

Working Paper

Modeling the Impacts of Diffuse Pollution on Receiving Water Quality

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WP-95-2
January 1995



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ABSTRACT

Nonpoint or diffuse pollutants represent a major cause of water-quality degradation of rivers, estuaries, lakes, and reservoirs and have become increasingly significant in countries where point sources of pollution are largely controlled. Nonpoint sources cause eutrophication, oxygen depletion, sedimentation, acidification, and salinization in receiving water bodies, introduce pathogenic organisms and other pollutants, and through shock loads of pollutants, cause mortality and morbidity of aquatic organisms. The major sources of nonpoint pollution include agriculture, silviculture, construction, and urban runoff.

The potential effects of nonpoint source pollutants on receiving water may be evaluated using water-quality models. A recommended procedure is to: 1) establish study objective and constraints, 2) determine pollutant interactions, 3) perform a screening analysis, and 4) select a water-quality model and complete the analysis. In the screening analysis, simple analytical tools are used to evaluate the potential severity of impacts and the time scales at which impacts occur. We describe a screening-level model that evaluates the response to a conceptual spike load of unit strength. Water-quality models may be either relatively simple analytical models or more complex numerical models. For many situations, an analytical model will provide sufficient analysis of a problem and no further evaluation is required. For more complex problems, numerical water-quality models can provide a detailed and rigorous analysis.

Analytical models are available to describe a variety of pollutant and receiving water situations. We describe models for the following: streams and rivers - simple conservative and nonconservative pollutant inflows, probabilistically described conservative pollutant inflows, biochemical oxygen demand and dissolved oxygen, suspended sediment, and adsorptive micropollutants; lakes and reservoirs - nutrient loading (eutrophication), and micropollutant loading; and estuaries - nonconservative pollutant inflows. A table of available numerical models and their applicability and capabilities is also provided.

A case study of Lake Balaton, Hungary – which formed the subject of a major policy oriented research of IIASA and the Hungarian Academy of Sciences at the late seventies, early eighties – illustrates the procedure for assessing nonpoint source pollutant impacts. Lake Balaton is a large but shallow lake which is experiencing water-quality degradation due to nutrient inflows and consequent eutrophication. Approximately 70% of the nutrient load comes from nonpoint sources. The largest single load is the inflow from the Zala River which varies over time with precipitation. Analysis of historical data shows that a monthly time scale should be used to capture the effects of this variability on the response in the most western, hypertrophic basin of Lake Balaton. Analysis with the unit-load screening model illustrates that short time scales must be considered to evaluate the response to nonpoint source loads of the Zala River itself while much longer time scales, on the order of a year, suffice for evaluating the response of the entire lake.

MODELING THE IMPACTS OF DIFFUSE POLLUTION ON RECEIVING WATER QUALITY

Peter Shanahan¹ and László Somlyódy²

1. INTRODUCTION

In this paper, we discuss the importance of nonpoint sources and their effects on water bodies and discuss the differences between point and nonpoint sources. Particularly, nonpoint sources may differ significantly from point sources in their distribution in time and space—a factor we illustrate through simple analytical models.

We also stress the importance of water-quality models as a tool for understanding and assessing nonpoint source impacts on water quality. Our recommended overall approach to water-quality modeling is to start with simple analytical models for screening and progress to more complex models only if necessary (after defining the outlines of the problem with screening models). We describe and provide equations for a variety of analytical models, and discuss and list a variety of numerical models. We also provide numerous additional references on models as well as sources of modeling computer codes. Selected analytical models are illustrated in several worked examples. The use of screening models in problem definition is illustrated on the example of Lake Balaton, Hungary, which was a major study of IIASA and various Hungarian institutions about fifteen years ago with the aim to use systems analysis for the development of eutrophication control strategies (Somlyódy and van Straten, 1986).

Ultimately, we hope to provide our readers with a sense for importance of nonpoint source impacts on receiving water quality as well as some tools and an approach to assessing those impacts.

2. WATER QUALITY IMPACTS

2.1 *RECOGNITION OF THE NONPOINT SOURCE POLLUTION PROBLEM*

Researchers began to recognize the significance of nonpoint source pollution in the late 1960s and early 1970s as water-quality models and mass balance calculations revealed significant sources of pollutants other than known point sources. Nonpoint sources were manifested in the field when improvements in wastewater treatment at point sources failed to produce fully the anticipated improvements in stream and river water quality. With continuing water-quality improvements and control of point sources, nonpoint sources became increasingly recognized in Europe and North America as an important aspect of environmental water quality. The control of nonpoint sources is usually more complex and difficult than for point sources in that nonpoint sources often involve complex transport and transformation through several media (e.g., atmospheric deposition, soil application, and chemical transformation in soils involving the air, soil, and water media). Moreover, one cannot in a practical sense directly regulate nonpoint source emissions, but only activities in the watershed that may cause emissions. Nonpoint source pollution therefore continues as an area of active research.

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There has been growing awareness of the importance and severity of nonpoint source pollutants in European and American legislation. New environmental directives for water quality by the European Community impose strict limitations for a variety of nonpoint source pollutants. In the U.S., nonpoint sources were recognized in the 1987 reauthorization of the federal Clean Water Act. Section 319 of the Act requires the U.S. Environmental Protection Agency and the individual states to assemble information that characterizes nonpoint source pollutant impacts on receiving water quality and to report to the U.S. Congress on the nature of NPS pollutant impacts and present a plan to address nonpoint sources.

The importance of diffuse sources on water quality has been confirmed by studies in Europe and North America. A recent report on the Danube River basin found that nonpoint sources contributed 60% of the nitrogen and 44% of the phosphorus load to the entire river basin (CEC, 1994). According to Cunningham (1988), nonpoint sources were the principal contributor of pollutants to 76% of the U.S. lakes and reservoirs that failed to meet stream-water quality standards. Nonpoint sources similarly impaired 65% of the U.S. streams failing to meet standards and 45% of the estuaries. These statistics show that diffuse sources figure significantly in those water bodies where pollution problems persist.

2.2 IMPACTS AND TYPES OF DIFFUSE SOURCES

The impacts of diffuse sources on receiving water quality are many and varied, being a function of the types of pollutants, their sources, and the receiving-water environment. The following partial list of impacts draws upon a similar list assembled by Varis and Somlyódy (1993):

- **Eutrophication** caused by the abundance of nutrients and other prerequisites of primary production;
- **Oxygen depletion** caused by the degradation of organic matter in water;
- **Hygienic problems** caused by pathogenic organisms including viruses, bacteria (*Salmonella*, *Yersinia*, etc.), and protozoa (*Schistosomiasis*, *Giardia*, *Cryptosporidium*, etc.);
- **Sedimentation** caused by sediment runoff and accumulation;
- **Acidification** of lakes caused by atmospheric deposition;
- **Salinization** caused by high salt concentrations in runoff and irrigation return flow; and
- **Mortality and morbidity** of aquatic organisms due to cumulative and/or shock loads of toxic micropollutants including heavy metals, pesticides, herbicides, and industrial chemicals.

The previously mentioned report to the Congress required under the 1987 Clean Water Act reauthorization was completed in 1992 (U.S. EPA, 1992a) and provides a comprehensive examination of nonpoint source pollution in the United States and characterization of the nature of NPS pollutants and sources. The EPA NPS study organized the data which it collected according to various data categories including the type of water body, type of nonpoint source, and type of pollution. The EPA definitions for these three categories are used here with minor modification and are summarized in Tables 1 through 3. Table 3 shows the water body categories defined by EPA along with the NPS sources and pollutants found in the EPA survey to affect those water bodies most often.

The EPA survey supports several overall conclusions. Agriculture is the single greatest cause of nonpoint source pollution in the U.S. and is the leading cause of NPS impacts in rivers, lakes, and wetlands. Two pollutants closely associated with agricultural sources, nutrients and sediments, are the most common NPS pollutants found in the survey. In ground water, which is used almost exclusively for water supply, toxic chemicals are the most common pollutants with pesticides from agriculture the leading single pollutant type. NPS pollution impacts are generally similar in Europe. Nitrate is a particularly significant ground-water contaminant in Europe as well as in parts of the U.S.

Despite the overall significance of agriculture, several specific nonpoint source types and pollutants merit special attention. Urban nonpoint sources are important because they particularly affect water bodies that are sensitive—estuaries and coastal waters. And by their very nature, urban sources affect water bodies that are used by and benefit the largest numbers of people. A particular class of pollutants, toxic chemicals, also have added significance because they are persistent and have the potential to affect the health of aquatic life and humans. Pathogens are similarly important with respect to their ability to affect human health.

Airborne pollutants constitute a special class of diffuse pollutants because of their unique mode of reaching receiving waters. While airborne pollutants may come from many sources originally, they reach receiving waters via direct deposition to the water, or deposition and subsequent washoff from the watershed. Nutrients, metals, and oxygen-demanding substances are deposited at significant rates in urban and suburban areas (U.S. EPA, 1992b). Another important class of airborne diffuse pollutants are those responsible for acid precipitation and deposition. The ecology of water bodies in watersheds with low alkalinity in Scandinavia, Canada, and United States have been severely affected by lowered pH due to acid precipitation.

Table 1. Nonpoint pollution source types (adapted from U.S. EPA, 1992a).

| Source Category | Examples |
|---------------------------------|--|
| Agriculture | Crop production, grazing, animal raising, feedlots |
| Silviculture | Timber production, reforestation, road construction, forest management |
| Construction | Highway, bridge and road construction, land development |
| Urban Runoff | Storm sewers, combined sewer overflows, surface runoff, marinas |
| Mining | Surface and subsurface mining, placer and dredge mining, mine and mill tailings |
| Land Disposal | Landfill, wastewater irrigation, sludge disposal, septic systems, hazardous waste disposal |
| Hydrologic/Habitat Modification | Stream channelization, dredging, dam construction, flow regulation, shoreline modification |
| Other | Atmospheric deposition, spills, in-place contaminants, storage tank leaks |

Table 2. Pollutants associated with nonpoint source pollution (adapted from U.S. EPA, 1992a).

| |
|---|
| Pesticides |
| Toxic organic chemical |
| Nontoxic organic chemicals |
| Metals |
| Ammonia |
| Chloride |
| Other inorganic chemicals |
| Nutrients (phosphorus and nitrogen) |
| pH |
| Sediment and siltation |
| Organic enrichment and dissolved oxygen |
| Salinity |
| Thermal modification |
| Flow alteration |
| Other habitat alterations |
| Pathogens |
| Radiation |
| Oil and Grease |

Table 3. Water body types and NPS impacts (adapted from U.S. EPA, 1992a).

| Water body Type | Leading U.S. Sources | Leading U.S. Pollutants |
|----------------------|---|---|
| Rivers and streams | Agriculture Unknown | Siltation Nutrients |
| Lakes and reservoirs | Agriculture Unknown | Nutrients Siltation |
| Estuaries | In-place contaminants Urban | Nutrients Pathogens |
| Coastal waters | Waste storage tanks Petroleum activities | Oil and grease Metals, pesticides, pathogens, inorganics |
| Wetlands | Agriculture Hydrologic modification | Siltation Metals |
| Ground water | Not specified | Pesticides Toxic compounds |

3. ASSESSMENT OF WATER-QUALITY IMPACTS FROM DIFFUSE SOURCES

Systematic evaluation of nonpoint source impacts requires using a water-quality model of some sort. Whether simple or complex, a model can offer several benefits. First, models provide a predictive ability which enables potentially expensive water-quality management options to be evaluated and tested prior to their implementation. This is obviously far more cost-effective than testing expensive infrastructure and management actions on a trial-and-error basis. Water-quality models also provide a systematic and rigorous framework for examination of water-quality impacts. Because of their requirements for precise and accurate data, models often serve to direct field water-quality data collection and to identify significant gaps in characterization or understanding of a water body.

3.1 APPROACHES TO ASSESS NONPOINT SOURCE IMPACTS

In this section we discuss procedures to assess the impact of diffuse pollutants on receiving water quality. We focus on the differences between nonpoint source and point source effects, the latter serving as a comparison standard because of the several decades of experience in analyzing point source problems. Thus, as a point of departure we consider the major steps and features in assessing point source discharges (see Table 4).

For traditional waste load allocation problems, point source discharges are considered steady-state. Their locations are well-defined, corresponding to municipalities or industrial plants, and the actual discharge loads for different pollutants (for example, BOD, nitrogen, and phosphorus) can be straightforwardly estimated from measurements, population equivalent values, production technologies, existing or planned treatment technologies, and so forth.

Assuming that loads from the different point sources are steady over time, the water quality at a specific receiving-water cross section depends primarily on the flow and consequent dilution. (For conservative pollutants, concentration depends strictly on dilution; for a real constituent, reactions and their dependence on various processes and parameters, such as temperature, may cause distortions). The well-known, theoretical relationship is that concentration is a hyperbolic function of the flow, as shown by the line for point source dominated conditions in Figure 1. Note however that this relation holds only for steady conditions and the figure does not apply when the streamflow rate is changing. Due to the hyperbolic dependence of concentration on flow, a statistical low flow is selected as a "safe" design condition. Typically this design flow is Q355 in Europe and 7Q10 in the United States. Q355 is the flow which is exceeded on average 355 days of the year and 7Q10 is the average low flow observed during 7 consecutive days once in 10 years.

A consequence of the above procedure is that the actual hydrology and characteristics of the watershed (slopes, vegetation, land use pattern, etc.) are unimportant (see Table 4). Nothing more than basic hydraulic and morphometric characteristics are required to compute the steady or gradually varied flow needed as input to model water-quality impacts. Even this step is often further simplified by assuming a trapezoidal cross section and estimating flow from the Manning equation.

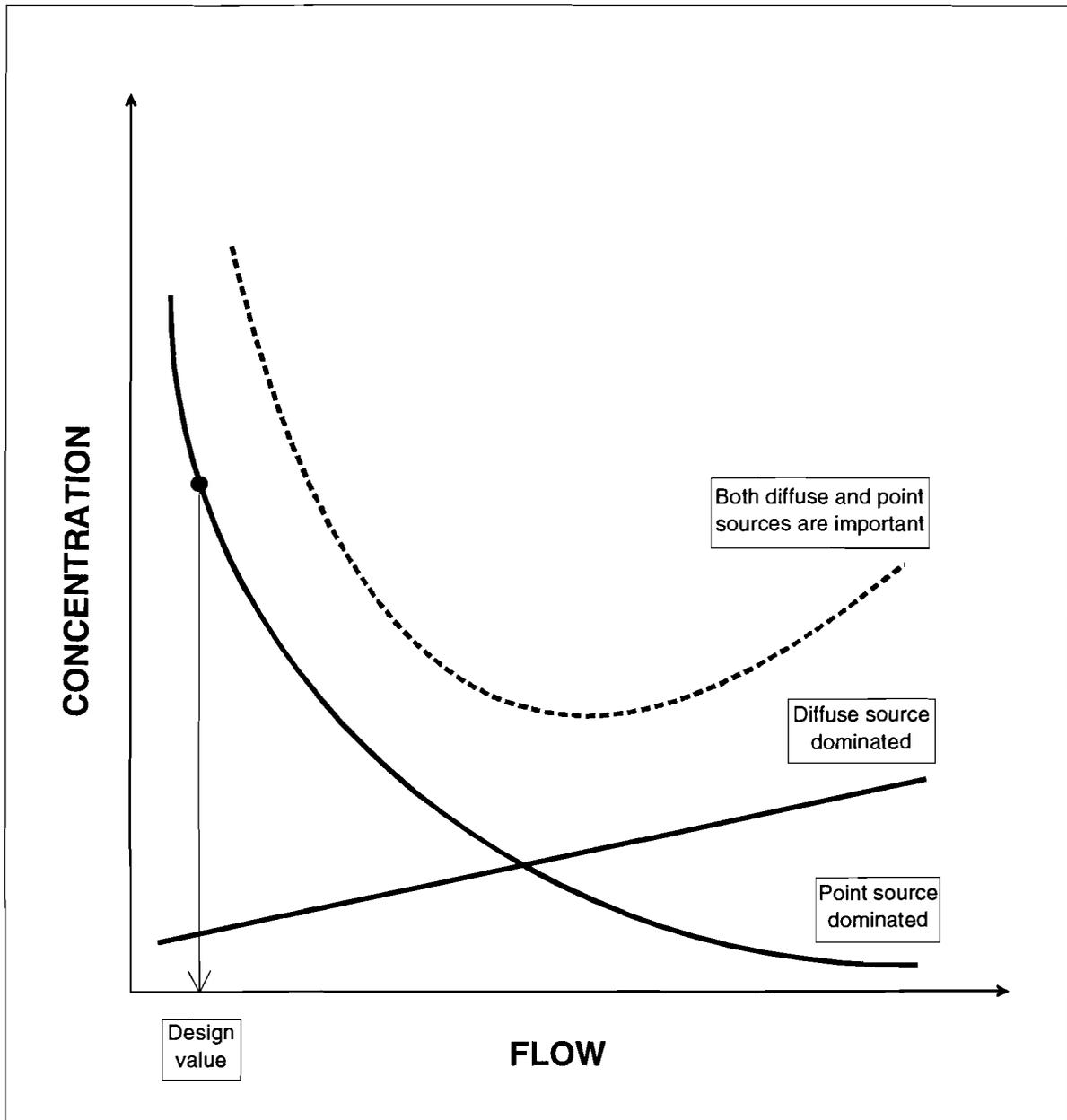


Figure 1

Schematic concentration versus flow relationships for different types of watersheds

