TECHNICAL GUIDANCE FOR ESTABLISHING TOTAL MAXIMUM DAILY LOADS (TMDLs):

Involving CSO and Stormwater Point Sources and Nonpoint Sources

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Introduction

The purpose of this document is to provide agencies responsible for developing Total Maximum Daily Loads (TMDLs) with guidance on the identification, selection, and use of technical frameworks or models which combine point and nonpoint, steady and episodic source components. In April, 1991, the U.S. Environmental Protection Agency (EPA) released a document entitled, "Guidance for Water Quality-based Decisions: The TMDL Process" (EPA 440/4-91-001). That document provides broad outlines for several aspects of the TMDL process including the regulatory basis, EPA policies and objectives, methods overview, and agency responsibilities. The present guidance is intended to augment that report by providing greater detail on actual methods of TMDL development, with particular emphasis on how to model wet-weather point and nonpoint source pollutant loading and resulting impacts within a given waterbody.

Historically, nonpoint source (NPS) pollutant runoff analyses and point source (PS) waste load allocations (WLAs) have often been separated. WLAs have focussed on municipal and industrial wastewater discharges, and only recently has greater attention been given to control of wet-weather, episodic PS discharges, such as urban stormwater and combined sewer overflows (CSOs). In fact, it is typical for the States and EPA to have separate groups addressing wastewater discharges, episodic urban PS runoff, and NPS runoff, and these separate groups may not interact on a frequent basis. With EPA's recent emphasis on TMDL development and watershed protection, however, these program functions are coming together. Agencies responsible for the the implementation of the water quality program are quickly learning that successful water quality management depends on an integrated PS and NPS control program. The challenge, therefore, has become how to develop analytical frameworks that support combined steady PS, episodic wet-weather PS, and NPS pollutant loading and impact analyses.

Integrating steady PS, wet-weather PS, and NPS analyses is not always a simple task. Most of the commonly used WLA methods, as well as many State water quality standards, are based on the concept of a relatively steady pollutant load. Steady loads are of greatest concern when dilution flows are at their lowest point in the receiving waterbody (i.e., during extended drought conditions), and are readily analyzed in terms of design low flows of a given probability of recurrence. In contrast, wet-weather loads vary significantly in time and are typically greatest during high precipitation periods.

Determining how to combine these episodic loading events into a decision-making framework originally based on design-flow analysis of PS impacts is often a formidable task. Nonetheless, integration of episodic, wet-weather loads into the TMDL process is a challenge that must be met to achieve an effective management strategy which protects the beneficial uses of impacted waterbodies. Accomplishing this integration will often require the use of mathematical models to assess control strategies and predict the frequency of water quality excursions. However, the presence of episodic, wet-weather loads can lead to difficult and complex modeling problems.

This document provides guidance on the identification and selection of an appropriate modeling strategy for estimating TMDLs involving episodic, wet-weather loads. Such a strategy must recognize the physical complexity of many episodic loading problems, yet also should recognize the practical constraints of time and money available to complete modeling studies. How do we balance these competing needs? Conceptually, the answer is straightforward: Use as simple a model as is appropriate to calculate and apportion the TMDL. However, trying to pin down exactly what constitutes an appropriate level of complexity is a thorny problem; its resolution is a principal aim of this document.

In addition to formulation of modeling strategy for estimating TMDLs, this guidance provides: (1) a survey of simple and complex models for estimating wet-weather loads and receiving water flows and impacts; (2) guidance on the development of monitoring plans to support the TMDL estimation process; (3) information on data collection and sampling for TMDLs involving wet-weather loads; (4) techniques for the implementation and interpretation of simulation models; and (5) a selection of useful case studies. The guidance is intended to be of general applicability to many types of loading problems and receiving waterbodies, with extensive reference to other useful EPA guidance.

Chapter I. TMDL Problem Assessment, Goal Setting, and the Role of Models

Purpose: This chapter provides a brief overview of the concepts involved in establishing a TMDL for watersheds with both point sources (PS) and nonpoint sources (NPS). It sets the stage and provides the context for the main purpose of this document, which is to provide guidance on the use of modeling to establish TMDLs which involve episodic, wet-weather point and nonpoint source loads. In this context, wet-weather point sources refer principally to permitted stormwater and combined sewer overflow (CSO) systems. The regulatory context of TMDLs, which are designed to achieve compliance with the requirements of the Clean Water Act (CWA), provides the basis for assessing what questions should be answered in a modeling application.

1.1 What exactly is a TMDL?

The concept of Total Maximum Daily Load (TMDL) was established in 1972 with the Clean Water Act (CWA) amendments to the Federal Water Pollution Control Act. The reference to TMDL is found in Section 303(d) of the Act which states,

> "Each State shall establish for the waters identified... [as water quality limited]..., and in accordance with priority ranking, the total maximum daily load, for those pollutants which the Administrator identifies under section 304(a) as suitable for such calculation. Such load shall be established at a level necessary to implement the applicable water quality standards with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality."

For the most part, however, implementation of this part of the act was delayed for approximately 20 years as both EPA and States struggled to determine what exactly constituted a TMDL.

In 1991, EPA took a giant step toward clarifying the meaning of TMDL with publication of a guidance document titled, "Guidance for Water Quality Based Decisions: The TMDL Process." Included among its contents are clarification of EPA's policies and principals applicable under the CWA, the relationship of TMDL development to the water quality planning and management process, and EPA/State responsibilities in the development and implementation of TMDLs. Because of the comprehensive nature of that document, it is not possible to recapitulate that material in full detail. Therefore, the reader is encouraged to review that document as background material for preparation in use of this guidance document. However, several of the main points will be summarized below, and some will be revisited to provide further clarification on the meaning of TMDL.

The statutory reference to TMDL in Section 303(d) has been interpreted by EPA to mean that TMDLs are comprised of the sum of individual wasteload allocations (WLAs) for point sources, and load allocations (LAs) for nonpoint sources and natural background levels that are established for a given waterbody segment such that water quality standards (WQSs) are maintained (U.S. EPA, 1991a). In addition, the TMDL must include a margin of safety (MOS), either implicitly or explicity, that accounts for the uncertainty between pollutant loads and the quality of the receiving waterbody. Conceptually, this definition is denoted by the equation:

$$\textbf{TMDL} = \Sigma\ \textbf{WLAs} + \Sigma\ \textbf{LAs} + \textbf{MOS}$$

where: TMDL →→→ maintains WQSs

Even with this clarification, however, several issues remain regarding what practically constitutes a TMDL. Certain questions are routinely being asked by State and EPA Regional personnel responsible for implementing TMDLs. Because these questions require practical answers in order for TMDL development to move forward in a consistent and orderly fashion, they are addressed below item by item:

Question: Is the TMDL comprised of all parameter restrictions that collectively protect water quality, or is there a separate TMDL for each water quality parameter?

Answer: EPA recommends that there be one TMDL per parameter for a defined waterbody management unit. While - under this approach - several TMDLs may be required to protect water quality standards, practical considerations necessitate that control strategies address specific parameters. It is possible, however, that several types of parameters will have to be controlled in order to address a single water quality standard. In these cases, the administering agency should make its own determination on the most effective and efficient way to establish and track the TMDL (i.e., as separate or collective requirements).

Question: Do TMDLs only apply to parameters for which numeric criteria have been adopted?

Answer: No, TMDLs are intended to address all State water quality standards including narrative criteria, numeric criteria, designated uses, and antidegradation provisions. Thus, parameters such as physical habitat and biological integrity are included.

Question: Are TMDLs always to reflect a single numeric value which constitutes the total loading capacity for the given waterbody?

Answer: No, it is not realistic to think that a single total loading restriction will always be definable for a given waterbody. The loading capacity (LC) or assimilative capacity (i.e., the greatest amount of loading that a waterbody can receive without violating water quality standards - 40 CFR 130.2(g)) of a waterbody does not necessarily reflect a fixed amount of loading. In this regard, the "pie" diagram commonly used to explain TMDL components (see figure 1-1) may actually be misleading because of its simplistic configuration. One tends to think of a pie as a fixed object that can be "sliced" or allocated into readily measurable parts. However, assimilative capacity often varies in time and space due to the dynamic, sometimes random nature of the ecological features (e.g., physical, chemical, and biological) which comprise a waterbody and its watershed. Thus, rather than being represented by a single numeric value, a TMDL is often comprised of a combination of management strategies for a given pollutant that collectively protect water quality standards. Those strategies may include seasonal or multi-level controls to address this variation.

In this manner, the TMDL at the mouth of a watershed is not simply the critical flow at the mouth of the watershed multiplied by the ambient criteria. Such a definition would ignore the assimilative properties (e.g., oxidation, volatilization, nitrification, sediment adsorption, etc.) of the watershed above the mouth. It is quite possible to have a much greater load distributed throughout a watershed that will not result in a violation of water quality standards than would be allowed at a fixed point at the mouth of the watershed. Alternatively, one could conceive of situations where allowable loading could increase as long as instream concentrations are maintained below levels necessary to protect the standard (e.g., cases where wasteflows dominate streamflow and increase without increases in pollutant concentration). Thus, the TMDL may be thought of as "a tool for implementing State water quality standards [which] is based on the relationship between pollution sources and in-stream water quality conditions. The TMDL establishes the allowable loadings or other quantifiable parameters for a water body and thereby provides the basis for States to establish water quality-based controls. These controls should provide the pollution reductions necessary for a waterbody to meet water quality standards." (U.S. EPA, 1991a).

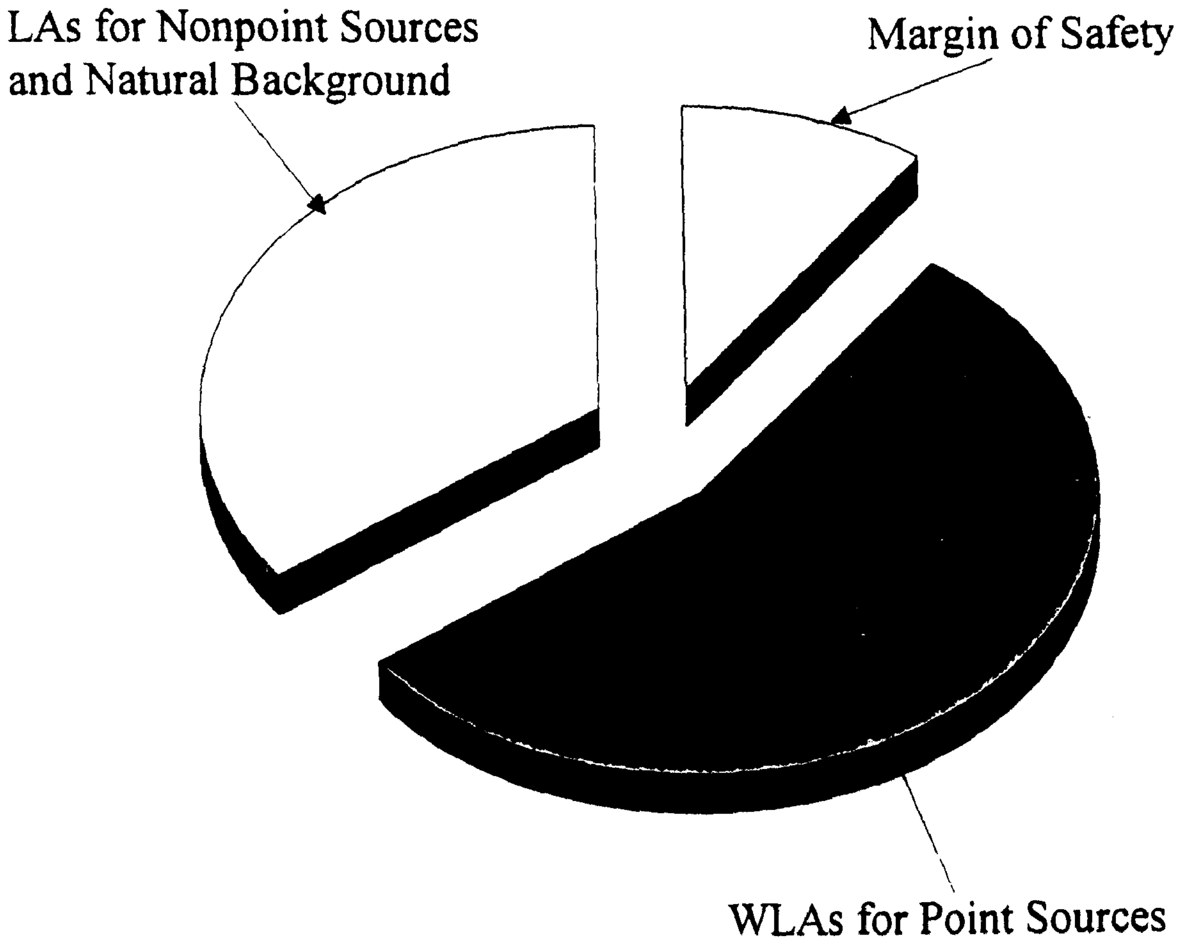

Figure 1-1. Allocation of Waterbody Assimilative Capacity

Question: Is there a specific management unit (i.e., size of watershed) for which the TMDL must be established?

Answer: Management units should generally be chosen to fit the extent of the problem being addressed. Therefore, there are no regulatory requirements for choice of waterbody segments. There are, however, a few conventions that make TMDL development more technically and scientifically sound. In general, management segments should be extended upward within a watershed to the basin boundaries. Since TMDLs include nonpoint source components, use of the watershed as a management unit ensures that all potential contributors can be addressed. While an isolated portion of the watershed may be the only segment exhibiting impairment, contributions to that impairment may be coming from upstream sources.

As with any rule, there are exceptions to it. There may be natural breakpoints (e.g., lakes or impoundments, etc.) or the system may be so large that it is more practical to break it into segments. EPA has been recommending that States track the TMDLs using the Waterbody System, so the management unit that corresponds to what States input to that system may be a logical choice for those that use it.

It may also be helpful to develop and track TMDLs according to hydrologic units that correspond with other agencies dealing with water such as the USGS and SCS. Even if management units are not exactly the same size between coordinating agencies, sharing and interpretation of information can be aided if the management units "nest" within each other so that information can be aggregated and compared at some level. For example, the State of Virginia uses SCS hydrologic units which have been nested within the USGS cataloging hydrologic units. This can be very important where tools such as geographic information systems (GIS) are used to manage, interpret, and present watershed information.

Question: Should the margin of safety (MOS) term always be made explicit and does it also include a reserve for future allocations?

Answer: The TMDL Process document (U.S. EPA, 1991a) indicates that the MOS is typically incorporated with conservative assumptions in calculations or models used to develop TMDLs (i.e., implicitly). This implies that the estimate is biased on the side of safety. However, the premise that assumptions actually provide conservative estimates may need to be scrutinized on a case-by-case basis. As the EPA Process document indicates, where an additional MOS is needed, an explicit MOS term can be added to the equation.

It should be recognized, however, that in some cases the TMDL developer may not be able to establish an MOS to completely account for uncertainty. This can occur when the uncertainty is so large that it becomes impractical to remove that large of a portion of loading. It also can occur when assimilative capacity has been exhausted and existing sources are already at state-of-the-art control levels. In these cases, agencies are often struggling just to find additional ways to reduce pollution sources to get near targeted reduction levels, and there is no additional capacity to hold in reserve to account for uncertainty.

On the other hand, the TMDL developer may run into cases where the sum of WLAs and LAs under current conditions leaves a portion of the assimilative capacity remaining. This is fine as long as the TMDL reflects control strategies that collectively keep sources of pollution at or below the loading capacity. In fact, this may be done intentionally to allow for future uses and thus the TMDL becomes an effective long-term planning tool. Translating TMDL strategies into LAs and WLAs will be addressed further in chapter 7.

1.2 The Water Quality Planning Context of the TMDL

Prior to beginning the TMDL development process, it's important to understand the overall planning context in which the TMDL development process fits. The TMDL developer does not act in isolation from the rest of the water quality management program for a given State. Rather, TMDL development occurs in response to water quality assessment and subsequent prioritization of a waterbody for management action. This process is known as a water quality-based approach to pollution control and is well outlined in chapter 2 of the EPA manual, "Guidance for Water Quality-based Decisions: The TMDL Process." The approach can be simplified into the following five step process (illustrated in figure 1-2):

Step 1: Identification of Water Quality-Limited Waters
Step 2: Priority Ranking and Targeting
Step 3: TMDL Development
Step 4: Implementation of Control Actions
Step 5: Assessment of Water Quality-Based Control Actions

Section 303(d) of the Clean Water Act (CWA) requires States to establish TMDLs for waterbodies where federally-based minimum guidelines for wastewater discharges

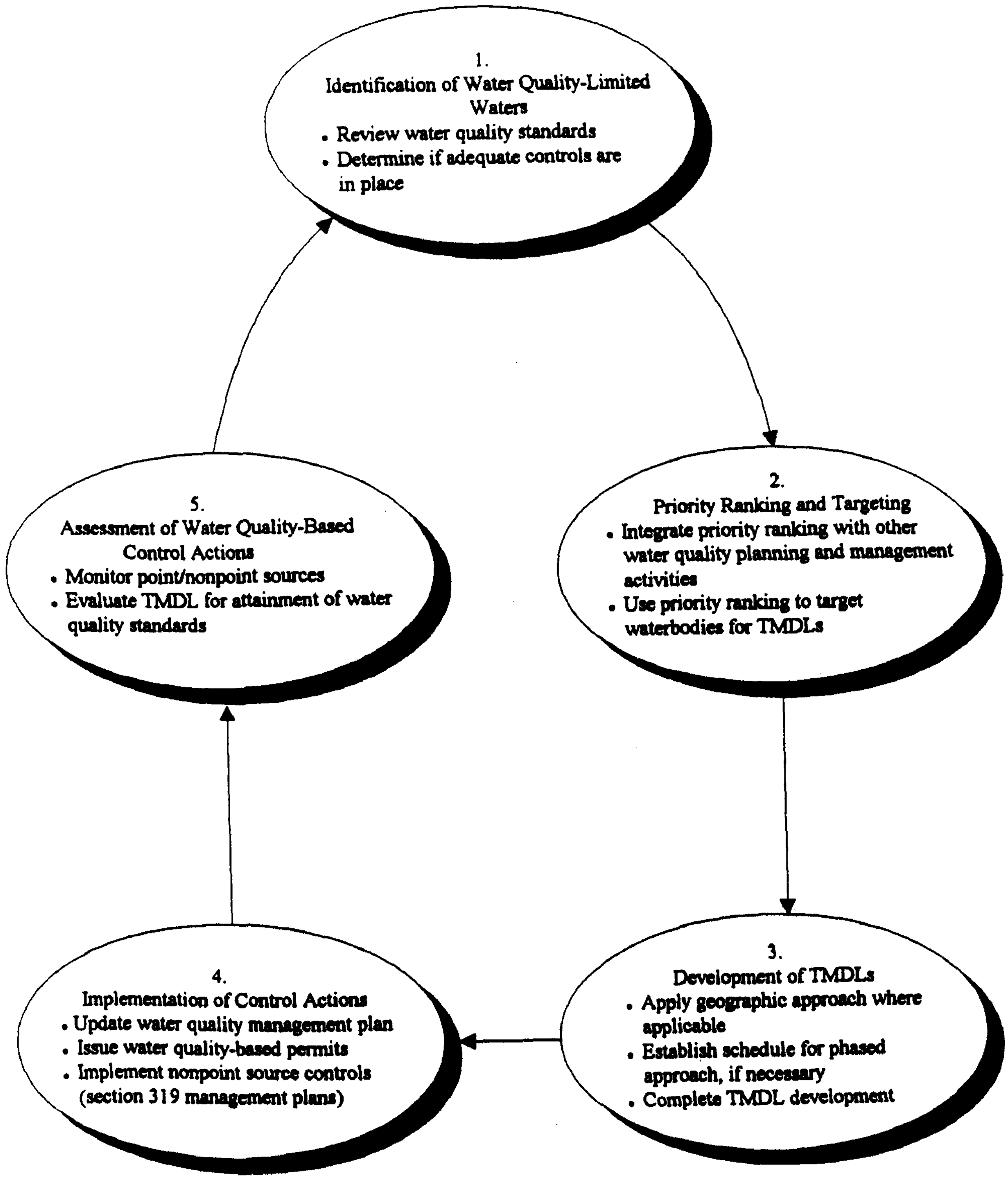

Figure 1-2. General Elements of the Water Quality-Based Approach (from U.S. EPA, 1991a)

are not stringent enough to protect water quality standards. Through the 303(d) process, States must identify and prioritize those waters in need of TMDL management strategies. Thus, to a large extent, water quality problems have already been subject to a preliminary assessment and the major causes and sources of impairment have been identified by the time at which the control agency is ready to establish a TMDL for a given waterbody.

The technical core of the process ; contained in Step 3. In this step, the TMDL developer begins with the identification of a prioritized, targeted waterbody and develops an analysis sufficient to develop appropriate control actions. It is in this area that modeling typically plays an important role, and subsequent chapters can be thought of as providing technical guidance in support of the completion of Step 3 of the TMDL development process for TMDLs involving wet-weather point and nonpoint sources.

1.3 The Relationship of TMDL Development to Watershed Protection

The method that EPA has chosen to implement the Section 303 planning process recognizes that States cannot address all identified water-quality limited waterbodies simultaneously. After recent revisions (see 57 Federal Register 33040; Friday, July 24, 1992), the regulations implementing Section 303(d) now only require that States identify the waterbodies that they have prioritized for TMDL development over a two year period. A critical part of the TMDL process is realistically determining which waterbodies can be handled at a given time and establishing a logical schedule for TMDL development.

EPA also recommends that development of TMDLs proceed along a geographic basis (i.e., watershed), since water quality concerns are usually area wide and caused by multiple sources (U.S. EPA, 1991a). Some States (e.g., NC and SC) have established their TMDL development schedule around a five-year basinwide planning cycle. Representatives of these States have indicated that using the "basin" as the unit of management has provided for greater efficiency, thereby allowing the development of a larger number of more comprehensive TMDLs in a given amount of time (U.S. EPA, 1992). Elements of the basinwide planning process include comprehensive water quality monitoring, assessment, waterbody management prioritization and modeling analyses, all of which combine to provide a ready-made foundation for TMDL development. These activities are scheduled over the five-year cycle such that by its end, all TMDLs in the basin are implemented via an approved basinwide management plan. Thus, TMDL problem assessment and goals are established within the watershed management planning process. This allows the TMDL developer to see the "big picture" of TMDL needs within the basin and consequently provides better background information to make critical decisions regarding the level of complexity and resources that will be devoted to a given TMDL.

For example, in North Carolina a basin is re-examined every five years and the basinwide management plan is updated accordingly. During the first part of the five-year period, the State identifies known problem areas and establishes intensive field sampling plans for model development. Both ambient and intensive monitoring occurs during years one through three of the five year period for that basin, and model development begins as information is collected. Model development and application should be completed during year four and TMDL results should be incorporated into the plan that goes through the public review and adoption process in year five. Therefore, those prioritized waters that did not receive a TMDL in one planning cycle can be targeted for completion within the next cycle. This occurred with their recent adoption of a plan for the Neuse River Basin.

The State of New Jersey recently proposed a rule that would formally adopt a TMDL approach within a basinwide planning context which specifically addresses modeling concerns in a cyclic review format. After dividing the State into five geographic regions and establishing a schedule for water quality program activities for each region, New Jersey's action plan includes the following TMDL development components for each watershed:

1. Select the parameters needing a water quality model for each watershed in a given permit cycle.

2. Determine the appropriate model complexity and accuracy for the TMDL for each parameter that needs to be addressed.

3. Determine the priority for completing the TMDL analysis for each parameter for each watershed.

4. Complete the TMDL for each parameter of interest.

Similar to North Carolina's program, the New Jersey approach would handle TMDL development within a five-year cycle:

Year 1: Preliminary instream data for all toxics are gathered at selected sampling points within the watershed. These data will be used to assist in determining which parameters might be targeted during the first permit cycle. All other available instream data are collated. Effluent data collected as a result of the monitoring requirements in the discharge permits are collated. A list of the potential parameters to be targeted is selected from this limited database. Preliminary evaluation may be undertaken with scoping models at this stage.

Year 2: Instream and effluent data are collected for the targeted parameters. If conventional and nonconventional parameters have not yet been addressed in the watershed, or in portions of the watershed, those parameters will be addressed in the sampling and subsequent model development.

Year 3: The hydrodynamic water quality model is developed focusing on selected parameters. Preliminary wasteload allocations for point sources and load allocations for nonpoint sources are developed for the selected parameters.

Year 4: The public input process for the selected parameters is completed. Discharge permits are drafted at the end of TMDL/WLA/LA public input process.

Year 5: By the end of Year 5, the affected permits in the watershed are issued as final permits.

During a given 5-year cycle, not all parameters in each watershed can be addressed. The objective is to begin with those parameters which have the highest priority. Lower priority parameters can be addressed in subsequent cycles.

1.4 Phased TMDL Development and Basinwide Planning

Problem assessment and goal setting for TMDLs can be limited by lack of information on sources and loads. In particular, accurate determination of nonpoint source loads is often difficult. However, EPA regulations (40 CFR 130.2(g)) provide that load allocations for nonpoint sources "are best estimates of the loading which may range from reasonably accurate estimates to gross allotments..." Thus, an incomplete understanding of nonpoint source loading to an impaired waterbody should not delay the implementation of water quality-based control measures.

To address the development of TMDLs in cases where estimates are based on limited information, EPA (1991a) has suggested use of a phased approach. The phased approach is defined "as a TMDL that includes monitoring requirements and a schedule for re-assessing TMDL allocations to ensure attainment of water quality standards." EPA guidance also states that the phased approach may be "necessary" where nonpoint source controls are involved. In order to allocate loads among both nonpoint and point sources, there must be reasonable assurances that nonpoint source reduction will in fact be achieved. However, the only federally enforceable controls under the CWA are those for point sources through the NPDES permitting process. With the phased approach, the TMDL includes a description of the implementation mechanisms and the schedule for the implementation of nonpoint source control measures.

1.4.1 Phasing Management Actions

The phased approach allows for an incremental approach to managing water-quality limited waters. Control agencies can either implement interim measures while a final TMDL strategy is being developed or they can hold off on implementing a strategy until the appropriate information is collected. In either case, however, the agency should develop a firm schedule for collection of needed information and final development of the TMDL management strategy.

1.4.2 Accomodating Long-Term Modeling Studies

A phased approach may be a good choice where extensive model development is required to provide TMDL developers with a mechanism for predicting the outcomes of alternative management strategies. Complex modeling typically requires a considerable amount of time for data collection, model calibration, and model validation. Therefore, a phased approach can allow the developer time to establish a coherent modeling strategy. Simple models can be used for initial scoping of the problem and preliminary assignment of WLAs and LAs. Often, earlier phases of data review and analysis focus on establishing relative loadings and therefore require less precision. The scoping exercise can lead to the formulation of a focused sampling plan, which in turn provides the basis for more detailed modeling in later phases of the process where a higher degree of precision may be warranted.

1.4.3 Piggy-backing With Basinwide Planning

The phased approach works well within a basinwide planning context, in which a regular cycle of review of TMDLs is established by scheduling concurrent review of all NPDES permits within a basin. Phased TMDLs are naturally complementary to regular basinwide TMDL reviews, and the timing of the phases can be timed to the overall review process. In North Carolina, the phased TMDL has been integrated with the five-year review basinwide review cycle. Of the more than fifty TMDL strategies outlined in North Carolina's 1993 plan for the Neuse River Basin, several were implemented in a phased format. For instance, while a TMDL strategy was adopted for the upper 180 mile stretch of the main stem of the river using a field calibrated QUAL2E model, an interim strategy was adopted for the lower estuary portion of the basin while a more complex, multi-dimensional, hydrodynamic model is being developed. The State will include any modifications to the Neuse Basin plan in 1998 as it comes up for renewal.

1.5 Matching the TMDL Analytical Framework to Management Goals

Choosing the right analytical framework (e.g., model or combination of models) for TMDL development requires the developer to consider factors that match management goals. The goals can range rom general water quality program objectives to meeting specific criteria within the waterbody of concern. In addition, the developer must evaluate methods to achieve the goals in light of resource constraints. The following details some of the factors that will affect the decision regarding model choice.

1.5.1 Meeting Water Quality Standards

The overall goal of TMDL development is to achieve compliance with water quality standards (WQSs), including narrative criteria and designated uses. Modeling requirements for the TMDL are necessarily constrained by the need to provide answers that address the excursions of WQSs, and thus depend on the form in which these WQSs are expressed (i.e., narrative versus numerical; frequency of excursion allowed; etc.).

TMDLs can address not only specific chemical concentration requirements, but other types of WQSs as well. As stated in EPA's Guidance for Water Quality-based Decisions...,"..it is becoming increasingly apparent that in some situations water quality standards - particularly designated uses and biocriteria can only be attained if non-chemical factors such as hydrology, channel morphology, and habitat are also addressed. EPA recognizes that it is appropriate to use the TMDL process to establish control measures for quantifiable non-chemical parameters that are preventing the attainment of water quality standards..."

In some cases, this means controlling parameters for which there are no specific standards. For instance, States rarely adopt instream phosphorus standards since the nutrient by itself is not a threat to human health or aquatic life and habitat. Rather, it is the eutrophication problems (e.g., substantial fluctuations in dissolved oxygen, excessive algae growth, fish kills, etc.) that are associated with excessive amounts of the nutrient being loaded to the system. The standards tend to reflect the response variables. However, as relationships are formalized between the response and dependent variables, TMDLs can be established for parameters for which no formal criteria have been adopted.

The statistical form of the standard should also be considered in making a decision on the appropriate model framework. Do we need to calculate a long-term average, a "typical" concentration at design flow, or actually evaluate the probability of water quality excursions? EPA recommends that water quality criteria statements should be developed in a duration-frequency format, to include requirements that a given concentration not exceed a critical value on average more than once in a given return period. As described in U.S. EPA (1991b), EPA criteria are developed as national recommendations to assist States in developing their standards and to assist in interpreting narrative standards. EPA criteria or guidance consist of three components:

- Magnitude: How much of a pollutant (or pollutant parameter such as toxicity), expressed as a concentration, is allowable.

- Duration: The period time (averaging period) over which the instream concentration is averaged for comparison with criteria concentrations. This specification limits the duration of concentrations above the criteria.

- Frequency: How often criteria can be exceeded.

(Note: While these components are often thought of in regard to numeric standards, its important to point out that the concepts of magnitude, duration, and frequency can be applied to non-chemical water quality problems as well.)

When expressed in this form, a typical aquatic life water quality criteria statement is recommended to contain a concentration, averaging period, and return frequency. For instance, U.S. EPA (1985) recommends that water quality criteria for the protection of aquatic organisms should typically be expressed in a generic form as follows: "...aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of [the pollutant] does not exceed [the lower of the chronic-effect or residue-based concentrations as the criterion continuous concentration] more than once every three years on the average and if the one-hour average concentration does not exceed [the acute effect-based criterion maximum concentration] more than once every three years on the average."

A duration-frequency criteria statement directly addresses protection of the waterbody; that is, it is expressed in terms of the acceptable likelihood of excursions of WQSs. While this appears ideal, it may not always be practical, as it requires estimation of long-term averages. Many states rely instead on the older concept of critical flows (e.g., 7Q10). Setting a standard based on a given critical flow can be shown to be equivalent to a specification of an average duration and frequency of excursion, at least for a load which is independent of flow. However, the concept of a single critical flow may not make sense when dealing with wet-weather episodic loads, as these are likely to be correlated with flow; thus the high loads are very unlikely to occur simultaneously with critical low-flow conditions. The critical flow based LA may then be overly conservative for parameters with a long response time, but might not be protective enough for a nonconservative pollutant, such as a toxic with a short response time, where critical concentrations are achieved not at low flow but during storm runoff events. A more accurate analysis would require predicting the actual duration and frequency of WQS excursions.

The modeling analysis will generally need to provide answers which match the form in which WQ criteria and permit conditions are expressed. If criteria and permit conditions take, or are recommended to take a duration-frequency form, then modeling for TMDLs should provide similar information: that is, the modeling and monitoring activities should eventually result in an estimate of the frequency distribution of receiving water concentrations, rather than just worst-case and/or average estimates. This is particularly important for episodic, wet-weather loading, as the concentrations and impacts of these types of sources are sensitive to variability in both runoff rates and streamflow. However, worst-case estimates (which imply use of a large MOS) may be useful in the early stages of phased TMDL development due to their relative simplicity.

1.5.2 The Role of Models in the TMDL Development Process

The link between WQSs and pollutant loads is usually provided by a combination of modeling and monitoring. In addition to addressing the WQSs, the TMDL developer must consider several other factors including the amount of information available on which to base the decision, the complexity of the problem, spatial and temporal resolution, and the availability of program resources for TMDL development and implementation. The first issue that the TMDL developer faces after receiving a prioritized waterbody is whether enough information already exists to develop a TMDL and whether a modeling analysis is needed to help establish that TMDL. The answer to this question will determine the future level of effort that needs to be employed and may lead to a series of decisions regarding model choice and TMDL development (see Figure 1-3).

Figure 1-3 illustrates the general flow of the TMDL development process, with particular emphasis on the decision-making junctures where model development is required. The remainder of this guidance document will focus on components of this decision-making framework and how they apply to TMDL development for problems involving wet weather point and nonpoint sources.

