# A SURVEY AND REVIEW OF MODELING FOR TMDL APPLICATION IN TEXAS WATERCOURSES

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# 1. Introduction and background

## *1.1 Definition of modeling*

A model in its broadest sense is a simplified depiction of a natural entity that in some way exhibits its important features while eliminating or suppressing matters of irrelevant detail. In science and engineering, an essential attribute of a model is that it be quantitative, that is, that it yield a numerical value for a feature of the natural entity, as a surrogate for a measurement. A quantitative model can be used to explore cause-and-effect relations and to determine values of physical variables that are too costly or difficult to measure directly.

The above paragraph is general and could apply to any discipline or field of study. In the specific area of water resources, examples of models include an arrangement of laboratory tanks and retorts in which microorganisms behave as they do in a lake, and a scale-model of a river or estuary in which the movement of contaminants can be visualized by dye plumes. Another, and very important, example of a model is a mathematical relation, which might be embodied in a graph or equation, referred to as a "mathematical model." Equations representing flow in a stream as a function of water level, and the longitudinal profile of dissolved oxygen downstream from a sewage outfall are mathematical models. As the equations are extended to include various, often interacting variables of the watercourse, and to accommodate the effect of more external factors, the resulting mathematical model can become extremely complex, until its solution must be carried out on a computer. For this reason, it is now common to refer to specific computer programs, which solve such equations, as "models."

Such complex computer-based mathematical models have grown in accuracy and utility over the past four decades, and have become fundamental tools for the management of water resources systems. This has arisen from two capabilities afforded by mathematical models. First, modeling provides a means of sorting out the complexities of a natural watercourse to identify those factors that are most responsible for some observed conditions in the watercourse, e.g. excessive contamination, flooding potential, and so forth, and therefore to suggest remedial actions. Second, models have a predictive capability, not (necessarily) in the sense of forecasting the future, but in the sense of displaying the behavior of the watercourse under a combination of conditions that has not been represented by direct observation. This allows the assessment of the watercourse under various extremes of conditions, and the evaluation of engineering projects or regulatory strategies, both of which are activities central to water resource management.

It is essential that a model used in water-resources management be sufficiently accurate for its intended purpose. Because a model is a simplified depiction of the natural system, its accuracy is subject to question until proven. The acceptability of a model can only be determined by a confrontation with observation. Therefore, the existence of a model does not obviate the need for data from the watercourse, but in fact imposes additional needs and requirements on the data base. The predictions of the model are directly compared with measurements for two purposes. First, most watercourse models include "free parameters," i.e. variables used in the mathematical formulation for which direct measurements do not exist. These can be estimated by adjusting their values until the resulting model prediction agrees with measurements, a process referred to as model "calibration." Second, the model is operated under the same external conditions as encountered during collection of a set of field data, and the model predictions compared to the field measurements, without any adjustment or "fitting" of the model, to evaluate the performance of the model, a process referred to as model "verification."

### *1.2 TMDL's and modeling*

The maximum rate of injection of a pollutant that a watercourse can accept without violating some level of aquatic health is referred to as its "assimilative capacity" or "loading capacity." This is an old concept underlying much of the early work in sanitary engineering regarding the response of receiving streams to effluent discharges (e.g., Merriman, 1918). With the adoption of water quality standards as a measure of minimum levels of aquatic health, the regulatory problem became establishing the limit on an effluent discharge that under critical conditions did not exceed the assimilative capacity of the receiving water, such a limit becoming the basis for granting of a waste discharge permit.

Modeling was an important part of the process of quantifying the assimilative capacity, and therefore the permitted levels for effluent discharges. For a simple, easily measured parameter of water quality and a single wasteload source, the relationship between wasteload magnitude and receiving water quality could often be quantified on the basis of analysis of field data and a conceptual model of the watercourse. More complicated dependencies necessitated mathematical models. When more than one waste discharge affected the watercourse, then a wasteload "allocation" was carried out, to achieve the best economical and environmental balance among the discharges, taking into account their magnitudes and locations, as well as the variation in response of the watercourse. The complexity of the wasteload allocation process for point source discharges was so great that a suitable mathematical model was indispensable to the task.

The concept of Total Maximum Daily Load (TMDL), as expressed in Section 303(d) of the Clean Water Act, is a generalization and formalization of the concept of wasteload allocation, that includes not only point source discharges but natural sources of the pollutant and so-called nonpoint sources that arise from the watershed (EPA, 1991) or environs of the watercourse. The important aspects of a TMDL include the following:

- TMDL's consider both point and nonpoint sources of a pollutant, which can operate under different conditions and at different times.
- The conditions deemed "critical" or "limiting" for point-source impacts are usually different than those for nonpoint source impacts.
- Nonpoint (or diffuse) sources include pollutants that enter the watercourse during runoff events, through interflow, or through atmospheric deposition.
- The time and external conditions of the maximum *impact* of a nonpoint source load may not coincide with the time and conditions creating the maximum load.
- The impacts, hence the TMDL's, of different pollutants may be controlled by different sets of conditions.
- There generally is not a unique allocation of loadings of pollution, among point and nonpoint sources, that results in the TMDL.

These emphasize the complexity of determining the impacts from a variety of loads of pollutant on a watercourse, and therefore of arriving at a TMDL allocation. Because of this complexity, the use of mathematical models of the watercourse in question is mandatory in the TMDL process.

The complexity of the TMDL determination is rendered even greater because of the geographical and hydrometeorological characteristics of Texas. Texas exhibits a remarkable range of climates. From the arid desert of the Trans-Pecos to the humid Eastern Forest segments of the Sabine basin, there is a sevenfold variation in annual rainfall, the largest of any state save California (and that only by dint of Death Valley). Part of this is due to Texas' situation along the inland-directed trajectory of onshore flow from the Gulf of Mexico, the single most important source of water vapor for the contiguous states. Most of Texas' precipitation is derived from deep convection. Such sources of streamflow as melting of winter snowpack or quasi-stationary stratiform systems do not operate in Texas. Rather, streamflow is derived, directly or indirectly, from convective storms. Most regions of the state exhibit a clear seasonality in this convective activity, though this varies across the state according to the relative importance of airmass thunderstorms, equinoctial frontal systems, or tropical disturbances. The range of hydroclimatology, and its associated seasonal variation, in concert with an equally complex range of soils and geology, results in a variety of topography and vegetation. The model(s) to be used for TMDL determination in Texas must be capable of addressing this variability.

The watercourses to be evaluated are equally varied. A TMDL will be required of almost every example of Texas watercourse. Rivers and streams in Texas generally exhibit substantial seasonality and year-to-year variation in their flow patterns, and are frequently flashy due to the convective source of runoff. Some of these streams are shallow, with riffle-pool morphology, while others are relatively deep and uniform in cross section. There are about 200 reservoirs of more than 5000 ac-ft capacity, almost all of which provide some sort of water supply function, of which some are deep systems that exhibit a seasonal stratification, and others are shallow and are replaced relatively rapidly by streamflow. Some of the Texas rivers have channelized tidal reaches, creating deep systems in which salinity intrusion from the sea is a major feature of their hydrography. Most of the rivers flow into the broad, shallow, productive bays of the Texas coast, which may raise special concerns in the potential impact of contaminants carried in the inflow.

In order to address TMDL determination in Texas, clearly models for a wide array of watercourses must be available to the State, which moreover must be suitable for accurately depicting its variable hydroclimatology and terrain, and their impact on the watercourse. Because the TMDL modeling must incorporate nonpoint source loadings, there needs to be a means of coupling the model to the watershed. This project seeks to summarize the models that are presently available and to offer judgment on their suitability for use in the TMDL process.

### *1.3 Project approach*

The purpose of the present review is to conduct an independent assessment of existing watershed-scale nonpoint-source loading models and instream water-quality models appropriate to Texas environments, with a focus on the relative ease of integration with an ArcView-based geospatial Graphical User Interface (GUI). Specific objectives of the review were:

(1) Compile list of candidate models.

(2) Obtain detailed information about the computer implementation of each model.

(3) Delineate capabilities and limitations of each model, with special emphasis on requirements of Texas watercourses.

(4) Determine (where appropriate) the capability of each model for incorporation into Arc-View environment.

(5) Formulate a "short list" of recommended models for the TNRCC.

This review is not intended to be a comprehensive review of water quality modeling for surface water management. Such a review would far exceed the time and resources available to this project. Rather, this project focuses on the appropriateness for a model to be applied to Texas surface-water environments, and the present report attempts to summarize the review for the non-specialist in modeling.

The list of candidate models was to be as comprehensive as possible, consistent with project resources. The preliminary list was combined from suggestions of the staffs of TNRCC and CRWR. In the early stages of the literature review, other models were added to the list if they seemed appropriate for at least preliminary consideration. In order to limit the scope of the review to ensure its completion within the limited resources for the project, a procedure of successive screening was applied to the list of candidate models. This screening was exclusionary, seeking to eliminate candidate models as early in the screening sequence as possible, so as to minimize the effort invested in review. Therefore, the first level of screening was fairly coarse and focused on crucial attributes that useful models were required to possess, such as being implemented in a transportable, modifiable computer code that is readily available and non-proprietary. The successive screening levels became increasingly detailed and technical as the review proceeded, so that the effort of detailed review was limited to a minority of those on the list of candidates.

In the following chapter, a brief survey of the technical aspects of surface water-quality modeling is presented, by way of introduction to the review, concluding with the screening criteria devised for the review and their application to the candidate models. A single comprehensive surface water model that encompasses all watercourses from the upper watersheds to the sea, with full geographic resolution and the ability to depict the range of flow conditions encompassed within a TMDL is presently not available, and is arguably undesirable. For the near future, at least, it will be necessary to address the different types of watercourses with specific models designed to treat each watercourse (although how one couples such models together is certainly a concern to the present review). Models addressing specific types of watercourses are therefore reviewed individually in the subsequent chapters.

A compilation of the individual model reviews is presented in a separate companion report, Ward and Benaman (1999). Although this companion report could be viewed as an appendix to the present document, it is intended to function as an autonomous reference, listing the watercourse models addressed in this study, in which more detail about the individual models and the specific evaluations according to the multi-tiered screening criteria are documented.

Constraints of time and budget, and the specific focus of this study on formulating a list of candidate TMDL models for Texas, dictated that this review rely upon sources in the technical literature and on discussions with recent users of the models under review. In this study, we did not acquire copies of model software and subject these models to independent evaluation, though such operational tests are recommended for the list of models that emerged as viable candidates for Texas TMDL application. Nor was this intended to be a comprehensive literature review of each model, but rather a review adequate for supporting a decision of including or excluding such models for use in Texas. This was, however, a *critical* review, not one of merely summarizing the features and capabilities of each model. Limitations in model formulation, range of application and software performance were explicitly documented where these might circumscribe or hamper utility of the model in application to the Texas environment.

We note that other comprehensive model reviews exist in the literature, which the reader may wish to consult for additional information on models considered here, or for those not included in this review. These references generally are noncritical, nor are they specific to the Texas situation. In particular, Singh (1995) is a useful summary of the features and operation of nearly 30 catchment models, including (in a companion CD) executables of model code or of model demonstrations (for proprietary models), authored mainly by the principal developers of the models. Shoemaker et al. (1997) present a catalog of aquatic models specifically identified as potentially useful in the TMDL process. Many of these transcend the domain of modeling of physico-chemical parameters in surface watercourses (which is the subject of the present study), addressing toxicological or biological responses, or including risk evaluation protocols.

# 2. Modeling in surface water management

By definition, a TMDL addresses the quality of a surface-water resource (since, in Texas, the target water quality is defined in terms of a surface-water standard or related criteria), so the models under consideration specifically address surface watercourses. In determining the applicability of a mathematical model to a specific watercourse, there are numerous aspects of that model to be considered, ranging from the processes incorporated in or excluded from the model formulation, the numerical treatment of the basic equations, how the model is coupled to the larger hydrometeorological system, what input data are required of the user, and the specifics of the computer program embodying all of the above. These are briefly surveyed here, to establish a basis for the reviews presented in subsequent chapters.

## *2.1 Model formulation*

### *2.1.1 Surface waters as a component of the hydrological cycle*

The movement and behavior of water in a surface watercourse are embedded in a larger system, the overall hydrological cycle, encompassing the circulation of water in its various phases (vapor, liquid and solid) through the atmosphere, ocean, and terrestrial components of the earth. A comprehensive model would of necessity include all of these processes and would therefore be global in extent. From the practical standpoint of managing a reach of a river system in Texas, the relevance of the Asiatic monsoon or calving from the Ross ice sheet would appear remote, so the modeler would justifiably seek to limit the geographical extent of the model to that area which has an immediate effect on the system of concern.

For a river system, one natural terrestrial boundary might appear to be the watershed, that region from which impingent precipitation eventually drains into the river. For a bay or estuary, however, the watershed is not a complete geographical boundary, because the estuary is also subject to marine influences. Thus, the processes governing the circulation of water at the estuary mouth must be addressed. For Texas, this is the Gulf of Mexico, and the comprehensive model would need to include all of the controls on the behavior of the Gulf, namely the inflows of the adjacent rivers, and their watersheds, as well as direct tidal and meteorological forcing, and the indirect effect of exchanges with the Atlantic.

Even, for the simple river system, for which the spatial boundary of the model could be confined to the watershed, it would still be necessary to include the precipitation process itself, which means modeling the cloud forms and the larger meteorological systems in which they are produced, which means modeling the flow of the westerlies over the state, and the influx of marine air from the Gulf of Mexico. The modeling problem clearly is expanding far beyond the boundaries of the river system that motivated the problem, and at this larger scale is manifestly intractable.

The means of confining the model "domain" to a sufficiently limited region so as to be practical is to excise this region from the larger hydrological cycle, and treat it in isolation. The effects of the larger hydrological cycle acting across the surface bounding the model domain now have to be specified, so that this strategy in effect trades one problem for another, but this new problem is generally easier to deal with. These effects of the larger hydrological cycle on the excised region are referred to, appropriately, as boundary conditions.

The watercourse could be addressed as a single entity, but even that might be too demanding, because of the different modeling requirements associated with the varying character of the watercourse. A schematic of a simple hypothetical river system is shown in Fig. 2-1, illustrating how the character of the watercourse changes from the upper watershed, to the river channel, to the reservoir, and finally to the estuary and bay at the mouth of the river. The same strategy applies as before to make the model more tractable, namely to further resect the various types of watercourses and treat each autonomously, as indicated in Fig. 2-2. In modeling each subsection of the watercourse, the transfers of water from the other subsections must be explicitly provided: the runoff from the watershed must be explicitly added to the river channel at the appropriate locations, the flow from the upper river channel must be applied to the river channel in the backwater of the reservoir, the flow from the river basin through the reservoir must be applied as

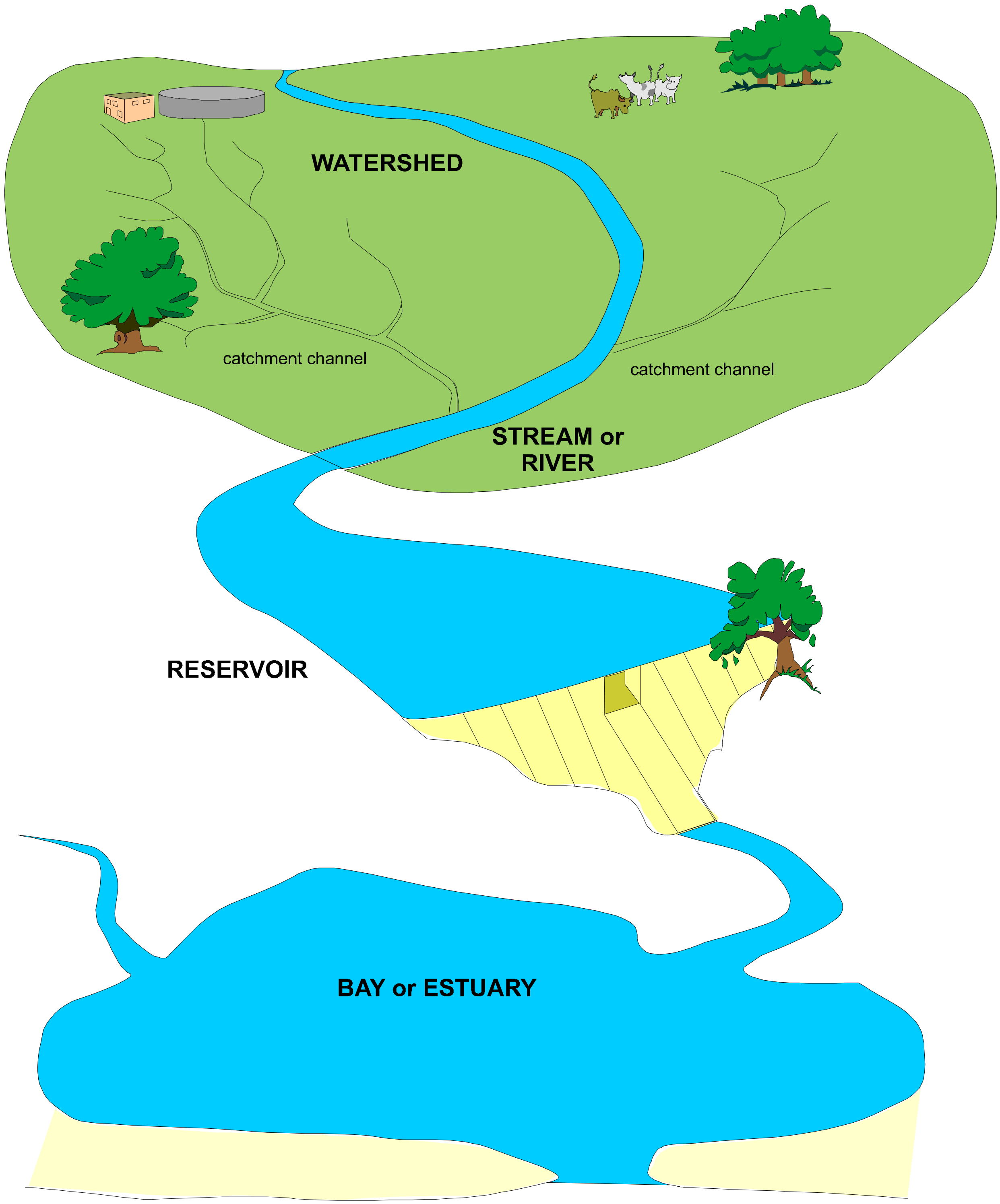


Figure 2-1. Anatomy of surface-water model

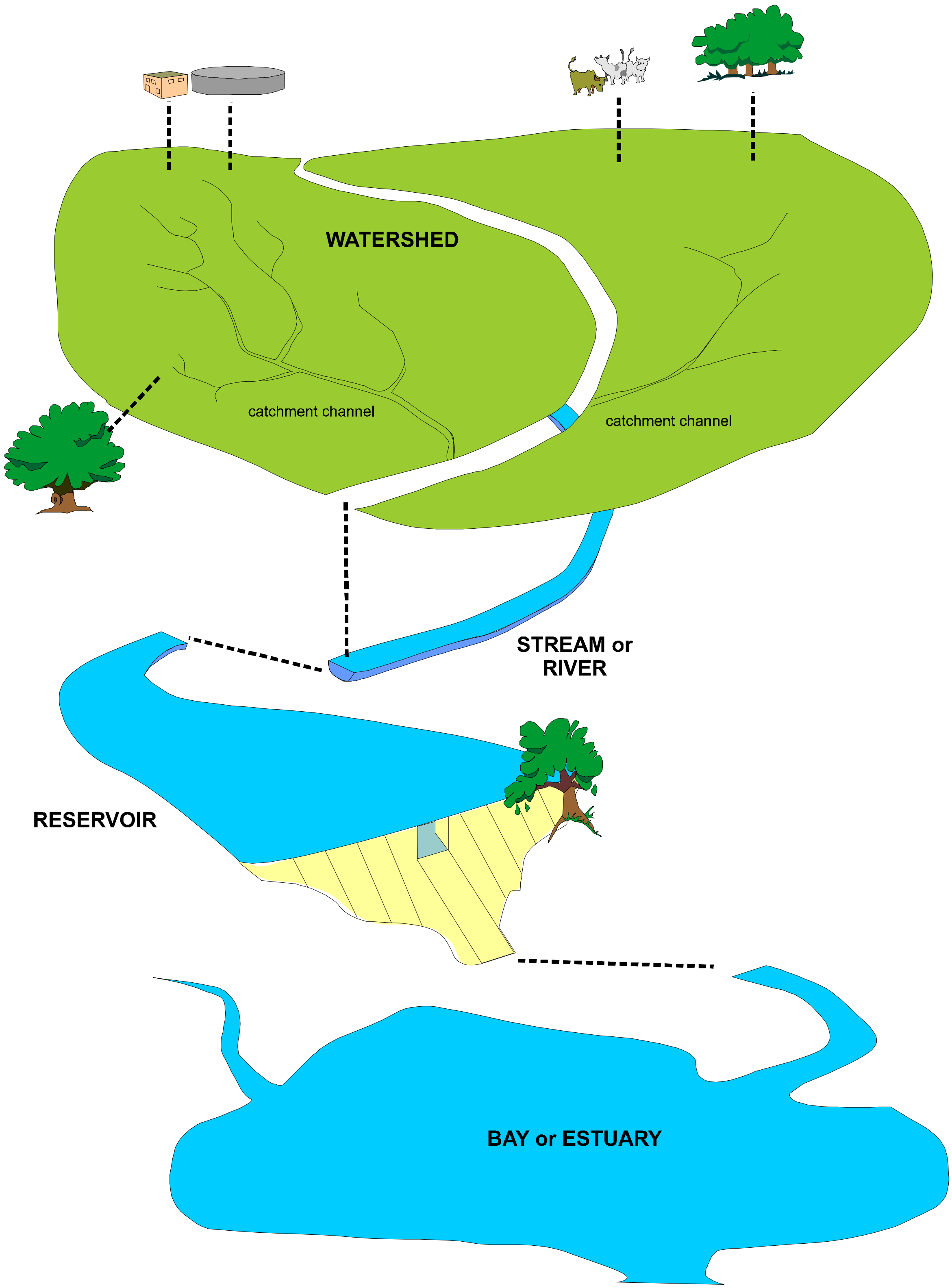


Fig 2-2  Exploded view of watercourse model

an inflow to the estuary. There are other boundary fluxes of water not shown in the schematic, e.g. the transpiration of water from vegetation on the watershed, diversions from the river, waste discharges, evaporation from water surfaces, tides and intrusion of seawater at the mouth of the bay, and so forth. One flux that can be especially important is the flux through the lower boundary, *viz*. infiltration from the land surface and river channel into the subsurface, or the reverse process of interflow into the river bed. This is indicated schematically in Fig. 2-3.

The treatment of different types of watercourses as though they are independent and autonomous is the philosophy followed in the development and application of surface-water modeling almost without exception. One major reason is that computer resources have not been adequate to model an entire drainage system from the edge of the watershed to the sea. Even apart from the demands on computer resources (which becomes less of an obstacle every year), the behavior of water flow in the different types of watercourses, and the differing nature of their management problems still argue for isolated models addressing the characteristics of the specific watercourse of interest.

### *2.1.2 Nature and types of models*

The general definition of a model provided in Chapter 1 admits a variety of devices, including scaled physical models, electrical and mechanical analogs, graphs and diagrams. This review is concerned solely with mathematical models, implemented for solution on a modern digital computer. Even at this, there is a bewildering range of models applicable to natural watercourses.

Probably the most fundamental property of a mathematical model is whether it is an empirical or mechanistic model. An empirical model, also referred to as a statistical model, is a mathematical relation between variables that is designed to fit, in some sense, a series of measurements. A traditional linear regression of streamflow on rainfall is an example. The properties of a statistical model include the following:

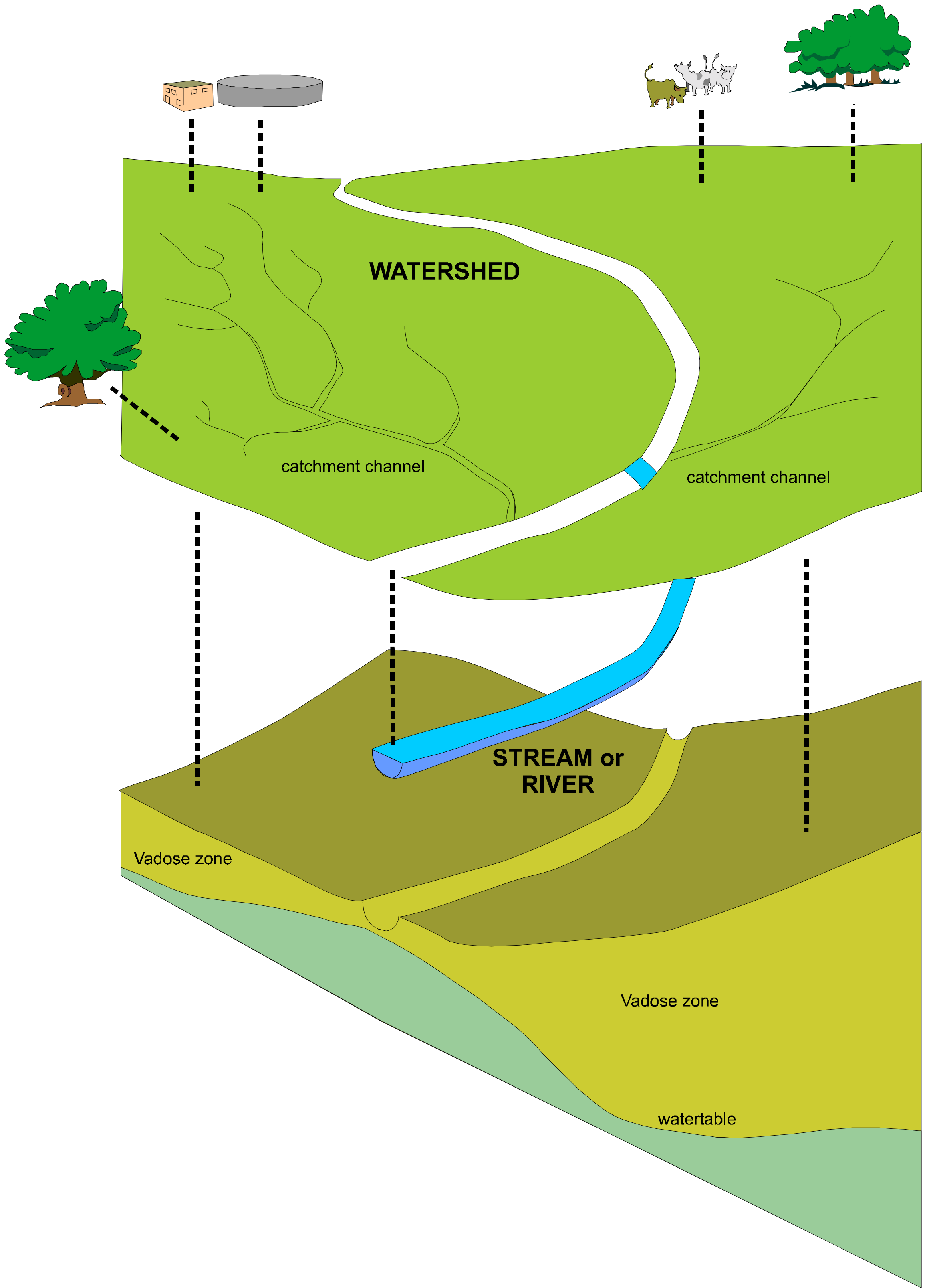


Fig 2-3 Exploded view with subsurface component

- The mathematical relationship is basically arbitrary, being assumed as a compromise between ease of computation and similarity to the trend of measurements.

- The parameters of the relationship (e.g., slope and intercept, for a linear relation) must be established by some curve-fitting procedure to a set of measurements.

- The specific parameters are dependent upon the data used to fit the relation: additional data will alter the relation; moreover, extrapolation beyond the range of data used to fit the relation can be aleatory.

- Any fundamental change in the physical system from which the measurements are obtained will invalidate the relation.

These properties of an empirical model might be considered deficiencies. The last is particularly important. In the example of a linear regression fitted to measurements of streamflow versus rainfall, if the watershed is altered, by urban development for example, the relationship derived would no longer be applicable. New data taken after the alteration and a new fitted relation will be needed.

A mechanistic model, also referred to as a deterministic or process model, is a mathematical statement of a physical law. As an example in watercourse modeling, Newton's second law of motion is the basis of mechanistic models of stream hydraulics. For pollutant distributions in a watercourse, the equation of conservation of mass is the mechanistic model. In principle—though rarely in practice—a mechanistic model avoids the above deficiencies of the empirical model. No assumption of functional form is made, but rather the dependency follows from the model itself (but mathematical solution of the model can be much more difficult). There is no direct dependency upon measurements, and if the physical system is altered, the same alterations can be accommodated in the mechanistic model.

There would appear to be a wide philosophical gulf between these two types of models. The empirical model depicts an association, while the mechanistic model is based upon cause and effect. The empirical model is a fit of an assumed relation to measurements, while the mechanistic model is a theoretical computation from first principles. The empirical model is a mathematical summary of data, while the mechanistic model is the solution to a problem in physics. The fact is that the mathematical models of practical utility in water-resource management are hybrids. A pure mechanistic model does not exist: some degree of empiricism is incorporated into any watercourse model. It is, however, important in characterizing a model to assess how far between these two poles the model lies. It is also the goal of model development to reduce as much as possible the use of empiricism. Generally, the more mechanistic the model, the greater its range of applicability and its usefulness in the management enterprise.

The ways in which empiricism becomes injected into mechanistic models are important in the present review because they provide a means for differentiating among models, for determining which kind of model might be preferable for a specific problem, and for delineating weaknesses of models for the TMDL process. The root source of empiricism is the complexity of the world. A pure mechanistic model reflects that complexity. Rather general mathematical expressions of mechanistic models for the flow of water are given in Table 2-1. (Some terms in these equations are indicated schematically, *viz*. "stress" and "flux", because the mathematical expressions are too messy to warrant being displayed here.) A mechanistic mathematical model, e.g. Table 2-1, is a partial-differential equation in four dimensions: time and the three spatial coordinates. Each of the terms in these equations, similarly, is a function of these four independent variables. Each constituent in the water necessitates a separate equation. Moreover, there may be coupling between the terms of the equations. For example, suspended solids and water temperature affect water density, expressions for which must be included in the momentum equations.

The numerical solution of such equations can become very complicated, and it can be questioned whether the results have any immediate utility in water-resource management, as the analysis of the model results becomes as difficult as analysis of the real world. In order to make these equations more tractable (and their solutions more useful), it is conventional to introduce various simplifications. These simplifications fall into several categories (Ward and Montague, 1996), as follows:

Table 2-1
Mathematical formulae for general mechanistic fluid-flow models

*momentum equation(s):*

$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + v\frac{\partial u}{\partial y} + w\frac{\partial u}{\partial z} = -\frac{1}{\rho}\frac{\partial p}{\partial x} + \frac{1}{\rho}\nabla(stress)_x \tag{1}$$

$$\frac{\partial v}{\partial t} + u\frac{\partial v}{\partial x} + v\frac{\partial v}{\partial y} + w\frac{\partial v}{\partial z} = -\frac{1}{\rho}\frac{\partial p}{\partial y} + \frac{1}{\rho}\nabla(stress)_y \tag{2}$$

$$0 = -\frac{1}{\rho}\frac{\partial p}{\partial z} - g \tag{3}$$

$(x,y,z)$ = position coordinates
$(u,v,w)$ = vector velocity
$p$ = pressure
$\rho$ = density

*continuity:*

$$\frac{\partial u}{\partial x} + \frac{\partial v}{\partial y} + \frac{\partial w}{\partial z} = 0 \tag{4}$$

*conservation of mass:*

$$\frac{\partial c}{\partial t} + u\frac{\partial c}{\partial x} + v\frac{\partial c}{\partial y} + w\frac{\partial c}{\partial z} + \frac{1}{\rho}\nabla(flux) \quad = \sum S_i \tag{5}$$

$c$ = mass concentration of substance
$S_i$ = kinetic source or sink of substance

- *Reduction of dimensionality*. If there is a prominent dimension in the watercourse morphology (and fine-scale variation is not of concern), the dimensionality of the equations may be reduced by averaging over the lesser important dimensions. For a watershed, the areal distribution is most important, so the equations might be averaged in the vertical. Similarly, a broad, shallow bay may be averaged in the vertical and treated with a two-dimensional (horizontal) geometry. A river or stream may be averaged over the cross section and treated as a one-dimensional longitudinal system. Deep channels or density-stratified reservoirs may have to explicitly depict variation in the vertical but can be averaged laterally from bank-to-bank, or perhaps even over their horizontal area.

- *Long-term temporal averaging*. If detailed time variability is not of concern, the model may address variables averaged over a longer period of time, thereby simplifying both the input-data burden and the mathematical solution. The extreme instance of reducing temporal variability is to assume a dynamic equilibrium, i.e. a steady-state, which in fact may obtain for sufficiently long temporal averaging.

- *Simplification of stresses and kinetics*. For many constituents, the internal source/sink terms may be quite complex, and simpler expressions may function nearly as well as the more complex and accurate terms. As an example, in modeling dissolved oxygen, the complex of aerobic bacterial degradation processes can often be written as a single BOD term with simple first-order kinetics. In the hydrodynamic (i.e., momentum) equation, the complex stress terms can often be stripped down to only one or two terms (i.e., stress on horizontal planes) and written as viscous shears.

When space averaging is applied, the resulting equations become considerably simpler, because of the reduced number of dimensions, but there are still terms that apply at the boundaries of the averaged dimension(s). For example, the vertical averaged momentum equation must still have the values of stress at the surface and bottom specified. Also, the nonlinear advective terms ($u\ \partial u/\partial x$ or $v\ \partial c/\partial y$, for example) do not average to zero, but rather produce residual product terms that can be quite large. All of these terms must be parameterized, that is, written as a mathematical function of variables included in the model. For the bottom stress, any of the well-known hydraulic stress terms can be used, such as the Chezy friction term, or Manning's equation. The residual product terms arising from the nonlinear advective terms are referred to as dispersion and frequently written as a diffusive flux. All of these sorts of replacements involve "free" parameters (e.g., Chezy coefficient, Manning's "n", dispersion coefficient), so called because their values are external to the model formulation and must be determined from other information. The precise selection of the parameterizing formulations and the source of the free parameters are features that distinguish one model from another, and must be considered in relative evaluation of a model.

For all of the models considered in this review, the need for generality in watercourse geometry and the complexity of the transport and kinetic terms require that the mathematics be solved by numerical methods. The modeled system must be discretized in space and in time, and the corresponding derivatives replaced by numerical approximations. The modeler is usually given some latitude in how the watercourse is discretized, which is a compromise between the increased accuracy obtained by a greater number of evaluation points, and the limits of computer resources: time-space resolution versus efficiency of operation. There is also a vast array of options for the numerical solution method itself. However, this is rarely at the disposal of the model user, but instead is part of the model design and built into the structure of the computer program. Although one might hope that the developers of a model have selected a numerical method that satisfies such minimal requirements as accuracy and stability, this may not be the case, so the numerical behavior of a model may be one consideration affecting its selection.

In summary, watercourse models may be differentiated according to:

- spatial integration, i.e. spatial dimensionality
- time resolution (steady state, slowly varying, pulse or short-term variation)
- form of hydrodynamic equations, including parameterized terms
- constituents represented, their coupling and kinetics
- spatial discretization convention
- numerical solution method

The various combinations of these properties implies a large potential number of different models.

### *2.1.3 Model boundaries*

When a model considers the watercourse to be separated from the hydrological cycle, the interaction of the other elements with the modeled watercourse has to be specified as boundary conditions. For a watershed model, these include precipitation on the surface. For a river model, these include river flow at the upstream limit of the modeled reach, runoff into the river channel across the banks (and carried by tributaries, if these are not explicitly included in the model network), and exchange with subsurface waters through the stream bed and banks. For an estuary model, these include tidal exchange with the sea and gravitational intrusion of seawater. Information for these boundary conditions come from three possible sources:

(1) coupling or inputs from other models
(2) observations, as a time series of measured data
(3) direct specification by the modeler

The last is frequently employed in "scenario runs" in which key conditions, usually idealized, drive the model. Examples are design floods and the 7Q2 low-flow regime.

The boundary conditions must be consistent with the spatial-temporal resolution of the model. A steady state model, for instance, requires steady boundary conditions. The boundary conditions can be applied only at the computational elements of the model, so have to be aggregated—or, as the case may be, disaggregated—to be consistent with the numerical discretization of the model.

For models which depict variation in time, the starting configuration must be specified, referred to as the "initial conditions" (in fact, another type of boundary condition). This requires that a value be specified for every variable in the model at every point in the space domain. The same three basic sources of information listed above for boundary conditions also apply to initial conditions. For "scenario" runs, the initial conditions may be some arbitrary but realistic distribution of values. For model validation runs, these conditions have to be as realistic as data permit, and may represent a source of error in the model prediction.

For water-quality models, which are the primary focus of this review, an important boundary condition is the introduction of contaminant loads carried by fluid flow into the model watercourse. For watersheds, these loads can physically originate by detachment or leaching from the terrain. In many cases, they are specified as a mathematical source function at each point in the model domain, and function therefore as an internal source function. For a river or lake, they may be introduced by lateral waste streams, which, being physically too small to warrant resolution in the model network, are input as a boundary condition.

Boundary and initial conditions are sometimes overlooked in model application as "matters of detail." They are, in fact, half the modeling problem, and the capabilities (or limitations) of a model for accommodating boundary conditions of various types are an important feature of that model.

### *2.1.4 Model compartments*

It is useful to separate various components of a model and consider them individually, as a convenient means of both delineating model requirements for a TMDL and differentiating among various types of models. In the present context, we consider the following conceptual components, or "compartments," of a watercourse model:

- hydrodynamics
- transport
- source/sink processes
- sediment processes

Our discussion of these compartments in this section is independent of whether the model is statistical or deterministic. However, for specificity, the mechanistic model of Table 2-1 will be used as reference.

The hydrodynamic compartment of a model determines the current velocity (with three-dimensional components $u,v,w$) together with other related factors, notably pressure, density, flow, and/or the distribution of level of the water surface $h(x,y)$. In Table 2-1, this is the part of the model represented by the momentum equation and the continuity equation, equations (1) - (4). Unless fairly restrictive spatial integration and time-averaging are imposed, the hydrodynamic model is a difficult and involved computation. For large watercourses in which the spatial variation of water properties must be explicitly determined, such as deep lakes or coastal bays, the hydrodynamic compartment may command the bulk of the modeling effort.