Table 4. Comparison of major steps in assessing point source and nonpoint source problems.

| | Point Sources | Nonpoint sources |
|--------------------------------------|---|--|
| Pollutant loads | Steady Well-defined locations Easily estimated | Unsteady Driven by precipitation Areal sources Difficult to estimate |
| Design conditions | Low flow Largely independent of hydrology Independent of watershed | Dynamic and stochastic Highly dependent on hydrology Highly dependent on watershed |
| System representation | Simple (One-dimensional representation of river with point sources) | Complex (Two-dimensional representation of both watershed and water body) |
| Impact assessment (modeling) | Requires only emission estimates and subsequent use of steady, deterministic water-quality models | Joint completion of areal load computations, flow routing, and impact assessment |
| Approximate approaches | Not typically required or necessary | Estimation of line sources and time scales Use of steady-state or shock-load models |
| Calibration and parameter estimation | Literature data Relatively simple measurements (time series and longitudinal profiles) | Even comprehensive field measurements may be inadequate to calibrate complex models with many parameters |

The schematization of the system is also extremely simple: a river basin is represented as a tree structure composed of the river main stem and its tributaries. Wastewater discharges are imposed on this tree as point information. Since both the loads and the longitudinally varied flow are assumed to be constant and steady, deterministic water-quality models can be employed to estimate the impact of point-source discharges. This is almost always an acceptable estimate and a more detailed approach is needed only for unusual problems, when more detailed spatial (and perhaps temporal) changes in the flow and water quality need to be computed.

Calibration and validation of models are never easy exercises. Still, for traditional point source problems there are many sources of information on how to estimate parameter values and on what measurements to make to determine water-body-specific parameters. Thus, a traditional water-quality model of a stream system dominated by point sources is relatively straightforward.

In contrast to the above procedure, the assessment of nonpoint source impacts incorporates more steps of greater sophistication as outlined below (see also Table 4).

First of all, most diffuse pollution is directly or indirectly driven by precipitation and thus loads are inherently dynamic in nature. Diffuse pollutants originate from areal sources which are translated in the receiving-water river or lake into line sources. Thus, nonpoint sources are dynamic line sources in contrast to the steady point loads discussed above.

The actual load to a water body is a result of a number of complex processes. At the most basic level, we must recognize that the carrier of the pollutants is water (and possibly sediment) as it moves through the hydrologic cycle. Thus, except in unusual circumstances, detailed hydrologic calculations are needed to describe diffuse pollutant sources. In addition, other watershed processes such as

erosion, plant uptake, and biological transformation may play important roles. Overall, it is clear that water and material cycling must be considered jointly if we wish to obtain a detailed load estimate.

From the point of view of planning, the identification of a single design scenario is difficult and often impossible for nonpoint source problems. In contrast to point source problems, pollutant concentrations in the receiving waters of a watershed dominated by diffuse loads typically increase as streamflow increases (as illustrated by Line 2 in Figure 1). For catchments where both diffuse and point sources are important (Line 3 in Figure 1), critical conditions may occur at either high or low flow, and there is no single obvious design flow. While it may be possible to identify a single "optimal" flow at which the receiving water concentration is the least, this flow is of limited utility for water-quality planning.

In practice, most nonpoint source problems involve multiple pollutants. For example, BOD, nitrogen, phosphorus, and heavy metals may be simultaneously discharged to a river or a lake by a municipal wastewater treatment plant and they may interact with each other in the water body. This is not a significant problem for water-quality assessment, since dilution affects all of the constituents similarly and thus a single design condition can be used. The situation is much more complex for nonpoint source pollution (even if we neglect its unsteady nature), since the flow dependency can vary from one pollutant to the other (see Table 4 for the basic types). For instance, BOD, phosphorus, and nitrogen jointly influence dissolved oxygen conditions in a river, but they may respectively follow lines 1, 2, and 3 in Figure 1. Thus, there may be no unique design condition for all pollutants in a nonpoint source problem.

In a strict sense, impact assessment for nonpoint source pollutants should require not only a dynamic analysis, but also a stochastic one accounting for all hydrologic, meteorologic, and other variables. Likewise, management alternatives should be evaluated for their ability to affect not only the mean but also the variance of water-quality indicators. Such an approach would fully integrate receiving waters with their associated watersheds, and would route flow and water quality dynamically and stochastically. This technology remains a distant reality, despite tremendous developments in monitoring, remote sensing, computer technology, and modeling. The approach would require combining a detailed watershed hydrologic-hydraulic model (such as the Danish Hydraulic Institute SHE model or the U.S. EPA HSPF model) with a watershed nonpoint source model like CREAMS and a receiving water-quality model all in a stochastic Monte Carlo framework. Obviously, approximations are necessary to achieve an alternative realistic modeling framework.

The starting point of an approximate assessment methodology is to recognize that the basic structures of water-quality models of rivers, lakes, and estuaries do not depend on whether pollution originates from point or nonpoint sources. Thus, nonpoint source assessments can be done using the same methods and models as used for point sources provided that the required input can be obtained at time and space scales consistent with the receiving-water body, the water-quality problem, and the nature of the impacts.

As noted above, diffuse pollution entails consideration of distributed or line sources. Generally, their derivation (for instance, converting from unit areal loads in the watershed to unit length loads to a river) does not cause serious difficulty (see for example Ryding and Rast, 1989). In contrast, the time scale is much more problematic. In many cases, shock-load or steady-state models provide an acceptable approximation: shock-load models for problems in which short-term impacts are important (see Section 2) and steady-state models for longer-term effects. In addition to such cases, for which analytical solutions can be used, there are a number of others for which loads can be established on the desired time scale (a month or a year) and state-of-the-art dynamic models can be employed (for instance to study lake eutrophication and its control). But for many problems, load estimates are desired, but unavailable, on a daily or shorter scale. There is no simple recipe for these problems except to base an evaluation on a number of "realistic" estimates in a sensitivity/uncertainty analysis with the primary goal to learn how the system may behave. Whatever the particular problem

involved, it is clear that the time scale and its determination is an important consideration. It is further discussed in Section 3.3.

Because analytical and simplified models often produce an acceptable approximation for nonpoint source problems, and can be used as a screening tool in any case, we turn most of our attention in the following sections to relatively simple models. These models could be used in the preliminary stage of a nonpoint source evaluation to acquire an order of magnitude impact estimation. Depending on the outcome of this preliminary evaluation and the importance of the problem, the analyst should decide whether more comprehensive modeling is required or not.

3.2 MASS-BALANCE PRINCIPLES

In the sections to follow in this paper, we emphasize simple water-quality models for use in screening-level calculations of potential receiving water impacts. The basic principle behind all of the equations presented is the mass balance. Basically, the mass balance is an accounting of all mass entering, leaving, and remaining in the system of interest. It can be summarized in the following equation which would be used to determine the change in mass over a selected period of time and a selected volume of water body:

$$\begin{aligned} \text{Change in mass} = & \text{Inflow due to advection} - \text{outflow due to advection} + \\ & \text{inflow due to dispersion} - \text{outflow due to dispersion} + \\ & \text{external load from point sources} + \\ & \text{external load from nonpoint sources} + \\ & \text{internal load} \pm \\ & \text{reactions} \end{aligned}$$

In the screening calculations that follow, a mass balance would be typically done for a well defined, natural volume of water such as a whole lake or well defined segment.

Completing the mass balance equation illustrates the relative contributions of point and nonpoint sources as well as processes occurring within the water body. However, the simple statement of the equation above gives little indication of the sometimes great disparity in the temporal character of the different components. As already noted, point sources are typically steady and relatively constant; nonpoint sources occur as sporadic intense events. These different time scales imply different modeling and monitoring requirements. The following section explores the implications of time-scale effects further.

3.3 TIME-SCALE EFFECTS IN MODELING NPS POLLUTION

A simple analytical model of receiving water response to a short duration pollution event is constructed in this section in order to provide an understanding of the nature of time-scale considerations. Figure 2 shows the assumed pollutant input: pollutant load increases linearly from an initial baseline value of L_0 to a maximum value of L_{max} in time $T/2$, and then decreases linearly over the same time period back to the original value. This spike-like inflow is conceptually similar to the pollution inflow associated with a rainstorm event. The pollutant of concern is assumed to experience loss from first-order decay and removal by flow out of the system. The water body is modeled as a single fully mixed volume. The governing equation is thus:

$$\frac{dC}{dt} = \frac{L}{V} - \frac{Q}{V}C - kC \quad (1)$$

where, C is the pollutant concentration in the water body [M/L^3];
 t is time [T];
 $L(t)$ is the load of pollutant as illustrated in Figure 2 [M/T];
 V is the volume of the water body [L^3];
 Q is the rate of flow through the water body [L^3/T];
 k is a first-order loss coefficient.

This equation has as its solution the following:

$$\begin{aligned} \frac{C}{C_\ell} = & 1 + \left(\frac{\ell_{\max}}{\ell_o} - 1 \right) \frac{t}{T/2} - \left(\frac{\ell_{\max}}{\ell_o} - 1 \right) \frac{2}{k' T} \\ & + \left[\frac{C_o}{C_\ell} - 1 + \left(\frac{\ell_{\max}}{\ell_o} - 1 \right) \frac{2}{k' T} \right] \exp(-k't) \quad \text{for } t \leq T/2 \end{aligned} \quad (2a)$$

$$\begin{aligned} \frac{C}{C_\ell} = & \frac{\ell_{\max}}{\ell_o} - \left(\frac{\ell_{\max}}{\ell_o} - 1 \right) \frac{t^*}{T/2} + \left(\frac{\ell_{\max}}{\ell_o} - 1 \right) \frac{2}{k' T} \\ & + \left[\frac{C_{T/2}}{C_\ell} - \frac{\ell_{\max}}{\ell_o} - \left(\frac{\ell_{\max}}{\ell_o} - 1 \right) \frac{2}{k' T} \right] \exp(-k't^*) \quad \text{for } T/2 < t \leq T \end{aligned} \quad (2b)$$

$$\frac{C}{C_\ell} = \left(C_T - \frac{\ell_{\max}}{\ell_o} \right) \exp(-k't^{**}) \quad \text{for } t > T \quad (2c)$$

where, $\ell_o = L_o/V$ is the base load normalized by the water body volume [$M/(TL^3)$];
 $\ell_{\max} = L_{\max}/V$ is the normalized maximum load ;
 $k' = k + 1/\tau$ is the apparent decay coefficient [$1/T$];
 $\tau = V/Q$ is the hydraulic residence time [T];
 $t^* = t - T/2$ is a transformed time variable [T];
 $t^{**} = t - T$ is also a transformed time variable [T];
 $C_\ell = \ell_o/k'$ is a reference concentration [M/L^3]; and
 $C_o, C_{T/2}, C_T$ are the water body concentrations at times 0, $T/2$, and T [M/L^3].

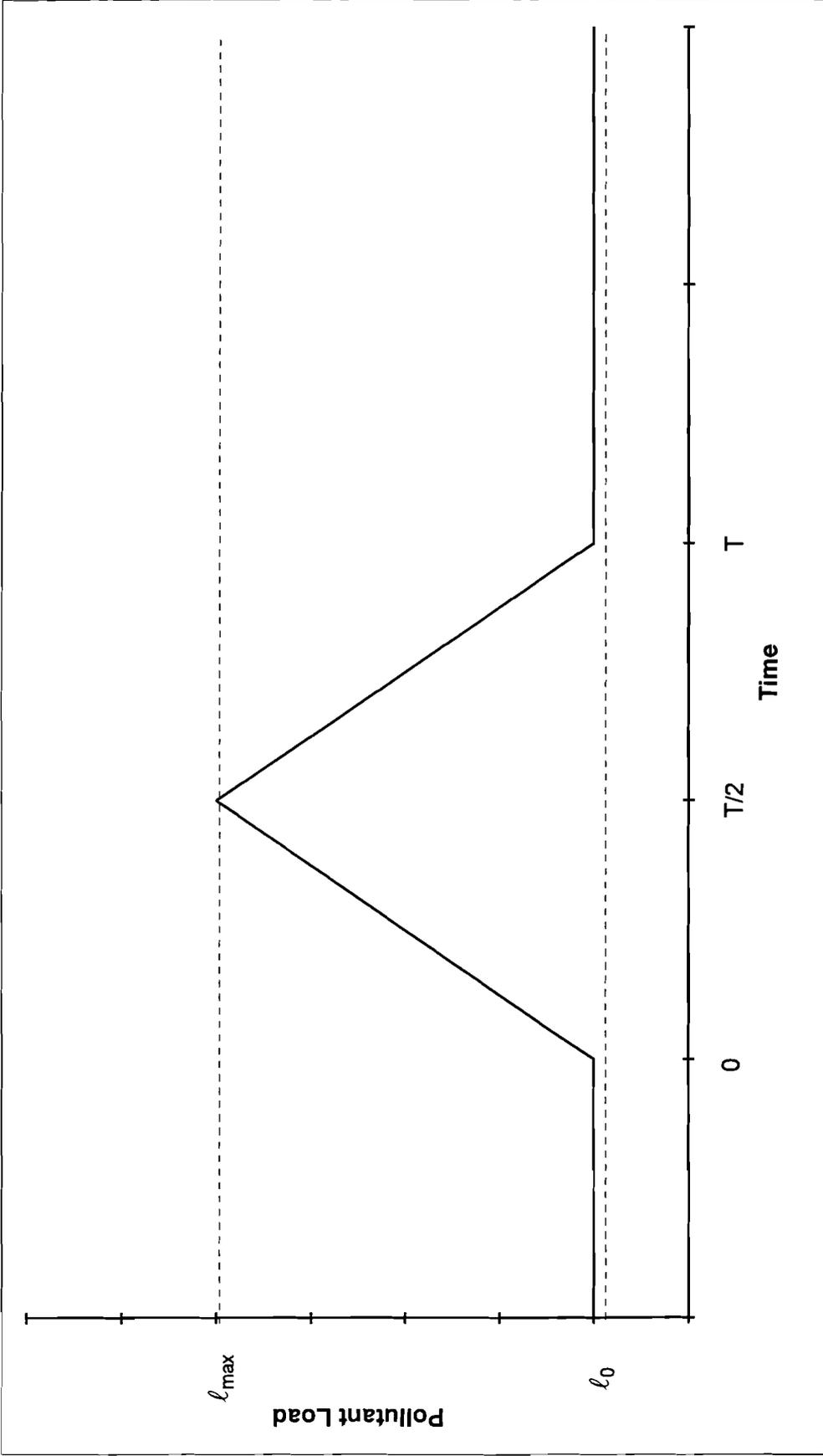


Figure 2
Assumed pollutant load history for screening model

It is instructive to consider the character of this solution as a function of the relationship between the loading time, T , and the residence time, τ . Figure 3 shows the solution in terms of normalized concentration, C/C_ℓ versus normalized time, t/τ , as a function of T/τ , the ratio of the event duration to the residence time. For large T/τ and no decay, the normalized receiving water concentration C/C_ℓ becomes large and in the limit approaches ℓ_{\max}/ℓ_0 (see Figure 4). This is because the transient effect of the event duration becomes dominant over the time scale of the water body and the water-body concentration follows closely the influent concentration. This might be the case of an NPS inflow to a river, in which the residence time for a river reach may be quite short, an hour or less, while a typical NPS event might last a day or more. In this case the river concentration is altered almost immediately and then follows closely the transient character of the NPS load. On the other hand, if T/τ is very small, the transient effect of the NPS event is damped out in the receiving water, and the load makes only a slight but long-lasting (relative to the event duration) change in the receiving water concentration. This would be the case of stormwater inflows to a lake: each individual storm event makes only a slight change in the receiving water, although the cumulative effect of many storms may be significant. Thus, it is a very acceptable approximation for a lake to ignore individual storm events and consider, for example, the total monthly NPS load to the lake.