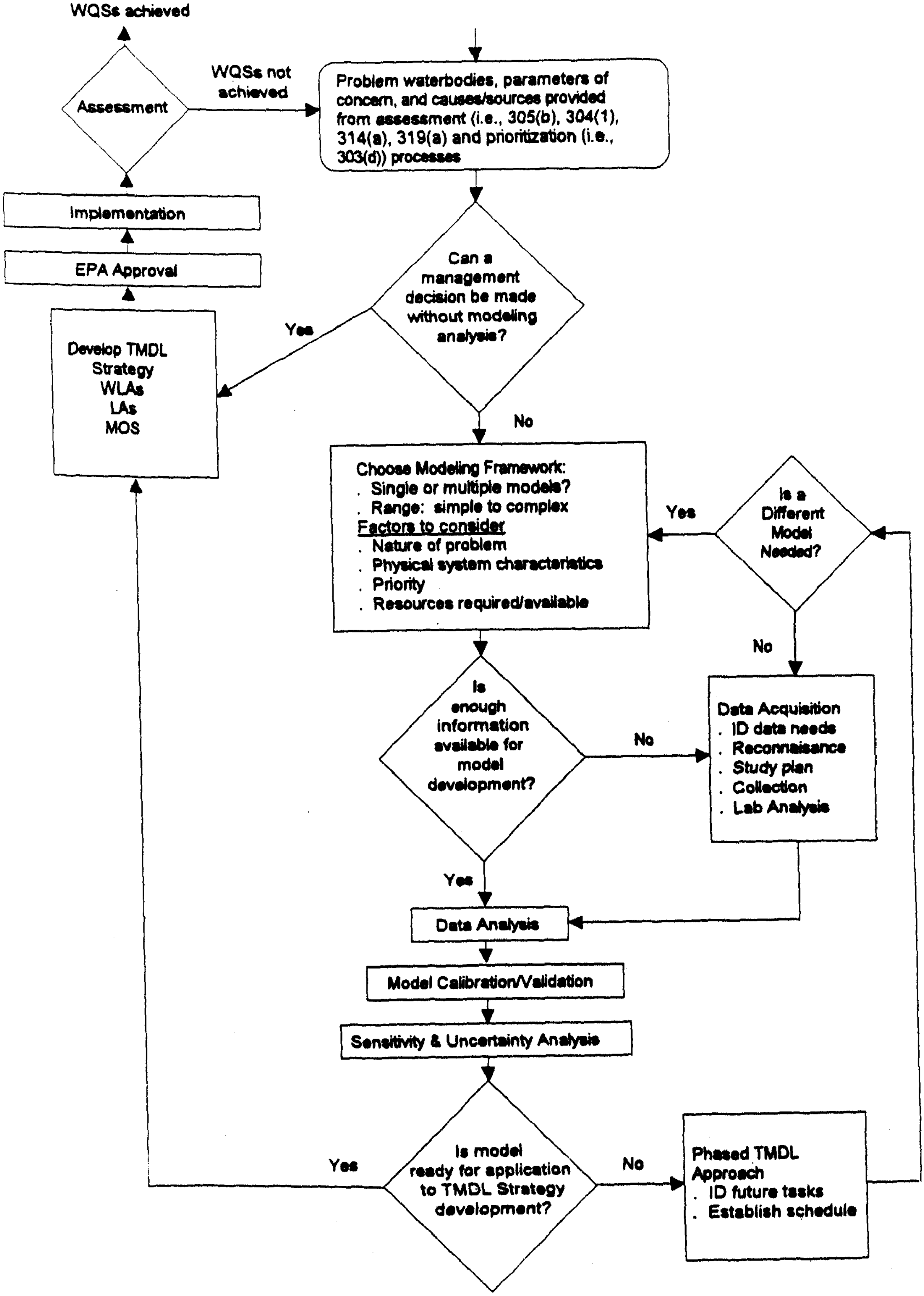

Figure 1-3. Flow Chart of the TMDL Development Process

1.5.3 Key Issues in Modeling for TMDLs

What sort of models should be used to establish a TMDL? The focus of the remainder of this guidance is the identification and application of models for calculating TMDLs which involve episodic, wet-weather loads. Key issues to be addressed in subsequent chapters include the following:

- *How should episodic, wet-u ather PS and NPS be represented in modeling for TMDLS?* TMDLs including wet weather point source and NPS necessarily involve loadings which are episodic, or unsteady, dependent on the pattern of rainfall, and which must be described statistically. Further, the TMDL will often address the interaction of PS and NPS on a watershed basis. Detailed simulation of PS and NPS loading in response to precipitation events can require extensive work with complex models. However, simpler approximations may also be available. What level of representation of the wet-weather PS and NPS loads is appropriate in terms of protection of the receiving waterbody?

- *What is the appropriate level of model complexity?* When is a complex model needed? And when is a simpler modeling approach appropriate? The questions are simple, but the answers are not. "Complexity" may refer to the hydrodynamic or pollutant transport component of models. These components do not necessarily need to be at the same level of complexity. Similarly, it may be appropriate to model NPS and receiving waters at different levels of complexity. Although wet-weather sources may be at issue, perhaps only relative long-term comparisons are needed and thus the complexity of the mechanisms can be ignored. In addition, increasing model complexity does not necessarily increase predictive accuracy. In general, we expect project costs to increase as model complexity increases. How do we strike the appropriate balance between complexity, accuracy and cost?

- *When are simple models appropriate?* It is clear that simple models have a definite place at the scoping level of most problems. That is, before sufficient data are available to undertake a complex model, approximate answers can be obtained using simpler approaches, as described in Chapter II. The results can provide the basis for the development of a monitoring plan, as well as the development of a more complex model of the waterbody. However, there will also be cases where it may be appropriate, or necessary, to calculate the TMDL using simple models alone. This may be adequate if (1) the uncertainty introduced into the analysis by use of the simplifying assumptions can be and is evaluated (at least qualitatively) and is incorporated into the margin of safety, and (2) the spatial resolution of the model is adequate to address the appropriate WQSs.

When are complex simulation models required? Some States have proposed that complex hydrodynamic models will be required to establish TMDLs in most situations involving wet-weather PS and NPS loads. Such situations may require use of complex models to achieve an accurate representation of response to specific precipitation events. However, it may be possible to meet the goals of the CWA and comply with the intentions of the TMDL process by using simpler, approximate models. The simpler models are attractive on the basis of lower cost of implementation, but may misrepresent the actual response of the waterbody. If resources are not available to apply complex models to all TMDLs, what are the conditions in which use of complex models is most essential to provide protection of the waterbody designated uses?

References, Chapter I

U.S. EPA. 1992. Watershed Cost Survey. Assessment and Watershed Protection Division.

U.S. EPA. 1991a. Guidance for Water Quality-based Decisions: The TMDL Process. EPA 440/4-91-001. Assessment and Watershed Protection Division.

U.S. EPA. 1991b. Technical Support Document for Water Quality-based Toxics Control. EPA/505/2-90-001. OWEP/OWRS.

U.S. EPA. 1985. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms. NTIS PB85-227049.

Chapter II. Use of Simple Models for TMDLs

Purpose: Chapter II will provide more information on how simple scoping methods can be effectively used for assessment and TMDL development, particularly where PS and wet-weather NPS pollutant loading must be considered jointly in the modeling analysis. Functions of simple models are discussed, along with factors to consider in the trade-off between model accuracy and the cost of implementation.

2.1 Avoiding the Automatic Tendency Toward Complexity

Given the complexities of the transport and ecological processes in most waterbodies, the modeler may be tempted to think that a complex model is required to capture the behavior of the system. However, it is important to recognize that capturing complexity may come at a high price. It also may not be necessary to address the issue at hand, which is compliance with WQSs and protection of designated uses. For example, although the real time transport of nutrients from various land use activities, stormwater runoff, and point sources in a lake's watershed may be complex, capturing that complexity may not be relevant to a TMDL if the lake's eutrophic response is more a function of average total seasonal or total annual loading. As a general rule of thumb, simple models are preferred when they satisfy the objectives of the user, because they are usually developed at a lower cost and their basis is typically better understood by decision-makers and stakeholders. Therefore, simple scoping models should not be ignored or bypassed in favor of complex simulation models without an assessment of whether a simple model can provide an adequate answer.

There are many factors that a modeler can consider in answering the question of how complex a model is needed. Chapter IV will cover these factors in greater detail, but in general the factors include technical, regulatory, and user criteria. While it may be a challenge to find the right balance among these factors, it may help the modeler to keep a few modeling fundamentals in mind when going through this decision process (Donigian and Huber, 1991):

1. Have a clear statement of project objectives. Verify the need for quality modeling.

2. Use the simplest model(s) that will satisfy the project objectives.

3. To the extent possible, utilize a quality prediction method consistent with available data.

4 Only predict the quality parameters of interest and only over a suitable time scale (i.e., don't attempt intra-storm variations in quality unless it is necessary).

2.2 Defining What is "Simple"

There is no explicit definition of what constitutes a simple model; rather, the definition must be made in relative terms. Any model is a simplification of reality. However, some models are more detailed and complex than others in their representation of the ways in which a watershed's processes vary in time and space. A "simple" model can be defined as one in which the finer scale details, although known to exist, are intentionally overlooked. For example, an approximate empirical prediction of an NPS event mean concentration (EMC) made based on average land use characteristics, without detailed simulation of the underlying processes, is obviously a simple approach.

In general, we can define "simple" models as approximative scoping tools that are relatively easy to implement and which provide an order of magnitude (or better) assessment of a water quality problem. They are characterized by relatively large spatial scales, and large temporal scales, often steady-state. Analytical solutions to transport equations provide a clear example of simple models that can be used for approximate scoping of water quality problems. Analytical models generally assume rather restrictive conditions, typically one-dimensional, constant parameters and steady state. While real world conditions may not meet any of these criteria, judicious application of analytical solutions is very useful to "home in" on the range of potential responses and relative importance of various phenomena.

Simple models may also utilize computer codes with numerical solutions, but implemented at a relatively coarse spatial and temporal scale. While there is no clear line between simple and complex, our intention is to show the practical uses of models which impose a relatively low burden of data collection and model calibration and treat dynamic phenomena in a manner which is highly averaged in time and space.

2.3 Function of Simple Models in the TMDL Process

Simple models can play a significant rôle in three areas of the TMDL development process: framing the problem, identifying and prioritizing problem areas, and in actually establishing LAs and WLAs. The first two uses are obvious; the third is somewhat more controversial.

2.3.1 Framing the Problem

Sometimes, when there are limited data, the modeler feels at a loss as to which step to take next. Simple models can often help in these situations by "framing" the problem. For instance, a lack of data may leave the modeler wondering about the magnitude of the problem. Use of a simple model (e.g., USGS regression model or FHWA model) may give the modeler a better approximation of the problem's magnitude, and perhaps place an upper nd lower bound to it.

In addition, the simple model application may provide a guide for future activities. Insights may be gained for development of a monitoring program such as key sample locations or parameters that warrant closer attention than others. Future modeling activities may be guided as well. For instance, evaluation of the results of the simple model application may show that a more complex model is needed to accurately address the problem. By exposing information or conceptual model gaps, the modeler is left with a better understanding of what needs to be done to achieve final objectives.

2.3.2 Targeting and Prioritizing

Simple models are often an excellent choice where the objectives are to identify and prioritize waterbodies or pollutant sources of concern. In these cases, the modeler is typically in need of only gross characterization for relative comparison. By comparing approximate loads between sources or estimated conditions among waterbodies, the modeler can target those areas of most interest or importance regarding further investigation or management.

Creation of a nutrient budget is a good example where simple scoping models provide these functions. Based upon a rudimentary breakdown of land use within a given watershed, the modeler can use export coefficients to estimate annual nutrient loading from each

New Watershed Screening Tool

EPA/OST recently released a prototype of a tool for identifying and targeting problem waterbodies titled the "Watershed Screening & Targeting Tool (WSTT)." WSTT summarizes STORET water quality and flow data for a watershed. It operates in a user-friendly PC environment, and allows the user to screen, compare, and target sections within a particular watershed to locate pollution hot spots that need special attention. It also features a watershed model that permits simple watershed assessments that predict daily runoff, streamflow, erosion, sediment load, and nutrient washoff. While currently only available for the States of Alabama and Georgia within Region IV, EPA is considering expanding the application of the WSTT to other regions of the country.

land use category (i.e., unit loads such as kg/hectare/yr). These data, combined with PS data collected through the NPDES self-monitoring program, can be used by the modeler to set up a budget table that provides an estimate of total loading and allows for comparison of source loads. Such comparisons can show which sources may be of greatest concern on a long-term average basis, and the modeler can determine whether a management decision should be based on that limited information. In cases where the implications of the decision are potentially far-reaching or costly, the modeler may choose to further investigate the problem focusing on those sources identified as most important via the screening process. Similarly, this process could be repeated for a number of smaller hydrologic units within a larger basin such that relative comparisons can be made between those watersheds, and efforts or decisions could be prioritized accordingly.

2.3.3 Establishing LAs/WLAs

Finally, simple models can also be used to establish TMDLs/LAs/WLAs under certain conditions. After evaluating appropriate factors (see Chapter IV), the modeler may select the simple model(s) as the best choice(s) to use for development of LAs and WLAs within the TMDL management strategy. This would imply that the reliability of the model predictions has been deemed adequate for management purposes. This can be true even where the MOS is relatively large, particularly where the outcome or response to the model result would not change substantially within the range of error in the prediction. For example, perhaps the decision to target a portion of a watershed for stormwater controls is simply based on the likelihood that the loading from uncontrolled sources would exceed a given threshold. If the model results show that the loads far exceed the threshold, even after considering a large potential model error, then a more accurate model is not needed to justify the application of controls.

It is well established that simple models may often suffice for TMDLs for waters whose impacts are largely or entirely due to point sources. Traditionally, EPA and States have focused on the control of point sources via WLAs written for an individual discharger, rather than from a whole basin perspective. Under these conditions, relatively simple steady-state analytical models are often sufficient to evaluate WLAs. Use of such models is well covered in EPA's series of technical guidance manuals for performing wasteload allocations, and these methods may continue to suffice in areas where point sources are the main concern.

TMDLs which address wet weather NPS, on the other hand, necessarily involve loadings which are episodic or unsteady - dependent on the pattern of rainfall - and which must be described in terms of a statistical probability of recurrence. Further, the TMDL concept typically includes examination of the interaction of PS and NPS on a watershed basis. Performance of an accurate TMDL (i.e., one that includes a minimal excess margin of safety) would typically involve use of a complex simulation model which can represent episodic loads and unsteady receiving water conditions. However, simpler models, including one-dimensional steady state models, may be adequate in some cases, if (1) the uncertainty introduced into the analysis by use of the simplifying assumptions is incorporated into the margin of safety (either through an explicit uncertainty analysis or through use of conservative assumptions), and (2) the spatial resolution of the model is adequate to address the appropriate WQSs and impairments of designated uses. For instance, a simple analytical mixing zone model might be sufficient to evaluate the impact of a rapidly degrading contaminant even in a complex estuarine hydrodynamic situation; however, a one-dimensional model could not assess the spatial distribution of a longer-lived pollutant in such a situation. Additional guidance on the applicability of simple models for TMDLs is provided in Chapter IV.

Also, from a practical standpoint, simple models may have to be used to establish LAs and WLAs where resources are in short supply (e.g., there is not enough time, or funds for sampling, or staff are short on expertise or computational power to perform more complex modeling analyses). In these cases, however, these limiting factors should always be made clear to the decision-maker and the potential implications of error in decision-making based on the simpler approach should be determined. A phased approach can always be used to follow up in such a situation should resources be freed up in the future.

In addition to their potential use for direct TMDL development, simple loading models can assist TMDL development by providing input for more complex receiving water models that are in turn used to establish the WLAs and LAs. For instance, some well recognized and frequently used receiving water models (e.g., QUAL2E, WASP) do not explicitly include routines for simulating runoff from various land uses. In these cases, rather than abandon these models in favor of a more complex watershed model, it might suffice for the modeler to use one or more of the simple runoff estimate models to arrive at discrete loads which could, in turn, be input to the existing receiving water model.

2.4 Assessing Scoping Model Accuracy

Upon application, the model user should always assess the accuracy of the model results. Since no model is perfect, the objective is to assess the magnitude of error and to determine whether that degree of error is important. From a practical standpoint, model inaccuracy is important if it causes the wrong decision to be made. Since simple models often have a large potential prediction error associated with them, close attention should be paid to the assessment of error on any application of significance.

This is typically done through a comparison with the problem setting. For instance, the modeler may ask, "to what degree does the simple scoping model capture the expected variability of the real world system? Are climatic conditions (i.e., wet vs. dry year) important?," etc. If the model only produces an "average" condition prediction and the extremes are of importance to addressing the problem, then a more sophisticated model that captures the variability would be warranted.

Timing also may be important to model accuracy. A simple model that produces annual average loading may suffice where receiving water impacts occur over a long period due to high retention times and slowly reacting pollutants. On the other hand, such a model would probably not accurately address a problem that occurs on a seasonal or shorter term (e.g., daily, hourly) basis. This issue is covered further in Chapter IV.

In addition, the modeler should be aware of the extent that the simplification of a system assumed through the scoping model biases the result. For instance, a reasonable worst-case event mean concentration (EMC) could be assumed for each loading event based on the NURP or USGS studies for urban runoff. If the worst-case loads are assumed to coincide with low flows in the receiving waterbody (an unlikely proposition), such an analysis is likely to overestimate the magnitude of concentrations seen downstream and thus biases the result. The impact of this bias is that the available fraction of the waterbody's LC that can be assigned for WLAs is underestimated. This may not be an issue if there is little need for WLAs. On the other hand, it could be of particular importance in situations where there is a large demand for WLAs and the remaining assimilative capacity is limited or exhausted. In the latter case, a more detailed modeling analysis could provide a more accurate estimate of NPS impacts, thereby reducing the excess MOS that was built into the scoping model via a conservative assumption (i.e., that worst-case concentrations occur during every runoff event).

Where actual spatial or temporal data distributions are available for response variables, the modeler can compare model predictions to these data for their degree of accuracy (e.g., check for patterns of over- or underestimation, magnitude of error, etc.). Evaluations can range from as simple as a bivariate plot comparison to more statistically rigorous techniques such as the chi square or Kolmogorov-Smirnov two-sample tests (see Chapter VII). The outcome of these statistical evaluations of model accuracy can answer whether the model's level of accuracy is acceptable for decision-making or whether a more accurate model is needed.

The following checklist may be helpful to modelers to review scoping model accuracy:

- Does the model's spatial dimension match the physical characteristics of the system of concern?
- Is the natural temporal variation of the response variable adequately covered by the model?
- Is there any evidence of model bias? (e.g., consistent over or under prediction)
- Is the error associated with the model prediction acceptable for decision making purposes?

In some cases, there may be a paucity of data for the modeler to use to make these assessments. However, if that is the case, then the modeler undoubtedly had to make gross assumptions in order to perform the modeling analysis at all. Therefore, it is probable that the lack of knowledge led to conservative assumptions such that there is an implicitly larger MOS. If the MOS is unacceptably large, then a more detailed modeling analysis aimed at reducing the uncertainty and improving on accuracy would need to be performed. See Chapter IV for more guidance on model selection, particularly where simplified methods may be adequate, and Chapter VII for greater detail on evaluating model accuracy.

2.5 Types of Simple Scoping Methods

A considerable amount on information regarding simplified modeling methods is available from EPA. Specific model or method descriptions are provided in the following references:

EPA. 1992. Compendium of Watershed-Scale Models for TMDL Development. U.S. Environmental Protection Agency, Office of Wetlands, Oceans and Watersheds. Report No. EPA841-R-92-002.

Donigian, A.S., and W.C. Huber. 1991. Modeling of Nonpoint Source Water Quality in Urban and Non-Urban Areas. U.S. Environmental Protection Agency. Report No. EPA/600/3-91/039.

EPA. 1985. Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water: Parts I & II. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA. Report No. EPA/600/6-85/002a & b.

Rather than repeat those specific model descriptions here, this guidance will focus on separating these methods into general categories that can be used for different types of applications and then discuss considerations in the use of those types of methods in TMDL decisions.

Simple methods generally fall into two category types, analytical or statistical. Analytical methods typically involve models that attempt a simplified representation of the physical, chemical and/or biological components and processes occurring within a waterbody. These types of models tend to ignore many of the real world complexities, but they are still causally based. The Streeter-Phelps equation coupling oxygen demanding substances to dissolved oxygen deficit in the water column is an example of a simple analytical model. Statistical methods, on the other hand, reflect observed relationships without explanation of causality. The regression equations developed by USGS (Tasker and Driver, 1988) to provide storm-mean pollutant loads and corresponding confidence intervals based upon monitoring data at 70 gaging stations in 20 States provide an example of a simplified statistical approach.

Given their relative ease of use, it's often possible to run both a statistical and analytical scoping model and compare the results. For example, the two different scoping methods might be used to estimate the annual nutrient loading into a lake or estuary. The analytical approach could involve a mass balance of loads from PS and NPS throughout the watershed, with source load estimates obtained from NPDES monitoring data and land-use export coefficients. The statistical approach, on the other hand, might involve the development of a regression model where flow and concentration monitoring data are used to establish a flow/nutrient loading relationship, and the model is used to estimate loading under an average annual flow regime. The statistical approach may be preferred where adequate monitoring data are available, whereas the analytical approach would be preferred in situations lacking field data. However, it is difficult to apportion the resulting loads from the instream regression model back to specific source categories. So if objectives extend beyond comparative watershed loading or total loading, a more detailed method would be needed.

One advantage to the use of statistical models is that they can often be used to calculate a frequency distribution of loading, which is particularly advantageous to assessing the impacts of wet weather events. For example, where these models describe a frequency distribution of event mean concentrations in loading, they can be combined with information of the frequency distribution of rainfall to yield a frequency distribution of loading to the receiving water. The Federal Highway Administration (FHWA) Model is a simplified scoping model (spreadsheet format) that is built upon this principle; summary statistics are produced on the magnitude and frequency of occurrence of instream pollutant concentrations (more details can be found in "The Compendium of Watershed Scale Models," U.S. EPA 1992).

2.6 Examples of Simple Model Application

The following applications are offered to illustrate the points made above:

2.6.1 A "Framing" Example

Acute toxicity effects have been observed in a small watershed of a western metropolitan area following rain events. Since the effects were noted following the storms, no data are available for specific toxic constituents that may have been present in the runoff during the storm. However, given the highly urbanized nature of the watershed, officials have hypothesized that heavy metals may be the cause. In particular, trace amounts of lead (Pb) have been found in water column and sediments during non-storm events and, while the levels observed were not acutely toxic, they may be residual levels and indicative of what might be washing off the land at higher concentrations during storm periods. In order to see whether its possible that Pb could be the cause, and before too much money is spent attempting to pursue Pb further, the manager decides to use a simple model to estimate the an average event mean concentration for Pb.

The USGS regression model (Driver and Tasker, 1988) developed for the western region of the U.S. for mean storm-runoff Pb concentration was used to get a rough idea of the probable magnitude of the stormwater Pb concentration in the watershed in question. The equation takes the form:

$$Pb = 141(TRN)^{-.347}(DA)^{-.145}(LUI+1)^{-.109}(LUC+1)^{0.034}(LUN+2)^{-.086}(MAR)^{.046}BCF \quad (ug/l)$$

(R2 = .19, Std. error = 88 %)

where:
TRN = Total Storm Rainfall in inches
DA = Total contributing drainage area in square miles
LUI = Industrial land use, as % of contributing area
LUC = Commercial land use, as % of contributing area
LUN = Nonurban land use, as % of contributing area
MAR = Mean annual Rainfall in inches
BCF = Bias correction factor (1.304 for Pb in Western Region)

The storm rainfall level (i.e., TRN) of interest is determined to be approximately 0.5 inches since that is the level at which effects have been observed. The drainage area is estimated to be about 1.0 square mile, and has a relatively high percentage of impervious area (IA = 50 %) due to its urbanized nature. The watershed is composed of 40 percent industrial land use (LUI), 30 percent commercial land use (LUC), and 15 percent nonurban land use (LUN). The mean annual rainfall is estimated at 10 inches.

Using the regression equation, an event mean concentration (EMC) of 152 ug/l is estimated. Comparing this to the final acute value (FAV) for Pb of 67 ug/l, the manager sees that the model prediction far exceeds the acute toxicity criterion (even after appropriate dilution calculations in the receiving stream). However, she's worried that the low R-square and relatively high standard error of the model may result in too high of a prediction error. She decides to calculate a 90 percent confidence interval and discovers that the lower bound of 78 ug/l is still above the acute threshold of concern. With this information in hand, the manager decides to initiate a pilot storm monitoring program that will include lead monitoring to further pursue the matter.

2.6.2 A Targeting and Prioritizing Example

A nutrient budget was developed for the Neuse River Basin in North Carolina for comparison of PS and NPS loads throughout the basin. Loads were estimated for each of 14 sub-basins using pollutant loading factors ("export coefficients") for NPS and NPDES discharger monitoring data for PS (NCDEHNR-DEM, 1993). Estimation of land uses at this scale (The Neuse Basin encompasses approximately 6200 square miles) was facilitated by the use of a geographic information system that contained data obtained from a LANDSAT study.

The results (summarized for phosphorus in Table 2-1.) of the simple estimation technique revealed portions of the watershed that merited further attention. For instance, the majority of phosphorus loading in the basin can be attributed to two sub-basins (030402 and 030407). Of these two areas, point sources are the biggest contributor in sub-basin 030402 whereas agricultural NPS was by far the biggest estimated contributor in 030407. The 030407 watershed is comprised entirely by the drainage to Contentnea Creek, a major tributary to the Neuse River. The State used the nutrient budget, along with assessment information indicating use impairment, to target Contentnea Creek and rank it as a high priority watershed for TMDL development.

Table 2-1. Point and Nonpoint Source Phosphorus Loading to the Neuse River Basin by Land Type and Sub-basin.

Subbasin	Agriculture (kg/year)	Forest (kg/year)	Urban (kg/year)	Wetland (kg/year)	Water (kg/year)	Other (kg/year)	Point Source (kg/year)	Totals (kg/year)	Totals Percent
03 04 01	50,974	11,413	26,658	2,674	2,957	1,341	17,603	113,620	10.7%
03 04 02	64,284	9,307	34,510	1,692	842	549	110,397	221,581	20.8%
03 04 03	16,173	1,299	4,999	273	97	93	8,414	31,348	2.9%
03 04 04	39,797	2,598	5,701	583	250	145	3,436	52,508	4.9%
03 04 05	68,685	6,299	1,025	371	486	568	14,884	92,317	8.7%
03 04 06	43,070	3,962	3,369	344	174	198	1,919	53,036	5.0%
03 04 07	145,823	12,699	1,269	699	661	650	24,614	186,414	17.5%
03 04 08	19,465	3,062	1,091	883	481	541	157	25,680	2.4%
03 04 09	30,570	5,267	186	771	45	530	415	37,784	3.5%
03 04 10	18,311	6,971	3,328	2,703	31,875	1,506	24,772	89,467	8.4%
03 04 11	27,569	4,920	2,457	1,727	152	1,926	405	39,156	3.7%
03 04 12	19,459	2,036	400	163	536	108	16,026	38,728	3.6%
03 04 13	6,377	732	1,564	1,517	24,541	357	0	35,088	3.3%
03 04 14	1,554	268	296	1,207	45,570	148	0	49,043	4.6%
Totals - kg/yr	552,111	70,832	86,854	15,608	108,666	8,659	223,043	1,065,772	
Totals - %	51.8%	6.6%	8.1%	1.5%	10.2%	0.8%	20.9%		100.0%

The following subtotals are for the Neuse Basin below Falls Lake Dam to New Bern (all subbasins except 01, 13 and 14)

References, Chapter 2

Donigian, A.S. Jr. and W.C. Huber. 1991. Modeling of Nonpoint Source Water Quality in Urban and Non-Urban Areas. EPA/600/3-91/039. ERL, Athens GA.

NCDEHNR-DEM. 1993. Neuse River Basinwide Water Quality Management Plan. Water Quality Section.

Tasker, G.D. and N.E. Driver. 1988. Nationwide Regression Models for Predicting Urban Runoff Water Quality at Unmonitored Sites. Water Resources Bulletin 24(5): 1091 - 1101.

U.S. EPA. 1992. Compendium of Watershed-Scale Models for TMDL Development. EPA841-R-92-002. Office of Water.

Chapter III. Simulation Models: Model Definitions and Available Models

As discussed in Chapter I, models are used to help establish the links between pollutant loads and WQSs in a receiving waterbody, with the basic objective of obtaining compliance with the CWA. Where any nonpoint sources (and particularly episodic, wet-weather sources) are involved in the impairment of a waterbody, the effects of nonpoint BMPs or point source load reductions on WQS excursions is difficult to predict quantitatively from past observations. The TMDL developer will often need to employ modeling to extrapolate beyond the monitoring record and to assess the potential impacts of proposed control strategies.

The main purpose of this chapter is to discuss the types of models that may be used in the estimation of TMDLs, and to provide a formal basis for their evaluation and comparison in terms of the needs of a TMDL developer at a specific site. It thus focuses on model type and availability, but also provides a catalog and reference to a number of useful models supported (to some extent) by Federal agencies. In Chapter IV we will come at the problem from the other direction, that is, given the characteristics of the TMDL problem and site, what are the characteristics of appropriate models. Chapter IV represents the process of model identification. Comparing the *desired* set of characteristics to the set of *available* models (this chapter) constitutes model selection.

While this chapter provides general information on the characteristics of simulation models, and also lists specific models available for various purposes, the focus is on model characterization. We do not provide comprehensive reviews of individual models. This is for two reasons. First, detailed reviews of most of the appropriate models are available in other EPA guidance (notably including, for loading models, "Compendium of Watershed-scale Models for TMDL Development" (U.S. EPA, 1992) and "Modeling of Nonpoint Source Water Quality in Urban and Non-urban Areas" (Donigian and Huber, 1991), and, for receiving water models, the WLA guidance). Secondly, the status and availability of simulation models is subject to continual change, and models other than those listed here will also likely be appropriate for TMDLs.

3.1 Types of Models Used in TMDL Development

Simulation models can be used to assess aspects of NPS loading, pollutant fate and transport in receiving waterbodies, and ecological and biological impacts within the receiving water. The exact goals of modeling will be determined by the need to evaluate the sources and control of waterbody impairment, expressed in terms of the applicable numeric or narrative criteria. Specific types of impacts most likely to be encountered in a given waterbody type are discussed in Section 5.6.

Thus, estimation of TMDLs can require modeling or analysis of a number of different physical domains or processes. (In the most general sense, we will use modeling to refer to any sort of structured analysis of a domain or process; sorting out whether a complex simulation model or a simple scoping or empirical analysis is to be used is a subsidiary question). For instance, to investigate the effects of agricultural nutrient loading delivered via a stream into a lake we might need to consider the following components:

1. Buildup and availability of nutrients under current agricultural practices,
2. Patterns of overland flow during storm events;
3. Transport of nutrients in washoff during overland flow events;
4. Flow regime in the receiving stream;
5. Transport of nutrients in the receiving stream;
6. Delivery and mixing of the nutrient load into the lake;
7. Water circulation and nutrient cycling within the lake;
8. Algal population/nutrient responses to the nutrient loading.

In most cases, not all of these aspects will (or can) be modeled in equal detail, and some may be bypassed with "appropriate assumptions". Various of the individual aspects may be addressed together within a single simulation model (or simple screening approach). However, in most cases there will not be any single, "as is" modeling package available to meet all the needs of a TMDL, particularly as the different components may need to be addressed at different levels of detail. To continue the nutrient loading example, the TMDL developer (at a fairly simple level of analysis) might need to combine (1) an empirical or loading function analysis of monthly nutrient delivery from agricultural land use, (2) a simple, mass-balance/dilution calculation of transport within the stream, (3) a seasonal model of nutrient cycling within the lake, and (4) an empirical model of lake nutrient response. This example combines four rather simple models. On the other hand, if the concern was with acute toxic effects of pesticides, it might be necessary to estimate or model (1) pesticide availability, (2) washoff during individual storm events, (3) dynamic transport and reactions within the stream, and (4) mixing and decay near the outfall into the lake. All four components could involve rather complex models, if sufficient resources were available to undertake such an analysis.

3.1.1 Taxonomy of Simulation Models for TMDLs by Physical Domain

Because a modeling strategy for estimating TMDLs usually will involve more that a single modeling component, and because TMDLs are applicable to all types of receiving waterbodies, this guidance must address a wide range of simulation models. For the purpose of discussion it is convenient to start with a formal categorization or "taxonomy" of the types of models that may be useful for TMDLs. Models may be classified on many different criteria, including types of pollutants considered, level of complexity, etc. However, the most natural division is on the basis of the physical domain addressed. The basic categorization by physical domain of types of models useful for TMDLs involving nonpoint sources is shown in Box 3-1.

It is important to emphasize in this classification that water flow ("quantity") and pollutant transport ("quality") constitute separate aspects of both nonpoint source and receiving water simulation. Indeed, these aspects are often addressed by different models or approaches. A common situation is that in which a hydrodynamic model is first run to describe receiving water flows. The results are then used to drive a separate water quality model. The analyses may be at rather different levels, e.g., we might require only a steady state description of the hydrology (perhaps a design flow) but then superimpose a time-dependent model of episodic loads of a reactive pollutant. For another example, we might consider analysis of a storm sewer system, in which a detailed hydrodynamic simulation is made of the flows, and pollutant transport is then estimated as an average concentration in flow based on drainage area land use - thus combining a complex model of flow with a very simple "model" of pollutant transport.

Box 3-1. TMDL Model Taxonomy by Physical Domain

1. **Loading Models.** Used to estimate urban and non-urban nonpoint source-derived loading to receiving waters and to predict BMP effectiveness. These models can generally address episodic, rainfall-dependent loads; some also address dry weather flows. There are two major subclasses:

   A. **Urban Loading Models,** which include combined sewer overflow and urban stormwater discharge models. These are generally point sources, in terms of discharge to the receiving waterbody through outfalls, which, however, include consideration of nonpoint washoff into the collecting system.