This is ironical, because in itself, the hydrodynamic compartment has no intrinsic interest to a manager addressing a water-quality problem. It is important because the distribution of currents can be a prime determinant of spatial variation in water quality. This operates through the transport compartment of the model. In Table 2-1, this is the collection of terms on the left-hand side of the mass-conservation equation (5). The currents are only part of the transport, the remainder being the mysterious "flux" term in (5), which includes turbulent diffusion and mass "dispersion." A transport model is in effect the mass conservation equation but with no explicit specification of the source/sink terms on the right-hand side.

These source/sink terms are regarded as a separate compartment, sometimes referred to generically (and imprecisely) as "kinetic" terms. They are specific to the particular constituent under consideration. For a conservative parameter, these terms will be zero, and the concentration of that parameter will be governed entirely by transport. Most water-quality parameters of interest in TMDL determinations are nonconservative, and the specific formulations of the $S_i$ terms in (5) depict the various chemical, physical and biological processes to which that parameter is subjected. The relative size of the source/sink compartment in comparison to the transport compartment is also an indicator to the sensitivity of the model and the relative allocation of effort needed for each of the compartments. For a constituent that is highly reactive, its source/sink terms will account for more of its variation in concentration. More effort is needed in the formulation and parameterization of the kinetic terms, while transport may be treated rather simplistically. Dissolved oxygen can usually be modeled by such an expedient. In contrast, a waterborne parameter subject to small kinetic processes will behave quasi-conservatively, and its in-stream concentrations will depend more on transport, which will have to be modeled with a higher level of accuracy. It is important to note that the relative importance of transport and kinetics depends upon the magnitude and time variation of the currents. Under steady low flow conditions, the kinetic terms may predominate, but under dynamic storm hydrographs, transport may become dominant.

It was noted above that each water quality parameter requires a separate mass-balance equation, because the sources, sinks and kinetics differ. This is further complicated if the kinetics of one water-quality parameter depend upon the concentration of another. The classical example is dissolved oxygen, whose kinetic sink due to organic consumption is often modeled in terms of BOD, which has sources of wasteload discharges and a degradative sink that follows approximately a first-order decay. Other examples include the conversion of organic nitrogen to ammonia, and ammonia to nitrate, so that each of these feeds into the source term for the next. (The rate of nitrogen oxidation may also appear as a sink term in the DO mass-balance equation.) An important characteristic of a model is its ability to accommodate such feedforward and feedback relations between the water quality parameters.

The last compartment, sediment processes, refers to a particular constituent, namely fine-grain sediments measured in a water sample as suspended solids, whose mass balance would strictly be comprised of a transport and a source/sink compartment. We list it as a separate compartment because

(1) modeling of sedimentary processes is extraordinarily difficult,
(2) many water quality models have no particular provision for treating sediment,
(3) watershed loads produced by storm runoff in the flashy hydrometeorology of Texas usually entail large concentrations and effluxes of sediment,

(4) many parameters of potential concern in the TMDL process are either closely associated with sediments or sorb to sediments, so that suspended sediment becomes the "carrier" for that parameter.

The review of sediment transport mechanics is too involved to explore here. (A recent survey is given by Julien, 1998.) From the standpoint of characteristics of water-quality models, the physical situation can be summarized as follows. For practical purposes, the presence of sediment in the water is due to current velocity. Thus currents not only affect the transport of sediment by entering the advection terms of (5) but also the source/sink terms. We differentiate between sources of sediment that are external to a watercourse and those that are internal. External sources are carried into the watercourse either from upstream or from peripheral drainage areas. Inflow is the usual agency for this, but bank caving and wasteloads are mass loads of sediment with little or no associated flow. Internal sources (which can also be sinks) include sediment re-mobilized from the banks and bed by moving water, at a rate limited by the "capacity" of the flowing water to transport sediment, dependent in turn upon the properties of the sediment particles, the velocity of the water, and the sediment already carried in the water. The source/sink terms of Table 2-1 apply to erosion and deposition on the bed of the watercourse.

Sediment transport can be source-limited or capacity-limited. In principle, the concentration of sediment can be modeled by a version of (5) in which the sources and sinks $\sum S_i$ include the influxes of sediment from lateral sources around the watercourse, and into and out of the water column. The latter would be sediment remobilized from the bottom due to water movement across the bottom and sediment settling out of the water column by gravity and turbulence. This works satisfactorily if the sediment concentrations are dilute and made up of silt- and clay-sized particles, a situation usually source-limited. If the flow is sediment-laden, then the interactions between water velocity, particle size and physical properties, turbulent entrainment, and gravitational settling become too complex to be conveniently depicted by simple mathematical expressions. In this case, usually capacity-limited, resort may be made to quasi-empirical equations relating bulk sediment transport to hydraulic properties of the flow, such as the Einstein, Simons-Li or Bagnold equations (Bagnold, 1966, Julien, 1998).

This is the fundamental difference between sediment transport in the watershed and sediment transport in a receiving stream. In the watershed environment, sediment transport is generally capacity-limited, governed by the detachment mechanics and the ability of the overland flow to remove the detached sediment. In a receiving stream, there is a greater volume of flow, the suspended sediment concentrations are governed largely by the lateral influx from watershed drainage, and are characteristically source-limited. A lake can be either: the system can be capacity-limited because of the low current velocities, but only if the sources of sediment from the inflowing tributaries are large enough.

In assessing the capabilities of various models for use in a TMDL determination, the extent to which each compartment is represented in the model helps establish the type of TMDL problem the model is capable of addressing. Some models have hydrodynamic capability but no transport, others have transport capability but no hydrodynamic (in which case the current velocities must be input by the user). Some have rudimentary options of kinetic specifications, while others have a wide range of parameters at the disposal of the user with quite complex source/sink terms. Any of these may or may not include the capability for modeling sediment transport.

## *2.2 Model Implementation*

### *2.2.1 Programming considerations*

The models considered in this review are not only mathematical formulations, but are also numerical solutions of these formulations implemented in an operational computer model. As a part of the model evaluation, we must also consider the features of the computer code. These include how the code is structured for routine use, properties of the code, and how amenable it is for alteration, which might be required to render it more applicable to Texas watercourses.

Of foremost importance is the integrity of the code, in the sense of correct operation. Errors occur in the coding of complex models. The obvious ones are caught in model development because they produce bizarre answers thereby calling attention to themselves. More subtle errors may not emerge until the model has been applied to several systems, and systematic departures from measurements or from "expected behavior" lead a modeler to examine the code itself as a source of the aberration. There is no absolute assurance that any computer model is bug-free, but as its history of use accumulates with different users on different platforms applying the model to different watercourses, the level of confidence increases that the model is really behaving as it should.

The intended users of models for TMDL determination in Texas are not expected to be model developers, and the TMDL process should not include computer programming. Therefore, the extent to which the model code has been designed for users without requiring intimate knowledge of the code is an important feature for this review. This can be difficult to assess, because one must distinguish between the complexity of a problem and the complexity of a model. Natural watercourses are complex, and the complexity of water-quality responses of these watercourses necessitate the use of models. There is, therefore, a limit to which a model can simplify the problem set-up without sacrificing accuracy. On the other hand, the structure of the computer code should not compound complexity. As is the case with model integrity, a trustworthy guide to the ease of use of a model is the massed experience of past users.

Of special interest in implementation of models, especially watershed models, is the use of Geographical Information Systems (GIS). These systems offer the capability to greatly simplify the burden on the user for preparation of massive input files, and furthermore provide a mechanism for the graphic display of model results. Their potential importance in TMDL modeling is so great that they are addressed separately in the following section and in Chapter 7.

Part of the code function and the user experience include special requirements of the model for the platform. For TMDL determinations in Texas, operation on high-end PC platforms is a requirement. Most older main-frame codes, FORTRAN-based, have made the transition to the PC environment. Some of the newer model codes, however, may use language compilers of limited distribution or require minicomputer workstation environments, especially for special-purpose file manipulations. Such requirements could severely limit the utility of the model code in supporting TMDL procedures in Texas, and therefore must be considered in the model evaluation.

The structure of model inputs, in addition to user convenience, can facilitate or hamper application of the model to situations characteristic of Texas watercourses. The range of variation of input parameters (or whether a particular parameter variable is even included in the input files), how these parameters are allowed to change over time, and whether functional dependencies upon the watercourse environment are allowed, are all important considerations. We must recognize that many models may have been designed for application in other geographical areas, and therefore may not include features of importance to Texas, but their set-up and operation may be sufficiently general that with minor modifications they might be usefully adapted to Texas settings. The ease with which this can be done therefore becomes a part of the review, e.g., whether the source code for the model is available, what code language(s) are used, and the difficulties that might be encountered in making modifications to the code.

### *2.2.2 Geographical Information Systems*

Geographical Information Systems (GIS) have gained considerable attention and use in the past decade in environmental modeling. Although originally utilized mostly by planning and development entities, the program's database management tools, along with its strong visualization techniques, have revolutionized environmental model development. Many environmental modeling projects have utilized GIS since the software's inception in the 1980's. A brief list of some projects utilizing GIS within the last five years is as follows:

- The implementation of EPA's Hydrologic Simulation Program – Fortran (HSPF) and GIS to model water quality and quantity on the Grand River Watershed in Ontario (Al-Abed and Whiteley, 1995).

- The application of EPA's Water Quality Analysis and Simulation Program (WASP) with a GIS connection to model dissolved oxygen in the Houston Ship Channel (Benaman *et al.*, 1996).
- A watershed modeling effort using the Precipitation-Runoff Modeling System (PRMS) on the Williamette River Basin performed by the United States Geological Survey (USGS) in Oregon (Laenen and Risley, 1995).
- Non-point source loading assessment of the San Antonio- Nueces coastal basin using GIS (Saunders and Maidment, 1996).
- A connection of the water quality model, WASP, and GIS to simulate biochemical oxygen demand in the Buffalo River (DePinto et al., 1994).

This list is just a very small subset of the number of projects completed or underway throughout the nation which take advantage of the capabilities of GIS within environmental management. Most state agencies now have GIS as an accepted department within their government structure and utilize the technology for both planning and environmental management studies. In addition, GIS departments within environmental consulting firms have grown in number and GIS capabilities have become a strong marketing tool for these companies. Academic research in GIS with environmental modeling has also expanded, with many environmentally-based departments developing curricula and programs that deal specifically with the use of GIS in their specialty. The popularity and increased use of GIS are exemplified in the increased number of presentations and publications on the subject. Many technical conferences on environmental management and analysis have dedicated entire technical sessions to the subject, while conversely, GIS based conferences include environmental modeling on their agenda. The three primary areas that GIS is applied in environmental analysis are database management, model development, and output visualization.

The ability of GIS to spatially display large data sets lends itself well to data management and analysis of natural systems. For the most part, sampling surveys conducted in the environment have a spatial component to them, e.g., bathymetry transects located along a river bed, chemical concentrations measured throughout the sediment of a lake, or land use mapped over a particular watershed. All of these data sets are best viewed in the context of their study area. GIS allows engineers and scientists to obtain a quick look at the data taken in a study area and query the data for analysis. For example, a data set within GIS may contain sampling locations of dissolved oxygen in a Texas lake or estuary. The attributes of these point locations may be date and time of sampling, water temperature, dissolved oxygen concentration, and sampling depth. The software would display the shoreline of the water body, along with a point at each sampling location, indicating the sampling density and dissolved oxygen conditions in the water. Another example would be land use, e.g., nonpoint source loading from a watershed to a receiving water body. GIS can assist the user in analyzing the current and future land use of a basin while providing insight into potential problem areas. For example, if major development is planned in a primarily agricultural river basin, the use of GIS would assist the planners and engineers in determining the impact of that development on the surrounding environment. Recent projects within Texas have utilized GIS for this purpose, including nonpoint source loading assessments of the San Antonio-Nueces coastal watershed, the Corpus Christi Bay system (extending from Baffin to Mesquite Bay), the Houston Ship Channel Basin, Dickinson Bayou, and the Trinity River Basin.

Water quality models are highly dependent on spatial information for input parameters. Data drives a model and, typically, data used in environmental models are spatially variable. Initial conditions, boundary conditions, and modeling parameters must be provided for each computational element of the model to solve the process equations. GIS provides a powerful tool for the development of these types of inputs, in addition to assisting in developing the model segmentation. GIS can assist the engineer in analyzing spatial characteristics and developing proper segmentation through data visualization and analysis. In addition to model segmentation, GIS has been used in the development of input parameters. For example, a recent modeling effort in Lavaca Bay used GIS to establish the model segmentation and determine various bed property parameters and initial mercury concentrations in each model segment through the spatial interpolation of data. It was estimated that the use of GIS in this modeling effort reduced the amount of time necessary for model development by at least 50% (Benaman and Mathews, 1997).

Besides database management and model development, a key use of GIS has been in visualization of model output and environmental data. The primary reason GIS has gained such popular use in the past decade has been because of its ease of use in displaying environmental conditions. Land use is color coded to represent different categories, sampling points are modified to signify their chemical concentrations, and model segmentation is color ramped to display output results. All of these display capabilities aid in the decision making process for TMDL calculations.

Along with the use of GIS comes the ever-pressing question of user interfaces within GIS to develop and execute these water quality models. There are three primary forms of connection between GIS and an environmental model. The first is a loose, or *ad hoc*, connection where the user formats the data for GIS, determines the required input, creates the input files manually, and runs the model outside of GIS. The model output is then manually reformatted for GIS and imported into the software for visualization. The second type is partial integration in which GIS plays more of a role in the input file generation and execution. However, in partial integration, the environmental model still stands outside of GIS and is accessible by the user. Full integration, the third level of connection, means that the model is fully integrated within the GIS program and the fate and transport equations are solved using GIS software. At this level, accessibility of the model by the user is limited and sometimes not possible (Tim and Jolly, 1992).

For all of these levels of GIS connection, the nature of the model development needs to be incorporated within the user interface. Figure 2-2 depicts an exploded view of the watercourse necessary for the implementation of modeling. For a GIS connection, this view is taken one step further , in which the environment is broken down into the basic components and characteristics as they would be interpreted in GIS. For example, the river in Figure 2-2 may be a box or segment in the model, but in GIS this box is a polygon with its attributes being the data necessary for model input, such as water depth, cross-sectional area, slope, flow, and chemical concentration. Once this structure is established, GIS can successfully communicate with the user and model to provide a powerful user interface. Programs can be written within the interface to read the input from available data, create the input files, execute the model, and display the output. The concept of user interfaces is addressed further within the discussion of the recommended models. In addition, Chapter 7 reviews some current user interfaces and their potential application to Texas TMDL's.

### *2.3 Role of data in modeling*

Measurements and observations are intrinsic and indispensable to the modeling process. Field data are needed to define the physiographic and morphological features of the watercourse. Many of the boundary conditions are derived from data, either directly (for example, in driving a river model with a time history of measured daily flows at the upstream boundary of the model reach), or indirectly (driving the model with a flow of determined by statistical analysis of a data record). The sources of contamination, either from mobilization from watershed terrain, or from discharges of effluent, are determined from measurements.

The processes of calibration and verification in model application were noted earlier. In both cases, a direct comparison of model computation and field measurement is made, but the purposes are different. Calibration is used to establish values of the free variables in the model formulation. Verification, on the other hand, is a direct test of the ability of the model to reproduce observations. (Terminology is not consistent in the field. "Validation" is often used as a synonym for "verification", and also as a collective description of the complete process of calibration and verification. One also encounters used of the term "verification" to mean calibration, a terminology that can be traced back to the construction of physical hydraulic models, which were "verified" by moving cobbles around on the model bed, or bending and snipping tin strips embedded in the model, to force the model water levels to agree with a set of data.)

Both types of exercise require a complete set of input data, to establish boundary conditions and loadings, to ensure that the model execution conforms to the external conditions encountered during the period of data collection. The points—in both space and time—at which data are collected to compare to model predictions must also be chosen with great care. There are many aspects of model verification, which can become a complicated process in which model sensitivity is analyzed and the model tested over a range of conditions. For present purposes, what is important is the extent to which a candidate model have been tested against field data for watercourses similar to those in Texas and over a similar range of hydrometeorological conditions.

### *2.4 Scope of the review*

In order to contain the scope of the present project and to focus on models that would directly contribute to the TMDL determination process in Texas, we limit our consideration to mechanistic models that determine the distribution of waterborne constituents in the watercourse. This implies a capability for computing both transport processes and kinetics. In most cases, the velocity components for the transport terms have to established by a separate hydrodynamic or hydraulic model, which might be a component of the model, or may require an independent model. We further confine our attention to models designed to represent general watercourse behavior, avoiding special-purpose models, such as plume/jet models or mixing zone models.

For application in a TMDL determination, watershed hydrology must be linked with receiving watercourses. For Texas, the most important such watercourses are streams and reservoirs. Generally, more spatial detail is necessary in these receiving watercourses than in the watersheds, because it is in the receiving watercourse that the target criteria must be met. Some watershed models include a receiving watercourse as part of the modeled system, which means that the properties of the receiving-water component must be evaluated as well. Most watershed models produce loadings at the downstream limit of the watershed (presumably, the receiving water boundary), so the properties of this model output must be considered, mainly the time-space resolution compared to that needed in the stream or reservoir model. From an operational point of view, the formatting and I/O options may be important in facilitating (or impeding) coupling of the watershed model output to receiving water models.

While the stream or reservoir is the more important receiving watercourse, for Texas TMDL problems there are three others that may be of significance. First is the estuary, while operationally has the same requirements of I/O coupling as a river or stream, so requires no additional consideration. The other two are components of the subsurface part of the hydrological cycle (see Fig. 2-3). The unsaturated or vadose zone may be important to some TMDL's, especially the uppermost root zone, in nutrient kinetics. Also, there may be situations in which the receiving watercourse receives flow from the watershed both from surface pathways, and from subsurface through interflow. To model the subsurface pathways will require coupling of the watershed with the involved portion of the subsurface, certainly the vadose zone, and perhaps a saturated zone as well. While such occurrences are expected to be rare in Texas, it may be useful to identify models which include a subsurface capability.

With the explosion in computer resources over the past two decades, many models have been promulgated that incorporate waterborne constituents into a larger computational problem, such as probabilistic risk assessment, monte carlo simulations, economic and cost/benefit analyses, or optimization procedures. An important category of these wider-strategy models is biological models, those that simulate the response of organisms, including major components of the ecosystem. Some of these models offer great promise in attacking complicated problems of water-resource management. But, in the present context, a TMDL is defined with respect to target concentrations of waterborne constituents, and the objective of TMDL modeling is to determine what range of loading configurations under various hydrometeorological conditions will ensure that these target concentrations are not violated. While there is a larger ecological context, we assume this to be external to the modeling task *per se*, but bound up in the establishment of target concentrations and critical conditions.

As noted earlier, the approach to the review of models was to develop a screening procedure to which candidate models were subjected. The screening procedure was three-tiered:

(1) representative of Texas watercourses
exists as operational nonproprietary program for PC
sufficient history and currency of application

deterministic (mechanistic) in philosophy

(2) suitable physical formulation

suitable numerical formulation

suitable coding features and hardware requirements

(3) Specific characteristics of watercourse modeled, for each of:

watersheds/basins, rivers/streams, reservoirs/lakes, bays estuaries:

adaptable to Texas watercourse properties

demonstrated applicability and acceptance for systems typical of Texas

capable of GIS implementation or coupling to GIS-based watershed model

Description and reviews of the individual models are given in Ward and Benaman (1999), including detailed screening criteria. Screening Level 1 is exclusionary, listing key requirements that must be met by a candidate model. Failure on any one of these criteria (more detailed explanation of which may be found in Ward and Benaman, 1999) is sufficient to exclude the model from further consideration. Application of the Level-1 criteria enabled us to pare down the list of candidates with a minimum of literature review, thereby conserving the resources of the project to be applied to the models which offer a real possibility of usefulness.

Screening Level 2 was also exclusionary, but required more effort in review of the model documentation and literature references. The term "suitable" is actually given fairly precise definition, as can be seen by consulting Ward and Benaman (1999). Emphasis of the Level-2 criteria is on the computer implementation of the model, nature of the program code, user-oriented operation, and related matters. From the standpoint of scientific evaluation of a model, these matters are largely irrelevant. However, in view of the fact that TMDL determinations will be made for a variety of watercourses by many workers of varied background in model applications, these programming aspects become important in ensuring the success of such TMDL projects.

Level-3 attempted to evaluate the potential of a watershed or stream model for application to the Texas environment. Lake/reservoir and bay/estuary models are considered to be even more specialized models, and their evaluation criteria are much more specific to the type of watercourse for which the candidate model is supposed to be applied. The criteria therefore differ between a basin model, on the one hand, and a reservoir or estuary model, on the other. Emphasis is given to the history of application and general acceptance of a model. This biases the evaluation in favor of models that have been developed for some time and have a long history of application. We note that often the better model may be the newer model, in taking advantage of modern software capabilities, of using more sophisticated numerical methods, or in improving the technical basis of a model by repairing weaknesses in older models that have been made evident in their application.

In the concluding chapter of this report (as well as in Ward and Benaman, 1999), we have identified those models that appear to us to offer substantial improvements on their predecessors, but have not had the benefit of extensive application experience. The TNRCC may be well advised to subject these models to a more extensive evaluation, which could not be undertaken within the scope of this review. In Chapter 7, we also review BASINS, in whose promulgation for TMDL determinations EPA has invested considerable effort. BASINS is not so much a model as a model shell, whose component models are addressed in the appropriate subsequent chapters as well as the model reviews of Ward and Benaman (1999).

Table 2-2 lists all of the candidate models evaluated in this review, by program name and program source, with the highest level of screening each model received. That is, those models listed with Level-1 screening were rejected from further consideration at that level. The reason for this rejection, along with other relevant information about the model, is indicated in the "comments" column. The models are grouped by watercourse: a model may appear in more than one place in the table if it has an advertised capability to treat more than one type of watercourse. A total of 47 models were reviewed in this project, of which 13 passed to Level-3 or Level-4. (Another dozen or so models were sought, including DECAL, P8-UCM, SITEMAP, STORM, and TPM, but no information could be found. Several more, such as MIKE-21 and HEM3D, are closely associated with models listed in Table 2-2, and included in their reviews.) More details about the models that survived Level-1 screening are given in the following chapters, and complete descriptions are given in Ward and Benaman (1999).

Table 2-2

Summary of model assessments
(Agency abbreviations at end of table)

| *model* | *source of model* | *screened to Level:* | *comments* |
|---|---|---|---|
| | | watershed models | |
| ADAPT | OSU | 1 | research model, limited history |
| AGNPS | ARS | 1 | insufficient currency |
| ANSWERS | NCSU | 3 | event model, dated code |
| ANSWERS-2000 | VTI | 1 | under development |
| CLAWS | LLL | 1 | under development, poorly documented |
| CREAMS | ARS | 2 | agricultural fields only, not adaptable to watersheds |
| DESERT | IIASA | 1 | under development, poorly documented |
| DR3M | USGS | 1 | urban runoff, limited history |
| EPIC | ARS | 2 | agricultural fields, not adaptable to watersheds |
| GLEAMS | ARS | 2 | farm-scale catchment, not adaptable to watersheds, but may have limited utility in manure or litter application BMP evaluation |
| GWLF | n/a | 1 | inadequate documentation, limited history |
| HSPF | USGS/CEAM | 3 | process models poorly documented, difficult to apply |
| IIHR | IIHR | 1 | no longer supported, limited history |
| MIKE-SHE | DHI | 1 | watersheds, drainage network, vadose zone & aquifers, proprietary |
| MODFLOW | USGS | 1 | vadose zone & aquifers, not adaptable to watersheds |
| PRMS | USGS | 3 | input demands less than HSPF, limited water-quality capability, GUI input management system under development |

(continued)

Table 2-2
(continued)

| *model* | *source of model* | *screened to Level:* | *comments* |
|---|---|---|---|
| | | watershed models (continued) | |
| RUSLE | ARS | 1 | limited applicability, agricultural fields, statistical model, sediment load only |
| SLAMM | USGS/W | 1 | urban watersheds only, inappropriate for Texas, limited history |
| SPUR | ARS | 1 | under development |
| SWAT | ARS | 3 | includes lakes & vadose zone, lumped formulation, statistical process models |
| SWIM | ICIR | 1 | under development |
| SWMM | CEAM | 3 | emphasis on urban catchments |
| SWRRB | ARS | 1 | agricultural fields only, replaced by SWAT |
| WAM | SWET | 1 | proprietary |
| WASH123D | WES | 1 | includes vadose zone & aquifers, rivers & streams, difficult to use, insufficient history of application, inadequate technical acceptance |
| WEPP | NSEL | 2 | agricultural fields only, not readily applicable to Texas watersheds, insufficient application |
| WMS | SSG | 1 | proprietary, interface "shell" only |
| | | stream and river models | |
| CE-QUAL-ICM | WES | 1 | insufficient application |
| CE-QUAL-RIV1 | WES | 1 | insufficient application |
| CHARIMA | IIHR | 1 | not in public domain, limited history |
| CLAWS | LLL | 1 | under development, poorly documented |
| DESERT | IIASA | 1 | under development, poorly documented |

(continued)

Table 2-2
(continued)

| *model* | *source of model* | *screened to Level:* | *comments* |
|---|---|---|---|
| | | **stream and river models (continued)** | |
| DYNHYD | CEAM | 3 | link-node 1-D, dated code |
| HSPF | USGS/CEAM | 3 | process models poorly documented, difficult to apply |
| MIKE-SHE | DHI | 1 | watersheds, drainage network, vadose zone & aquifers, proprietary |
| QUAL2E | CEAM | 3 | limited to steady-state conditions |
| QUALTX | TNRCC | 3 | same limitations as QUAL2E, specific to Texas watercourses |
| RIVER3 | GSC | 1 | insufficient application, inappropriate for Texas hydrology |
| RIVMOD | CEAM | 1 | hydraulics only, limited history |
| SMPTOX | CEAM | 1 | not suitable for TMDL-type problem, inappropriate for Texas hydrology, not current, limited history |
| WAM | SWET | 1 | proprietary |
| WASP | CEAM | 3 | must be coupled with suitable hydro-dynamic/transport model |
| | | **lake and reservoir models** | |
| BATHTUB | WES | 1 | statistical, limited history |
| CE-QUAL-ICM | WES | 1 | insufficient application |
| CE-QUAL-W2 | WES | 4 | deep stratified reservoirs, application difficult, code may contain bugs |
| DYNHYD | CEAM | 3 | link-node 1-D, dated code |
| EUTROMOD | NALMS | 1 | dated, limited history, inadequate acceptance |
| EXAMS | CEAM | 1 | insufficient application |
| $IDOR^{2D}$ | MU | 1 | proprietary |
| PHOSMOD | NALMS | 1 | not current, insufficient application |

(continued)

Table 2-2
(continued)

| *model* | *source of model* | *screened to Level:* | *comments* |
|---|---|---|---|
| | | lake and reservoir models (continued) | |
| POM | PU | 4 | complex to operate, limited water-quality capability; mainly estuary model, but has been applied to large lakes |
| QUAL2E | CEAM | 3 | 1-D, mainstem reservoirs only, limited to steady-state conditions |
| QUALTX | TNRCC | 3 | same limitations as QUAL2E, specific to Texas watercourses |
| WASP | CEAM | 3 | must be coupled with suitable hydro-dynamic/transport model |
| | | estuary or bay models | |
| CE-QUAL-ICM | WES | 1 | insufficient application |
| CE-QUAL-W2 | WES | 4 | deep channel estuaries, application difficult, code may contain bugs |
| CHARIMA | IIHR | 1 | not in public domain, limited history |
| DYNHYD | CEAM | 3 | link-node 1-D, dated code |
| EFDC | CEAM | 4 | complex to use, insufficient history of application, inadequate acceptance |
| IDOR$^{2D}$ | MU | 1 | proprietary |
| POM | PU | 4 | complex to operate, limited water-quality capability |
| QUAL2E | CEAM | 3 | limited to 1-D systems under long-term steady-state conditions |
| QUALTX | TNRCC | 3 | same limitations as QUAL2E, specific to Texas watercourses |
| TxBLEND | TWDB | 4 | 2-D horizontal, no water-quality capability, limited technical acceptance |
| WASP | CEAM | 3 | must be coupled with suitable hydro-dynamic/transport model |

(continued)

Table 2-2
(continued)

Agency abbreviations

| | |
|---|---|
| ARS | Agricultural Research Service<br>U.S. Department of Agriculture |
| CEAM | Center for Exposure Assessment Modeling<br>U.S. Environmental Protection Agency |
| DHI | Danish Hydraulic Institute (Hørsholm) |
| GSC | Geological Survey of Canada<br>Natural Resources Canada |
| ICIR | Potsdam Institute for Climate Impact Research<br>(Germany) |
| IIASA | International Institute for Applied System Analysis<br>(Austria) |
| IIHR | Iowa Institute of Hydraulic Research<br>University of Iowa |
| LLL | Lawrence Livermore Laboratory |
| MU | Water Resources Environmental Information Systems Laboratory<br>McMaster University |
| NALMS | North American Lake Management Society<br>(Madison) |
| NCSU | Biological and Agricultural Engineering Department<br>North Carolina State University |
| NSEL | National Soil Erosion Laboratory<br>Purdue University |
| OSU | Food, Agriculture & Biological Engineering<br>Ohio State University |
| PU | Program in Atmospheric and Oceanic Sciences<br>Princeton University |
| SSG | Scientific Software Group<br>Washington, DC |
| SWET | Soil and Water Engineering Technology, Inc.<br>Gainesville |
| TNRCC | Texas Natural Resource Conservation Commission |
| USGS | U.S. Geological Survey |
| USGS/W | U.S. Geological Survey, Wisconsin District |
| VTI | Biological Systems Engineering Department<br>Virginia Tech |
| WES | Waterways Experiment Station<br>U.S. Corps of Engineers |

# 3. Watershed models

## *3.1 Role of the watershed*

The ultimate source of water is precipitation, and the ultimate concern of a TMDL is the quality of water in a watercourse. The intermedium through which precipitation is transformed to streamflow is the watershed. In the physical system, therefore, the watershed occupies a central role in the quantity and quality of water in the watercourses: it acts as a processor of precipitation to create streamflow. The importance of the watershed as a processor is indicated by the fact that only a modest fraction of the quantity of precipitation falling on a watershed actually reaches the drainage system.

Figure 3-1 displays isopleths of this fraction, the ratio of runoff to rainfall, based upon long-term average values, for Texas. There is clearly a wide variation in this ratio across the state, a consequence of climatology and landscape (which are, themselves, interdependent). Even in the humid eastern section of the state, only one-fourth of the impingent precipitation actually appears as runoff in the surface drainage. Farther west, this fraction decreases to less than three per cent. Moreover, what this figure does not communicate is that there is a strong variation in time of this ratio at any fixed point, depending upon the condition of the land surface and the time characteristics of the meteorological event producing the runoff.

Until recently, explicit consideration of the watershed has been largely avoided in surface water management by considering the watercourse in isolation from its environment (cf. Fig. 2-2), and characterizing the flux of water into the watercourse as a boundary condition. For considerations relating to impacts and management of point discharges of wastes into a stream or river, for which streamflow provides a diluting and transport mechanism, this is not an inappropriate simplification. But many waterborne constituents are introduced into the water as a consequence of the processing by the watershed, some of which are themselves pollutants, and some of are not, but do affect how the receiving water responds to pollutants. Since a TMDL considers all

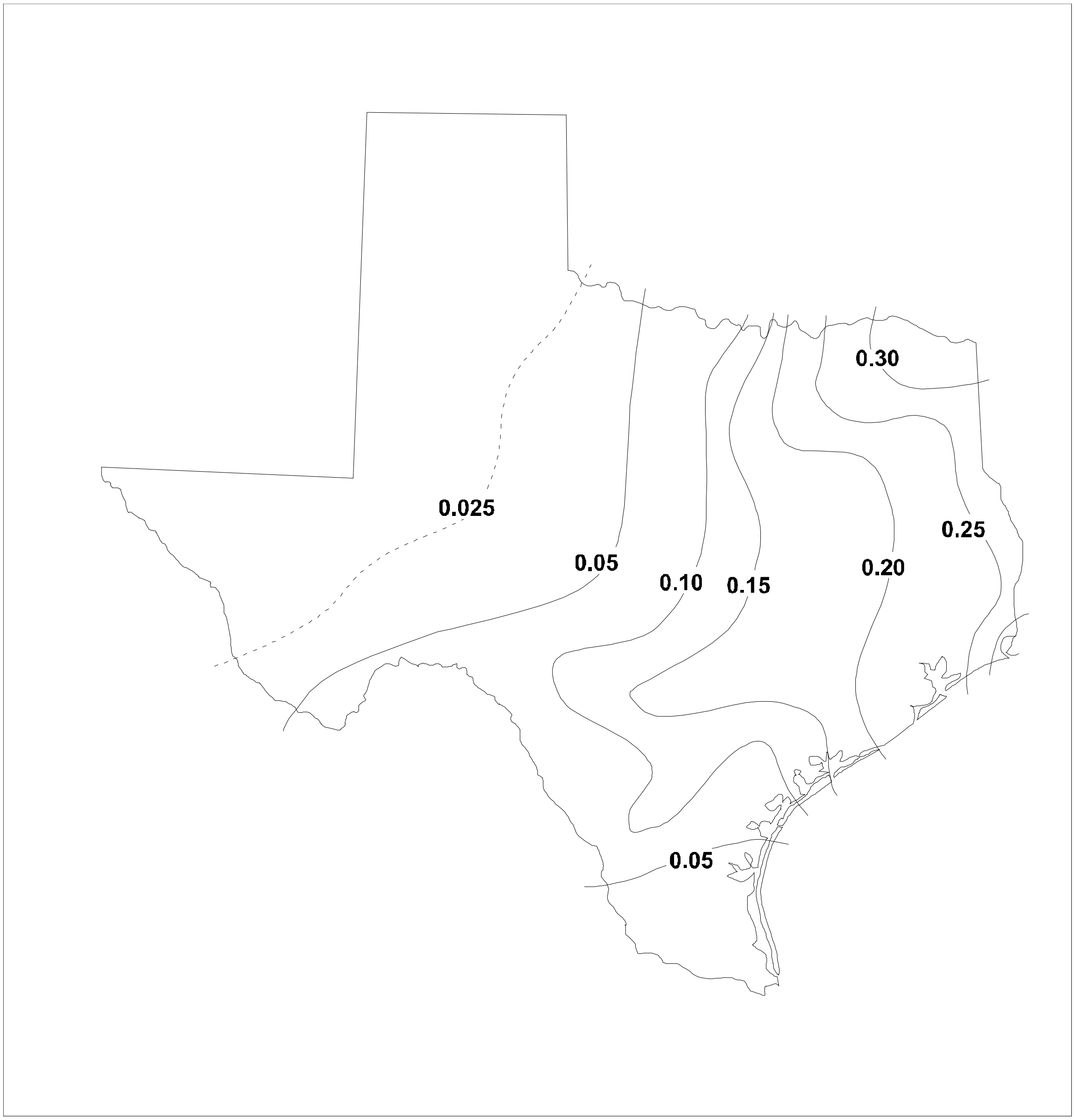


Figure 3-1 Runoff/rainfall contours for Texas (after Ward and Valdes, 1994)

sources of pollutants in an integrated analysis, it must explicitly address the role of the watershed processor. Thus, a model is needed that allows direct evaluation of the processing of precipitation by the watershed.

There are two central processes in the effect on stream water quality of the hydrology of watersheds. First is the process of infiltration, by which water impingent on the surface of the watershed penetrates the ground and is thereby removed from the surface-water budget. It is often stated that runoff is the excess of the rate of precipitation at the surface over the "capacity" of the soil for infiltration (Horton, 1933). While this is accurate, this form of statement is suggestive that one need only debit a fixed rate (of infiltration) from precipitation to determine runoff. It is a short jump with this line of reasoning to the assumption that runoff can be estimated as a fixed percentage of precipitation. Consideration of Fig. 3-1 makes it clear that the infiltration "debit" in Texas is at least three-fourths of the precipitation, usually more, so that errors in its estimation become magnified as errors in the associated runoff. If infiltration averages around 90% of precipitation and is estimated within ±10%, the corresponding estimate of runoff will be ±100%. Thus in evaluating watershed models, one of the important aspects to be examined is how infiltration is modeled.

The second central process is the mobilization ("detachment") and transport of surface particles by the movement of water on the land surface. The role of water movement in erosion of the land surface is well known: this process of mobilization and transport is basic to erosion. From the standpoint of TMDL determination, whatever is removed by erosion from the land surface is carried into the drainageways. Many of the pollutants of concern to TMDL's are transported by exactly the same mechanism (some may be sorbed onto particles), and even those constituents that occur in the receiving water in solution, are dissolved in this process. The modeling of sediment transport in a candidate watershed model is therefore a major aspect of its evaluation.

There are other processes and depictions of watershed hydrology that can be important as well. Precipitation is also lost by vegetative interception and by ponding. The soil water budget, including vegetative transpiration, dictates the state of desiccation of the land surface and therefore affects infiltration at the outset of a rainfall event. Water flow (as runoff) on the watershed moves over the land surface as overland flow until intercepted by a drainageway then is routed through the network of tributaries to the receiving stream. The hydraulics are different for these two modes of movement. There is also an issue of scale: at the smallest scales water is captured by rills, these conflow into depressions and gullies, which feed channels then tributaries. In any model, the full drainage network generally cannot be completely resolved, but must be aggregated in some way. The capacity of a watershed model for explicitly addressing these features, and the associated parameters of the land surface, dictate its ability to represent modifications to the land surface, such as dikes, ponds, channels, and vegetation, all of the elements of BMP controls.

