer models such as WASP5 and HSPF.

In large rivers, lateral mixing may occur over large distances and bacterial counts or toxics concentrations on the same shore as the discharge can be calculated using the following expression, as a conservative estimate (U.S. EPA, 1991a):

$$C_x \approx \frac{C_e Q_e W}{Q_s \sqrt{\frac{\pi D_y X}{u}}}$$

where: D_y = lateral dispersion coefficient (L^2/T)
W = stream width (L)
π = 3.14159...

This equation is conservative because it neglects any discharge-induced mixing. Simulating over the correlated probability distributions of C_e, Q_e, Q_s, and Q_u can provide an estimate of the frequency of WQS exceedances at a specific distance from the outfall. The method requires the estimation of a lateral dispersion coefficient, which can be measured in dye studies or by methods described in Mixing ***in Inland and Coastal Waters*** (Fischer et al., 1979). Fischer's methods calculate the lateral dispersion coefficient D_y as follows:

$$D_y = 0.6\ du^* \pm 50\%$$

where: ***d*** = water depth at the specified flow (L)
u^* = shear velocity (L/T).

In turn, the following equation estimates shear velocity:

$$u^* = (gds)^{1/2}$$

where: g = acceleration due to gravity (L/T^2)
s = slope of channel (L/L)
d = water depth (L).

The model DYNTOX (LimnoTech, 1985) is specially designed for analysis of toxics in rivers and can handle all three dynamic modeling techniques. U.S. EPA (1991a) and the WLA series by Delos et al. (1984) address the transport of toxics and heavy metals in rivers.

Oxygen Demand/Dissolved Oxygen. The time scales and distances affecting DO processes are greater than for bacteria and toxics. Lateral mixing therefore results in approximately uniform conditions over the river cross section and one-dimensional models are usually appropriate for simulation. The WLA guidance (U.S. EPA, 1995g) discusses the effects of steady and dynamic DO loads, and provides guidelines for modeling impacts of steady-state sources. Simple spreadsheet models such as STREAMDO IV (Zander and Love, 1990) have recently become available for DO analysis.

In general, screening analyses using classical steady-state equations can examine DO impacts to rivers as a result of episodic loads. This approach assumes plug flow, which in turn allows an assumption of constant loading averaged over the volume of the plug (Freedman and Marr, 1990). This approach does not consider longitudinal diffusion from the plug, making it a conservative approach. The plug flow analysis should correlate with the duration of the CSO. For example, a plug flow simulation of a 2-hour CSO event would result in a downstream DO sag that would also last for 2 hours. Given the plug flow assumption, the classic Streeter-Phelps equation can estimate the DO concentration downstream:

$$D = D_o e^{-K_a t} + \frac{W}{Q}\left(\frac{K_d}{K_a - K_r}\right)\left[e^{-K_r t} - e^{-K_a t}\right]$$