   B. **Non-urban Loading Models,** which address runoff from agriculture, forestry, suburban development, and other distributed runoff sources.

   For either urban or non-urban loading, models can address some or all of the following aspects:

   i. **Runoff hydrology**: the generation and routing of overland, sewer, and drainage flows.

   ii. **Source availability**: the buildup and availability for washoff of pollutants and sediment.

   iii. **Sediment transport**: The movement and delivery of sediment (and sediment bound pollutants) from sources to the receiving waterbody.

   iv. **Chemical transport**: the movement, reactions and delivery of chemical pollutants (dissolved or sorbed) to the receiving waterbody.

2. **Receiving Water Models.** Used to estimate receiving water flow and transport, and responses to pollutant loading, and to establish TMDLs/WLAs/LAs to meet water quality standards. Three general types of physical domains are addressed, distinguished by the differing importance of advective and dispersive transport processes. An additional distinction may be made between far-field and near-field (local mixing zone) models:

   Far-field Models

   A. **Rivers** and streams, in which unidirectional advection is the dominant transport process.

Box 3.1 (continued)

B. **Lakes** and reservoirs, in which flow is not unidirectional and macrodispersion is often the dominant transport process.

C. **Estuaries**, in which both advection and dispersion (driven by tidal fluxes) are important.

<u>Near-field Models</u>

D. **Mixing Zone Models**, which consider near-source dilution only.

Within each of these physical model types, receiving water models may address some or all of the following:

i. **Hydrology**, or the movement of water.

ii. **Sediment** transport, scour and deposition.

iii. **Chemical** transport and reactions.

3. **Ecological Receptor Models.** Used to predict ecological responses to pollutant loading to a receiving waterbody. Their "physical" domain thus may be the ecosystem. These may be needed to establish TMDLs/WLAs/LAs to meet certain narrative criteria for the protection of aquatic life, habitat criteria and biocriteria. Two classes of Ecological Receptor models are commonly used in TMDLs:

A. **Habitat Modification Models**, which may involve consideration of stream flow dynamics or stream morphology (e.g., channel down-cutting, bank erosion, bar deposition, etc.) in the context of habitat suitability.

B. **Biological Community Models**, under which we may include:

i. Population dynamics (or biomass production) models, which might address phytoplankton, zooplankton, periphyton, benthic macroinvertebrates, fish, or piscivores or other animals dependent on the aquatic food chain.

ii. Biodiversity or ecological indicator models.

3.1.2 Other Bases for Model Classification

Physical domain is perhaps the most obvious means of classifying models, but is not the only basis for classification. That is, if we are modeling impacts in a lake, we don't simply choose among all possible lake models at random. Instead, other issues must be considered (e.g., form of relevant WQSs, cost of model implementation, etc.), and all these other considerations can also be used to classify available models. As noted above, Chapter IV will consider the issue of model identification, while this chapter is concerned with setting the stage for model selection, in which the user must evaluate which among many potential models and modeling approaches provide a good match to the modeling requirements of the site. The other bases for model classification which are relevant to the issue of model identification will also need to be considered. Among these are, in addition to physical domain:

- **Temporal Representation and Scale.** Models may be classified as steady state or dynamic in their representation of a given process. Steady state models do not include a derivative with respect to time in their formulation; that is, they represent the ultimate response to a steady forcing function given infinite time. Dynamic models represent temporal variability. Some simulation packages may combine steady representations of some components with dynamic representation of others - for instance, steady flows with time varying loads superimposed. Where dynamicprocesses are represented there are usually limits on the timesteps or temporal representations that are appropriate for a given model. Numerical models often have a maximum time step determined by conditions of solution stability. On the other hand, lumped parameter models usually have a minimum time step below which the simplifying assumptions used in model development do not permit accurate representation.

- **Spatial Representation and Scale.** The physical representation employed by a simulation model may be in zero, one, two or three dimensions (zero dimension refers to receiving water models which treat the receiving waterbody as a completely mixed tank or continuously stirred reactor). Similar to the temporal representation, there are usually limits on the spatial increments that are appropriate for a given model. Obviously, a model employed in the estimation of a TMDL should be able to produce results at a spatial scale appropriate to the WQSs under consideration. For instance, in lake eutrophication problems, the spatial dimension of interest may be the whole lake (or whole epilimnion), while, on the other hand, consideration of CSO discharges of fecal coliforms may require a fine spatial scale to assess impacts on beaches, drinking water intakes, etc.

- **Constituents and Processes Simulated.** Simulation models have differing abilities to represent types of constituents. Distinctions may include conservative vs. degrading, dissolved vs. sediment sorbed, organic vs. inorganic. Some models only handle one group of constituents (e.g., DO and BOD). Besides chemical contaminants, TMDLs may also need to consider modeling of sediment itself, temperature, and habitat and geomorphological parameters.

- **Practical Considerations and Costs.** Besides their ability to represent physical processes, models differ widely in their ease of use, data requirements, and the level of effort required for successful implementation. These are all valid considerations in the selection of a simulation model for use in TMDLs.

The various means of classifying models will be revisited in Section 3.3, which provides a summary (and classification) of various federally supported models useful for the calculation of TMDLs. This provides a language, or formal set of criteria for describing which models are useful in which situations. In Chapter IV, the identification of appropriate models will be developed using the same set of criteria.

3.2 Introduction to Simulation Models

This section provides a brief introduction to the major concepts and terminology of simulation modeling relevant to TMDLs. This is essentially an annotated glossary of concepts useful for the discussions which follow.

3.2.1 Typical Forms of the Governing Equations for Flow

Simulation models which attempt to provide a causal, physically-based representation of flow and transport (as opposed to empirical or statistical formulations), are generally based on the principle of conservation of mass, and, where appropriate, the principle of conservation of energy or momentum, expressed through partial differential equations. Performance of a simulation model is dependent on the way in which these equations are formulated, and the way in which the equations are solved.

Equations of Flow. All physically based descriptions of flow consider conservation of mass. That is, for a control volume of a simulation, input of water minus output must equal storage within the control volume. Conservation of mass alone is sufficient to describe certain processes, such as reservoir storage under gradually changing inflows. However, flowing water also possesses energy or momentum, and the principal of conservation of energy must also be considered to develop a complete description of flow, particularly where gradients are relatively large or changing over time.

For flow in a channel, the complete equations of flow are typically represented by the St. Venant equations. The St. Venant equations may apply in three dimensions, but can be written in one dimension as

$$\frac{\partial V}{\partial t} + V\frac{\partial V}{\partial x} + \frac{g}{A}\frac{\partial(\bar{y}A)}{\partial x} + \frac{Vq_l}{A} = a_g - a_f + a_w$$

$$\frac{\partial Q}{\partial x} + \frac{\partial A}{\partial t} = q_l$$

where

V is the velocity of flow
t is time
x is distance along the axis of flow
g is the gravitational constant
A is cross sectional area
$\bar{y}$ is the depth of the centroid of flow
q_l represents net flows into the channel for a given control volume
a_g is acceleration due to gravity
a_f is acceleration due to friction
a_w is acceleration due to wind
Q is the flow through the control cross section (= V•A)

The first equation is the conservation of momentum or energy equation, with 7 terms, while the second equation is the continuity or conservation of mass equation, with 3 terms, for a total of 10 terms. In order, these 10 terms represent for a given control section:

1 and 2. Taken together, these terms represent the total differential of velocity with respect to time, i.e., the local acceleration.

3. represents the hydrostatic pressure change across the section.

4. represents the component of momentum associated with inflows to the section.

5. is the acceleration due to gravity.

6. is the acceleration due to bed friction.

7. is the acceleration due to wind.

8. is the change in flow across the section.

9. is the change in storage in the section.

10. is the lateral inflow volume into the section.

A solution of a particular form of these equations yields a method of flow *routing*. In many situations, many of the terms within the momentum equation are relatively insignificant on an order of magnitude basis. For steady state analysis, the partial derivatives with respect to time are assumed to be zero. The representations of channel flow may be classified on the basis of which terms are included or omitted.

Hydrologic Routing methods make use of the continuity equation only; that is, they implicitly assume that changes in energy need not be considered to model the changes in flow. This is usually adequate to describe longer term averages, and situations in which gradients do not change rapidly. Hydrologic routing is contrasted to hydraulic routing, in which some form of the momentum equation is included. Typical methods of hydrologic routing include the Muskingum method and the SCS Convex Method for channels and the Storage Indication method for reservoir routing.

Kinematic Wave Routing considers the momentum equation, and is thus a hydraulic routing method. However, only the most significant terms of the momentum equation are introduced and in steady state: it is assumed that acceleration due to gravity and due to body friction are in balance, that all other terms of the momentum equation are insignificant, and that the only time derivative present is that in the continuity equation. The method is thus applicable to uniform water surface profiles and (locally) steady flows, although discontinuities in the flow regime are considered. Kinematic wave methods can estimate the rate of propagation of flood waves or changes in flow and are more sophisticated than hydrologic routing methods. However, they are more commonly used for overland flow than channel flow. The SCS (1979) Att-Kin TR-20 method represents a combination of storage indication and kinematic wave routing techniques for channel flow.

Dynamic Wave Routing methods address unsteady and nonuniform flow by including acceleration and internal momentum energy terms. This is necessary to obtain accurate representation of flow effects in which the friction slope of the water is not equal to the bed slope, such as occurs in backwater effects. Various representations are possible, and not all the terms of the momentum equation are always included. For channel flow, the effects of hydrostatic pressure change (overpressure) across the control volume (term 3 above) are often ignored, as in the formulation for the hydrodynamic model DYNHYD in the WASP package (Ambrose et al., 1988). Complete dynamic wave formulations are important for the study of extreme flood events, and are incorporated into several National Weather Service models (e.g., DWOPER).

The above terminology is most commonly applied to streams, rivers and estuaries. The same physical principles apply to overland flow, and kinematic wave methods are often used to describe the time history of flow generation on small, and particularly impervious areas. However, a detailed description of the energy component of overland flow from a larger, heterogeneous area is usually neither practical nor relevant. For TMDL development we will usually be interested in total runoff generated from a land area during a precipitation event, or, at most, an approximate time history of runoff. Therefore, overland flow is usually addressed via empirical or mass balance methods. The most commonly used empirical approach is the SCS Curve Number Procedure (SCS, 1968). This predicts runoff from a storm based on antecedent conditions, soil classification, and type of cover. Mass balance approaches generally estimate runoff based on precipitation minus infiltration, and employ a model of the time-dependent decay (and recovery) of infiltration capacity.

Simulation of flows in lakes often involve some rather different problems. This is because lakes can present very different dimensions from flowing streams, and typically show three dimensional patterns of flow, often with thermal stratification. In most lakes, wind is the principal mixing mechanism, and temperature gradients the principal resistance to mixing. Detailed simulation of the internal flows of lakes requires consideration of the thermal energy balance and density gradients due to temperature and dissolved solids.

3.2.2 Equations for Sediment and Pollutant Transport

Simulation of sediment and pollutant transport is usually built atop a description of flow, which may either be derived from a flow model, or described from observation. Transport processes include advection, which is the movement of constituents with the bulk flow of water, and dispersion, which is the result of mixing processes. In practice, dispersion may be thought of as containing the net results of all those flow processes which occur at a scale too small to be captured by the flow model, as well as mixing due to molecular diffusion. Transport is expressed via an advection-dispersion equation.

In one dimension (i.e, averaged over the cross sectional area of a channel) the advection-dispersion equation can be written in the following form (using the notation of Ambrose et al., 1988):

$$\frac{\partial(AC)}{\partial t} = -\frac{\partial(U_x AC)}{\partial x} + \frac{\partial}{\partial x}\left(E_x A \frac{\partial C}{\partial x}\right) + A\,(S_L + S_B) + A\,S_K$$

where

- A is the cross-sectional area (L^2)
- C is concentration of the water quality constituent (M/L^3)
- t is time (T)
- x is distance in the direction of flow (L)
- U_x is the longitudinal advective velocity (L/T)
- E_x is the longitudinal dispersion coefficient (L^2/T)
- S_L is the direct and diffuse loading rate (M/L^3-T)
- S_B is the boundary loading rate (including upstream, downstream, benthic and atmospheric loading; M/L^3-T)
- S_K is a net source or sink rate, representing reactions and transformations (M/L^3-T).

A similar representation may be made in two or three dimensions as required. A separate equation is required for each constituent. Often, these are linked through the last term, representing transformations including internal sources and sinks. For instance, the equations for transport of BOD and DO are linked through a term in which the decay of BOD causes a corresponding removal of dissolved oxygen (or increase in oxygen deficit).

Depending on the process, a variety of kinetic formulations have been proposed in the literature. However, in most surface water quality assessments, chemical transformations are approximated by a first order rate law with rate constant *k* of the form

$$\frac{dC}{dt} \propto k\,C$$

which is equivalent to an exponential decay process.

Another important constituent rate determination is for the boundary loading rates (distinct from advective and dispersive transport). For instance, in simulating sediment these include exchange across the benthic boundary representing losses due to deposition and gains due to scour. For many dissolved constituents we may need to consider volatilization into the atmosphere and sorption on to suspended sediment.

EPA provides references on typical dispersion coefficients and rates and constants for conventional pollutants in Bowie et al. (1985) and for conventional and nonconventional pollutants in Mills et al. (1985). Another important reference for estimation of chemical reaction and transformation rates is Lyman et al. (1990).

3.2.3 State Variables, Boundary Conditions, and Initial Conditions

State Variables are those variables whose behavior a simulation model attempts to reproduce and predict. Choice of state variables is a critical part of model implementation. The more state variables that are included, the more difficult the model will be to implement and calibrate as the model is likely to be over-specified relative to the data; however, if important state vari)les are omitted from the simulation the model may produce unrealistic results, or be unable to answer necessary questions for the TMDL. For instance, if DO is being simulated, should phytoplankton be included as a state variable (i.e., should phytoplankton biomass be simulated), or should it be represented as a constant DO source/sink? The answer depends in part on whether proposed load allocations to control oxygen demand are also likely to affect phytoplankton density, and thus affect DO. The general rule proposed by Thomann (in Freedman et al., 1992) is: "Keep state variables to a minimum; model only those for which data exists; but always include those state variables which will be impacted by a WLA." However, while it is desirable to model only state variables for which data are available, this is not always appropriate, as Thomann himself notes, when unmonitored components are essential to the understanding of key state variables. For instance, both dissolved and particulate phases must generally be included.

Boundary Conditions and Initial Conditions constrain the partial differential equations from a general to a particular solution. Dynamic flow and water quality problems can be thought of as initial boundary value problems, in which a solution is propagated in time from an initial condition but also depends continuously on boundary conditions. In contrast, steady state solutions may be boundary value problems, which do not depend on initial conditions.

Boundary conditions, as the name implies, describe the state variable along the boundaries of the solution domain at all times during the simulation. The boundary conditions may constitute the specification of prescribed values at the boundary (a first-type or Dirichlet boundary condition), the specification of the normal derivative or gradient on the boundary (second-type or Neumann boundary condition), or the mixed specification of a function of the state variable and its derivative on the boundary (third-type or Robin boundary condition). For instance, in simulating hydrology of a coastal plain river in one dimension in which channel cross-sectional area is a state variable, we might specify a time series of inflows at the upstream end of the simulation grid and tidal elevation at the downstream end. This specifies a rate of change of cross-sectional area upstream and a specified cross-sectional area at the downstream boundary. Initial conditions are also required for solution of time propagation models. In some cases (e.g., river flow simulations) the solution after a few days simulation will be rather insensitive to initial conditions. However, in other cases, such as water quality simulation for a slug discharge into a lake, the predicted time history may be highly sensitive to initial conditions of circulation and background concentrations.

3.2.4 Solution Methods for Numerical Models

A simulation model is specified by writing the partial differential equations and specifying an appropriate set of initial and boundary conditions. These must be combined into a solution form. The behavior of a simulation model is determined not just by its specification but also by the method of solution of the governing equations. In terms of choosing an appropriate model, the solution method is of interest primarily because of restrictions it may place on the representation of spatial and temporal variability or, conversely, the maximum time or spatial increment that can be used in simulation.

For very simple model specifications (generally those where the parameters and fluxes are constant in space and time) it may be possible to derive an analytical solution, which allows a direct and exact analysis of the model. A good example is provided by the classic Streeter-Phelps equation, which predicts dissolved oxygen deficit downstream of a constant source in terms of reaeration, BOD deoxygenation and BOD loss rates. Analytical solutions are often useful for screening, due to their ease of implementation. Because the solution is exact, they are also not subject to instability from numerical roundoff errors. However, the usefulness of the simple formulation is limited by its assumptions of constant loading, constant flow, and constant reaction rates.

To solve a more complex model specification, i.e., one where parameters vary in space and/or time, it is necessary to resort to numerical solution methods. Most of the more complex quantity and quality models encountered in TMDL development use an *explicit finite difference* method. In this method, the partial differential equation is solved directly. Each partial differential is represented by a difference approximation. This is equivalent to representing the continuous problem by a step-function or gridded approximation. For instance, we might form a difference approximation for a partial differential with respect to time evaluated at time t as the "slope" of the line between discrete values at time t and time t+1:

$$\frac{\partial x}{\partial t} \approx \frac{x_{t+1} - x_t}{\Delta t}$$

In this case we have used a forward difference operator (over time step Δt); if we evaluated from time t-1 to t+1, we would have a central difference operator (i.e., centered at time t), and if we evaluated from t-1 to t we would have a backward difference operator. Of course, the approximation introduces a truncation error into the solution (see Figure 3-1), and the magnitude of this error can be evaluated through rearrangement of the Taylor series expansion (see, for instance, Anderson et al., 1984).

Once the partial differentials are expressed in an appropriate finite difference form, we have an algebraic expression in terms of current, future (and possibly past) values of the parameter of interest. If the algebraic expression is rearranged to solve for the future value in terms of the known current and past values, this yields an explicit finite difference solution, e.g.

$$C = \frac{\partial x}{\partial t} \qquad C \approx \frac{x_{t+1} - x_t}{\Delta t} \qquad x_{t+1} \approx x_t + C\,\Delta t$$

When an explicit finite difference scheme is used, there are constraints imposed on solution stability. In general, a solution is unstable if the approximation errors tend to magnify as the solution is propagated; it is stable if approximation errors tend to damp out. Methods are well developed for the analysis of finite difference solution stability. For instance, in a first order wave equation in one dimension solved with a central difference representation of the time derivative, the stability criterion is given by the classic Courant-Friedrichs-Lewy condition, which requires that the absolute value of the Courant number, $c\Delta t/\Delta x$, be less than one, where c is the wave speed or celerity. The stability criterion thus typically places limits on the ratio of the time step to spatial step - i.e., to achieve a certain degree of resolution in the spatial grid, the user must employ a time step that is below a certain critical value. The documentation for models employing explicit finite difference solution methods should provide an analysis of the computational stability criteria.

An alternative to explicit finite difference solutions is the *implicit* finite difference solution method. In this approach, the values at all locations on the spatial grid for the next time step are solved simultaneously as a system of algebraic equations. Implicit finite difference equations are generally unconditionally stable, and truncation errors do not magnify without bound, thus allowing use of a larger time step. However, truncation errors are still present and degrade the quality of the solution as the time step grows larger. In the implicit method, this is usually expressed as dissipation or artificial dispersion, in which the numerical method results in the "smearing" of a wave front that is properly sharply defined. Thus an implicit solution does not remove the burden of finding an appropriate simulation timestep. Implicit solution methods have been less commonly used in water quality simulation problems because of the high computational burden of solving equations for all points simultaneously.

An alternative numerical method to the finite difference approach is the finite element method. Finite elements work with an integral formulation of the problem, rather than directly with the partial differential equations. Typically, a weighing

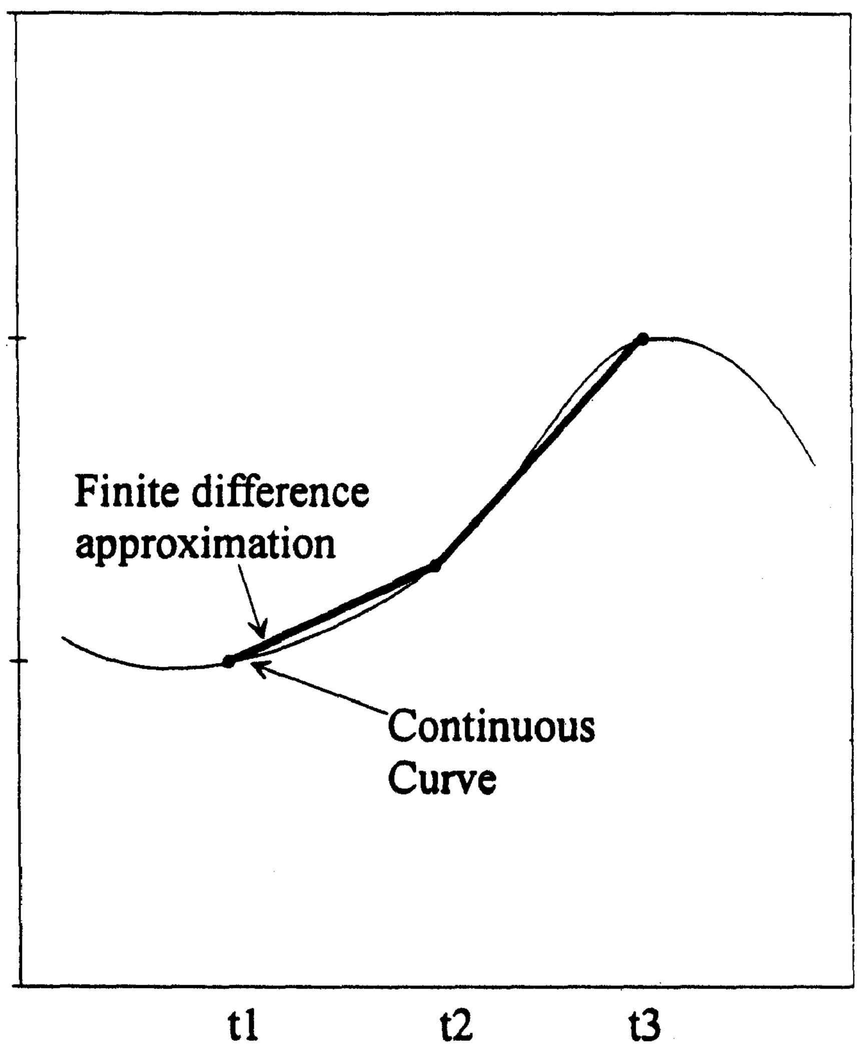

Figure 3-1. Finite Difference Approximation to a Curve

function is derived based on the condition that the weighted average of the residuals of the approximate solution at the grid nodes will be zero. This method has been extensively developed in mechanical engineering, and has received significant application to groundwater flow problems (see Istok, 1989). Application to surface water problems has been limited; however, finite element solutions have been used for the sediment-contaminant transport model SERATRA (Onishi and Wise, 1982).

3.3 Characteristics of Available Simulation Models

As noted in the introduction to the chapter, this guidance is not intended to provide detailed reviews of all available simulation models useful for estimating TMDLs. Instead, the focus is on the development of an appropriate modeling *strategy*, which identifies the necessary characteristics for whatever models are employed (see Chapter IV). Therefore, available simulation models are presented in terms of salient characteristics which can be matched to the requirements determined in the model identification process. The TMDL developer should treat this section as primarily an example of how to logically classify available models, rather than an exhaustive list of the universe of models.

That said, this section does attempt to provide a fairly comprehensive list of a certain subset of models. This subset is essentially those models which are (1) non-proprietary, (2) readily available at no or minimal cost, and (3) adequately documented and subject to some degree of support. These criteria mean that we consider primarily simulation models developed and supported by Federal agencies, with the addition of a few models distributed by universities and State agencies. We have purposefully omitted commercially available models, which in many cases represent state-of-the-art in terms of ease of use and presentation of results, but typically represent user-friendly enhancements of public-domain models. While such models may require a significant initial cost, they should not be ignored by the TMDL developer, where appropriate. (For example, a number of sophisticated commercial products are available for the simulation of stormwater and combined sewer flow). Also omitted from the lists are models which do not seem to be in current use or whose availability appears to be in question. Finally, the same sort of criteria are relevant to the development of site-specific simulation models.

The presentation of available models is organized according to the major subheads of the model taxonomy displayed in Box 3-1. We discuss loading models and receiving water models, and, under each of these heads, present the characteristics related to (1) the flow of water or hydrology, and (2) the transport of sediments and pollutants. We have not provided a survey of ecological receptor and habitat models, because of the diverse character of such models, which does not lend itself to a presentation of general applicability. However, the use of such models is discussed further in Chapter VI. This results in for tables of model characteristics, covering Runoff Quantity, Runoff Quality, Receiving Water Quantity, and Receiving Water Quality. Of course, many simulation models cover more than one category of simulation - for instance, combining simulation of rural NPS flow and loading.

3.3.1 Models of Point and Nonpoint Wet-weather Runoff

Table 3-1 surveys models for Runoff Quantity (including models for rural NPS and urban point sources, such as CSOs and stormwater, which result from the collection and point discharge of episodic, wet-weather flows.) The table provides the following information:

Col. 1 lists the model name.

Col. 2 lists the other general categories of simulation addressed by the software package, and is repeated throughout the following tables. NR stands for wet-weather runoff (both nonpoint and point) quantity or flow simulations (this table); NQ stands for wet-weather runoff quality simulation; RF stands for receiving water flow simulation; and RQ stands for receiving water quality simulations.

Col. 3 provides a key to the main reference for the most recent release of a given model (keyed to the bottom of the table).

Col. 4 provides a key to reviews of a given model in four selected sources, three of them EPA guidance.

Col. 5 indexes the land use applications of a given model, with U standing for urban and R for rural.

Col. 6 indicates the method of simulation of flows, whether overland or collection system. The major categories are (a) empirical runoff coefficient methods, (2) SCS curve number ("Curve Number") methods, (3) water balance methods, based on the principle of conservation of mass, without hydraulic simulation of momentum; (4) kinematic wave methods, which include a simplified representation of the energy equations; and (5) dynamic wave methods, which address (more or less) the full momentum equations.

Col. 7 indicates the temporal resolution, or time step which can be achieved by a given model. There are two issues here: first, whether a model is applicable to continuous simulation of flows or just response to individual events, and second the minimum time step representation which can be reasonable achieved.

Col. 8 indicates the agency supporting a model. Most EPA models have been supported by the Center for Exposure Assessment Modeling (CEAM) at the Environmental Research Laboratory, Athens. Other agencies cited include the Federal Highway Administration (FHWA); U.S. Geological Survey (USGS); Army Corps of Engineers Hydraulic Engineering Center (HEC) in Davis, CA; Army Corps of Engineers Waterways Experiment Station in Vicksburg, MS (WES); the Agricultural Research Service of the USDA (USDA/ARS); and the Soil Conservation Service (SCS); as well as several state agencies.

Table 3-1. Runoff Quantity Models

Model	Other Uses	Main Ref.	Reviews	Land Use	Simulation Type	Time Step	Agency
EPA Statistical	NQ	1,2	a,b,c	U,R	Runoff Coeff.	Annual, Event Ave.	EPA
USGS Regression	NQ	3	a,b	U,R	Regression	Annual, Event Ave.	USGS
FHWA	NQ, RQ	2	a,b	Highway	Runoff Coeff.	Annual, Event Ave.	FHWA
GWLF	NQ	4	a	U,R	Curve Number	Continuous Monthly	Cornell Univ.
AGNPS	NQ,RF, RQ	5	a,b	R (Ag)	Curve Number	Continuous Hourly	USDA/ ARS
STORM-RWQM	NQ,RF, RQ	6	a,b,c,d	U	Runoff Coeff./ Curve #	Continuous Hourly	HEC
ANSWERS	NQ	7	a,b	R	Water Balance	Event	Univ.
DR3M	NQ	8	a,b,c	U	Kinematic Wave	Continuous Subhourly	USGS
SWRRBWQ	NQ,RF, RQ	9	a,b	R	Curve #/ Water Balance	Continuous Daily	USDA/ ARS
SWMM	NQ	10,11	a,b,c,d	U	Kinematic & Dynamic Wave	Continuous Subhourly	EPA/ CEAM

Table 3-1 (continued)

HSPF	NQ,RF, RQ	12	a,b,c,d	U,R	Water Balance, Hydrologic Routing	Continuous Subhourly	EPA/ CEAM
Auto Q-ILLUDAS	NQ	13	a	U	Water Balance	Continuous Event	Illinois State Water Survey
CREAMS	NQ	14	a,d	R (field scale)	Water Balance	Continuous Daily	USDA/ARS
TR-20	RF	15		R	Curve Number	Event, Sub-event	SCS
HEC-1	RF	16		R	Multiple (UH to Kinematic)	Event, Sub-Event	HEC
TR-55		17		U	Curve Number	Event	SCS

Key to References
1. Hydroscience, 1979
2. Driscoll et al., 1990
3. Driver and Tasker, 1988
4. Haith et al., 1992
5. Young et al. 1986
6. HEC, 1977a
7. Beasley & Huggins, 1981
8. Alley and Smith, 1982a
9. Arnold et al., 1991
10. Huber & Dickinson, 1988
11. Roesner et al., 1988
12. Johanson et al., 1984
13. Terstriep et al., 1990
14. Knisel, 1980
15. SCS, 1973
16. HEC, 1985
17. SCS, 1986

Key to Reviews
a. U.S. EPA, 1992
b. Donigian & Huber, 1991
c. WPCF, 1989
d. McKeon & Segna, 1987

3.3.2 Models of Point and Nonpoint Wet-weather Loading

Table 3-2 discusses Runoff Quality Models, i.e. the transport and loading of pollutants from episodic, wet-weather sources. The first five columns are identical to those in Table 3-1. The additional columns summarize the following characteristics:

Col. 6 summarizes the types of pollutants which can be addressed by a given simulation model. Some models are rather general in their potential application. Others are characterized as applicable to nutrients (N), oxygen and oxygen demand (O), metals (M), conservative organic pollutants (C), and nonconservative, reactive organic pollutants (NC).

Col. 7 addresses the manner in which pollutant loads are simulated. Most models use one of two methods: Loading Functions, in which event mean concentrations are empirically related to land use; and buildup-washoff formulations, in which the time-dependent availability of pollutants is attempted to be simulated. Several models rely on sediment potency factors; i.e. pollutant load is based on a fixed fraction of sediment scoured. This method is generally available in any model which simulates sediment erosion.

Col. 8 summarizes sediment transport in overland flow. Many models use the Universal Soil Loss Equation (USLE) or the Modified USLE (MUSLE). Other models attempt to simulate sediment detachment and transport by physical processes.

Col. 9 indicates the time step achievable by the runoff quality routine, analogous to the time step presented in Table 3-1.

Col. 10 indicates whether the model is capable of addressing pollutant routing (timing within the runoff), transformation and degradation during transport.

Col. 11 states the supporting Agency, as in Table 3-1.

Table 3-2. Runoff Quality Models

Model	Other Uses	Main Ref.	Review	Land Use	Constituents	Load Generation	Sediment Erosion	Time Step	Routing - Transformation	Agency
EPA Statistical	NR	1,2	a,b,c	U,R	General	Loading Function	USLE/ MUSLE	Annual, Event Ave.	no	EPA
USGS Regression	NR	3	a,b	U	N,O,M,C	Loading - Regression	N/A	Annual, Event Ave.	no	USGS
FHWA	NR, RQ	2	a,b	High-way	N,C,M	Loading - Median Conc.	N/A	Annual, Event Ave.	no	FHWA
Watershed		8	a	U,R	General	Loading Function	USLE	Annual	no	USGS
GWLF	NR	4	a	U,R	N,S	Loading Function	MUSLE	Continuous Monthly	no	Univ.
AGNPS	NR	5	a,b	R(Ag)	N,S	Potency Factors	MUSLE	Continuous Hourly	no	USDA/ ARS
STORM-RWQM	NR,RF, RQ	6	a,b,c,d	U	N,O,M,S	Buildup-Washoff	USLE	Continuous Hourly	no	HEC
ANSWERS	NR	7,11	a,b	R(Ag)	N,S	Potency Factors	Detach-ment	Event	yes	Univ.
DR3M-QUAL	NR	13	a,b,c	U	N,S,C,M	Buildup-Washoff	MUSLE	Continuous Subhourly	no	USGS
SWRRBWQ	NR,RF, RQ	9	a,b	R	S,N,C,NC	Buildup-Washoff	MUSLE	Continuous Daily	yes	USDA/ ARS

Table 3-2 (continued)										
SWMM	NR	10	a,b,c,d	U	General	Buildup-Washoff	MUSLE	Continuous Subhourly	yes	EPA/CEAM
HSPF	NR,RF, RQ	12	a,b,c,d	U,R	General	Loading-Washoff	Detach-ment	Continuous Subhourly	yes	EPA/CEAM
CREAMS	NR	15	a,d	R (field scale)	S,N,C,NC	Potency Factors		Continuous Daily	yes	USDA/ARS
Auto Q-ILLUDAS	NR	14	a	U	S,N,C, NC,O	Buildup-Washoff		Continuous Event	no	Illinois SWS
Watershed Management Model	RQ	16	a	U,R	N,M	Loading Function	NA	Annual	no	Florida DER

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8. Walker et al., 1989
9. Arnold et al., 1991
10. Huber & Dickinson, 1988
11. Dillaha et al., 1988
12. Johanson et al., 1984
13. Alley & Smith, 1982b
14. Terstriep et al., 1990
15. Knisel, 1980
16. CDM, 1992

Key to Reviews

a. U.S. EPA, 1992
b. Donigian & Huber, 1991
c. WPCF, 1989
d. McKeon & Segna, 1987

3.3.3 Models of Receiving Water Flow

Table 3-3 summarizes a number of models for the simulation of receiving water flows. As above, the list focusses on models supported by Federal agencies. This constraint necessarily omits some of the more complex and interesting model applications available in the literature; a good summary of such models available for estuarine applications is provided in Ambrose et al. (1990). The first four columns of Table 3-3 are similar to those in the pre ous tables.