## *3.2 Important characteristics of watershed models*

### *3.2.1 Infiltration and runoff*

The method of modeling infiltration, and by differencing, runoff, in available operational watershed models falls into two categories: process-based or statistical. The process-based approach employs a mathematical formula for rate of infiltration. Several such equations have been developed, as reviewed by (e.g.) Swartzendruber and Hillel (1973), Smith (1976), Rawls et al. (1993), Hillel (1998). The expressions that appear to be most popular in watershed models are those of Horton (1940), Green and Ampt (1911), and Holtan (1961), see Table 3-1. These equations tend asymptotically with time (after rainfall begins) to a constant infiltration "capacity", really a rate, but differ in soil parameters and in the expression for the time-decaying part of the equation. Both the Green-Ampt and Holtan expressions have an initial (i.e., at the beginning of the rainfall) infiltration equal to the rainfall rate, while the Horton equation assumes a maximum feasible infiltration rate. The Holtan equation has been criticized because the parameters are not "physically based" and are therefore difficult to estimate from soil properties (Bouraoui and Dillaha, 1996). A considerable literature has been developed on the estimation of Green-Ampt parameters from more common soil features, e.g. Rawls et al. (1983). Although the parameters of the Horton equation can in principle be determined from soil properties (see

Table 3-1
Examples of mathematical infiltration models used in watershed models

| | | |
|---|---|---|
| Horton | $f(t) = f_c + (f_o - f_c) \exp\{-kt\}$ | (6) |
| Green-Ampt | $f(t) = K\,[F(t) + s] / F(t)$ | (7) |
| Holtan | $f(t) = f_c + a(F_p)^n$ | (8) |

$F(t)$ = cumulative infiltration — $f(t) = F'(t)$ = infiltration rate
$f_c$ = ultimate (asymptotic) infiltration — $f_o$ = initial infiltration rate
$k$ = time decay constant — $K$ = hydraulic conductivity
$s$ = parameter of capillary suction at wetting front
$F_p$ = potential cumulative infiltration when constant rate is acquired
$a,n$ = parameters of soil & vegetation

Eagleson, 1970), they are more commonly estimated by fitting the equation to runoff data (Rawls et al., 1993). For all of these equations, once the parameters of the equation are established (e.g., from soils data), the amount of infiltration is given as a function of soil moisture and rainfall rate, from which the excess in rainfall rate is equated to runoff.

Another family of mechanistic infiltration models are derived directly from some form of Darcy's equation, e.g. the nonlinear Fokker-Planck equation. These tend to be more theoretical, see, e.g., Philip (1969) and Parlange (1974), are more difficult to relate to larger scale ("bulk") soil parameters, and do not appear to have been applied in operational watershed models.

The statistical method uses assumed functional relations between runoff and rainfall fitted to measured data. The most important statistical model in the watershed models reviewed here is the SCS curve-number method. This method was published as part of the SCS *National Engineering Handbook* based upon data from "small watersheds" collected by ARS in the 1950's and early 1960's (Mockus, 1972c). The method relies upon two assumed relations between cumulative volumes of precipitation, infiltration, and runoff, each a function of time, (9) and (10) in Table 3-2, from which a single equation (11) for runoff can be obtained by eliminating one of the variables. The model uses two parameters: A, the "initial abstraction," i.e., the volume of rainfall absorbed before runoff begins, and S the "maximum possible storage" of the watershed. The parameter S is considered to be governed by the physical properties of the soil, the surface treatment of the soil (especially tillage and vegetation), and the antecedent moisture conditions, i.e., the soil moisture at the time of the beginning of the storm.

A "curve number" CN is defined by equation (12), Table 3-2, which is the basis for parameterizing soil properties. Figure 3-2 shows the relation between this CN and the potential storage parameter S. Mockus (1972c) does not present the data from which the various CN values are derived, except for a selection of data in his Fig. 10.2, which suggests the vast majority of measurements of S to range 0.3 - 10 in, corresponding to a range of CN of 50 - 98. The initial abstraction A is assumed to be a fixed fraction of S, taken to be $a = 0.2$ in all of the SCS tabulations and graphs. The wetted infiltration rate of a soil ("runoff potential") is categorized into four "groups" (A, B, C, D, from highest infiltration to lowest), and Mockus (1972a) assigns over 4000 soil types to these four groups, an assignment made largely by judgments of "soil scientists and correlators". Watersheds are also characterized by "land use"

Table 3-2
Mathematical infiltration model used in SCS Curve-number method

$$F(t) = [P(t) - A] - Q(t) \tag{9}$$

$$F(t) / S = Q(t) / [P(t) - A] \tag{10}$$

$$Q(t) = (P - aS)^2 / [P + (1-a)S] \tag{11}$$

$$S = 1000/CN - 10 \tag{12}$$

| | |
|---|---|
| P(t) = cumulative precipitation | A = initial abstraction |
| Q(t) = cumulative runoff | S = maximum possible storage of watershed |
| F(t) = cumulative infiltration | $a$ = A/S |
| CN = curve number | |

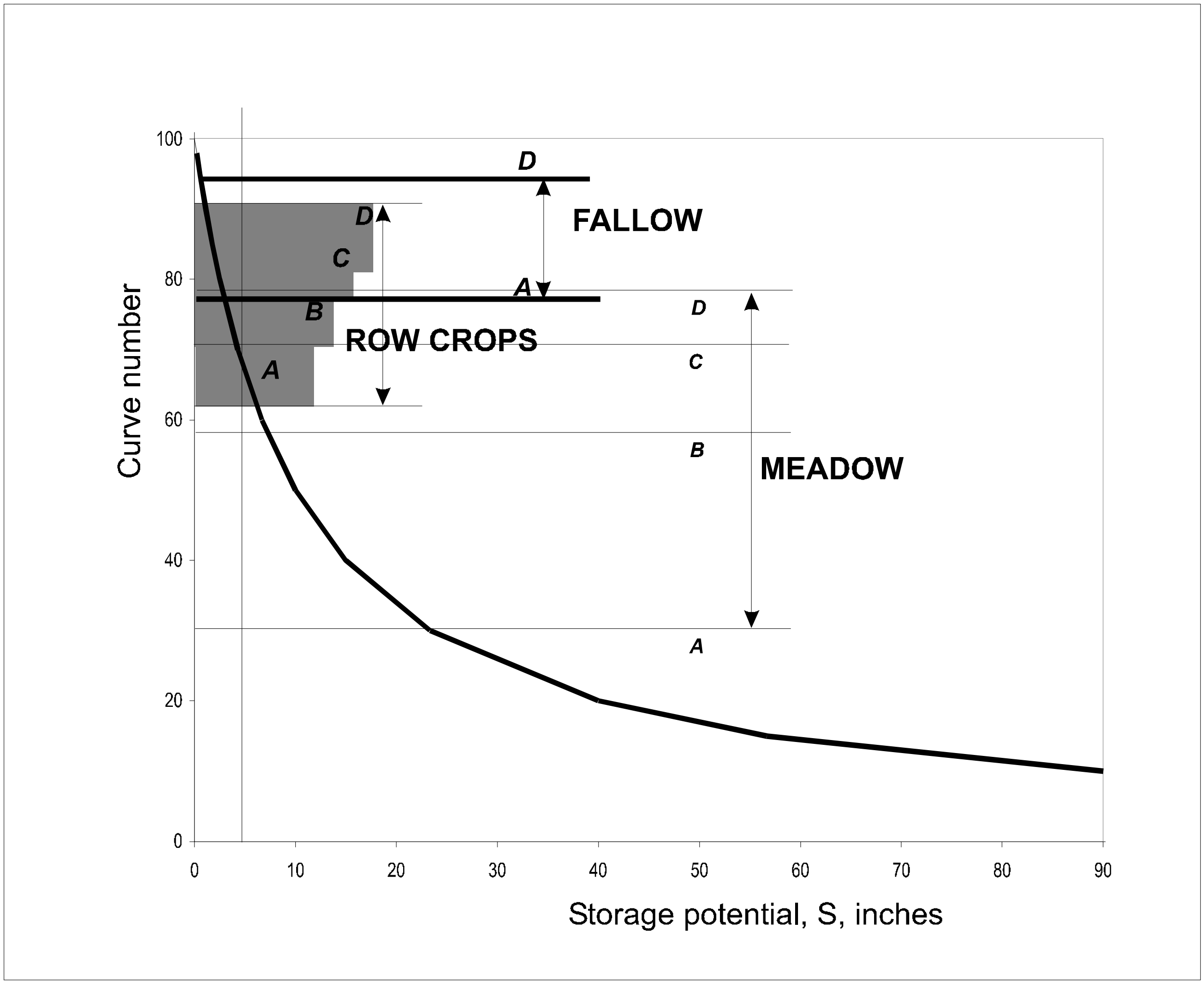


Figure 3-2 - Relation of SCS curve number to storage potential S (e.g., meadow, woods, fallow, row crops, etc.) and "hydrologic condition", a qualitative estimate of vegetation density and condition, "good" promoting infiltration, and "poor" promoting runoff (Mockus, 1972b). This leads to a three-dimensional categorization of soil and land-use, each of which is assigned a curve number, in a master look-up table in Mockus (1972b). Within the agricultural categories of "land use" there is a further subdivision according to agricultural practices. Three of the soil/land-use categories are plotted in Fig. 3-2.

Antecedent moisture conditions are considered to affect the value of S and therefore the corresponding curve number. SCS defines three categories, based on rainfall in the preceding five days, *viz*. AMC-I ("low"), AMC-II ("average"), and AMC-III ("high"). The master look-up table (Table 9.1 in Mockus, 1972b) and the soil/land-use categories shown in Fig. 3-2 assume AMC-II (average) conditions (Mockus, 1972c). In Fig. 3-3 is shown the range of correction in CN necessary to account for antecedent moisture, evidently considerable in comparison to the range of CN with soil/land-use categories (Fig. 3-2). A more revealing display results from back-transforming the curve number from equation (12) to recover the values of S (data for which the SCS relations have presumably been fitted), shown in Fig. 3-4. The effect of antecedent moisture conditions is seen to modify the range of S from 50% to 250% of the "average" value implicit in the published relations.

The SCS method is regarded rather ambiguously by workers in the field. On the one hand, it is part of a design method promulgated by the USDA that provides a well-defined and definite means of computing the surface water budget for a watershed, based upon careful measurements in experimental watersheds, and has therefore received wide application. On the other hand, the basis for the model is not well-accepted. The method relies upon nomographs and look-up tables, and a great degree of "intuition" is necessary to properly assign parameters for cover treatment and condition. While a computerized version has been presented by Mack (1995), it can be debated whether the user is done a service by not having access to the background information for assignment of cover and condition parameters given in the users manual. The second relation (10) has been criticized as an unrealistic physical model of infiltration (e.g., Smith, 1976, Singh, 1989). The model is apparently sensitive to the value of *a*. The selected

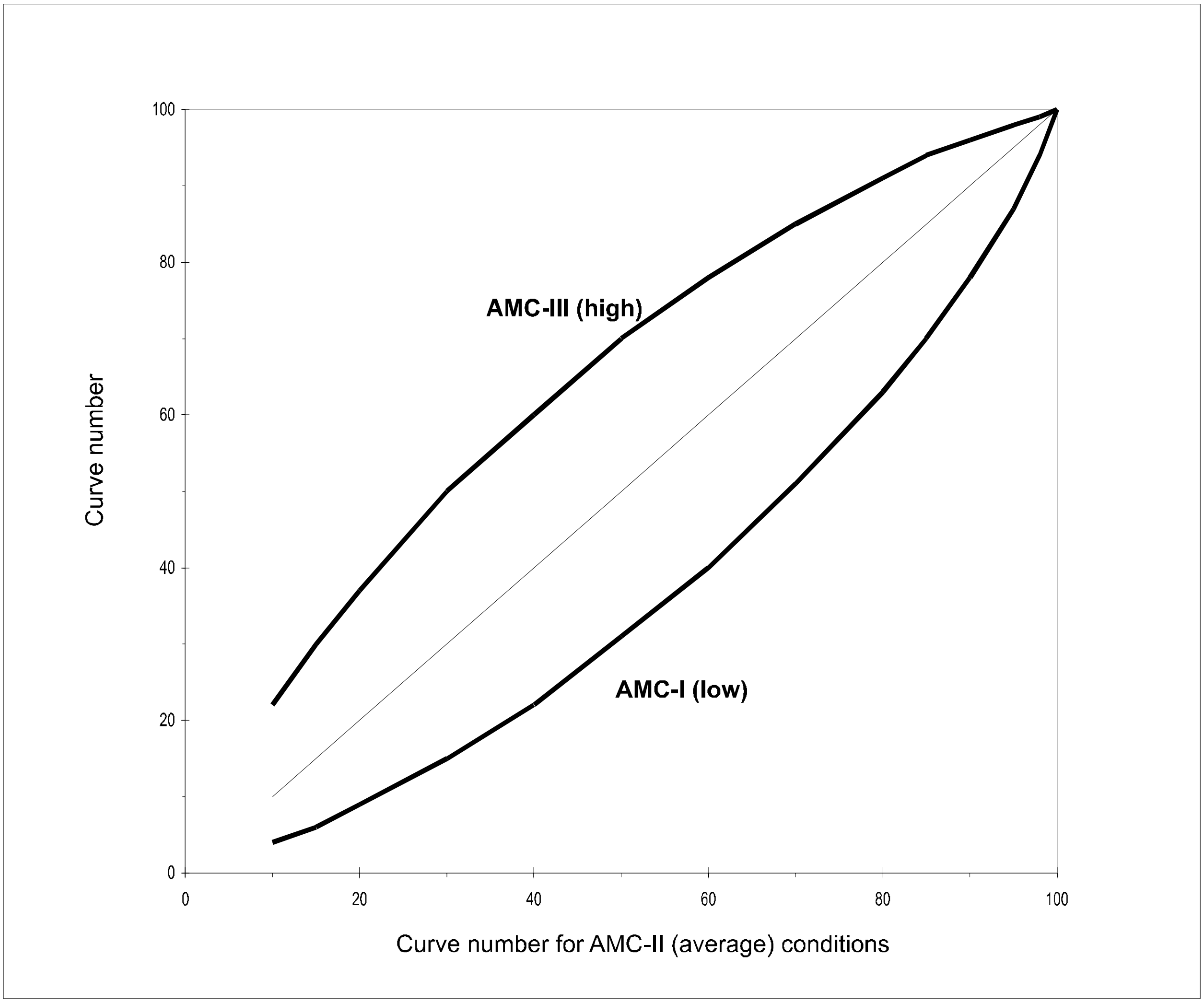


Figure 3-3 - Correction of average Curve Number to account for antecedent moisture conditions

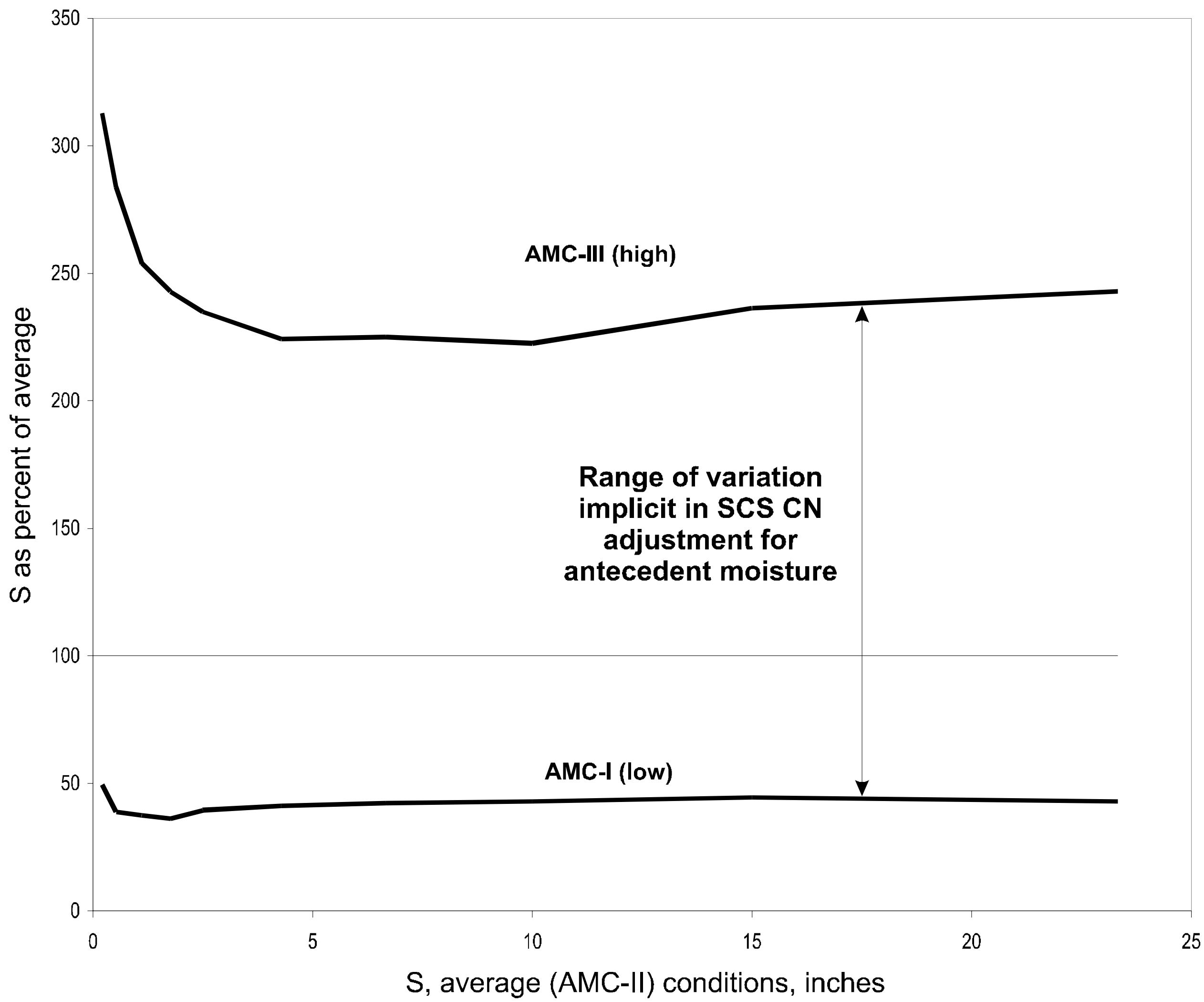


Fig. 3-4. Effect of antecedent moisture conditions on range of S

data given in Mockus (1972c) indicate a range of 0.02 - 2.1, with 50% of the data lying *outside* the range 0.09-0.36, yet the method (including the determination of CN from data and the various tabulations of CN versus soil conditions) assumes a value of 0.2. The role of S in the model formulation is contradictory. For the computation itself it is taken to be constant, but it is also a function of antecedent moisture conditions and therefore controlled by previous infiltration events. (Why, therefore, would it not vary during the course of a storm?) Rawls and Brakensiek (1986) compared the Green-Ampt and CN methods, finding the former to be superior in accuracy. Probably the greatest problem is that the original data upon which the method is based do not appear to have been published, and some of the legitimate statistical issues of how well the model really represents the data to which it is fitted have not been addressed. It is noteworthy that the recent comprehensive text on soil physics by Hillel (1998), which includes extensive discussion of infiltration, maintains a stony silence about the SCS method.

### *3.2.2 Spatial resolution: lumped versus distributed*

Once water accumulates at the surface, i.e. the rate of precipitation exceeds that of infiltration, it can be moved by hydraulic forces, *viz*. pressure gradients driven by higher water levels where water has accumulated or where the terrain is at higher elevations, and affected by frictional drag at the land surface. The resulting flows are conceived to take place over the terrain surface itself as "overland" flow, or to move into surface drainageways, higher-order distributaries that form the network of channels eventually conflowing into the larger tributaries of the watershed. Watershed models vary in the extent to which they depict these separate flow mechanisms, and also vary in their ability to resolve such details of the drainage network.

The spatial resolution of the watershed model is one of its most salient features. Watershed models are described as being "lumped" or "distributed" but the meanings of these terms are not precise. Generally, these terms indicate opposite poles of the degree of spatial integration implicit in the model. A model that is formulated in terms of average characteristics across the watershed and computes the areal-averaged values of infiltration, runoff, etc., is "lumped," that is, the model is integrated over the entire surface of the watershed. A model that discretizes the watershed area into many points at each of which rainfall, elevation, soil, vegetation and other surface features are specified, and infiltration, runoff and other hydrological responses are computed, is "distributed." Clearly, input specification for a distributed model requires much more effort (and information) than for a lumped model. The "early" (ca. 1970) distributed models necessitated manual determination of each of the landscape properties and separate input via punched cards, a tedious, onerous procedure that led to favoring lumped models over distributed. With the advent of GIS, there has been a renascence of distributed models.

There are, however, many varieties of distributed models. Some, like SWAT, subdivide the model watershed into several or many subwatersheds, so that it in effect replaces a lumped watershed model with a concatenation of smaller, but still "lumped" subwatersheds. Others, like ANSWERS, subdivides the watershed into many square elements, at each of which a surface water budget is computed. This can be viewed as subdividing the watershed into small square catchments, each of which is "lumped." Whether a model is "lumped" or "distributed," we submit, is a matter of whether the model formulation seeks spatial resolution of all of the elements of the surface water budget, and is a matter of degree rather than dichotomy.

Other features of formulation differentiate various "distributed" models, all relating to the issue of resolution. One important such feature is how hydraulic movement of water is determined within the watershed computational elements, and the related aspect of how network channels are depicted. Again, there are two extremes, whether flow direction is specified *a priori* by the user, or whether it is computed from topography as part of the model solution. The method may differ for overland and channel flow. Channel configurations may be specified in some *a priori* way by the user, or computed directly from topography (and how this is done from a discrete network of elevations is yet another source of variation between models).

### *3.2.3 Sediment mobilization and transport*

For TMDL applications, an aspect of a watershed model that is equally important to the infiltration-runoff model is how hydraulic sediment removal from the land surface is formulated.

As noted in Section 3.1 above, in watershed models this process is viewed as erosion, but in terms of the effect on a receiving watercourse, this is the watershed sediment load. As is the case in modeling infiltration and runoff, for erosion there are mechanistic models and statistical models.

Of the statistical methods for erosion computation, the most important in operational watershed modeling is the so-called Universal Soil Loss Equation (USLE). This is a relationship originally developed to quantify agricultural soil loss, in which the long-term soil loss is given as a product of six factors: rainfall runoff erosivity, soil erodibility, topography, cover management, and supporting practices, equation (13) in Table 3-3. Some version of this equation has been in use by soil conservationists since the early 1940's, referred to variously as the "slope-practice" method or the Musgrave equation (Wischmeier, 1965). Its widespread use began with promulgation by the USDA in 1965 (Wischmeier, 1965) and was superceded and extended in 1978 (Wischmeier and Smith, 1978), and again in 1997 (Renard et al., 1997). The philosophy of the USLE is "scaling," in that the various factors in the product are chosen to scale the computed soil erosion from that measured for a standard reference, a soil plot with the following properties:

Length 22.1 m (72.5 ft)
Uniform slope of 9%
Fallow and barren (no plant cover)
tilled up- and down-slope

As is clear from these properties of the reference plot, the USLE was originally designed for agricultural lands in the U.S. east of the Rockies. It has been extended to rangelands and forests in the remainder of the U.S. and to cultivated lands in the Tropics.

Each scaling factor affecting erosion is determined from some data set, of variable range and extent, and is presented in the manual (Wischmeier and Smith, 1978) as isoplethed maps, nomographs, or table look-ups. The rainfall factor (R) is related to the kinetic energy of the raindrop and the maximal 30-minute intensity. This factor R varies from less than 20 for the

Table 3-3

Universal Soil Loss Equation formulation for watershed sediment erosion

$$A = R \cdot K \cdot L \cdot S \cdot C \cdot P \qquad (13)$$

$$R = 0.01 \sum (916 + 331 \log I)\, I$$

$$LS = \sqrt{(L)}\,(0.0076 + 0.53S + 7.6S^2)$$

| | |
|---|---|
| A = annual soil loss, tons/ac/yr | R = rainfall-runoff erosivity |
| K = soil erodibility | S = slope steepness |
| C = cover management | L = slope length |
| P = supporting practices | I = rainfall intensity, in./hr |

high basins of the Rockies to 600 in the New Orleans - Mobile area of the Gulf Coast, and in Texas ranges from 75 in the Trans-Pecos to 500 at Sabine Pass, the isopleths (called isoerodents and having nothing to do with small mammals) more or less conforming to isohyets of annual rainfall. The soil erodibility factor K measures the ease of erosion of a soil, generally ranging from 0.01 (practically nonerodible) to 1.0 (very erodible, such as silt and fine sands), but is somewhat less well-established than R. A nomograph for computing K based upon percent silt and fine sand, permeability and four organic matter structures is given by Wischmeier and Smith (1978), and a tabulation of "generalized" K values for 8000 soil series is given in SCS (1975). Topography is represented by the value of the product, normalized so that the 22.1-m 9%-slope reference plot has LS = 1. Erosion is conceived to increase with run length L and grade steepness S. Figure 3-5 displays the variability of LS based upon tabulated data in Moore and Wilson (1992). Both the cover factor C and the practice factor P are complicated and based upon a smaller data resource than the previous factors. The cover factor accounts for species, density and distribution of vegetation, and ranges 0.05 - 1.0. The practice factor P accounts for contouring, strip planting, terracing, and related strategies, ranging 0.25 to 1.0. (Since the standard reference is barren, both C and P will be less than unity.) Onstad et al. (1979) present

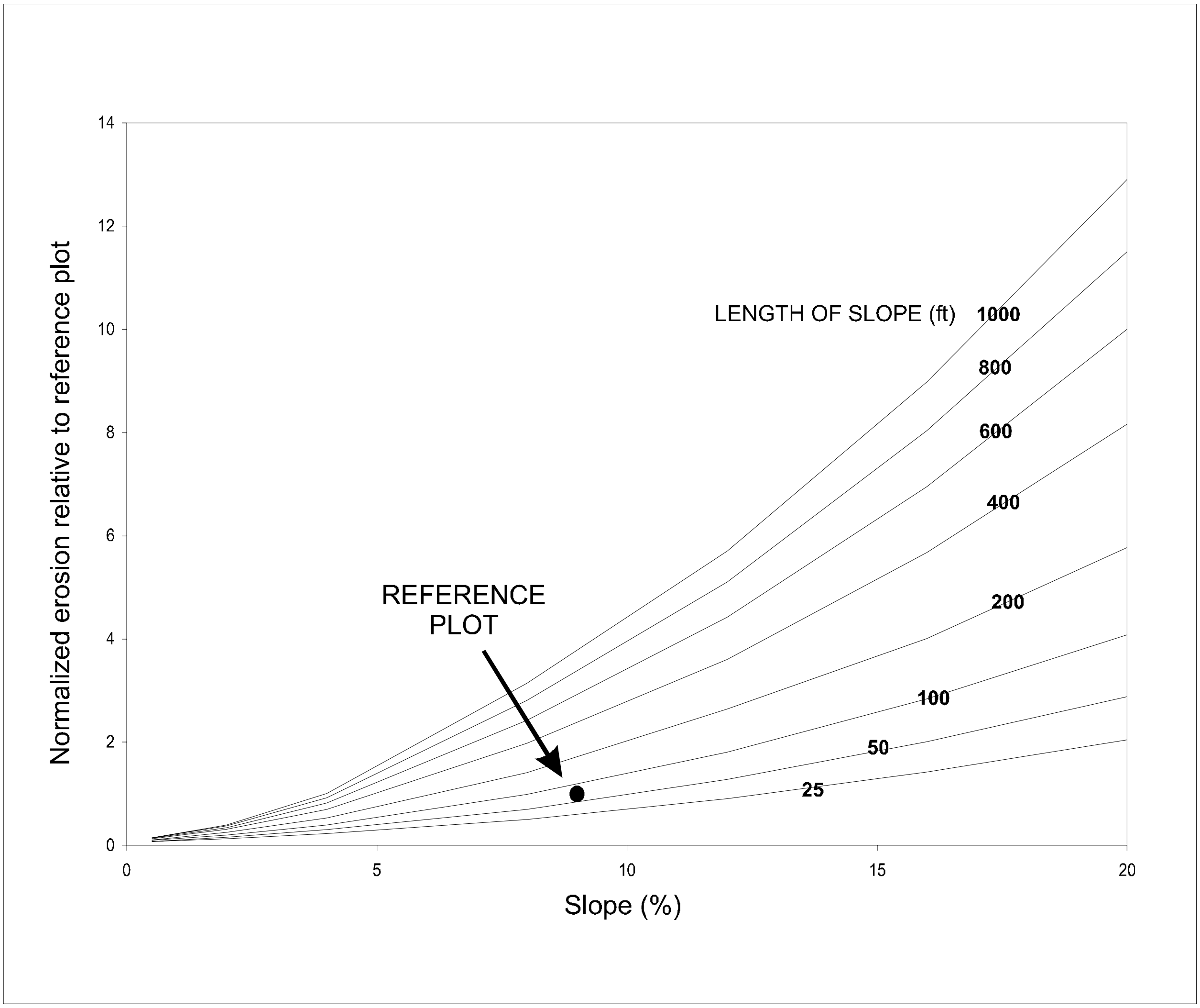


Figure 3-5. Length-slope factor of USLE (from data of Moore and Wilson, 1992)

data on the precision of each factor in the USLE, based upon 42 research field plots distributed over eight midwestern states, giving values of the (within-treatment) coefficients of variation as:

| | | | |
|---|---|---|---|
| R | 34 % | K | 39 % |
| L | 19 % | S | 5 % |
| C | 92 % | P | n/a |

Recently, a revised version of the USLE has appeared, named (appropriately) the Revised Universal Soil Loss Equation (RUSLE), Renard et al. (1997), see also Renard and Ferreira (1993). The primary changes in the RUSLE, relative to USLE, are (1) a slight reformulation of R to better reflect storm "energy", (2) a more accurate estimate of LS, incorporating effects of surface residues, decreasing infiltration over the course of a storm, converging and diverging terrain, deposition on the watershed, and (3) implementation in a computer program. Though RUSLE is available as a stand-alone product (Renard et al., 1997, Ward and Benaman, 1999), it is also incorporated into the ARS model WEPP.

The popularity of the USLE is an example of how nature abhors a vacuum. The need for a means of quantifying soil erosion, where no reliable methodology existed, was filled by the USLE. It was developed to address an important, but, from the standpoint of watershed modeling, a rather narrow problem, namely the loss of soil from small-scale rill and sheet erosion, particularly croplands (Wischmeier, 1976). It has been widely used, especially by soil scientists and agents in the field, and has been applied to situations remote from the original intended application, e.g. urban construction areas, highway embankments, mine tailings, coal piles, and recreational sites (Foster, 1977, Renard et al., 1991). It has been applied by Los Alamos to estimating plutonium loads from fallout, and was adopted by EPA as the basis for determining conservation compliance, per the 1977 Clean Water Act amendments.

The method has, however, been widely criticized (Miller and Donahue, 1995, Hillel, 1998). The mathematical relation (13) upon which it is based has been dismissed as over-simplified (Hillel, 1998). The data upon which the various factors are determined have been criticized as inadequate and the analysis faulted for failure to take explicit account of sources of variability.

Even the best-established of these factors, the R factor, has many problems.  No account is made for duration of the storm or the occurrence of overland flow.  The tabulated and graphed R factors represent 22-yr averages, but the year-to-year variability in R is estimated to range a factor of 2 about this 22-yr mean (see Miller and Donahue, 1995), which translates to an error of estimation of the same magnitude in computed sediment loss A.  The method was intended to apply to agricultural plots, and the very nature of the method limits it to small watersheds: the L values only range up to 1000 ft.  Yet in the watershed applications it is applied to areas that are orders of magnitude greater.

Probably the most telling indication of the technical status of USLE is that it has gradually been de-emphasized in WEPP (Ward and Benaman, 1999).  While the USLE/RUSLE technology remains a option to the user of WEPP, most of the development and application of WEPP in the last decade has been directed toward its process-based distributed component models.  The current version is reported to have eliminated USLE-type parameters entirely from the calculations of interrill sediment loading (Foster et al., 1995).  (However, the replaced interrill term is still a product form, each factor of which is empirical, and identified with a physical process or parameter of the soil or rainfall.)

The mechanistic models utilize directly the rainfall rates applied at the land surface and the resulting water depths and current speeds inferred from the runoff calculation to quantify each step in the erosion process: the initial "detachment" of a particle, i.e. its mobilization from the soil-plant matrix of the surface, its transport with overland flow, its subsequent deposition elsewhere on the surface or its discharge into a network drainageway.  (Some workers use the generic term "detachment" to also apply, with a change of sign, to deposition, e.g. Foster et al., 1981.)  Sediment is mobilized from the land surface by the direct impact of rainfall drops and by entrainment in the movement of water over the land surface.  On the smallest scale, this "interrill" source is still modeled by empirical relations, but based upon careful, detailed measurements on experimental "rainulator" plots.  As the runoff is organized into rills, then channels and gullies, the bed entrainment becomes a more important part of the source of sediment to the flowing water, and the transport of sediment in the flow becomes increasingly important in the mass budget of sediment.

The process of sediment transport has been much more intensively studied than the processes of erosion and deposition. If the water is sufficiently deep and swift, two modes of sediment transport occur, that of sediment suspended in the free movement of the water as "suspended load" and that in a near-bottom, loosely aggregated, mainly saltating mode referred to as "bedload." Several equations have been developed for both types of load, most common are those due to Einstein, Bagnold, Simons and Yalin (Bagnold, 1966, Yalin, 1977, Julien, 1998).

Mobilization of sediment is conceived as dependent upon the stress exerted by the flowing water exceeding a critical value, and the rate of entrainment being proportional to the departure from this critical stress. The generic relationships are summarized in Table 3-4. If D>0 then sediment

Table 3-4
Mechanistic equations for sediment detachment (a.k.a. mobilization)

$$D = \begin{cases} D_c \, (T_c - G) / T_c & \text{for } T_c > G \\ \beta V_f \, (T_c - G) / h\bar{u} & \text{for } T_c < G \end{cases} \tag{14}$$

$$D_c = K_r \, (\tau - \tau_c) \tag{15}$$

$G$ = sediment load in flowing water (e.g., kg/s)
$D_c$ = detachment (a.k.a. entrainment, mobilization, resuspension) capacity
$\tau$ = stress exerted on soil surface by flowing water
$\tau_c$ = value of critical stress for sediment texture and cohesiveness
$T_c = f(\tau)$ = sediment transport capacity in flowing water
$h$ = depth of flow
$\bar{u}$ = vertical mean current
$\beta$ = turbulence coefficient
$K_r$ = coefficient relating to erodibility of soil

is mobilized from the bed and entrained into the flow, and if $D<0$ then deposition takes place. A model must compute G from a mass balance on sediment, so it is evident that G is both the dependent variable and a term in the source/sink part of the equation. Moreover, the parameters governing D (as well as sediment transport) are dependent upon the hydrodynamics, so a feed-forward model calculation is necessary. The same basic equations of Table 3-4 are applicable to both sheet (or interrill) flow and channelized flow, though the expressions for the parameters are different, and depend upon the scale of the process, e.g. an interrill area versus a field versus a small catchment. For overland flow, considered to be shallow and erosive, the sediment transport capacity is assumed to be bedload, and can be computed by any of the relations cited above.

Several candidate models carry the watershed processes only through hydrology and sediment loading (or erosion), e.g. PRMS. For TMDL purposes, depiction of entrainment and transport of chemical constituents is needed as well. There are two broad classes of chemical transport processes: in association with sediment for those constituents sorbed to particulates, and in solution in the water itself. For the former, the kinetic processes may include interaction with plants, soils and soil water, and might include expressions for foliar interception, degradation, and washoff, as well as adsorption, desorption, and degradation in the soil. For the latter, the key factor is solute transfer to surface runoff, which generally assumes a mixing depth in the upper layer of soil (for CREAMS and GLEAMS, this is 1 cm, see Leonard and Wauchope, 1980), with specified pore concentration of constituent from which surface entrainment is computed, conventionally treated as first-order (e.g., Frere, 1973, Lai and Lo, 1992).

### *3.3 Model summary and evaluation*

Of the twenty-six (26) watershed models initially considered in this review (see Table 2-2), nine (9) were given more detailed review than the Level-1 screening. These nine models were reviewed with respect to capabilities and methods for the following processes:

- Runoff computation methods
- Erosion/sediment load computation
- Contaminant loads
- Model program structure and coding
- Model input requirements

The comparative capabilities of these models are summarized for each of these categories in Tables 3-5 through 3-9. It must be emphasized that this review is based upon the documentation for each model, applications reported in the literature, and (in a few cases) inspection of the computer code, but not upon set-up and operation of the individual models, which was beyond the scope of this study. Discussions of the individual models are given in Ward and Benaman (1999).

### *3.3.1 SWAT*

For many years the U.S. Department of Agriculture through its Agricultural Research Service (ARS) and Soil Conservation Service (SCS, now Natural Resources Conservation Service, NRCS) has engaged in a process of model development to support management of agricultural lands, notably effects on small agricultural catchments of various tillage and soil management practices to loss of topsoil, and results of application of fertilizer and pesticides. A part of these models must necessarily address the efflux from the agricultural catchment of sediment and chemicals in order to characterize the erosion and chemical budget of the catchment. With recent increasing concern about the impact of runoff from agricultural areas on downstream water quality, attention has been directed toward these efflux terms as loads, and the agricultural management models have become to be regarded as agricultural loading models. This change in focus is evident in the increasing geographical scale of these models: earlier models like CREAMS and EPIC were designed as "field-scale" models, whereas later models like SWRRB and SWAT were designed to have the capability of addressing several to many subcatchments in an overall watershed model. One such model emerged from the screening of this review as a viable candidate of TMDL determination in Texas, *viz*. SWAT.