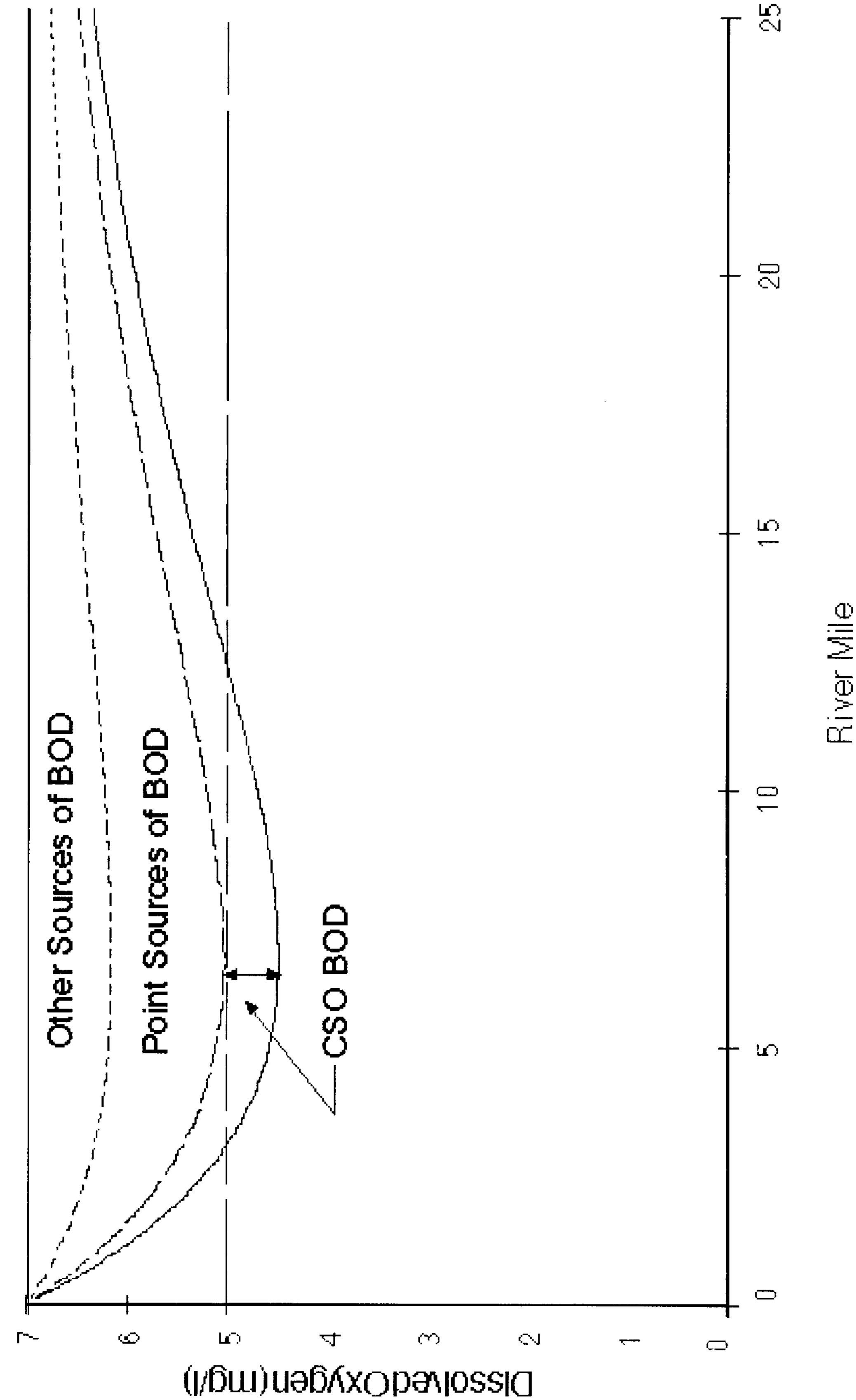
where:
D = DO deficit downstream (M/V)
D_0 = initial DO deficit (M/V)
K_a = atmospheric re-aeration rate (1/T)
t = time of passage from source to downstream location (T)
W = total pollutant loading rate (M/T)
Q = total river flow (V/T)
K_d = biochemical oxygen demand (BOD) deoxygenation rate (1/T)
K_r = BOD loss rate (1/T).

This method can address the joint effects of multiple steady sources through the technique of superposition (Exhibit 8-1). Superposition is used when linear differential equations, such as the Streeter-Phelps equation, govern pollutant concentrations along a receiving stream. For such linear systems, the concentration of a pollutant in a river due to multiple steady-state sources is the linear summation of the responses due to the individual sources. Superposition techniques are also used to estimate pollutant concentrations due to multiple steady-state sources of toxic pollutants. However, it cannot address multiple sources that change over time, nor can it address the effects of river morphology. When such issues are important, more sophisticated modeling techniques are necessary.

More sophisticated modeling techniques are also necessary to assess the effects of sediment oxygen demand (SOD) and plant respiration (which remove oxygen from the receiving water), and photosynthesis by aquatic plants (which adds oxygen to the water). The Streeter-Phelps equation makes the simplifying assumption that there are only point sources of CBOD, so SOD, photosynthesis, and respiration are assumed to be zero. If photosynthesis, respiration, and SOD are significant, more complex analysis is needed to evaluate these factors. These distributed sources and sinks of DO and BOD are addressed by Thomann and Mueller (1987) and by several computer models, including QUAL2EU and WASPS.