The analysis scheme and solutions provided above can be used in screening calculations to evaluate the important time scales in an NPS water-quality problem. This analysis can provide useful insight into the character of the problem and the type of analysis that would be effective. For example, if one's problem is at either extreme of T/τ , then an analytical solution may be an effective model of the system. But, if T/τ is an intermediate value, then a detailed analysis using numerical models may be required.

3.4 WATER-QUALITY IMPACT ASSESSMENT USING MODELS

The selection and development of an appropriate and useful water-quality model requires a prior understanding of the purpose or purposes for which it will be used. Modeling can serve many purposes: it can contribute to basic understanding of the water body and pollutant sources; it can guide the planning and design of monitoring programs; it can be used to design and compare pollution control alternatives; and it can be used as tool for basin-scale water-quality management. The nature of nonpoint sources will in many cases force a wider modeling purpose than would point sources. The diverse character of nonpoint sources and their frequent connection to land use implies that more people may be affected by water-quality management decisions, thus creating a greater burden on the water-quality manager and modeler. Frequently, modeling of nonpoint sources will require an examination of management alternatives and their economic implications. Tradeoffs between control of different nonpoint sources, and thus different sectors of the population and economy, may need to be considered. And finally, the stochastic character of nonpoint sources may necessitate special modeling approaches not typically needed for point sources. Thus, the overall character and complexity of nonpoint sources may force a broader purpose to modeling than does the typical point source problem. Particularly, providing input to a complicated water-quality management process may be a key purpose of a nonpoint source modeling effort.

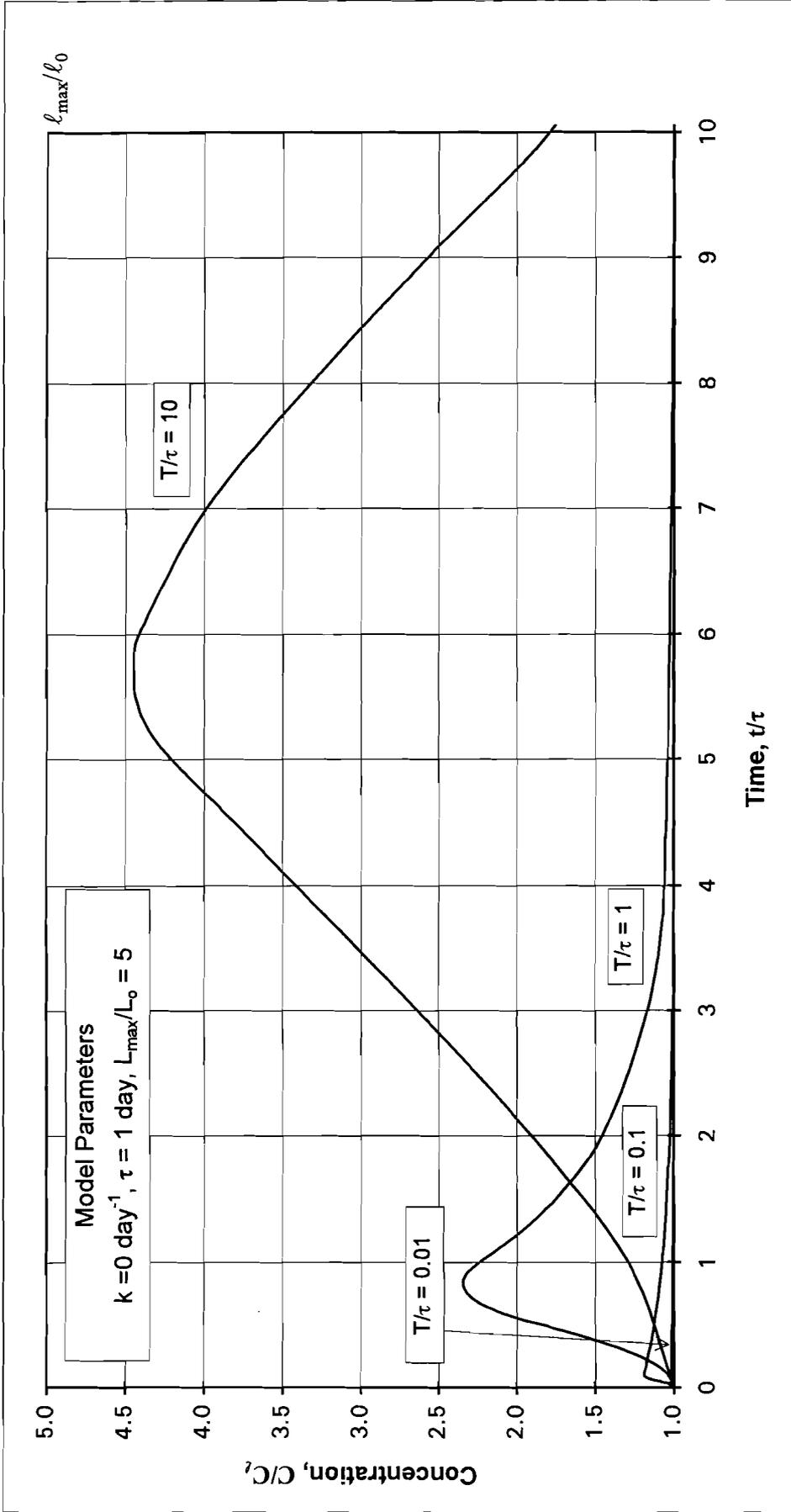


Figure 3
 Receiving water concentration histories predicted by screening model

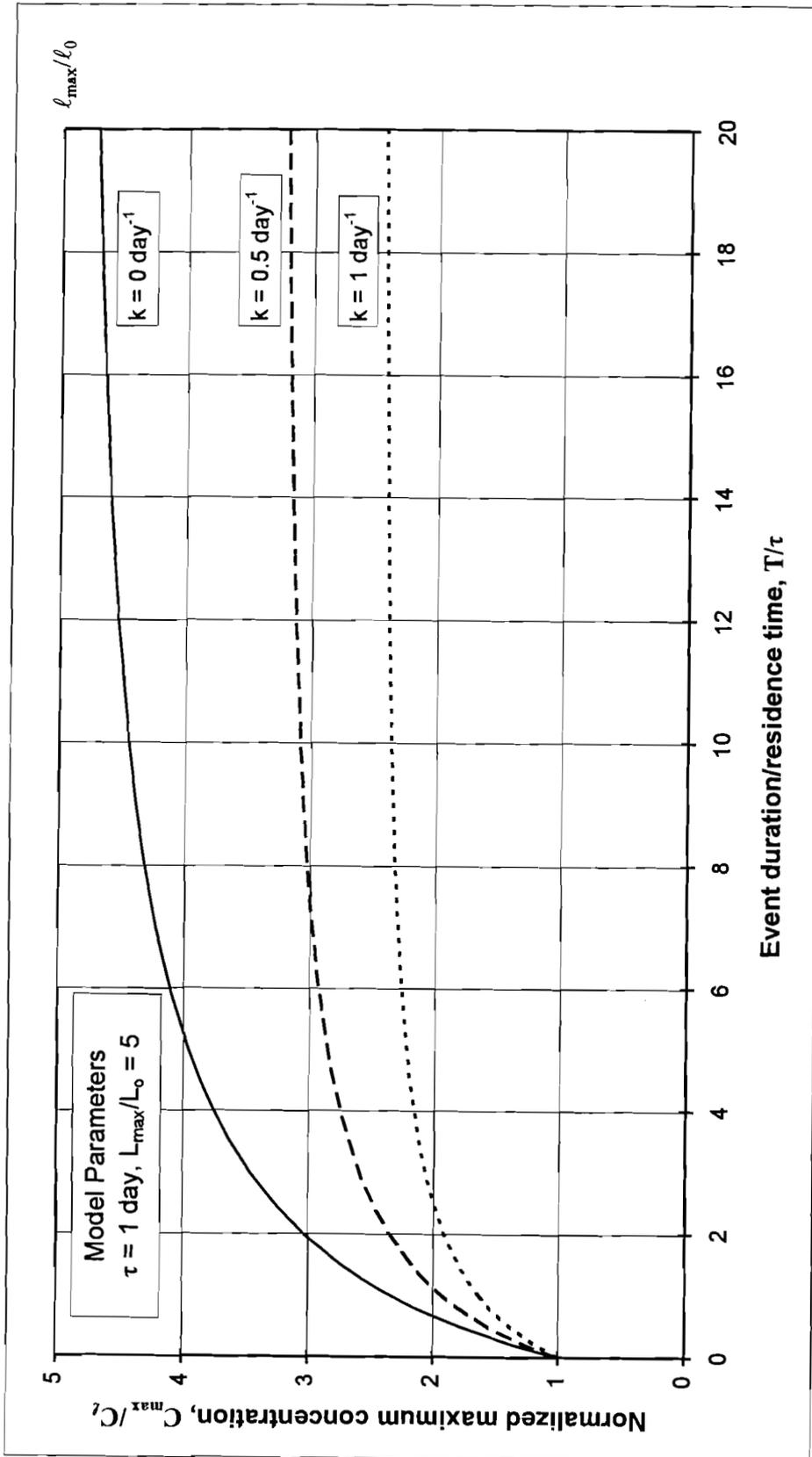


Figure 4
 Maximum receiving water concentration predicted by screening model

Section 2 makes clear that many different types of water-quality problems are encompassed by nonpoint sources. Indeed, the lists of pollutant types and sources in Tables 1, 2, and 3 are so expansive as to embrace all water-quality problems other than those specifically associated with a point-source discharge (that is, a discharge from a single pipe outfall). This diversity of pollutants, sources, and receiving-water environments requires the user to carefully select the model or models that best fit the requirements of a specific impact assessment. The following provides some general guidelines for selecting a model. The presumption is that a model will be selected from available solutions or computer codes. If a model is to be developed from first principles, we recommend the procedure outlined by van Straten and Somlyódy (1986) as a guide to model identification and development.

The following outlines the major steps in selecting a model for a diffuse pollution problem:

1. Establish study objectives and constraints. The objectives of a particular water-quality assessment typically derive from the need to meet environmental regulations or some other water-quality goal. The most successful water-quality modeling studies are those in which the objectives are clearly and specifically defined prior to the start of any actual modeling. The particular pollutants to be controlled and modeled should be determined as well as the conditions (time of year, locations in water body, hydrological conditions) under which predictions will be required. However, constraints on the study must also be factored into the study design and model selection. Constraints may be posed by limitations in the available field data as well as by time and resource limitations. Clear identification of both objectives and constraints prior to the modeling effort helps to ensure that a realistic program can be planned and completed.

2. Determine pollutant interactions. The specific pollutants and conditions identified in Step 1 should be analyzed to determine the water-quality processes that control the fate and transport of the pollutants in the specific water environment being assessed. Both hydrodynamic and physical-chemical processes must be considered. The primary hydrodynamic parameter is the hydraulic residence time of the water body. In some problems, such as open coastal zones, mixing and dispersion may also be significant. Physical-chemical processes include settling or adsorption of pollutants to the sediments, biochemical reaction and conversion to different materials, volatilization, chemical reactions, and any other reaction that removes or alters the pollutant. The modeler must identify these processes and develop an understanding of the circumstances under which they are important to the water body and problem being assessed. Typically, this will entail both an evaluation of field data collected in the water body and review of technical literature that addresses the problem and processes being considered.

3. Perform screening analysis. Steps 1 and 2 will have at least partially identified the length and time scales for which an assessment is required and within which important processes will act. Screening analyses may be performed using the solutions provided in Section 3.3 above or using similar simple models. The goal of the screening calculations is to determine factors to which the receiving water is insensitive, and which may therefore be ignored in the modeling effort. Ultimately, one should be able to identify an approximate modeling approach or determine that such an approach is inadequate. Section 4 describes a variety of analytical solutions from the literature which may be used in screening analyses.

4. Select model and perform final analysis. Based on the results of the screening analysis, an appropriate model should be selected. The analytical solutions provided in Section 4 of this paper may, in many cases, be used for final modeling. For other problems, we provide a list of available numerical models in the discussion to follow. Selecting an appropriate numerical model may be complicated for many nonpoint source problems. Traditionally, water-quality models have been oriented towards simulating point sources, and thus carry certain inherent assumptions regarding time and space scales. Particularly, traditional water-quality models are usually steady-state models which

assume steady streamflow and pollutant discharge. These assumptions may be inappropriate for many nonpoint sources, and model assumptions, both explicit and implicit, should be carefully considered.

The potentially wide diversity of nonpoint sources makes it difficult to differentiate nonpoint source models from point source models in a systematic and general way. The problem is simplified in the following by considering the primary types of nonpoint sources: urban stormwater flows and agricultural/nonurban runoff. Both of these major sources are driven by rainfall runoff and therefore they share similarities with respect to temporal characteristics. However, nonurban nonpoint sources are far more widely distributed in space than urban stormwater discharges, a factor which creates significant differences in the spatial characteristics of the urban and nonurban sources.

For both agricultural/nonurban runoff and urban stormwater, individual pollutant discharge events are caused by rain storms and/or snowmelt events. Individual NPS events occur over short time periods and are necessarily random in their duration, frequency, and severity. However, for some water bodies, such as lakes, and types of pollutants, such as nutrients and accumulative toxics, individual events may have little direct influence on the receiving water, which instead responds to the cumulative effect of multiple events. Thus, the water-quality problem and pollutants dictate the time scale of the impact and the nature of the required model. Pollutants which exert immediate, short-term effects will require a transient water-quality model with detailed temporal resolution. These pollutants include dissolved oxygen depletion, acute toxicity due to "shock" loads, and sudden sediment and turbidity loads. Steady-state or quasidynamic models will suffice for pollutants which affect the water body more slowly, such as lake eutrophication due to nutrients or sediment contamination due to bioaccumulative or adsorbing toxics.

Based on the considerations outlined above, Table 5 presents a classification scheme according to the types of sources and pollutants, and temporal and spatial characteristics. Following the classification scheme of Table 5, Table 6 summarizes the space and time scales associated with selected water-quality problems and the type of water-quality model thereby required. Finally, in Table 7, we provide an abbreviated listing of available numerical water-quality codes which may be suitable for nonpoint source problems. Various volumes in the U.S. EPA Technical Guidance Manual for Waste Load Allocation (see the list of references) include additional lists and expanded discussions of available models and their capabilities.