Col. 5 identifies the major waterbody types for which the flow simulation is appropriate. These include rivers (i.e., advection dominated systems), lakes/ reservoirs, and estuaries. Models designated "Lake" are usually appropriate for both lakes and reservoirs; those intended specifically for reservoirs are indicated as "Reservoir."

Col. 6 shows the dimensionality of the flow simulation. Note that some models (e.g., DYNHYD) are essentially one-dimensional in their equations, but address multi-dimensional simulation by using a link-node network.

Col. 7 summarizes the basic method of computing or routing flows. Water balance and hydrologic routing procedures are based on continuity only, without solution of the energy equation. Kinematic and dynamic wave methods address time-dependent momentum at various levels of complexity (see Section 3.2.1). Steady state solutions to the energy equations are provided by Bernoulli equation type solutions.

Col. 8 identifies whether flows are simulated as steady-state or dynamic, and what approximate temporal resolution is achieved. The latter is a rather subjective determination, as a model's ability to accurately *represent* flows is often at a rather coarser time step than that needed for internal simulation purposes.

Table 3-3. Receiving Water Quantity (Flow) Models

Model	Other Uses	Main Ref.	Rev.	Waterbody Type	Dimensionality	Routing Type	Time Step	Agency
QUAL2E	RQ	1		River Estuary	1-D	Water Balance	Steady	EPA/ CEAM
DYNHYD (WASP4)	RQ	2	a,d	River Lake Estuary	1-D+ (to 3D link node)	Dynamic Wave	Transient Subhourly	EPA/ CEAM
AGNPS	NQ,NR, RQ	3	a,b	River	1-D	Unit Hydrograph	Transient Hourly	USDA/ARS
SWRRBWQ	NF,NQ, RQ	4	a,b	Lake River	1-D, 0-D	Water Balance	Transient Daily	USDA/ARS
HSPF	NF,NQ, RQ	5	a,b,c,d	River Lake	1-D	Hydrologic Routing	Transient Hourly	EPA/ CEAM
HEC-2		6	d	River	1-D	Bernoulli Equation	Steady	HEC
HEC-6	RQ	7	d	River	1-D	Bernoulli Equation	Series of steady evts.	HEC
DWOPER		8	d	River	1-D	Dynamic Wave	Transient Subhourly	NWS
TR-20	NR	9		River	1-D	Storage + Kinematic	Transient Subhourly	SCS
HEC-3		11		Reservoir	1-D	Water Balance	Monthly	HEC

Table 3-3 (continued)								
HEC-5		12		Reservoir River	1-D network	Hydrologic Routing	Transient Hourly	HEC
HEC-1	NF	14		River Reservoir	1-D	Kinematic & others	Event	HEC
CE-QUAL-R1	RQ	15		Reservoir	1-D	thermal & density	Transient	WES
CE-QUAL-W2	RQ	16		Lake Estuary	2-D	thermal & density	Transient	WES
RWQM (STORM)	RQ,NF, NQ	17		River	1-D	Kinematic Wave	(to support longer term quality)	HEC
WQRSS	RQ	18		River Reservoir	1-D	Multiple methods	Transient Subhourly	HEC
TABS-2	RQ	19		River Estuary	2-D		Transient Subhourly	WES
WIFM-SAL	RQ	20		River Estuary	2-D		Transient Subhourly	WES

Key to References

1. Brown & Barnwell, 1987
2. Ambrose et al., 1988
3. Young et al., 1987
4. Arnold et al., 1991
5. Johanson et al., 1984
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11. HEC, 1976
12. HEC, 1979b
14. HEC, 1985
15. WES, 1988
16. draft available
17. HEC, 1979a
18. HEC, 1978
19. Thomas and McAnally, 1985
20. Schmalz, 1985

Key to Reviews

a. U.S. EPA, 1992
b. Donigian & Huber, 1991
c. WPCF, 1989
d. McKeon & Segna, 1987

3.3.4 Models of Receiving Water Quality

The final table, Table 3-4, summarizes model capabilities for simulation of receiving water quality. In addition to categories represented in Table 3-3, the following data are included:

Col. 6 indicates the types of constituents addressed, using the same abbreviations as in Table 3-2, with some additions; that is, they are characterized as applicable to nutrients (N), oxygen and oxygen demand (O), metals (M), conservative organic pollutants (C), nonconservative, reactive organic pollutants (NC), sediment (S), and temperature (T).

Col. 8 summarizes sediment transport simulation abilities. The options are not addressed ("NA"), transport of sediments ("transp"), and simulation of erosion ("eros") and deposition ("dep").

Col. 10 indicates ability of the pollutant transport routines to simulate degradation and transformations. Many models incorporate simple decay routines, but cannot handle transformation products. Those which address both are denoted "full".

Table 3-4. Receiving Water Quality Models

Model	Other Uses	Main Ref.	Rev.	Water-body type	Constituents	Dimensionality	Sediment Sim.	Time Step	Transf. Degrd.	Special Characteristics	Agncy
QUAL2E	RF	1		River Estuary	O,N,A,C, NC,T	1-D	NA	Steady	Degr	Uncert. analysis	EPA/ CEAM
WASP4	RF	2	d	River Estuary Lake	General	1-D+ (to 3-D link node)	transp	Cont. Subhrly	full	EUTRO & TOXI versions	EPA/ CEAM
AGNPS	NQ,NF, RQ	3	a,b	River	N,S,C	1-D	transp eros/ dep	Cont. Hourly	no		USDA/ ARS
SWRRBWQ	NQ,NF, RF	4	a,b	River (sed) Lake (N)	S,N,C,NC	2-D Compart ment	transp eros. dep.	Cont. Daily	Degr	Uncer-tainty Analysis	USDA/ ARS
HSPF	NQ,NF, RF	5	a,b, c,d	River Lake	General	1-D	transp	Cont Subhrly	full	Freq-Dur module	EPA/ CEAM
EXAMS-II		6	d	General	C,NC	1-D to 3-D	NA	Steady-Monthly	full	Flows spec external	EPA/ CEAM
MEXAMS		7	d	General	M	1-D to 3-D	NA	Steady-Monthly	full	Flows spec external	EPA/ CEAM
HEC-6	RF	8	d	River Reservoir	S	1-D	transp eros dep	Contin Daily	NA	stream bed profiles	HEC

Table 3-4 (continued)

STREAMDO IV		10		River	O	1-D	NA	Steady	NA	ammonia tox.	EPA R VIII
AMMTOX		11		River	Ammonia	1-D	NA	Monthly Diel cycles	Degr	Freq Analysis	EPA R VIII
SMPTOX3		12		River	C,NC	1-D	NA	Steady	Degr	PS only Sens. Anal.	EPA/ CEAM
FHWA	NF,NQ	13	a,b	River Lake	N,C,M	Statistical	NA	Stats on Events	no	Highway Runoff	FHWA
Watershed Management Model	NQ	14	a	Lake	N,M	0-D	NA	Annual	simple Degr	Spread-sheet	Fla. DER
DYNTOX		17		River	C, NC (PS + backgrd)	1-D	NA	Daily	Degr	Freq- Dur analysis	EPA/ CEAM
SEDDEP		18		Estuary	S	2-D	trans depp		NA	outfall depos.	EPA/ CEAM
CE-QUAL-R1	RF	19		Reservoir	T,C,NC, N,O	1-D	transp dep	Hourly	full	Monte Carlo	WES
CE-QUAL-W2	RF	20		Lake Estuary	T,C,NC, N,O	2-D	transp	Hourly	full	full energy	WES
RWQM (STORM)	NF,NQ, RF	21		River	O,N,T	1-D	NA	long-term	Degr	Linked to STORM	HEC

Table 3-4 (continued)

WQRSS	RF	22		River Reservoir	General	1-D	transp	Contin. Subhrly	full	temp., ecologic sim.	HEC
CORMIX		23		Estuary Lake	Dilution	2-D, 3-D	NA	Steady	no	near field	EPA/ CEAM
SEM		24		Estuary	O,C,NC	1-D	NA	Steady	degr	flows input	EPA
AUTOQUAL		25, 26		River Estuary	O,N	1-D	NA	Steady	no	flows input	EPA
FETRA		27	d	Estuary Lake	S,C,NC	2-D	transp eros dep	Contin.	degr	finite element	NUREG
TABS-2	RF	28		River Estuary	S,C,NC	2-D	transp dep	Contin.	degr		WES
WIFM-SAL	RF	29		River Estuary	bacteria	2-D	NA	Contin. Subhrly	degr		WES
PLUMES		30		Estuary Lake	Dilution	2-D, 3-D	NA	Steady	no	near field	EPA/ CEAM

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Chapter IV. TMDL Modeling Strategy and Model Identification

Purpose: This chapter is intended to guide the user in formulating an appropriate modeling **strategy** for establishing a TMDL. Key aspects of the modeling strategy are the identification and selection of an appropriate model or models. Following Ambrose et al. (1990), the goals of model identification and selection are to **identify** the simplest conceptual model that includes the physical and chemical phenomena that should be addressed in establishing the TMDL, and to **select** the most useful analytical formula, statistical model, or computer simulation model for calculating the TMDL.

We first discuss the principles of establishing a modeling strategy. This is followed by a discussion of general criteria for model identification for TMDLs, and specific considerations for TMDLs involving wet-weather PS and NPS loads. In Section 4.4 we have endeavored to apply these principles to develop practical guidance for model selection. This is presented in the form of a set of structured decision criteria, or decision trees.

The purpose of the decision trees is to assist in identifying an optimal, site-specific modeling approach, taking into consideration the characteristics of the loads, characteristics of the receiving waterbody, data availability, regulatory context, and available resources for analysis. This is done by leading the user step-by-step through a hierarchy of issues which should be considered in the model identification process. The modeling decision trees are primarily concerned with model identification, rather than selection; they are not an 'expert system' that will pick *which* model to use. Rather, the presentation consists of a logically structured summary of suggestions and guidance to the *type* of models appropriate to a specific TMDL situation, and the corresponding monitoring requirements. The output of the process is a statement of necessary model criteria for a given TMDL. The user can then match these criteria to the characteristics of available models to complete the model selection process. Where no match is found this may indicate the need to develop a site-specific model.

4.1 Establishing a Modeling Strategy

In the TMDL process, modeling may be used to accomplish several related functions. First, modeling, combined with monitoring information, can be used to establish the total assimilative capacity of the waterbody. Modeling, combined with monitoring, may also be used to estimate the frequency distribution of loads from wet-weather episodic loads and any resulting excursions of WQSs. Finally, modeling often provides the basis for the evaluation of BMPs and optimal assignment of LAs and WLAs. The quality of the TMDL may thus be critically dependent on implementation of an appropriate modeling strategy.

Of course, not all TMDLs require modeling; in some cases decisions can be made directly from observational data. Even when models are needed, it is not always necessary to employ a detailed simulation model if simple scoping models are adequate (Figure 4-1). However, most TMDLs will require some form of modeling. The type of modeling employed will be closely linked with the availability of data. While models are often used to extrapolate beyond m sured data, the type of model that can be employed is also constrained by the availability of data for calibration and verification. EPA's Science Advisory Board (U.S. EPA 1989) has concluded that, ideally, modeling should be linked with monitoring data in regulatory assessments. Thus the modeling and monitoring strategies for TMDLs should be developed in tandem. Further, the modeling strategy will likely need to be periodically refined as additional data are collected and initial modeling efforts applied.

A complete modeling strategy has a number of components, which range from setting objectives to the identification, selection and implementation of appropriate models. Many of these aspects can be thought of as analogous to the QA/QC measures which are applied to measurements (see 57 Federal Register 22907, May 29, 1992). The modeling strategy has four major components:

4.1.1. Modeling Study Objectives

The first step in development of the modeling strategy is to clearly define the goal of the modeling exercise, i.e. how a model can help address the questions and problems presented by the TMDL. Note that the modeling study objectives are likely to change as additional data are developed, particularly when a Phased TMDL is pursued.

Modeling study objectives should include a clear statement of what information the model will help estimate, and how this estimate will be used. For TMDL or WLA analysis, the objectives will address WQSs or beneficial uses. Therefore, the first step is to review the applicable WQSs and uses to be protected. Local, state, and federal regulations may contribute to a set of objectives and constraints. Each may specify particular pollutants or classes of pollutants, and imply time and space scales that must be resolved by the model. For example, proscription of "toxic pollutants in toxic amounts" implies simulation of whole effluent toxicity dilution. Ammonia or metals standards imply simulation of those specific chemicals (Ambrose et al., 1990). The modeling study objectives also must be consistent with known project constraints (i.e., schedule, budget, and other resources), as well as the objectives of the receiving water analysis.

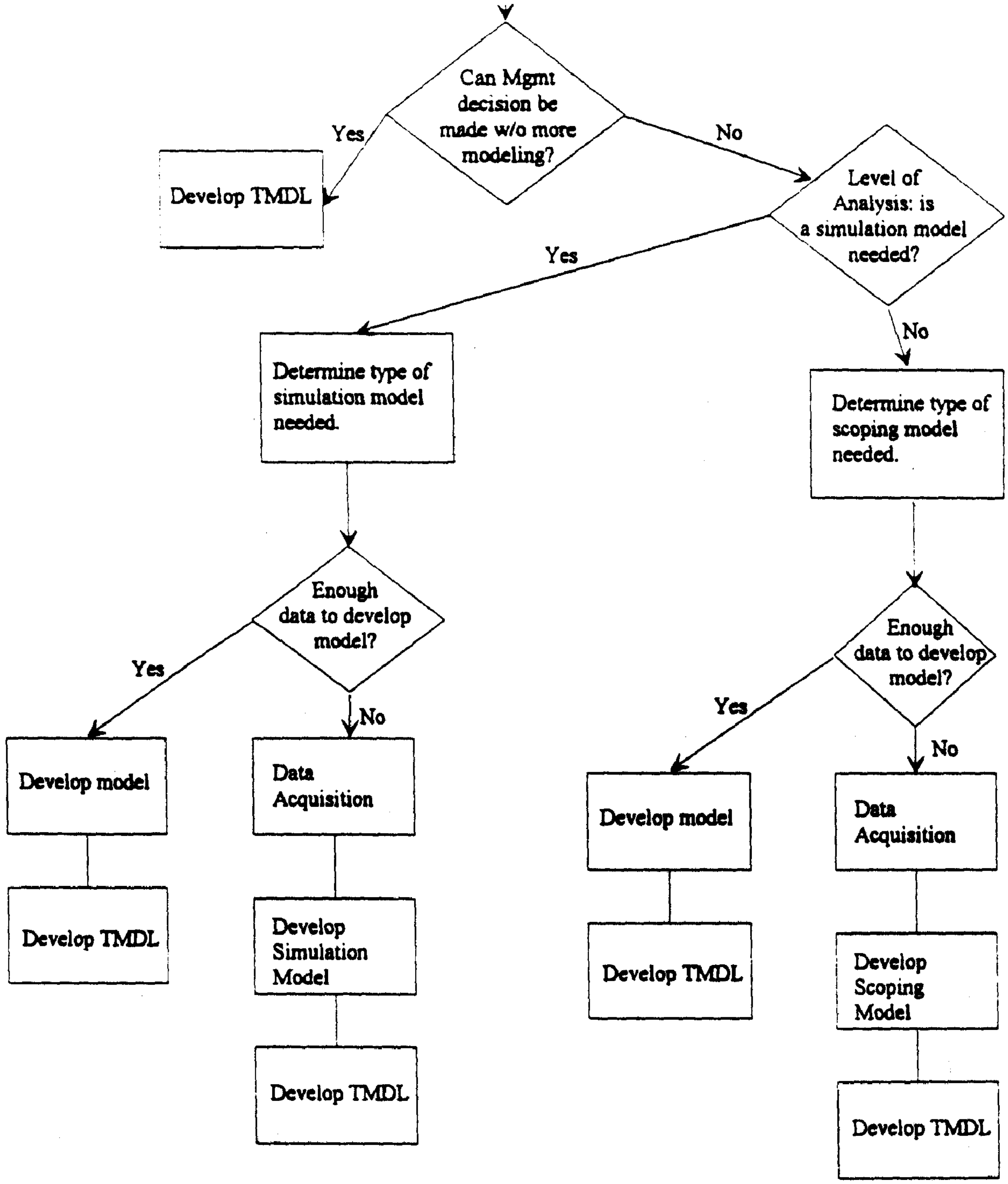


Figure 4-1. Basic Conceptual Decision Tree For Model Choice

The modeling study objectives provide the basis for model identification, and are developed further in Section 4.4.

4.1.2. Model Identification and Selection

The second component of a modeling strategy is the identification and selection of appropriate models (whether simple or complex). This is the focus of Section 4.4.

4.1.3. Model Data Collection Plan

Modeling and monitoring efforts will typically develop in tandem: In most cases it will not be possible to specify all the necessary components of the modeling strategy at the beginning of the process, when data may be limited. Instead, it will be necessary to refine the goals and requirements of the modeling as additional data are collected. Similarly, the monitoring plan may need to be adjusted based on preliminary modeling runs. For instance, in a DO/BOD problem involving steady point and episodic sources, the initial stage of analysis, with limited data in hand, might consist of steady state stream modeling covering a number of different scenarios for the impacts of the steady plus episodic loads under a variety of streamflow conditions. This analysis would reveal the general location of critical points - although the identification will likely be much less exact than in a problem involving only steady point loads. This initial analysis would yield only a relative, and not quantitative estimate of the occurrence and probable location of WQS excursions. It would also yield an indication of which episodic loads are likely to be of most significance to the overall impairment of the waterbody. This initial, simple modeling analysis could provide the basis for a targeted monitoring strategy, designed to further quantify those loads which *might* be important contributors to waterbody impairment. The additional data collected by this targeted monitoring program could then form the basis for a dynamic (unsteady) modeling application.

These observations indicate that the development of a final modeling strategy is an iterative process, involving close interplay between the modeling and monitoring plans. Development of monitoring plans is discussed further in Chapter V.

4.1.4. Model Quality Assurance

Just as quality assurance (QA) principles are applied to the collection of analytical data, analogous QA measures can be applied to simulation modeling. Quality assurance consists of a plan to assure that a product meets defined standards of quality with a stated level of confidence. Once a modeling approach is identified, a QA approach should be developed to provide guidelines for the project and establish a baseline against which the effectiveness of model applications can be evaluated.

As with analytical QA, modeling quality assurance begins with the establishment of data quality objectives (DQOs). The DQOs should reflect the modeling study objectives, and provide a pre-established benchmark against which to evaluate the success and validity of the modeling effort. For instance, it is advisable to establish *a priori* goals for the desired accuracy of model predictions, goodness of fit for model calibration and verification, as well as sampling uncertainty in the determination of measurable physical parameters. Such goals should be defined in terms of the type of model used, and the objectives of the modeling study. For instance, the DQOs for a preliminary study concerned with average nutrient levels in a waterbody would be quite different from those for a model required to predict point-in-time coliform concentrations at a public beach. In the first case, a simple model which gave an accurate prediction of seasonal averages is likely to be quite sufficient for estimating a TMDL; in the latter case it may be necessary to conduct more detailed modeling sufficient to determine the time history of waterbody impacts from individual loading events.

Of course, the relative success and accuracy of most environmental fate and transport model applications is more difficult to predict and control than the analysis of chemical samples. When model DQOs are not met, this should certainly prompt a reexamination of model objectives, model application, and data collection. However, failure to attain desired levels of accuracy does not mean that the model results are unusable (unlike laboratory QA). Instead, it may imply that (1) the modeling strategy should be revised, <u>or</u> (2) the model results should be used to estimate the TMDL, but with a correspondingly large MOS to account for the potential error in model predictions, limit excursions of WQSs, and protect beneficial uses. Evaluation of model accuracy, uncertainty and bias is covered in Chapter 7.

To further the analogy with laboratory QA, the user may consider formalizing a quality control method as well. Quality control (QC) is a part of QA, and ensures the product of the analysis is satisfactory and economical through the specification of and adherence to certain procedures and protocols. Quality control procedures for modeling may include testing the installed version of computer code for correct performance, validating accuracy of input data sets, good record keeping and documentation of model calibration, verification and application, and so on.

4.2 General Criteria for a Modeling Strategy for TMDLs

This section addresses the general question of determining a modeling strategy for calculating TMDLs. Formulation of such a strategy depends on many factors. It is more than just choosing the "best" model of a physical system, as the strategy must take into account such real-world constraints as available resources for a project. This guidance takes a somewhat different approach from some of EPA's Waste Load Allocation modeling guidance in that it recognizes the inevitability of such constraints and attempts to define an appropriate level of analysis to perform a TMDL with finite resources.

Models used to calculate TMDLs must first adequately represent the significant features of the physical system describe . Passing this test, a key aspect of model selection is the cost and effort required for implementation, which must be balanced against the benefits achieved by use of the model. There is an obvious tension between an optimal technical representation of the physical system and the cost of implementation, as increasing model resolution (the fineness of the spatial and temporal scales of model prediction) and model accuracy (the extent to which predictions differ from observations) usually involves increased expense and effort. Further, complete accuracy can never be achieved (especially for simulations involving wet-weather PS and NPS loading), and there is typically a point at which increased modeling effort provides rapidly diminishing returns in terms of increased accuracy. Also, as noted by Thomann in Freedman *et al.* (1992), increasing model complexity beyond a certain point actually results in a decline in model credibility (Figure 4-2). This is because increasing complexity generally requires specification of more and more parameters and state variables, all of which require a detailed database for complete assessment. Indeed, attempts to develop models with extremely high levels of accuracy and resolution can have the unintended effect of delaying the analysis and implementation of controls that may be necessary to prevent impairment of the waterbody. On the other hand, simple modeling approaches which offer only approximate representations of the system can often be used to implement TMDLs. Ideally, the effects of the inaccuracies and approximations introduced by use of the simpler approach can be explicitly incorporated into the MOS. At a minimum, the simple analysis should yield an indication of the relative risk posed by different sources. This will allow the TMDL developer to target the most promising sources for control during a phased TMDL development process.

An *appropriate* modeling strategy must be focused on performing the TMDL in a practical manner. It therefore involves more than selecting modeling techniques that provide an optimal description of the physical system and pollutants or impact mechanisms of interest. In general, we advise designing a modeling effort to provide answers that are as detailed and accurate *as needed*, at the lowest corresponding expense and effort - *e.g.*, don't run a complex simulation with HSPF if a spreadsheet dissolved-oxygen simulation will be adequate to estimate the WLAs and LAs. In EPA's estuarine WLA guidance (Freedman *et al.*, 1992), this point of view is succinctly summarized by Thomann:

> The best models are often the simplest...[advocate] doing estuarine water quality modeling in as simple a fashion as possible and only after all simplicity has been

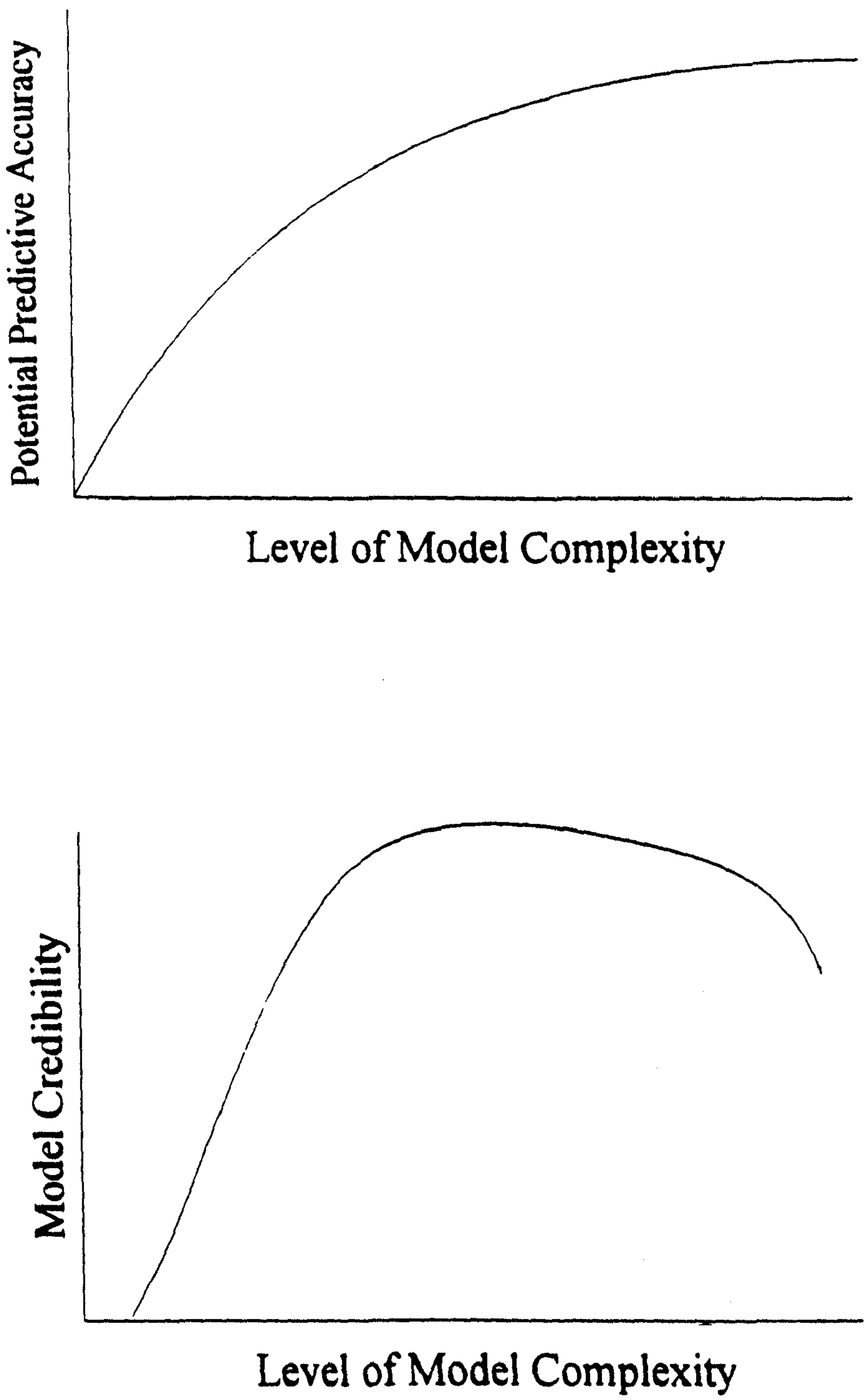


Figure 4-2. Relation of Model Complexity, Accuracy, and Credibility

exhausted should increasing complexity be introduced and then only after careful consideration is given to the improvements in the model that might be realized. The reasons for this bias are: (a) most analysts have only limited experience, time and resources available, and (b) unnecessarily complex models sometimes tend to obscure uncertainty behind a facade of "reality".

Defining the optimum balance between ease-of-implementation and accuracy is always difficult, yet is a key issue in model identification.

4.2.1 Types of Criteria for Model Identification

While the physical characteristics of the system certainly must guide selection of models, other factors must also be taken into account. The criteria for model identification for TMDLs fit into three general categories (expanding on the classification of Mao, 1992): Technical Criteria, Regulatory Criteria, and User (Functional & Operational) Criteria.

Technical Criteria comprise the match of the model to the physical characteristics of the system. They reflect whether the model is appropriate to the physical system being described. For instance, a one-dimensional transport model designed for rivers cannot provide a good description for the distribution of contaminants in a stratified estuary, and slug releases of reactive chemicals cannot be accurately described by steady state models. These considerations first involve the comparatively simple question of whether a model's governing equations and boundary conditions are a good match to the characteristics of the waterbody, washoff process, and pollutants requiring TMDLs. However, evaluation relevant to the TMDL process involves additional, and potentially competing considerations: For instance, as shown in the simple example above, it is possible to perform a TMDL that is protective of WQSs using a model which does not provide a truly accurate description of the system (*e.g.*, no dispersion). Technical criteria must then 1) recognize the differences between the model and reality, and 2) insure that all such discrepancies result in errors on the side of safety.

Regulatory Criteria reflect the fact that the modeling effort is driven by compliance with the CWA, and should be framed in the appropriate regulatory context. That is, the TMDL is based on attaining water quality criteria and the modeling effort should provide answers framed in similar terms. Determining whether a numeric criterion will be achieved may require a different strategy than would be required for a narrative criterion. Further, modeling for a numeric criterion that specifies an average concentration over a large area will require a different level of detail than for the evaluation of a numeric criterion based on point concentrations in time.

User Criteria comprise the functional and operational needs of the user.

Assuming a variety of models are available to provide an adequate physical description of the system, and to provide answers to the appropriate regulatory questions, then selection among the candidate models will involve consideration of such things as available resources, ease of use, and so on. *Functional* needs refer to such issues as ease of use and communication of results, availability and adequacy of documentation, and model complexity and data requirements. The level of effort required to couple particular washoff and receiving water models can be an important functional criterion. Use of a model that is poorly documented and not well known will increase the difficulty of communicating and gaining acceptance of the results. Another key issue is the interface between modeling and monitoring. A complex model implementation is of little value unless adequate data are available for calibration and verification. In many cases, the best approach may be an incremental effort, in which simple screening models are used with initial, usually limited, data. These results may then suggest a more detailed modeling and monitoring strategy. Implementation of the more detailed model will then require additional refinements in monitoring for calibration/verification, and so on. *Operational* needs reflect both the requisite technical ability to implement the model, and the estimate of cost and time requirements for the implementation. These criteria provide the cost side for a cost-benefit analysis of model selection. In general, use of more complex models can result in significantly higher costs, due both to implementation effort and amount of data required for model calibration. This cost must be balanced with available resources, and the benefits to be achieved through the effort. The benefits of modeling are defined through 1) additional protection against excursions of WQSs, and 2) minimization of the economic loss incurred by requiring the maintenance of an excess MOS to compensate for uncertainty in predictions regarding the system. In many instances, lower cost, simpler modeling approaches will often be available which will enable completion of a TMDL, given that an adequate MOS is calculated to account for the simplification. Greater modeling effort can then result in the benefit of a reduced MOS required to protect designated uses (and thus a lesser burden on PS and NPS dischargers), but at the cost of increased resource requirements for modeling.

4.2.2 Idealized Approach to Modeling Strategy

As noted in the previous section, there are three general sets of considerations or modules that control the determination of a modeling strategy: 1) technical criteria (physical characteristics of the chemical and system); 2) regulatory criteria; and 3) user criteria (functional and operational needs). These three modules should interact as a three-dimensional matrix to determine the optimal modeling strategy (see Figure 4-3).

In theory, one could evaluate all three modules simultaneously, leading to the determination of an optimal modeling strategy. However, simultaneous evaluation,

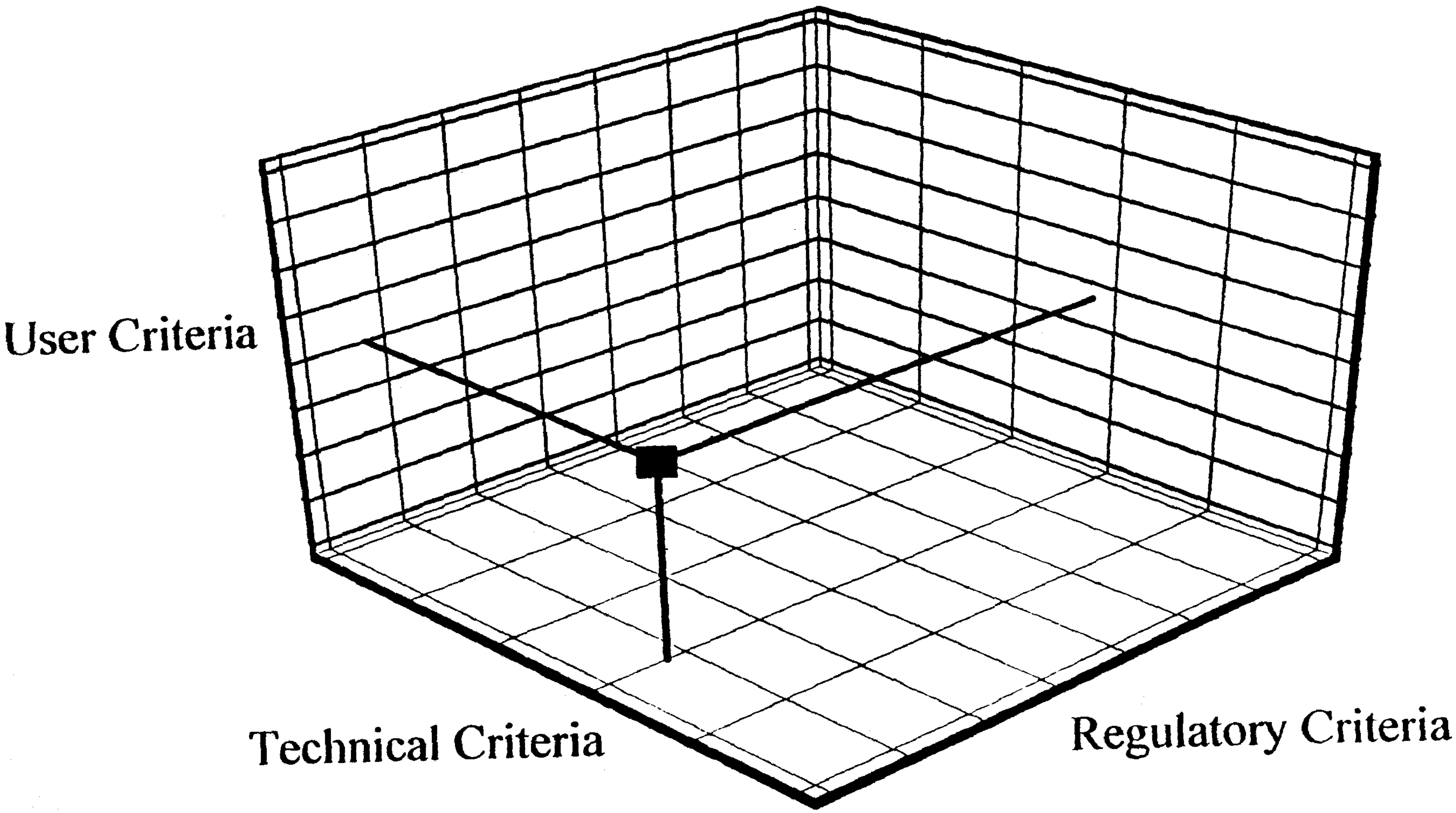


Figure 4-3. Determining an Optimal Modeling Strategy for the TMDL

while it could be addressed in an artificial intelligence context, is difficult to formulate into written guidance. It is also desirable to separate the determination of physical characteristics from the other modules, as it may be necessary to address multiple chemicals, as well as differing levels of regulatory criteria and functional needs for a single set of physical characteristics. (Noting, however, that some physical characteristics of the receiving water may also change in response to changes in loads.)