Nutrients/Eutrophication. Nutrient discharges affect river eutrophication over time scales of several days to several weeks. Nutrient/eutrophication analysis considers the relationship between

Exhibit 8-1. Dissolved Oxygen Superposition Analysis



nutrients and algal growth. Analysis of nutrient impacts in rivers is complex because nutrients and planktonic algae,[3] which are free-floating one-celled algae, usually move through the system rapidly.

The current WLA guidance (U.S. EPA, 1995g) considers only planktonic algae (rather than all aquatic plants) and discusses nutrient loadings and eutrophication in rivers primarily as a component in computing DO. The guidance applies to narrative criteria that limit nuisance plant growth in large, slowly flowing rivers.

LAKES

Bacteria and Toxics. Mixing zone analysis can often be used to assess attainment of WQS for bacteria and toxics in lakes. For a small lake in which the effluent mixes rapidly, the concentration response is given by the following equation (Freedman and Marr, 1990):

$$C = \frac{M}{V} e^{(-K - \frac{Q}{V})t}$$

where: C = concentration (M/L^3)
M = mass loading (M)
Q = flow (L^3/T)
K = net decay rate (bacteria die-off, settling and resuspension, volatilization, photolysis, and other chemical reactions) (1/T)
V = lake volume (L^3)
t = time (T).

For an incompletely-mixed lake, however, a complex simulation model is generally necessary to estimate transient impacts from slug loads. The EPA WLA guidance series contains a manual on chemical models for lakes and impoundments (Hydroqual, Inc., 1986). This guidance, which also applies to bacteria, describes simple and complex models and presents criteria for selecting models and model parameters.

[3] Aquatic plants can be divided into those that move freely with the water (planktonic aquatic plants) and those that are attached or rooted in place.

Oxygen Demand/Dissolved Oxygen. Simple analytical approximations can model oxygen demand and DO in cases where DO mixing occurs quickly relative to depletion by COD/BOD. Where lateral mixing occurs rapidly but vertical temperature stratification exists, DO concentration can be addressed for a two-layer stratified lake under the following simplifying assumptions (from Thomann and Mueller, 1987):

- The horizontal area is constant with depth
- Inflow occurs only to the surface layer
- Photosynthesis occurs only in the surface layer
- Respiration occurs at the same rate throughout the lake
- The lake is at steady-state.

With these severe restrictions, the solution is given by:

$$c_1 = \left(\frac{q}{K_L + q}\right)c_o + \left(\frac{K_L}{K_L + q}\right)c_s + \frac{pH_1 - RH - S_B}{K_L + q} - \frac{K_{d1}H_1L_1 - K_{d2}H_2L_2}{K_L + q}$$

and

$$c_2 = c_1 - \left(\frac{S_B + RH_2 - K_{d2}H_2L_2}{E/H_i}\right)$$

where the subscripts 1 and 2 refer to the epilimnion (top layer) and hypolimnion (lower layer), respectively, and variables without subscripts refer to the whole lake, and where:

q	=	Outflow rate (L/T)
K_L	=	DO transfer rate at lake surface (L/T)
c	=	DO concentration (M/L^3)
c_0, c_s	=	Initial and saturation dissolved oxygen concentrations (M/L^3)
p	=	Gross photosynthetic production of DO (m/L^3-T)
H	=	Depth (L)
H_i	=	$H/2$ when $H_1 = H_2$ and H_1 when $H_2 >> H_1$ (L)
R	=	Phytoplankton DO respiration (M/L^3-T)

S_B = Sediment oxygen demand (M/L^2-T)
K_d = Deoxygenation coefficient (1/T)
L = Steady-state CBOD concentration in water column (M/L^3), = $W/(Q+K_rV)$, where W is the mass loading rate, Q is the rate of flow through the lake, V is the volume, and K_r is the net loss rate.
E = Dispersion coefficient (L^2/T).