Table 3-5
Watershed models - Runoff computation methods

| *Model* | *Infiltration** | *Runoff** | *Land surface* | *Drainage* | *Subsurface treatment* | *Non-rainfall period* |
|---|---|---|---|---|---|---|
| ANSWERS | Holtan | H | varied soils | surface slope & channel network | n/a | event only |
| CREAMS Option 1 | | CN | single crop, soil | CN slope term | multiple layers | CS, ET, percoln |
| CREAMS Option 2 | G-A | H | single crop, soil | surface slope | multiple layers | CS, ET, percoln |
| EPIC | | CN | single crop, soil | CN slope term | multiple layers | CS, ET, percoln |
| GLEAMS | | CN | single crop, soil | CN slope term | multiple layers | CS, ET, percoln |
| HSPF | quasi-empirical | H | homogeneous subwatersheds | surface slope | 2 zones† | CS, ET, percoln |
| PRMS | G-A | H | variable soil & surface | HRU surface slope | 2 zones† | CS + event, ET |
| SWAT | | CN | homogeneous subwatersheds | CN slope term | multiple layers | CS, ET, percoln |
| SWMM | G-A | H | urban watersheds | drainage network | 2 zones‡ | CS + event |
| WEPP | G-A | H | subwatersheds | rills, gulleys | multiple layers | CS, ET, percoln |

Key:

| | |
|---|---|
| CN | SCS Curve Number method |
| CS | continuous simulation, can accommodate non-precipitation weather |
| ET | evaportransipiration |
| event | storm only simulation, no capability for interstorm water budget |
| G-A | Green-Ampt equation |
| H | Hortonian, i.e. runoff is excess of rainfall over infiltration |

* One of the two is modeled, from which the other is computed    † both vadose    ‡ one vadose, one saturated

Table 3-6
Watershed models - Erosion/sediment load computation methods

| *Model* | *Method* | *Detachment* | *Transport* | *Deposition* | *Surface* |
|---|---|---|---|---|---|
| ANSWERS | empirical | product | Yalin | Yalin | USLE coeffs |
| CREAMS | USLE | n/a | n/a | n/a | USLE coeffs |
| EPIC | USLE | n/a | n/a | n/a | USLE coeffs |
| GLEAMS | USLE | n/a | n/a | n/a | USLE coeffs |
| HSPF | empirical | power-law | linear w/Q | none | USLE coeffs |
| PRMS | empirical | rill/interill | linear w/Q | none | ? |
| SWAT | USLE | n/a | n/a | n/a | USLE coeffs on subwatersheds |
| SWMM | USLE or user-input rating | n/a | n/a | n/a | USLE coeffs |
| WEPP | excess stress | rill/interill | Yalin | Yalin | field-scale |

Table 3-7
Watershed models - Contaminant loads

| *Model* | *Parameters* | *Source functions* | *Kinetics* | *Surface treatments* |
|---|---|---|---|---|
| ANSWERS | none | n/a | n/a | n/a |
| CREAMS | N, P, pesticides | plant uptake sorption | first-order | multiple soil layers, root uptake |
| EPIC | none | n/a | n/a | n/a |
| GLEAMS | N, P, pesticides | plant uptake sorption | first-order | multiple soil layers, 1 crop |
| HSPF | N, P, C, BOD, arbitrary tracers | buildup/ washoff | first-order or isotherms | 4 soil layers, plant uptake parameters |
| PRMS | no capability | | | |
| SWAT | N, P, pesticides | sorption | first-order | 1 soil layer, 1 crop |
| SWMM | anything* | buildup/ washoff | none | combinations of porosity, imperviousness & field capacity |
| WEPP | none | n/a | n/a | n/a |

* user must be able to supply source functions

Table 3-8
Watershed models - Model program structure and coding

| *Model* | *Source code* | *GIS linked* | *Output format* |
|---|---|---|---|
| ANSWERS | FORTRAN | no | ASCII tables |
| CREAMS | FORTRAN | no | large ASCII files |
| EPIC | FORTRAN | no† | |
| GLEAMS | FORTRAN | no† | large ASCII files |
| HSPF | FORTRAN | yes | graphs & tables |
| PRMS | FORTRAN | yes | plots, tables, statistics |
| SWAT | FORTRAN | yes | |
| SWMM | FORTRAN | no* | Tables & *.wmf graphics in Windows interface |
| WEPP | FORTRAN | no | Windows interface |

† academic integration with GIS has been done

* commercial products exist

Table 3-9
Watershed models - Model input requirements

| *Model* | *topography* | *soils* | *land surface* | *hydrometeorology* | *time interval* |
|---|---|---|---|---|---|
| ANSWERS | Grid topo | field cap (%) | grid point input | breakpoint rainfall | arbitrary |
| CREAMS | CN slope factor | field cap. | 1 slope & length | precip record | arbitrary |
| EPIC | none | field cap | 1 slope & length | precip record | daily |
| GLEAMS | none | field cap | 1 slope & length | precip record | daily |
| HSPF | subwatersheds | infiltration parameters | slope & length of each segment | precip record | arbitrary* |
| PRMS | HRU's | field cap | slope & length of each HRU† | precip record | daily or "storm scale" |
| SWAT | CN slope factor | field cap | slope & length | precip record | daily |
| SWMM | subcatchments | infiltration parameters | slope & length of each subcatchment | precip record | arbitrary |
| WEPP | topography | roughness, hydr. conductivity | slope & length | precip | daily or longer |

* can range from minutes to days, but some subroutines have fixed pre-set intervals
† in "storm mode" each HRU can be subdivided into multiple plane surfaces

The principle purpose of SWAT is computation of runoff and loadings from rural, especially agriculture-dominated, watersheds (Williams and Arnold, 1993). SWAT is the latest incarnation of a family of watershed models developed by USDA extending back to CREAMS and ROTO. The immediate predecessor of SWAT is SWRRB, which SWAT has now replaced at the ARS. The fundamental spatial unit for the model is the "subbasin" assumed to be homogeneous in all watershed parameters. While SWRRB was designed for application to a rather small watershed, which could be subdivided into no more than 10 subbasins, SWAT extends SWRRB by allowing multiple subbasins, up to 10,000. Not only does this permit more spatial resolution, it also allows SWAT to address larger watersheds than was the case for SWRRB, as much as several thousand miles in area.

Each subbasin in SWAT has an associated interior channel modeling the principal drainageway within that subbasin. In addition the outlet of each subbasin is conceived of being connected to the outlet of the basin by a routing channel. The soils of each subbasin are subdivided into several layers extending from the surface throughout the root zone. The first (uppermost) soil layer of thickness 10 mm determines the disposition of water and controls sediment and chemical quality of the runoff water. The model includes a plant growth submodel, a somewhat simplified version of that in EPIC, see Ward and Benaman (1999)

The heart of the hydrological component of SWAT is a computerized version of the SCS Curve Number method, by which runoff is directly computed, and the remaining surface water available for infiltration, plant uptake, and evaporation determined by differencing (Arnold and Williams, 1995, Arnold et al., 1990). Sediment mobilization and transport are computed by digital versions of the USLE. Most of the soil data for SWAT can be taken from the SCS Soils-5 database. To specify crops and agricultural practices requires vegetation types, tillage operations, number of crops in rotation, planting and harvest dates, curve numbers, biomass conversion factor, water stress yield factor, harvest index, and if irrigation is an option, the date and the amount of irrigation, or the water stress and irrigation runoff ratio.

SWAT is being used by the NRCS Blackland Research Center to determine loadings into Lake Waco. Its watershed was subdivided into 47 subbasins, organized into 7 basins, of which four are main tributaries (North Bosque, Middle Mosque, South Bosque and Hog Creek) and three are small peripheral catchments of the lake. The model was calibrated using the detailed flow and water quality data acquired by Tarleton State University for 1993-97. One of the key calibration parameters in this application was the curve number, and BRC reduced the CN by 5 units throughout all the subbasins in order to achieve calibration versus monthly flows, typically within 20%. For sediment loads, the minimum C factor of the USLE (see Table 3-3) was doubled.

The wide variety of agricultural surface treatments in SWAT are its greatest advantage. These, in effect, place a number of structural and nonstructural BMP's at the disposal of the modeler for evaluation in alternative strategies of runoff control from agricultural areas, a ubiquitous problem in Texas. The greatest weakness of SWAT in the view of these reviewers is its reliance upon the empirical formulations of the CN and USLE methods. An alternative agricultural management model that is more deterministic in concept is WEPP, also being developed within ARS, see Ward and Benaman (1999). It is not recommended for consideration at present because it is still largely developmental and has not had the application experience of the SWRRB-SWAT family. Its development should, however, be monitored for future applicability in Texas TMDL's.

### *3.3.2 HSPF*

One general-purpose watershed model is recommended for consideration as a candidate TMDL tool, *viz*. HSPF (Donigian et al., 1984, 1995). (A stripped-down version of HSPF, called NPSM, is incorporated into the EPA TMDL model shell BASINS, see Chapter 7.) HSPF is based upon the concepts of the mechanistic Stanford Watershed Model. There are three "application modules" in HSPF that address types of watercourses: PERLND and IMPLND are watershed loading models treating, respectively, pervious and impervious catchments, and RCHRES is a one-dimensional (section-mean) stream model that functions as the receiving watercourse. HSPF is fully dynamic and includes provision for continuous modeling of runoff and sediment mobilization with an array of both generic water-quality parameters, and specific coupled kinetics, including BOD-DO, P- and N-nutrients and phytoplankton interactions in the watercourse, and pesticides. An array of options is available for depicting various agricultural land treatments. One-dimensional lakes can be incorporated into the stream segmentation. The subsurface budget is modeled as a two-layer system, which can interact with the surface resource through plant function and interflow, and provision for percolation to a deep aquifer is included. (The aquifer itself is not modeled, but is treated as a sink of water.)

The watershed to be modeled is subdivided into computational catchments based upon distribution of meteorological stations and soil types, which define catchment segment "groups" each of which is assumed homogeneous in climatology and soils. Each such group is further subdivided according to "land use" classifications, which can include vegetational assemblages, agricultural cropping patters and urbanization. The boundaries of these watershed segments then define reaches of the receiving watercourse, which are modeled as completely-mixed (both laterally and longitudinally). Because the complexity of the input file structure increases geometrically with the number of such segments, the user is advised to be parsimonious in their specification.

The hydrology component of HSPF for pervious surfaces computes surface storage, infiltration flux and storage through two soil zones and two groundwater layers, one of which is active in the simulation and drives baseflow in the receiving stream, and one of which represents the deep percolation sink of water. There is also a separate storage accounting attributed to interflow to downslope segments or the receiving stream. The hydrology component for impervious surfaces is stated to include most of the accumulation and wash-off functions of SWMM and related urban runoff models.

Operation of the model is complicated, but exemplifies the problem of coding a general model for time-varying simulation. Such a simulation necessitates long time series of all of the input data streams, which will be defined for all watershed segments or (in the case of meteorology) segment groups, and for many (perhaps all) stream segments. Acquisition, re-formatting and management of these input time series files represent much of the effort of application of the model. Manipulation of input and output time-series files is controlled by a series of modules (more, in fact, than the number of *computational* modules), and the large array of options leads to a complex input-file structure. The operating module of HSPF constantly updates the state variable time series based on the user-specified time scale. This allows the user to dictate the time scale of all state variables, allowing the user the flexibility for integrating over the proper time period for any given simulation.; i.e., a flood period could be updated on a scale of hours, whereas a longer simulation can be updated on a daily basis.

The direct incorporation of a receiving water component RCHRES in HSPF offers the capability of directly linking the watershed outputs into the stream response in a seamless way. Few watershed loading models have this capability. But it can be debated whether it is preferable to having a completely external receiving watercourse model for which the user must manipulate output files to drive the receiving model. The hydraulic subroutine of HSPF, which determines the advection transport, is based upon a time-interval budget of water volume based on inflow from the above reach, user-specified outflows, and discharge to the next downstream reach. The hydraulics by which the last is computed is a user-defined relation between Q and depth. The model does not carry out this hydraulic computation, but must have the functional rule as an input for each segment reach. Even more limiting is the fact that the resolution in the receiving watercourse is the boundaries of the subwatershed segments. These will typically be several tens of kilometres in length, over which HSPF computes average concentration values. This resolution is wholly inadequate for depicting water quality variation in a stream.

Of the models reviewed here, HSPF *formally* presents probably the most capabilities for addressing the range of problems likely to be encountered in Texas TMDL determinations, at least controlled by watershed sources. While literally every process that is identified in the surface water budget corresponds to an equation in HSPF, it is difficult to judge the relation of these equations to the standard models for those processes as treated in the literature. This has greatly frustrated this review. The mechanisms employed for the key processes, such as sediment detachment, overland flow, and surface erosion, are given limited description in the model documentation (Bicknell et al., 1996), and must be dug out of the user's manual or the code algorithms. Most of the sources for the model are "gray" literature. For example, the sediment detachment and transport model references a Stanford Technical Report from the 1960's (Negev, 1967) with surface practice modeling "influenced" by the Universal Soil Loss

Equation (USLE). Many of the discussions in the user's manual (Bicknell et al., 1996) are based upon qualitative sketches of how a process "ought to work" followed by mathematical equations representing the curves in the sketches. Although this is certainly one means of developing a process model, the separate relations must be tested against measurements, which does not appear to have been accomplished for HSPF.

The net effect of all of this is a bewildering array of options for the user, requiring specification of various coefficients appearing in the numerous process equations, and which generally have limited relation to physical properties of the natural system or conventional formulations of the processes. In Release 7 of HSPF (Donigian et al., 1984), the estimated number of parameters requiring user input is over 1000. In Version 11 this must be doubled. Up to ten waterborne constituents can be modeled in a single simulation. The constituents can travel via overland flow, interflow, or groundwater flow. The user specifies which mechanisms are considered for each constituent. The interflow and groundwater transports are simple loading relationships, the product of concentration and flow, the first of which the user specifies in each watershed area. Chemical processes modeled include hydrolysis, oxidation, photolysis, biodegradation, volatilization, and sorption, all of which require coefficients and rate constants. Constituents can be transported by overland flow in different phases: dissolved or entrained in the water, or sorbed to the solids in the water column. The association of constituents to solids is based on simple relationships with sediment and water yield. The user can input partition coefficients and the kinetic rates for adsorption for each parameter involved. Or, the constituents can be proportional to sediment removal based on “potency factors,” which indicate the constituent’s strength relative to the sediment removed from the surface, and which, again, are user inputs.

As a specific example, consider the subroutine that computes sediment transport in the stream channel. HSPF uses three sediment types: clays, silts, and sands, for which suspended and bedload transports are budgeted separately. The user must partition the watershed runoff load into the three grain-size categories. The user must then select among three choices of relations for modeling deposition and scour that are extracted from an earlier modeling project at Batelle (Onishi and Wise, 1979). Again, this is an example of HSPF relying upon "gray" literature as its primary source for a process formulation. (Note that this particular report is labeled "draft", 20 years after its completion.)

This array of options not only creates a burden of operation for the user, but also poses a great danger of incorrect specification of model parameters. With so many to choose from, little guidance from literature, and no information on even relative sensitivity of response, the user can inadvertently specify combinations that produce unrealistic results, but which may be hidden in the internal calculations of the model. As an example, in a recent attempt to apply HSPF to a TMDL in Texas, hydrology parameters were adjusted in accordance with guidance (Donigian et al., 1984) and achieved very good calibration, or so it appeared from a comparison of modeled streamflows to gauged data. Upon closer inspection, however, it was discovered that most of the streamflow was being percolated to the upper soil layer then re-entering the stream channel through interflow, a completely nonsensical pathway which is not displayed to the user without explicit printout commands (J. Miertschin, pers. comm., 1999).

In conclusion, there is much in HSPF to recommend it for use in Texas TMDL's, such as its explicit detail in the hydrology and transport modules, and its well-thought-out approach to manipulating the large data files associated with a time-varying model. In its present form, however, every new application will become a major model development enterprise with a great degree of freedom for incorrect process specification. We suggest that TNRCC develop a "Texas" version of HSPF, with (1) pre-set default values appropriate to the state, (2) new subroutine codes depicting key processes based firmly on current knowledge, again appropriate to the Texas environment, and (3) a considerably simplified user interface. With respect to (2), the mechanistic relations embodied in ANSWERS, ANSWERS-2000, and WEPP, may be the most viable possibilities, since all of these have been separately subjected to field validation.

We also recommend that the receiving water component of HSPF in its present form not be used in TMDL evaluations, except for unusual circumstances such as treating a shallow run-of-the-river reservoir which has no longitudinal gradients in quality. There are two options available to TNRCC: (1) export the loading results from HSPF and use to drive a stream/river water quality model appropriate for the system of concern, (2) extend the RCHRES subroutine in HSPF to disaggregate the stream reach into a more highly resolved submodel. We believe the former to be the more practical option.

### *3.3.3 PRMS*

One additional watershed modeling system recommended for consideration is MMS/PRMS. Like HSPF, PRMS is a watershed model for the evaluation the impacts of various combinations of precipitation, climate, and land use on streamflow, sediment yields, and general basin hydrology. The model, which is a modular-design, distributed-parameter system, is developed and supported by the US Geological Survey. Basin responses to normal and extreme rainfall can be simulated to analyze the changes in water-balance relationships, flow regimes, flood peaks and volumes, soil-water relationships, sediment yields, vadose zone flow and groundwater recharge.

The concept behind the model is to divide a watershed into a series of homogeneous response units (HRUs) based on basin characteristics such as slope, aspect, elevation, vegetation type, soil type, land use and precipitation distribution. Use of HRUs was found to be more accurate than the approach used by HSPF in a study by Flugel (1995). The sum of the HRU responses, weighted by area, produces the daily system response and streamflow for a basin. Spatial variability in soil type is included in the modeling of infiltration, evaporation and transpiration from the soil, subsurface flow and other watershed processes. Vegetation cover parameters which can be specified in each HRU include vegetation cover density and the maximum storage available on this vegetation, which is a function of precipitation form (snow, winter rain or summer rain). Sediment detachment and transport is modeled using a rill-interrill concept approach (Leavesley and Stannard, 1995). Input requirements include the sediment concentration (mass/volume) and parameters controlling the rainfall detachment rate of sediment and the overland flow detachment rate of sediment.

The number of parameters is quite large for PRMS, but is certainly fewer than the number required by the HSPF model. Therefore, PRMS may be expected to be somewhat easier to use but also somewhat less general than HSPF. Moreover, HSPF and PRMS use the same data-management front end, *viz*. ANNIE to help users interactively store, retrieve, list, plot, check, and update spatial- and time-series data for hydrologic models, which in turn uses the Watershed Data Management (WDM) direct access file. At present, however, PRMS does not include a water-quality capability, nor is it set up to be coupled to a receiving stream model. The Modular Modeling System, MMS, is presently under development by USGS, and includes PRMS as a component. MMS uses a master library that contains compatible modules for simulating a variety of water, energy, and biogeochemical processes. Depending on the system, different algorithms can be chosen to model desired chemicals and responses. In addition, a GIS interface is being developed to aid in the use of the MMS (Leavesley and Stannard, 1995).

At this point, therefore, neither PRMS nor MMS can be considered a viable candidate for use in the Texas TMDL process. However, the PRMS model has many potential capabilities in its hydrological formulation that represent an advance over HSPF and SWAT, and we recommend that the development and application of the PRMS model continue to be monitored for future application.

### *3.3.4 Specialized models*

Several models were identified in this review as having specialized capabilities that might prove useful in isolated circumstances in Texas. CREAMS and EPIC are both "field-scale" agricultural models that have a range of specialized depictions of surface practices, root zone kinetics, and vegetation effects. There may be specific TMDL-related problems that would benefit from their capabilities. GLEAMS is an extension of CREAMS that better depicts the soil and root-zone behavior, especially with respect to the leaching of constituents such as nutrients and pesticides. GLEAMS may be of use in considering the impacts of CAFO's on surface and subsurface water. An example application is reported by Yoon et al. (1994) who used GLEAMS to predict nutrient (N and P) losses in surface and subsurface runoff and their concentrations in soil layers, following application of different rates of poultry litter.

The management of storm water has been a perennial concern of urban engineering, and the EPA model SWMM has a long history of development and application to address this problem.  Its potential for use in Texas TMDL projects is not considered to be high, however, precisely because it is such a specialized model.  Its complexity for dealing with the vagaries of urban plumbing would appear to make it much too detailed for the strategic-level evaluations of a TMDL.  There may of course be isolated TMDL problems in urban areas that will require the level of detail afforded by SWMM, but the modeling requirements of these problems will have to be addressed *ad hoc*.

# 4. Stream and river models

## *4.1 Role of streams and rivers*

From a hydrological viewpoint, the stream is the predominant type of receiving water in Texas in terms of length of regulated watercourse and in terms of distribution over the area of the state. The defining properties of a stream are a predominant longitudinal dimension, following a well-defined channel, in which flow is unidirectional (except for rare backwater effects), so that the direction of "upstream" is unambiguous. We consider a river to be a stream with relatively larger dimensions and a perennial flow.

For evaluation of stream water quality, the primary variation is considered to be longitudinal, and for most purposes in Texas, a cross-sectional mean is an adequate approximation of the constituent concentration at any position along the watercourse. We limit our review of TMDL candidate models to those with these properties for the modeled streams. (This means, for example, that special-purpose spatially detailed models, such as crossing-jet discharges, plume or mixing-zone models, or models with detailed vertical variation, are not part of this review. This does not mean that special circumstances somewhere in Texas might not dictate need for such a model, but simply that we do not expect this to be necessary for most TMDL's.)

The Texas environment imposes additional constraints on stream models and also obviates the need for capabilities that might be required elsewhere. Most streams in Texas occur in alluvial settings in which the stream channel configuration is determined by erosion into the landscape. There are relatively few occurrences of controls of streambeds by geological structure. This means that sinuosity is a common feature of stream channels. It also means that sedimentary processes can be important in stream dynamics, and that certain morphological features associated with alluvial systems are common in the state, including cut banks and point bars, pool and riffle environments, bank caving, and channel migration and abandonment.

The climatology of Texas exerts strong controls on streamflow. It has already been noted that the dominant source of streamflow in Texas is deep convection. Streamflow therefore tends to be flashy, with large spikes of runoff and flow separated by protracted periods of baseflow. (The state is also subject to longer-term variations from wet years to drought.) The loading associated with storm-derived runoff events can be considerable. Both the storm hydrograph and the low-baseflow condition can create different but equally important responses in the water quality of the stream. Particularly for the former, the stream model must be capable of accommodating dynamic time variation in the inputs and in the response of the stream.

On the other hand, freezes and winter storage of water in solid phases are of negligible importance in the water budget of Texas. Some stream models are designed for the protracted severe winter climates of the northern tier of states, capabilities which are unnecessary in Texas. Similarly, models that are designed to address larger, stable flows of continental-drainage rivers (the Mississippi, Columbia, St Lawrence, Susquehanna, Ohio, and so forth) have little utility in Texas.

### *4.2 Important characteristics of stream & river models*

The one-dimensional equations are obtained from equations (1) -(5) (Table 2-1) by integrating over a cross section. For freshwater streams it is conventional to assume a constant density $\rho$, so that the pressure $p$ can be eliminated in favor of water level $h(x,t)$ above some datum. The boundary stress $\tau_o$ is written as one of the standard fluid resistance terms used in open-channel hydraulics, such as the Chezy, Darcy-Weisbach or Manning formulae. The longitudinal mass flux term is usually written as a diffusive flux $F_x = - \rho E \, \partial c / \partial x$. One version of the resulting equations (assuming zero surface stress and atmospheric pressure gradients) is given in Table 4-1.

Table 4-1
Mathematical formulae for mechanistic model of one-dimensional stream
(cf. Table 2-1)

---

*momentum:*

$$\frac{\partial Q}{\partial t} + \frac{\partial uQ}{\partial x} = -gA\frac{\partial h}{\partial x} - g\overline{D}\frac{Q^2}{C^2} \tag{16}$$

$Q = uA$ = longitudinal flow
$D$ = water depth
$h$ = water surface elevation
$C$ = Chezy friction coefficient

*continuity:*

$$\frac{\partial Q}{\partial x} + B\frac{\partial h}{\partial t} = q \tag{17}$$

$B$ = surface width
$q$ = lateral inflow per unit length along channel

*conservation of mass:*

$$\frac{\partial c}{\partial t} = \frac{1}{A}\left[-\frac{\partial Qc}{\partial x} + \frac{\partial}{\partial x}EA\frac{\partial c}{\partial x}\right] + \sum S_i \tag{18}$$

$c$ = section-mean mass concentration of substance
$E$ = longitudinal dispersion coefficient
$S_i$ = source or sink of substance (including kinetics and loads)

---

The complete set of equations for a stream is comprised of both a hydrodynamic model and a mass-conservation model for concentration of constituent $c$. A separate mass-balance equation is needed for each water-quality parameter. The equation of continuity (17) is considered part of the hydrodynamic model, i.e. (16) and (17) together, and is often mathematically combined with (16) to result in a single equation for streamflow $Q$. The complete model for a constituent $c$ is an example of a feed-forward model, in which the hydrodynamic model is solved first to obtain the flow in the stream channel $Q$ (or, equivalently, the section-mean velocity), which is then supplied to the advective transport term in the mass-conservation equation (18) for $c$.

Open-channel hydraulics uses (16) and (17) as the starting point (called the St. Venant equations, see Chow et al., 1988), and employs further simplifications to obtain equations appropriate for uniform, nonuniform steady, slowly varying (quasi-steady), unsteady, and other flow configurations (e.g., Chow, 1959, Henderson, 1966). In water-resources modeling, these equations are solved by any of a number of numerical methods (e.g., Vreugdenhil, 1989) that involve discretizing the stream channel into short segments or nodes. Some models treat only a single channel, while others can accommodate a network by allowing interconnection and confluence of stream reaches.

If external data are available for streamflow $Q_o$ at the upstream end of the model reach, and this streamflow is steady, then the need for the hydrodynamic model is obviated, because

$$Q(x) = Q_o + \int q(x)\, dx$$

In point of fact, this has been true for the vast majority of applications of the one-dimensional stream model, because the most common historical problem is that of low steady flow. In Texas, establishment of permit limits for a point discharge and development of a wasteload allocation for a stream assume a constant streamflow $Q_o$ equal to the 7Q2 at some upstream gauge. Since this is an extreme low flow, almost always $q(x) = 0$, and the hydrodynamic model boils down to $Q(x) = Q_o$. The need for TMDL's to also consider time-dynamic flows means that the hydrodynamic part of the model will have to be given more careful consideration than has been traditionally the case in wasteload allocation. Even if lateral inflows $q(x)$ can be neglected, the upstream inflow will still be a strong function of time $Q_o(t)$ so the variation in flow with distance downstream will depend upon bed friction, lateral storage, and response of the water surface. Determining $Q(x,t)$ downstream from a point where $Q_o(t)$ is specified is referred to as the "routing" problem.

The bracketed terms in (18) are together the transport of substance *c*, cf. equation (5) of Table 2-1. The dispersion term includes the effects of turbulent transport and variations in *u* and *c* across the section that are correlated (see Fischer, 1973, Valentine and Wood, 1979). Like the friction coefficient (*C* in Table 4-1), there is usually no reliable means of specifying *a priori* the dispersion coefficient *E* and it will have to be established by model calibration. Monitoring the transport of a substance without sources or sinks (i.e., one that is conservative), such as a release of fluorescent dye, is the simplest means of determining *E*.

Just as various simplifications of the hydrodynamic equations are obtained by assuming away some of the terms, so also is the solution of the mass conservation equation (18) simplified. One of the more important such simplifications is how time variation is treated. The steady-state approximation is obtained by taking $\partial c/\partial t = 0$ *and* by assuming constancy in the transport, loading and kinetic terms of (18). With these assumptions, (18) reduces to an equation with only one independent variable *x*. If the stream geometry, as measured by *A* and *D* in (16), is assumed as well to have simple variation (e.g., constant with *x*) then an analytical solution may even be possible. Even without this strong an assumption, the numerical treatment of the equation is greatly simplified in the steady state.

Sediment concentration is a special case. As noted in Section 2.1.4, for practical purposes, the presence of sediment in the water is due to current velocity, either through influx or through resuspension of bed sediments. We differentiate between sources of sediment that are external to the stream channel and those that are internal, the former carried into the stream by flow, the latter mobilized and entrained by moving water. The relations of Table 3-4 apply to erosion and deposition on the stream bed. In principle, the concentration of sediment can be modeled by a version of (18) in which the sources and sinks $\sum S_i$ include the influxes of sediment from lateral sources along the channel and into and out of the water column. The latter would be sediment remobilized from the stream bed due to water movement in the channel and sediment settling out of the water column by gravity and turbulence. This works satisfactorily if the sediment concentrations are dilute and made up of silt- and clay-sized particles, a situation usually source-limited.

If the streamflow is sediment-laden, then the interactions between water velocity, particle size and physical properties, turbulent entrainment, and gravitational settling become too complex to be conveniently depicted by the model of Table 4-1. In this case, usually capacity-limited, resort may be made to quasi-empirical equations as given in Table 3-4, relating bulk sediment transport to hydraulic properties of the flow, such as the Einstein, Simons-Li, or Bagnold equations (Bagnold, 1966, Julien, 1998).

This is the fundamental difference between sediment transport in the watershed and sediment transport in the receiving stream. In the watershed environment, sediment transport is generally capacity-limited, governed by detachment mechanics and the ability of the overland flow to remove the detached sediment. In the stream channel, the sediment concentration is characteristically source-limited: because there is a greater volume of flow, the suspended sediment concentrations are governed largely by the lateral influx from watershed drainage.

### *4.3 Model summary and evaluation*

Compared to the review of watershed models, fewer stream models were examined in this project. This was mainly because of the historical success of treating the stream environment, compared to the other surface-water environments considered, which implies the existence of several well-tested, widely accepted, general-purpose models that are appropriate for addressing water quality in a stream. For TMDL purposes, the greater issue is whether these (or other) models are suitable for adaptation to the dynamic transient conditions attending storm-water influxes, that we expect to be of significance for TMDL determination in Texas. We reviewed models designed exclusively for the stream setting and more general-purpose water-quality model whose range of application includes (but is not limited to) flowing streams. None of the former, such as CE-QUAL-ICM, CE-QUAL-RIV1, or RIVER3, survived the Level-1 screen.

Five stream and river models, most of which are more generally applicable to other watercourses as well, were given detailed review. These five models were reviewed with respect to capabilities and methods for the following processes:

- Hydraulic formulation
- Sediment load and transport
- water quality parameters and response
- Model program structure and coding
- Model input requirements

The comparative capabilities of these models are summarized for each of these categories in Tables 4-2 through 4-6. As with the watershed models, this review was based upon the documentation for each model, applications reported in the literature, and (in a few cases) inspection of the computer code, but not upon set-up and operation of the individual models, which was beyond the scope of this study. (The authors of this review do have specific experience with several of these models from past projects.) Reviews of the individual models are given in Ward and Benaman (1999).

The watershed model HSPF was described in Section 3.3.2 above. It is included in this section because its receiving water module is in effect a model of stream water quality. As noted earlier, it is debatable whether combining the receiving water model with the watershed model is in fact desirable. Apart from this, the receiving watercourse model of HSPF suffers from two major problems as a stream model. First, there is no resolution in HSPF beyond the upstream and downstream boundary points of a watershed segment. This means that the model computes water quality averaged over a substantial reach of the stream. Practical water-quality determinations, however, generally require much finer spatial resolution. It would be impossible, for example, to depict the DO sag in the Trinity River downstream from Dallas using

Table 4-2
Stream and river models - hydraulic formulation

| *Model* | *Time resolution* | *hydraulics* | *friction* | *density variation* | *comments* |
|---|---|---|---|---|---|
| | | Stream models embedded within watershed models | | | |
| HSPF | variable | input table of h=f(Q) | n/a | none | Recommends use of Mannings eqn |
| | | General receiving water model | | | |
| DYNHYD | variable | 1-D St Venant eqn | Manning | none | allows reversing (upstream) flow |
| QUAL2E | steady-state | input power-law h=f(Q) | n/a | n/a | |
| QUAL-TX | steady-state | input power-law h=f(Q) | n/a | n/a | |
| WASP | variable | user input from separate hydrodynamic model | n/a | n/a | typically coupled with DYNHYD |

Table 4-3
Stream and river models - sediment transport

| *Model* | *channel sources* | *watershed sources* | *deposition* | *texture resolution* | *comments* |
|---|---|---|---|---|---|
| | | | Stream models embedded within watershed models | | |
| HSPF | scour above critical stress | TSS load | critical stress + Stokes | 3 classes | functional models are obscure |
| | | | General receiving water models | | |
| DYNHYD | no capability | | | | hydrodynamic model only, usually coupled with WASP |
| QUAL2E | none | steady TSS load | n/a | n/a | can model only as simple parameter |
| QUAL-TX | none | steady TSS load | n/a | n/a | can model only as simple parameter |
| WASP | none | TSS | none | none | would require user-defined processes |

Table 4-4
Stream and river models - water-quality capability

| *Model* | *time resolution* | *parameters* | *coupling sources* | *loads &* | *kinetics* | *comments* |
|---|---|---|---|---|---|---|
| | | Stream models embedded within watershed models | | | | |
| HSPF | fully time-varying, arbitrary timestep | organics, TSS, DO, nutrients, pesticides, temperature, metals, chlorophyll | activated in groups | point & nonpoint (surface washoff) | first-order | limited capability to de-couple variables, e.g., cannot treat BOD-DO sag problem |
| | | tracers (up to 10) | none | nonpoint + point | first order | user supplies coeffs & partitioning |
| | | General receiving water models | | | | |
| DYNHYD | fully time varying | no capability | n/a | n/a | n/a | hydrodynamic model only, usually coupled with WASP |
| QUAL2E | steady-state | organics, BOD, DO nutrients, temperature, chlorophyll | activated in groups | point sources, & steady lateral loading | first-order | includes general tracer with with first-order kinetics |
| QUAL-TX | steady-state | organics, BOD, DO, nutrients, temperature, chlorophyll | activated in groups | point sources, & steady lateral loading | first-order | includes general tracer with with first-order kinetics |
| WASP | time varying | organics, BOD, DO, | user-specified | user-specified | user-specified loadings | general transport model, but user must input (or code) the kinetic processes |

Table 4-5
Stream and river models - Model program structure and coding

| *Model* | *Source code* | *GIS linked* | *Output format* |
|---|---|---|---|
| | Stream models embedded within watershed models | | |
| HSPF | FORTRAN | no* | graphs & tables |
| | General receiving water models | | |
| DYNHYD | FORTRAN | no | tables & ASCII files |
| QUAL2E | FORTRAN | no | tables & ASCII files |
| QUAL-TX | FORTRAN | no | tables & printer images |
| WASP | FORTRAN | no | tables & ASCII files |

*The BASINS version of HSPF is linked via GIS, but has no receiving stream capability

Table 4-6
Receiving water models - Model input requirements

| *Model* | *Physio-graphy* | *water quality parameters* | *transport parameters* | *loads & sources* | *kinetics* | *comments* |
|---|---|---|---|---|---|---|
| | | | Stream models embedded within watershed models | | | |
| HSPF | x-scn areas, channel breaks at land segment boundaries | nutrients, DO, algae arbitrary tracers (up to 10) | none | watershed applications, point sources, channel sed | 1st-order rates or M-M coeffs 1st-order | M-M coeffs are user-supplied user supplies coeffs & partitioning |
| | | | General receiving water models | | | |
| DYNHYD | link-node config x-scn areas | no capability | n/a | inflows, tidal bndies | n/a | hydrodynamics only, usually coupled with WASP |
| QUAL2E | channel network, A=f(Q) rating | BOD, DO, nutrients, tracers | flow, dispersion | point, bed fluxes, steady lateral loadings | 1st-order rates | steady-state model |
| QUALTX | channel network, A=f(Q) rating | BOD, DO, nutrients, tracers | flow, dispersion | point, bed fluxes, steady lateral loadings | 1st-order rates | aeration, some rate terms specific to Texas aquatics |
| WASP | watercourse cell network, depths, areas | arbitrary | flows & "mixing" coeffs between cells | loads into each cell | 1st-order rates | transport model, user-supplied kinetics |

Key:
DO dissolved oxygen
M-M Michaelis-Menten

HSPF.  Second, HSPF has no hydrodynamic compartment: rather, the user must input a relation between $h$ and $Q$, which means that the flow response to a storm runoff event will not be accurately depicted.  Yet this is exactly the problem that the model should be able to address to serve as a satisfactory TMDL method.  Following the recommendation of Chapter 3, if a "Texas BASINS" including a version of HSPF were to be developed, one provision would have to be an accurate external calculation of stream hydraulics under dynamic unsteady conditions.

Two of the models listed in Table 4-2 are the classical one-dimensional models QUAL2E of EPA and QUAL-TX of TNRCC.  As noted in Ward and Benaman (1999), both models have a common ancestor in QUAL developed by the Texas Water Development Board in the early 1970's.  QUAL2E has more options, but QUAL-TX has several kinetic terms based upon field experiments in Texas watercourses and designed for application in the state.  Otherwise both models are equivalent.  The models solve the mass balance equation (18) of Table 4-1 by a method of finite differences, for which the watercourse is discretized along its longitudinal axis as a series of computational "elements," grouped into reaches in which the major transport terms and reaction coefficients are considered to be constant.  Branching watercourses, i.e. tributary drainage, can be depicted.  The most important aspect of these models is that they are steady state.  Because of this, the QUAL models do not include a hydrodynamic compartment, since the (steady-state) flows are supplied by the user.  Dissolved oxygen is perhaps the most important modeled constituent within QUAL2E and QUAL-TX, and most of their history of application is in addressing DO problems.  Inputs for DO include saturation concentration, rates of $O_2$ production and uptake due to algae, chemical oxidation, sediment oxygen demand rate, and reaeration rate.  QUAL-TX contains Texas-specific options for reaeration and sediment oxygen demand.

The greatest limitation of the QUAL models is, of course, their inability to depict the response of the stream to time varying inflows, such as would result from storm runoff.  For TMDL problems that are amenable to treatment by a steady or quasi-steady approximation, QUAL-TX would be the recommended vehicle.

DYNHYD is a hydrodynamic model promulgated by EPA, that addresses time-varying flows in response to inflows or water-level variations. It is a descendant of the Orlob-Shubinski estuary model of the 1960's, originally developed for San Francisco Bay, whose best-known East Coast applications are the EPA Potomac and Delaware models (see Ward and Espey, 1971). DYNHYD is a link-node hydrodynamic model simulating velocity, volume, and water depth subject to river flow and tidal phenomena. The equations of conservation of mass and energy are solved by the method of finite-differences to predict water velocities, flows, water heights, and volumes. Bed characteristics are parameterized using Manning's *n*. The model is strictly one dimensional, but it can be used to depict a two-dimensional system by use of a branching network of links and nodes. For the stream environment, DYNHYD is a well-tested model of unsteady hydraulics.

DYNHYD does not, however, include a transport capability. For this DYNHYD output is typically coupled into WASP, effectively a link-node transport-model counterpart to DYNHYD, but in fact derived from the Thomann (1963) estuary DO model of the 1960's. As a well-tested model combination capable of handling time-varying water-quality responses, the DYNHYD-WASP combination is probably the best option available to TNRCC. It is not without problems, however. The application experience of DYNHYD to dynamic storm-runoff events is limited, based upon the literature survey of Ward and Benaman (1999), and there is indication that the model may prove problematic for especially abrupt storm events common in Texas. The WASP model has the disadvantage of relying upon the user to supply much of the source/sink terms in the model.

It is surprising to us that no carefully developed, well-tested fully dynamic model of stream hydraulics and water quality emerged from the review. The closest approach is CE-QUAL-RIV1, developed by the Corps Waterways Experiment Station, but this model has received relatively limited application, and has primarily been used in evaluating hydraulic structure operation, such as the effects on water quality due to the regulation of streamflow by hydropower dams.

The State has invested many years of development and data collection in the QUAL-TX model. It appears that the most expeditious means of having a fully dynamic model appropriate for Texas systems would be to exploit the capabilities in QUAL-TX by recoding it as a time-varying model, and incorporating one-dimensional unsteady hydrodynamics into the model operation.

# 5. Lake and reservoir models

## *5.1 Role of lakes and reservoirs*

Natural lakes in Texas (discounting playas) are slightly more common than the poultry dentition of proverbial notoriety. Reservoirs, on the other hand, are numerous: there are over 200 major reservoirs (greater than 5000 ac-ft), which serve a variety of purposes including flood control, electric power cooling water, and water supply.