Table 5. Classification scheme for NPS receiving-water problems.

| |
|--|
| Types of sources |
| Urban |
| Non-urban (predominantly agricultural) |
| Types of pollutants |
| Traditional pollutants (BOD, nitrogen, phosphorus, suspended solids, pH) |
| Inorganic micropollutants (toxic metals, other inorganic toxics) |
| Organic micropollutants (toxic organic compounds) |
| Spatial scale |
| Localized nonpoint sources |
| Distributed or dispersed nonpoint sources |
| Temporal character |
| Steady loads |
| Shock loads |

Table 6. Model selection for NPS problems.

| Type of Source and Pollutant | Space Scale | Time Scale | Model Structure |
|--|------------------------------------|-----------------|--|
| Urban Stormwater Problems | | | |
| Acute pollutants in all waters (coliform bacteria, sediment, acute dissolved oxygen depletion) | Near field | Hours | Spill-event models (transient stream models, dilution models) |
| Chronic pollutants in rivers (chronic toxic materials, continuous coliform discharges, continuous DO depletion) | Far field | Days | "Traditional" stream models (river toxics models, first-order decay models, Streeter-Phelps models) |
| Nutrients in lakes and seas (phosphorus for lakes, nitrogen for seas) | Entire water body | Months to years | Long-term loading models (Vollenweider-type models, one- or multiple-box models) |
| Accumulative pollutants in all waters (PCBs, pesticides) | Entire water body | Months to years | Long-term toxics models (one- or multiple-box models, river toxics models) |
| Agricultural/Nonurban Runoff Problems | | | |
| Chronic pollutants in rivers (chronic toxic materials, sediment loads, continuous DO depletion) | Far field, entire water body | Days to months | "Traditional" stream models (river toxics models, sediment models Streeter-Phelps models) |
| Accumulative pollutants in all waters (pesticides and herbicides) | Entire water body | Months to years | Long-term toxics models (one- or multiple-box models, river toxics models) |
| Nutrients in lakes and seas (phosphorus for lakes, nitrogen for seas) | Entire water body | Months to years | Long-term loading models (Vollenweider-type models, one- or multiple-box models) |

Table 7. Available numerical models.

| Model | Environments | Time Domain | Spatial Domain | Pollutants | Reference |
|-----------------------------|--------------------------|-----------------------------|---------------------------|--|----------------------------|
| WASP4 | Lakes, rivers, estuaries | Steady-state, dynamic | 3-dimensional | Organic toxics, organic wastes and nutrients | Ambrose et al. (1988)* |
| DELWAQ | Lakes, rivers, estuaries | Steady-state, dynamic | 1-, 2-, or 3-dimensional | Organic toxics, organic wastes and nutrients | Postma (1990) |
| IMPAQT | Lakes, rivers, estuaries | Steady-state, dynamic | 1-, 2-, or 3-dimensional | Organic toxics, metals | Delft Hydraulics (1990) |
| EXAMS-II | Lakes, rivers, estuaries | Steady-state, quasi-dynamic | 3-dimensional | Organic toxics | Burns and Cline (1985) |
| WQRRS | Lakes | Dynamic | 1-dimensional (vertical) | Organic wastes and nutrients | Smith (1978) |
| CE-QUAL-R1 | Lakes | Dynamic | 1-dimensional (vertical) | Organic wastes and nutrients | WES (1986a) |
| CE-QUAL-W2 | Lakes and estuaries | Dynamic | 2-dimensional (laterally) | Organic wastes and nutrients | WES (1986b) |
| QUAL2E | Rivers | Steady-state | 1-dimensional | Organic wastes and nutrients | Brown and Barnwell (1987)* |
| DYNTOX | Rivers | Dynamic | 1-dimensional | Organic toxics and metals | Limno-Tech, Inc. (1985)* |
| Dynamic Estuary Model (DEM) | Estuaries | Quasi-dynamic | Quasi-2-dimensional | Organic wastes and nutrients | Chen and Orlob (1972) |

* The computer codes for these models are distributed by the U.S. Environmental Protection Agency's Center for Exposure Assessment Modeling (CEAM) and can be run on IBM-compatible microcomputers. Models can be obtained free of charge from CEAM either by corresponding with the Center or by accessing the Center's computer bulletin board: Center for Exposure Assessment Modeling, U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA 30613-0801 USA, Telephone: 1 (706) 546-3130, Fax: 1 (706) 546-2018, Bulletin Board: 1 (706) 546-3402

4. WATER QUALITY MODELS

The following discussion of water quality models describes both simple models, which can be used as screening level models to identify the severity and character of nonpoint source impacts, and detailed models, which can be used for a more complete analysis. The simple models are presented more completely, so that the reader can use the information provided in this paper to perform screening analyses. For the more detailed models, available computer codes are summarized in Table 7 for readers wishing to pursue that level of analysis.

4.1 MODELS FOR RIVERS AND STREAMS

TRADITIONAL POLLUTANTS

The most elementary model for stream pollution is the simple dilution model which assumes that pollutants entering a flowing stream immediately and fully mix within the flowing water, but are not degraded or otherwise removed from the water. The model is generally conservative in the sense it will predict higher concentrations than will actually occur. However, in wide rivers where mixing across the channel is slow it will likely underestimate concentrations. Nevertheless, it provides a quick and logical first estimate of pollutant impact on the receiving stream. Where the nonpoint source makes a negligible contribution to the streamflow, the simple dilution model can take the form:

$$c_2 = c_1 + \frac{m}{Q_R} \quad (3)$$

where, c_1 is the average concentration of pollutant in the stream upstream of the area where pollutant enters the stream [M/L³];
 c_2 is the average concentration of pollutant downstream of where pollutant enters the stream [M/L³];
 m is the mass of pollutant that enters the stream [M/T]; and
 Q_R is the streamflow [L³/T].

Where the nonpoint source makes a substantial contribution to streamflow, which is often the case for stormwater sources, the simple dilution model is:

$$c_2 = \frac{Q_R}{Q_R + Q_{NPS}} c_1 + \frac{Q_{NPS}}{Q_R + Q_{NPS}} c_{NPS} \quad (4)$$

where, c_{NPS} is the concentration of pollutant in the nonpoint source inflow [M/L³]; and
 Q_{NPS} is the flow of the nonpoint source inflow [L³/T].

The simple dilution model is admittedly a crude estimator of stream concentrations. Nevertheless, it is the basis for the probabilistic dilution model which considers the probabilistic nature of urban stormwater flows (DiToro, 1984). The model is based on Equation 4, but c_1 , c_{NPS} , Q_R , and Q_{NPS} are taken to be lognormally distributed, which implies that c_2 is approximately lognormal as well. Using probability distribution parameters for the input variables, the probability distribution parameters for c_2 can be derived. This method has the distinct advantage of explicitly acknowledging the

probabilistic nature of nonpoint source pollutant inflows. The U.S. Federal Highway Administration has developed a computer program based on these principles and on data from the U.S. National Urban Runoff Program to determine probabilistic pollution loads from highway runoff (Driscoll *et al.*, 1990).

The probabilistic model (DiToro, 1984) determines the mean and variance of the downstream concentration c_2 (as in Equation 4) as:

$$\mu(c_2) = \mu(c_{NPS}) \mu(\phi) + \mu(c_1) [1 - \mu(\phi)] \quad (5)$$

$$\sigma^2(c_2) = \sigma^2(\phi) [\mu(c_{NPS}) - \mu(c_1)]^2 + \sigma^2(c_{NPS}) [\sigma^2(\phi) + \mu^2(\phi)] + \sigma^2(c_1) [\sigma^2(\phi) + (1 - \mu(\phi))^2] \quad (6)$$

where, $\mu(x)$ is the mean of variable x ;
 $\sigma(x)$ is the standard deviation of variable x ;
 $\sigma^2(x)$ is the variance of variable x ;
 ϕ is the dilution factor = $Q_{NPS}/(Q_{NPS} + Q_R) = 1/(1 + D)$; and
 D is the dilution = Q_R/Q_{NPS} .

All other variables are as defined for Equation 4 above. To evaluate Equations 5 and 6, it is necessary to determine the α -quantile of the dilution factor, ϕ , which in turn depends upon the mean and standard deviation of ϕ . If the α -quantile of ϕ , ϕ_α , is such that the probability $\Pr\{\phi < \phi_\alpha\} = \alpha$, then:

$$\phi_\alpha = \frac{1}{1 + \exp[\mu_\ell(D) - z_\alpha \sigma_\ell(D)]} \quad (7)$$

where, $\mu_\ell(x)$ is the log mean of variable x ;
 $\sigma_\ell(x)$ is the log standard deviation of variable x ; and
 z_α is the α -quantile of a standard normal variable ($z_\alpha = \pm 1.645$ for $\alpha = 0.95, 0.05$).

(Driscoll *et al.* (1990) give a slightly different formulation for the value of ϕ_α)

From the assumption that the nonpoint source and stream flow are both lognormally distributed, it follows that the log mean and standard deviation of the dilution, D , are:

$$\mu_\ell(D) = \mu_\ell(Q_R) - \mu_\ell(Q_{NPS}) \quad (8)$$

$$\sigma_\ell^2(D) = \sigma_\ell^2(Q_R) + \sigma_\ell^2(Q_{NPS}) - 2\sigma_\ell(Q_R)\sigma_\ell(Q_{NPS})\rho_\ell(Q_R, Q_{NPS}) \quad (9)$$

where, $\rho_\ell(Q_R, Q_{NPS})$ is the cross correlation between $\ln(Q_R)$ and $\ln(Q_{NPS})$ (often assumed to be zero for computational simplicity).

Next, it is assumed that ϕ is approximately lognormal and held to be exactly lognormal at the α and $\alpha-1$ quantiles, which leads to:

$$\mu_\ell(\phi) = \frac{1}{2} [\ln(\phi_\alpha) + \ln(\phi_{1-\alpha})] \quad (10)$$

$$\sigma_\ell(\phi) = \frac{1}{2z_\alpha} [\ln(\phi_\alpha) - \ln(\phi_{1-\alpha})] \quad (11)$$

The normal moments of a variable x are directly related to the lognormal moments by:

$$\mu(x) = \exp \left[\mu_\ell(x) + \frac{1}{2} \sigma_\ell^2(x) \right] \quad (12)$$

$$v^2(x) = \exp [\sigma_\ell^2(x)] - 1 \quad (13)$$

$$\sigma(x) = \mu(x) v(x) \quad (14)$$

and conversely:

$$\mu_\ell(x) = \ln \left(\frac{\mu(x)}{\sqrt{1 + v^2(x)}} \right) \quad (15)$$

$$\sigma_\ell(x) = \sqrt{\ln(1 + v^2(x))} \quad (16)$$

For some parameters, the median, $m(x)$, may be available rather than the mean. For a lognormal distribution, the arithmetic and log means are related to the median as:

$$\mu(x) = m(x) \sqrt{1 + v^2(x)} \quad (17)$$

$$\mu_\ell(x) = \ln(m(x)) \quad (18)$$

where, $m(x)$ is the median of variable x .

EXAMPLE 1: PROBABILISTIC DILUTION MODEL

Problem conditions.

Use the Probabilistic Dilution Model to evaluate the impacts of highway runoff on the concentration of total suspended solids (TSS), chemical oxygen demand (COD) and lead (Pb) in the receiving stream. Analysis of streamflow records has yielded the following statistics:

$$\text{Mean streamflow, } \mu(Q_R) = 0.28 \text{ m}^3/\text{sec}$$

$$\text{Standard deviation of streamflow, } \sigma(Q_R) = 0.42 \text{ m}^3/\text{sec}$$

$$\text{Coefficient of variation of streamflow, } v(Q_R) = \sigma(Q_R)/\mu(Q_R) = 1.5$$

There are no direct measurements of runoff flow, but a rainfall-runoff model has been used to estimate runoff as:

$$\text{Mean runoff, } \mu(Q_{NPS}) = 0.13 \text{ m}^3/\text{sec}$$

The coefficient of variation of runoff flow is assumed equal to that for rainfall intensity, which has been found from analysis of precipitation records:

$$\text{Coefficient of variation of runoff flow, } v(Q_{NPS}) = 1.3$$

Analysis of historical stream water quality data gives the following:

$$\text{Mean upstream TSS concentration} = \mu(c_{1,TSS}) = 85 \text{ mg/l}$$

$$\text{Mean upstream COD concentration} = \mu(c_{1,COD}) = 25 \text{ mg/l}$$

$$\text{Mean upstream Pb concentration} = \mu(c_{1,Pb}) = 0.004 \text{ mg/l}$$

$$\text{Standard deviation upstream TSS concentration} = \sigma(c_{1,TSS}) = 153 \text{ mg/l}$$

$$\text{Standard deviation upstream COD concentration} = \sigma(c_{1,COD}) = 16.5 \text{ mg/l}$$

$$\text{Standard deviation upstream Pb concentration} = \sigma(c_{1,Pb}) = 0.019 \text{ mg/l}$$

Based on data from the U.S. EPA National Urban Runoff Program (NURP), Driscoll *et al.* (1990) give the following statistics for concentrations in urban highway runoff:

$$\text{Median concentration of TSS, } m(c_{NPS,TSS}) = 142 \text{ mg/l}$$

$$\text{Median concentration of COD, } m(c_{NPS,COD}) = 114 \text{ mg/l}$$

$$\text{Median concentration of lead, } m(c_{NPS,Pb}) = 0.4 \text{ mg/l}$$

$$\text{Coefficient of variation for all pollutants, } v(c_{NPS}) = 0.75$$

Problem solution.

Using the streamflow statistics with Equations 15 and 16 results in:

$$\text{Log mean streamflow, } \mu_\ell(Q_R) = -1.86$$

$$\text{Log standard deviation of streamflow, } \sigma_\ell(Q_R) = 1.086$$

From the runoff flow parameter values, the following are computed using Equations 14, 15 and 16:

$$\text{Standard deviation of runoff flow, } \sigma(Q_{NPS}) = v(Q_{NPS})\mu(Q_{NPS}) = 0.169 \text{ m}^3/\text{sec}$$

$$\text{Log mean runoff flow, } \mu_\ell(Q_{NPS}) = -2.53$$

$$\text{Log standard deviation of runoff flow, } \sigma_\ell(Q_{NPS}) = 0.995$$

EXAMPLE 1: PROBABILISTIC DILUTION MODEL (CONTINUED)

From the NURP data statistics, the lognormal statistics are computed from Equations 16 and 17 as:

- Log mean runoff concentration of TSS, $\mu_{\ell}(C_{NPS,TSS}) = 4.96$
- Log mean runoff concentration of COD, $\mu_{\ell}(C_{NPS,COD}) = 4.74$
- Log mean runoff concentration of lead, $\mu_{\ell}(C_{NPS,Pb}) = -0.916$
- Log standard deviation for all pollutants, $\sigma_{\ell}(C_{NPS}) = 0.668$

The arithmetic moments are then computed from Equations 12, 13, and 14:

- Runoff TSS concentration: $\mu(C_{1,TSS}) = 178 \text{ mg/l}$ $\sigma(C_{1,TSS}) = 133 \text{ mg/l}$
- Runoff COD concentration: $\mu(C_{1,COD}) = 143 \text{ mg/l}$ $\sigma(C_{1,COD}) = 107 \text{ mg/l}$
- Runoff lead concentration: $\mu(C_{1,Pb}) = 0.500 \text{ mg/l}$ $\sigma(C_{1,Pb}) = 0.375 \text{ mg/l}$

The statistics of the dilution factor are computed from $\mu_{\ell}(Q_R)$, $\sigma_{\ell}(Q_R)$, $\mu_{\ell}(Q_{NPS})$, and $\sigma_{\ell}(Q_{NPS})$ using Equations 8 and 9. The cross correlation between $\ln(Q_R)$ and $\ln(Q_{NPS})$ is assumed to be zero:

$$\begin{aligned} \mu_{\ell}(D) &= \mu_{\ell}(Q_R) - \mu_{\ell}(Q_{NPS}) = 0.673 \\ \sigma_{\ell}^2(D) &= \sigma_{\ell}^2(Q_R) + \sigma_{\ell}^2(Q_{NPS}) = 2.168 \qquad \sigma_{\ell}(D) = 1.47 \end{aligned}$$

The 5th and 95th percentiles of the dilution factor distribution are computed from Equation 7:

$$\phi_5 = 0.0433 \quad \text{and} \quad \phi_{95} = 0.852$$

From this, the statistics of the dilution factor are given by Equations 10 through 14:

$$\begin{aligned} \mu_{\ell}(\phi) &= -1.65 & \sigma_{\ell}(\phi) &= 0.905 \\ \mu(\phi) &= 0.289 & v(\phi) &= 1.13 & \sigma(\phi) &= 0.326 \end{aligned}$$