Because this analysis assumes steady-state loading and because measuring some of the parameters proves difficult, the method may only have limited application to CSOs. A modeler able to define all of the above parameters may choose to apply a more spatially resolved model.

In many cases, complex simulation models are necessary to analyze DO in lakes. These are either specialized lake models or flexible models, such as EUTROWASP, that are designed to address issues specific to lakes. Some experienced modelers have been successful in modeling thermally stratified lakes with one or two dimensional river models (e.g., QUAL2EU) that assume the river bottom is the thermocline.[4]

Nutrient/Eutrophication Impacts. For lakes, simple analytic equations often can analyze end-of-pipe impacts and whole-lake impacts, but evaluating mixing phenomena frequently requires a complex computer model (Freedman and Marr, 1990). Simple analytical methods can be applied to lake nutrient/eutrophication impacts in situations where the CSOs mix across the lake area within the time scale required to obtain a significant response in the algal population. In most lakes, phosphorus is considered to be the limiting nutrient for nuisance algal impacts and eutrophication. Mancini et al. (1983) and Thomann and Mueller (1987) have developed a procedure for calculating the allowable surface loading rate. The following steps are drawn from this procedure:

Step 1. Estimate the lake volume, surface area, and mean depth.

Step 2. Estimate the mean annual inflow and outflow rates. Where urban areas draining to the lake constitute a significant fraction of the total drainage area, flow

[4] Such techniques should not be used by inexperienced modelers as they can lead to inaccuracies if they are not used with caution.

estimates from urban runoff and CSOs should be included in the hydrologic balance around the lake. For lakes with large surface areas, the estimate should include surface precipitation and evaporation.

Step 3. Determine the average annual total phosphorus loading due to all sources, including all tributary inflows, municipal and industrial sources, distributed urban and rural runoff, and atmospheric inputs. ***Technical Guidance Manual for Performing Waste Load Allocation*** (Mancini et al., 1983) discusses techniques for estimating these loadings.

Step 4. For total phosphorus, assign a net sedimentation loss rate that is consistent with a local data base.

Step 5. Select trophic state objectives of either total phosphorus or chlorophyll-a consistent with local experience. Calculate the value of the allowable phosphorus area1 loading, W^1, from:

$$W' = a\bar{z}\left(\frac{Q}{V} + v_s\right)$$

where: W' is the allowable area1 surface loading rate (M/L^2-T)
a is the trophic state objective concentration of total phosphorus or chlorophyll-a (M/L^3),
Q is outflow (L^3/T),
V is lake volume (L^3),
$\bar{z}$ is mean depth (L), and
v_s is the net sedimentation velocity (L/T).

Step 6. Compare the total area1 loading determined in Step 3 to the value of W^1 obtained in Step 5.

Additional approaches are discussed in Reckhow and Chapra (1983b).

ESTUARIES

Unlike most rivers, estuaries are tidal (i.e., water moves upstream during portions of the tidal cycle and downstream during other parts of the cycle). When averaged on the basis of tidal cycles, pollutant transport in narrow, vertically mixed estuaries with dominant longitudinal flow is similar to that in rivers. However, due to tidal reversals of flow, a narrow estuary may have a much larger effective dispersion coefficient since shifting tides may cause greater lateral dispersion. In such a system, the modeler can apply approximate or screening models used for rivers, provided that an appropriate tidal dispersion coefficient has been calculated. In wider estuaries, tides and winds often result in complex flow patterns and river-based models would be inappropriate. WLA guidance for estuaries is provided in several EPA manuals (Ambrose et al., 1990; Martin et al., 1990; Jirka, 1992; Freedman et al., 1992).

In addition to their tidal component, many estuaries are characterized by salinity-based stratification. Stratified estuaries have the horizontal mixing due to advection and dispersion that is associated with rivers and the vertical stratification characteristic of lakes.

In complex estuaries, accurate analysis of far-field CSO impacts-such as nutrients/ eutrophication, DO, and impacts on particular sensitive areas-typically requires complex simulation models. Simpler analyses are sometimes possible by treating the averaged effects of tidal and wind-induced circulation and mixing as temporally constant parameters. This approach may require extensive site-specific calibration.