The key defining property of a lake that differentiates the lacustrine environment from the other watercourses addressed in this review is the long detention time entailed by the large ratio of volume of the lake to its inflow. Because of this property of detention, lakes pose the possibility of retention and accumulation of pollutants, and the resulting effects on water-quality can be more problematic than the purely flux-dominated stream or river. An additional consequence of the long detention time is the possibility of warm-season stratification due to receipt and accumulation of heat in the near-surface layers. For lakes that are sufficiently deep, the vertical decline in water temperature from the heated surface to the cool near-bottom waters takes place in a narrow zone, the thermocline, whose corresponding vertical density gradient restricts and dampens vertical mixing. If sufficiently stable, the thermocline will divide the lake into two non-exchanging layers, the epilimnion above, and the hypolimnion below.

Because of Texas' southern climatology, seasonal ice formation is not a concern for the hydrography of Texas lakes. Seasonal stratification can occur, however, but the temperature of maximum water density (4°C) is not acquired in the cooling season. Turnover in such systems does occur, but the dissolution of the thermocline is due to diminished insolation combined with increasing surface heat loss and increasingly intense vertical mixing caused by fall frontal passages. Whether stratification occurs in a reservoir, and how stable the vertical temperature gradient is, both depend upon the depth of the reservoir, its location in the State (i.e., the regional climate), and the degree of topographic sheltering. Reservoirs with a significant throughflow due to dam releases may have their stratification potential further disrupted by these releases.

Also, lakes with large internal circulations, notably power-plant cooling reservoirs, can have a considerably disrupted stratification due to the heat load and the associated mechanical mixing.

There is a surprising range in vertical stratification in Texas reservoirs. The summer thermocline may be comparatively shallow, such as the 5-6 m thermocline depth in Toledo Bend, or quite deep, such as the 20-m depth in Amistad. Stability is also variable: Lake LBJ, for example, has weak vertical temperature gradients, even in late summer, while Toledo Bend has a sharp thermocline that even supports internal waves. Some reservoirs relatively close together can exhibit very different stratifications (Sam Rayburn compared to Toledo Bend, for instance). As a general rule, the only reliable means for establishing the stratification characteristics of a reservoir in Texas is by measurement.

Stratification concerns also apply to other parameters. Many of these have their principal source at the surface of the lake, e.g., oxygen, and the vertical distribution is dictated by turbulent flux, which varies inversely as the vertical gradient in density (i.e., temperature). Thus as the seasonal stratification develops, the flux of oxygen to deeper levels of the lake is diminished, and the concentration of DO in these deeper waters drops as it is consumed by aerobic organisms, notably bacteria. In this case, there is a seasonal oxycline that forms, generally in or above the thermocline layer. An example is shown in Fig.5-1 from Lake Buchanan, which is weakly stratified in the summer.

Most of Texas' major lakes are run-of-the-river throughflow systems, so that a substantial proportion of the volume of the lake is replaced by inflows during an average year. (Some reservoirs are constructed on minor tributaries and rarely spill, being made-up by pumpage from nearby watercourses.) This throughflow character combined with the flashy hydrometeorology implies the potential for highly time-varying loads to the lake, but the detention is a mechanical long-term average, and many lake water-quality problems acquire their limiting conditions in the low-flow summer season. The fact that some of these lakes serve as a drinking-water source means that the applicable criteria for some parameters will be more stringent than normally the case for other water-use categories. A cursory inspection of the current 303(d) list for Texas indicates about 30 reservoirs listed, for problems ranging from low DO to excessive metals.

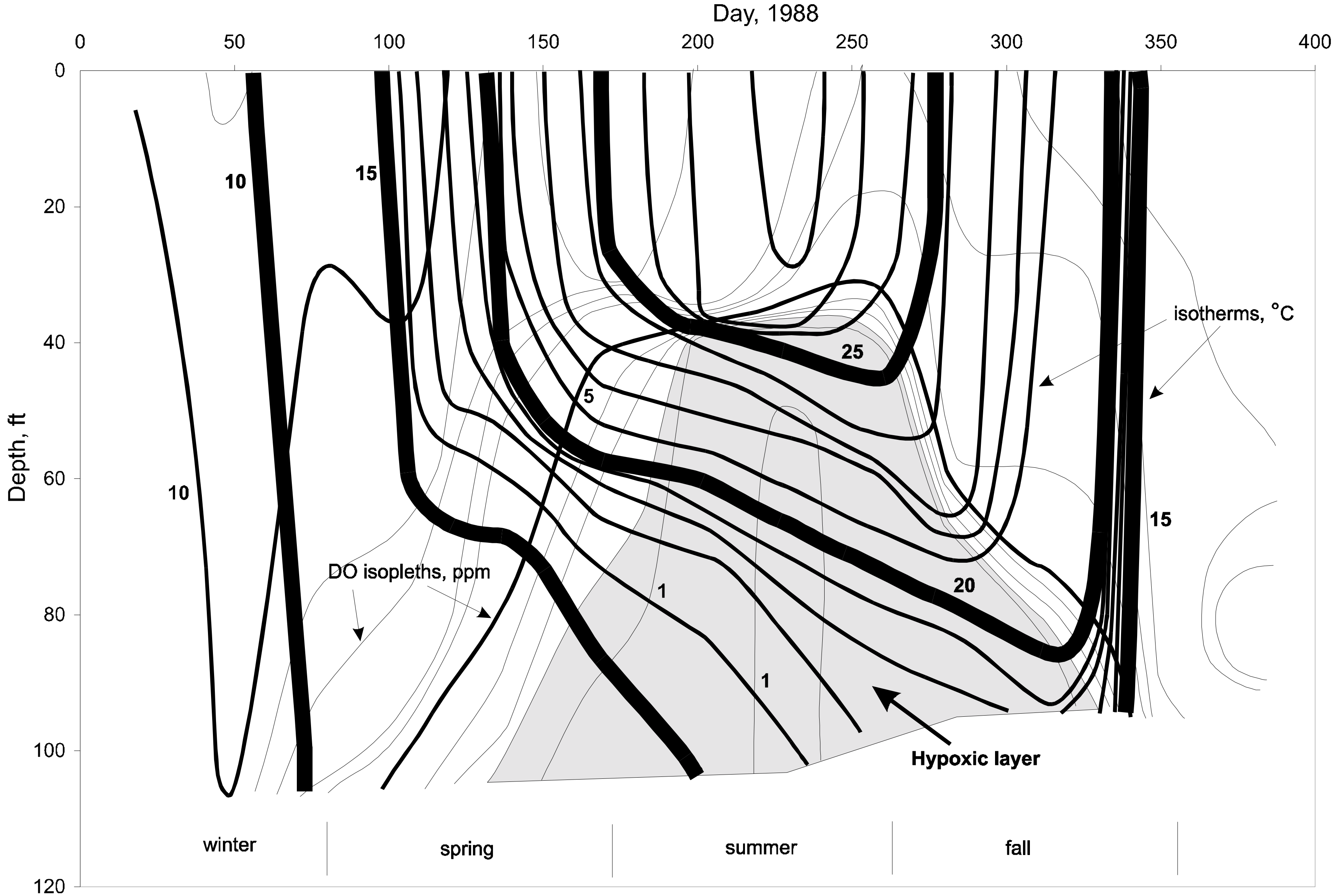


Figure 5-1 Temperature and DO structure in Lake Buchanan, 1988. (Data courtesy of Lower Colorado River Authority.)

### *5.2 Important characteristics of lake and reservoir models*

As with other watercourse models, one of the primary features of a lake model is the degree of spatial simplification created by averaging in the model. This must be determined by the variability in the morphology of the lake, and the associated variation in water-quality parameters. The occurrence of systematic or consistent poor water quality or criteria violations in specific geographical regions of a reservoir, such as one of the tributary arms, or in the epilimnion near the dam, may dictate the need for a model that resolves that spatial variation.

The geometrically simplest lake model treats the lake as a well-mixed homogeneous volume of water with inflow and outflow. This is the Continuously Stirred Tank Reactor (CSTR) depiction of the lake, and is appropriate either for reservoirs that are indeed shallow and well-mixed, or for rather qualitative "bulk" analyses for which volume-mean values of the water-quality parameters provide adequate detail. Trophic state is frequently compared between reservoirs, as an index to eutrophication, based upon CSTR analysis. Also, the general response of a small reservoir to a time signal of loading (e.g., seasonal influx) can be estimated by treating the reservoir as a CSTR. The CSTR is essentially a zero-dimensional model, since no spatial parameter variation is depicted.

While it is certainly possible that the CSTR may be an appropriate model for determining a TMDL for some reservoirs in Texas, in most cases some spatial variation will have to be resolved. It seems likely that one of the following will be necessary:

- horizontally averaged, preserving vertical variation
- cross-section averaged, preserving longitudinal variation
- laterally averaged, preserving longitudinal and vertical variation

The equations for these types of models, in a generic form, are given in Table 5-1. The first type of model, (19) and (20) in Table 5-1, would be applied to reservoirs that are deep enough that seasonal stratification develops, and are broad and horizontally well mixed so that there is little significant variation in any horizontal direction. The most important diagnostic of the

Table 5-1
Mathematical formulae for spatially averaged models of lake and reservoir transport

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***Surface (horizontal)-mean model:***

*conservation of heat:*

$$\rho c_p \frac{\partial \overline{T}}{\partial t} = -\frac{\partial}{\partial z} flux_z \left\{\frac{\partial T}{\partial z}\right\} + \sum H_z \qquad (19)$$

*conservation of mass:*

$$\frac{\partial \overline{c}}{\partial t} = -\frac{1}{\rho}\frac{\partial}{\partial z} flux_z \left\{\frac{\partial c}{\partial z}\right\} + \sum S_i \qquad (20)$$

***Section-mean model:***

*momentum:*

$$\frac{\partial Q}{\partial t} + \frac{\partial uQ}{\partial x} = -gA\frac{\partial h}{\partial x} - gh\frac{Q|Q|}{C^2} \qquad (21)$$

*conservation of mass and heat:*

$$\frac{\partial c}{\partial t} = \frac{1}{A}\left[-\frac{\partial Qc}{\partial x} + disp\right] + \sum S_i \qquad (22)$$

***Lateral mean model:***

*momentum:*

$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + w\frac{\partial u}{\partial z} = -g\frac{\partial h}{\partial x} + \frac{1}{\rho}\nabla(stress) + (disp) \qquad (23)$$

*conservation of heat:*

$$\frac{\partial T}{\partial t} + u\frac{\partial T}{\partial x} + w\frac{\partial T}{\partial z} = -\frac{1}{\rho}\nabla flux_z \left\{\frac{\partial T}{\partial z}\right\} + (disp) + \frac{1}{\rho c_p}\sum H_z \qquad (24)$$

*conservation of mass:*

$$\frac{\partial c}{\partial t} + u\frac{\partial c}{\partial x} + w\frac{\partial c}{\partial z} = -\frac{1}{\rho}\nabla flux_z \left\{\frac{\partial T}{\partial z}\right\} + \sum S_i \qquad (25)$$

$T$ = temperature
$h$ = elevation of water surface above bottom
$c_p$ = specific heat
$flux_z$ = turbulent flux at level z
$\rho$ = density, a function of temperature $T$
$H_z$ = thermodynamic heat flux

---

applicability of this type of model is the lack of horizontal slope in the thermocline or oxycline. In (19) and (20), the functional dependence of the vertical flux $flux_Z$ on vertical temperature gradient $\partial T/\partial z$ is indicated in the notation. This is the greatest hydrodynamic complexity in this type of model, and for its accurate formulation requires explicit treatment of stability effects on turbulence.

A simplification of (20) is the two-layer model, which treats the average parameter values above and below the thermocline. The depth of the thermocline is determined by either a separate (perhaps empirical) model, or by a parameterized mixed-layer model. This two-layer model is intermediate in complexity between the CSTR and the complete vertical variation depicted in (20). From the standpoint of modeling average constituent concentrations in the epilimnion, this is, in effect, the CSTR with a specified flux as a lower boundary condition (at the thermocline, rather than the lake bottom).

The second type of model, (21) and (22) in Table 5-1, would be applicable to a system whose stratification is negligible, but in which substantial variation is found along the horizontal longitudinal axis. Many of the dendritic reservoirs of Texas have a prominent horizontal dimension and are shallow enough that seasonal stratification is unimportant, for which this can be an appropriate model. A slight variation may necessitate a dendritic network with junction points, to depict the effect of conflowing tributaries. The equations of this model, it will be noted, are formally the same as those for the section-mean stream, (16) in Table 4-1. Often the value of Q is small or can be established by external conditions (such as the circulating flow of a power-plant, or release rules for the dam), in which case (21) is obviated, and the model becomes simply (22).

The last model, (23) - (25) in Table 5-1, is the most complex, requiring detailed spatial geometry and well as complicated hydrodynamics and transport terms. Such a model may be necessary for long, deep reservoirs, such as Lake Travis, Texoma, Falcon, and similar systems. The vertical-flux terms can be particularly complex since, like the horizontal-surface-mean model, depiction of stability effects on turbulence is needed. Nor is the complexity limited to the transport processes. Interaction of vertical distributions of nutrients with light penetration is fundamental to phytoplankton productivity, and may have to be considered for hyperstimulated ("eutrophic") systems. For those reservoirs in which significant seasonal stratification occurs with a de-oxygenated hypolimnion, the kinetics can become complicated because of the change of oxidation states in the vertical. For some water quality parameters, however, it may be sufficient to consider only the distribution of values in the epilimnion.

Loads enter a lake through stream and tributary flows and from the peripheral drainage around the lake body. Backwater effects of the reservoir impoundment extend many miles up the channels in inflowing rivers and tributaries, especially in the low-gradient topography of much of Texas. Backwater interferes with the ability to accurately measure streamflow in the field. This, and the paucity of streamflow gauges on minor tributaries to a reservoir may imply the need for modeling runoff for a large proportion of the reservoir watershed. Landscape-derived loads would be equally difficult to measure, and modeling would be mandated for these as well.

### *5.3 Model summary and evaluation*

The reservoir system, like the estuary to be addressed in the next section, was considered to have several unique features so as to require a special-purpose model. These requirements include treatment of stratification, seasonal structural variation, effects of release works and operating rules on water quality, and special water-quality considerations deriving from the long-detention characteristics of a lake. It was surprising that few general-purpose lake and reservoir models seem to exist, especially given widespread concern with the quality of the lake environment. We can only surmise the reason for this, but suggest that it may be that the long-detention feature has dominated the modeling philosophy for lakes. This feature means that lakes are long-term integrators, and their quality aspects can often be treated on a volume-mean (i.e., zero-dimensional) basis, e.g. the CSTR or the two-layer system. Those reservoirs in which spatial variation is important tend to be dendritic systems with a prominent longitudinal dimension, so that an adaptation of a one-dimensional through-flow model can be suitable for most problems. Thus, general-purpose models such as QUAL2E and WASP have found occasional application to the reservoir setting. (Also, part of the reason for the poor selection of reservoir models may be a general lack of appreciation for the hydrodynamic aspects of reservoir water quality.)

For TMDL purposes, though Texas reservoirs receive storm-water influxes, apart from exceptional storms, these tend to be trapped and substantially detained, if not retained, by the large volume of the reservoir. It is not clear how important dynamic response will be to TMDL determinations for reservoirs. In the model review, we considered both the long-term quasi-steady models as well as models with a shorter-time response capability. Based upon the nature of problems stated in the current Texas 303(d) list, it would appear that long-term trends and low-flow summer conditions comprise the most frequent requirements. There is a potential for each of these to be indirectly affected by storm runoff and associated loads, even if not directly, in that such loads would represent an influx of contaminants that could exert a water-quality impact much later when the system stratifies or the throughflow diminishes.

Four general-purpose hydrodynamic or transport models, and one stratified lake model emerged from the screening procedures (Table 2-2) and were given detailed review. The former were DYNHYD, QUAL2E, QUAL-TX and WASP. DYNHYD is strictly a hydrodynamic model, but is frequently coupled with WASP: for application to a reservoir, both would be needed. The stratified lake model reviewed was CE-QUAL-W2. In addition, HSPF was considered, because it purports to include the capability for modeling a run-of-the-river reservoir. These models were reviewed with respect to capabilities and methods for the following processes:

- Hydrodynamic formulation
- Sediment loading and accumulation
- water quality capability
- Model program structure and coding
- Model input requirements

The comparative capabilities of these models are summarized for each of these categories in Tables 5-2 through 5-6. As with the watershed and stream/river models considered earlier, this review was based upon the documentation for each model, applications reported in the literature,

Table 5-2
Lake and reservoir models - hydrodynamic formulation

| *Model* | *Time resolution* | *dam operation capability* | *friction* | *stratification* | *density variation* | *comments* |
|---|---|---|---|---|---|---|
| | | | <u>General receiving water models</u> | | | |
| HSPF | time varying | limited | Chezy | none | none | reservoir treated as single well-mixed segment |
| DYNHYD | time varying | none | Mannings *n* | none | none | not presently set-up to model a dam junction, re-coding necessary |
| QUAL2E | steady-state | none | none | none | none | no hydrodynamic capability |
| QUAL-TX | steady-state | none | none | none | none | no hydrodynamic capability |
| WASP | steady-state | none | none | none | none | no hydrodynamic capability, usually coupled with DYNHYD |
| | | | <u>Receiving Water Models: lakes & reservoirs</u> | | | |
| CE-QUAL-W2 | time varying | general | quadratic stress | yes | yes | two-dimensional, vertical-longitudinal |

Table 5-3
Lake and reservoir models - sediment loading and accumulation

| *Model* | *watershed sources* | *deposition/ resuspension* | *texture resolution* | *comments* |
|---|---|---|---|---|
| | | General receiving water models | | |
| HSPF | yes | critical stress + Stokes settling | 3 classes | reservoir treated as single well-mixed segment |
| DYNHYD | n/a | n/a | n/a | hydrodynamics only, no transport modeling capability |
| QUAL2E | steady lateral & upstream | none | arbitrary | no explicit capability, user would treat as conservative or nonconservative tracer |
| QUAL-TX | steady lateral & upstream | none | arbitrary | ditto |
| WASP | yes | none | arbitrary | no explicit capability, user would treat as conservative or nonconservative tracer |
| | | Receiving water models: lakes & reservoirs | | |
| CE-QUAL-W2 | | no explicit provision | | re-coding would be necessary |

Table 5-4
Lake and reservoir models - water-quality capability

| *Model* | *Time resolution* | *parameters* | *loads & sources* | *kinetics* | *comments* |
|---|---|---|---|---|---|
| | | General receiving water models | | | |
| HSPF | time varying | nutrients, DO, tracers | watershed, point source | first-order, user input | mainly used for ag runoff-type problems |
| DYNHYD | time varying | none | n/a | n/a | hydrodynamics only, see WASP |
| QUAL2E | steady state | DO, BOD, nutrients | point source, steady lateral | conventional & 1st-order | 1-D model only |
| QUALTX | steady state | DO, BOD, nutrients | point source, steady lateral | conventional & 1st-order | 1-D model only, Texas-based relations |
| WASP | time varying | DO, temp, nutrients, tracers | point source, user-specified loadings at each cell | user-specified | heavily dependent upon user input or re-coding |
| | | Receiving water models: lakes & reservoirs | | | |
| CE-QUAL -W2 | time varying | DO, temp., algae, nutrients | point source & trib inflows | conventional | 2-D horizontal & vertical, mainly used in eutrophication problems |

Table 5-5
Lake and reservoir models - Model program structure and coding

| *Model* | *Source code* | *GIS linked* | *Output format* | *Comments* |
|---|---|---|---|---|
| | | General receiving water models | | |
| HSPF | FORTRAN | no* | graphs & tables | Time series of volume-mean concentrations |
| DYNHYD | FORTRAN | no | tables & ASCII files | Will require some re-coding for reservoir |
| QUAL2E | FORTRAN | no | tables & ASCII files | Concentration profile with distance along reservoir axis |
| QUAL-TX | FORTRAN | no | tables & printer line images | ditto |
| WASP | FORTRAN | no | tables & ASCII files | |
| | | Receiving water models: lakes & reservoirs | | |
| CE-QUAL-W2 | FORTRAN | no | graphs & tables | |

*The BASINS version of HSPF is linked via GIS, but has no receiving stream capability

Table 5-6
Lake and reservoir models - Model input requirements

| *Model* | *Physio-graphy* | *water quality parameters* | *transport parameters* | *loads & sources* | *kinetics* | *comments* |
|---|---|---|---|---|---|---|
| | | | General receiving water models | | | |
| HSPF | | — same as Table 4.6 — | | | | treats reservoir as CSTR |
| DYNHYD | | — same as Table 4.6 — | | | | hydrodynamics only, usually coupled with WASP |
| QUAL2E | | — same as Table 4.6 — | | | | steady-state model, little application history to reservoirs |
| QUALTX | | — same as Table 4.6 — | | | | no example application to reservoir available to this review |
| WASP | | — same as Table 4.6 — | | | | transport model, user-supplied kinetics |
| | | | Receiving water models: lakes & reservoirs | | | |
| CE-QUAL-W2 | depths & A = f(h) | nutrients, DO, temperature | none (internally computed) | point sources & trib inflows | conventional, 1st order, M-M, & many others | climatological data for heat budget also needed |

Key: CSTR - continuously stirred tank reactor DO - dissolved oxygen M-M - Michaelis-Menten and (in a few cases) inspection of the computer code, but not upon set-up and operation of the individual models, which was beyond the scope of this study. (The authors of this review do have specific experience with several of these models from past projects, however.) Reviews of the individual models are given in Ward and Benaman (1999).

It was somewhat surprising that so few candidate models appropriate for addressing water quality of lakes emerged from the screening. Of the models screened out, see Table 2-2, CE-QUAL-ICM is applicable to one-dimensional systems, uses the nutrient budget kinetics of the Corps Chesapeake Bay model, but has a limited history of application. EUTROMOD, PHOSMOD and BATHTUB are all rather dated, and have a limited application history. IDOR is much more current but is proprietary. HSPF has can depict a lake as a single, long, well-mixed receiving water segment, as a part of its RCHRES module, but lacks many of the kinetic capabilities necessary to depict the lake environment, and cannot treat the reservoir in isolation (as would be desirable for investigating summer low-flow conditions).

We speculate that one reason for the dearth of lake models exploiting modern computational capabilities is that many lake problems have been addressed by CSTR-type models on an *ad hoc* basis, but these models are not formalized and promulgated as general-operation computer programs. Another possible reason is the re-direction of the federal concern with deteriorating lake quality to controlling loads from the watershed, hence the shift of activity from the EPA Clean Lakes program to the Section 319 projects. A model structured along the lines of BATHTUB, perhaps with a two-layer capability, would be of great potential value to many reservoir problems in Texas. (The greatest deficiency of BATHTUB *per se* is its statistical character.)

# 6. Estuary and bay models

## *6.1 Role of estuaries and bays*

The coastal zone of Texas is one of the major surface-water resources of the State, and has been the recipient of extensive research, monitoring, and regulatory development. The jewels in the Texas coastal diadem are its bays, from Sabine Lake on the Louisiana border to the Lower Laguna Madre, almost at the international boundary of Mexico. These are examples of estuaries, and exhibit the diversity and productivity implied by that category of watercourse. The traditional, but imprecise, definition of an estuary is a waterbody in which seawater and freshwater intermix. What is important is that estuaries are transitional between freshwater and marine systems, and therefore are influenced by both terrestrial and oceanic processes. Ward and Montague (1996) comment:

> Estuaries contain some of the most productive areas of the world, with respect to both ecology and economics. This is not a coincidence. At the coastal land-water interface, natural subsidies of production accrue to both nature and humanity. For humanity, the land-water interface is advantageous for development, affording access both to the sea and the interior. While this statement is true for the coastline in general, the estuary affords a special attraction, its sheltered morphology offering a natural protective harbor, with access to freshwater supplies in its upper reaches, and a bounty of fish and game. Many of the great population centers of the world are established on estuarine harbors. … Estuaries subsidize heavy industry. Costs of transporting raw materials and finished goods over water are much less than those of other forms of transportation, and the sheltered land-water interface of an estuary is a natural transfer zone. Heavy industry is also subsidized by the large volume of water available in estuaries, as a source of process water and for dilution of wastes. In modern society, the estuary function of a rich food supply remains important. The land-water interface is the base for the commercial fishing industry, and is especially intimately connected to the estuaries.

This statement is made about estuaries in general, but it applies exactly to the coastal bays of Texas.

Unanimity among workers in estuary processes is lacking on the precise definition of an estuary, but most definitions include the following characteristics (Ward and Montague, 1996):

- coastal waterbody
- semi-enclosed
- free connection to open sea
- influx of sea water
- freshwater influx
- small to intermediate "scale"

(The last characteristic differentiates an estuary from larger systems, such as the Gulf of Mexico or Mediterranean Sea, which possess the other properties.) These features suggest some of the complexity of an estuary. For example, an estuary is governed by hydrographic processes that are both riverine and marine, e.g. floods and tides, respectively. But there are also hydrographic processes that are unique to the estuarine environment, consequences of the interaction of marine and riverine influences within the semi-enclosed morphology of the estuary.

The principal hydrographic features of an estuary are: (1) morphology and bathymetry, (2) hydrology, (3) tides, (4) meteorology, (5) density currents. Estuaries tend to be broad, well-circulated systems. This is certainly true of the Texas bays, which are shallow compared to their surface area and therefore are particularly responsive to wind forcing. The bays are connected to the sea through narrow inlets through barrier islands except for Sabine Pass whose inlet threads through a chenier plain. The estuary mouth, or inlet to the sea, is one of the fundamental morphological controls, since it determines the exchange with the sea. Littoral sand supply, riverine sediment loads, and internal re-working of sediments further establish patterns of shoal areas that are sculpted and shaped by waves and currents.

The principal terrestrial control on an estuary is the inflowing river. As noted in Section 1.2, the deep-convective origin of precipitation in Texas leads to river flows that are flashy. This is transmitted to the Texas bays. The timing and volume of the seasonal freshets are the most important hydrometeorological feature of these systems. Inflow affects the hydrography of the estuary by establishing a gradient of salinity across the system, and further influences water quality by its associated influx of constituents of terrestrial origin, frequently including anthropogenic wasteloads. There is a gradient in climatology across the watersheds of Texas from the humid east to the arid west that translates to a companion gradient in hydroclimatology of the bays. The bays on the upper coast tend to be bimodal in annual inflow with maxima in the spring and fall, while the bays on the lower coast have more widely spaced inflow pulses and are frequently hypersaline during protracted low-flow periods.

The tide is, of course, the most obvious marine influence on estuary hydrography. The tide in the Gulf of Mexico has been described as "bush league" (Ward, 1997). To a first approximation, this tide is the superposition of semidiurnal and diurnal components with a 13.6-day variation (arising from the cycle of lunar declination), modulated by a secular seasonal rise and fall in water level with low waters in winter and summer. The tidal range is about a meter at great declination and less than 15 cm at zero declination. As the tide propagates through the inlets into a bay, the semidiurnal component is substantially reduced, and the diurnal component attenuated.

Because of the broad, shallow morphology of the Texas bays, in conjunction with the dramatic coastal weather variations, they tend to be responsive to meteorological forcing, of which the wind is the most important factor. Suddenly varying winds can induce "wind tides" by effecting a tilt in the water surface across the estuary and an abrupt water-level differential between the estuary and the adjacent sea. The Texas bays are additionally subject to the variety of meteorological conditions peculiar to the coastal zone, including tropical storms and the seabreeze (Ward, 1997).

The density current is perhaps the least obvious and most poorly understood aspect of estuary hydrography, but it is a basic element of the circulation. This is the current generated by a horizontal difference in density. In an estuary, this arises from the horizontal salinity gradient, the more saline water being denser than the fresher water. Essentially, the density current is the flow of denser water displacing lighter water, but modulated by mixing processes and the shape of the estuary. In a longitudinal estuary, this is manifested as a mean circulation directed upstream in the lower layer and downstream in the upper. In broad, shallow bays, the density current can be manifested as a flow directed upstream in the deeper sections of the estuary, compensated by a seaward return flow in the shallower sections. The density current is the prime vehicle for salinity intrusion and often establishes a dynamic equilibrium with the steady freshwater inflow from the river. The density current is governed by the horizontal gradient in salinity. The flow in the density current circulation is nominally an order of magnitude *greater* than the fresh-water inflow that maintains the salinity gradient (Ward and Montague, 1996), so it is clearly a major factor in the transport processes operating in the estuary.

All of these hydrographic factors interact to produce an environment with intense turbulence and mixing, and prominent internal circulations. These affect the transport of constituents and the establishment of constituent concentrations, and represent a degree of complexity that is rarely equaled in inland watercourses.

The importance of the Texas bays, and their potential for impacts due to human activities, including contamination, have led to their occupying a central position in Texas environmental management and regulation. Moreover, the range of estuarine watercourses on the Texas coast has necessitated a variety of analysis and modeling capabilities. These types of estuaries include, most notably:

- tidal reach of a river
- salinity intrusion reach of a river
- small, shallow embayment
- distributary network of a delta
- deep navigation channel
- broad, major bay system

The question of which specific modeling capabilities will be needed to treat TMDL's for the Texas estuaries depends on the type of estuary that is involved, addressed in the next section, but also on the nature of the problem requiring a TMDL. In the current 303(d) listing for Texas the only estuary systems listed for violation of dissolved oxygen standards are tidal/salinity-intrusion reaches of streams or deep landlocked navigation channels such as the Houston Ship Channel and Inner Harbor of Corpus Christi. The one exception to this statement is Oso Bay in the Corpus Christi system. Virtually all of the bay segments of Galveston Bay are on the current 303(d) list. However, the basis for these listings are one or more of: (i) fish consumption advisories for dioxin, (ii) closure of oyster reefs, (iii) elevated metals (mercury or copper), (iv) bacteria levels in excess of contact recreation standard. East Matagorda Bay and Cedar Lakes, all of the secondary bays of the Matagorda system, Nueces Bay and Oso Bay in the Corpus Christi system, and sections of San Antonio, Aransas, Copano, Corpus Christi Bay and the Laguna Madre are listed for oyster reef closure. In addition, Lavaca Bay and Cox Bay are listed for tissue contamination by mercury, described as "residual from historic sources," presently the subject of a Superfund project.

### *6.2 Important characteristics of estuary models*

Estuaries have a special place in the development and application of water-quality models in the U.S. While models for streams were the first to be developed for routine water-quality evaluation, the estuary environment was addressed almost as early, and over the years has received the most intense modeling study of any type of watercourse (e.g., from a quarter-century ago, Ward and Espey, 1971). Several reasons motivated the early and intensive development of models for estuaries. The consequences of pollution on the estuarine environment are perceived as pressing because of the variety of economically and ecologically important biota. Contaminant impacts are also more visible than rivers or reservoirs, due to intense recreational use of estuaries. The wasteloads to estuaries are greater than most inland systems because they receive the cumulative wasteloads from upstream, and also because urban and industrial development tends to concentrate on the estuary periphery.

As noted above, the estuary is much more complex than freshwater systems, being affected by both marine and terrestrial factors, but also by processes unique to the estuarine environment itself, such as the salinity-driven density current. The principal sources of this complexity are: (1) variation in salinity, (2) extreme variation with time, (3) irregular morphology. Salinity variation equates to density variation. Streams and rivers are usually treated as constant density systems, so this complexity does not arise. Deep lakes can exhibit vertical stratification in density because of differential heat absorption. Unlike the vertically stratified lake, in an estuary the principal gradient in density is horizontal, from the sea to the land. As noted above, this produces horizontal accelerations that drive autonomous internal circulations. Moreover, the density gradient (as is the case in a stratified lake) significantly alters the vertical turbulent flux of mass. Extreme variation with time is intrinsic to the estuarine environment. Unlike the case with rivers or lakes, in the estuary there is never a steady state. Sources of time variation include freshets in the inflows and meteorological forcing. But even if these are not operating, the tide always is, and it prevents occurrence of a true steady state.

All of these features can be dealt with by applying a fully three-dimensional time-varying density-coupled model, such as the generic model of Table 2-1 in which density is assumed to vary and thereby enter the pressure gradient term. Such models exist, and two (POM and EFDC) are reviewed in the next section. However, the computational resources required and the difficulties in completely specifying such a model have made more simplified treatments desirable. Following the strategies of model simplification outlined in Section 2.2, geometrical complexity is reduced by integrating over one or more spatial dimensions. The most common such models are:

- laterally averaged, preserving longitudinal and vertical variation
- vertically averaged, preserving horizontal variation
- cross-section averaged, preserving longitudinal variation

Table 6-1 displays the types of equations that result from such spatial averaging. The laterally averaged model is applied to deep systems with significant variation in the vertical, such as a ship channel. The vertically averaged model is applied to systems with significant horizontal variation but which are well-mixed in the vertical, such as broad shallow bays. The cross-section averaged model is applied to systems in which the most important variation is along the main, longitudinal axis of the estuary, such as tidal rivers.

These model equations are different in two important ways from, e.g., Table 4-1. First, these are no longer feed-forward equations, in which the hydrodynamics are solved first, then the currents used to determine transport in a mass-balance equation, as is done in the models of Tables 2-1 or 4-1. Rather these are coupled equations, that must be solved simultaneously. Second, the dispersion terms are of a different character. Any systematic variation in the current velocity and

Table 6-1
Mathematical formulae for spatially averaged models of estuary transport

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***Lateral mean model:***

*momentum:*

$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + w\frac{\partial u}{\partial z} = -\frac{g}{\rho(s)}\int_z^h \frac{\partial \rho(s)}{\partial x}\,dz - g\frac{\partial h}{\partial x} + \frac{1}{\rho}\nabla\,(stress)_x + (disp) \tag{26}$$

*conservation of salt:*

$$\frac{\partial s}{\partial t} + u\frac{\partial s}{\partial x} + w\frac{\partial s}{\partial z} = -\frac{1}{\rho}\nabla flux_z(u, \frac{\partial s}{\partial z}) + (disp) \tag{27}$$

*conservation of mass:*

$$\frac{\partial c}{\partial t} + u\frac{\partial c}{\partial x} + w\frac{\partial c}{\partial z} = -\frac{1}{\rho}\nabla flux_z(u, \frac{\partial s}{\partial z}) + \sum S_i \tag{28}$$

***Vertical-mean model:***

*momentum:*

$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} + v\frac{\partial u}{\partial y} = -\frac{gh}{\bar{\rho}(s)}\frac{\partial \rho(s)}{\partial x} - g\frac{\partial h}{\partial x} + \frac{1}{\rho h}(\tau_h - \tau_o) + (disp) \tag{29}$$

*conservation of salt:*

$$\frac{\partial s}{\partial t} + u\frac{\partial s}{\partial x} + v\frac{\partial s}{\partial y} = -\frac{1}{\rho h}F_h + (disp) \tag{30}$$

*conservation of mass:*

$$\frac{\partial c}{\partial t} + u\frac{\partial c}{\partial x} + v\frac{\partial c}{\partial y} = -\frac{1}{\rho h}(F_h - F_o) + \sum S_i \tag{31}$$

(continued)

---

Table 6-1
(continued)

---

***Section-mean model:***

*momentum:*

$$\frac{\partial Q}{\partial t} + \frac{\partial uQ}{\partial x} = -gA\frac{\partial h}{\partial x} - gh\frac{Q|Q|}{C^2} + disp \tag{32}$$

*conservation of salt:*

$$\frac{\partial s}{\partial t} = \frac{1}{A}\left[-\frac{\partial Qs}{\partial x} + disp\right] + \frac{F_h}{\rho h} \tag{33}$$

*conservation of mass:*

$$\frac{\partial c}{\partial t} = \frac{1}{A}\left[-\frac{\partial Qc}{\partial x} + disp\right] + \sum S_i \tag{34}$$

$s$ = salinity
$h$ = elevation of water surface above bottom
$F_z$ = mass flux of substance at level z
$\tau_z$ = stress in horizontal on plane at level z
$\rho$ = density, a function of salinity

---

the constituent (*s* or *c*) in the direction over which the average is performed produces a net flux term involving the average of the product of these variations, which looks superficially like the traditional Reynolds flux term in turbulence, but has a completely different origin. This net flux is also referred to as "dispersion," shown generically (as *disp*) in Table 6-1, in analogy to the product-departure terms in the constant-density stream model, but the systematic variation arises from density-driven circulations. When both lateral and vertical variations are averaged out, as is done for the section-mean model, equations (32)-(34), the dispersion terms can become quite large.

As they stand, these equations of Table 6-1 still contain full time-variation. The time variability can be handled as in other systems, by re-casting the model as a long-term average. In the case of the estuary, the averaging period has to be at least one tidal cycle (24.8 hours for Texas systems), usually several such cycles. Longer periods may be needed to average out perturbations from meteorology and inflow. If the controlling factors are sufficiently steady and the average is sufficiently long, a useful "tidal-mean" steady state may be constructed. Again, however, systematic time variations in current and constituent concentration that are correlated (as will be the case for tidal variations) produce net flux terms in the averaged model, referred to as before as "dispersion" but originating from variation quite different than that for a unidirectional stream.

The dispersion terms in a space-averaged, time-averaged estuary model are often written in a diffusive flux form analogous to (18) of Table 4-1, but with a dispersion coefficient *E* that is much larger than would be employed in a constant-density unidirectional stream or river. Ward and Montague (1996) tabulate some example estuary dispersion coefficients, whose values are several hundred $m^2/s$. Clearly, dispersion is responsible for much of the flux in estuarine transport. There is no theoretical basis for expecting its mathematical form to be diffusive, so the dispersion term may also represent a source of error in the model. For a constituent that is highly reactive, its kinetic terms will account for more of its variation in concentration than transport, and it may be sufficiently accurate to use a highly averaged model with dispersive transport. Dissolved oxygen can usually be modeled by such an expedient. For a waterborne parameter that is more conservative, and whose distribution is therefore dependent more on transport, such averaging can be a significant source of error. When feasible, it is better to explicitly depict time- and space-variation in the model for such a parameter. The expediency of using such artifices as dispersion coefficients is one of the reasons for requiring extensive model validation in estuaries.

Models have been applied in Texas estuaries to determine the response of temperature gradients in the bays due to power-plant returns, the effects of deep navigation channels on salinity and sediment, the concentrations of pollutants resulting from waste discharges, and the effects of upstream reservoir construction and freshwater diversion on the estuary ecosystem. Despite this history of activity, there are many aspects of contaminant transport and transformation in the Texas bays that have not been addressed by past models, in part because of the problem-specific nature of these past modeling efforts. Table 6-2 summarizes the principal estuary systems in Texas and the *minimal* estuary model necessary to adequately model their quality.

In summary, the estuary system exhibits several unique features that require models specially formulated to depict these features. These include density stratification, tides, salinity intrusion, and large (and irregular) spatial dimensions. Estuaries continue to stimulate model application, and it is safe to say that the most complex and sophisticated watercourse models have been developed for specific application to estuaries. Some of these, including models with a history of use in Texas, are considered in the next section.