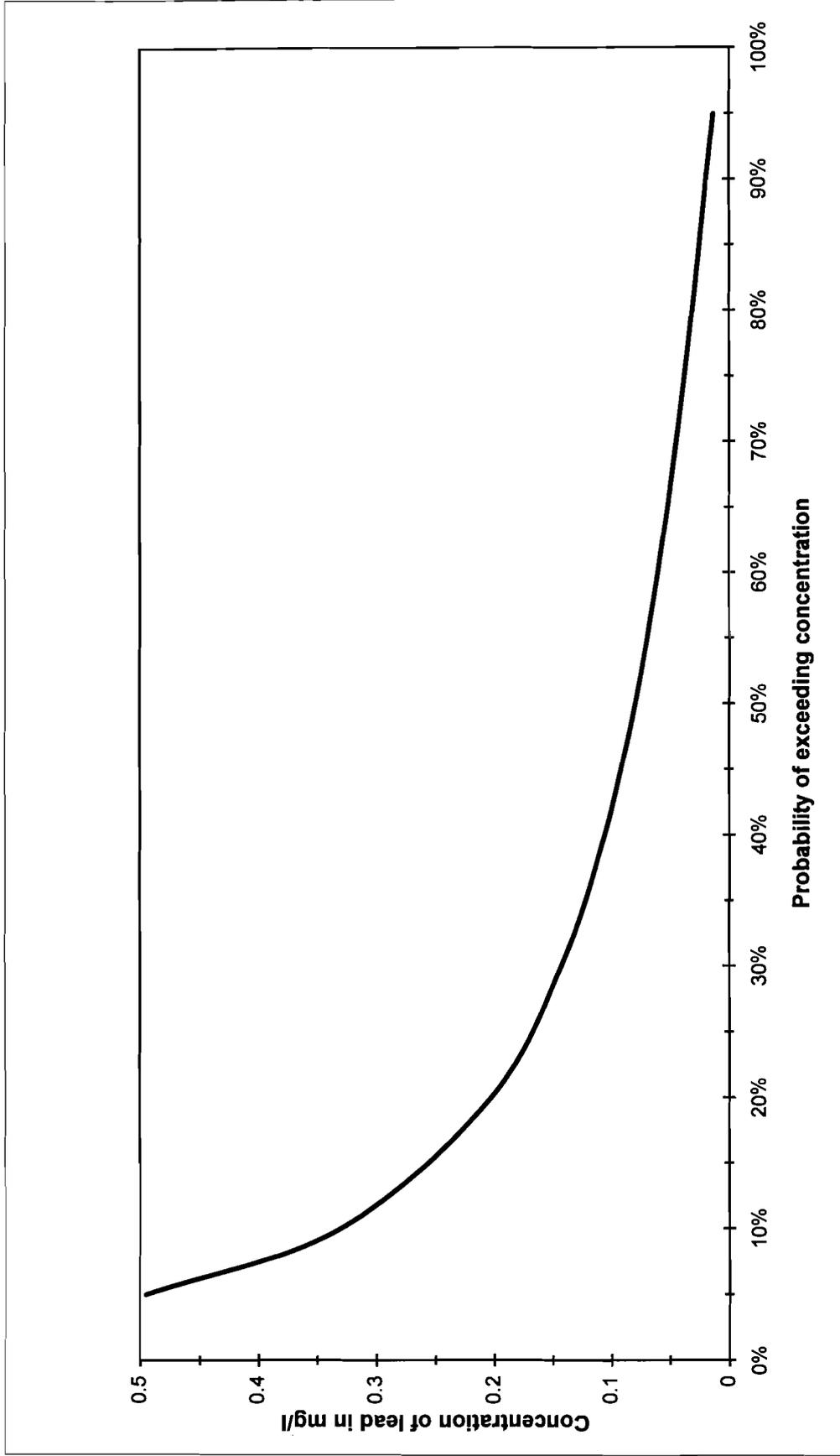
Equations 5 and 6 are then solved for the stream receiving water mean and variances:

$$\begin{aligned} \mu(C_{2,TSS}) &= 112 \text{ mg/l} & \sigma(C_{2,TSS}) &= 136 \text{ mg/l} \\ \mu(C_{2,COD}) &= 59.0 \text{ mg/l} & \sigma(C_{2,COD}) &= 61.7 \text{ mg/l} \\ \mu(C_{2,Pb}) &= 0.148 \text{ mg/l} & \sigma(C_{2,Pb}) &= 0.231 \text{ mg/l} \end{aligned}$$

Finally, Equation 7 can be used with these means and standard deviations to determine the probability distribution of downstream concentration. For example, for lead:

| α | 5% | 10% | 20% | 30% | 40% | 50% | 60% | 70% | 80% | 90% | 95% |
|------------------|--------|--------|--------|--------|--------|-------|-------|-------|-------|-------|-------|
| Exceedance prob. | 95% | 90% | 80% | 70% | 60% | 50% | 40% | 30% | 20% | 10% | 5% |
| z_{α} | -1.645 | -1.282 | -0.842 | -0.524 | -0.253 | 0 | 0.253 | 0.524 | 0.842 | 1.282 | 1.645 |
| Concentration | 0.0128 | 0.019 | 0.031 | 0.044 | 0.060 | 0.080 | 0.105 | 0.142 | 0.202 | 0.331 | 0.495 |

The concentration distribution is plotted below.



Example 1

Probability distribution for concentration of lead in receiving water
from Probabilistic Dilution Model

The procedure for applying the probabilistic dilution method is to first obtain historical records of the streamflow and stream concentration from which the arithmetic and log means and standard deviations can be determined by direct statistical analysis. The nonpoint flow is measured in rare circumstances but alternatively the mean runoff can be estimated from the mean rainfall event using standard rainfall-runoff calculations and the standard deviation of runoff can be assumed equal to that for rainfall (Driscoll *et al.*, 1990). Next, the log mean and standard deviation of D are computed from Equations 8 and 9. Then, the values of ϕ at the 5th and 95th quantiles are computed from Equation 7; the log moments of ϕ from Equations 10 and 11; and finally the arithmetic moments from Equations 12, 13, and 14. These values, together with the streamflow and concentration moments, are used in Equations 5 and 6, to find the moments of the downstream concentration.

The simple dilution model and probabilistic dilution model are strictly applicable only to pollutants which are conservative (are not degraded or otherwise removed from the stream water). The models provide a first approximation for all but the most rapidly degraded pollutants, are reasonably accurate for total dissolved solids and dissolved salts, and are often used as a reasonable model of metals. More sophisticated analysis of pollutant dynamics is often desired, however, particularly for pathogenic bacteria that die in the stream and organics that lead to dissolved oxygen depletion. The next level of model complexity considers pollutant degradation or removal as a first-order process:

$$c = c_0 e^{-kt} = c_0 e^{-kx/U} \quad (19)$$

- where,
- c is the concentration of pollutant as a function of t or x [M/L³];
 - c_0 is the concentration in the stream at the head of the reach being modeled [M/L³];
 - k is a first-order coefficient to represent the rate of degradation, removal, or decay [1/T];
 - t is time of travel along the stream [T];
 - x is distance along the stream [L]; and
 - U is the average streamflow velocity [L/T].

Equation 19 assumes a starting stream concentration of c_0 , for example as would be found in the stream downstream of a pollutant input. Many pollutant degradation or removal processes are taken to follow first-order decay, including die-off of fecal coliform bacteria (Bowie *et al.*, 1985); biodegradation of oxygen-demanding substances (Thomann and Mueller, 1987); biodegradation of pesticides and toxic organic chemicals (Schnoor *et al.*, 1987); and cooling of heated waters (Edinger *et al.*, 1974). The references indicated provide guidance and literature values for selection of first-order coefficient values that are appropriate for particular pollutants and processes.

A powerful application of the principle of first-order degradation is the well known Streeter-Phelps model, which couples equations for the first-order degradation of organic matter (represented as biochemical oxygen demand or BOD) and the accompanying consumption of dissolved oxygen (DO) and counteracting replenishment through the water surface. This equation has applicability to nonpoint sources because urban runoff and other nonpoint source loads often include oxygen demanding materials. The Streeter-Phelps equation for the concentration of dissolved oxygen in the stream is (Thomann and Mueller, 1987):

$$c = c_s - \left\{ \frac{k_d}{k_a - k_r} \left[\exp\left(-k_r \frac{x}{U}\right) - \exp\left(-k_a \frac{x}{U}\right) \right] \right\} L_0 - (c_s - c_0) \exp\left(-k_a \frac{x}{U}\right) \quad (20)$$

- where,
- c is the concentration of dissolved oxygen in the water [M/L³];
 - c_s is the saturation concentration of dissolved oxygen [M/L³];
 - k_d is the deoxygenation rate (the rate of DO consumption as BOD is degraded) [1/T];
 - $k_r = k_d + k_s$ is the overall loss rate of BOD from the water column [1/T];
 - k_s is the settling rate of BOD from the water column [1/T];
 - k_a is the reaeration rate, the volumetric rate at which oxygen enters the water column through the surface [1/T];
 - c_0 is the concentration of dissolved oxygen at $x=0$ [M/L³]; and
 - L_0 is the concentration of BOD at $x=0$ [M/L³].

This solution predicts the classic “DO sag curve” in which the concentration of dissolved oxygen decreases with distance downstream from the source, reaches a minimum at the DO sag point, and then recovers with further distance downstream (Figure 5).

The dissolved concentration at its minimum, at the sag point, is given by the equation (Thomann and Mueller, 1987):

$$c_{\min} = c_s - \frac{k_d}{k_a} L_0 \exp(-k_r t_c^*) \quad (21)$$

- where,
- c_{\min} is the minimum concentration of dissolved oxygen downstream of the source, and
 - t_c^* is defined as follows:

$$t_c^* = \frac{1}{k_a - k_r} \ln \left\{ \frac{k_a}{k_r} \left[1 - \frac{(c_s - c_0)(k_a - k_r)}{k_d L_0} \right] \right\} \quad (22)$$

The Streeter-Phelps-type analytical equation can be extended to include oxygen demand by nitrification and sediment sources, oxygen loss and gain associated with algal respiration and photosynthesis, and other oxygen-demanding or contributing processes (Thomann and Mueller, 1987).

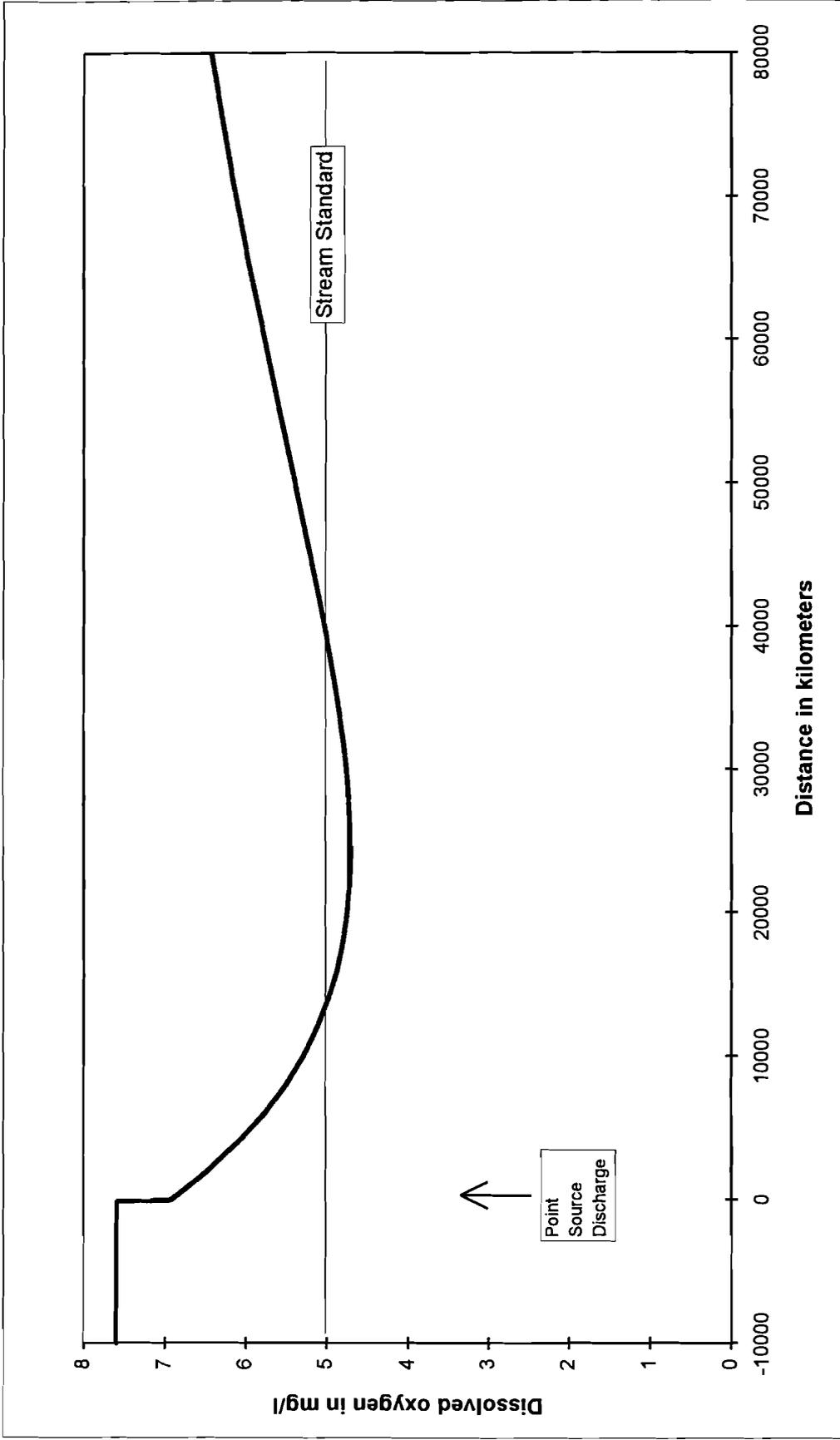


Figure 5
Dissolved oxygen concentration in a stream below a point source discharge

Equations 20 through 22 apply to a point source of BOD or a localized distributed source. For widely distributed nonpoint sources, an equation that considers the distributed input of pollution is desirable. Li (1962) gives a generalized solution for a distributed source. For this solution, we assume a constant NPS inflow rate, q , given in units of volume of inflow per unit time per unit length of stream [L^2/T] (e.g., m^3/s of inflow per meter of stream length) with a constant BOD concentration, L_{NPS} [M/L^3], and dissolved oxygen concentration, c_{NPS} [M/L^3]. Also, unlike Li, we have distinguished the BOD decay rate, k_d , from the BOD settling rate, k_s . With these changes, Li's solution yields:

$$c = c_s - \left\{ \frac{k_d}{k'_a - k'_r} \left[\exp\left(-k'_r \frac{x}{U}\right) - \exp\left(-k'_a \frac{x}{U}\right) \right] \right\} \left[L_0 + \frac{L_{NPS}q}{k'_r A} \right] - (c_s - c_0) \exp\left(-k'_a \frac{x}{U}\right) - \left\{ \frac{q}{k'_a A} \left[1 - \exp\left(-k'_a \frac{x}{U}\right) \right] \right\} \left[(c_s - c_{NPS}) + \frac{L_{NPS}k_d}{k'_r} \right] \quad (23)$$

where, k'_a is the 'effective' reaeration rate [$1/T$] = $k_a + q/A$;
 k'_r is the 'effective' BOD loss rate [$1/T$] = $k_r + q/A$;
 L_{NPS} is the BOD concentration of the NPS inflow [M/L^3];
 q is the NPS inflow per unit length of stream [L^2/T]; and
 A is the cross-sectional area of the stream [L^2].

The accompanying change in the stream BOD is given by the equation:

$$L = L_0 \exp\left(-k'_r \frac{x}{U}\right) + \frac{L_{NPS}q}{k'_r A} \left[1 - \exp\left(-k'_r \frac{x}{U}\right) \right] \quad (24)$$

Note that the effect of the distributed NPS load appears in these equations as a modification of the reaeration and BOD loss rates. This has the potential to confuse the calibration of these parameters in models of stream dissolved oxygen for either point or nonpoint sources.

More sophisticated models of dissolved oxygen, BOD, and nutrients in streams are available. Probably the most widely used is the QUAL2 model, developed by the U.S. EPA (Brown and Barnwell, 1987) and available in a microcomputer version. In addition to DO, BOD, and nutrients, QUAL2 has capabilities to model temperature, conservative substances, and pollutants which degrade (such as fecal coliform) and settle (such as sediment). The model also considers distributed sources as well as point sources. Other available models for dissolved oxygen are reviewed by Driscoll *et al.* (1983).

Parameter values for stream dissolved oxygen models are typically determined from a combination of field data and literature values. Useful reviews of literature values are provided by Thomann and Mueller (1987) and Bowie *et al.* (1985).

EXAMPLE 2: STREETER-PHELPS MODEL FOR STREAM DISSOLVED OXYGEN

Point-source discharge

Problem conditions.