Near-field mixing zone analysis in estuaries also presents special problems, because of the role of buoyancy differences in mixing. Jirka (1992) discusses mixing-zone modeling for estuaries.

8.3.2 Computer Models Supported by EPA or Other Government Agencies

This section describes some computer models relevant to receiving water modeling. Most of these models are supported by EPA's Center for Exposure Assessment Modeling (CEAM). CEAM maintains a distribution center for water quality models and related data bases.[5] CEAM-supported models relevant to modeling impacts on receiving water include QUAL2EU, WASPS, HSPF, EXAMSII, CORMIX, MINTEQ, and SMPTOX3. The applicability and key characteristics of the CEAM-supported models are summarized in Exhibit 8-2.

[5] See Section 7.3 for information on obtaining models from CEAM.

Exhibit 8-2. EPA CEAM-Supported Receiving Water Models

Applicability to Hydraulic Regimes and Pollutant Type

	Rivers & Streams			Lakes & Impoundments			Estuaries			Near Field Mixing
Model	**Nutrients**	**Oxygen**	**Other**	**Nutrients**	**Oxygen**	**Other**	**Nutrients**	**Oxygen**	**Other**	
QUAL2EU	✓	✓	✓							
WASP5	✓	✓	✓	✓	✓	✓	✓	✓	✓	
HSPF	✓	✓	✓	✓	✓	✓				
EXAMSII			✓			✓			✓	
CORMIX	Near-field mixing model for all water body types									✓
MINTEQ	Equilibrium metal speciation model									
SMPTOX3			✓							

Key Characteristics and References

Model	Pollutant Loading Type	Transport Dimensionality	Current Version	Key References
QUAL2EU	Steady	1-D	3.22	Brown & Barnwell, 1987
WASP5	Dynamic	Quasi-2/3-D (link-node)	5.10	Ambrose, et al., 1988
HSPF	Dynamic (integrated)	1-D	10.11	Johanson, et al., 1984
EXAMSII	Dynamic	User input (quasi 3-D)	2.96	Burns, et al., 1982
CORMIX	Steady (near field) [1]	Quasi-3-D (zonal)	2.10	Doneker & Jirka, 1990
MINTEQ	Steady	None	3.11	Brown & Allison, 1987
SMPTOX3	Steady	1-D	2.01	LimnoTech, 1992

[1] CORMIX was originally developed assuming steady ambient conditions; Version 3 allows for application to some unsteady environments (e.g., tidal reversal conditions) where transient recirculation and pollutant build-up can occur (CEAM, 1998).

QUAL2EU is a one-dimensional model for rivers. It assumes steady-state flow and loading but allows simulation of diurnal variations in temperature or algal photosynthesis and respiration. QUAL2EU simulates temperature, bacteria, BOD, DO, ammonia, nitrate, nitrite, organic nitrogen, phosphate, organic phosphorus, algae, and additional conservative substances.[6] Because it assumes steady flow and pollutant loading, its applicability to CSOs is limited. QUAL2EU can, however, use steady loading rates to generate worst-case projections for CSOs to rivers. The model has pre- and post-processors for performing uncertainty and sensitivity analyses.

Additionally, in certain cases, experienced users may be able to use the model to simulate non-steady pollutant loadings under steady flow conditions by establishing certain initial conditions or by dynamically varying climatic conditions. If used in this way, QUAL2EU should be considered a screening tool since the model was not designed to simulate dynamic quality conditions.

WASP5 is a quasi-two-dimensional or quasi-three-dimensional water quality model for rivers, estuaries, and many lakes. It has a link-node formulation, which simulates storage at the nodes and transport along the links. The links represent a one-dimensional solution of the advection dispersion equation, although quasi-two-dimensional or quasi-three-dimensional simulations are possible if nodes are connected to multiple links. The model also simulates limited sediment processes. It includes the time-varying processes of advection, dispersion, point and nonpoint mass loading, and boundary exchanges. WASP5 can be used in two modes: EUTRO5 for nutrient and eutrophication analysis and TOXI5 for analysis of toxic pollutants and metals.