An *idealized* approach, which might be followed by an expert with detailed knowledge of a specific waterbody, is shown in Figure 4-4. This represents a "top down" approach, in which a thorough, pre-existing understanding of the site is used to select an appropriate modeling strategy. (This will be contrasted with a *practical* approach in 4.2.3.)

The top box in Figure 4-4 addresses the determination of physical characteristics, and also shows the functional relationships of some of the major subcomponents within this box. The assessment of the physical characteristics (of the chemical, receiving water, episodic source loads, etc.) forms the first module, and yields the necessary evidence to formulate an optimal descriptive model strategy for the system - i.e., the model strategy which might be chosen to advance scientific understanding of the behavior of the pollutant in the system if time, money, and the particular form of the applicable regulatory criteria were not of concern.

This module identifies important physical characteristics of chemicals, sources and hydraulics relevant to modeling. It has a primary aim of describing the level of detail appropriate to simulations. This in turn depends on characteristics of the chemical (e.g., reactive or conservative), of the waterbody (e.g., complexity of flow) and loading (e.g., constant vs. wet-weather episodic). The basic strategy is shown in the top box of Figure 4-4. We begin by identifying important characteristics of the chemical(s) under study as well as hydraulic behavior of the receiving water body. The degree of spatial and temporal detail required in a receiving water model is determined as a function of both chemical and hydraulic characteristics. For instance, if we are studying impacts of nutrient loadings where the response time of the waterbody is relatively slow it may not be necessary to study short time or small spatial variability in nutrient concentrations in the waterbody. In other words, we might make do with a description of average transport processes, at a time scale appropriate to the response of the waterbody, rather than requiring a minute-by-minute dynamic simulation.

Modeling of wet-weather loads is, of course, partially dependent on pollutant characteristics. However, the spatial and temporal scale for wet-weather load modeling will also be driven by the requirements of the receiving water model. That is, if the receiving water model needs daily loads only, it may not be necessary to simulate hourly loads, unless this is the best way to get at daily totals.

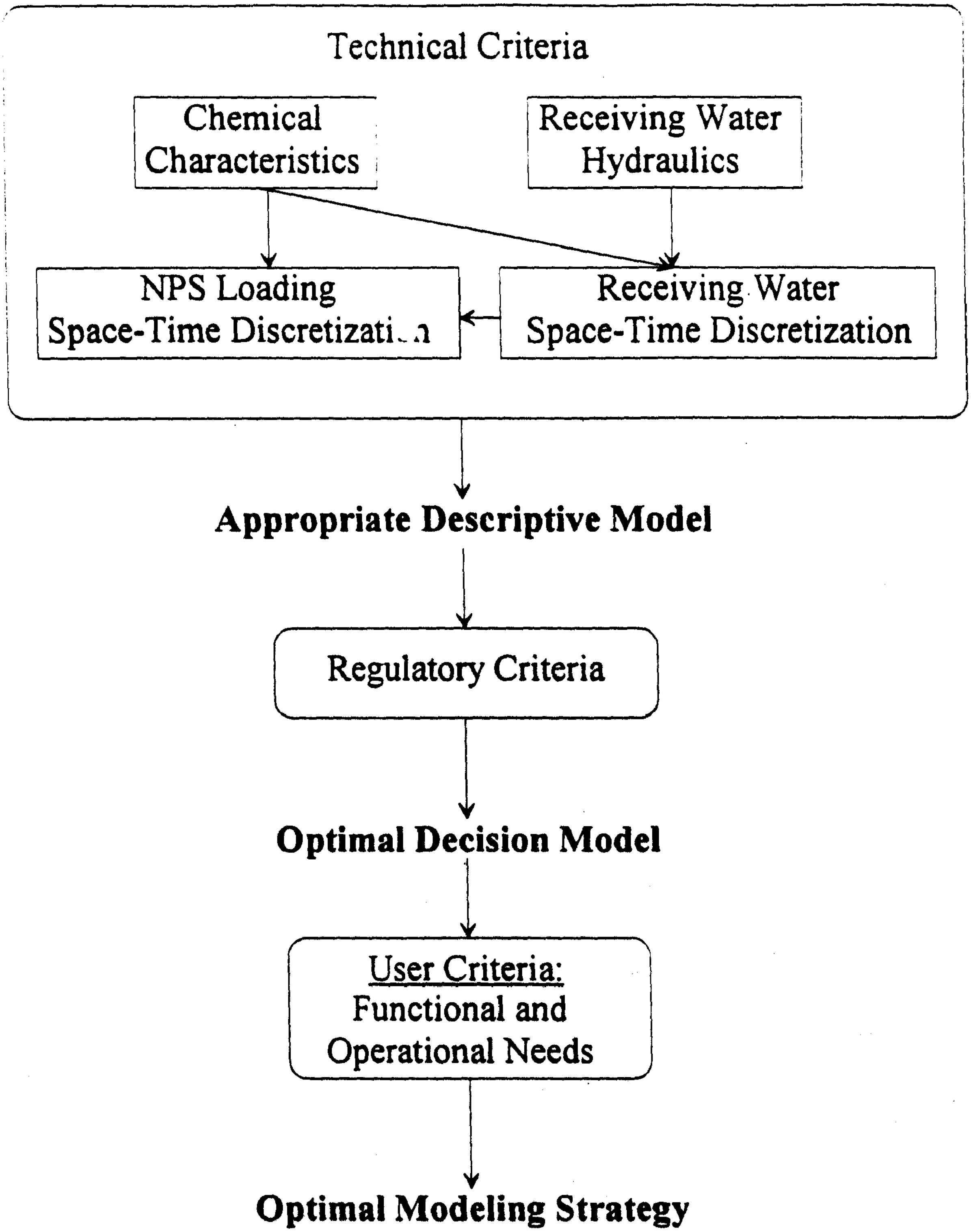


Figure 4-4. Idealized Approach to TMDL Modeling Strategy

In the second module, the descriptive modeling strategy is filtered through the applicable regulatory criteria. Here the modeling strategy is tuned to provide an efficient way to answer specific regulatory concerns. Combining the requirements of the descriptive model with the filter of regulatory criteria results in the specification of an optimal decision model strategy. This is the strategy which might be chosen if both the physical characteristics and regulatory criteria were taken into account, but issues such as cost, availability of models and so on were still not included.

In the third module, the decision modeling strategy is filtered through these functional and operational needs. This results in the formulation of a practical modeling strategy which can be applied to the site. Derived in this way, such a strategy attempts to honor the physical characteristics, and the regulatory decision criteria, while also seeking an optimal expenditure of money and effort to reach a target level of precision in the answers.

When substantial expert knowledge of the system is already available, the idealized top-down approach to modeling strategy shown in Figure 4-4 could be applied. However, a practical and generalized approach to the selection strategy, applicable to a wide range of sites, will proceed somewhat differently. This should recognize that the TMDL analysis typically begins before perfect knowledge of the characteristics of the site is attained, and that the analysis may proceed in phases, from simple to complex. The idealized procedure of Figure 4-4 must be rearranged in terms of specific strategic criteria, which interact with one another and with various components of the three modules.

The real-world, practical problem of model selection with limited resources and incomplete data may be rather different from the idealized, theoretical process schematized in Figure 4-4. A dichotomy often exists between the two viewpoints, which can result in technical modeling guidance (expressing what ought to be done, given unlimited resources) being perceived as of little relevance to the solution of real-world problems (where practical constraints on resources are a primary determinant of modeling strategy). These practical constraints, which come under the heading of Functional and Operational Needs, assume up-front significance in the practical approach to model selection.

How can these two viewpoints be reconciled in addressing the problem of TMDLs with episodic, wet-weather PS/NPS loading? In general, the *most accurate* mathematical description of wet-weather PS/NPS loading to a receiving water would require use of sophisticated, time-varying simulation programs, requiring a high level of expertise and considerable expenditure of time and money. However, the objective of the TMDL is the achievement of WQSs and protection of designated uses, to which accuracy is a tool, rather than an end to itself. This means that a less accurate, but more readily implemented, steady state representation of episodic loading can often be used to provide at least a preliminary assessment, given the determination of an adequate MOS. As discussed in Section 4.2.1, there is generally a tradeoff between increased accuracy (and smaller required MOS) and cost/resources required to calculate the TMDL.

In practical terms, most every TMDL will commence with the use of simple steady-state mass balance models for scoping the problem (unless sophisticated model representations of the water body are already available and calibrated). This can be called a Level 0 representation (see Table 4-1). A Level 0 representation is relatively low cost, but cannot represent time variability of the wet-weather loading, or frequency of point concentrations, and therefore will require a relatively high MOS (often accomplished through the imposition of worst-case assumptions).

If the Level 0 representation is unsatisfactory, because the uncertainties in the analysis or the required MOS are too high, additional, more sophisticated effort must be brought to bear on the analysis. For instance, the mass balance scoping could be replaced with a steady-state simulation analysis of the receiving water body. This would help to refine the estimation of point concentrations in the receiving water, but would still require use of worst case assumptions to provide an adequate MOS for episodic wet-weather loading (e.g., maximum probable event loading combined with antecedent drought conditions in a receiving stream).

We can conceive of the process of moving from simple, lower cost representations to more complex, higher cost representations as a ladder, in which we start from Level 0, and climb up to higher levels. How far is it necessary to climb? This must be determined on the basis of the tradeoff between cost (and available resources) and accuracy (and required MOS). The exact specification of the components of this ladder will vary from case to case. However, the general concepts remain the same. A typical example is provided in Table 4-1. This could represent the succession of Levels for a problem involving conventional pollutants, such as BOD, where response of the waterbody is spread over space and time. For a decaying constituent with acute toxic effects the Levels would likely be defined somewhat differently, with greater emphasis on the time variability of the loading.

Corresponding to the example specifications of Table 4-1, it will be necessary to specify the spatial and temporal resolution of the simulation of the wet-weather loadings, the receiving water hydraulics, and the chemical transport and reactions. These will depend on the characteristics of the system and pollutant. For example, the types of description of receiving water body hydrodynamics which might correspond to the levels in Table 4-1, by waterbody type, are shown in Table 4-2.

Table 4-1. Example Levels of Analysis for TMDLs with Episodic NPS Loading and Receiving Water Models (Conventional Pollutant)

Level of Analysis	NPS Loading	Receiving Water	Cost	Required MOS	Data Require-ments
Level 0 (Scoping)	Steady/ Generic or RM	Mass Balance/ Worst Case	Low	Very Large	Low
Level 1	Steady/ Average or RM	Steady	Low	Large	Low
Level 2	Seasonal/ Average or RM	Seasonal	Low-Moderate	Large	Low - Moderate
Level 3	Seasonal/ Event Mean	Daily	Moderate	Moderate	Moderate
Level 4	Event	Dynamic	High	Small	High

In sum, an "ideal" modeling strategy must reflect the physical characteristics of the site and pollutant, yet also reflect available resources, prioritization, state of phased TMDL analysis, and acceptable levels of uncertainty. This presents a complex set of interlocking goals, the resolution of which will vary from site to site. Specifying the complete criteria for an ideal modeling strategy is not possible *a priori*, as each receiving water body is likely to exhibit its own peculiarities. However, it is possible to construct logically ordered suggestions to assist in the proper formulation of such a strategy, recognizing that the picture must be completed with site-specific knowledge.

The structured decision process is presented in Section 4.4 as a sequence of logically constructed questionnaires and decision trees. These are designed to help the user develop a set of formal model selection criteria, which may then be matched to the available models, such as those described in Chapter III.

Table 4-2. Example Levels of Analysis for Receiving Water Models

Level of Analysis	River	Lake	Estuary
Level 0	Mass Balance	Mass Balance	Not Applicable
Level 1	1-D Steady	1-D/ 2-D Steady	1-D/ 2-D Tidal Average Screening
Level 2	1-D Seasonal Steady	2-D Steady with Overturn	2-D Seasonal Tidal Average
Level 3	Routing of Daily Average Flows	Time-Varying Mass Balance	2-D/ 3-D Inter-Tidal Quasi-Dynamic
Level 4	Dynamic Flows	Energy Budget Circulation	2-D /3-D Intra-Tidal Flows

4.3 Specific Model Issues for Simulation of Loading and Impacts from Episodic, Wet-weather Events

Before presenting detailed suggestions for the model selection strategy in Section 4.4, it is desirable to provide a discussion of some of the specific model issues that arise in attempting to model episodic, wet-weather PS/NPS for TMDLs. Why does TMDL modeling involving such sources tend to be difficult? Some, among many, causes are summarized below.

4.3.1. Episodic Nature of Loads

WLA modeling is often conducted with the assumption that loading from sources is constant or changes infrequently, and that source strength is monitored and known. This allows use of simulation models using steady-state loading assumptions. Episodic, wet-weather events can contribute loading to the receiving water in infrequent, short but intense pulses. The loading function is thus very much dynamic, or non-steady state, and models which assume steady, or quasi-steady loading rates cannot be expected to represent the short-time variability of pollutant concentrations. However, this does not mean that dynamic receiving water models are always required. The fact is that the quality response of most receiving waters is relatively insensitive to short term variations in load rate, at least for conventional pollutants. For instance, the response time of lakes and bays to nutrient loadings is generally on the order of weeks to years, while the response time of large rivers to OD is on the order of days (Donigian and Huber, 1991). These relatively long response times mean that steady state receiving water models can yield appropriate results, if applied correctly, when the pollutant of concern is mixed into a relatively large area prior to manifestation of the response. On the other hand, response to acute toxins may require a small time step and careful consideration of the dynamic timing of the load.

Where steady state receiving water models can be applied, we may need only average loading rates from wet-weather events. However, the episodic nature of loading must be taken into account in simulating wetd-weather loading if the receiving water models demand any more detail than annual average or totals. This usually involves at a minimum the simulation of the occurrence of wet-weather runoff events from precipitation. However, in most cases it is sufficient to resolve only to the event total load scale. The case studies include successful TMDLs performed by linking episodic loading models to steady state receiving water models. Section 4.4 provides suggestions on the conditions under which steady state receiving water models with averaged wet-weather loading can be used, and when, conversely, a dynamic approach to loading should be used.

4.3.2 Definition of Critical Events

WLAs for constant discharges are often calculated on the basis of a design flow (e.g., the 7Q10 flow, which is the seven day average low flow with a 10 year return period). Design flows are chosen to provide a certain degree of protection against water quality excursions by evaluating effects of the source under the most stressful conditions which reoccur with a certain specified frequency.

EPA has recommended to the States that water quality criteria statements be made with an appropriately defined duration (averaging period) and frequency of excursion (U.S. EPA, 1991). In 40 CFR 122.45(d), EPA requires that all NPDES limits be expressed, unless impracticable, as both average monthly and maximum daily values. Similarly, the *Guidelines for Deriving National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (U.S. EPA, 1985) recommends a generic form for a water quality criteria statement:

> . . . aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of [pollutant] does not exceed __ µg/L more than once every three years on the average and if the one-hour average concentration does not exceed __ µg/L more than once every three

years on the average.

While a frequency-duration format is preferable, a design flow approach is acceptable for WLAs, because the design flow is translatable into a corresponding recurrence frequency (U.S. EPA, 1986). Use of design flows thus serves as a *surrogate* for a probability of water quality excursions (as defined on a specified duration).

Unfortunately, the direct correspondence between frequency of excursions and design flows is based on the assumptions that (1) loads are steady, and (2) rate of loading is not correlated with instream flows. This is clearly not the case for episodic, wet-weather loads. For example, consider the case of overland NPS BOD loading from agricultural runoff into a stream. It is evident first that the bulk of the loading occurs during storm events which yield sufficient precipitation to produce overland flow and sediment transport to the stream. The loading is not steady, but occurs in intense pulses, although, in the case of BOD, the effects may be felt for days to weeks afterward. Further, the same precipitation event which caused the loading to occur is also likely to increase flow in the stream, and little loading will take place during extended droughts. Thus, the loading and streamflow are positively correlated with one another. Now suppose we try to impose a conventional design flow criterion and approximate the total annual load in one of two ways: If we assume that the load is steady, and divide the total load among 365 days, the LA resulting from a steady-load design flow may seriously misrepresent the potential effects that may occur if the bulk of the total loading actually occurs in a small number of major storm-washoff events. This could lead to the analysis not being protective of water quality. On the other hand, if we try to examine the actual episodic loads in combination with the steady-state design flow criterion, we are likely to be extremely over-protective - because high loads are very unlikely to occur in combination with low flows.

How then do we determine for episodic sources WLAs/LAs that are likely to meet WQSs (with a given degree of assurance or probability of exceedance) and will be protective of designated uses? One alternative would be for the States to formulate and apply wet weather water quality criteria. This is an alternative which EPA has encouraged, but which is not currently available in most circumstances. Another reasonable alternative is to recognize that the design flow approach cannot serve as a reasonable surrogate for frequency-duration analysis of water quality excursions when a significant component of the loading to the waterbody results from wet-weather, episodic loads. This implies that one should simulate the actual frequency of WQS excursions, rather than relying on design flows, when evaluating wet weather-dominated TMDLs.

Accurate estimation of frequency of excursions of WQSs for waterbodies with wet-weather dominated loading generally involves continuous simulation over a number of years of precipitation records, and is a logical way to proceed, when sufficient resources are available to undertake such an analysis. However, continuous simulation is not always feasible during scoping or the early phases of a Phased TMDL, due to lack of data or constraints on available resources to perform the modeling analysis. In such cases, a quick, but conservative analysis is often needed to provide a preliminary analysis of the problem, constituting an analogy to the steady-state design condition.

Without simulation of the correlation between wet-weather loads and receiving water flows, such an approach cannot be expected to yield highly accurate results. However, for a scoping analysis it is often appropriate to construct a design condition which consists of (1) a wet-weather event load of a given recurrence frequency, combined with (2) steady-state or (seasonal) average flows in the receiving waterbody. In essence, this approach makes a "good guess" at the appropriate design condition, involving high loads and associated flows (because the highest loads are likely to coincide with higher than normal flows, use of an average flow is a relatively conservative approach to determining the appropriate load-flow pair, yet avoids the obvious inconsistency of combining high, precipitation-driven loads with drought flows). At the lower, or less complex Levels of analysis described in Section 4.2, we will refer to this as the combination of a "reasonable maximum" load and a steady-state, average flow. "Reasonable maximum" can be defined in terms of the desired recurrence interval of the load; combination with average flows then results in a somewhat more conservative analysis of recurrence of WQS excursions (i.e., an inflated MOS), appropriate to the early phases of analysis.

4.3.3. Stochastic Nature of Loads

In terms of the available assimilative capacity for WLAs, the conceptual equation for TMDLs (presented in Chapter 1) may be rewritten as

$$WLAs = LC - LAs - MOS$$

The analysis is complicated by the fact that the LAs reflect an estimate of a process with a significant random component. That is, wet-weather loading is a function of storm intensity, duration, and frequency, which must be described probabilistically. As noted above, and as EPA has recommended in the WLA guidance, and stated most clearly in the recent TSD for toxics (U.S. EPA, 1991), criteria statements should be made in a format with an appropriately defined duration (averaging period) and frequency of excursion. The fact that criteria and permit conditions take, or are recommended to take a duration-frequency form requires that modeling for TMDLs should provide similar information: that is, the modeling and monitoring activities should result in an estimate of the frequency distribution of receiving water concentrations, rather than just worst-case and/or average estimates. This is particularly important for episodic, wet-weather loading, as the concentrations and impacts of these types of sources are sensitive to variability in both washoff rates and streamflow. Further, in the typical case washoff rates and receiving water flow are strongly, but not completely correlated with one another, complicating the analysis.

In the TSD for toxics, EPA (1991) recommends three dynamic receiving water modeling techniques to yield the equency distribution of receiving water concentrations: continuous simulation, Monte Carlo simulation, and lognormal probability modeling.

Continuous Simulation Models combine daily (or other time step) measurements or synthesized estimates of effluent flows, effluent loads, wet-weather source concentrations/loads and receiving water flows to calculate receiving water concentrations. A deterministic model is applied to a continuous time series of these variables, so that the model predicts the resulting concentrations in chronological order with the same time sequence as the input variables. This enables a frequency analysis of concentrations at any given point of interest. The analysis automatically takes into account the serial correlation that may be present in flows and other parameters, as well as the cross-correlations between measured variables. This is potentially the most powerful method available for accurate prediction of the frequency of receiving water concentrations. However, it does have its disadvantages. First of all, it is very data intensive. Further, long time series of wet-weather loads will generally not be available, and these will have to be synthesized from precipitation records. This introduces uncertainty, and if any other input time series are lacking the uncertainty will likely be so great that Monte Carlo methods are preferable (see below). EPA (1991) recommends that if recurrence intervals of 10 or 20 years are to be examined for WQS excursions, at least 30 years of flow data should be available to provide a sufficient record to estimate the probability of such rare events.

Monte Carlo Simulation Models combine probabilistic and deterministic analyses. That is, this approach uses a deterministic water quality model with statistically described inputs. The model is run repeatedly, at each application drawing a random realization from the input statistical distributions. If all the time-varying inputs (such as flows) and uncertain parameters are described statistically, the result is a simulated set of receiving water concentrations which reflects the statistical distribution of the model inputs; however, these concentrations will not follow the temporal sequence of real data. The Monte Carlo method is potentially very powerful, and requires somewhat less data than continuous simulation. A particular strength is its ability to provide a direct assessment of model uncertainty by use of statistical representations of uncertain parameters.

Lognormal Probabilistic Dilution Models have been developed by EPA to provide a simpler method of frequency analysis. As implemented in EPA's DYNTOX model (LimnoTech, 1985), the lognormal probabilistic approach takes a simple stream dilution model, assumes that all the input parameters can be represented by lognormal distributions, and uses numeric integration to derive the resulting distribution of receiving water concentrations. While simple to use, the model is limited to application to rivers and streams, does not include instream fate processes, and cannot correctly analyze more than a single pollutant lo: 'ing source. The approach also assumes that all input parameters, including effluent flow and concentration, can be described by lognormal distributions. The latter assumption will often not be appropriate for analysis of impacts of episodic wet-weather loading over a period longer than the average runoff event, as wet-weather flows can be equal to zero for long periods between events. For these reasons, lognormal probabilistic modeling will in many cases not be applicable to calculating TMDLs which involve episodic, wet-weather sources. However, it may be of use in certain simple cases, such as short-response impacts from a localized wet-weather source, or situations in which the wet-weather source can be regarded as a relatively constant background in relation to a dominant point source.

For some cases involving TMDLs with episodic, wet-weather loads, a detailed prediction of the frequency distribution of concentrations may be desired, but neither a full continuous simulation (requiring a long time series of wet-weather load estimates) nor a full Monte Carlo simulation (which simulates receiving water flows and thus ignores the effects of serial correlation) may be appropriate. In such cases, hybrid methods may provide a powerful alternative. For instance, simulation over a continuous time series of observed receiving water flows could be combined with a Monte Carlo representation of wet-weather loads. To do this properly, however, the probabilistic simulation of the wet-weather component would need to take into account the correlation between wet-weather loading and receiving water flows (since both are driven by precipitation). For instance, observations could be used to develop a conditional probabilistic model in which wet-weather loads were simulated conditional on flow and/or precipitation.

4.3.4. Special Issues for CSO/SW Loading Models

Urban pollutants loaded via CSO/SW represent nonpoint washoff processes, which are however collected in a sewer and discharged to the receiving water as a point source. Most available dynamic urban loading models combine hydraulic modeling of the sewer system with modeling of wet-weather contaminant loading to the system. However, these represent two very different aspects of the process. Because the hydraulic behavior of sewer systems is well understood, and the physical characteristics of the system can often be accurately defined, hydraulic calibration can often be accomplished with relatively high accuracy. On the other hand, the state of the art in predicting pollutant loading from urban sources is much less satisfactory, and event mean concentrations may be subject to a high degree of random, or at least poorly understood, noise. We therefore suspect that it will often be appropriate to undertake detailed simulation of sewer hydraulic response combined with simpler, spatially and temporally averaged estimates of pollutant concentrations in order to derive an estimate of the frequency distribution of pollutant loadings to the receiving water.

4.3.5. Other Loading Mechanisms

In dealing with wet-weather loading, we usually think of wet-weather washoff processes. However, certain other mechanisms can also be important in the total delivery of a given pollutant to a water quality-limited waterbody. Pathways to consider include:

- Secondary generation from sediments: Sediment bound pollutants in washoff may be deposited episodically, but release to the water column only gradually. In some cases, the loading may have both a direct (episodic) and indirect (quasi-steady sediment release) component.

- Dissolved constituents in rain itself may be of importance (acid rain).

- For waterbodies with large surface areas, atmospheric deposition of fine particulates or volatiles may be an important loading mechanism. For instance, in the Great Lakes, atmospheric deposition is thought to constitute a major portion of the ongoing loading of PCBs.

- In some climates, wind erosion of sediments may be an important loading mechanism.

- Dry-weather NPS can sometimes be important, involving loading from groundwater flux to streams or surface seepage. Diffuse acid mine drainage is a particularly salient example.

The loading mechanisms addressed in the preceding bullets are not within the focus of the present guidance. However, their occasional importance should be kept in mind by the TMDL developer.

This and preceding sections summarize only a few of the key issues of importance to modeling of the quality of receiving waters impacts by episodic, wet-weather loads. These and other issues are incorporated into the next section, which attempts to provide general guidance on the selection of appropriate models for the calculation of TMDLs which involve episodic, wet-weather loading.

4.4 Decision Criteria for TMDL Model Identification

We discussed the general criteria for model objectives for TMDLs in Section 4.2, while Section 4.3 presented in more detail some of the specific technical issues associated with modeling wet-weather, episodic loading. How can these many factors be combined into a logical procedure for model identification and selection?

In the following pages we provide an aid to TMDL model identification in the form of a set of questionnaires and brief decision trees. Working through these steps will enable the user to come up with a set of formal criteria (*e.g.*, necessity of simulating time-varying flows) that represent the characteristics of simulation models or scoping procedures that are appropriate to the waterbody and level of analysis of the TMDL procedure. The procedure does not recommend specific models, as the realm of available models is constantly changing and expanding. Rather, it yields a description of model characteristics which can be matched to the available models summarized in Chapter 3, or to other models identified by the user.

The decision trees are framed in terms of the Levels of Analysis for TMDLs, described in Section 4.2 (see Table 4-1), in which it is assumed that the process will begin at the simplest level that is appropriate, and add complexity as necessary. These were defined to range from Level 0 analysis (simple scoping) to Level 4 analysis (generally involving complex dynamic simulation).

The user should note that the decision trees provide only generic suggestions, which incorporate the authors' opinions and experience regarding common modeling situations. Specific exceptions to many of these guidelines can no doubt be identified. Therefore, results of this process should **always** be reviewed to make sure that they make sense in terms of site-specific conditions and needs - and modified as necessary.

STEP 0. BASIC CHARACTERIZATION (Questionnaire)

Step 0 consists of forming a catalogue of important data and characteristics to be utilized in the subsequent decision trees. The data needs are addressed in more detail in other sections; a recapitulation of the most important items appears below.

0.1. **Technical Criteria** Data requirements for technical criteria are concerned with the important physical characteristics of the receiving waterbody, pollutants of concern, and wet-weather loading sources. These data requirements are discussed in more detail in Chapter 6. Important points to consider include:

0.1.1 Identify parameters or group of parameters of concern for the present TMDL. These may include chemical pollutants, sediment, physical habitat modifications, etc. List constituent(s) under study, and important chemical and biological characteristics. Does the analysis require consideration of secondary effects (e.g., BOD load evaluated in terms of effect on DO downstream), or do only direct effects (e.g., a numeric criterion for the discharged constituent) need to be considered?

0.1.2 Catalog known and suspected sources of loading. Are there point sources? Are there wet-weather, episodic sources of the constituent? What are the relative magnitudes of the various sources? What is the spatial relationship of the sources?

0.1.3 If episodic, wet-weather sources are present and of significant magnitude, what are the associated land use types? What is the pattern of variability in wet-weather loading? Are nonpoint sources of types which may be amenable to control through BMPs?

0.1.4 Characterize initial physical boundaries for impact analysis in the receiving waterbody. Include a descriptive catalog of designated uses, extent of known impairment, location of any sensitive areas, etc.

0.1.5 Characterize receiving water flow regime. Usually, this question boils down to whether the receiving water is a river, lake, estuary, etc. Answering the question is not always as simple as it appears. This issue is addressed further in Step 4. Basic data requirements are: length scales of the system, advective velocities, and estimates of dispersion coefficients and reaction rates. These can be used to form dimensionless numbers and characteristic mixing times, which index the relative importance of advection, dispersion, stratification and chemical reactions in receiving water. Calculation of these numbers is shown in Box 4-1 (following Step 0). Note that more than one receiving water body type may be involved in the analysis (e.g., a river discharging into an estuary).

0.1.6 Identify chemical/impact parameter characteristics. For chemical pollutants, an estimate of rates of reaction or decay is often of great importance (see Bowie et al., 1985 for a basic reference). Is the chemical preferentially sorbed to sediment or dissolved? Does the chemical accumulate within the system so that prior years' loadings are important to understanding present effects? Where physical habitat impairment or ecosystem response is of concern, how is the wet-weather loading thought to be expressed in waterbody impairment?

0.2 **Regulatory Criteria** The regulatory criteria will determine what types of questions will need to be answered through use of models. These issues are discussed in more detail in Chapter 1.

0.2.1 What is the format of the relevant WQSs to be addressed by the TMDL? Do they represent numeric or narrative criteria, or both? If numeric criteria apply, they may include not-to-be-exceeded values, target concentrations expressed at critical design flows, or frequency-duration specifications for long-term averages. Do relevant narrative criteria allow a one-at-time approach to individual pollutants, or do they imply that a whole effluent toxicity approach should be followed (e.g., "toxic substances in toxic amounts")? Is a mixing zone allowed for point sources, and what criteria apply within and at the boundary, and beyond, of the mixing zone?

0.2.2 If a Phased TMDL is being pursued on the waterbody, what is the stage of phased TMDL process? Has previous TMDL or WLA analysis been made of this waterbody-pollutant pair, or of this waterbody and other pollutants? Was a phased approach undertaken involving BMPs for LAs which must be reevaluated?

0.2.3 What is the priority ranking for this TMDL?

0.2.4 Do regulations impact the area to be simulated by imposing jurisdictional boundaries?. The most natural unit for TMDL analysis with wet-weather loads is the watershed or basin. However, jurisdictional boundaries or regulatory constraints may preclude treatment of the whole basin within the TMDL. Describe any regulatory constraints on the boundaries of the watershed to be simulated in this step.

0.3 **User Criteria** These include the functional and operational needs of the user, including limitations of resources available. These issues are discussed further in Section 4.2.

0.3.1 Practical constraints and resources: What is the available level of resources for this TMDL? What is the time frame for completion?

0.3.2 What models and expertise are available in-house for the analysis? What is the level of experience available for analysis of similar problems?

0.3.3 Form a preliminary estimate of the acceptable MOS for the TMDL, which will depend on the level of uncertainty present in the modeling analysis and monitoring data. The level of uncertainty acceptable in the modeling analysis should be specified *a priori*, as noted in the discussion of modeling DQOs. This is included among the User Criteria because the acceptable level of uncertainty, or, alternately, the required level of accuracy, will be an important factor in determining the effort that must be expended to establish the TMDL. If previous TMDL/WLA analysis has been undertaken, the level of uncertainty in the earlier work should be evaluated.

Box 4-1. Calculation of Dimensionless Parameters & Characteristic Mixing Times

The use of dimensionless parameters to characterize receiving water bodies is largely due to work of Fisher et al. (1979) and Schnoor (1985), and is summarized in McKeon and Segna (1987).

The relative importance of advection and diffusion is given by the Peclet number, P_E, defined as

$$P_E = ul/D$$

where:

u is the mean velocity (L/T)
l is the segment length under consideration (L)
D is a dispersion coefficient (L^2/T)

This dimensionless parameter is a ratio of the advective transport process to the dispersive transport process. If the Peclet number is significantly greater than 1.0, the system is advection-dominated; if it is much less than 1.0, dispersion dominates the transport, at least for dissolved conservative substances.

The importance of stratification in an estuary can be evaluated through the Estuarine Richardson Number, proposed by Fischer (1972):

$$R = \frac{[(\Delta\rho/\rho)\, g\, Q_f]}{W}\, U_t^3$$

where:

R is the Estuarine Richardson Number,
$\Delta\rho$ is the difference in density between the river and ocean water (M/L^3),
ρ is the density of the ocean water (M/L^3),
g is the acceleration of gravity (L/T^2),
Q_f is the freshwater inflow (L^3/T),
W is the channel width (L), and
U_t is the root mean square tidal velocity (L/T).