### *6.3 Model summary and evaluation*

Compared to the situation for lakes and reservoirs, a number of candidate models for estuaries emerged from the screening process, Table 2-2. In general, for each of the dimensional categories of Table 6-2, there is at least one model represented in those reviewed. Moreover, there also exist several three-dimensional models applicable to estuaries, of which two are considered in this review.

Table 6-2
Types of Texas estuaries and minimal transport model capabilities

| *physical system* | | | *model depiction* | | |
|---|---|---|---|---|---|
| *type* | *time variation* | *spatial variation* | *time resolution* | *dimensionality* | *transport* |
| tidal river | long-term average | longitudinal variation | tidal-mean | 1-D | dispersive |
| | short-term | longitudinal | dynamic | 1-D | |
| river saline reach | long-term average | longitudinal variation | tidal-mean | 1-D | dispersive |
| deltaic distributaries | time-varying | longitudinal | dynamic | 1-D | network w/ flats storage |
| small bay | long-term average | longitudinal variation | tidal-mean | 1-D | dispersive |
| | long-term average | transverse gradients | tidal-mean | 2-D horizontal | dispersive |
| | time varying | transverse gradients | dynamic | 2-D horizontal | dispersive |
| navigation channel | long-term average | longitudinal variation | tidal-mean | 1-D | dispersive |
| | long-term average | vertical-longitudinal | tidal-mean | 2-D incl. vertical | gravity circulation |
| major embayment | time-varying | transverse gradients | dynamic | 2-D horizontal | dispersive |

Four general-purpose transport models, and four estuary-specific models were given detailed review. These eight models were reviewed with respect to capabilities and methods for the following processes:

- Hydrodynamic formulation
- Sediment loading and accumulation
- water quality capability
- Model program structure and coding
- Model input requirements

The comparative capabilities of these models are summarized for each of these categories in Tables 6-3 through 6-7. As with models for the previous watercourse types, this review was based upon the documentation for each model, applications reported in the literature, and (in a few cases) inspection of the computer code, but not upon set-up and operation of the individual models, which was beyond the scope of this study. (The authors of this review do have specific experience with several of these models from past projects.) Detailed reviews of the individual models are given in Ward and Benaman (1999).

General-purpose receiving water models such as QUAL2E, QUAL-TX and WASP have found frequent application to the estuary setting. Both QUAL2E and QUAL-TX are steady-state models and are therefore applicable only to tidal-mean conditions in an estuary under steady inflow and loadings. However, many of the water-quality management problems in Texas estuaries can be satisfactorily addressed under these conditions. DYNHYD is strictly a hydrodynamic model but is usually coupled with WASP, which has no hydrodynamic capability and for which the user must supply currents and dispersion coefficients. (Although DYNHYD is applicable to a variety of watercourses, it was originally billed as a dynamic estuary model, since it was derived from the old link-node bay model of Orlob and Shubinski.)

CE-QUAL-W2 is a laterally averaged hydrodynamic and water quality model, designed for application to watercourses with prominent longitudinal variation that are deep enough for density stratification to be important (Cole, 1994). It was developed from a model of

Table 6-3
Estuary and bay models - hydrodynamic formulation

| *Model* | *Time resolution* | *space resolution* | *bed stress* | *density variation* | *comments* |
|---|---|---|---|---|---|
| | | | <u>General receiving water models</u> | | |
| DYNHYD | dynamic | 1-D horiz | Mannings n | none | longitudinal (section-mean) model |
| QUAL2E | steady state | 1-D horiz | n/a | none | hydrodynamic inputs: dispersion & Q |
| QUAL-TX | steady state | 1-D horiz | n/a | none | hydrodynamic inputs: dispersion & Q |
| WASP | dynamic* | user-specified | n/a | n/a | no hydrodynamic capability: user inputs in transport terms |
| | | | <u>Estuary models</u> | | |
| CE-QUAL-W2 | dynamic† | 2-D vert | Chezy | S & T | applied to deep longitudinal estuaries with density (gravitational) circulation |
| EFDC | dynamic‡ | 3-D | log BL | S & T | Mellor-Yamada turbulence closure |
| POM | dynamic‡ | 3-D | log BL | S & T | Mellor-Yamada turbulence closure |
| TXBLEND | dynamic | 2-D horiz | Mannings *n* | S | finite element solution |

* long-term tidal mean only † tidal or tidal-mean ‡ mode splitting into barotropic & baroclinic variations

Key: log BL - logarithmic boundary layer above bed
Q - throughflow
S & T - salinity and temperature part of model solution

Table 6-4
Estuary and bay models - sediment transport

| *Model* | *channel sources* | *watershed sources* | *deposition/ resuspension* | *texture resolution* | *comments* |
|---|---|---|---|---|---|
| | | General receiving water models | | | |
| DYNHYD | | — no capability — | | | hydrodynamics only |
| QUAL2E | | same as Table 4-3 | | | |
| QUAL-TX | | same as Table 4-3 | | | |
| WASP | | same as Table 4-3 | | | |
| | | Estuary models | | | |
| CE-QUAL-W2 | | — no capability — | | | would require re-coding to include |
| EFDC | yes | trib inflows | both cohesive & noncohesive | user-defined | broad capability for sediment budget |
| POM | | — no capability — | | | S & T only |
| TXBLEND | | — no capability — | | | Version 3.0 will include a limited TSS capability (deposition w/o resuspension) |

Key: S & T - salinity and temperature
TSS - total suspended solids

Table 6-5
Estuary and bay models - water-quality capability

| *Model* | *Time resolution* | *parameters* | *loads & sources* | *kinetics* | *comments* |
|---|---|---|---|---|---|
| | | | General receiving water models | | |
| DYNHYD | dynamic | | — no capability — | | hydrodynamic model only, usually coupled to WASP |
| QUAL2E | steady state | DO, nutrients, salinity | PS & steady lateral loads | standard 1st-order | transport usually calibrated to salinity |
| QUAL-TX | steady state | DO, nutrients, salinity | PS & steady lateral loads | standard 1st-order | transport usually calibrated to salinity |
| WASP | dynamic | DO, nutrients, user-specified | user-specified at each cell | user-specified | extensive user input & re-coding may be necessary |
| | | | Estuary models | | |
| CE-QUAL-W2 | dynamic | nutrients, DO, salinity, temp | PS & trib inflows | standard 1st-order | probably most useful as long-term tidal-mean model |
| EFDC | dynamic | N, organics | point/NP | complex | Adaptation of WES Chesapeake Bay model |
| POM | dynamic | | — no capability — | | transport of S&T only |
| TXBLEND | dynamic | salinity | n/a | n/a | calibrated by "Big G", (an array of element corrections) |

Key:
- DO - dissolved oxygen
- N - nitrogen
- PS - point source
- S&T - salinity & temperature
- WES - Waterways Experiment Station

Table 6-6
Estuary and bay models - Model program structure and coding

| *Model* | *Source code* | *Spatial scheme* | *GIS linked* | *Output format* |
|---|---|---|---|---|
| | | General receiving water models | | |
| DYNHYD | FORTRAN | link-node | no | tables, ASCII files |
| QUAL2E | FORTRAN | 1-D branching | no | tables, ASCII files |
| QUAL-TX | FORTRAN | 1-D branching | no | tables & printer line images |
| WASP | FORTRAN | linked cells | no | tables, ASCII files |
| | | Estuary models | | |
| CE-QUAL-W2 | FORTRAN | FD grid | no | ASCII files, variety of graphical outputs |
| EFDC | FORTRAN | FD curvilinear | no | tables, files, variety of graphical outputs |
| POM | FORTRAN | FD curvilinear | no | tables, files* |
| TXBLEND | | FE 2-D | | files, graphical output* |

*User supplies visualization software

Key: FD - finite difference
FE - finite element

Table 6-7
Estuary and bay models - Model input requirements

| *Model* | *Physio-graphy* | *transport parameters* | *loads & sources* | *kinetics* | *comments* |
|---|---|---|---|---|---|
| | | | General receiving water models | | |
| DYNHYD | link-node network, depths, lengths, widths | friction parameters | tides at mouth, diversions, inflows | n/a | hydrodynamics only, coupled with WASP |
| QUAL2E | 1-D segmentation, areas = f(Q), depths | dispersion coeffs. | steady point & lateral influxes | classical BOD-DO rate coeff., N,P 1st order rates | steady-state, applicable only to long-term tidal mean |
| QUAL-TX | 1-D segmentation, areas = f(Q), depths | dispersion coeffs. | point sources, steady lateral in-fluxes | ditto above, Texas SOD and Ka rates | steady-state, applicable only to long-term tidal mean |
| WASP | segmentation net-work, depths | dispersion (mixing) coeffs. | time-history of loads at each cell | user-specified rate coeffs. | transport only, currents must be imported from separate model, typically DYNHYD |
| | | | Estuary models | | |
| CE-QUAL-W2 | 2-D grid structure, widths, bathymetry | friction (Chezy coeff), horizontal diffusion coeff. | tides/head & salnity at mouth, point loads & trib inflows | nutrient kinetics, DO coeffs (e.g., Ka), radiation terms | tidal or long-term tidal-mean operation, density currents modeled internally |
| EFDC | curvilinear grid, depths | roughness (zo), horizontal diffusion | tides & salinity at mouth, heat budget, loads at each grid | nutrient & DO rate coeffs, radiation terms, | |
| POM | curvilinear grid, depths | roughness (zo), horizontal diffusion | tides & salinity at mouth, heat budget | radiation terms | no water quality capability besides salinity & temp |
| TXBLEND | FE grid, depths | Mannings *n*, horiz disp., Big G array | tides & salinity at mouth | n/a | no water quality capability salinity & temperature |

temperature-structure in a power-plant cooling water reservoir created by John Edinger and Ed Buchak, which they later adapted to the density-current circulation in a longitudinal estuary. The Corps Waterways Experiment Station has further developed the model to include detailed nutrient and oxygen budgets. Despite the "user friendly" objective of the structured, commented code and the substantial users manual, model set-up and execution are difficult. The WES website offers a "word of caution to the first time user," that model application is a complicated and time consuming task. Furthermore, the model has had relatively few applications in the recent literature, most of which have been to lakes, not estuaries—despite its being extant for almost 20 years—and many parts of the code have not been adequately tested.

TxBLEND evolved from a model developed by Gray and Lynch in the 1970's (see Lynch and Gray, 1979, Gray, 1987) for application to tidally dominated circulations of shallow coastal embayments. For the past two decades, the Texas Water Development Board (TWDB) has invested a considerable effort in the expansion and validation of TxBLEND for application to the Texas bays. The primary objective of the modeling work has been the capability for salinity prediction, a parameter judged to be central in evaluating the effect of freshwater inflows on the ecology and productivity of these bays. The formulation and operation of the model is summarized in the draft user's manual (Matsumoto, 1999). Briefly, the model is a numerical solution to both a hydrodynamic and a mass-transport equation, the latter being specifically applied to salinity. These equations are integrated in the vertical so the vertical dimension is eliminated, and the model calculations are for the two-dimensional circulation. While the original Gray-Lynch model focused on tidally dominated environments, the TWDB has incorporated horizontal salinity gradients into the hydrodynamics and coupled the mass-balance solution for salinity. Numerical integration is effected by the method of finite elements, using a tiling of triangular elements. The model equations also include dispersion coefficients, additional diffusive-type viscosities (to control nonlinear instability) and an empirical parameter "big G," which is in effect a calibration parameter whose value must be specified for every computational element in the model domain. Moreover, different inflow regimes require different Big G arrays. The model has been criticized for this over-reliance on empiricism.

In many respects, TxBLEND is an attractive alternative for TMDL modeling where this would be necessitated in a Texas Bay. It has been designed for specific application to the Texas systems, and finite-element input grids are already available for each of the Texas estuarine systems, except for the Laguna Madre. The vertical-mean geometry, i.e., two-dimensional horizontal, is suitable for most water-quality distribution issues in the Texas bays, because of the extreme shallowness of these systems and the lack of significant vertical gradients in concentration. TxBLEND would be especially appropriate for TMDL problems in which tidal action is the predominant hydrodynamic control, since this is the type of dynamics for which the model is eminently suited. On the other hand, in its present form, TxBLEND does not include a water-quality module. It does have a mass-transport capability, but this is limited at present to salinity. No kinetics specific to traditional water-quality parameters have been incorporated into the model. There is no capability for wasteload injection, either point or nonpoint. Perhaps the best use that could be made of TxBLEND in the TMDL process is to output the current field into a suitable mass transport and kinetics model such as WASP.

One other type of model is represented in the candidates of Table 6-2. This is the "new family" of very general, hydrodynamically based three-dimensional coastal models that have begun appearing within the last decade, a consequence of the great strides in computing power and the hunger for academic dissertation topics. Two representatives were reviewed in this project, POM and EFDC. Certainly, the best established of this "new family" is popular POM (Blumberg & Mellor, 1987), which has been applied in dozens of estuary and coastal settings and for which an active users group is established on the Internet. EFDC (Hamrick, 1996) is more recent and is promulgated by EPA as a candidate model for TMDL determinations is estuary situations. The principal author of EFDC, John Hamrick, is presently employed at Tetra Tech, Inc., which is involved in several TMDL projects and developed BASINS for the USEPA. A setting where EFDC was selected for a TMDL project involving three-dimensional modeling is South Puget Sound (Cusimano, 1999). In this case EFDC was chosen by the Washington State Department of Ecology because it is in the public domain and is broad in scope, including hydrodynamics, sediment transport and nutrient cycling.

Both models solve the governing hydrodynamic equations, *viz*. momentum and volume conservation equations. In addition, both use the Mellor-Yamada level 2.5 turbulence closure scheme to compute vertical mixing coefficients (eddy viscosity and eddy diffusivity). In both models orthogonal curvilinear horizontal coordinates are used and sigma (stretched) coordinates are employed in the vertical. The use of orthogonal curvilinear horizontal coordinates, as opposed to rectangular grid cells, allows the user some flexibility in generating model grids to fit the boundaries of the waterbody. The numerical methods of EFDC and POM are similar, both employing the method of finite differences, and both models use a mode-splitting technique in which the depth-averaged currents are solved in the "external" (barotropic) mode and vertical shears are computed in the "internal" (baroclinic) mode.

The principal differences between EFDC and POM are:

(1) EFDC incorporates a mass-transport submodel, so that constituent distributions can be obtained as part of the model run; POM is strictly a hydrodynamic/salinity/temperature model;
(2) The model boundary specifications are more general and allow a wider range of options than POM. EFDC can depict river/floodplain regions and peripheral shallow marshes or tidal mudflats that are exposed and inundated on the tide cycle;
(3) I/O routines are incorporated into EFDC to facilitate grid generation and to display model results; external software must be used for these purposes with POM.

Other differences between EFDC and POM include a greater range of options in the numerical solution of the equations for the former, and (in principal) more precise, higher-order procedures. EFDC can also output transport fields for input to independent water quality models. The model presently has an option to be coupled with WASP5 and CE-QUAL-ICM (Hamrick, 1996), in that hydrodynamic output files can be generated already in the format for input into these water-quality models. These and other options, though extraneous to most simulations, can be highly useful or even necessary for some specific projects. However, they also significantly increase the complexity of preparing model inputs and increase the probability that incorrect or incompatible options will be chosen by model users. Moreover, these increase the computational demands for executing the program, even if most of the options are disabled. To give some idea of the degree of complexity of this model, the user's manual for EFDC has 133 pages of text describing only model inputs and outputs (Hamrick, 1996). POM is also complex to set up and operate, relying upon user-supplied FORTRAN code for much of the input structure.

Although EFDC is becoming associated with TMDL projects, it is not well established in the academic/research environment. Unlike virtually all of the recently developed and widely used three-dimensional hydrodynamic models, the numerical methods and their coding in a solution algorithm of EFDC have not been published in peer-reviewed literature but instead in "gray" literature. More significantly, there is a rather sparse history of application of this model. Programming flaws, omissions in the development and analysis of a numerical method, and failures of process formulations are frequently disclosed as a model receives wide application by a variety of users and in a variety of applications, all of which are promoted by publication in the peer-review process. (This is why a history of application and acceptance by competent modelers is among the screening criteria for this review.) For this reason, there is a certain amount of risk entailed by adopting a new model such as EFDC since it cannot be expected to be as reliable as models which have been applied by a variety of users and repeatedly documented in peer-reviewed literature.

The greatest liability for adopting either POM or EFDC for TMDL determinations in Texas is the great complexity of the models and the requisite expertise of the user. While in principle the sophistication of these models in time resolution, dynamics and spatial geometry would allow the model to subsume all of the categories of Table 6-2, the effort in learning and implementing either of these models for a TMDL determination in a Texas estuary, in our view, prohibits their use unless otherwise unavoidable (and no clear TMDL problem has emerged yet that would require such sophistication). On the other hand, it may be desirable to implement a special project with the objective of training model users and setting up one of these models for general application in Texas, from which TMDL determination, as well as other problems of estuary management, would benefit.

## 7. Integrated modeling systems

Comprehensive management of a water resource will probably entail the operation of several models whose results are interdependent. This may arise from the sequential effect of one compartment on another (see Section 2.1.4), for example hydrodynamics affecting transport of a substance, whose concentration is further affected by kinetic processes, which in turn are controlled by other water-quality parameters. It may arise from the sequential effect of different components of the hydrological cycle (see Section 2.1.1), such as watershed-derived loadings injected into a stream or lake. In addition, the modeling process will require comparison of measurements, perhaps subjected to various statistical analyses, to the predictions of models, which may in turn dictate adjustments in the operation of the models. All of this necessitates the manipulation and display of potentially large files of information, including model input and output, and field measurements and their statistics. The user interface is designed to facilitate this process.

Since the inception of computer-based watercourse modeling over 40 years ago, there has been a continuous effort to make computer models easier to set up and execute. With the advent of distributed personal computers, the continuing development of faster processors, and the open availability of high-level programming technology, user interfaces to models have become commonplace. A major investment of research and programming activity has been dedicated to developing graphical user interfaces (GUIs) in order to assist the user in developing modeling input files, running the model, and coupling models together. The database management tools available in GIS combined with its powerful visualization capabilities make it an obvious choice for assisting in environmental modeling.

Because the TNRCC desires to exploit the advantages of GIS within its TMDL program, this chapter primarily concentrates on the implementation of model coupling and GUIs within GIS, referred to here as an integrated modeling system. Of course, many modeling GUIs exist in DOS, Windows, and UNIX platforms, without GIS. The objectives of this project specifically focus on GIS GUI's, especially those embodied in Environmental Systems Research Institute (ESRI) software Arc-View or Arc-Info.

The design and operation of a GIS GUI depends upon (1) the methods by which different watercourse models are coupled, (2) the capabilities for access and manipulation of environmental data, and (3) the degree of integration of the GUI software with the modeling and data management programs. While it is true that with a properly designed GUI, these aspects will be "transparent to the user," nonetheless their design and implementation will govern the capabilities of the interface or impose limitations on its use.

## *7.1 Functional integration*

### *7.1.1 Model coupling*

Examples of model coupling have already been presented in preceding chapters, e.g., the coupling between runoff produced by a watershed model and the inflows of a receiving stream model, or between a hydrodynamic model and a water-quality transport model, in which the output from the former is velocities that are used in the advective terms of the latter. Another example is using the output from a watershed loading model as the boundary condition of chemical loading for a receiving waterbody model to calculate water quality conditions. Some models have this coupling intrinsically incorporated into their program. For example, HSPF is primarily a watershed model, but it also has a receiving water component built into its coding for water quality calculations. Other—and rather common— instances of model coupling are the manual transporting of an output file from one model into the input stream of another, in which the user typically reformats the output file to agree with the input requirements of the second model. Application of WASP is typically carried out in this way.

Model coupling, especially of the manual type, occurs on environmental projects which involve the analysis of more than one system. Recently, a full-scale modeling effort of the Hudson River in New York was completed by QEA (1999). This project modeled hydrodynamics, sediment transport, chemical fate (i.e. water quality), and bioaccumulation, see Figure 7-1. The five models developed for this study were manually coupled in a feed-forward process through direct coding in the FORTRAN model programs. Although neither a user interface nor GIS was directly used, the practices of this analysis exemplify the basic procedures underlying model coupling. The hydrodynamic model provided flows and velocities to the sediment transport water quality model, the sediment transport output (as well as the flow and velocity fields) was used to define the movement of solids in the water quality model, and the chemical concentrations determined from the water quality model were fed into the bioaccumulation model to ultimately determine the chemical concentrations in fish (QEA, 1999).

When model coupling is used, it can either be "loose" as in the Hudson River example above, in which the file transports are accomplished directly by the user, or it may be "tight" in which the data transfers are accomplished by the computer programs without the intervention of the user. Of course, the limit of tight coupling is when the program code directly combines two or more models. A classic example of this is the BOD-DO problem in a stream, in which the model solves the longitudinal distribution of BOD and uses this in the kinetic terms of the solution for DO. Another example is the coupling of temperature structure, reservoir circulation and DO concentrations in CE-QUAL-W2. A similar example is the coupling of hydrodynamics and salinity distribution in the estuary models EFDC and POM. In these cases, the coupling of temperature or salinity with the hydrodynamics is one of mutual feedback: circulation establishes the variation of temperature or salinity, hence density, but the variation of density is one of the accelerations in the hydrodynamics. Coupling also occurs between model watercourses and/or compartments. In HSPF, the computed runoff and chemical loads from the watershed submodel are immediately input into the mass balance calculation of the receiving reach submodel. In the case of the receiving water submodel of HSPF and the DO submodel of CE-QUAL-W2, the coupling is of a feedforward nature, hence is one more of convenience to the user rather than dictated by the basic physics.

The more important type of tight model coupling in the present context is the linking of two autonomous models through a code that seamlessly creates input for one from the output of the other. One example of this type of "tight" model coupling is EPA's BASINS. This model

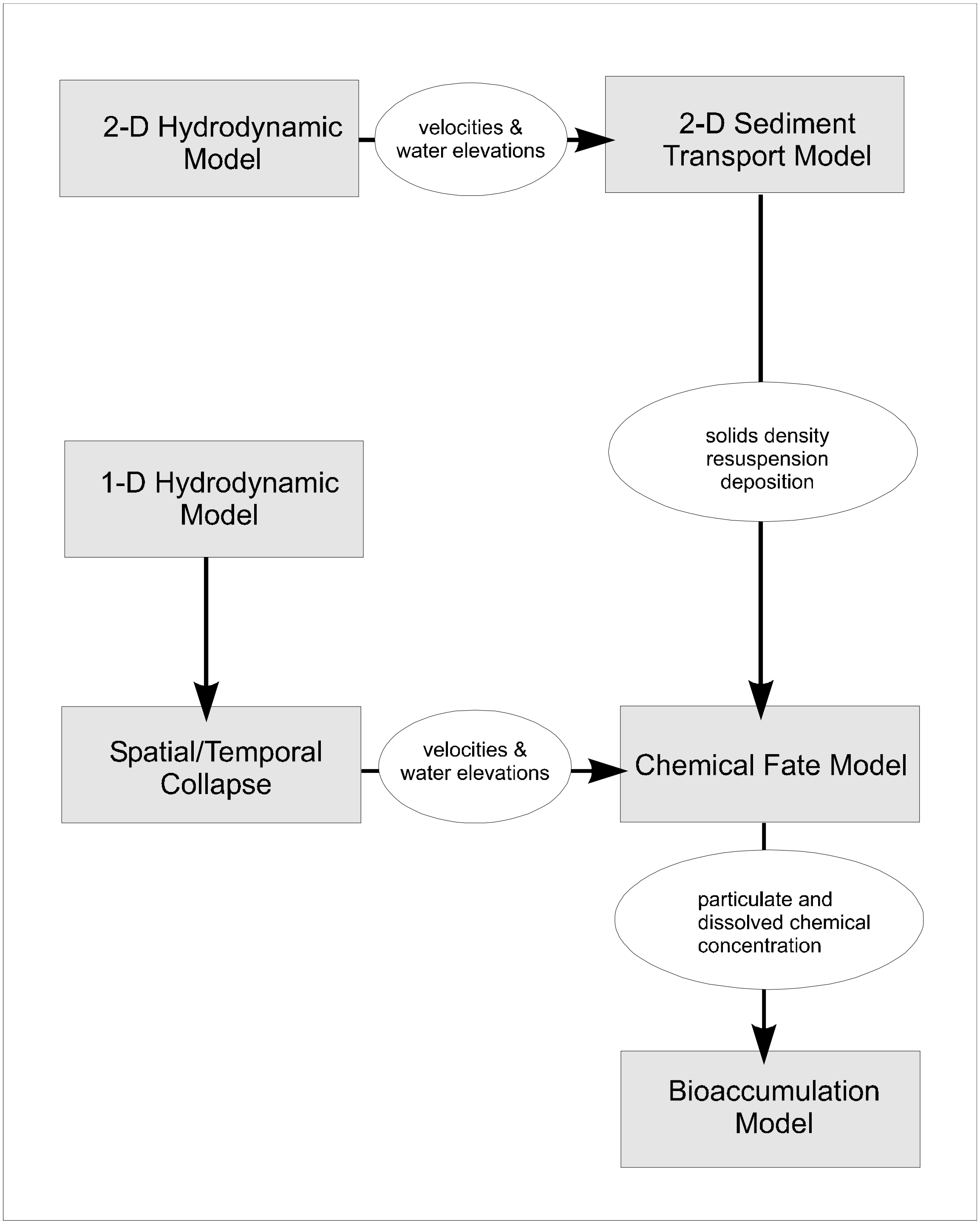


Figure 7-1. Example of forward feed model coupling performed manually on a recent modeling project for the Hudson River, New York (QEA, 1999).

"shell" uses output from a watershed loading model as direct input to a number of receiving water quality models (EPA, 1998, see Section 7.2 below). Another example is the Integrated Watershed Management Model (IWWM) which couples equations from a hydrologic model to simulate flows, infiltration, and other hydrologic processes with a water quality component that determines nutrient cycling and other constituents within the environment (Chen, *et al.*, 1996 and Chen, *et al*., 1995). In both of these instances, the model coupling is carried out by a model interface so that information needed from one simulation to the next is formatted by the program, not the user. It is this type of tight coupling via user interfaces, incorporating more than one model, that is of primary interest to the TNRCC.

A practical problem associated with this type of model coupling is that of maintenance – how can model coupling be accomplished and maintained as the component models are upgraded and changed? The answer to this question is in part dependent upon the planning, foresight and design effort invested by the programmers in the coupling interface. As would be expected, in most cases, programming sophistication is needed to create a seamless connection between two different models – the program has to successfully read output from the first model, know what part of that output is necessary for the second model, and format this output, with any other additional needed data, into input files. While current capabilities of present-day computers and programming languages certainly permit this type of integrated connection, it should be noted that environmental models tend to be rather data-intensive. As a result, developers may find that much "data accounting" is necessary to create satisfactory input files, which usually leads to the implementation of a comprehensive database manager (e.g., GIS). Additionally, as models are revised and upgraded, the coupling program will need to be updated in order to maintain the connection. New versions of a model do not always result in major changes with their input format or data. (For example, the changes made in EPA's WASP code from version 4 to 5 were minor, relative to the input file structure.) However, occasionally new versions of a model entail major changes in their input structure, necessitating major modifications to the coupling code.

### *7.1.2 Data management*

A Geographical Information System is, by definition, a data management system, one that exploits the georeferencing of environmental data in its analysis and display features. GIS incorporates a broad capability for data importation and structuring, and can utilize all of the standard file formats employed by database managers. Since water resource modeling is a data-intensive enterprise, especially in input-file development, as well as calibration and verification, the innate data-management capabilities of GIS can be a particularly useful adjunct to model operation. For that reason, it is tempting to incorporate data-management capabilities into an integrated modeling system.

It is useful to distinguish between an archival data base and an analytical data base (e.g., Ward and Armstrong, 1997). An *archival* data base is intended to preserve the measurements of a data-collection program, maximizing the information retained, without modifying or corrupting the data in any way. This includes compiling all ancillary data, employing no units conversions, and not pre-processing the basic measurements in any way (such as depth-compositing, time averaging, interpolating to standard space-time intervals, substituting values for measurements below detection limits, etc.). An *analytical* data base, in contrast, manipulates the basic measurements, however necessary to facilitate the desired analyses. This may include averaging, smoothing, subsampling the basic data, or combining the measured data with measurements from programs of other agencies.

In the modern world of digital technology, all data-collection programs should have an archival data base. Examples include the U.S. Geological Survey STORET data base, the EMAP data base maintained by EPA, and the coastal fisheries data bases of Texas Parks and Wildlife. Many agencies will also require various analyses based upon combined and processed data. When an agency attempts to have a single data base serve both objectives, conflicts arise which can limit the utility of the data. (Such a situation has arisen with the TNRCC Statewide Monitoring Network data base, the resulting problems documented in Ward and Armstrong, 1997.)

Detailing of the structure and function of data base management is a diversion from the subject of this report. Suffice it for the present context to observe that the data resource of a GIS-based GUI for an integrated modeling system is a type of analytical data base. The capabilities of the GIS for analysis and display, including data importation and merging, greatly strengthen the model support of the system, but should not be employed as well as an archival data base manager.

### *7.1.3 GIS-based user interfaces*

To actually list all of the model user interfaces that have been developed in the last decade would be daunting. Even if we confined the discussion to just interfaces developed within GIS, the list would still be too extensive for this review to include all of the recent applications. Instead, this section will present a brief overview of some selected GUIs that have been developed and applied in watershed/water quality modeling.

As enumerated by Tim and Jolly (1994), three levels of integration are possible with GIS. Most modelers who utilize GIS in environmental analysis apply Level-1 integration, which is an *ad hoc* connection, where model input developed within GIS is manually entered into the input files of a model. After the model is run, the output is reformatted and imported back to GIS for visualization. At this level, GIS is not truly a user interface, but merely a tool to aid in model development and execution. Levels 2 (partial) and 3 (fully integrated) utilize GIS as more of a true GUI, where the software is actually developed beyond it's original state to produce a tool which communicates with the process model – either internally or externally. These types of connections have become more and more popular in current practice, in part because they are "transparent" to the user.

The University of Buffalo and Limno-Tech, Inc. developed one of the first GIS-model connections with the ESRI software ArcInfo and EPA's WASP model in the early 90's. This interface, called Geo-WAMS, links a watershed loading model with a modified version of the WASP4 subprogram, EUTRO4, which models nutrients, algae production and eutrophication. The interface automates various model development tasks such as spatial and temporal analysis of watershed data, model input configuration, model input data editing and conversion, model processing, data transfer between models, and model calibration. The entire GUI incorporates the watershed loading and water quality models; a database management system; model-data linkage and assistance tools; application tools for calibration, sensitivity analysis and diagnostics; and an output manipulator for querying and visualization – all within a GIS program. This GUI was successfully applied to the Buffalo River to model dissolved oxygen (DePinto, *et al.,* 1996).

The GEO-Wams GUI is an example of a partial, or Level-2, GIS-model connection. Another example of a partial connection is a hydrodynamic/pollutant transport model recently developed at McMaster University (Boyle and Tsanis, 1998). Although this modeling system has limited application (see review of IDOR in Ward and Benaman, 1999), it still illustrates the basic concepts in user interface design and model coupling. The GIS-based GUI combines a two-dimensional, depth-averaged hydrodynamic pollutant transport model, IDOR$^{2D}$ with ESRI's ArcView GIS through the GIS programming language *Avenue*. The ArcView overlay is used for data management, model input generation, model execution, and output display (Boyle and Tsanis, 1998). Also under development at the university is a GIS coupling to IDOR$^{2D}$'s three-dimensional counterpart, IDOR$^{3D}$ (Tsanis and Boyle, 1998).

Besides new models, well-established models have also been linked to GIS for use in a GUI. This includes the USDA's Agricultural Non-Point Source Pollution (AGNPS) model and SWAT. A recent undertaking connected these two models with a ArcInfo-based user interface (Fulcher, *et al.*, 1996). Using ArcInfo's Arc Macro Language (aml), the two models were coupled and partially integrated into a GUI, which generated the input files, executed the models, and displayed the final output. Another partial connection is illustrated through the USGS's Modular Modeling System (MMS),which enables a user to selectively couple the "most appropriate" process algorithms from various models and create an "optimal" model for the desired area of study. The interface has components for the three stages in model application, pre-processing, model execution, and post-processing, that aid the users in developing their own customized model with algorithms from PRMS and various other watershed/water quality models. The UNIX-based framework, which includes a GIS interface, also allows modelers to import and develop their own algorithms (Leavesley, *et al.*, 1996). GIS is used in MMS to delineate watersheds, develop spatial model parameters, generate input files, and analyze model output (Leavesley, *et al.*, 1996).

Finally, Level-3, i.e. nearly full, or complete, connections have been accomplished in GIS user interfaces through the programming of the basic process equations necessary for the desired simulation within the GIS software itself. Examples of this type of GUI include EPA's BASINS (Lahlou et al., 1998) and the recently developed Watershed Analysis Risk Management Framework (WARMF – Chen et al., 1999). BASINS is not technically a complete connection, as the executable for some of the models or model subroutines are still intact within the interface, but the program can be viewed as fully integrated because of the format in which the user interacts with the models (see Section 7.2 for further review).

WARMF, on the other hand, is a complete connection because it takes process equations from existing models and programs them into the Windows-based interface with GIS components (Chen et al., 1999). The GUI, which is developed and distributed by Systech, Inc., utilizes equations and concepts of various existing models: The main computing code was taken from the Integrated Lake-Watershed Acidification Study (ILWAS) model; ANSWERS provides the algorithms for sediment transport, erosion, resuspension, and deposition; pollutant accumulation on the land surface is treated using modified equations from SWMM and the kinetic subroutine of WASP5 is used for conventional pollutants, nutrients and algal dynamics (Chen *et al.*, 1999). The precursor to WARMF, IWMM, is also a management model developed by Systech, which utilized process equations from different models and was built in a GIS framework (Chen et al., 1995 and Chen et al., 1996). Both of these GUIs (WARMF and IWMM) have had limited application, perhaps because they are proprietary. Recently, however, Systech has begun marketing WARMF as a TMDL tool to compete directly with EPA's BASINS (e.g. Chen *et al.*, 1999), so its use may increase as the need for TMDL analysis increases. (The WARMF interface actually calculates a TMDL based on watershed and water quality modeling, while BASINS just provides the models for water quality simulation, leaving it up to the user to apply these in a way that produces a TMDL.)

### *7.1.4 User interfaces: important characteristics*

The most obvious advantage, indeed the purpose, of GUIs is to provide a modeler with the easy use of an otherwise not-so-user-friendly computer model. The advent of user interfaces has made environmental models much more accessible to all types of engineers, scientists, and planners. This advantage, however, is also its most significant disadvantage. User interfaces can "gloss-over" the complexity of a model, making it seem as if the model is quite simple, while in reality the interface may have made many assumptions in order to simplify the model and these assumptions may greatly impact the results of a simulation.

Because interfaces tend to "hide" the models within their programming, the model may be reduced to a "black box" where data is input and output is processed, but the user is relieved of needing to understand the processes being modeled. Although this problem is inherent with or without a GUI, it has been exacerbated because GUI's make modeling a much more accessible tool in environmental management. This is not to say that interfaces should not be developed and applied, much can be said in their support, such as their ease in strong database maintenance characteristics, input file generation, output processing and display.

In the preceding section, a few examples of the various user interfaces developed over the last several years were presented. The impetus behind their development may have varied, but the overall purpose was the same: to provide an user-friendly connection between the modeler and the model. Although the evaluation of any particular user interfaces is beyond the scope of this project (excluding the brief review of BASINS at the end of this chapter), it is useful to summarize several desiderata, which are important when considering the implementation of a model GUI. These are meant as general guidelines: these are considered to be necessary properties of a model interface but may not be sufficient to ensure that the GUI and the underlying models are applicable to a given modeling problem. As always, the user should evaluate any prospective GUI relative to the specific modeling requirements of the problem context.

*Preservation of the model's process equations* The equations defined within a modeling framework are the backbone of the program. These equations should be unchanged when the model is incorporated into a GUI, especially if the model has been tested and successfully calibrated and verified in numerous situations. Even if the interface is meant to be a "complete" connection, where the process equations are embodied within the interface programming (as opposed to being preserved within the model executable), the overall algorithms developed within the original model code should be transferred unchanged to the interface code and extensively tested against results from the original program.

*Access to the model executable (and sometimes, model code)* The ability to analyze the modeling code and execute the model without the interface can become vital, especially in the model development phase of a project. Sometimes, an interface provides a good tutorial for a new user, but over time may hamper the modeling process once the modeler fully understands how to successfully develop the input files and execute the model. Inability to independently access the model may eventually frustrate the user. For this reason, it is advantageous to be able to execute the model without the interface, if desired. In these circumstances, even though the interface may no longer be executing the model, it still may be useful for input generation and output processing. In some circumstances, access to the model source code and the ability to create a revised executable may be necessary. This may become crucially important if modifications to the model are necessary to accommodate regional conditions.

*Database interaction and versatility* As noted earlier, environmental models tend to be very data intensive. As a result, the ability to access and manipulate a comprehensive database through the interface is a potentially valuable attribute. GIS presents an obvious choice, but it is not the only option – many other databases exist which may be better suited for a project, depending on the data format (e.g. Microsoft Access, Oracle, FoxPro, etc.). Sometimes, these large databases can be connected to GIS through object-oriented database connection (ODBC) drivers and other tools. In addition, the GUI should allow the user to import a custom data sets for analysis and update existing databases. The interface should explicitly display any pre-set or default modeling parameters that may otherwise be defined by the modeler. For example, in a watershed model, a parameter such as depth of rainfall before infiltration begins may be required for the runoff calculation. If the interface pre-sets this number, it is important to that the user be informed of this fact, and the interface should, in addition, offer the functionality to alter the default setting.

*Debugging tools and process checks* Typically, no model will run correctly the very first time the user attempts to execute it. Almost always, some type of debugging will need to occur before a model will run for the full simulation time. Although the use of an interface cuts down on simple errors in input files (such as units conversions and text formatting), the GUI should still provide some sort of debugging tools to assist the modeler with potential run-time errors. In addition, process checks are vital, especially for new users. As a simple example, if a watershed and water quality model are interfaced, the watershed loads need to be determined before the execution of the water quality model – if an attempt is made to run the water quality model before the loads are calculated, the GUI should raise a flag to the modeler. Other checks may not be as obvious and straight-forward – perhaps the flows need to be set in an input file before the GUI can accurate calculate volumes or residence times. Straight-forward or hidden, the interface should ensure that the proper steps are followed in the correct order for model execution.