WASP5 is essentially a pollutant fate and transport model. Transport can be driven by another hydrodynamic model such as DYNHYD5. DYNHYD5 is a one-dimensional/quasi-two-dimensional model that simulates transient hydrodynamics (including tidal estuaries).

[6] A conservative substance is one that does not undergo any chemical or biological transformation or degradation in a given ecosystem. (U.S. EPA, 1995g)

HSPF is a one-dimensional, comprehensive hydrologic and water quality simulation package which can simulate both receiving waters and runoff to CSSs for conventional and toxic organic pollutants. HSPF simulates the transport and fate of pollutants in rivers and reservoirs. It simulates three sediment types: sand, silt, and clay.

EXAMSII can rapidly evaluate the fate, transport, and exposure concentrations of steady discharges of synthetic organic chemicals to aquatic systems. A recent upgrade of the model considers seasonal variations in transport and time-varying chemical loadings, making it quasi-dynamic. The user must specify transport fields to the model.

CORMIX[7] is an expert system for mixing zone analysis. It can simulate submerged or surface, buoyant or non-buoyant discharges into stratified or unstratified receiving waters, with emphasis on the geometry and dilution characteristics of the initial mixing zone. The model uses a zone approach, in which a flow classification scheme determines which near-field mixing processes to calculate. The CORMIX model cannot be calibrated in the classic sense since rates are fixed based on the built-in logic of the expert system.

MINTEQ determines geochemical equilibrium for priority pollutant metals. Not a transport model, MINTEQ provides a means for modeling metal partitioning in discharges. It provides only steady-state predictions. The model usually must be run in connection with another fate and transport model, such as those described above. A number of assumptions (e.g., equilibrium conditions at the point of mixing between a CSO and the receiving water) must be made to link MINTEQ predictions to another fate and transport model, so it should be used cautiously in evaluating wet weather impacts.

SMPTOX3 is a one-dimensional steady-state model for simulating the transport of contaminants in the water column and bed sediments in streams and non-tidal rivers. SMPTOX3 is an interactive computer program that uses an EPA technique for calculating concentrations of

[7] In some applications CORMIX has proven inaccurate for single port discharges.

toxic substances in the water column and stream bed as a result of point source discharges to streams and rivers. The model predicts pollutant concentrations in dissolved and particulate phases for the water column and bed sediments, as well as total suspended solids. SMPTOX3 can be run at three different levels of complexity: as described above (highest complexity), to calculate toxic water column concentrations but no interactions with bed sediments (medium complexity), or as a total pollutant toxics model (low complexity) (LimnoTech, 1992).

The following additional models are supported by EPA or other government agencies:[8]

DYNTOX is a one-dimensional, probabilistic toxicity dilution model for transport in rivers. It provides continuous, Monte Carlo, or lognormal probability simulations that can be used to analyze the frequency and duration of ambient toxic concentrations resulting from a waste discharge. The model considers dilution and net first-order loss, but not sorption and benthic exchange. DYNTOX Version 2.1 and the draft manual are available from the Office of Science and Technology in EPA's Office of Water (202-260-7012).

CE-QUAL-W2 is a reservoir and narrow estuary hydrodynamics and water quality model developed by the Waterways Experiment Station of the U.S. Army Corps of Engineers. The model provides dynamic two-dimensional (longitudinal and vertical) simulations. It accounts for density effects on flow as a function of the water temperature, salinity and suspended solids concentration. CE-QUAL-W2 can simulate up to 21 water quality parameters in addition to temperature, including one passive tracer (e.g., dye), total dissolved solids, coliform bacteria, inorganic suspended solids, algal/nutrient/DO dynamics (11 parameters), alkalinity, pH and carbonate species (4 parameters).

[8] McKeon and Segna (1987), Ambrose et al. (1988a) and Hinson and Basta (1982) have reviewed some of these models.

8.4 USING A RECEIVING WATER MODEL

As was the case for CSS models (see Section 7.4), receiving water modeling involves developing the model, calibrating and validating the model, performing the simulation, and interpreting the results.