Determine the dissolved oxygen profile downstream of a municipal sewage discharge to the Blackstone River during low-flow conditions. The effluent discharge is as follows:

Effluent BOD, $L_{PS} = 30$ mg/l

Effluent DO, $c_{PS} = 6$ mg/l

Effluent flow rate, $Q_{PS} = 90,700$ m³/d

The river conditions are:

BOD upstream of discharge, $L_R = 4.0$ mg/l

DO upstream of discharge, $c_R = 7.6$ mg/l

Saturation DO, $c_S = 7.7$ mg/l

Deoxygenation rate, $k_d = 0.3$ day⁻¹

BOD settling rate, $k_s = 0.1$ day⁻¹

Deoxygenation rate, $k_r = k_d + k_s = 0.4$ day⁻¹

Reaeration rate, $k_a = 0.8$ day⁻¹

River flow rate, $Q_R = 1.5$ m³/s

Problem solution.

First determine the initial concentrations immediately downstream of the discharge assuming full mixing. The equation is similar to Equation 4:

$$c_0 = \frac{Q_R}{Q_R + Q_{PS}} c_R + \frac{Q_{PS}}{Q_R + Q_{PS}} c_{PS}$$

The parallel equation holds for the initial BOD, L_0 .

The solutions for c_0 and L_0 are:

$$c_0 = 6.9 \text{ mg/l}$$

$$L_0 = 14.8 \text{ mg/l}$$

The solution for the stream dissolved oxygen concentration as a function of x , the distance downstream of the point-source discharge is determined from Equation 20. The solution is plotted below.

EXAMPLE 2: STREETER-PHELPS MODEL FOR STREAM DISSOLVED OXYGEN (CONTINUED)

Nonpoint-source discharge

Problem conditions.

Assume conditions such that the nonpoint-source discharges at the same concentration and flow as the point source above, except that the flow is uniformly distributed over a 2000-meter reach. Therefore,

$$\text{NPS inflow per unit length, } q = Q_{PS}/2000 = 46 \text{ m}^3/\text{d}$$

$$\text{NPS BOD, } L_{NPS} = 30 \text{ mg/l}$$

All other parameters are the same as for the point-source problem. The effective reaeration and BOD loss rates are:

$$\text{Effective deoxygenation rate, } k'_d = k_d + q/A = 3.6 \text{ day}^{-1}$$

$$\text{Effective reaeration rate, } k'_a = k_a + q/A = 4.1 \text{ day}^{-1}$$

Solution.

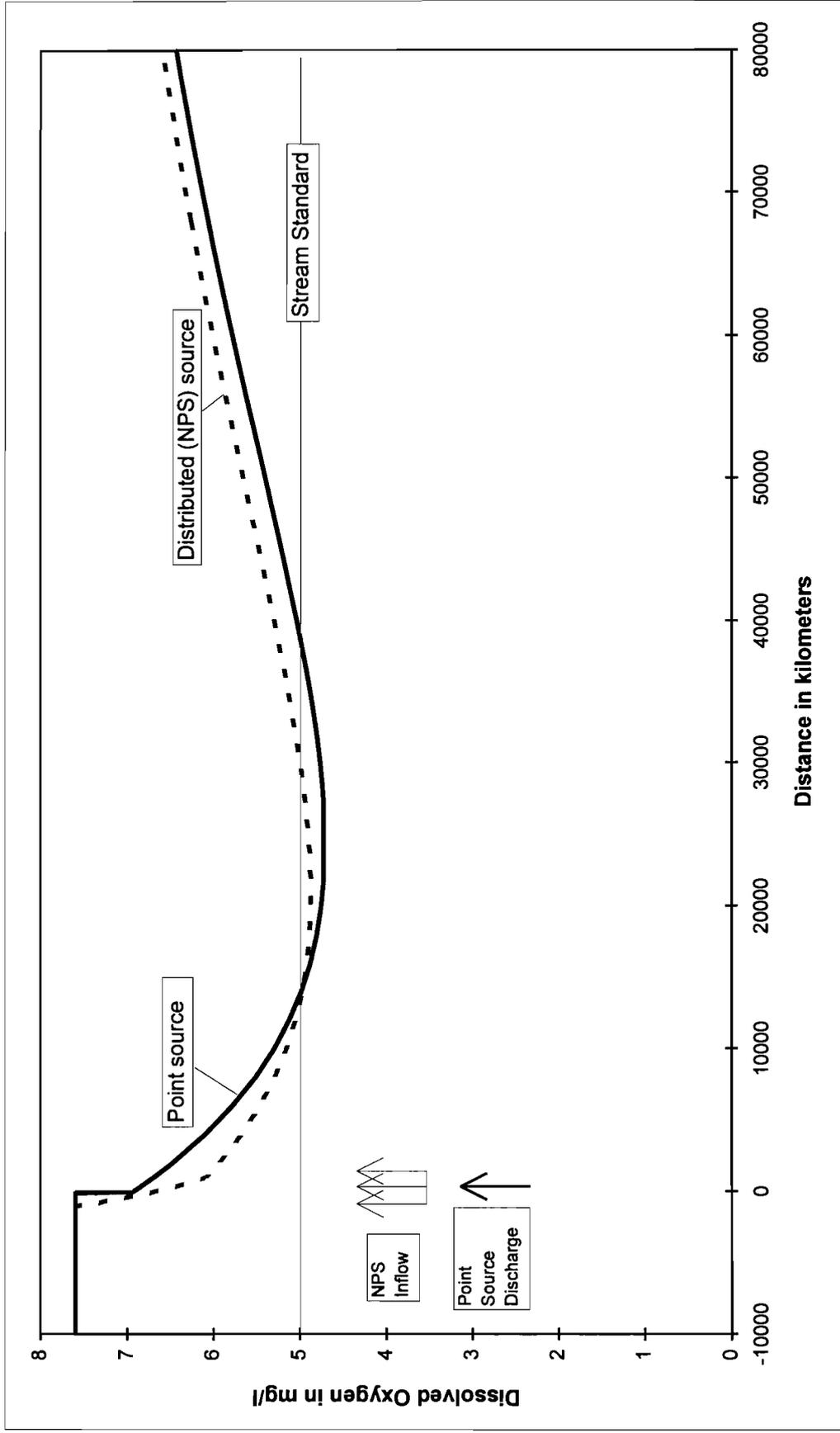
The problem solution must be broken into reaches, with Equations 23 and 24 used in the reach of NPS inflow, and Equation 20 used for the remaining downstream section. Taking the NPS inflow reach to extend from $x = -1000$ to 1000 meters, Equations 23 and 24 are solved by replacing x with $x' = x + 1000$. The solution at selected points is:

| x | c | L |
|-------|-----|------|
| -1000 | 7.6 | 4.0 |
| -500 | 7.0 | 7.0 |
| 0 | 6.6 | 8.7 |
| 500 | 6.2 | 10.7 |
| 1000 | 5.9 | 12.5 |

The values of c and L at the end of the reach, not become c_0 and L_0 for the solution of Equation 20. Because the reach solved by Equation 20 begins at $x = 1000$ m, the x variable must again be replaced, this time with $x'' = x - 1000$. The solution then comes directly from Equation 20 and is also plotted in the figure, along with the NPS solution for -1000 to 1000 m.

Conclusions

The figure shows some differences between the point and nonpoint source discharges. The NPS discharge causes a more rapid decrease in dissolved oxygen over the 2000-meter inflow reach, but then stream DO recovers somewhat more quickly downstream. The overall difference is minor and at large distances from the source, the two solutions converge.



Example 2

Dissolved oxygen concentration in a stream below a point source and nonpoint source of equal strength

Suspended sediment is a pollutant commonly associated with nonpoint sources. Modeling of suspended sediment in surface water bodies recognizes that the sediment in the water column is the result of a balance between resuspension and settling (Thomann and Mueller, 1987). Presuming that there is a net settling of sediments from the water to the bed, the net settling rate, v_n , is:

$$v_n = v_s - v_u \quad (25)$$

where v_n is the net settling rate [L/T];
 v_s is the rate of sediment settling in the water column [L/T]; and
 v_u is the rate of sediment resuspension from the bed [L/T].

Typically, the net settling rate is determined by model calibration to field data. Presuming that an appropriate and representative net settling rate can be defined, the equation for suspended sediment in the water column downstream of a point input is (Thomann and Mueller, 1987):

$$S = S_0 \exp\left[-\frac{v_n}{HU} x\right] \quad (26)$$

where S is the concentration of suspended sediments as a function of x [M/L³];
 S_0 is the concentration at the source [M/L³];
 x is distance downstream from the source [L]; and
 H is the water depth [L].

All of the solutions given in this section are applicable to problems in which stream dispersion can be neglected. Dispersion is almost always neglected for steady-state problems, in which both the pollutant discharges and streamflow are steady. For many other problems, neglecting dispersion is a conservative assumption which results in predicting a higher concentration than if dispersion were accounted for. However, for some other problems such as shock loads associated with a sudden spill or NPS load, dispersion is important. Solutions provided below for estuaries, in which dispersion is always a factor, can be used stream and river problems with appropriate dispersion coefficients.

MICROPOLLUTANTS

Models of toxic substances in the environment generally focus on their propensity to adsorb to solids such as suspended sediments in stream water. The partitioning between the phase of chemical dissolved in the water column (represented by the variable c) and that adsorbed to sediment (c_{ss}) is usually represented as an equilibrium process:

$$c_s = K_d c^b \quad (27)$$

where, c_s is the mass of chemical adsorbed to solids per unit dry weight of solids [M/M];
 c is the mass of chemical dissolved in the water [M/L³];
 K_d is the partition coefficient [L³/M]; and
 b is a constant [dimensionless].

Often, a linear partitioning model (Freundlich isotherm) is presumed to apply at the low concentrations typical in water-quality problems, and b is equal to 1. O'Connor (1988) provides a helpful perspective on the partition coefficient. He points out that there is disagreement within the literature on phenomenological aspects behind the partition coefficient and that it is essentially an empirical coefficient that represents a possible multitude of processes. He goes on, however, to emphasize the parallels between the partition coefficient and the widely used first-order deoxygenation rate for BOD, which successfully but empirically represents a multitude of microscale processes that lead to biodegradation. As with the BOD deoxygenation rate, the best results are achieved when measurements are made in the stream to define an appropriate value of the partition coefficient.

Other factors that may affect organic chemicals in water are volatilization and chemical or biochemical degradation. Volatilization may be represented as a volatilization rate [L/T] divided by the depth of the water body [L] to arrive at a pseudo-first-order coefficient, k_v , [1/T]. The rate of volatilization will depend on the partial pressure of the chemical in the atmosphere overlying the water surface, which for most situations can be taken as zero. Degradation reactions are typically assumed to follow a first-order decay also, with a rate constant of k_d [1/T]. Incorporating the assumptions of zero overlying partial pressure and a constant suspended sediment concentration, S , yields the solution (Mills *et al.*, 1985):

$$c = c_0 \exp\left[-\frac{(k_v + k_d) x}{1 + K_d S U}\right] \quad (28)$$

The effect of partitioning to suspended sediments appears in the solution as effectively more rapid decay, by a factor of $(1 + K_d S)$.

The next level of complexity in modeling toxic substances in streams involves the simultaneous solution of stream sediment and chemical concentrations. Thomann and Mueller (1987) and O'Connor (1988) develop similar models for toxic chemicals in streams with O'Connor's being the more complex. Thomann and Mueller's solution for a steady-state condition with no net loss or gain of sediment is:

$$c = \frac{m}{Q} \exp\left[-\frac{v_T x}{H U}\right] \quad (29)$$

where, c is the chemical concentration in the water column [M/L³];
 m is the mass of pollutant entering the stream per unit time at $x=0$ [M/T];
 H is the depth of the water [L]; and
 v_T is the net chemical loss rate [L/T].

Thomann and Mueller (1987) recommend determining the net loss rate, v_T , from field studies in the stream but also give formulae to define the loss rate from chemical and stream characteristics. The formulae, which are complex, may be found in Thomann and Mueller (1987) or O'Connor (1988). The solution for the concentration of chemical in the sediments is similar in form to that for the concentration in the water column, but differs by a factor of proportionality that depends upon the differing partition coefficients, diffusion rates, porosities, and solids concentrations in the sediment versus the water. Again, Thomann and Mueller (1987) or O'Connor (1988) give complicated equations for this factor.

Several sources provide information on the parameter values required to model toxic chemicals. References oriented specifically to water-quality modeling include Delos *et al.* (1984), Schnoor *et al.* (1987), and Mills *et al.* (1985). In addition, a number of references provide more general information on chemicals in the environment. Useful handbooks include Verschueren (1983), Lyman *et al.* (1990), and Howard *et al.* (1991).

More sophisticated models of chemical substances in streams are available, and are reviewed by Delos *et al.* (1984) and Schnoor *et al.* (1987). Perhaps the most widely used model is TOXIWASP, a version of the U.S. EPA WASP model designed specifically for toxic substances (Ambrose *et al.*, 1988). The WASP modeling system includes several separate computer programs: a hydrodynamics program, DYNHYD, which simulates the movement of water under tidal conditions; a eutrophication water-quality model, EUTROWASP, which simulates dissolved oxygen, BOD, and nutrients; and the toxics water-quality model, TOXIWASP. The hydrodynamics program is optional; hydrodynamic flows can alternatively be specified in the WASP program inputs. The WASP water-quality models share common algorithms to model the mixing and transport of pollutants, and to solve for pollutant concentration over time. The TOXIWASP and EUTROWASP components supply appropriate biochemical and physical reaction information. Because of the model's generic structure, the WASP models can be used to simulate a variety of water-body types. The TOXIWASP program itself provides algorithms to model a full range of processes affecting chemicals in the environment including ionization, adsorption, biodegradation, volatilization, photolysis, hydrolysis, oxidation, and transformations between chemicals.

4.2 MODELS FOR LAKES AND RESERVOIRS

NUTRIENTS AND EUTROPHICATION

Nonpoint source pollutants have the potential to cause cultural eutrophication of lakes as well as to add sediments and toxic chemicals. Cultural eutrophication is a term used to describe the undesirable effects in the water quality of a lake when it is unnaturally enriched by fertilizers and other sources of nutrients from man's activities. The most important nutrient relative to eutrophication of lakes and reservoirs is phosphorus, an essential element for the growth of aquatic plants and algae. When phosphorus is overabundant, it can lead to an excessive growth of algae and aquatic weeds, and an accompanying reduction in water quality.

Prediction of lake eutrophication effects requires an accounting of the phosphorus entering the lake. The response of the pond to phosphorus may be examined through a variety of analytical tools of increasing complexity. Simple loading models are a classical technique to make a rough estimation of a lake's trophic status. The concept of the loading model was advanced and refined by Richard Vollenweider and numerous variations have been developed from his basic idea. These models are useful in identifying the potential magnitude of the lake's eutrophication problems.

The Vollenweider equation gives the steady-state concentration of phosphorus in the lake water as (Reckhow and Chapra, 1983):

$$P = \frac{W}{\bar{z}} \frac{1}{\frac{1}{\tau} + \sqrt{\frac{1}{\tau}}} \quad (30)$$

where, P is the steady-state phosphorus concentration in the lake [M/L^3];
 W is the phosphorus loading rate per unit surface area of the lake [$M/L^2/T$];
 \bar{z} is the average depth of the lake [L];
 τ is the hydraulic residence time equal to V/Q [T];
 Q is the average inflow into the lake [L^3/T]; and
 V is the lake volume [L^3].

Although the Vollenweider model simplifies the lake's dynamics considerably, it is built upon a solid base of empirical data and has proven to be a highly useful predictor of lake eutrophication (see for example, van Straten, 1986). Variations on the loading model concept have been developed by a number of authors and are reviewed by Reckhow and Chapra (1983).

EXAMPLE 3: VOLLENWEIDER MODEL FOR LAKE EUTROPHICATION

Problem conditions.

Estimate the steady-state concentration of phosphorus in Lake Balaton, Hungary for the whole lake and Keszthely Bay (see Section 5).