Calculation of Characteristic Mixing Times (continued)

observations of the presence of stratification will usually be available. However, the degree of stratification may change seasonally, and the Estuarine Richardson Number provides a quick means for evaluating the potential for change, either seasonally or with altered inflow regimes.

Finally, the residence time of a degrading or reactive contaminant in the water body has an effect on the importance of the physical processes that must be modeled. For instance, if degradation occurs much more quickly than advective transport, it may not be necessary to model the advective transport. These relationships may be evaluated through the use of characteristic mixing times, suggested by Eschenroeder (1983):

The advection time is proportional to the principal length scale of the domain or area of interest, l, divided by the mean velocity, u:

$$t_A \sim l/u$$

The diffusion time is proportional to the square of the distance scale, W, divided by the dispersion coefficient, D:

$$t_D \sim W^2/2D$$

and may be defined for longitudinal dispersion along the main axis of flow (t_{DL}) or for transverse dispersion normal to the axis of flow (t_{DT}).

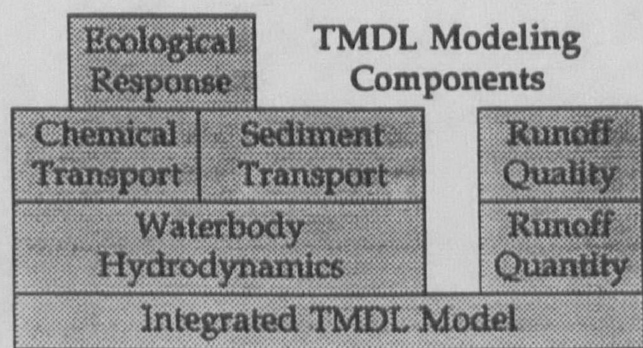
The importance of chemical reactions can be evaluated via the transformation time. The transformation time is proportional to the reciprocal of the first-order rate coefficient, k:

$$t_T \sim 1/k$$

The following steps provide a series of brief, relatively simple decision trees, in tabular format. These may be worked through in succession, and are designed to yield a description of appropriate modeling identification components for a TMDL analysis of a particular site.

In building a site-specific surface water modeling application we usually start from the basics (i.e., the flow of water) and add on additional compartments and complexity as necessary. For the purpose of sorting out a modeling strategy it is, however, convenient to work in the opposite direction: After an initial determination of the level of complexity to be pursued, we begin with the top-level components (e.g., ecological response) and work backwards to the base level of hydrodynamic simulation. This is because, in the context of decision for a TMDL, the necessary complexity of all the component models is driven by the level of complexity required in the response variable model, which will occupy the top level of the set of models chosen. At each step we first ask the basic question "Is a model needed?", and, if yes, set forth some generic decision criteria.

STEP 1. APPROPRIATE LEVEL OF ANALYSIS (Decision Tree)



Step 1 sets the general framework for developing an appropriate modeling strategy for calculation of a TMDL, and, as such, relates to all the component models to be employed (see box). The key to identifying a modeling strategy which is appropriate not just to the physical characteristics of the waterbody, but also to the objectives of the TMDL, and practical constraints of available resources is to select an appropriate Level of analysis (see Table 4-1) for the identification of simulation models. The Levels increase in complexity, and required effort, from Level 0 (simple scoping) to Level 4 (detailed dynamic simulation). As described in Section 4.2, the Levels may be defined in general terms as:

Level 0	Simple scoping analysis
Level 1	Application of simplified modeling representation, typically involving steady state modeling of waterbody hydrodynamics in one or two dimensions with annual average or reasonable maximum wet-weather loads
Level 2	Elaboration of Level 1 to include factors such as seasonal variability

and a more detailed spatial representation, but still utilizing a quasi-steady state representation of hydrodynamics

Level 3 Simplified dynamic simulation of receiving water hydrodynamics, which may involve hydrologic routing of daily or tidally averaged flows and some form of continuous, event-based representation of wet-weather loading.

Level 4 Detailed dynamic modeling of receiving waterbody, usually coupled with continuous simulation of wet-weather loading.

Selection of the appropriate Level of analysis for the current effort begins with the consideration of any past TMDL analyses:

1. Current state of TMDL analysis for the site and pollutants under examination:
   a) New TMDL (no previous analysis for this pollutant on the waterbody); or preliminary assessment to identify waterbody impairment -> 2
   or
   b) Previous TMDL work has been undertaken on the waterbody (addressing the pollutant of present interest). Current effort may involve a reevaluation of WLAs and LAs based on additional monitoring data obtained under a Phased TMDL approach, or a scheduled review of permits within a basin -> 3

2. New TMDL for this pollutant/waterbody, or preliminary assessment.
   a) Established modeling base available for waterbody from prior TMDL or other work -> 2.1
   or
   b) No established modeling base -> 2.2

2.1 New TMDL, established modeling base. Evaluate appropriate level of analysis based on review of previous modeling.

2.2 New TMDL, no established modeling base.
   a) High priority TMDL, impairment of special areas of concern, or reported violations of human health-based WQSs at beaches, drinking water intakes, and other exposure points -> 2.2.1
   or
   b) Other -> 2.2.2

2.2.1 High priority new TMDL.
   a) Seasonal variability in wet-weather loading significant : Begin analysis at Level 2
   or

b) Impairment of waterbody involves nonlinear interactions among different pollutants: Begin analysis at Level 2.

or

c) Flow in receiving water body is intermittent: Begin analysis at Level 2

or

d) wet-weather loading (on average) appears to be relatively stable over seasons, and (b) and (c) do not apply: Begin analysis at Level 1 and increment if necessary.

or

e) Analysis of excursion of WQSs at human exposure points or special areas of concern requires analysis of upstream movement of pollutants in an estuarine system: Begin analysis at Level 3.

2.2.2 Lower priority new TMDL or preliminary assessment.

a) Known impairment of waterbody involves nonlinear interactions among different contaminants: Begin analysis at Level 2.

or

b) Flow in receiving water body is intermittent: Begin analysis at Level 2

or

c) Tidal estuaries and other systems in which advective and macro-dispersive forces are both prominent in the transport of pollutant loads. This is generally characterized by the mixing times t_A and t_D being of approximately equal magnitude: Begin analysis at Level 1.

or

d) Other, including preliminary assessments: Begin analysis at Level 0

3. Continued TMDL for a given pollutant and waterbody. Examine results of previous modeling analyses:

a) Ongoing monitoring shows previous model predictions not within acceptable range of accuracy -> 3.1

or

b) Prior MOS unacceptably large, or level of uncertainty too high -> 3.1

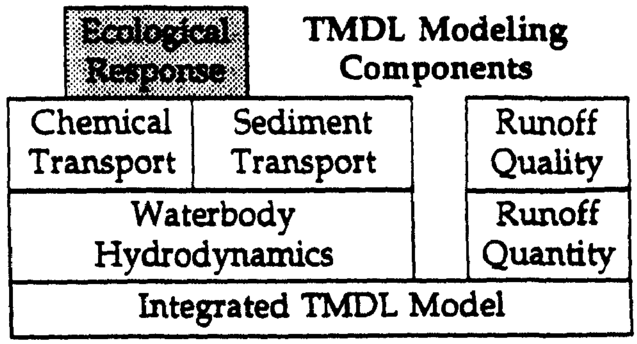
or

c) Neither (a) nor (b) applies -> 3.2

3.1 Previous TMDL effort needs revision: Increment to next level of analysis, of, if previously at level 4, reexamine modeling assumptions.

3.2 Previous TMDL level of modeling acceptable. Revisit analysis at same level, incorporating additional data gained in monitoring.

STEP 2. ECOLOGICAL RESPONSE MODELS



After having established the general Level of analysis in Step 1, we now return to a potential top-level component of the modeling application: ecological response. Ecological response models are used to predict ecological responses to ecosystem degradation or restoration. They are used to establish TMDLs/WLAs/LAs to meet habitat criteria and biocriteria. Many other TMDLs will not involve ecological response models.

1. First, determine if an ecological response model is needed.
   a) Physical habitat degradation is of concern in waterbody impairment, or interpretation of waterbody impairment or restoration requires examination of responses of biological community, via population dynamics, biomass production, or biodiversity -> 2

   or

   b) Evaluation of waterbody impairment or restoration does not require direct evaluation of biological responses - for instance, the impairment can be characterized by excursions of numeric criteria for chemical constituents or by narrative criteria to be evaluated in terms of concentration of the pollutant, <u>and</u> physical habitat restoration techniques are not under consideration -> 3

2. Situation requires use of some form of ecological response model (whether empirical or simulation).
   a) Ecological response of interest involves physical habitat modification -> 2.1

   or

   b) Ecological response of interest involves population dynamics, biomass production or biodiversity measures -> 2.2

2.1 Habitat modification models. Analysis is often piggybacked on to receiving water models and/or population dynamic models. Effects usually considered to fall into one of two broad classes:

   a) Alterations in stream morphology (e.g., variations in width and depth, creation of erosion controls, settling ponds, etc.) -> 2.1.1

   or

   b) Alterations in stream flow volume or timing, transport dynamics, or chemical dynamics which alter input to other modeling components (e.g., creating reaeration drop structures or planting streambank trees to reduce solar energy input) -> 2.1.2

2.1.1 Stream morphology models.
• TMDL analysis Levels 0-2: Use empirical methods and guidelines to estimate impacts, such as those outlined by Leopold et al. (1964), Dunne and Leopold (1978), Rosgen (1993) and others. Simulation of chemical and sediment transport may not be needed.
• TMDL analysis Levels 3-4: Consider combination of empirical methods with simulation of sediment movement in Step 4. Simulation of chemical transport may not be needed.

2.1.2 Habitat modifications which alter coefficients or inputs to other model components (such as chemical dynamics in the receiving water, receiving water hydrodynamics, or biological community models): Evaluate representation in the appropriate model component in subsequent steps.

2.2 Biological community models. Sophisticated simulation models for biological communities are generally experimental/research tools with limited predictive ability, and, at present, little used for TMDL development. The primary exception is algal biomass models, which are frequently applied.

a) Algal biomass models for systems dominated by planktonic algae (macrophytes and periphytic algae do not play a major role in nutrient cycling) -> 2.2.1

or

b) Other biomass production and population dynamic problems -> 2.2.2

or

c) Biodiversity or ecological indicator problems -> 2.2.3

2.2.1 Planktonic algal biomass models.
• TMDL analysis Levels 0-1: Employ empirical prediction methods and site specific correlations to relate biomass to chemical and hydrologic regime. These will usually require steady-state estimates of flow and nutrient availability.
• TMDL analysis Levels 2-3: Use steady state algal response models to simulate average response to seasonal average nutrient loading (e.g., algal components of QUAL2E and similar models).
• TMDL analysis Levels 3-4: Consider applicability of dynamic simulation of algal response to time varying loads (e.g., algal components of EUTROWASP). Nutrient concentrations will likely be required on a daily scale.

2.2.2 Other biomass production and population dynamic models.
• TMDL analysis Levels 0-3: Use empirical prediction methods and site specific

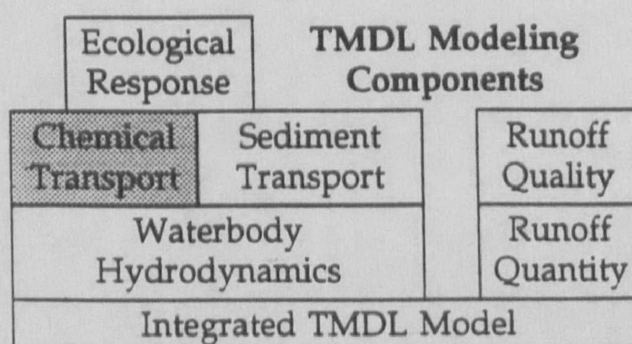
correlations. A useful review of models that attempt to predict standing crop of stream fish is provided by Fausch et al. (1988).

- TMDL analysis Levels 3-4: Consider quasi-steady state (gradually varying) simulation of population response, given adequate resources and data, in combination with empirical and observational methods. In many cases the focus of resources should be on effective monitoring.

2.2.3 Biodiversity and ecological indicator problems. Working simulation models are generally not available to address this class of problems; instead, a qualitative assessment is often based on comparison to reference sites. The TMDL developer should consult EPA's (1990) biological criteria guidance, as well as EPA's series on Use Attainability Analyses. Depending on the type of pollutant under consideration, estimates of biological criteria may require average or maximum chemical concentrations.

3 No ecological response model is needed; proceed to Step 3.

STEP 3. CHEMICAL TRANSPORT MODEL TEMPORAL RESOLUTION



In some cases, the TMDL will be driven by an ecological response to chemical concentrations; in others it will depend on chemical concentrations themselves. The time step required for receiving water simulation will depend to large extent on the temporal resolution required for chemical or sediment modeling. We therefore first examine the receiving water chemical simulation time step and related characteristics. This results in a decision on whether steady state or dynamic water quality simulation should be pursued, and, if dynamic, a qualitative indication of the maximum allowable time step. The spatial resolution for water quality simulation is evaluated in Step 6 along with the spatial resolution of the hydrodynamic model.

The time step discussed here simply reflects the desired temporal resolution of the model results, and will typically be considerably larger than the actual simulation time step used in a dynamic model of water quality. This is because most water quality models use explicit finite-difference solutions, in which case there are stability constraints on the maximum time step that can be used to ensure model stability (i.e., solution schemes in which errors introduced in the finite difference approximation tend to damp out, rather than amplifying without bound). These stability criteria relate the time step to the spatial step, flow velocity, dispersion coefficients and decay rate. In these cases, the size of the time step will be constrained by the spatial resolution required in the model, among other things. In determining the simulation time step, the user should consult documentation for the specific model.

1. Is a chemical transport model needed (as distinct from a sediment transport model alone)?

   a) Chemical concentration driven situations: WQS based on dissolved or suspended concentration; or TMDL based on ecological response expressed through dissolved or suspended concentration -> 2

   or

   b) Non-chemical problems involving physical habitat modification, alterations in flow (without a chemical component), or sediment (as a pollutant itself) ->3

2. Given that a chemical transport model is needed, start by sorting on the Level of the TMDL analysis.

   a) Concentration-driven analysis at Level 0, 1, or 2; or ecological response-driven analysis requiring averaged concentrations (as opposed to a continuous simulation of concentrations at points in space and time) -> 2.1

   or

   b) Concentration driven analysis at Level 3 or 4; or ecological response-driven analysis which requires a description of concentrations at specific points in space and time -> 2.2

2.1. Analysis at Levels 0 through 2. For the simpler analyses it is often, although not always, possible to use steady-state water quality approximations. The analysis can be based on average concentrations (for long response times) or reasonable maximum concentrations (for short response times). Note, however, that use of the steady state approximation generally means that an accurate estimate of the frequency of excursion of WQSs cannot be obtained; therefore the MOS must be increased to account for this uncertainty. The alternative, dynamic (time-varying) simulation of water quality (Levels 3-4) can provide more accurate results, but at the cost of considerably more modeling effort. The error introduced by the steady-state assumption, and possible means of compensating for it, differ widely with the type of waterbody under consideration. Therefore, next identify water body type (hydrodynamics and transport properties) as dominantly 1-D advective (e.g, river), dispersive (e.g., lake), or 2,3-D advective-dispersive (e.g., estuary).

   a) Typical river hydrodynamics: dominantly advective system with $P_E >> 1$ or region of interest -> 2.1.1

   or

b) Typical lake/reservoir hydrodynamics: dominantly dispersive system with $P_E \ll 1$, and t_{DL} and $t_{DT} \ll t_A$ -> 2.1.2

or

c) Situations neither clearly (a) nor (c), including estuaries, complex and wide rivers, etc. -> 2.1.3

2.1.1 Level 0-2 analysis for rivers (dominantly advective systems). For advective systems, an assumption of steady loading will often provide a relatively good representation of the maximum concentration response to a single episodic or slug input, whether analysis is required for near field concentrations, far-field concentrations, or secondary effects. This is because the pollutant is advected away from the source. In the far-field, assumption of a steady source is then equivalent to examination of the behavior of a discrete slug under the assumption that the concentration is not reduced by longitudinal dispersion. The simplifying analysis thus has a built in conservative assumption. For near-field analysis, the steady state assumption is merely a dilution calculation (with reactions considered if appropriate), and the steady representation of a slug input is again conservative. When multiple episodic sources are considered, the steady state approximation is equivalent to assuming that the peak concentrations from all sources coincide, which may be extremely conservative. General suggestions are:

- Levels 0-1: Use steady state water quality simulation.
- Level 2: Use seasonally steady water quality simulation, or superpose analytical solutions to the advection-dispersion equation for slug or episodic loads on a steady state hydrodynamic model.

2.1.2 Level 0-2 analysis for lakes and impoundments (dominantly dispersive systems). Short term variability in water quality in dispersive systems subjected to episodic loads is not as readily described by steady state analysis as advective systems. On the other hand, the response time of lakes to many pollutant inputs tends to be long, in which case analysis of average rates of wet-weather loading is often sufficient. It is important to examine the relative length of the response time of the waterbody compared to the average duration of a loading event. Are we looking at a short-term (acute) or long-term (chronic) response? For some pollutants, both pathways may need to be followed (e.g., when concentration limits for both shorter and longer averaging periods are included in a WQS). The criteria can also be applied to analysis of secondary or indirect effects, such as effects of BOD loading on a DO criterion. The user should, however, base the analysis on the response time of the parameter of interest (in this example, DO).

a) Numeric criterion with averaging period less than, or of the same order of magnitude as the average interevent time between loading events; <u>or</u> evaluation of short-term (acute), non cumulative direct effects of pollutant -> 2.1.2.1

and/or

b) Numeric criterion with averaging period greater than the average interevent time between loading events; <u>or</u> evaluation of long-term or cumulative effects where the response time of the waterbody is greater than the average inter-event loading time -> 2.1.2.2

2.1.2.1 Level 0-2 analysis for lakes, "short" response, where spatial concentration gradients must be considered to obtain a reasonable estimate of excursions of WQSs, and thus full lateral mixing of episodic loads is unreasonable to assume. This includes both near-field dilution and subsequent part. i mixing into the epilimnion. For dominantly dispersive systems, a steady loading approximation is not useful for episodic loads. Instead, screening approximations can be based on analytical solutions to the 2-D diffusion equation. Use of analytical solutions avoids the problem of specifying a simulation time step. For lakes, the episodic wet-weather inputs will generally be represented by a set of tributary discharges, and thus can be represented as intermittent point sources.

- Level 0: Use time-varying diffusion equation quality solution with worst case or reasonable maximum event loading. (Superpose solutions from different loads.)
- Levels 1-2: Use time-varying diffusion equation quality solution with observed or simulated loads.

2.1.2.2 Level 0-2 analysis for lakes, "long" response. The long response time means that a steady state analysis, based on average loading, can be used for the initial Levels of analysis. (Problems in spatial resolution for this approach are addressed in Step 5). The analysis can be based on a steady-state water quality simulation with average loads per year or season, or maximum average load over the response period, as appropriate to the pollutant under study. Do we need to consider accumulation of the pollutant in the system? This is often the case for low solubility, sediment bound toxics which may release slowly to the water column and result in a very slow response to changes in loading.

a) Role of accumulation is <u>not</u> of major significance; responses to change in loading should be fully expressed within a year -> 2.1.2.2.1

or

b) Role of accumulation needs to be considered -> 2.1.2.2.2

2.1.2.2.1 Level 0-2, "long" response without significant accumulative effects. Use modeling based on annual or seasonal average loading. As above, a steady state solution can readily incorporate first-order reactions, given steady loading and advective and dispersive fluxes in the receiving water body. Attainment of numerical concentration standards may be estimated by steady-state dilution of average loading, with sufficient MOS to account for year to year variability. For secondary effects, particularly eutrophication problems, the following water quality temporal representations can be used (see Mancini et al., 1983):

- Level 0: Preliminary scoping can often be accomplished by using empirical or regression approaches based on average loading. This is particularly well developed for analysis of lake trophic response to nutrient loads.

• Level 1-2: Steady state water quality simulations are generally appropriate, based on average loading rates.

2.1.2.2.2 Level 0-2, "long" response with significant accumulative effects. As in 2.1.2.2.1, steady state water quality simulations are usually appropriate; however, bed interactions may result in changes in loading taking a long time to reach steady state. Calculation of time to steady state is presented in Hydroqual (1986).

2.1.3 Level 1-2 analysis for estuaries and other advective-dispersive systems. (Level 0 analysis is not appropriate for this type of complex system.)

a) WQSs or responses require analysis of near field (mixing zone) concentrations or response for at least some loading sources -> 2.1.3.1

and/or

b) Far-field analysis required -> 2.1.3.2

2.1.3.1 Level 1-2 estuarine analysis including near field response. Evaluate dilution of discharge using steady state water quality mixing zone models. If far-field analysis is also required, complete simplified far-field analysis (2.1.3.2) and evaluate potential impact of far-field concentrations on local mixing zone concentrations.

2.1.3.2 Level 1-2 estuarine analysis for far-field responses. Steady state or quasi-dynamic simulations of water quality are appropriate at these Levels, utilizing loading estimates which are averaged at a time frame longer than the tidal cycle (e.g., daily to yearly). Appropriate water quality models are:

• Level 1: Apply steady state water quality analysis driven by annual or seasonal average loading to obtain an approximate prediction of annual or seasonal mean conditions.

• Level 2: Water quality simulation may be steady-state or tidally averaged (quasi-dynamic). Quasi-dynamic models should be able to predict water quality variations on the order of days to months.

2.2. Analysis at Level 3-4. These levels will generally involve time-varying (dynamic) simulation of water quality and pollutant routing in the receiving waterbody. The intention is to predict the probability distribution of receiving water concentrations, rather than just a worst-case or average concentration. EPA (1991) has recommended three dynamic modeling techniques for establishing WLAs: continuous simulation, Monte Carlo simulation, and lognormal probability modeling. However, lognormal probability modeling is generally not applicable to TMDLs involving episodic, wet-weather loads, as described in Section 4.3.3.

a) Analysis at Level 3 -> 2.2.1

or

b) Analysis at Level 4 -> 2.2.2

2.2.1 Analysis at Level 3: Quasi-dynamic water quality simulation is usually applicable. For rivers, this will usually mean modeling quality on a day to day basis, while for lakes a time-varying mass balance approach can be used with a maximum time step of days to weeks. For estuaries, a tidal period average time step is usually appropriate. The preferred type of dynamic or quasi-dynamic analysis can be determined as follows (see EPA, 1991):

a) Relatively complete time series of receiving water flow, and daily rainfall and evaporation (to estimate runoff), or direct measurements of runoff, representative of current conditions are available for a period approximately twice as long or longer than the return period that must be predicted; <u>and</u> sufficient data are judged available to obtain an adequate calibration of all model parameters: Continuous simulation over the historic time series is preferred as the most powerful technique to estimate frequency of WQS excursions.

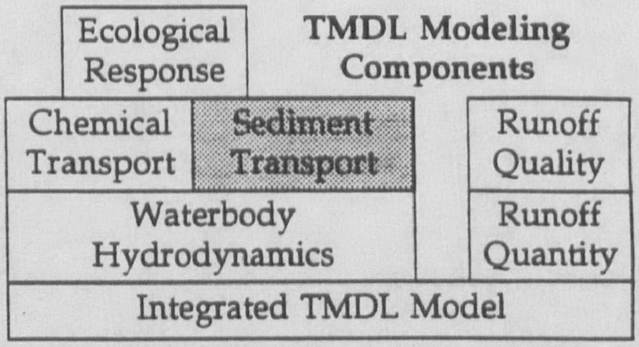
or

b) Time series records are not sufficiently long; <u>or</u> significant alterations in the system have taken place so that historic receiving water flows are not representative of current conditions; <u>or</u> role of parameter uncertainty must be examined: Monte Carlo simulation may be preferable, or a combined Monte Carlo-continuous approach (see Section 4.3.3). Note that the time step for full Monte Carlo simulation input should be defined as the averaging time in the relevant criterion under investigation.

2.2.2 Analysis at Level 4: Use fully dynamic water quality simulation. In terms of the requirements of the TMDL analysis, the maximum receiving water chemical time step should be the smaller of the averaging time specified in the criteria statement and the typical duration of a loading event. For cases in which the specified averaging time is greater than the typical event duration, the maximum time step is the event scale. The actual simulation time step will usually be smaller than this maximum value, due to model stability criteria, desired spatial resolution, etc. Determine the appropriate dynamic simulation method (continuous or Monte Carlo) as in 2.2.1.

3. No chemical transport model required.

STEP 4. SEDIMENT TRANSPORT MODEL



This step evaluates the need for and general type of sediment transport modeling to be employed in the receiving waterbody. Sediment transport will be of concern when it is a significant factor in transport of the pollutant of interest, or when sediment itself is of concern, either as a pollutant or a driving factor in morphological change.

1. Is a sediment transport component needed?
   a) Sediment itself is pollutant of concern -> 2
   or
   b) Stream morphology is to be modeled (assumed applicable only to streams/rivers)
   -> 3
   or
   c) Pollutant of concern tends to be sorbed to sediment or particulate matter. This includes hydrophobic organic compounds and most metals -> 4
   or
   d) Level 3-4 simulation of DO problems where sediment oxygen demand is significant, or Level 3-4 simulation of secondary water quality effects significantly influenced by cycling of sediment, organic carbon or particulate organic matter -> 4
   or
   e) Pollutant of concern is dominantly dissolved and transport in receiving waterbody is not controlled by movement of sediment or organic matter ->5

2. Sediment TMDL: Sediment component is needed. Return to Step 3 and analyze the time step based on sediment as a "chemical" parameter. For the purposes of the analysis of modeling strategy, treat sedimentation as a first order "decay" process.

3. Stream morphology TMDL, which may be based on empirical analyses or simulation of sediment transport.
   a) Analysis at Levels 0-2 -> 3.1
   b) Analysis at Levels 3-4 -> 3.2

3.1 Stream morphology TMDL at Levels 0-2. If changes in morphology are evaluated via empirical relationships no sediment model component will be needed.

3.2 Stream morphology TMDL at Levels 3-4. Sediment transport modeling may be required to support any stream morphology modeling determined in Step 2. Usually, only averaged measures of transport will be required. However, as bulk sediment movement is largely driven by infrequent flood events, a detailed analysis of hydrodynamics may be needed.

4. Strongly-sorbed pollutant simulation, or other situation in which movement of sediment and/or organic carbon will be needed to support the chemical simulation.

a) Analysis of organic compounds which preferentially sorb to organic carbon or organic matter; or dynamic analysis of sediment oxygen demand -> 4.1

or

b) Analysis of metals or other ionic compounds whose movement is associated with the movement of sediments -> 4.2

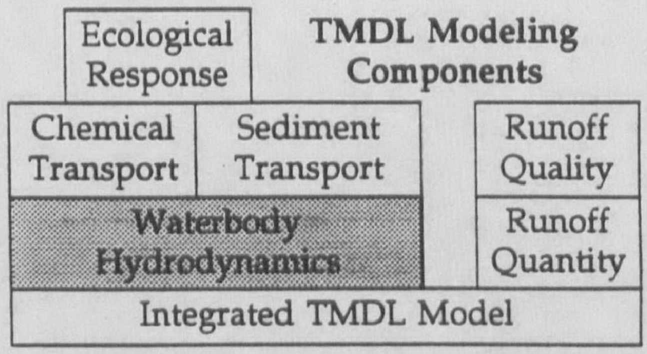
4.1 Simulation of strongly-sorbed organic pollutants. Accurate representation will need to consider organic carbon as a state variable. This may further need to be divided into particulate and dissolved organic carbon, the first of which is driven by fine sediment transport and exchange with the benthos. Algal organic carbon production may also need to be simulated for a full representation. Such complex models will usually not be feasible unless substantial resources are available; they are thus inappropriate for the simpler Levels of analysis.

- Levels 0-2: Dynamic simulation is not feasible at these levels; therefore detailed models of sediment and organic carbon transport are not needed. Sorbed transport may be approximated by consideration of sediment rating curves, literature partition coefficients, etc. - with the recognition that significant errors in prediction may result.
- Levels 3-4: The more advanced levels of analysis will usually involve dynamic or quasi-dynamic simulation of water quality. Where sorbents play an important role they should also be simulated, generally at the same time scale as the hydrodynamic model. Chemical reactions will also generally need to be considered.

4.2 Analysis of contaminants, such as many metals, associated with inorganic sediment component. Redox reactions, metal speciation, precipitation and dissolution may all need to be considered for a detailed understanding of the problem. Sediment transport can be simulated with a degree of sophistication similar to that described under 4.1.

5. No sediment transport component is needed for the TMDL.

STEP 5. HYDRAULIC MODEL TIME STEP/SOLUTION METHOD



This step addresses the time scale of the receiving water hydrodynamic component of the simulation, including whether a steady state or dynamic representation should be used. The actual simulation time step used in a model application will also involve consideration of chemical dynamics and regulatory criteria.

1. Level of present analysis:
   a) Level 0 scoping analysis, not including tidal estuaries -> 2
   or
   b) Subsequent level -> 3

2. Level 0 scoping analysis. Begin scoping with steady state mass balance or empirical approaches (no detailed simulation of flow). For instance, lake eutrophication problems might be scoped with regression methods relating trophic status to nutrient loading, while concentration of toxics could be estimated by dilution calculations.

3. Level 1 or greater TMDL analysis. Identify water body hydrodynamics and transport properties as dominantly 1-D advective (e.g, river), dispersive (e.g., lake), or 2,3-D advective-dispersive (e.g., estuary).
   a) Typical river hydrodynamics: dominantly advective system with $P_E >> 1$ or region of interest -> 3.1
   or
   b) Typical lake/reservoir hydrodynamics: dominantly dispersive system with $P_E << 1$, and t_{DL} and $t_{DT} << t_A$ -> 3.2
   or
   c) Situations neither clearly (a) nor (c), including estuaries, complex and wide rivers, etc. -> 3.3

3.1. Rivers and other dominantly 1-D advective systems
Level 1,2 analysis -> 3.1.1
or
Level 3,4 analysis -> 3.1.2

3.1.1 Level 1,2 analysis for rivers. Flow routing will usually not be necessary.

• Level 1: Use steady description of flow; select appropriate worst-case design condition

• Level 2: Use steady description of hydraulics with separate description for each season.

3.1.2 Level 3,4 analysis for rivers. Unsteady (time-dependent) flow effects should be considered.

• Level 3: Set maximum time step to smaller of maximum chemical time step (from Step 3), WQS averaging time, or 1 day. Route daily average flows using hydrologic or hydraulic routing; interpolate from daily flows if needed.

• Level 4: Simulate with hydraulic (momentum) routing. Appropriate maximum time step is likely to be determined by desired spatial resolution of output and model solution stability criteria relating time step to spatial increment.

3.2 Dominantly dispersive systems, primarily lakes. In these systems, the appropriate *hydrodynamic* representation at a given Level of analysis is generally the same whether nutrients, toxics, near-field, or far-field problems are under consideration.

a) Level 1,2 analysis -> 3.2.1

or

b) Level 2,3 analysis -> 3.2.2

3.2.1 Level 1,2 analysis for dominantly dispersive systems. At these levels of analysis, lakes can generally be represented by a steady state mass balance, with constant inflow and outflow to represent advective sources and sinks (Mancini et al., 1983). Within lake flows are generally not represented. However, at Level 2 it may be useful to consider seasonal variability in volume and rates of inflow and outflow. For short-response problems, the steady internal hydrodynamics may be combined with assumptions of reasonable maximum loading in inflows, or frequency distribution of such inflows.

3.2.2 Level 3,4 analysis for dominantly dispersive systems. At these Levels of analysis, time variability should be taken into account. However, because of the relatively slow response time of lakes to many pollutants, the time step may still be large compared to that required for simulation of advective systems. For any changes in loading, time to steady state may be quite long for many lakes, particularly where bed sediment interactions are important. In the WLA guidance, Hydroqual (1986) provides methods for calculation of time to steady state for simulation modeling of lakes.

• Level 3: This level corresponds to the Time-Varying Mass Balance models for lakes described by Mancini et al. (1983) and Hydroqual (1986). The lake may be represented as completely mixed, or as a set of mixed segments. Time variability in inflows and outflows (from the lake or between segments) is generally represented on a scale of days to weeks. This level of hydrodynamic representation can often be used to support at least approximate water quality simulations at a somewhat smaller time step, if needed.

• Level 4: This level comprises dynamic models of circulation within lakes. The hydrodynamic simulation will generally require use of hourly or sub-hourly maximum time step sufficient to model insolation and wind energy induced circulation patterns for continuous simulation.

3.3. Advective-dispersive systems, including most estuaries, some wide or hydraulically-complicated rivers, short- sidence time reservoirs, etc.

a) Systems with tidal boundary (estuaries) -> 3.3.1

or

b) Other advective-dispersive systems -> 3.3.2

3.3.1 Estuaries

a) Level 1,2 analysis -> 3.3.1.1

or

b) Level 3,4 analysis -> 3.3.1.2

3.3.1.1 Level 1,2 analysis for estuaries. Investigate using steady state approximations with tidal mixing approximated as effective dispersion.

• Level 1: Use steady description of flows based on observations and mass balance; select appropriate worst-case design conditions.

• Level 2: Use seasonally steady description of flows, perhaps based on hydrologic routing and allowing for seasonal changes in stratification where appropriate.

3.3.1.2 Level 3,4 analysis for estuaries.

• Level 3: Use daily inflows and tidally averaged internal flows to build a daily average advective-dispersive model. Simulation time step will be determined by model solution stability and desired spatial and temporal resolution of results.

• Level 4: Use hourly or sub-hourly maximum time step to model intra-tidal variability. Much shorter internal time steps (e.g., seconds) will be required many models to maintain stability in solutions.