*Easy display of output with data for model validation* Often user interfaces overlook or provide limited capability for the coordinated review of data and model results. A strong interface provides the ability to display model output with available data, and to compute various summary statistics, which greatly facilitate the modeling tasks of calibration and verification. Considering that a major investment of effort will be made in the collection and analysis of field data, and the comparison of that field data with modeling results, any means of expediting this process will greatly benefit the overall modeling program.

*Comprehensive and comprehensible help file or manual* It is understandable that many GUI's are developed under contract, and are therefore limited by time, scope, and finances of the project; however, the developer should not ignore the need for a proper interface documentation. Because the generation of a full-scale manual is cumbersome, and sometimes unnecessary, the programmers should at least provide an online help system to aid in the use of the GUI.

To a great extent, proper documentation and the aforementioned "process checks" can help ensure that the sophistication implicit in the model processes are not lost in the novelty of an interface. Also, a clear separation needs to made between a model development project, which includes input formulation, model calibration and verification, and routine management application of a developed model. The former type of project should be undertaken by experienced modelers with direct access to the model codes and operations, while the latter can be reduced to a user-friendly operation exploiting the simplifications of a GUI.

### *7.2 BASINS*

The Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) system was developed under the direction of EPA's Office of Science and Technology, Standards and Applied Science Division and in cooperation with an interdisciplinary team from Tetra Tech, Inc., see Lahlou et al. (1998). BASINS, an interface designed within the GIS platform, ArcView 3.x, was developed to achieve a more efficient way to approach watershed and water quality management. It integrates several environmental data sets with analysis techniques and environmental models, and assists in various stages of environmental management and planning. BASINS was, in part, created to support the development of TMDLs. Because of the recent enforcement of Section 303(d) of the Clean Water Act, some states are under court order to develop greater than 100 TMDLs per year (e.g. Georgia and Idaho). This is an enormous increase in work based on the relatively small number of TMDLs that had been developed prior to 1995 (Battin et al., 1998). BASINS is being developed to with the hope of assisting the states in this daunting task. This section present a brief overview of BASINS 2.0 with its tools and utilities, and attempts to review its applicability to the Texas TMDL process.

The BASINS system combines six components for performing watershed and water quality analysis:

- National databases with local data import tools
- Assessment tools (*TARGET*, *ASSESS*, and Data Mining) that address needs ranging from large-scale to small-scale basins

- Watershed Characterization Reports
- Utilities including Data Import, Land Use Re-Classification, DEM Reclassification, and Watershed Delineation
- Watershed and water quality models including NPSM/HSPF, TOXIROUTE, and QUAL2E
- Post-processing output tools.

### *7.2.1 Data Sets, Assessment Tools, Utilities and Characterization Reports*

The BASINS physiographic data, monitoring data, and associated assessment tools are integrated in a customized GIS environment – ESRI's ArcView 3.1a. The data provided for BASINS can be classified as three types: spatially distributed data, environmental monitoring data, and point source data, summarized in Table 7-1 (Lahlou et al., 1998).

There are three BASINS system assessment tools: *TARGET*, *ASSESS*, and Data Mining. *TARGET* is designed to integrate and process areas that include more than one watershed (EPA hydrologic unit code, HUC). This tool is designed to assemble and process a large amount of detailed, site-specific data associated with a particular region and to summarize the results on a watershed basis. Using these water quality or point source loading summaries, watersheds are then ranked based on the level of selected evaluation parameters. Figure 7-2 shows an example screen.

*ASSESS* uses the same data as *TARGET* but provides a different perspective on the locational distribution of potential pollution problems. It operates on a single watershed (HUC) or a limited set of watersheds and focuses on the status of specific water quality stations or discharge facilities and their proximity to water bodies, see Figure 7-3. Data Mining is a tool that allows the user to retrieve and visualize BASINS water quality and point source loading data using a dynamic linkage between station locations and their corresponding loading or concentrations for all parameters monitored.

Table 7-1
Data sets included within BASINS

| *Spatially Distributed Data* | *Environmental Monitoring Data* | *Point Source Data* |
|---|---|---|
| Land use/land cover (GIRAS) | Water quality monitoring station summaries | Permit Compliance System |
| USGS Hydrologic unit boundaries | USGS gaging stations | Resource Conservation & Recovery Act (RCRA) sites |
| Urbanized areas | Water quality observation data | Industrial Facilities Discharge (IFD) sites |
| Drinking water supplies | Fish and wildlife advisories | Mineral Availability System |
| Populated place location | Bacteria monitoring station summaries | Mineral Industry Location |
| Dam sites | National Sediment Inventory (NSI) | Toxic Release Inventory (TRI) sites |
| Reach File, version 1(RF1) | Weather Station sites (477) | Superfund National Priority List sites |
| EPA region boundaries | Shellfish Contamination Inventory | |
| Reach File, version 3 (RF3) | Clean Water Needs Survey | |
| State boundaries | | |
| Soils (STASGO) | | |
| County boundaries | | |
| Elevation (DEM) | | |
| Federal and Indian Lands | | |
| Major Roads | | |
| Ecoregions | | |

In addition to these assessment tools, there are utilities to aid in a watershed/water quality analysis:

- The Watershed Delineation tool is used to create subwatershed boundaries within a cataloging unit, thereby allowing the user to evaluate and model water quality conditions on a subwatershed scale.
- The Import tool allows the user to import additional data sets and to prepare the data to make them compatible with BASINS GIS functions and models. Presently, it is designed to function on four data types: watershed boundaries, land use, Reach File Version 3, and Digital Elevation Model (DEM). It also provides the capability for users to import locally developed data, which might be more accurate, at a higher resolution, or more reflective of current conditions.

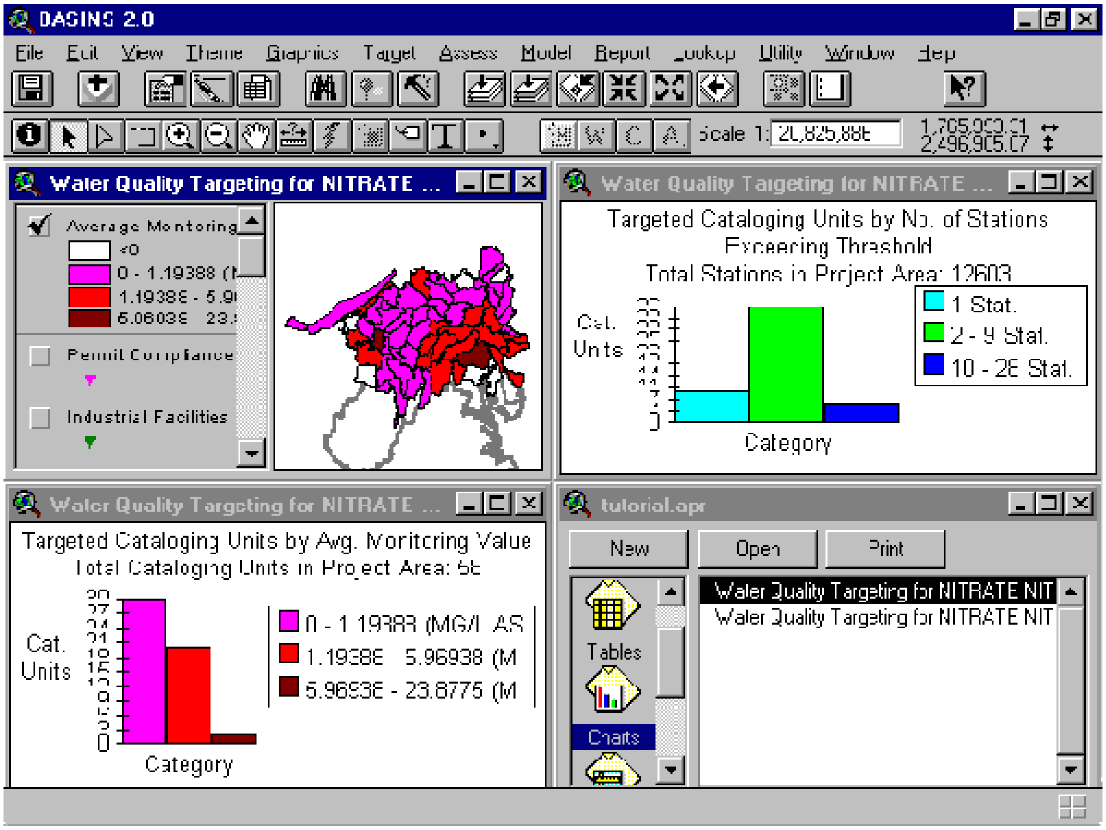


Figure 7-2. Example of display output from *TARGET* Assessment Tool. The three output windows are: (1) a map displaying the average monitoring value computed for each watershed based on the user-specified parameter, statistical summary, and threshold value; (2) a bar chart showing the distribution of cataloging units with respect to the number of stations exceeding the selected threshold value; and (3) a bar chart that summarizes the distribution of cataloging units with respect to the average monitoring values (Lahlou, *et al.*, 1998).

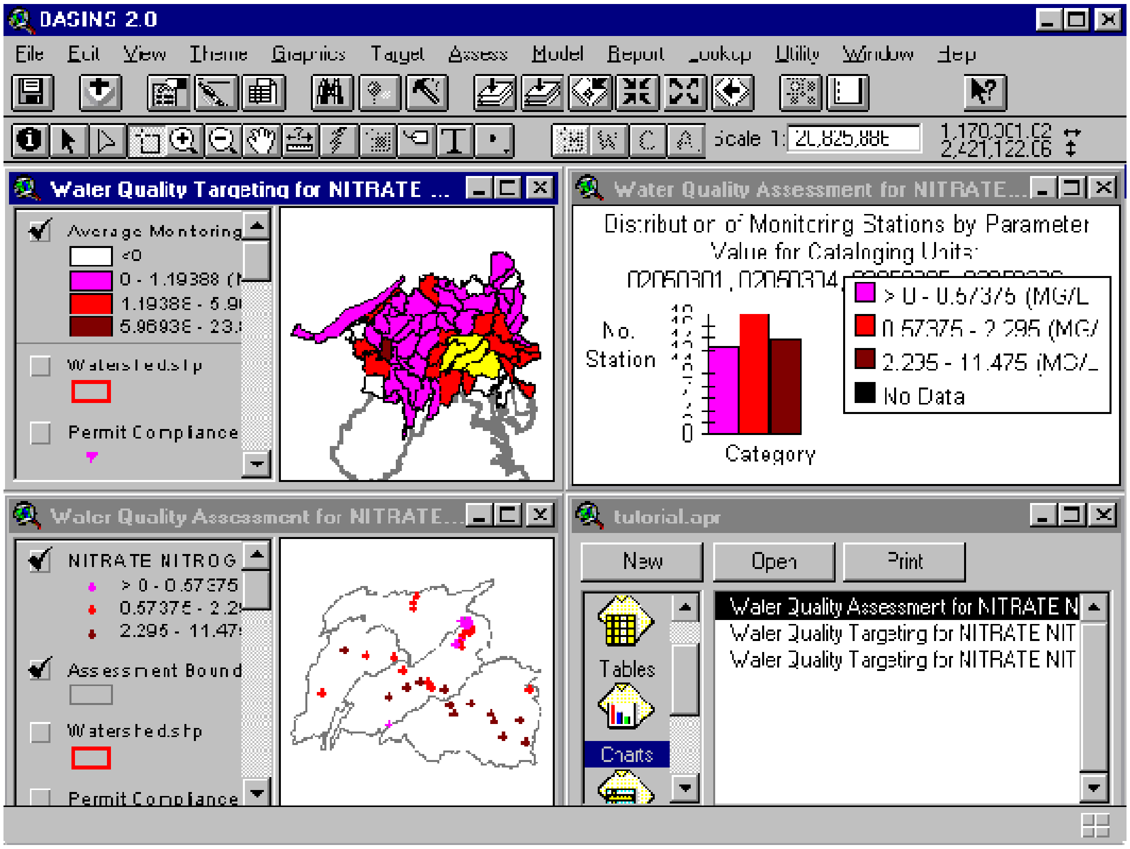


Figure 7-3. Example display of *ASSESS* Assessment Tool. The two output windows from this assessment tool are: (1) a map with water quality stations ranked according to the average monitoring value for the selected time period, selected water quality parameter, and corresponding statistical summary data; and (2) a bar chart displaying the distribution of the stations based on the monitoring value (Lahlou, *et al.*, 1998).

- The Land Use Reclassification tool is used to change land use classifications within an existing data set. This tool is particularly useful for modeling purposes while evaluating the effect of land use changes on water quality.
- The Water Quality Observation Data Management tool is used to manage water quality data by allowing the user to add new stations, delete unnecessary stations, relocate misplaced stations, and incorporate new data into existing stations.
- The Digital Elevation Model (DEM) Reclassification tool is used to tailor the display of the topographical data by providing for a user-defined interval input.
- The Lookup Tables tool provides the user with access to several reference tables, including water quality criteria data, Standard Industrial Classification (SIC) codes and definitions, and STORET agency codes and definitions.

Beyond assessment tools, BASINS includes six different types of Watershed Characterization Reports:

| | |
|---|---|
| Point Source Inventory | Water Quality Summary |
| Toxic Air Emission | Land Use Distribution |
| State Soil Characteristics | Watershed Topographic Reports. |

The Point Source Inventory Report relies on the EPA Permit Compliance System (PCS) database to identify permitted facilities in the selected study area and provides a discharge loading summary for a given year (1991-1996), for a given pollutant. The Water Quality Summary Report displays in table format, or map format if selected, water quality data as statistical summaries of the mean and selected percentiles of the observed data. It provides a summary of water quality monitoring stations within the selected watershed that monitored a particular pollutant during a given time period. This data was originally obtained from USEPA's Storage and Retrieval System (STORET). The Toxic Air Emissions Report lists facilities that are part of the Toxic Release Inventory (TRI) and have estimated air releases of a particular pollutant in a selected watershed.

The Land Use Distribution Report summarizes land use distribution data originally obtained from the USGS Geographic Information Retrieval and Analysis System (GIRAS) using the

Anderson Level II classification in both table and map format. The State Soil Characteristic Report summarizes, in table or map format, the spatial variability of selected soil parameters within one or a set of subwatersheds. Parameters considered include water table depth, bedrock depth, soil erodibility, available water capacity, permeability, bulk density, pH, organic matter content, soil liquid limit, soil plasticity, percent clay content, and percent silt and clay content. This information was originally obtained from the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) State Soil and Geographic Database (STASGO).

The Watershed Topographic Report provides a statistical summary and distribution of discrete land surface elevations in the watershed and generates an elevation map of the selected watershed. The source elevation map in BASINS for this report is derived from converting USGS one degree Digital Elevation Map (DEM) into a vector map product.

### *7.2.2 BASINS Models*

The BASINS system includes three watercourse models to accomplish the common objective of water quality modeling to predict the impact of different point and nonpoint source loading scenarios on surface water bodies. These models range from simple to fairly complex operative dynamics. TOXIROUTE calculates final and average concentrations of general water quality constituents based on a dilution and first-order decay algorithm. QUAL2E uses complex algorithms to simulate nutrients, biochemical oxygen demand, dissolved oxygen, temperature, algae, and conservative and nonconservative substances in a one-dimensional watercourse. The Nonpoint Source Model (NPSM) can be used in situations where a continuous simulation model of the fate and transport of water quality constituents in surface water bodies is required. It integrates both point and nonpoint sources and it capable of simulating nonpoint source runoff and associated pollutant loadings, accounting for point source discharges, and performing flow and water quality routing through stream reaches and well-mixed reservoirs.

TOXIROUTE is a simple first-order decay solution to simulate the transport of selected pollutants in streams and rivers. This model is useful for an initial examination of concentrations of discharged pollutants in receiving waters. The model does not explicitly consider nutrient or chemical reactions or transformations. In cases where algal growth or other significant chemical processes are a concern, this simplified model might be inappropriate. It assumes steady-state conditions, where the system has reached equilibrium, which may present limitations in cases where wet weather processes, such as nonpoint source runoff, predominate. TOXIROUTE only simulates pollutant transport in certain reach types included in the Reach File, Version 1 database. When applied within BASINS, the model receives point source discharge and reach data from themes on the BASINS View (Permit Compliance System, Reach File Version 1).

To initiate the model, a cataloguing unit must be selected. Then, TOXIROUTE is chosen from the *Model* pull-down menu. The steps in its execution are the following:

(1) The user is prompted to select a pollutant from a list of pollutants. If available, BASINS generates the point source data for the selected cataloguing unit and TOXIROUTE automatically loads this information.

(2) The user also enters background concentration, parent molecular weight, child molecular weight, and half-life. The parent molecular weight and child molecular weight are not significant if there is no degradation product (child chemical). The parent and child molecular weight are used to calculate the child chemical concentration.

(3) The *Stream Flow* selection box lets the user select 7Q10 or mean flow.

(4) The *Reach List* screen shows all of the reaches in the cataloging unit, including lengths and stream flows.

(5) Using the *Discharger List* screen the user can view and edit point source loading information as well as add or delete a facility.

(5) The *Output* screen lists, in tabular format, final concentrations on a reach basis. The Average Concentration column lists mean concentrations (averaged over the total length of the reach) of the pollutant in reaches and the Final Concentration column lists the concentrations of the pollutant at the end of the reaches (or downstream end of the reaches). The Child Concentration column shows the final concentration of the chemical produced during the decay of the parent chemical. The window also displays reach number, name, length (meters) and flow (meters per second cubed).

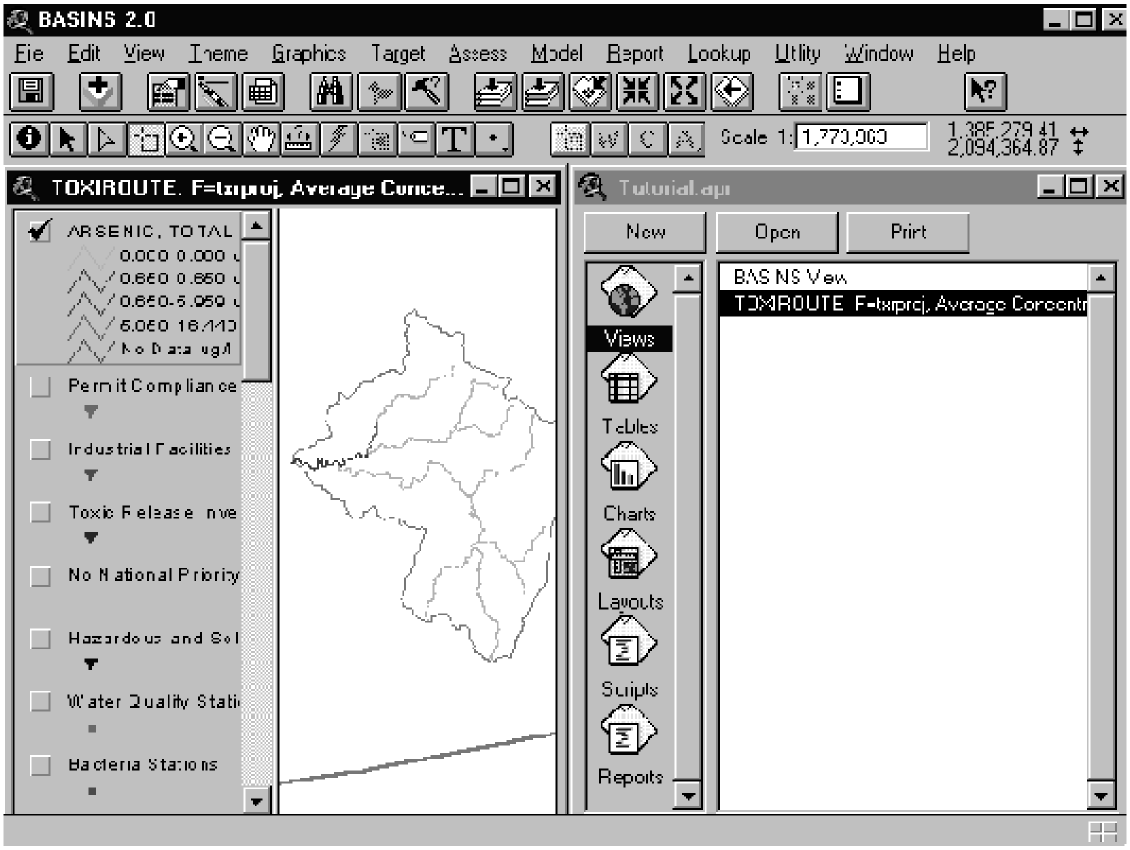


Figure 7-4. Example of output from TOXIROUTE model run (Lahlou, *et al.*, 1998).

To visualize TOXIROUTE output with the *Visualization* tool, the user selects one element to visualize (average concentration, final concentration, child concentration, and stream flow). A second dialog box provides the option to classify the output data using ArcView's legend editor tools. Results are displayed in a new view within ArcView where the active theme has the pollutant name and the legend displays the classification scheme, see Figure 7-4.

The QUAL2E model was described in Chapters 4, 5 and 6, see especially the tabulations of model properties, as well as reviewed in Ward and Benaman (1999). QUAL2E is accessed in BASINS by selecting it from the *Models* menu.. The user is prompted to select a year for modeling, and is informed that, by default, BASINS generates input data for CBOD, dissolved oxygen, fecal coliform, nitrogen species, and organic and dissolved phosphorus. Any point source data on these substances is automatically selected and processed in the QUAL2E simulation. The user may then select up to three conservative substances and one nonconservative substance. Information on the number of discharges in the selected reaches as well as the total number of pounds discharged per year will be provided to the user.

Geographic selection of QUAL2E simulation is different from that in the other two models included in the BASINS program because individual reaches in a cataloging unit are selected instead of a whole cataloging unit or watershed. BASINS first checks the data pertaining to the selected reaches to find out whether the selected reach network is acceptable for simulation with QUAL2E. Then, BASINS modifies the reach data slightly such that QUAL2E reach input requirements are met. In addition, if some necessary information for the input file is not available, a reasonable value is assigned to fill the blank. This default information can be seen using any text editor to view the DEFAULT.Q2E file. For example, QUAL2E uses 7Q10 flow as default stream flow, but this can be changed by the user to simulate other conditions.

Once the input file is complete, the QUAL2E interface window becomes active to the user, see Fig. 7-5, which allows the user to execute QUAL2E and also to set modeling parameters such as run time and simulation type. Within this interface, choosing the *Run File* option from the *Import* menu will load the generated file: QUALINP.RUN, which loads data from BASINS into QUAL2E. The *Next* and *Back* buttons (Fig. 7-5) move from screen to screen within the QUAL2E interface in order to view data input and system set up. In addition, the *Index* button can be used to view a list of all active and inactive screens. Before proceeding with the model run, the user can select and modify data on any screen by clicking on the right button.

In QUAL2E, each reach is divided into computational elements of equal length such that each reach has an integral number of such elements. The length of each computational element is 1.642 km. This implies that the length of the reach has to be adjusted, which is done by BASINS. As a result, reach lengths in the model may appear slightly different from those in the BASINS View. QUAL2E uses its own numbering scheme for reaches. The reach number in the Reach File v1 (RF1) database appears in the Reach Name column of Screen 2 of the QUAL2E

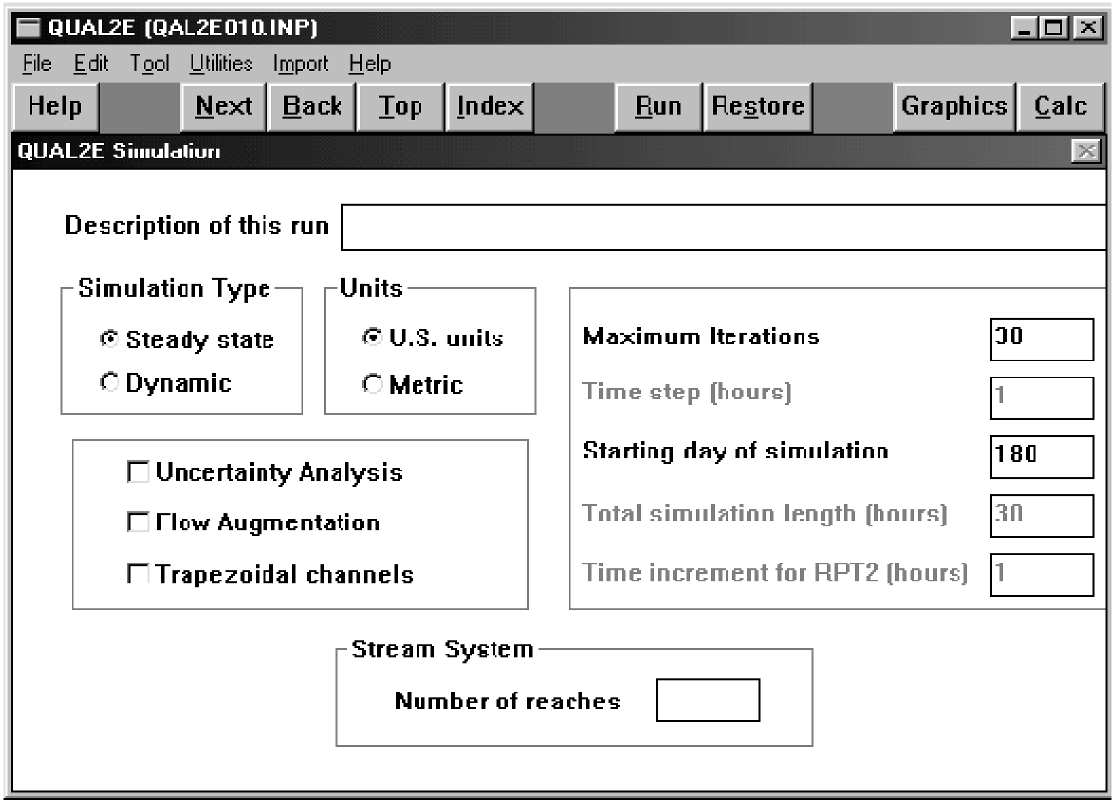


Figure 7-5. QUAL2E windows interface within BASINS (Lahlou, *et al.*, 1998).

Interface (Stream Reach System). The *Reaches* button displays the reach network and the computational elements used for the simulation, see Figure 7-6. Once the model has been executed, the output file can be viewed by selecting *Visualize* from the *Models* menu which displays the output using a text editor. In addition, clicking on the *Graphics* button begins the plotting program. To generate a graph, some data elements not available from the RF1 database have to be entered in Screen 11 of the QUAL2E Interface (Hydraulic Data). To plot a graph, a starting reach and an ending reach must be selected. Types of graphs available are flow vs. distance and water quality constituents vs. distance (Figure 7-7).

A significant limitation of QUAL2E in TMDL modeling is the fact that it is a steady-state model, and therefore is unable to simulate a time-varying problem during which both the stream flow in

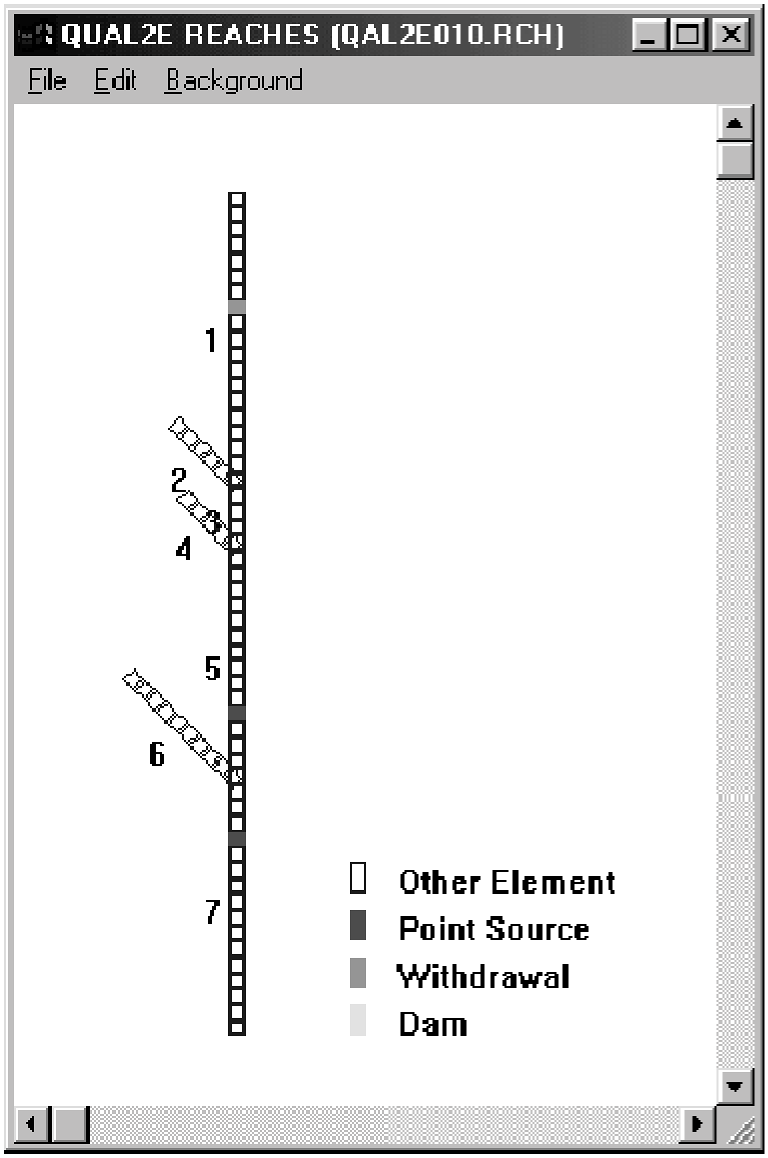


Figure 7-6. Display of reach network with computational elements for a QUAL2E run (Lahlou, et al., 1998).

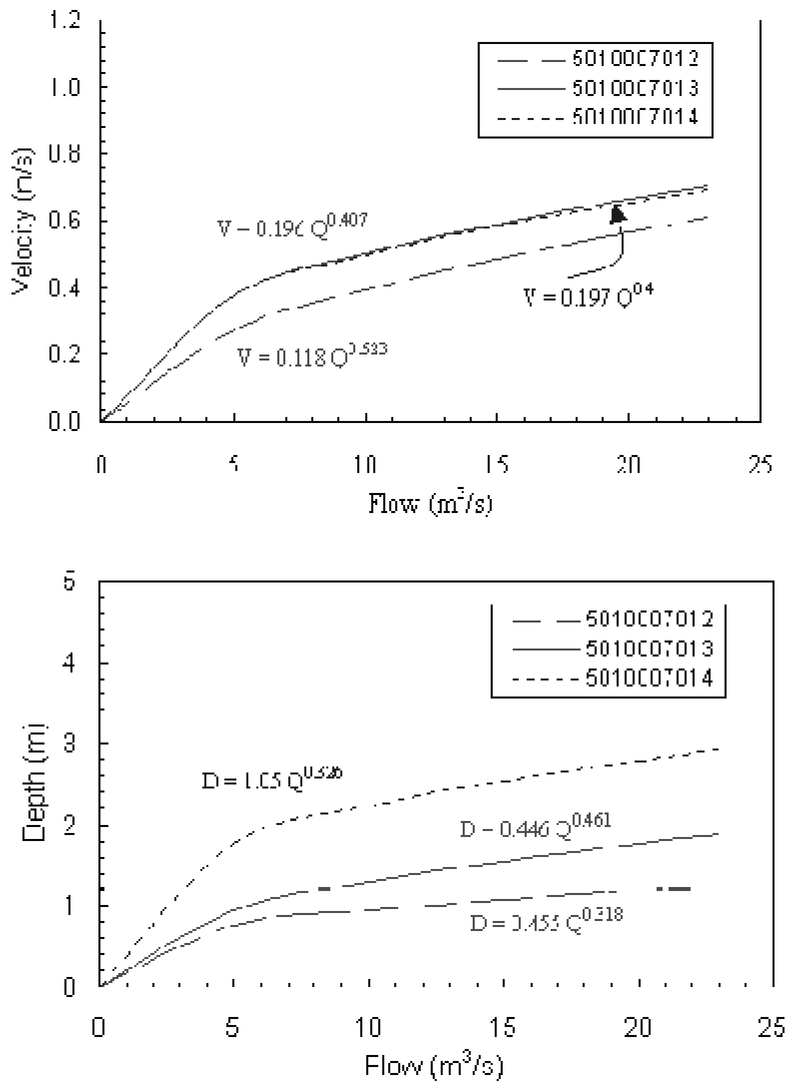


Figure 7-7. Output from QUAL2E's graphing program:
above, velocity vs. flow for three different reaches;
below, depth vs. flow for three different reaches (Lahlou, *et al.*, 1998).

river basins and input waste loads are not constant. As a result, the effects of dynamic forcing functions, such as headwater flows or point loads, cannot be modeled in QUAL2E. This includes, notably, runoff from storm events. Another limitation is that QUAL2E cannot handle multiple point sources discharging to a single computational element: BASINS must total all the discharges in each computational element while preparing a QUAL2E input file. In addition, QUAL2E does not allow the endpoints—either the upstreammost or downstreammost elements of a reach—to receive point source discharges, therefore if such discharges are specified, the discharger location is shifted one computational element to the interior of the reach.

It should also be noted that BASINS assumes a selected reach is a headwater even when it may have an upstream reach not included in the present simulation. To carry over the effect of upstream discharges not included in a simulation, it is necessary to model upstream reaches separately, record the output flow and concentrations, and keyboard these numbers in the Headwater Source Data screen in QUAL2E.

The NonPoint Source Model (NPSM) is the Windows version of HSPF, see Chapter 3 and Ward and Benaman (1999). HSPF is a distributed watershed model that can simulate the hydrologic and associated water quality processes on pervious and impervious land surfaces, and route the runoff and associated loads into well-mixed stream segments and impoundments (Bicknell et al., 1996). Each watershed is defined as a hydrologic unit containing a series of point and nonpoint sources discharging to an associated stream reach.

NPSM provides a GUI that can be launched directly from the BASINS View or as a stand-alone program. To execute NPSM from the BASINS View, the watershed theme must be activated and the user must select a watershed or a system of hydrologically connected subwatersheds to model. Subwatersheds can be developed either through an onscreen delineation using the *Watershed Delineation* tool or imported using the *Import* tool. Selecting *NPSM* from the Model pull-down menu will launch the model. For a NPSM session to begin, a project name is entered (i.e. name to model run) and a discharge year from the Permit Compliance System (PCS) is chosen to incorporate point source data in to the model. The BASINS GUI appears in the background of the NPSM Interface, and it contains four graphical windows:

Watershed Data Management (WDM) information
land use distribution
a 'BASINS' view
the BASINS project window.

The NPSM interface contains a series of 11 buttons:

| | | |
|---|---|---|
| Reach Editor | Simulation Time | Land use editor |
| NPSM control cards | Pollutant Selection Screen | Point Sources |
| Default Data Assignment | | Input Data Editor |
| Output Manager | Run NPSM | View Time Series Output |

see Figure 7-8. It is recommended that the user proceed through the NPSM functional buttons from left to right because once data "behind" a button is edited, it is necessary to edit data in every button located to the right of this one. The information entered during a "project session" is stored within a NPSM project file with the name designated earlier, at start up (see above).

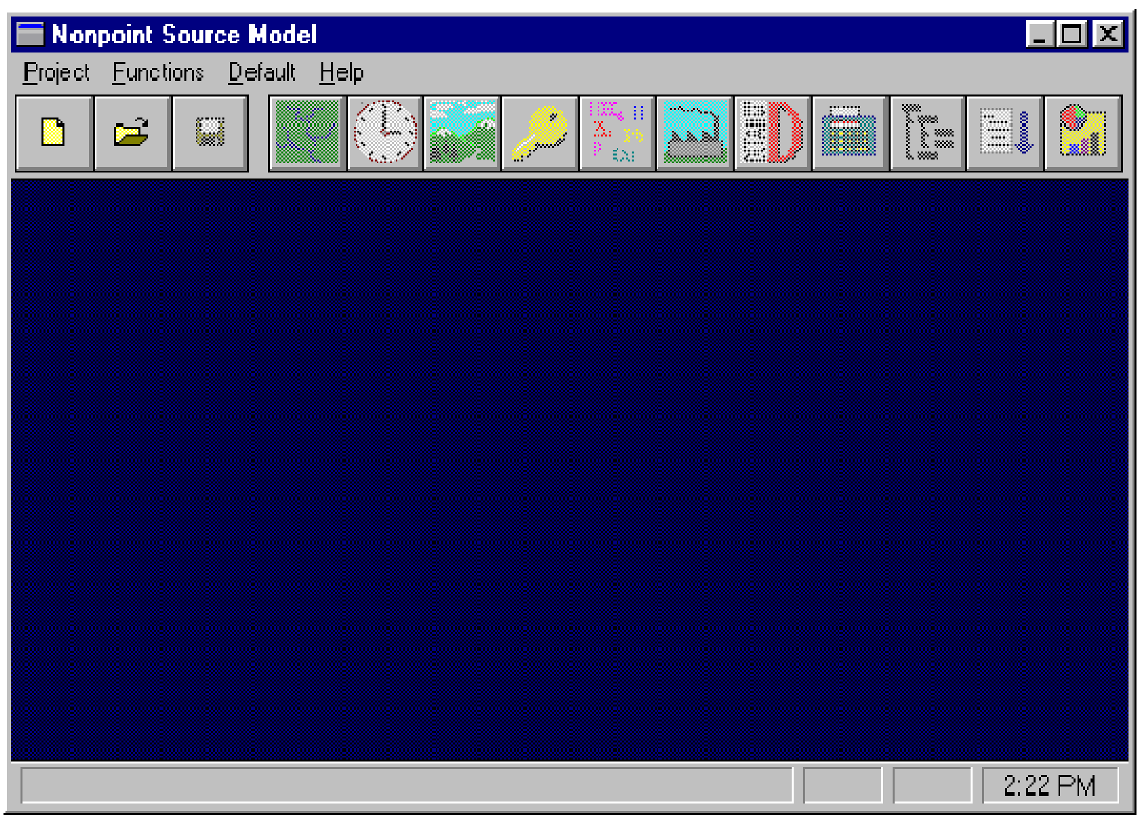


Figure 7-8. NPSM Interface as shown either through BASINS or when executed ‘stand alone’ (Lahlou, et al., 1998).