Somlyódy and van Straten (1986) give the following properties of the lake:

| | <u>Whole lake</u> | <u>Keszthely Bay</u> |
|-----------------------------------|---------------------|----------------------|
| Volume, V (m^3) | 1.907×10^9 | 8.2×10^7 |
| Average depth, \bar{z} (m) | 3.2 | 2.3 |
| Average inflow, Q (m^3/s) | 30 | 10.4 |
| Surface area, A (km^2) | 596 | 38 |
| Total phosphorus load, L (kg/d) | 850 | 259 |

Problem solution.

From these, the other Vollenweider model parameters are computed as:

| | | |
|---|-----|------|
| Hydraulic residence time, $\tau = V/Q$ (years) | 2.0 | 0.25 |
| Phosphorus loading rate, $W = L/A$ ($mg/yr\text{-}m^2$) | 530 | 2500 |

Next use the Vollenweider model, Equation 30, to solve for the lake phosphorus concentration, P :

| | | |
|---|------|------|
| Steady-state phosphorus concentration, P (mg/l) | 0.14 | 0.18 |
|---|------|------|

Conclusions

The observed annual average total P concentration is approximately 0.07 mg/l in Keszthely Bay but only about 0.04 mg/l averaged across the lake (Somlyódy and van Straten, 1986). Thus, in this instance at least, the Vollenweider model predicts concentrations that are comparable to but somewhat higher than actual observed concentrations. The Vollenweider model was developed for deep lakes and is not strictly applicable to the shallow Lake Balaton (although there are other formulas for deep lakes—see van Straten, 1986). A significant factor in the applicability of the Vollenweider model to a specific lake is the apparent settling rate. As noted by Thomann and Mueller (1987), the settling rate can be an order of magnitude higher than presumed in the Vollenweider formulation. A high settling rate in Lake Balaton is likely responsible for the difference between predicted and actual concentrations.

MICROPOLLUTANTS

By analogy with the phosphorus loading model, Chapra (1991) developed a toxicant loading model for toxic chemicals in lakes. The model considers both sediments and toxic chemicals, and their interaction through adsorption. The lake is modeled as a single fully-mixed body in the same fashion as assumed by the phosphorus loading models. Chapra provides equations for the solids and chemical concentrations (S and c) as:

$$S = \frac{q_s}{q_s + (1 - F_r)v_s} S_0 \quad (31)$$

$$c = \frac{q_s}{q_s + kH + v_v f_{d1} + (1 - F_r')(v_s f_{p1} + v_d f_{d1})} \quad (32)$$

- where,
- q_s is the lake's hydraulic loading rate = Q/A [L/T];
 - A is the lake surface area [L²];
 - F_r is the dimensionless resuspension ratio = $v_u/(v_b + v_u)$; and
 - v_b is the burial velocity or sedimentation rate for permanent burial of sediment [L/T];
 - H is the lake depth [L];
 - k is the lumped first-order decay coefficient for chemical in the lake [1/T];
 - v_v is the volatilization mass-transfer coefficient [L/T];
 - f_{d1} = $1/(1+K_d S)$ is the fraction of chemical in dissolved form in the lake;
 - f_{p1} = $1 - f_{d1}$ is the fraction of chemical adsorbed to suspended sediment in the lake;
 - F_r' is the dimensionless recycle ratio: $F_r' = \frac{v_u + v_d f_{d2}}{v_u + v_d f_{d2} + v_b + k_s z_s}$
 - f_{d2} = $1/(\phi + K_d[1-\phi]\rho)$ is the ratio of the sediment pore-water chemical concentration to the total concentration in the sediments;
 - ϕ is the sediment porosity [dimensionless]; and
 - ρ is the density of the sediment [M/L³].

All other variables are as defined previously.

The next level of complexity in modeling lake water quality accounts for the significant temperature stratification that occurs during the summer in lakes deeper than about three meters in temperate climates. During the summer, the surface layer of the lake (the epilimnion) is warmed by the sun, and becomes distinctly warmer than the underlying waters (the hypolimnion). The warmer and cooler layer are separated by the thermocline, a zone in which water temperature decreases rapidly with depth. The thermocline acts as a strong barrier to exchange of water between the epilimnion and hypolimnion, leading to distinct chemical environments in the two layers.

Thomann and Mueller (1987) present an analytical model which considers the separation of the epilimnion and hypolimnion. The model, which is not specific to phosphorus, gives the concentration of chemical in the epilimnion and hypolimnion. Inflow to the lake is considered to flow into the epilimnion only.

$$c_e = \frac{(W_e + \beta W_h)/Q}{1 + (1 - \beta) E_v'/Q + V_e k_e/Q} \quad (33)$$

$$c_h = \beta \left(c_e + \frac{W_h}{E_v'} \right) \quad (34)$$

where c_e, c_h are the chemical concentrations in the epilimnion and hypolimnion [M/L^3];

W_e, W_h are the chemical loads [M/T];

Q is the flow into the epilimnion [L^3/T];

E_v' is the vertical dispersive flux defined below [M^3/T];

V_e, V_h are the volumes [L^3];

k_e, k_h are lumped first-order decay coefficients; and

$$\beta = \frac{E_v'}{E_v' + V_h k_h}$$

The vertical dispersive flux is a function of the lake geometry and the vertical dispersion coefficient:

$$E_v' = \frac{E_v A_h}{\bar{z}_{eh}} \quad (35)$$

where E_v is the dispersion coefficient [L^2/T];

A_h is the interface area between the hypolimnion and epilimnion [L^2]; and

\bar{z}_{eh} is the vertical distance between middepth in the epilimnion and middepth in the hypolimnion [L].

Typically, the vertical dispersion coefficient is very small and can be assumed equal to the molecular diffusion coefficient ($5 \times 10^{-5} \text{ m}^2/\text{sec}$) as a first approximation (Wang and Harleman, 1982). At least for phosphorus, the lumped first-order decay terms, k_e and k_h are empirical and need to be defined on a lake-specific basis.

More sophisticated alternatives to these models include a variety of numerical lake simulation models. A review of available models is provided in U.S. EPA guidance by Mancini *et al.* (1983).

4.3 ESTUARIES

The estuarine environment is characterized by considerable longitudinal mixing as a consequence of back-and-forth tidal flow. For screening level analysis, this mixing can be approximated by a tidal dispersion coefficient, E , which must be empirically estimated. There are clear limitations to this approach, and it has been severely criticized (see for example, the exchange between Thatcher and Harleman (1981, 1983) and Fischer (1981)). Nonetheless, it remains a practical and useful approach for screening analysis. Thomann and Mueller (1987) provide guidelines for estimating the tidal dispersion coefficient from field data and Thatcher and Harleman (1981) provide an empirical formula.

The solution for a substance undergoing first-order decay and discharged continuously at $x = 0$ is (Thomann and Mueller, 1987):

$$c = \frac{W}{Q \sqrt{1 + \frac{4kE}{U^2}}} \exp \left[\frac{Ux}{2E} \left(1 \pm \sqrt{1 + \frac{4kE}{U^2}} \right) \right] \quad \text{for } x \leq 0, x \geq 0 \quad (36)$$

where W is the waste load [M/T];
 U is the freshwater (nontidal) flow velocity = Q/A [L/T];
 Q is the freshwater flow rate [L^3/T];
 A is the cross-sectional area [L^2];
 k is the first-order decay coefficient [1/T]; and
 E is the tidal dispersion coefficient [L^2/T].

The solution predicts an exponentially decreasing concentration both upstream and downstream from the source, with more rapid decrease in the upstream direction.

Thomann and Mueller (1987) also give the solution for a source distributed along a length of the estuary, as would characterize many nonpoint sources. For a source that contributes a load of w mass per unit length and is distributed between $x = 0$ and $x = a$, the solution is:

$$c = \frac{w/A}{k} \left(\frac{\alpha - 1}{2\alpha} \right) \left[1 - \exp \left(- \frac{Ux}{2E} (1 + \alpha) \right) \right] \quad \text{for } x \leq 0 \quad (37a)$$

$$c = \frac{w/A}{k} \left[1 - \left(\frac{\alpha - 1}{2\alpha} \right) \exp \left(\frac{U(x-a)}{2E} (1 + \alpha) \right) - \left(\frac{\alpha + 1}{2\alpha} \right) \exp \left(\frac{Ux}{2E} (1 - \alpha) \right) \right] \quad (37b)$$

for $0 \leq x \leq a$

$$c = \frac{w/A}{k} \left(\frac{\alpha + 1}{2\alpha} \right) \left[1 - \exp \left(- \frac{U(x-a)}{2E} (1 - \alpha) \right) \right] \quad \text{for } x \geq a \quad (37c)$$

where $\alpha = \sqrt{1 + (4kE / U^2)}$
 w is the load per unit time per unit length along the estuary [M/LT]

The two solutions provided above for estuaries can also be applied to stream and rivers where longitudinal mixing (dispersion) is important. For rivers and streams, the tidal dispersion coefficient, E , is replaced by the typically much smaller riverine longitudinal dispersion coefficient.

A complete model of pollutant concentrations in an estuarine environment requires full consideration of the complex tidal circulation. In estuaries that are not mixed vertically, the additional complication of salinity stratification must be factored in. These varied and complex conditions require the use of numerical models for all but the simplest of problems. A number of available estuarine water-quality models are reviewed by U.S. EPA (1987) and further guidance on using estuarine models is provided by Ambrose and Martin (1990) and Martin *et al.* (1990).

5. CASE STUDY - LAKE BALATON AND THE ZALA RIVER, HUNGARY

5.1 BACKGROUND

Lake Balaton in Hungary is one of the largest shallow lakes in the world (Figure 6) with a surface area of close to 600 km² but an average depth of only 3.1 m. The lake and its vicinity is the most important recreational area in Hungary, drawing tourists from both inside and outside the country. The lake faces deteriorating water quality due to artificial eutrophication caused by increased nutrient loads. Nutrient loads are rising due to the increased application of fertilizers in the watershed as well as the development of tourism, water supply, and sewerage collection (but without accompanying wastewater treatment). In the late 1970s the trophic state of the lake, particularly in its most western basin (Keszthely Bay) worsened to a critical condition, prompting a comprehensive study to be launched by IIASA and the Hungarian Academy of Sciences (Somlyódy and van Straten, 1986). The study's primary objective was to understand the watershed processes influencing nutrient loads, the behavior of the lake's ecosystem, and nutrient cycling, and to develop an optimal control policy. There was a strong focus on basic data collection and the development of various models both to develop technical understanding and provide management tools. The study was completed late 1982 and was followed by governmental decisions in 1983. The government's plan defined remedial measures and an implementation schedule. Remedial measures, which were specified in detail, included phosphorus removal at existing wastewater treatment plants, the export of sewage out of the lake's catchment, and the construction of pre-reservoirs at the mouths of tributaries to filter and remove nutrients – mostly of diffuse origin – before they enter the lake.

5.2 NUTRIENT LOADS TO THE LAKE

Lake Balaton drains an area of about 6000 km². Based on multiple years of data collection, the study estimated the annual average total phosphorus load (TP) to the lake to be about 850 kg/d (or 300 t/yr) and the total nitrogen (TN) load to be about ten times more. The contribution of sewage discharges was about 30%, while that of agricultural nonpoint sources was approximately 50%. The remaining load came from other diffuse sources, mostly urban runoff and atmospheric pollution.

The major inflow to Lake Balaton is the Zala River which discharges to Keszthely Bay at the lake's western end. The Zala drains about 40% of the lake's total catchment area, but then discharges to Keszthely Bay, which comprises only about 4 percent of lake's total volume. Thus, on a unit volume basis, the load to Keszthely Bay is an order of magnitude higher than the load to the lake as a whole. For this reason the water quality (trophic state) is poor in Keszthely Bay but gradually improves towards the eastern end of the lake.

5.3 ZALA RIVER LOAD AND THE ROLE OF THE TIME SCALE

As stated, the Zala River drains about 40% of the lake's total catchment and is thus the single greatest source of nutrients to the lake. About 60% of the TP load carried by the Zala River is of agricultural nonpoint source origin. Beginning in 1975, daily measurements of phosphorus and nitrogen were made at the mouth and at other locations on the Zala. Figure 7 shows streamflow and total phosphorus measured at the mouth of the Zala during 1978 along with precipitation. Obviously, precipitation strongly influences both the flow and TP load. As seen in Figure 7, the load can increase from the background value of about 200 kg/d to above 2000 kg/d.

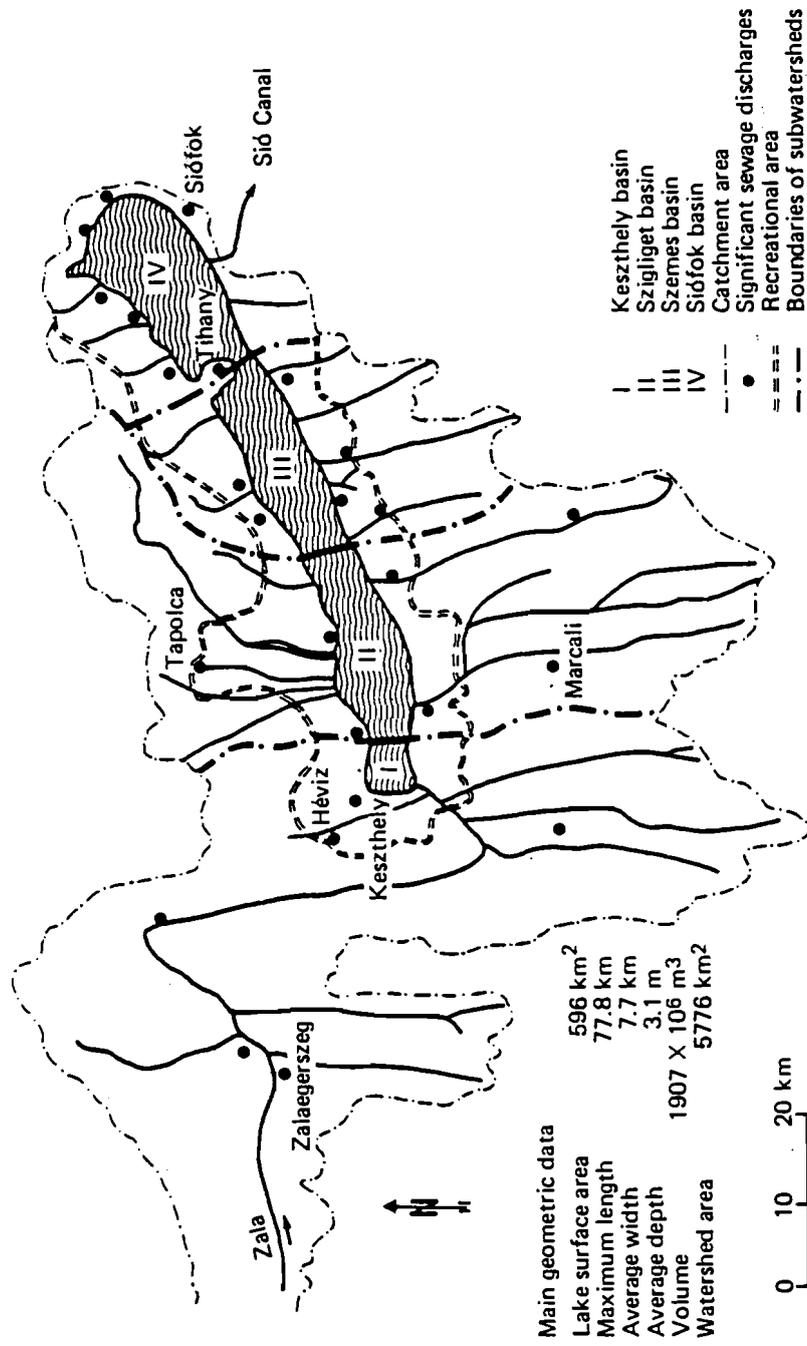


Figure 6
Lake Balaton and its watershed, Hungary

Figure 7 also shows the results of calibrating a discrete-time multiple-input single-output time-series model using instrumental variables (Beck, 1982). Although the figure shows the calibration stage, and not the usually less accurate prediction stage, the relatively poor agreement between observations and simulation is apparent. This state-of-the-art study illustrates that even deterministic simulation of historic nutrient loads on a daily time step is beyond the capabilities of existing models. This creates a serious difficulty if it is desired to predict changes in the water quality of Keszthely Bay on a short time scale.