3.3.2 Other (non-tidal) advective-dispersive systems. Exact determination of hydraulic time step will depend on site-specific conditions. General suggestions are:

• Level 1: Use annual average steady state approximation, with consideration of worst case design flows, where feasible.

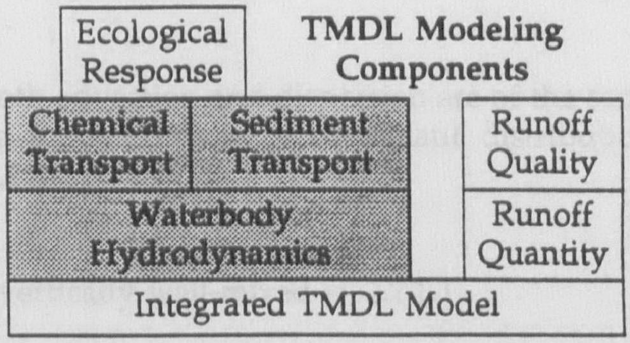
• Level 2: Continue steady state analysis, but consider seasonal variability.

• Level 3: Drive time-varying model with daily average flows; set maximum time step to smaller of chemical time step, WQS averaging time, or 1 day.

• Level 4: Set time step sufficiently small to allow dynamic simulation of hydraulics.

STEP 6. RECEIVING WATER MODEL SPATIAL REPRESENTATION

This step is concerned with determining the type and dimensionality of the receiving water simulation models - which include both hydrodynamic and water quality (chemical and/or sediment) components. Length of the spatial step will be influenced by prediction needs as well as model solution stability in dynamic simulation models.



1. Is a far-field receiving water model needed?
   a) TMDL considers far-field effects outside of the mixing zone or immediate area of wet-weather discharge; near-field effects may also be considered -> 2

   or

   b) TMDL considers near-field effects only, where only concentrations close to point of discharge into the receiving waterbody, or in a defined mixing zone are of concern. Situations in which this may apply include WQSs which address only mixing zone concentrations and narrative criteria for reactive substances which decay with sufficient rapidity such that essentially no transport occurs in the waterbody. The latter condition can be expressed in terms of characteristic mixing times as $t_T << t_A$ and $t_T << t_D$ -> 3

2. Far-field receiving water models. State of TMDL analysis is at:
   a) Level 0, not including tidal estuaries -> 2.1

   or

   b) Estuaries, and Level 1 or greater for other water bodies -> 2.2

2.1. Level 0 analysis. Use screening mass-balance approach in which the waterbody is treated as a zero-dimensional stirred reactor (for lakes/dispersive systems) or as a string of linked zero-dimensional segments (for rivers/advective systems).

2.2. Identify water body hydrodynamics and transport properties as dominantly 1-D advective (e.g, river), dispersive (e.g., lake), or 2,3-D advective-dispersive (e.g., estuary). This may be done using the dimensionless numbers and characteristic mixing times developed in Step 0.

   a) Dominantly advective systems, typically rivers. In general, $P_E >> 1$, $t_A <<$

t_{DT}, and $t_A << t_{DL}$ for the region of interest -> 2.2.1

or

b) Dominantly dispersive systems, typically lakes. Generally characterized by $P_E << 1$, $t_D << t_A$ -> 2.2.2

or

c) Complex systems in which both advection and dispersion are of the same general order of magnitude in determining contaminant distribution. Typically tidal estuaries -> 2.2.3

2.2.1. Dominantly 1-D advective system

a) No significant stratification, vertically well-mixed -> 2.2.1.1

or

b) Stratification is significant (rare in true rivers; may apply to short-residence reservoirs) -> 2.2.1.2

2.2.1.1. Unstratified, 1-D advective system

a) Perennial flow throughout region of impact -> 2.2.1.1.1

or

b) Intermittently flowing streams -> 2.2.1.1.2

2.2.1.1.1. 1-D hydrodynamic and water quality simulation appropriate beyond zone of lateral mixing. If WQS evaluation is required within mixing zone, or in regions of complex flow junctions, 2-D analysis may be required.

- Levels 1-2: Use input description of flows or steady state 1-D hydrologic simulation, coupled with 1-D simulation of quality. Consider uncertainty introduced by assumption of complete lateral mixing.
- Level 3: 1-D simulation of dynamic hydrology with 1-D water quality simulation (laterally averaged).
- Level 4: 1-D dynamic hydraulic simulation with 2-D water quality simulation.

2.2.1.1.2. Intermittent flow streams. Same analyses as 2.2.1.1.1 generally appropriate for periods of flow; however, special techniques may be required to model dryout phase, including ground water interactions.

2.2.1.2. Stratified systems with dominantly advective flow. Hydrodynamics are essentially 2-D (vertical) and water quality simulation may involve transport between upper and lower levels; 1-D analysis of the layer receiving discharge is likely sufficient for the scoping levels of analysis (Level 0-1). Beyond Level 1, use of a 1-D water quality analysis should be explicitly justified.

- Level 1: Use input description of flows or 1-D steady routing, with 1-D simulation of water quality in upper layer. Transport between layers should be characterized as net source, net sink, or no net interaction.

- Level 2: Input description of flows or use quasi-2-D steady routing (2 layers), with quasi-2-D water quality simulation (1-D representation of each layer with estimated fluxes between layers).
- Level 3: Quasi-2-D dynamic simulation of flow and quality.
- Level 4: Quasi-2-D dynamic simulation of flow, 3-D simulation of water quality.

2.2.2. Dominantly dispersive systems, typically lakes, which may show thermal or density stratification.

a) Stratification generally maintained for all or most of the summer -> 2.2.2.1

or

b) Unstratified, or weakly stratified and subject to frequent overturns -> 2.2.2.2

2.2.2.1 Stratified, dominantly dispersive system

a) Laterally well mixed, in terms of pollutant and averaging time of interest. Lateral concentration gradients are either not significant or dissipate in less time than appropriate for averaging time for numeric standards or waterbody response time for narrative criteria -> 2.2.2.1.1

or

b) Not laterally well mixed; (a) does not apply -> 2.2.2.1.2

2.2.2.1.1 Laterally well mixed, stratified, dominantly dispersive system. 1-D vertical analysis is sufficient to describe average distribution of pollutant.

- Levels 1-2: Input or simulate steady 1-D vertical description of flow, 1-D vertical simulation of water quality, is generally appropriate.
- Level 3: Use 1-D (vertical) dynamic simulation of hydrodynamics; 1-D or 2-D (longitudinal-vertical) simulation of water quality.
- Level 4: 2-D or 3-D simulation of hydrodynamics and water quality may be necessary to reproduce the temporal effects of vertical mixing at the most detailed level of analysis.

2.2.2.1.2 Not laterally well mixed in terms of averaging time or response of interest, stratified dominantly dispersive system. The distribution of pollutants and hydrodynamics of the system are inherently three-dimensional, but simpler approximations may be useful.

- Levels 1-2: Input or simulation steady 1-D vertical description of flow, use 1-D vertical simulation of quality on most-impacted segments of the waterbody. This incorporates conservative assumptions by neglecting diffusion out of most-impacted segments to regions of lower concentration.
- Level 3: Improve analysis by using 2-D (vertical) representation of water quality of most-impacted segments with specified concentration or flux boundary with remainder of waterbody.

• Level 4: 2-D or 3-D simulation of hydrodynamics with 3-D simulation of water quality in whole waterbody or portion of waterbody.

2.2.2.2 Unstratified dominantly dispersive system; pollutant discharges are mixed vertically within a time frame that is short relative to the needed analysis.

a) System is laterally well mixed, in terms of pollutant and averaging time of interest -> 2.2.2.2.1

b) System cannot be considered laterally well mixed -> 2.2.2.2.2

2.2.2.2.1 Laterally well mixed, unstratified, dispersive systems. Pollutants are well-mixed throughout the volume of the waterbody within the response time scale of interest. A zero-dimensional (completely stirred reactor) simulation will likely be appropriate for most applications.

2.2.2.2.2 Not laterally well mixed, unstratified, dispersive systems. Because the system is vertically mixed, the simulation of water quality is naturally 2-D (plan view).

• Levels 1-2: Input steady zero-dimensional description of flow (input-output). Consider use of zero-dimensional representation of most-impacted segments of the waterbody. This incorporates conservative assumptions by neglecting diffusion out of most-impacted segments to regions of lower concentration.

• Level 3: Improve analysis by using 2-D (plan view) water quality representation of most-impacted segments with specified concentration or flux boundary with remainder of waterbody.

• Level 4: 2-D simulation of hydrodynamics; detailed 2-D or 3-D simulation of waterbody or portion of waterbody.

2.2.3. Advective-dispersive systems, including most estuaries. For estuaries, the concept of Levels of Analysis 0-4 presented here is similar to Levels I through IV of estuarine simulation models described by Ambrose et al. (1990).

a) Laterally well mixed in direction transverse to main axis of flow, in terms of pollutant and averaging time of interest. Concentration gradients are not significant across axis of flow. This will generally mean that $t_{DT} < t_A$ -> 2.2.3.1

or

b) Not laterally well mixed -> 2.2.3.2

2.2.3.1 Laterally well mixed, advective-dispersive system.

a) Strongly stratified system. For estuaries, Fischer et al. (1979) suggest that the transition from vertically well-mixed to stratified estuaries generally occurs in the range of $0.08 < R < 0.8$, where R is the estuarine Richardson number (see Step 0). For $R > 0.8$, the system can be treated as strongly stratified. Within the transition range, the determination should be based on observation -> 2.2.3.1.1

or

b) Vertically well-mixed system -> 2.2.3.1.2

2.2.3.1.1 Laterally well mixed, stratified, advective-dispersive system. The hydrodynamics of the system are essentially 3-D, while pollutant transport can be described as 2-D (vertical); however, simpler analysis may be sufficient for many problems.

- Levels 1-2: 1-D analysis of the upper layer is usually sufficient, although the effects of ignoring interactions with lower layer should be considered. Note that the lower layer, adjacent to the sediments, may be a net source of certain pollutants (such as BOD).
- Level 3: 2-D (vertical) simulation of circulation could be used to improve resolution.
- Level 4: Detailed representation of transport processes likely needs a 3-D hydrodynamic simulation.

2.2.3.1.2 Laterally well mixed, unstratified, advective-dispersive system. Hydrodynamics can be described in two dimensions, while contaminant distribution varies significantly in one direction only.

- Levels 1-3: 1-D simulation of hydrodynamics suggested as adequate for analysis purposes, except insofar as the laterally well mixed assumption is not exactly true.
- Level 4: 2-D (vertical) or 3-D simulation of hydrodynamics can capture more of the full spatial and temporal variability of transport processes.

2.2.3.2 Not laterally well mixed advective- dispersive system.

a) Strongly stratified system. For estuaries, Fischer et al. (1979) suggest that the transition from vertically well-mixed to stratified estuaries generally occurs in the range of $0.08 < R < 0.8$, where R is the estuarine Richardson number (see Step 0). For $R > 0.8$, the system can be treated as strongly stratified. Within the transition range, the determination should be based on observation -> 2.2.3.2.1

or

b) Vertically well-mixed system -> 2.2.3.2.2

2.2.3.2.1 Stratified, not laterally well mixed, advective-dispersive system. Both hydrodynamics and pollutant transport are inherently three-dimensional.

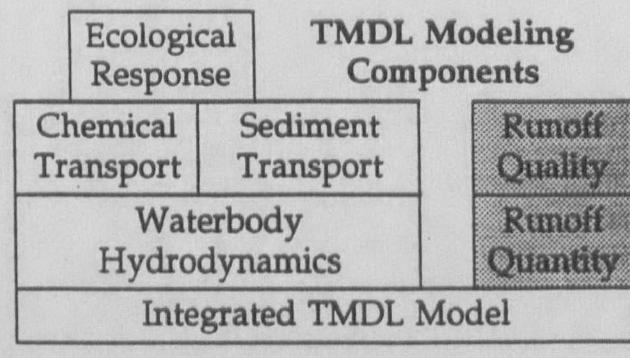
- Levels 1-2: Obtain approximate results with a 1-D analysis (hydraulic and transport) of the receiving layer. The fact that lateral variability and potential role of input from lower layer is ignored will result in the need to develop an estimate of potential errors in inherent in the approximation and either assign a large MOS or move to a more sophisticated analysis.

- Levels 3-4: A two-dimensional approximation or a full three-dimensional analysis of the estuary is needed. See EPA's estuarine WLA guidance (Ambrose et al., 1990) for further details.

2.2.3.2.2 Unstratified, not laterally well mixed advective-dispersive system. 2-D analysis generally required.

3. Near-Field modeling only required for TMDL. This implies that the important sources of pollutants resulting in water quality limited waters are separable, and do not interact with one another in any significant way. The TMDL thus reduces to a set of wasteload allocation problems.

STEP 7. WET-WEATHER LOADING MODELS - GENERAL



Choice of wet-weather loading models will first reflect the physical characteristics of the system, such as land use and pollutant types. For instance, urban and non-urban wet-weather loading problems are typically simulated with different models. However, certain general considerations apply to all types of wet-weather loading models.

In the context of TMDL development, the role of wet-weather source modeling and monitoring is to provide input to a receiving water quality analysis of waterbody impairment - whether in the form of an estimate of the total load delivered as a driver for a receiving water quality model or analysis. We thus need "end of the pipe" type of measurements. From this narrow viewpoint, we are not necessarily interested in the details of land surface processes, except as those details help improve estimates of loading. In addition, the spatial and temporal resolution of our loading predictions does not need to be any finer than is required to drive the receiving water analysis.

Given these caveats, it is clear that wet-weather load modeling for TMDLs will often be quite different from state-of-the-art descriptive simulation models. The interplay between monitoring and modeling is particularly important here: in many cases, the best course of action may involve site specific determination of pollutant concentrations via monitoring coupled with an event-total simulation of runoff volume, thereby yielding an estimate of the time series of loads. Therefore, we first need to inquire whether any wet-weather load model is needed, and, if so, whether quantity and quality, or just quantity should be simulated.

1. Are any wet-weather loading models needed? In some cases, no modeling analysis at all may be needed for the wet-weather sources. This will generally be when characterization through monitoring is adequate to answer the questions at hand.

   a) Wet-weather loads are relatively constant, well-characterized by monitoring, and not amenable to or not considered for control in this phase -> No wet-weather load modeling; rely on monitoring data.

   or

   b) Scoping analysis at Level 0, with adequate monitoring data base for preliminary analysis -> No wet-weather load modeling for the preliminary analysis.

   or

   c) Other situations, requiring wet-weather load modeling -> 2

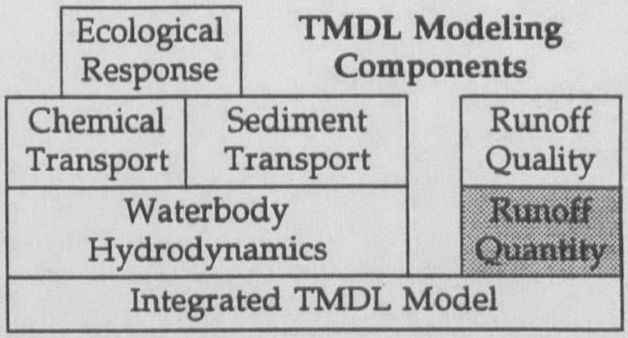
2. Wet-weather load modeling needed. It is assumed that this will require some degree of quantity (runoff) modeling. The quality component may or may not be addressed by modeling.

   a) Receiving water model requires loading on annual, average or event-mean basis (not intra-event); the source is susceptible to accurate monitoring (e.g., CSOs) and event-mean concentrations are well characterized by monitoring; and extrapolation beyond the monitoring record is not required at this stage -> User should consider representing wet-weather loading via analysis/simulation of flows (quantity) coupled with estimates of event mean concentration derived from monitoring. If so, only quantity modeling will be needed for wet-weather loads at this level, complete Step 7A and 7B

   or

   b) Other situations, in which modeling of wet-weather loads is needed -> Wet-weather quantity and quality modeling needed, complete Steps 7A, 7B, 8A and 8B

STEP 7A. WET-WEATHER QUANTITY MODELING - TIME STEP



The temporal resolution for modeling of wet-weather quantity (flows) will be largely determined by the needs of the receiving water analysis.

1. Establish the appropriate level of temporal detail for hydrologic modeling of wet-weather load contributions.

a) Chemical time step for receiving water established in Step 3 is steady, or annual or seasonal average -> 2

or

b) Chemical time step for receiving water is continuous, event or sub-event -> 3

2. Steady, or annual or seasonal average loadings required for receiving water model. Establish qualitative indication of degree of variability with time within the year or season.

a) Wet-weather loading is relatively predictable based only on annual or seasonal rainfall totals. This is likely the case when concentrations are relatively predictable, runoff volumes can be estimated from rainfall, and event mean concentrations are not strongly correlated with flow. -> 2.1

or

b) Wet-weather loading over the year or season is dominated by a few major events, as is often the case for combined sewer overflows, and day to day hydrology must be simulated to obtain a reasonable estimate of annual totals - 2.2

2.1. Use of annual or seasonal runoff totals is adequate: detailed simulation of wet-weather quantity is not required.

2.2. Average wet-weathher flows and loading need to be determined for event-dominated, episodic runoff.

a) Analysis at levels 0-2 -> 2.2.1

or

b) Analysis at Levels 3-4 -> 2.2.2

2.2.1 Level 0-2 analysis for event-dominated, episodic runoff. If, at these Levels, the receiving water analysis does not involve continuous simulation it is usually not appropriate to undertake continuous simulation of the wet-weather hydrology either. Therefore, use average annual or seasonal volumes, recognizing that this will introduce uncertainty by not accounting for the role of antecedent moisture conditions. Year to year variability in precipitation can be represented by frequency distribution or sensitivity analysis.

2.2.2 Level 3-4 analysis for event-dominated, episodic runoff. More sophisticated analysis requires a more exact analysis of the frequency distribution of the annual or seasonal average totals. This may be obtained by forming the time series of averages from a continuous series of events. Wet-weather quantity should thus be simulated at the event scale.

3. Continuous, event or sub-event receiving water chemical time step required.
   a) Receiving water model requires reasonable maximum load or event mean loads -> 3.1

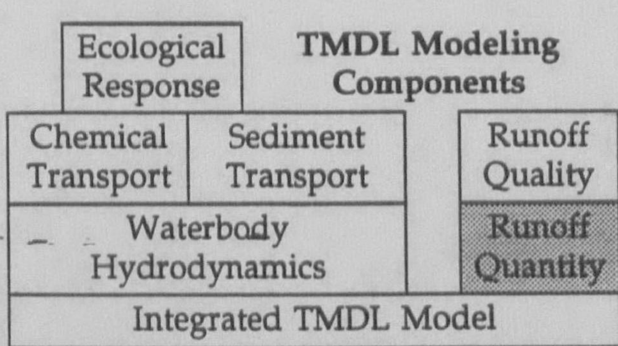
   or

   b) Receiving water model requires detailed pollutograph at the sub-event scale (hourly to subhourly) - 3.2

3.1 Event based model required; determination of reasonable maximum may require continuous simulation of events to establish frequency distribution with antecedent conditions taken into account.

3.2 Continuous model required with hourly to sub-hourly chemical time step as required by receiving water model.

STEP 7B. WET-WEATHER QUANTITY MODELING - SPATIAL DETAIL/ ROUTING



Virtually any wet-weather loading model is capable of reproducing spatial variability of sources to some degree. For instance, the simplest model may be run several times for each land use area, thus building up a representation of spatial variability. Choice of the level of wet-weather model spatial detail in a modeling application for TMDL will reflect factors including 1) level of the analysis; 2) availability of data and financial and manpower resources; 3) degree of heterogeneity in watershed land use patters; 4) extent to which significant wet-weather sources are focused (e.g., entering through CSO outfalls or drainage swales), or distributed; and 5) spatial resolution required by the receiving water model. However, because most wet-weather loading models can be used to express spatial variability, these factors are not generally categorizable determinants for model selection (although they may be significant in a site-specific context). Therefore, this section concentrates on whether or not a space-time history of loading is required, versus simply calculating event (or seasonal) totals. Essentially, this reflects whether any hydrologic *routing* of overland, storm sewer, or first order streamflow needs to be performed to provide input to a receiving water model. Capabilities of wet-weather loading models vary widely in this regard.

1. Determine whether receiving water model demands detailed time history or pollutograph.

   a) Analysis is at Level 0 (simple scoping) -> 2

   or

   b) Analysis is at Level 1 or greater, and receiving water model does not require intra-event time step -> 3

   or

   c) Receiving water model chemical time step requires loading model time step that is intra-event (hourly, sub-hourly) -> 4

2. Level 0 scoping or screening analysis. Only average loading estimates will be used. No hydraulic routing is required.

3. Level 1 or greater analysis with required loading time step at daily or event scale or greater.

   a) Level 1,2 analysis -> 3.1

   b) Level 3,4 analysis -> 3.2

3.1 Level 1 or 2 analysis with loading time step at daily or event scale or greater. Time dependent routing is not necessary (although it may be used). Runoff coefficient and curve number methods for hydraulic simulation may be acceptable.

3.2 Level 3 or 4 analysis with loading time step at daily or event scale or greater. Determine need for routing:

   a) Load delivered during event depends on peak or timing of flow within event -> 4

   or

   b) Daily loads are required, <u>and</u> events typically last longer than one day, <u>and</u> significant variability in concentration occurs with time within the event -> 4

or

c) Neither (a) nor (b) apply -> 3.2.1

3.2.1 Daily/event load calculations without intra-event timing. Water balance (input-output) and hydrologic routing methods are acceptable (although more sophisticated routing techniques may also be used.)

4. Intra-event loading time history required by receiving water model, or intra-event time history required to determine event loading. Routing of wet-weather flows is required.

a) Hourly time step is sufficient and detailed replication of flood waves or other momentum changes is not required -> 4.1

or

b) Flood wave propagation is thought to be significant to analysis of flow, but flow is gradually varying and effects of surcharging and backwaters (in pipe flow) or flood wave attenuation (in overland and channel flow) are of minor importance -> 4.2

or

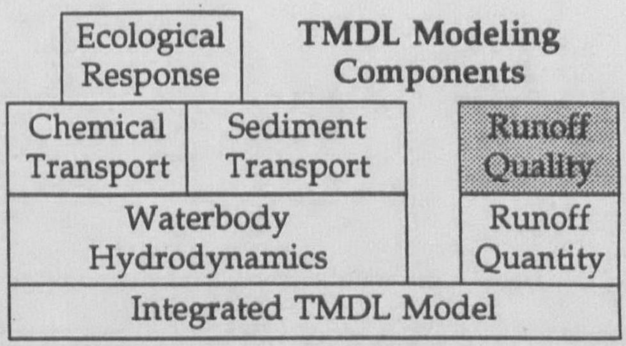
c) Effects of surcharging and backwaters (in pipe flow) or flood wave attenuation (for overland and channel flow) or other internal momentum effects must be considered -> 4.3

4.1 Intra-event loading without momentum effects. Can use water balance or hydrologic routing methods. (More sophisticated routing methods, such as kinematic and dynamic wave, are also acceptable.)

4.2 Intra-event loading with gradually varying flow. Can use kinematic wave or dynamic wave routing solutions.

4.3 Intra-event loading with strong internal momentum effects. Desirable to use a dynamic wave routing solution to provide accurate description.

STEP 8A. WET-WEATHER QUALITY MODELING - TEMPORAL RESOLUTION



This step is designed to aid in determination of the time detail requirements of the quality component of a wet-weather loading model - if indeed the quality aspects are simulated.As wit the chemical time step in receiving water models, it is framed in terms of an applicable maximum time step: the user is suggested to choose a model which can implement a temporal resolution of this scale or finer.

As noted above, there will be various instances in which attempting to simulate wet-weather load *quality* will not result in much, if any, improvement in predictions over use of concentrations obtained from monitoring data

1. Determine need for wet-weather quality simulation.
   a) Sufficient monitoring database exists so that it is not necessary to simulate wet-weather quality (i.e., use wet-weather quantity information and site-specific relation to quality), <u>and</u> it is not desired to predict land use changes to situations for which monitoring is not available -> 2

   or

   b) Wet-weather quality simulation is required -> 3

2. No simulation of wet-weather quality is required. Measured quality data are combined with wet-weather flows for the analysis.

3. Simulation of wet-weather quality is required. Primary determinant of wet-weather time step is the Level of analysis.
   a) Level 0 (screening) analysis -> 3.1

   or

   b) Level 1-4 analysis -> 3.2

3.1 Level 0 (screening) modeling of wet-weather. Apply annual loading function type models. An approximate partitioning of the total annual load within the year may be obtained by assessing the frequency of runoff producing precipitation events above a certain minimum transport capacity.

3.2 Level 1-4 simulation of wet-weather quality.
   a) Level 1,2 analysis -> 3.2.1

   or

b) Level 3,4 analysis -> 3.2.2

3.2.1 Level 1, 2 simulation of wet-weather quality. Loading required by receiving water analysis is:

a) Steady state, annual or seasonal average loading -> 3.2.1.1

or

b) Event average or reasonable maximum event loading -> 3.2.1.2

3.2.1.1 Level 1,2 simulation of annual or seasonal average wet-weather loading. Detailed resolution of timing of wet-weather loading not required; simple models with annual event time scale are adequate; finer resolution models may also be employed.

3.2.1.2 Level 1,2 simulation at event level. Wet-weather model should be able to simulate typical and extreme events, but not necessarily the detailed time history of events. Maximum simulation time step is daily or event. Event-based or continuous wet-weather quality models may be used.

3.2.2 Level 3,4 simulation of wet-weather runoff quality, in which a more accurate time series of loads is required. Continuous or probabilistic simulation should be employed. The time step limit established for simulating loading to receiving waterbody is:

a) Daily or event -> 3.2.2.1

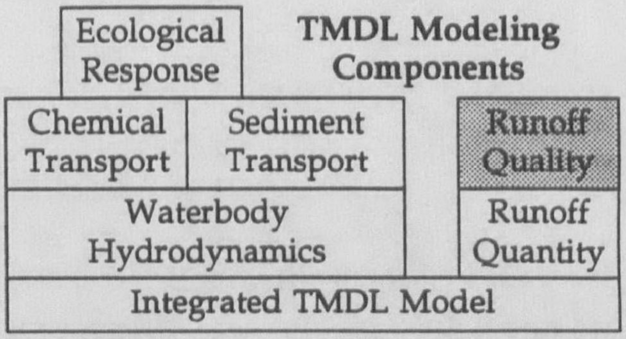
or

b) Sub-event, hourly or sub-hourly -> 3.2.2.2

3.2.2.1 Continuous time series of wet-weather event loads is needed, but not intra-event timing of loads. Use maximum wet-weather quality simulation time step at the daily or event level.

3.2.2.2 Continuous time series of wet-weather loads is needed with intra-event loading history. User should evaluate whether to use a daily-event time scale model (with flow-weighted partitioning to the desired loading time scale) or a sub-event time scale model.

STEP 8B. WET-WEATHER QUALITY MODELING - SPATIAL DETAIL



As with quantity modeling, most wet-weather load models can represent spatial heterogeneity of sources by repeated application to subareas, if more sophisticated methods are not built in. The question addressed here is whether the model should address pollutant routing: i.e., the space-time distribution of pollutant loads, rather than just the loading history predicted by transport by advective flows. This may be important when the rate of transport of the pollutant differs significantly from the rate of flow of water, or where reactions are taking place within the wet-weather flow.

1. Evaluate based on wet-weather quality simulation temporal resolution.
   a) Wet-weather quality temporal resolution is the event scale or greater, <u>and</u> it is not necessary to simulate the effects of in-transit treatment -> 2

   or

   b) Wet-weather quality temporal resolution is less than event length, <u>or</u> detailed analysis required of in-transit treatment effects ->3

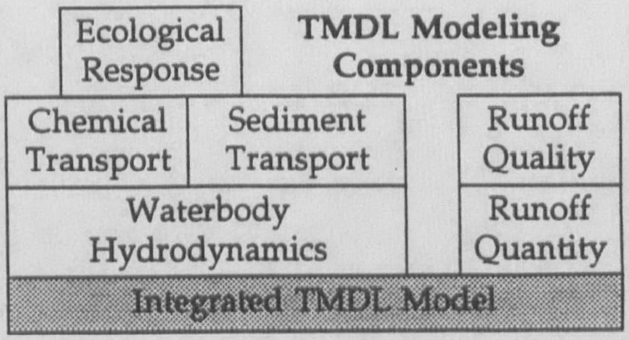
2. Detailed quality routing capabilities are not needed from the wet-weather load model.

3. Intra-event wet-weather quality simulation desired. A sensitivity analysis may be performed to determine if it makes any significant difference to predictions whether the exact arrival timing of component parts of the event load, beyond that predicted by simple advection, is considered. This should usually only be necessary at Levels 3-4 of analysis.
   a) Sensitivity analysis indicates non-advective intra-event timing of components of load should be considered: Use a model with wet-weather quality routing capabilities.

   or

   b) Sensitivity analysis indicates non-advective intra-event timing of components can be ignored: Wet-weather quality routing capabilities are not required in the model (as distinct from hydraulic routing capabilities).

STEP 9. ASSEMBLE AND REVIEW MODEL IDENTIFICATION CRITERIA



Step 0 provides the TMDL developer with a table of important physical, regulatory and functional characteristics of the system to be modeled. Steps 1 through 8 then yield suggestions regarding the characteristics of models which may be appropriate for TMDL analysis of the system. These can be assembled to provide the basis for model identification. (Note that Step 9 is not a decision tree; the following items should be addressed in sequence)

9.1. Form Table of Model Identification Criteria. The user should compile the results of all these steps in a tabular form (see, for example, Table 4-3). This table will summarize the types of models which might be employed for TMDL analysis, and some of their important functional requirements.

Table 4-3. Technical Criteria Summary Form for Model Selection

Model Component	Is a model needed?	Physical Domain	Pollutant or Impacts Addressed	Time Scale of Simulation	Flow Routing Capabilities	Quality Routing & Transformations
Ecological Response						
Receiving Water Quantity						
Receiving Water Quality						
Sediment						
NPS Quantity						
NPS Quality						

9.2. Review Model Identification Criteria Table. Once the table is assembled, it should be subjected to a thorough review. It is important to remember that Steps 1 through 7 provide only generic recommendations. These have been formulated to cover a wide range of circumstances; however, they may not be appropriate for the site-specific conditions of the problem under study. Two important questions that should be addressed in the review are:

- Do the model identification criteria make sense in terms of the known conditions of the waterbody and pollutant under study? To the extent that they do not, the decisic process should be revisited and alternative specifications examined and justified.
- Do the model identification criteria make sense in terms of the proposed level of effort and degree of accuracy required for the evaluation? If the analysis is thought to be too simple and uncertain, this may indicate that the decision trees should be revisited with the choice of the next higher Level of analysis. Where the analysis is thought to be too complex or expensive, it may sometimes be possible to move to a lower, more approximate Level of analysis, with a correspondingly larger MOS. However, in this case the user should also examine whether it is feasible to perform the TMDL at all within available resource limitations.

9.3. Analysis of Candidate Models. After the model identification criteria have been revised, a list of candidate models can be formed. The different model components (e.g., wet-weather loading quality and receiving water quality) may be addressed by separate models, or, if available, within single modeling packages. The candidate models should possess technical characteristics which match those identified in the model identification criteria table. Tabular guides to many Federal agency supported simulation models were provided in Chapter 3. Additional models available to the user are readily added to the list.

9.4. Model Selection. The previous step should identify available simulation models, or analysis methods, which meet the technical and regulatory needs of the site TMDL analysis (see section 4.2), with some recognition of user functional and operational needs, via choice of an appropriate Level of analysis. Where multiple candidate models are identified, the final choice of simulation models can further reflect user functional and operational needs, including cost and time requirements for the implementation, ease of use, and familiarity of staff with the models.

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Chapter V. Monitoring Plans and Impact Assessment for TMDLs

Purpose: This chapter addresses potential needs for monitoring to support modeling and to track potential benefits from implementing new wastewater control and/or watershed management practices. It introduces necessary considerations for designing a monitoring program to provided results adequate for addressing the program's general monitoring goal and specific monitoring objective. Chapter 6 introduces sampling techniques, primarily focusing on the general mechanics and potential problems with sampling. It also provides guidance on some techniques available for summarizing and assessing collected data. Particular attention in both chapters is paid to special considerations for sampling related to wet weather TMDL modeling efforts.

The chapter focuses on considerations important for developing monitoring plans. It introduces existing documents that contain information useful in designing monitoring plans, especially those applicable to wet-weather TMDL studies. Distinctions between monitoring goals to support modeling efforts, monitoring goals to track benefits of control, and management efforts are discussed, as are considerations for defining specific monitoring objectives. Also, special considerations are introduced regarding impacts and associated monitoring requirements for various types of waterbodies and land-uses.

5.1. Interaction of Monitoring and Modeling for TMDLs

An appropriate modeling strategy should be designed to provide answers appropriate to the TMDL process, within the constraints of the physical characteristics of the waterbody, pollutants and impacts under study, available resources, and other functional and operational needs of the TMDL developer. In most cases it will not be possible to specify all the characteristics of the system at the beginning of the process, when data may be limited. (This is, indeed, why a model selection strategy is required). Instead, TMDL developers will need to refine the goals and requirements of modeling as they gain additional knowledge through system characterization and monitoring. Similarly, TMDL developers will likely need to adjust monitoring plans as the characterization and assessment effort proceeds. The modeling strategy will help determine the calibration data requirements from the monitoring program, and model results may suggest critical areas where monitoring will be most effective.