The *Reach Editor* button has five options and it allows the user to add or delete reaches and to edit number, id, name, number of exits, type, and watershed for each reach. NPSM currently only supports single watershed outlets and stream reach types within its modeling framework. The Reach Network Visualization option under the Reach Editor button provides the user with a simple graphical representation of the reach network system, shown in Figure 7-9. The Reach Characteristic option includes reach number, reach name, length, delta-h, and elevation (either input manually or read from the River Reach version 3 file). Length refers to the length of the reach segment (in miles), delta-h refers to the change in vertical elevation over the length of the reach (in feet), and elevation refers to the average elevation of the reach segment (in feet). The F-Tables (function tables) option displays rating curves used for flow calculations, see Fig. 7-10. Data from the Reach File V1 database in BASINS is typically used to calculate the rating curve based on channel geometry (assuming a trapezoidal cross-section). If the F-table is not complete, the user must complete information in order to run a successful simulation. The *Import/Export* button can be used to import required stream characteristic data from the Reach File Version 1 database into BASINS to calculate F-Tables (rating curves) or the geometry may be entered or modified manually.

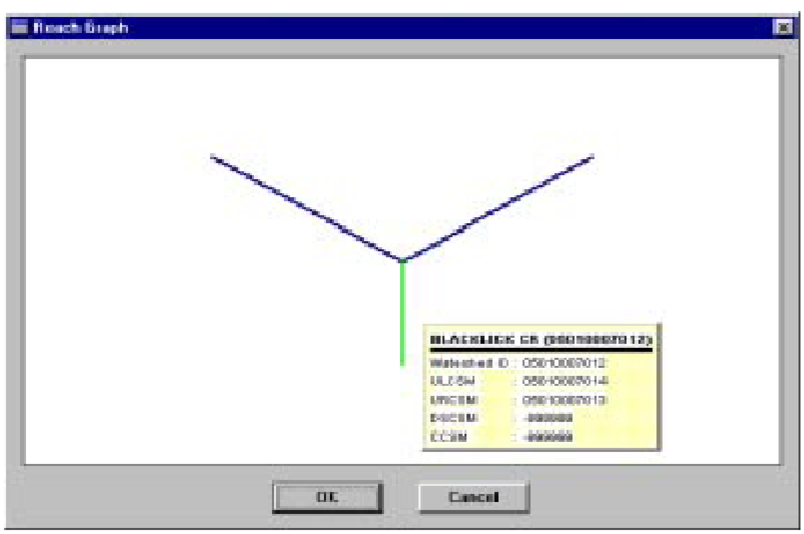


Figure 7-9. Reach Visualization tool within NPSM which shows the user a schematic representation of the modeled reach network (Lahlou, *et al.*, 1998).

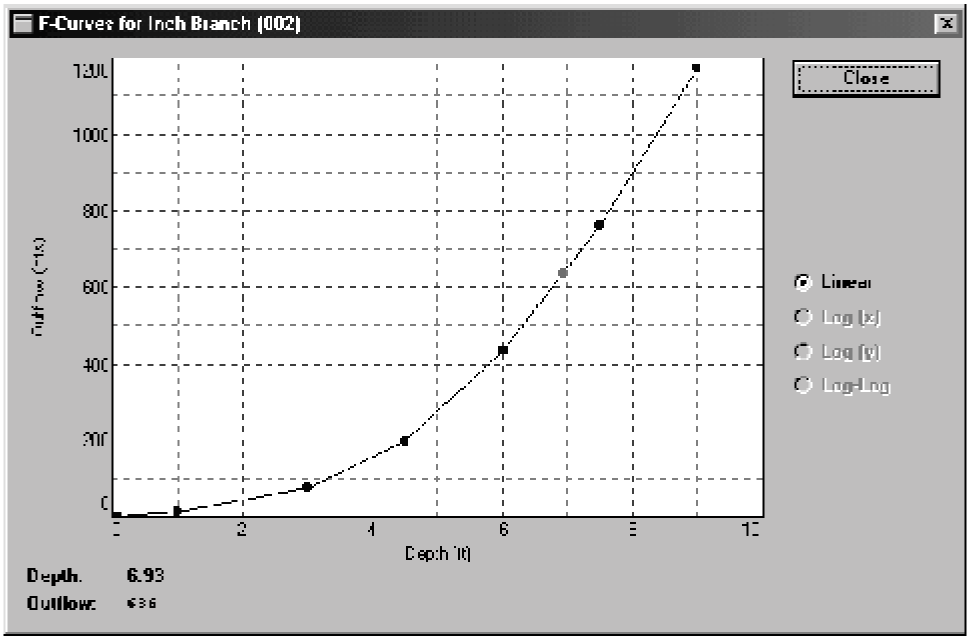


Figure 7-10. Example of rating curve generated from the NPSM F-Table (Lahlou, *et al.*, 1998).

The NPSM interface also provides windows to specify simulation time, meteorological data (which weather station data is to be used) and a land use editor, among other features, to control the time and spatial conditions considered in the model. The *Simulation Time and Meteorological Data* window allows the user to select the most appropriate meteorological data set. These are compiled by weather station, in Watershed Data Management, (WDM) files, which are binary files that contain the hourly data required by NPSM. The period of record is generally January 1, 1970 to December 31, 1995, however, a number of stations contain shorter periods due to limited data availability.

The *Land Use Editor* window displays each pervious and impervious land unit defined for modeling. Land Name, Land Type, Area, and Watershed define these. Land units in this table can be edited. The user can also modify the default percent perviousness value associated with each land use category. The *Control Cards* window displays options for choosing the proper

NPSM (HSPF) modules to model pervious land (main module: PERLND), impervious land (IMPRLND), and reaches (RCHRES). For example, in pervious land some of the modules are snow simulation, water flow, sediment transport, soil temperature, gas concentration, general water quality, soil moisture, pesticide, nitrogen, phosphorous, and tracer. In some situations, selection of one HSPF module results in automatic selection of another HSPF module because some HSPF modules are required to run other modules. The *Pollutant Selection* screen allows for selection of several pollutants from four classifications: eutrophication parameters (nitrogen cycle and/or phosphorus cycle), gasses (dissolved oxygen and/or dissolved carbon dioxide), general quality, and sediment/solids. In addition to general quality constituents (e.g. nitrogen and phosphorous), up to three pesticides and one tracer can be simulated.

When NPSM is first started for a given watershed, BASINS creates a file containing average flow and loading values for each source facility in the watershed(s) from Permit Compliance System (PCS) data. The *Point Sources* button uses this data to present discharger names and the associated point source pollutants as options for modeling. Point sources may be modeled as constant or time variable inputs. Additional point source data can be imported with the *Import* button, which allows the selection of a user-generated file containing time series flow and pollutant loading data. The *Default Data Assignment* button and associated window enables the user to select an NPSM default data file and assign default data to land units, reaches, and pollutants in the project. There is a starter default data file packaged within BASINS, but the user can create and modify his/her own default data files.

Finally output management and visualization is controlled through the *Output Manager* and *View Time Series Output* buttons. The *Output Manager* is used to specify the simulation parameters to print, the print intervals, and the grouping of output parameters. NPSM operates on an hourly time step; therefore, the user can choose from hourly, daily, monthly, and yearly print interval options. The *View Time Series Output* button allows the user to view results in text format within a text editor or graphical format within the NPSM postprocessor, see Figure 7-11. The postprocessor supports daily, monthly, and annual NPSM output. The postprocessor displays NPSM simulation output, BASINS water quality observation data, and USGS flow data in a graphical format. It also performs basic statistical calculations and data comparison.

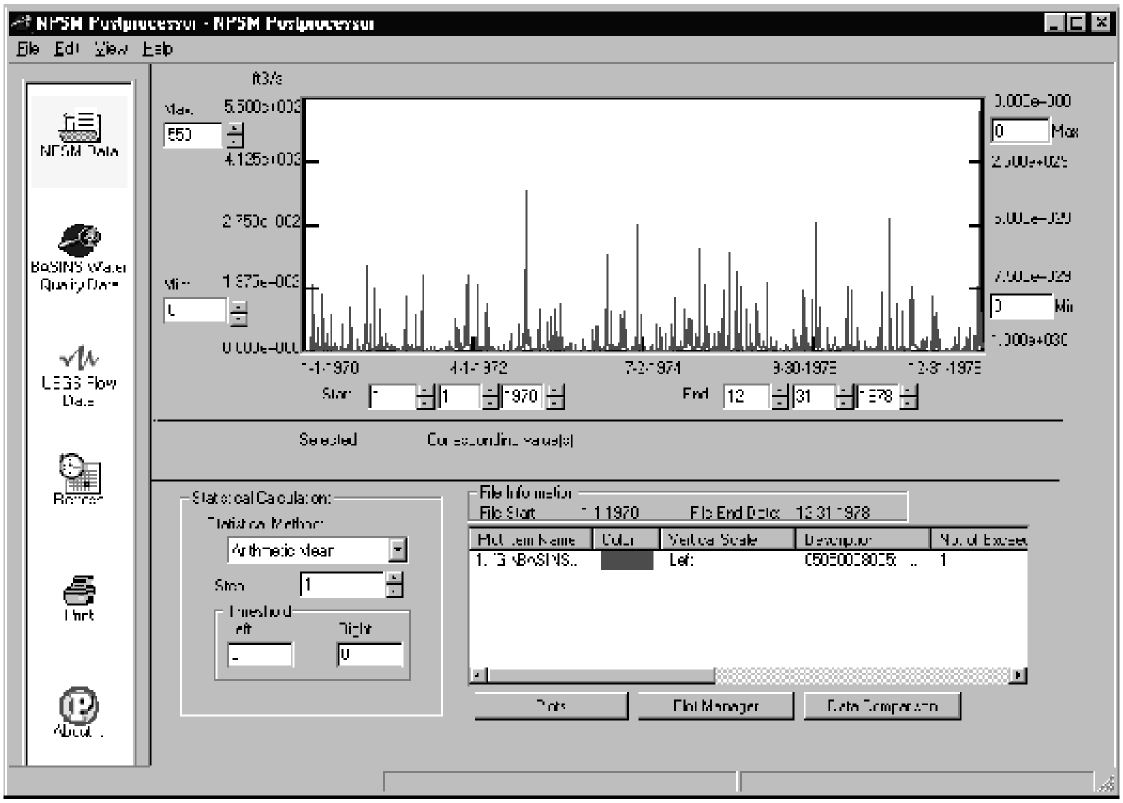


Figure 7-11. Example of output visualization post-processor for NPSM data (Lahlou, et al., 1998).

### *7.2.3 Application of BASINS in TMDL determination*

There are several ongoing projects that are applying or considering BASINS as a tool for watershed management and TMDL development (Paul Cocca, BASINS Team Engineer, EPA Office of Water, pers. comm., 1999). Some of these are:

- Evaluation of the potential use of BASINS in the Nashua watershed (EPA Region 1);
- Pathogen TMDL development in West Virginia (Muddy Creek) (EPA Region 3);
- Pathogen TMDL development in Georgia (EPA Region 4) using a beta version of BASINS;

- TMDL development on Kokomo Creek and Wildcat Watershed with additional manual setup of QUAL2E (EPA Region 5).
- Hydrologic and water quality loading assessment by the Army Core of Engineers in various basins (EPA Region 6).
- Estimation of nutrient loadings in Mobile Bay by the National Estuary Program (Region 6) .

The usefulness of BASINS consists largely on the fact that it is a starting point for the modeler. It provides a varied array of data files and it allows the user to perform model runs and calibrations with minimal time spent on data management itself. The most obvious disadvantage may be that this software is too easily accessible to the inexperienced user: it does require certain degree of knowledge and experience in order to use it correctly and make sense of the results. It is noteworthy, though, that it is user-friendly enough to encourage training through practice and usage.

Although well formatted, the BASINS interface could still be made easier to follow if there was even more format uniformity for all windows operated within the system. There could be “Help” features that, instead of being separate from the system in the form of technical notes or text files, were incorporated into the BASINS menu bar and windows. Similarly, it would be of great advantage to the inexperienced user if this help feature or one similar to the “lookup” tool would clarify the meaning of parameters that appear in their abbreviated parameter name form. Brief description of theoretical concepts providing equations and theory that the model uses to run its modules could greatly aid the modeler during calibration.

As user-friendly as BASINS seems, it is still quite difficult to develop and implement for a given watershed. The manual provided with BASINS, although complete in its explanation of the provided "tutorial", does not provide ample discussion on areas that may deviate from the tutorial set up. For example, the tutorial steps the user through modeling fecal coliform on a given watershed, However, if nitrogen or phosphorous need to be modeled, an explanation of the parameters and modules necessary for the simulation would have to be deciphered either from trial and error or from the HSPF manual. This causes much headache in trying to modify and customize BASINS for a particular TMDL. NPSM is extremely sensitive to which of the three modeling modules are active or inactive (Pervious, Impervious, and Reaches). Users will be further frustrated to find that the model run will crash without explanation or give obviously erroneous numbers without an indication of what may be wrong with the simulation. In addition, there are still some programming bugs in BASINS.

The fact that many parameters are pre-set within the program could be a concern with modeling the differing environmental systems that exist in Texas, since the sources for the parameters values and their appropriateness for Texas systems are unknown. Overall, the danger in BASINS is the same danger which holds true for many user interfaces: the processes governing the model simulations are hidden and difficult to get to when needed. For an experienced modeler, BASINS is frustrating in that the user has difficulty determining the model processing during an input file generation and model run. On the other hand, inexperienced modelers could incorrectly apply BASINS without realizing their simulation is flawed. Although it looks would appear to have great potential, the universal applicability of BASINS to the Texas TMDL process seems improbable without major customization.

# 8. Conclusions and recommendations

## *8.1 The enterprise of watercourse modeling*

At its core, modeling is the exploitation of a mathematical relation between measurable parameters of a watercourse to quantify the response of one set of parameters to changes in another. Confidence in the mathematical relation is improved by the extent that it is derived from physical principles and tested against measurements.

Implementation of models on modern computing platforms has introduced new aspects of the modeling process and obscured, at least to some extent, the core enterprise. With the increase in computing power, the ability to depict temporal and spatial detail and to accommodate increasingly complex relationships among variables has led to computer codes whose complexity appears to rival that of the real systems, and whose operation is so complicated as to demand specialized information-processing capabilities of the user completely removed from knowledge of the physical system to be modeled. The capacity of GIS to manipulate enormous geospatial data bases is yet another computational device that is particularly advantageous for depicting spatially complex systems, such as watersheds or coastal embayments, but also represents a quantum increment in required computer resources, both hardware and software.

To facilitate the operation of such systems and to obviate to some degree the requirement for specialized user skills, a number of user-interface programs have appeared, most important of which are those that use the graphical capabilities of modern desktop computers. These GUI's provide a capability for processing and display of massive data files, especially those used to generate model input, and those that represent model predictions of watercourse features. A notable benefit of this technology is the ability to employ graphical displays to communicate the results of rather technical and abstruse calculations to the general public, thereby facilitating the involvement of the public in water-management decisions.

In embarking upon a program of TMDL determination in the state, TNRCC must weigh the various features and attributes of different models, computer codes and GUI's in selecting a modeling system. For each TMDL problem, this is a major decision, because the structure and capabilities of a selected model may affect other elements of the project strategy, e.g. monitoring design, data compilation and analysis, scenario formulation, and possible treatment options. The decision, we submit, should be foremost founded upon the adequacy of the core watercourse model, i.e. its physical basis and validation, and its suitability for Texas systems. This review has therefore focused upon the features of the various watercourse models available, summarized in Section 8.2 below.

There is a philosophical tension in this decision between the desire for a model that has a range of capabilities for treating however much complexity is necessitated by the problem at hand, and the need to simplify the model by eliminating extraneous detail and emphasizing only the major controls on water quality. This philosophical issue has been with modeling from the outset, in issues of how much a modeled system can be "lumped" in space and time without eroding accuracy, and how mathematically troublesome terms in the relations can be simplified by using these lumped values. But with the growth of computing resources, the debate has sharpened to the question of how much understanding of the behavior of a complex system can be attained by increasing the number of variables considered and their space-time resolution. (There has been a subtle shift in the meaning of the word "overparameterization" in this debate, from its early meaning of a mathematical relation in which complex terms are replaced by simpler forms involving unspecified parameters, to its recent application to a model with a profusion of user-supplied parameters for which there is no measurement basis for assigning values.) Farrell (pers. comm. quoted by Renard, 1993) remarked, "The much used and abused approach of building models of greater and greater complexity, and the overparameterization that results from this approach must be avoided. Admittedly, this will raise some concern. However, the false sense of confidence that these synthetically parameterized models give to less-informed scientists and users is decidedly more dangerous."

The input file structures for both HSPF and SWAT are daunting: as noted earlier, HSPF requires many hundreds of parameters to be input by the user, and SWAT is nearly as demanding. This detail of input data notwithstanding, it is the accuracy of the individual process formulations that govern the utility of model. In this review, we found significant limitations of both models with respect to the formulation of the basic hydrological and water quality processes. Avoiding any pronouncement on one side or the other of the overparameterization debate, we observe that whichever model might be selected for a given watercourse, it will be to the advantage of TNRCC to create a Texas-specific input structure in which appropriate default parameters are pre-set, so that a user need only be concerned with those that are truly significant to the TMDL problem at hand.

It would also be remiss to overlook the important aspects of information-handling software, especially GIS and GUI's, in the model-selection decision. As noted in Section 2.2 and Chapter 7, the structural integration of GUI's (and GIS in particular) with a watercourse model can range from rather loose, facilitating only the user's manipulation of files, to tight, in which GUI processing is built into the model code, or (even more tightly) the model process code is built into the GUI. One of the properties of a GIS-based model shell, for example BASINS, particularly one that has a tight integration of model and interface, is that the initial set-up and execution of the model system for a given watershed can be handled efficaciously and intuitively using the GUI. For a management project in which a few model runs are to be made for each of many watersheds, this is particularly advantageous. On the other hand, for a project in which many model runs are to be made on a single (or few) watersheds, this may be less than advantageous, analogous to changing the oil in one's car before each use.

For the present requirements of TNRCC TMDL projects, we conclude that the optimal interaction of GUI and model is to maintain the separation of the preprocessing and postprocessing steps of model implementation from the execution of the model itself. Moreover, many aspects of the input file, once developed, will not be significantly changed for various alternative model runs, such as topography, soils, vegetation, rainfall- or streamflow-pattern scenarios. The GUI should allow the capability to fix or standardize those features of the input files.

### *8.2 Watercourse models for TMDL's*

Surface watercourses of Texas fall into a range of categories. Following the hydrological cycle, these proceed from small natural, urban or agricultural catchments, to basin-scale watersheds, through small evanescent streams to major perennial rivers, from small uncontrolled lakes to multi-purpose run-of-the-river reservoirs, and finally to the tidal and salt-intrusion reaches of the principal rivers, and the coastal embayments into which these rivers debouch. For many of these watercourses, there is significant interaction with the subsurface components of the hydrological cycle: the surficial soils, the root zone, vadose zone and aquifers. In principle, these are all manifestations of the flow of water so are equally amenable to treatment by the principles of fluid dynamics. A naive view might be that there should therefore be a single model equally applicable to all of these watercourses, for each of which the user merely alters the spatial geometry. In fact, the controlling processes, the nature of the hydraulic responses, and the parameterizations of the hydrodynamic and kinetic terms are so variable among these watercourses that the only viable strategy is to develop special-purpose models appropriate for specific types of watercourses. This is indeed the strategy that has been pursued in the development of the various models reviewed in this study.

In selecting a model for application to a TMDL, the first requirement is to clearly define the nature of the water quality problem addressed. This includes specification of:

- probable source of contaminants (e.g., landscape practices or environments, point source discharges, interaction with natural sources, production by kinetic reactions, etc.)
- categories of watercourse(s) involved,
- time scale of contamination (steady discharge, flashy loads due to storms, reaction rates)
- time-space manifestation of degraded water quality (high detention, low dilution, extremes of temperature)
- interaction among parameters (co-reacting constituents, particulate sorption)
- time-scale of response in watercourse (steady-state or equilibrium concentration, asymptotic variation, rises or spikes of high concentrations, etc.)

- spatial variability in water quality response (vertical stratification, zones of toxicity or arrested biological activity, nuisance algae formation), hence the spatial dimensionality necessary in the analysis of water quality

These together dictate the capabilities required of the model or models to be employed.

The categories of watercourses that we anticipate to represent the bulk of TMDL projects in Texas are:

(1) streams and rivers, in which the longitudinal variation of water quality is of concern
(2) run-of-the-river reservoirs that exhibit little to limited vertical stratification
(3) larger, deep reservoirs which exhibit seasonal stratification
(4) reservoirs of either type with substantial internal circulation due to power-plant operations
(5) tidal and/or saline intrusion reaches of rivers, in which the longitudinal variation of water quality is of concern
(6) deeper tidal systems, primarily navigation channels, affected by tides and salt intrusion, in which both longitudinal and vertical variations of water-quality parameters are important

We note that TMDL problems may be encountered on other types of watercourses, such as the open, shallow bays of the coast, the small-scale or "mixing zone" regions of rivers, lakes and estuaries, or the nearshore coastal environment, but we expect relatively few such situations to arise in Texas.

To address water quality of these systems, we anticipate the need to include modeling of the following *additional* watercourse environments:

(1) the contributing catchment, including soil and vegetation, and the effect of different surface properties,
(2) tributaries and small drainageways

(3) elements of the root zone or vadose zone in the watershed
(4) small reservoirs for erosion-control or agricultural water-supply purposes

While degraded water quality or limitations on use of regulatory concern will probably not arise often (or at all) with these systems, they do have a potential impact on the watercourses listed above and would need to be included in the TMDL model. The single most important of these is the contributing catchment, which for most Texas watercourses represents the primary source of nonpoint pollution.

Finally, a TMDL determination also has an implicit time scale that must be accommodated by the selected model. A differentiation must be made between the time scale of the problem context and the time resolution in a model. Water quality management problems can entail any of the following time scales:

(1) sudden excursions in water quality due to short-term rise and recession of flood hydrographs
(2) sudden excursions in water quality as in (1) separated by periods of steady or slowly varying flows
(3) longer term, slower variation in water quality due to seasonal or longer term variation in hydroclimatology and associated wasteloads
(4) equilibrium (steady-state) water quality under critical external conditions

A problem motivating the context of (1) is one in which contaminants are flushed into the watercourse by storm runoff, or are mobilized from the bed by the higher stream velocities resulting from storm flows. A dramatic example is the notorious "Black Rise" on the Upper Trinity River. If the watercourse exhibits degraded quality both as short-term storm responses and during the periods of lower flows between such events, a time scale context of (2) may be necessary. Another problem context in which this time scale of variation is required is when the objective is to determine a long-term average response to a variety of storm and nonstorm events. Agricultural land management often necessitates this problem context.

The problem context entailing time scale (3) is similar except that the specific short-term responses to storm events are not central to the occurrence of degraded water quality, so the fine details of storm hydrographs and the water-quality response are not needed. An example would be contaminants stored in the watercourse by runoff events, which then influence water quality long after the storm hydrograph has receded. Another example would be determining the effects of long-term variation in wasteloading or hydroclimatology. In this case, the integrated loads from storm events are needed but not their fine time detail. Yet another special case would be the seasonal variation in a waterbody, such as seasonal freshets in a river, or summer stratification in a lake.

The last time scale (4) results when time variations in loadings, hydrology and water quality response are not material to the management problem. This is the problem context for a point-source assimilative-capacity determination in which the critical conditions are usually summer low flows.

In order for a model to be capable of depicting one of these time scales, it must have an appropriate time resolution in the model operation (including inputs), and must have process formulations that are suitable for that time resolution. One index to the time resolution of a model is the smallest time step the model can accommodate (or for which adequate validation has been accomplished). Whether the processes are properly formulated for a given time scale is a more subtle matter, and hinges on the relative importance of the transport terms in comparison to the source/sink terms for various flow scenarios. Four types of model time resolution are distinguished:

- storm event
- continuous simulation
- slowly varying
- steady state

*Storm event* models are designed to depict the highly variable, immediate response of a storm hydrograph. A *continuous simulation (a.k.a. continuous time)* model includes this storm-response capability but also treats the very different hydrological behavior during the interstorm periods, when the watershed is desiccated by evapotranspiration and infiltration, and interflow plays a greater relative role in producing streamflow. "Continuous" simulation is an unfortunate choice of terminology, because these models are discretized in time, but this is the common patois among modelers. *Slowly varying* models have an integration timestep that is long in comparison to storm hydrographs. Storm loadings, if included in the model at all, are integrated over the model time step. *Steady state* models usually involve a model equation in which the time derivative is assumed *ab initio* to be zero, but there are a few steady-state models that use time as an asymptotic parameter, integrating forward until the solution equilibrates. (In fact, a time varying model can be used to determine the steady state response by this tactic.) A reliable indicator of whether a model is steady-state is if time variation is prohibited in all of the external input parameters.

A list of the models reviewed in this study and the extent to which they would appear to be of potential value in the Texas TMDL process are summarized in Table 2-2. Where a model has been eliminated from recommendation, the principal reason(s) for this is given in the last column. The most common reasons proved to be (1) an inadequate history of usage, as reflected in the technical literature, (2) insufficient demonstrated application to watercourses typical of Texas environments, which includes the extent of field verification that the model has received, and (3) constraints on access to the model, either because it is proprietary or that it is "in development" or limited to "research use." Table 2-2 also lists attributes that may hamper use of the model in a TMDL context, e.g., the model is steady-state only, there is no hydrodynamic or no water quality capability, or the model code may be difficult to apply.

Those models which are recommended for consideration for use by TNRCC are shown in Table 8-1. Table 8-2 categorizes the most likely modeling requirements for Texas TMDL's according to watercourse type and model time resolution, showing how the models recommended for consideration (Table 8-1) meet the requirements of the State. Those combinations of watercourse type and time resolution that are unlikely to be needed for TMDL's are indicated by gray cells in this table.

Table 8-1

Models recommended to TNRCC for consideration for use in TMDL determination
(Agency abbreviations at end of Table 2-2)

| *model* | *source of model* | *remarks* |
|---|---|---|
| | | recommended watershed models |
| HSPF | USGS/CEAM | process models dubious, overparameterized, much overhead due to many options, most of which are irrelevant to Texas TMDL's |
| PRMS | USGS | input demands less than HSPF, but limited water-quality capability, GUI input management system under development |
| SWAT | ARS | includes small lakes & vadose zone, but relies on statistical process models (CN & USLE) |
| | | watershed models of potential special application |
| ANSWERS | NCSU | event model, dated code, but mechanistic process formulae may be useful |
| EPIC | ARS | for agricultural fields, wide range of treatments |
| GLEAMS | ARS | field-scale catchment w/ good leaching depiction, limited utility for manure application problems |
| SWMM | CEAM | may be useful for detailed urban runoff problems |
| WEPP | NSEL | agricultural fields only, but mechanistic process formulae may be useful |
| | | recommended stream and river models |
| DYNHYD | CEAM | link-node 1-D, dated code, hydrodynamics only |
| HSPF | USGS/CEAM | difficult to apply, stream resolution too coarse, hydraulic relations must be input by user |
| QUAL2E | CEAM | steady-state only, useful for low-flow steady-load problems |
| QUALTX | TNRCC | ditto, but kinetics designed for Texas streams |
| WASP | CEAM | transport only, must be coupled with hydrodynamic model, user must supply kinetics |
| | | recommended lake and reservoir models |
| CE-QUAL-W2 | WES | deep stratified reservoirs, application difficult, code may contain bugs |
| DYNHYD | CEAM | link-node 1-D, dated code, may be adaptable to long time variation in unstratified dendritic reservoirs |

(continued)

Table 8-1
(continued)

| *model* | *source of model* | *remarks* |
|---|---|---|
| | | recommended lake and reservoir models (continued) |
| HSPF | USGS/CEAM | treats reservoir as single well-mixed segment, see above comments |
| QUAL2E | CEAM | 1-D steady state, may be applicable to low-flow scenario in shallow mainstem reservoirs |
| QUALTX | TNRCC | ditto |
| WASP | CEAM | transport only, must be coupled with hydrodynamic model, can be applied to time-varying quality in shallow dendritic reservoir |
| | | lake and reservoir models of potential special application |
| BATHTUB | WES | statistical, limited application history, but strategy may be adapted to mechanistic CSTR or 2-layer model |
| EFDC | CEAM | complex 3-D model, potentially applicable to reservoir |
| POM | PU | complex 3-D model, but has been applied to large lakes |
| | | recommended estuary or bay models |
| CE-QUAL-W2 | WES | deep channel estuaries, application difficult, code may contain bugs, useful for stratified channels |
| DYNHYD | CEAM | link-node 1-D, useful for tidal-dominated currents in channel estuary, has been applied to bays |
| QUAL2E | CEAM | 1-D steady state, potentially applicable to tidal-mean of channel estuary |
| QUALTX | TNRCC | ditto, kinetics developed for Texas systems |
| TxBLEND | TWDB | 2-D horizontal, no water-quality capability, limited technical acceptance, most useful for tidal dominated transports in bay regions |
| WASP | CEAM | must be coupled with hydrodynamic model for which DYNHYD or TxBLEND may be suitable |
| | | estuary or bay models of potential special application |
| EFDC | CEAM | 3-D dynamic model, complex to use, insufficient history of application |
| POM | PU | 3-D dynamic model, complex to operate, water-quality capability limited to salinity & temperature |

Table 8-2

Texas TMDL modeling requirements by watercourse type and time resolution and models satisfying requirements.
Combinations not expected to be widely necessary are filled in gray

| | *TIME RESOLUTION* | | | |
|---|---|---|---|---|
| *WATERCOURSE TYPE* | *steady state* | *slow time variation* | *continuous time variation* | *storm event* |
| *field or lumped catchment* | GLEAMS | | SWAT | ANSWERS, SWAT* |
| *watershed* | | HSPF, SWAT, PRSM | HSPF, SWAT, PRSM | HSPF, PRSM, SWMM, ANSWERS |
| *stream/river* | QUAL-TX | DYNHYD/ WASP | | |
| *reservoir, unstratified* | QUAL-TX | DYNHYD/ WASP | | |
| *reservoir, stratified* | | CE-QUAL-W2 | | |
| *estuary reach, longitudinal* | QUAL-TX | DYNHYD/ WASP | | |
| *estuary reach, stratified* | CE-QUAL-W2 | CE-QUAL-W2 | | |
| *coastal embayment, vertically averaged* | | | TXBLEND, POM, EFDC | |
| *coastal embayment, 3-dimensional* | | | POM, EFDC | |

*average conditions over long simulation period

One immediate observation to be made about Table 8-2 is that there are several places in the table, representing combinations of time resolution and watercourse type, for which there does not exist a suitable model. Even for those combinations for which there are one or several models, there are other problems. As discussed in Section 2.1.4, models can be declassified into "compartments" treating hydraulics (i.e., hydrodynamics), transport, waterborne parameter source/sinks (i.e., kinetics), and sediment mobilization and transport. Almost every one of the models shown in Table 8-1 lacks one or more of these compartments, which will hamper that model's utility in a TMDL determination. Even for those that do include all of these compartments, some of the process formulations are inadequate. SWAT, for example, relies upon the SCS curve number method for its runoff hydrology and the Universal Soil Loss Equation for sediment loading, see Section 3.2. This is why the developers of SWAT caution that it should be used to determine long-term average loadings from a watershed, and may not perform well for individual storm events (Dugas, pers. comm., 1999)

The lack of availability of suitable reservoir water quality models is particularly problematic. CE-QUAL-W2 is difficult to apply, does not have sufficient simplified default parameters as an alternative to its overparameterized input requirements, and may contain programming bugs. For some shallow reservoirs QUAL-TX may work, but this is pressing the range of applicability of this type of model.

Another significant lack evidenced by Table 8-2 is a receiving stream model capable of treating the dynamic response of water quality to a storm event. (HSPF, we note, includes a receiving stream submodel, but there is no hydraulic capability, and, moreover, it is a very poor model for water-quality management because of the limited spatial resolution, see Section 3.3.2.)

### *8.3 Conclusions*

The principal conclusions emerging from this review are as follows:

(1) The concept of a TMDL generalizes that of a wasteload allocation to include point source discharges of a pollutant, natural sources, and nonpoint sources that arise from the watershed and airshed of the watercourse. The TMDL is a determination relating to the quality of surface waters, and is based in part upon a target constituent level, which may be a scientifically or publicly mandated concentration, or may be a surface-water standard. Establishment of a procedure of TMDL determination for Texas is particularly problematic because of the range of hydroclimatology in the state, an equally complex range of soils and geology, a variety of topography and vegetation, and—especially—the variety of watercourses. The principal agency for nonpoint source loading is transient runoff deriving from convective storms and the resultant flashy streamflow characteristic of the state. The water-quality response in a receiving watercourse to such events may be immediate and transient, or may be delayed and long-term due to storage of contaminants in the system.

(2) Although there are many watercourse models on the market, there is no one model suitable for all (or even the majority) of TMDL projects anticipated in Texas. Even for specific combinations of watercourse characteristics and problem time scale, the existing models may not be entirely adequate to the problem. For some combinations, there do not exist suitable models.

(3) For watersheds, the most appropriate extant models are HSPF, SWAT and PRMS. Each of these, however, has significant weaknesses and limitations for Texas environments. Many of the process terms in these models may require additional study, validation or re-formulation for application to TMDL's. Between HSPF and SWAT, the deterministic basis of HSPF hydrology and sediment loading is preferable to the empirical basis of SWAT, which employs the SCS curve number and the USLE (see Table 8-1). PRMS appears to have a better formulation of both hydrology and sediment mechanics but lacks a water-quality capability as well as application experience in systems typical of Texas.

(4) For streams and rivers, the most appropriate extant models are QUAL-TX for the steady-state, low-flow-dominated problems, and the DYNHYD/WASP combination for time varying problems. The hydraulic basis and computational strategy of DYNHYD are dated, and there are some indications that it may not handle a dynamic storm hydrograph. There is not available a truly suitable model for the short-time response in water quality resulting from a storm hydrograph.

(5) Many reservoir quality problems in Texas could be addressed by treating the reservoir as a CSTR (perhaps with two layers). Surprisingly, there does not appear to be a suitable, mechanistic model for this configuration (BATHTUB being statistical). For lakes and reservoirs whose quality is dominated by kinetics and the only substantial spatial variation is along the longitudinal axis, QUAL-TX or DYNHYD/WASP may be suitable. This will probably be true for smaller, shallow reservoirs with loading at one end of the system. For those that are deeper, subject to high internal circulations (notably due to power plant operations), or evidence important vertical stratification in water quality, these models will not be suitable. The only proven, but rather undesirable, option for such deeper systems at this point is CE-QUAL-W2.

(6) For one-dimensional estuaries, i.e. tidal or salt-intrusion reaches of a river, there is no suitable model for short time-resolution problems: in this type of system, intratidal variation can be as problematic as storm event response. For management problems amenable to longer time-scale averaging, *viz.* intertidal, QUALTX or DYNHYD/WASP may be appropriate. Validation studies will be necessary, and high dispersion coefficients (e.g., Ward and Montague, 1996) will probably be necessary.

(7) For the large, open, spatially complex system of a coastal embayment, there are three models considered in this review with capabilities for addressing this sort of system: EFDC, POM and TXBLEND. Each might be suitable, but each would require additional development work and field testing to be useful. (However, it is unlikely that many TMDL problems in Texas will require modeling these complex system.)

(8) BASINS as a model shell offers some potentially important advantages, in facilitating set-up of a watershed model, and in allowing preliminary simplified model evaluations. However, there are bugs in the present model that will have to be worked through, in all likelihood it will be better to develop a complete HSPF application rather than relying upon the NPSM version in BASINS, and more accurate receiving water models will be needed for final TMDL determinations than afforded by BASINS.

### *8.4 Recommendations*

(1) Selection of a model for a Texas TMDL determination should be first based upon the type(s) of watercourse involved and the time scale dictated by the water-quality problem(s) of that watercourse. Selection should be further based upon the adequacy of the processes represented in the model. Availability of a model code that includes a (perhaps purportedly) user-friendly interface should not *per se* be a criterion of adoption.

(2) As this project did not include actual operation of the models reviewed, we recommend that the models deemed candidates for consideration (Table 8-1) be subjected to operational testing and evaluation. In some cases, these models are being used in TMDL projects underway in Texas, in which case the modeling task should be expanded to include validation and evaluation of the model. Operation and comparison of two or more candidate models to the same watercourse evaluation can be especially useful.

(3) For those models selected for TMDL application, we recommend that a Texas-specific version of the model and input file be developed, with default values for parameters appropriate for the watercourse under evaluation and possibly process formulations suited for Texas watercourses. This will greatly facilitate the routine operation of the more complex models such as HSPF or CE-QUAL-W2. The issue of "overparameterization" would be met by this approach and by increased emphasis on model validation against field measurements.

(4) We recommend, in particular, that TNRCC develop a "Texas" version of HSPF, with (1) pre-set default values appropriate to the state, (2) new subroutine codes depicting key processes based firmly on current knowledge, again appropriate to the Texas environment, and (3) a considerably simplified user interface. With respect to (2), the mechanistic relations embodied in ANSWERS, ANSWERS-2000, and WEPP, may be the most viable possibilities, since all of these have been separately subjected to field validation. We also recommend that the receiving water component of HSPF in its present form not be used in TMDL evaluations, except for unusual circumstances such as treating a shallow run-of-the-river reservoir which has no longitudinal gradients in quality. There are two options available to TNRCC: (1) export the loading results from HSPF and use to drive a stream/river water quality model appropriate for the system of concern, (2) extend the RCHRES subroutine in HSPF to disaggregate the stream reach into a more highly resolved submodel. We believe the former to be the more practical option.

(5) It has been many years since TNRCC (and its predecessor agencies) has carried out substantial projects in model development. From an early role of a national bellwether in developing and applying the "new" technologies of modeling and rigorous field monitoring to water quality management, the State has retrenched to a reliance on off-the-shelf software, or its venerable fallback QUALTX. The requirements for TMDL determinations in the State will necessitate at least major adaptations and modifications to the candidate models listed here, and in some cases new models. We recommend that TNRCC initiate a program of model development and validation addressing the specific features of Texas watercourses and hydroclimatology to meet this need. The TMDL process offers an ideal vehicle for coordinating with such a model development program, as it will marshal the data resources and engineering effort needed.

(6) Specific models recommended for development are:

(a) a low-dimensional model for a lake environment, depicted as a CSTR or a two-layer system, including nutrient kinetics and sediment-water interactions. A model formulated along the lines of BATHTUB, but with less reliance on statistical formulations of nutrient responses, would be especially useful.

(b) extension of QUAL-TX to a fully dynamic format, including integration with a suitable one-dimensional hydrodynamic model, to specifically simulate storm hydrograph effects on water quality

(c) a "Texas BASINS" model, incorporating the most useful aspects of the GUI of BASINS with versions of HSPF, QUAL-TX, and other watercourse models most suitable for Texas environments. see (4) above

(d) a combination of the lake-environment model (a) with the time-varying longitudinal model (b) to address dendritic shallow reservoirs, including capability for various dam operations, power-plant circulations, and tributary loading functions

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