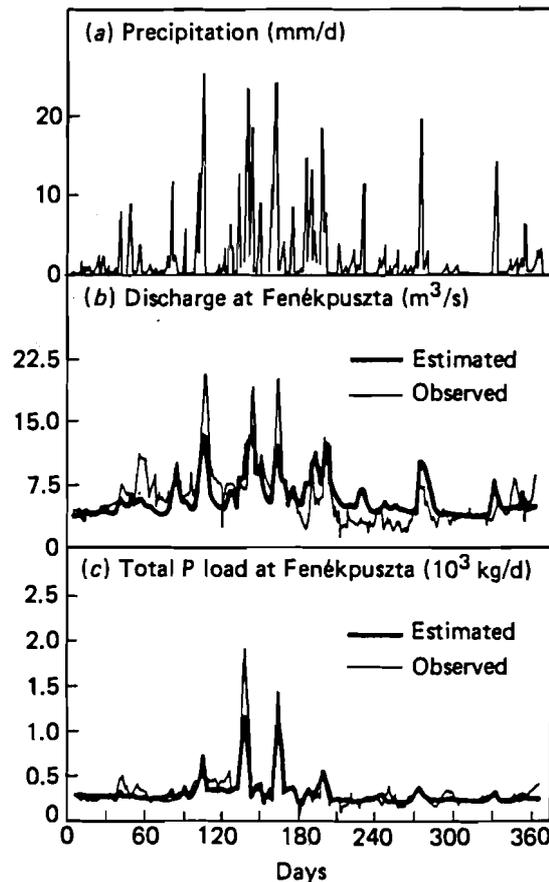


Figure 7
 Calibration of the Lake Balaton time-series loading model:
 (a) observed precipitation during 1978
 (b) observed and estimated Zala River discharge
 (c) total P load at Fenékpuszta

This raises at least two questions: First, what is the time scale at which acceptable load estimates can be obtained? And, second, are these estimates sufficiently detailed relative to the load response of the lake and Keszthely Bay? To find an answer to the first question monthly average data were generated for flow, suspended solids load, and total phosphorus load. In addition, the standard deviation of the daily flow was also calculated in order to characterize the month-to-month variability of the flow due to rainfall-runoff events. Subsequently, linear regression analyses were performed using data for 1976 through 1979 for calibration and data for 1980 through 1983 for validation (see Somlyódy and Jolánkai, 1986 for details). These analyses tested the dependence of the TP load on the flow alone, on the combination of the flow and the suspended solids load (assuming TP also depended on the

dynamics of particulate material), and on the standard deviation of the flow. All models gave satisfactory results (see Figure 8 for validation results), but because it was the simplest and easiest to implement, the first model was used to generate a synthetic time series of monthly TP loads for use as input to a model used later in the program for management purposes.

The exercise clearly demonstrated that as the time scale is increased temporal changes are smoothed (compare Figures 7 and 8) making it easier to describe (and generate) a loading history. In this sense, estimating annual average loads (often done using unit areal loads and transmission coefficients) is straightforward because year-to-year fluctuation is small (between 200 kg/d and 400 kg/d for the Zala River). However, an annual average load estimate was found to be too crude to capture the dynamics properly (Somlyódy and van Straten, 1986). These simulations were inadequate to predict temporal changes in the algal biomass or the annual peak biomass, which is the primary indicator used in eutrophication management. It was found that a monthly load estimate was the minimum needed to adequately model the lake.

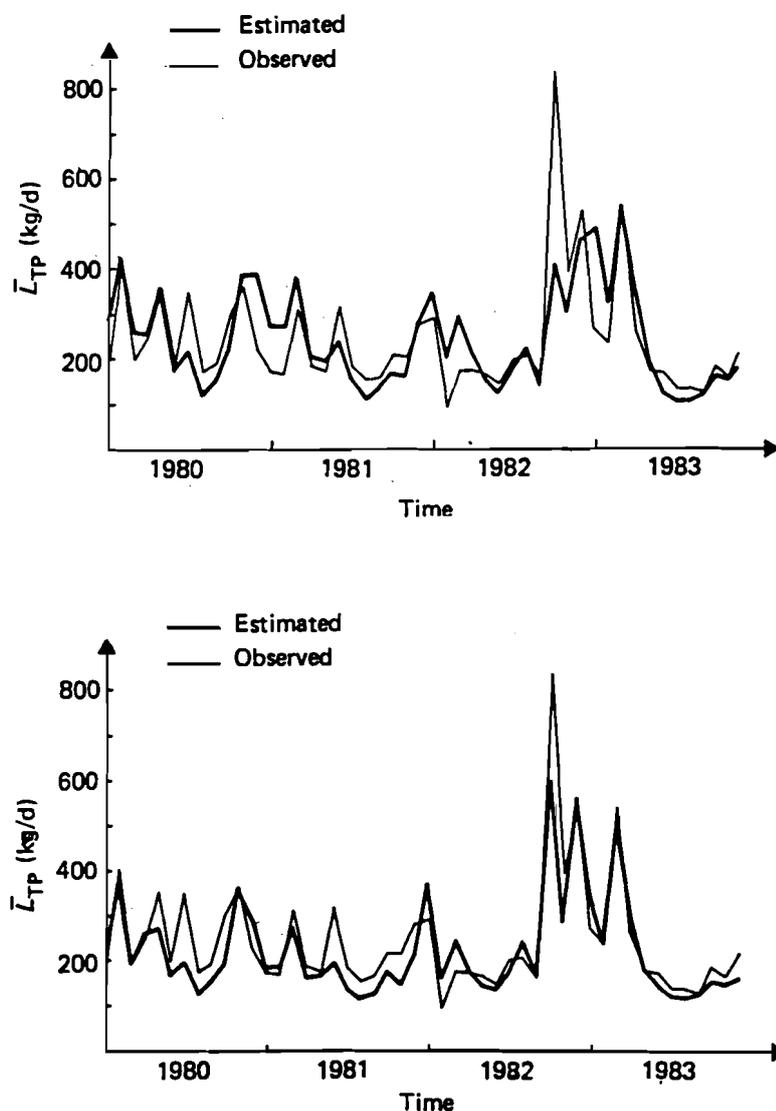


Figure 8
 Validation of the Lake Balaton regression loading model:
 (a) Regression based on Zala River flow
 (b) Regression based on Zala River flow and suspended sediment load

The hydraulic residence time of Keszthely Bay is about 0.25 year, which one would expect to be the desired order of magnitude of the time scale of the load estimate. To test whether a monthly average loading estimate provides sufficient detail to predict water quality, a detailed numerical model of the phosphorus cycle was run under monthly average input loads (and hydrologic and meteorologic data). Negligible differences were obtained in comparison to simulations with daily data. The example clearly demonstrates for the particular example of Lake Balaton and the Zala River the crucial role of the time scale in linking the watershed and water body. It also stresses that the estimation of the time scale should be one of the first steps of all nonpoint source impact analyses, influencing significantly the approach to be selected and subsequent analysis.

As a further example of the importance of estimating time scales, we return to the analysis approach discussed in Section 3.3 and apply it to Lake Balaton. This example considers the response to a fivefold increase in pollutant load as shown in Figure 1. The duration of the storm event, T , is taken as 4 days. The response of a typical reach of the Zala River (hydraulic residence time, τ , taken as 1 day), Keszthely Bay ($\tau = 75$ days), and the entire lake ($\tau = 720$ days). Figure 9 shows the results of the time-scale analysis introduced in Section 3.3. The Zala River shows a large response, with a nearly fourfold concentration increase in response to the fivefold load increase. The response in time follows closely the four-day loading event. The response in Keszthely Bay is much less, but persists much longer relative to the loading event (note that for Keszthely Bay, the loading event lasts only 0.05τ). Finally, for Lake Balaton as a whole, the loading event is barely perceptible because of the great length of the hydraulic residence time relative to the loading event duration.

This time scale analysis implies that for very short residence times, such as the Zala River, a simple solution such as the simple dilution model, Equation 4, would probably suffice for analyzing stream water quality. Similarly, for very long residence times, such as the entire lake, a simple model such as the Vollenweider model, Equation 30, would likely result in an adequate approximation of the annual average TP concentration. However, at intermediate residence times, such as in the Keszthely Bay, there is a dynamic response that would not be captured by the Vollenweider or simple dilution models. For this type of response, a more complicated model would be required to represent adequately the receiving-water response to a nonpoint source event.

The Lake Balaton case study eventually incorporated a full range of models, from the simple mass balance models to a two-dimensional model which linked hydrodynamics and phosphorus dynamics (Somlyódy and van Straten, 1986). The mass balance models were used as a tool to gain basic understanding of the response of the lake and its response to varying internal and external loads. However, numerical models were used to complete detailed analyses and forecasts of eutrophication in the lake and to design and test control strategies.

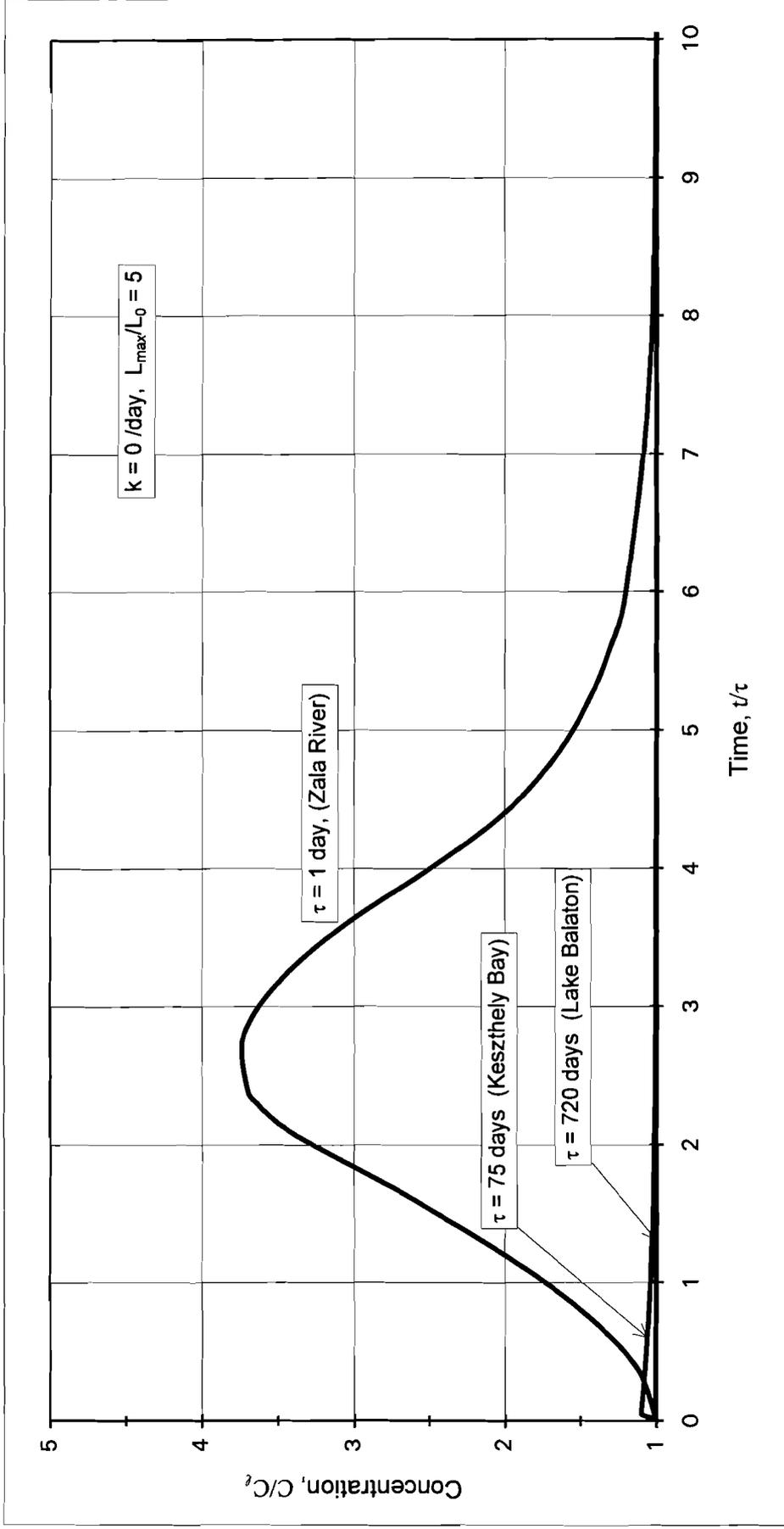


Figure 9
Time scale analysis for Lake Balaton

6. SUMMARY AND CONCLUSIONS

There are numerous types of receiving-water impacts caused by nonpoint source pollution. Agriculture is the single greatest cause of nonpoint source pollution, although urban runoff is also a significant contributor. Other sources are many and varied, and include silviculture, construction, mining, and land disposal.

Point and nonpoint sources are distinguished by very different spatial and temporal scales. Point sources are essentially steady discharges to a single point in the water body while nonpoint sources are typically highly dynamic and may vary widely in their spatial distribution. We recommend water-quality modeling as the most practical means for assessing the effects of point and nonpoint sources, but the variety of nonpoint sources creates special challenges for the modeler. Although all water-quality analyses are based on the simple principle of mass balance, the wide disparities between point and nonpoint sources and between different types of nonpoint sources necessitates a preliminary analysis to identify a modeling approach appropriate for a specific nonpoint source problem. To this end Section 3 of this paper provides an analytical equation for the response of a water body to a transient load for use in assessing the time scale of the receiving water response. This time-scale analysis can be used to determine if a particular problem is amenable to screening calculations using simple analytical water-quality models or if more complicated numerical models are instead required.

The preliminary time-scale identification discussed above is a key part of the following structured procedure we recommend for selecting a model for assessment of a diffuse pollution problem:

1. Identify the study objectives and constraints: What pollutants are to be controlled under what conditions?
2. Determine pollutant interactions: What water-quality processes control the fate and transport of the pollutants of interest?
3. Perform time-scale analysis: Based on the analytical model of water-body response, what are the time scales that govern the problem and what is an appropriate modeling approach (analytical or numerical) for problem assessment?
4. Select a model and perform a final analysis: Choose an appropriate analytical or numerical model and complete the water-quality assessment.

The wide disparity in nonpoint source types and effects makes this systematic procedure, and especially the second and third steps, critical to defining accurately the problem to be assessed and to selecting an appropriate modeling approach.

With respect to model selection, available numerical water-quality models are listed in Section 3 of the paper but our main emphasis is on analytical models for use in screening analyses. We favor analytical models as a first approach to defining and understanding a problem; in many cases these models are adequate for the complete assessment. A variety of such models are described in Section 4 and include:

- | | |
|-------------|--|
| for rivers: | the Probabilistic Dilution Model for stochastic loads of conservative pollutants the Streeter-Phelps model for stream dissolved oxygen simplified models of suspended sediment models of micropollutants considering adsorption and other processes |
| for lakes: | the Vollenweider phosphorus-loading model a toxics-loading model |

a two-layer pollutant-loading model

for estuaries: an estuarine pollutant dispersion model

Many of the models are equally applicable to point and nonpoint source pollution although preference is given to models particularly suited to nonpoint source problems. As illustrated in the worked examples, several of the analytical models are particularly useful for nonpoint source analysis: the Probabilistic Dilution Model for stream water quality; a model of stream dissolved oxygen affected by distributed BOD loads; and the Vollenweider model for lake eutrophication.

The importance of the system time scale is illustrated by the case study of Lake Balaton in Hungary. Cultural eutrophication of this important resource was examined in a multi-year modeling study completed in the early 1980s. The residence time of the entire lake is about two years, but only 75 days for the critical Keszthely Bay. Time-scale analysis shows that the Zala River, the major tributary of the lake, responds to loads on the time scale of days. The response time of the Keszthely Bay is about a month, whereas the whole lake responds only very slowly. Modeling studies conducted as a part of the original project of IIASA confirmed that shorter term dynamics were indeed important to understanding the lake's behavior, dictating, for example, that phosphorus loads be captured on a monthly rather than annual time scale. The time-scale analysis also illustrates that some aspects of the Lake Balaton system can be captured with simple analytical models. This too was confirmed in the original study where mass balance models provided critical understanding of the lake's behavior and its response to internal versus external phosphorus loads.

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U.S. Environmental Protection Agency
401 M Street, S.W.
Washington, DC 20460 USA
Telephone: 1 (202) 382-7056

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