8.4.1 Developing the Model

The input data needs for a specific receiving water model depend upon the hydraulic regime and model used. The permittee should refer to the model's documentation, the relevant sections of the WLA guidance, or to texts such as *Principles of Surface Water Quality Modeling and Control* (Thomann and Mueller, 1987). Tables B-2 through B-5 in Appendix B contain general tables of data inputs.

8.4.2 Calibrating and Validating the Model

Like CSS models, receiving water models need to be calibrated and validated. The model should be run to simulate events for which receiving water hydraulic and quality monitoring were actually conducted, and the model results should be compared to the measurements. Generally, receiving water models are calibrated and validated first for receiving water hydraulics and then for water quality. Achieving a high degree of accuracy in calibration can be difficult because:

- Pollutant loading inputs typically are estimates rather than precisely known values.
- Three-dimensional receiving water models are still not commonly used for CSO projects, so receiving water models involve spatial averaging (over the depth, width or cross-section). Thus, model results are not directly comparable with measurements, unless the measurements also have sufficient spacial resolution to allow comparable averaging.
- Loadings from non-CSO sources, such as storm water, upstream boundaries, point sources, and atmospheric deposition, often are not accurately known.
- Receiving water hydrodynamics are affected by numerous factors which are difficult to account for. Those include fluctuating winds, large-scale eddies, and density effects.

Although these factors make model calibration challenging, they also underscore the need for calibration to ensure that the model reasonably reflects receiving water data.

8.4.3 Performing the Modeling Analysis

Receiving water modeling can involve single events or long-term simulations. Single event simulations are usually favored when using complex models, which require more input data and take significantly longer to run (although advances in computer technology keep pushing the limits of what can practically be achieved.) Long-term simulations can predict water quality impacts on an annual basis.

Although a general goal is to predict the number of water quality criteria exceedances, models can evaluate exceedances using different measures, such as hours of exceedance at beaches or other critical points, acre-hours of exceedance, and mile-hours of exceedance along a shore. These provide a more refined measure of the water quality impacts of CSOs and of the expected effectiveness of different control measures.

CSO loadings commonly are simulated separately from other loadings in order to assess the relative impacts of CSOs. This is appropriate because the equations that best approximate receiving water quality are usually linear and so effects are additive (one exception, however, is the non-linear algal growth response to nutrient loadings).

8.4.4 Using Modeling Results

By calculating averages over space and time, simulation models predict CSO volumes, pollutant concentrations, and other variables of interest. The extent of this averaging depends on the model structure, how the model is applied, and the resolution of the input data. The model’s space and time resolution should match that of the necessary analysis. For instance, the applicable WQS may be expressed as a 1-hour average concentration not to exceed a given concentration more than once every three years on average. Spatial averaging may be represented by a concentration averaged over a receiving water mixing zone, or implicitly by the specification of monitoring locations to establish compliance with instream criteria. In any case, the permittee should note whether the model predictions use the same averaging scales required in the permit or relevant WQS.

When used for continuous rather than event simulation, as suggested by the CSO Control Policy, simulation models can predict the frequency of exceedances of water quality criteria. Probabilistic models, such as the Monte Carlo simulation, also can make such predictions. In probabilistic models, the simulation is made over the probability distribution of precipitation and other forcing functions such as temperature, point sources, and flow. In either case, modelers can analyze the output for the frequency of water quality criteria exceedances.

The key result of receiving water modeling is the prediction of future conditions due to implementation of CSO control alternatives. In most cases, CSO control decisions will have to be supported by model predictions of the pollutant load reductions necessary to achieve WQS. In the receiving waters, critical or design water quality conditions might be periods of low flows and high temperature that are established based on a review of available data. Flow, temperature, and other variables for these periods then form the basis for analysis of future conditions.

It is useful to assess the sensitivity of model results to variations in parameters, rate constants, and coefficients. A sensitivity analysis can determine which parameters, rate constants, and coefficients merit particular attention in evaluating CSO control alternatives. The modeling approach should accurately represent features that are fully understood, and sensitivity analysis should be used to evaluate the significance of factors that are not as clearly defined. (See Section 7.4.4 for additional discussion of sensitivity analysis.)