These observations indicate that the development of a final modeling strategy (and, indeed, determination of the need to employ sophisticated modeling) is an iterative process, which will involve close interplay between modeling and monitoring. For instance, a TMDL developer might use a simple screening model for initial scoping of the problem. This can aid in designing the monitoring plan, which in turn can lead to 1) determination of the need for more sophisticated modeling, and 2) the formulation of a targeted modeling strategy for the completion of the TMDL assessment.

5.2 Existing Guidance for Information on Monitoring Designs and Sampling Procedures

A number of existing guidance documents include information valuable to the design of monitoring programs and on selection and use of appropriate sampling procedures. Table 5-1 presents an list of nearly 25 documents produced or sponsored by EPA. Table 5-2 presents a selection of other guidance documents available from non-EPA sources that also contain much valuable information useful to the design and implementation of monitoring programs. While these lists are extensive, they are not comprehensive lists of available guidance. Many of the these documents were used to help develop some of the discussion included in this chapter, and they should be consulted where users of this document require additional information on the topics discussed.

Table 5-1. Existing EPA guidance on monitoring designs and sampling procedures

Guidance for Water Quality-based Decisions: The TMDL Process (U.S. EPA, 1991a) - Introduces the steps of the TMDL process, including general requirements for modeling and monitoring.

Water Quality Standards Handbook (U.S. EPA, 1983a) - Provides general guidance on water body surveys and assessments for conducting use attainability analyses and guidelines for developing site-specific water quality criteria.

Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses [streams and rivers] (U.S. EPA, 1983b) - Detailed guidance on conducting physical, chemical, and biological evaluations of stream and rivers, and appropriate methods for summarizing and assessing collected data.

Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses. Volume II: Estuarine Systems (U.S. EPA, 1984a) - Introduction to the physical dynamics of estuarine systems and the characteristics of plant and animal communities; detailed guidance on methods to characterize physical, chemical, and biological conditions, and on approaches to synthesize and interpret collected data.

Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses. Volume III: Lake Systems (U.S. EPA, 1984b) - Review of physical, chemical, and biological characteristics of lakes and reservoirs; detailed introduction to sampling methods and methods to summarize and assess collected data.

Technical Guidance Manual for Performing Waste Load Allocations. Book II. Stream and Rivers. Chapter 1, Biochemical Oxygen Demand/Dissolve Oxygen. (Driscoll et al., 1983) - Introduction to the assessment of BOD and DO in flowing water systems; minimum guidance on sampling and monitoring.

Technical Guidance Manual for Performing Waste Load Allocations. Book II. Stream and Rivers. Chapter 2, Streams and Rivers: Nutrient/Eutrophication Impacts (U.S. EPA, 1983c) - Introduction to evaluations of nutrient and eutrophication problems in flowing water systems; emphasis on assessment of phytoplankton appropriate only for large, slowly flowing rivers; guidance on minimal sampling requirements.

Technical Guidance Manual for Performing Waste Load Allocations. Book III. Estuaries. Part 1: Application of Estuarine Waste Load Allocation Models (Martin et al., 1990) - Provides monitoring protocols for calibration and validation of estuarine models, including types of data, frequency of collection, spatial coverage, and quality assurance.

Technical Guidance Manual for Performing Waste Load Allocations. Book IV. Lakes and Impoundments. Chapter 3: Toxic Substances Impact (Hydroqual, Inc., 1986) - Introduction to problem of toxic substances in lakes and reservoirs; presents data requirements for models, including minimum sampling frequencies, problems of time scales, needs for analysis of dissolved or total concentrations, sample handling, preservation, and documentation; does not specifically address WLA or TMDL processes.

Table 5-1. (continued)

Handbook: Stream Sampling for Waste Load Allocations Applications (Mills et al., 1986) - Provides guidance for designing stream surveys to support modeling applications for WLAs; includes potential sampling requirements for meteorological data, water quality data, times, frequencies, etc.

Guidance for State Water Monitoring and Wasteload Allocation Programs (U.S. EPA, 1985) - Provides general introduction on determining needs for monitoring program designs, monitoring for water quality based controls, monitoring for compliance and enforcement, water quality assessments, quality assurance, data reporting, EPA's monitoring strategy, and monitoring checklists.

Technical Support Document for Water Quality-based Toxics Control (U.S. EPA, 1991b) - Introduces needs for biological criteria, biological assessments, and biological surveys; monitoring needs in establishing permit limits.

NPDES Stormwater Sampling Guidance Document (U.S. EPA, 1992a) - Focuses on technical aspects of stormwater sampling, particularly to meet requirements for NPDES permits; includes extensive discussion of fundamentals of sampling for flow measurements and water quality evaluations.

Watershed Monitoring and Reporting for Section 319 National Monitoring Program Projects (U.S. EPA, 1991c) - Introduces water quality monitoring and assessment approaches under the 319 Program, including the design of monitoring programs; sampling station location; physical, chemical, and biological monitoring parameters; data analysis; and reporting requirements.

Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters (U.S. EPA, 1993) -Includes an extensive chapter on monitoring and tracking techniques to accompany management measures.

Developing Nonpoint Source Load Allocations for TMDLs - A Quick Reference Guide (Tetra Tech, Inc., 1992) - Includes a brief discussions on the role of water quality monitoring and follow-up monitoring; highlights additional sources for NPS monitoring guidance.

Biological Field and Laboratory Methods for Measuring the Quality of Surface Waters and Effluents (Weber, 1973) -The classic EPA guide to sampling methods and data analytical approaches for plankton, periphyton, macrophytes, macroinvertebrates, fish, and toxicity bioassays.

Macroinvertebrate Field and Laboratory Methods for Evaluating the Biological Integrity of Surface Waters (Klemm et al., 1990) - Provides thorough guidance on monitoring designs, sampling procedures, and analysis approaches for macroinvertebrate communities inhabiting aquatic environments.

Rapid bioassessment Protocols for Use in Streams and Rivers - Benthic Macroinvertebrates and Fish (Plafkin et al., 1989) - Presents five specific protocols, including detailed guidance on sampling and data analysis methods, for assessing biological conditions in streams and rivers.

Microbial Methods for Monitoring the Environment - Water and Wastes (Bordner and Winter, 1978) - Presents detailed guidance on sample collection and analytical methods for microbial samples.

Table 5-1. (continued)

Monitoring Lake and Reservoir Restoration (Wedepohl et al., 1990) - Presents example monitoring designs and sampling methods for lake, lake tributary, and watershed monitoring programs under Clean Lakes Program projects.

Monitoring Guidelines to Evaluate Effects of Forestry Activities on Streams in the Pacific Northwest and Alaska (MacDonald et al., 1991) - Provides detailed guidance on designing monitoring programs and on assessing monitored physical, chemical, and biological variables; includes a discussion on using the TMDL process within these assessments.

Ecological Assessments of Hazardous Waste-Sites: A Field and Laboratory Reference Document (Warren-Hicks et al., 1989) - Introduces uses and development of ecological effects endpoints, assessment strategies and approaches, field sampling designs, toxicity testing, biomarkers, and field assessment procedures for aquatic and other environments.

Table 5-2. Additional guidance on monitoring designs and sampling procedures

Design of Water Quality Monitoring Systems (Ward et al., 1990) - Excellent source of guidance to monitoring plan design, includes the role of statistics, quantifying information expectations, data analysis, and network design.

Sampling Design and Statistical Methods for Environmental Biologists (Green, 1979) - Includes "Ten Principals" of design, "optimal" impact study designs, selection of monitoring variables, and presentation of results.

Field Methods and Statistical Analyses for Monitoring Small Salmonid Streams (Armor et al., 1983) - USDI Fish and Wildlife Service guidance on assessing land use impacts, selecting and applying physical, chemical, and biological assessment methods, and statistical analyses of aquatic environmental monitoring data.

Methods for the Assessment and Prediction of Mineral Mining Impacts on Aquatic Communities: A Review and Analysis (Mason, 1978) - Proceedings from a USDI Fish and Wildlife Service workshop that produced detailed guidance on designing impact studies, analyzing collected data, and sampling and assessing bacteria, algae, zooplankton, benthic macroinvertebrates, fish, and other aquatic-related taxa.

Methods for Evaluating Riparian Habitats with Applications to Management (Platts et al., 1987) - USDA Forest Service document that provides extensive guidance on collecting data and assessing riparian areas, emphasizing Western streams.

Methods for Evaluating Stream, Riparian, and Biotic Conditions (Platts et al., 1983) - USDA Forest Service guidance on selecting study sites, developing sampling designs, using sampling transects, and assessing riparian, fish population, and benthic macroinvertebrate conditions in stream studies.

Table 5-2. (continued)

Estimating Total Fish Abundance and Total Habitat Area in Small Streams Based on Visual Estimation Methods (Hankin and Reeves, 1988) - Benchmark paper that provides habitat characterization methods that provide a basis for many of the more easily used approaches for monitoring stream habitats.

Fisheries Habitat Surveys Handbook (USDA Forest Service, 1989a) - One of several handbooks developed by the USDA Forest Service's Regions including intensive procedures to standardize survey and data collection procedures used to assess stream and lake habitats on Forest Service lands.

A Basin-Wide Stream Inventory Process Using Habitat Type Classifications for Resident Salmonids (USDA Forest Service, 1989b) - An example of broadly used alternative procedures used by individual National Forests, which are based on the Hankin and Reeves (1988) method, to characterize and assess stream habitat.

5.3 Defining General Monitoring Program Goals

Most monitoring programs *collect data* to *generate information*. The distinction between *data* and *information* is important. Put simply, data are measurements or observations made on a system (e.g., a wastewater stream or an ecological system), while information is knowledge with which one can make decisions. Poorly thought-out or incomplete monitoring programs often produce "data rich" but "information poor" results (Ward et al., 1990). For example, samples may have been taken at locations selected solely because of convenient access, at times when a staff member was near the area or when the weather was "fair." Such programs often proceed with only vague notions of what data might be needed and without a good idea of how to analyze the data that are obtained. When this approach continues over extended periods the results are often data rich and information poor, because monitoring did not proceed with clear goals, objectives, or designs.

Table 5-3 shows a series of useful steps to consider when developing monitoring programs. In the first step, TMDL developers should identify the goal(s) of monitoring. In the context of this guidance, monitoring generally would be conducted to address one or both of two general goals. The first of these goals is to support identification model calibration, and validation. Here the monitoring program can be strictly a data-collection effort, and the program ends when its collected data are transferred to the modelers. This is an example of monitoring that has a "data end-point goal" (Table 5-3). Often, however, monitoring aims to achieve an "information goal" (Table 5-3). Thus, the second common monitoring goal is to track and assess (1) potential benefits from implementing new treatment, control, and management strategies, or (2) potential impacts from new discharges. This goal requires analysts to extrapolate or transform monitoring data into *information* of known reliability about the characteristics of the system. In other words, although one intermediate end point of monitoring design and sampling collection is a set of data, most users are interested in the information conveyed by the data and not the data *per se* (except as noted above). They want answers to such questions as:

- How do the ambient conditions of this water body compare to those of other, similar water bodies in the region?
- How has the new control program improved conditions for aquatic life in the receiving water?
- How much precipitation-produced runoff will result in NPS runoff and discharges to the receiving waters?
- How does the discharge volume change when the runoff is due to snowmelt?
- What are the average ambient conditions in the river?
- How does a waste water discharge of volume "X" alter ambient water quality conditions in the receiving water?

To address such questions and to address the kinds of information goals shown in the above examples requires that monitoring designs also include plans for data management and analyses to derive available information from the collected data.

Table 5-3. General Steps for Monitoring Program Development

1. **Identify Monitoring Goal(s) (Purpose)**

   1.1. Data end-point goal

   1.1.1. Gather data for system model identification and selection
   1.1.2. Gather data for system model calibration and verification

   1.2. Information goal

   1.2.1. Establish characteristics of system variables (e.g., for water quality compliance monitoring)
   1.2.2. Assess ecological impacts to biological populations and communities inhabiting receiving waters

2. **Define Monitoring Objective(s)**

   2.1. Characterize effluent

   2.2 Define system component(s) or parameter(s) (i.e., system variables or system locations) for which monitoring data are needed, the kind(s) of data needed, and minimum accuracy and precision of data required to meet the information needs under each monitoring objective(s)

   2.3. Define the limits of variability in system conditions (e.g., time of day, season, section of runoff hydrograph) to be characterized under each monitoring objective

   2.4 Define Data Quality Objectives

   2.5 Define how data will be used, what comparisons made, what procedures used

3. **Design Sampling Program**

   3.1. Data end-point objectives

   3.1.1. Schedule sample and data collection periods, frequencies, and locations to correspond to requirements established under each monitoring objective

   3.1.2. Identify and select field sampling, data collection techniques, and laboratory analytical techniques that will provide data of sufficient precision and accuracy to achieve each monitoring objective

Table 5-3. (continued)

3.2. Information objective

3.2.1. Identify and select statistical method(s) capable of providing information required to meet each monitoring objective

3.2.2. Define minimum data requirements (data quality and quantity) that allow each selected statistical method(s) to be used to produce the information required to meet each monitoring objective

3.2.3. Schedule sample and data collection periods, frequencies, and locations to correspond to requirements established under each monitoring objective

3.2.4. Identify and select field sampling and data collection techniques and laboratory analytical techniques that will provide data of sufficient precision and accuracy to achieve each monitoring objective

4. Monitoring Program Implementation (see Table 5-4)

4.1. Sample and field data collection

4.2. Laboratory analysis

4.3. Data management

4.4. Data analysis

4.5. Reporting

4.6. Information use

Table 5-4. Checklist of Considerations for Documenting Monitoring Program Designs and Implementation (expanded from Ward et al., 1990)

Sample and Field Data Collection

Pre-Sampling Preparations

- __ Personnel selection and responsibility identification
- __ Personnel safety training and health care verification (first aid training, CPR, wet-weather training, safety guides, vehicle safety, vaccinations current, etc.)
- __ Site access prepared and legal consents obtained
- __ Scientific sampling or collecting permits needed
- __ Formats for field sampling logs and diaries
- __ Equipment availability, acquisition, and maintenance
- __ Sample collection schedule (random? regular? same-time-of-day?)
- __ Preparation of pre-sampling checklist

Sampling procedures

- __ Procedures documentation
- __ Staff qualifications and training
- __ Pre-sampling preparation (e.g., purging sample lines, instrument calibration)
- __ Sampling protocols
- __ Quality control procedures (equipment checks, replicates, splits, etc.)
- __ Required sample containers
- __ Sample numbers and labeling
- __ Sample preservation (e.g, "on ice" or chemical preservative)
- __ Sample transport (delivery to laboratory needed?)
- __ Sample storage requirements met
- __ Sample tracking and chain-of-custody procedures
- __ Field measurements
- __ Field log and diary entries
- __ Sample custody and audit records

Post-Sample Follow Up

- __ Filing sample logs and diaries
- __ Equipment cleaning and maintenance
- __ Proper disposal of chemical wastes
- __ Review documentation and audit reports

Table 5-4. (continued)

Laboratory Analysis

Pre-Sample Analysis Preparations
- ___ Verify use of proper analytical methods
- ___ Analyses scheduling
- ___ Verify sample number
- ___ Define a recording system for sample results
- ___ Apply a system to track each sample through the lab
- ___ Equipment maintenance and calibration
- ___ Quality control solutions on hand

Sample Analysis
- ___ Sample analysis methods and protocols
- ___ Use of reference samples, duplicates, blanks, etc.
- ___ Quality control and quality assurance compliance
- ___ Sample archiving
- ___ Proper disposal of chemical wastes
- ___ Full documentation in bench sheets

Data Record Verification
- ___ Coding sheets, data loggers
- ___ Data verification procedures and compliance with project plan
- ___ Analysis of splits are within data quality objectives
- ___ Assignment of data quality indicators and explanations

Data Management

- ___ Selection of appropriate hardware and software
- ___ Data entry practices and data validation (e.g., entry range limits, duplicate entry checking)
- ___ Data tracking
- ___ Characteristics of data achieving system
- ___ Data exchange protocols
- ___ General data availability

Table 5-4. (continued)

Data Analysis

- __ Software selection
- __ Handling missing data and non-detects
- __ Identification and use of data outliers
- __ Graphical procedures (e.g., scatter plots, notched-box and whisker)
- __ Parametric statistical procedures
- __ Non-parametric statistical procedures
- __ Trend analysis procedures
- __ Multivariate procedures
- __ Quality control checks on statistical analyses

Reporting

- __ Scheduled reports—timing, frequency, and lag times following sampling
- __ Report contents and formats
- __ Planned tables and graphics
- __ Report sign-off responsibility(ies)
- __ Report distribution recipients and availability
- __ Use of paper and electronic formats
- __ Presentations

Information Use

- __ Identification and application of decision or trigger values, resulting action
- __ Implementation of construction, control, and/or monitoring design alternatives
- __ Public release procedures

General

- __ Contingencies
- __ Follow-up procedures

5.4 Defining Specific Monitoring Program Objectives

With the goal(s) for monitoring defined, the next task when designing monitoring programs is to define the specific monitoring objective(s) (Table 5-3). Monitoring program objectives refine the broad goal(s) established for the monitoring program by, most simply, formally defining the information needs for why monitoring is to be undertaken. These objectives delimit the specific needs for monitoring data collection and specifies how these data will be used after collection, for example, by management or by regulatory agencies. They also provide limits for information expectations. With many monitoring programs, sampling and data collection efforts may require a phased implementation to address progressively more specific and detailed unknowns (objectives).

An important step during the process of defining specific monitoring objects is to compile all available relevant data for the system of concern. Then, at least a preliminary assessment of these data should be completed to determine kind and detail of information already available for the system. This often serves to focus needs for additional monitoring by step clearly establishing what is already known and what more needs to be learned about the system. For example, available information for the system should be compiled that, to the extent possible, identify, locate, and characterize:

- Point source discharges,
- Known and suspected nonpoint source discharges,
- Sites that have extensive existing data bases that warrant continued or intensified monitoring to provide long-term data sets useful to the goals of the planned monitoring program,
- Ambient receiving flow regimes and associated water qualities,
- Potentially affected sensitive receiving water, riparian, and wetland habitats, and
- Key indicator species, species of special concern, and species listed as Federal or State threatened or endangered species or are candidates for such listing.

Defining specific monitoring objectives requires focusing on realistic targets for specific monitoring activities. In this step, the program designer must identify the system variables and locations for which new monitoring data are needed. This step also requires defining the quality (accuracy and precision) of data needed to meet the monitoring objective. Monitoring programs provide sample data from limited locations and times across a range of conditions. Hence, selected sampling locations and times must provide "representative" samples under the conditions of concern to meet the monitoring objective. Each monitoring objective should specify the limits of system variation to be covered by the monitoring samples. Section 5.5 introduces implications of effluent, environmental, and sampling variability in defining monitoring designs.

Definitions of monitoring objectives should be as focused as possible. They generally should be clear and simple enough to allow fulfillment of the objectives without the needs for inventive or otherwise unproven sampling, analysis, or assessment methods. To achieve complex monitoring goals, program designers can group several monitoring objectives and address them through coordinated efforts or through a series of linked studies to address each objective until achieving the overall monitoring goal.

Various factors implicit in the these issues fall into a generally larger category of considerations required for developing *data quality objectives* (DQOs). In general, EPA's policy is that DQOs should be developed for all data collection efforts that require or result in substantial commitment of resources. It is not uncommon to include DQOs as part of the quality assurance plans for a project.

DQOs are qualitative and quantitative statements developed by a team of technical experts (including someone with statistical expertise), program staff, managers, decision makers, and other data users to specify the quality of data needed to support specific technical decisions or regulatory actions. Often, during this development process, decision makers will describe information needs, reasons for the need, how the information will be used, and specifications of any time or resource constraints on data collection. Next, technical staff and decision makers interact to detail specifications required for the problem assessment and any technical constraints possibly imposed on data-collection activities. Then, alternative approaches for data collection are defined and the approach(es) to be used selected. A clear definition of data objectives and selection of correct data-collection methods help ensure successful study completion. U.S. EPA (1985) describes the process in detail for developing DQOs; and Plafkin et al. (1989) also summarizes this material and its importance in quality assurance and quality control (QA/QC) procedures. We also include additional discussion on QA/QC considerations in Section 6.5.

5.5 Specifying Sampling Designs: Sample Sizes, Frequencies, and Locations

After clearly defining the monitoring objectives, the next step is to design the monitoring program (Table 5-3), sometimes also described as the monitoring system (e.g., Ward et al., 1990). Thus, monitoring program designers must not only define clearly the monitoring's short-term and (often) long-term objectives, but also the details of the monitoring process. A well-defined monitoring design should address:

- What are the objectives of monitoring?
- When will monitoring begin?
- What environmental (samp e) variables will be monitored?
- What methods will be used to analyze monitored variables?
- Where and how often will monitoring occur?
- How will collected data be managed and analyzed?
- What information will satisfy the program's goals, thus allow this phase of monitoring and, in some cases, the entire monitoring program eventually to end?

Monitoring objectives may require sampling designs of varying degrees of complexity, for instance:

- Point-in-time, single event, samples to broadly describe the general character of the sampled variable.
- An intensive, short-term series of collected samples over a predetermined period to detail patterns of change in monitoring variables associated with particular events, e.g., runoff from nonpoint pollution sources or a combined sewer overflow. Sample collections for such studies may occur at, for example, 5-minute, hour, or day intervals, depending on the information needs of the monitoring objective.
- Long-term samples collected at regular intervals (e.g., weekly, monthly, quarterly, or annually) to establish ambient or background conditions, or to assess, for example, general long-term trends of change or seasonal time series patterns in the monitored variables.
- Reference site studies to provide data for monitored variables against which results from other monitoring sites can be compared to judge relative changes in the variables between sampling dates (see Section 7.3).
- Near-field studies to sample and assess receiving waters within the

immediate mixing zone of discharges or runoff. These studies can be used, for example, to address monitoring objectives regarding possible impacts due to short-term, acute toxicity or long-term alterations to the habitat.

- Far-field studies to sample and assess receiving waters not within the mixing zone, but potentially affected by impacts that are delayed due to longer travel times required for discharged materials to reach these areas. Studies at such site may include assessments of possible chronic toxicity or nutrient enrichment effects.

The design of monitoring plans can involve a broad array of technical disciplines. Previous experience with these diverse problems can greatly aid in developing appropriate study designs. For example, evaluating and assessing loadings from point and nonpoint source discharges and their potential impacts—both on beneficial uses and on receiving water ecosystems—can involve consideration of:

- precipitation patterns,
- watershed hydrology and runoff coefficients,
- collection system hydraulics,
- surface water hydrology,
- water chemistry,
- aquatic toxicology,
- biological assessments,
- system modeling, and
- related areas of science and engineering.

When the monitoring program's goal is only, for example, to provide data for use in modeling applications, the design must match sample and data collection periods, frequencies, and locations to the scheduling requirements established under the monitoring objectives. Similarly, field sample and data collection techniques and laboratory analytical methods must be able to provide data that meet the quality requirements specified by the monitoring objectives before the monitoring program is ready to implement.

But, when the monitoring goal includes information objectives, monitoring program designers must identify and select statistical, graphical, or other data analysis techniques that can explicitly address the questions posed by each of these objectives. Data analysis techniques often differ in their assumptions about the data (e.g., the distribution of population or sample data) and in their sensitivity to variations in the data. Also, some techniques require larger sample sizes or more frequent sample collections than others. Therefore, to collect as much data as efficiently as possible, the sampling program's design should clearly identify the data analysis methods to be used. This will help ensure that data collection matches the data needs of the analysis methods. Definition of optimal monitoring program design must gather sufficient data to ensure that the monitoring objectives can be met, without collecting data in excess of the information needs. When first designing a monitoring program or when addressing new or unusual data problems, TMDL developers may wish to consult statisticians or other experts in the design of field monitoring programs.

As progressively more data are collected during implementation of monitoring programs, the relative amount of new information provided may progressively decreases. One problem in designing an optimal monitoring program is to determine an information/cost effective break-point for data collection efforts. This generally involves determining the minimum number of locations and times to be sampled, and minimum samples to be collected during monitoring to provide the information needed about the monitored system at the minimum level of acceptability (e.g., statistical significance).

The number and placement of sampling locations depends on the length of flowing water system or the extent of the lake, estuary, wetland, or other standing water system for which the monitoring characterization is needed. It also depends on the horizontal and vertical variability occurring throughout the area needed characterization and the degree or resolution of characterization required.

In total, establishing sampling locations must be based on objectives of the study and must be done considering various scales (Ward et al., 1990). First, at the macro level the sampling stations locations define the river reach, or lake or estuary area characterized through the sampling. These locations should directly relate to the needs defined by the monitoring plan objectives. The micro scale relates to the specific locations sampled at each sampling station. If the study objective is to characterized the aquatic community of a stream reach, then the samples should be collected from all habitat types found in that stream reach (riffles, pools, runs, side channels, etc.). If, instead, the objective is to define baseline stream conditions against which future possible changes in the system could be assessed, then sampling locations may be, for example, limited to riffle areas occurring in 0.75 to 1.5 feet of water. Most importantly, care must be taken in developing monitoring and sampling plan to avoid sampling the "wrong water." That is, collecting samples that do not the supply needed information and cannot be used to address the sampling objectives.

Determining the intervals and frequencies for sampling at monitoring locations also depends on monitoring objectives. For statistical reasons, it would generally be best to collect samples at random intervals, but practical constraints often require regularly spaced intervals. Defining the frequency of sampling (e.g., time between samplings) can also depend on the rate of natural or other change inherent in the variable sampled. Water chemistry can change in minutes and some water quality models need calibration and verification data collected at 5-minute intervals, while other monitoring objectives can be satisfied by weekly, biweekly, monthly, seasonal, or even annual samples. Summer phytoplankton communities can often replace themselves at 3- to 5-day intervals and zooplankton populations may turnover at 5-day to 2-week intervals. Summer periphyton communities can require two to five weeks to colonize and establish reasonably stable communities on artificial substrates. Benthic invertebrates colonization on such substrate can require similar periods, with their communities on natural substrates undergoing nearly daily changes accompanying emergence of adults from the water and migration by the remaining populations.

Defining locations to be sampled, frequency of sampling, and number of samples to be collected depends on the area and range of conditions to be characterized through monitoring. Further, when designing sampling networks, requirements inherent in the statistical or other data analysis procedures intended to be used to evaluate the collected data must be considered. For example, most simple statistical analyses are based on samples collected at random times and locations encompassing the range of all possible conditions of interest. Under these considerations, all possible locations and times within the spectrum of possible combinations to be characterized should have equal probabilities of being sampled. Thus, if the object is to characterize water quality conditions "in the estuary," all locations in the estuary should, technically, have an equal opportunity of being sampled to allow statistical inference of the sample results across the area. (Here, a grid of randomized locations may be placed over a map of the sampling area to identify sampling locations.) Similarly, if the objective to verify that dissolved oxygen at point 1000 feet from a point source discharge remains at concentrations greater than 6 mg/L, there ideally should be equal probabilities of collecting samples any time over each 24-hr period, seven days a week.

Restricting the scope of possible sampling times and locations correspondingly restricts the range of correct statistical inference. So, when samples are routinely collected along the south shore from the designated monitoring location between 10:00 and 11:00 AM on the first and third Tuesday of each month, the collected samples characterize *only* conditions during this one hour period on these two days per month along the south shore at these locations.

In practice, most monitoring programs rarely adhere to these randomization requirements strictly or even loosely. Typically, a set of "representative" monitoring locations are defined, a "regular" schedule of sampling dates established, with sampling beginning on each of these dates whenever the field crew can "get it together" in the morning. Not uncommonly, "representative locations" are places having ready access. Monitoring dates often occur early in the week to allow the remainder of the week for analyzing collected samples. And, sampling crews general reach their first sampling sites sometime between early- and mid morning.

For such studies, conventional statistics remain appropriate to summarize and characterize conditions *at the locations and the times sampled.* They also can be used to evaluate possible differences *among these locations and times sampled.* It further remains possible to produce deductive, subjective, and best-professional-judgement based extrapolations to conditions at other times and places, *but* the claim cannot be made that these qualitative extrapolations have a statistical, inductive basis. Finally, it should be additionally noted that the strengths of deductive inferences can tend to increase when similar results occur across increasing numbers of studies at the same sites or at other sites.

Fortunately, there are various methods to increase the potential worth of collected data and the ability extrapolate from these data with greater confidence by including aspects of randomness within the monitoring design. For example, when the objective of a study is reasonably served by regular spaced locations radiating out into the lake or estuary from point source, the compass direction from the source for each successive monitoring location might be selected using random procedures. Or, specific locations for sampling collections may be positioned in "the representative reaches" using a randomized grid method, with new sampling locations selected prior to each sampling date.

To reduce sampling time bias in collected samples, the first of the predefined monitoring locations to be sampled could be selected prior to each sampling trip using randomized techniques. For example, say five routine stream monitoring stations are identified as S1 to S5, downstream. Always sampling S1 first followed by sampling from each subsequent downstream can result in an early sample bias (e.g., early morning bias) at the upstream site, similar progressive time biases at each downstream site. But the bias can be eliminated by randomly selecting the first of the five stations be sampled and randomly determining whether the sampling occurs upstream or downstream. Three sampling order randomization for three successive sampling trips might result in sampling in the orders

- S3, S4, S5, S1, S2;

- S1, S2, S3, S4, S5; and
- S3, S2, S1, S5, S4.

Such efforts, however, can result in increased sampling costs associated with travel times between stations, but these costs may not be significant, particularly if the study design requires eliminating the sampling-time bias from collected data.

Sometimes intentional sampling-time bias can provide a useful feature in study designs. For example, when low dissolved oxygen concentrations in receiving waters, due to either organic or nutrient discharges, are a suspected problem requiring assessment and monitoring, available data or professional judgement generally can point to the area in the receiving water where the greatest oxygen depressions are most likely to occur. Since lowest dissolved oxygen concentrations usually occur in surface waters near and shortly following the dawn, sampling a station in that receiving water area first, in the early morning, during each sampling trip can help to better understand the occurrence, frequency and magnitude of the problem in the receiving water. The early morning bias in the data from that station can be use as an indicator of "worse-case" dissolved oxygen conditions over the extent of the receiving water body studied.

Overall, options for using randomized procedures are defined by the objectives of the monitoring program, as limited by practical considerations regarding the nature of the water body, possible access problems, and sampling costs. Thus, while considerations and options for including randomization in monitoring programs can provide valuable enhancements to the quality and the ability to extrapolate information produced, including such procedures in these programs generally should not result in sampling requirements that are either markedly more costly and less doable. Rarely, and perhaps never, should demands for randomized field sampling procedures obstruct completion of a monitoring program. The advice of Green (1979) is particularly relevant here. "When possible emphasize use of statistics for choosing a sample number and design that will allow convincing mean differences to be demonstrated without additional statistics, rather than to prove slight, unconvincing differences."

Adding special, supplemental studies to assess conditions not included within the routine monitoring program can also increase the potential confidence in possible extrapolations made using routinely collected data. For example, special studies can be conducted at "key" sampling stations to evaluate diurnal trends in monitoring variables collected at either random or regular intervals over a 24-hour period. (Here, random selection of the sampling location(s) and the first time of the regular sampling intervals increases abilities to correctly use the results in statistical-based extrapolations of the sampled data.) Other special studies can focus on following discrete slug flow down the stream or through the reservoir or estuary.

Determining the number of samples to be collected from each sampling station on each sampling date to provide information on a monitored variable sometimes is limited by practical considerations. For example, how much water can be reasonably packed out of a wilderness area? Or, how much space exists on the specially constructed artificial substrate holder for benthic invertebrate or periphyton samples? A single sample provides only a point estimate about the state of the sampled variable at the specific time and location sampled; no information is provided regrading the possible variability that may exist in th: estimate. Compositing two or more samples into a single sample provides information about "average" conditions from the sampled locations when sampled, but the results still cannot provide quantitative information about the variability inherent in the sample variable when sampled. Analyzing pairs of collected samples again can provide information on "average" and a possible "range" of conditions for the sampled locations, but again this sampling approach provides only weak information on variability. Progressively adding more samples enables determining progressively better information on variability.

The statistically based equations for determining sample size for a given probability level require the specification of the equations in Box 5.1 and the following quantities: WQ standard (C_s), the mean concentration where the standard should be declared attained with a high probability (μ_1), the false positive rate (α), the false negative rate (β), and the standard deviation ($\hat{\sigma}$). Box 5.2 gives an example of calculating sample size.

Box 5.1
Formulae for Calculating the Sample Size Needed to Estimate the Mean

$$n_d = \sigma^2 \left\{ \frac{z_{1-\beta} + z_{1-\alpha}}{C_s - \mu_1} \right\}^2 \qquad 5.1$$

where $z_{1-\beta}$ and $z_{1-\alpha}$ are the critical values for the normal distribution with probabilities of $1-\alpha$ and $1-\beta$ (see any basic statistical text).

The sample size may also be written in the following equivalent form:

$$n_d = \frac{\left(z_{1-\beta} + z_{1-\alpha}\right)^2}{\tau^2} \text{ where } \tau = \frac{\left(Cs - \mu_1\right)}{\sigma} \qquad 5.2$$

The term τ (Greek letter tau) expresses the difference in units of standard deviation.

Box 5.2
Example of Sample Size Calculations

Suppose it is desirable to verify compliance when the mean concentration is .2 mg/L below the WQS of .5 mg/L ($Cs = .5$, $\mu_1 = .3$) with a power of .80 (i.e., $\beta = .20$). Also suppose $\sigma = .43$, $\alpha = .05$, and 99 percent of the sample points will result in analyzable samples, then

$$\tau = \frac{(Cs - \mu_1)}{\sigma} = \frac{(.5 - .3)}{.43} = .465$$

From Standard Normal tables

$$z_{1-\alpha} = 1.645,\ z_{1-\beta} = 0.842.$$

Using Equation 5.2 from Box 5.1,

$$n_d = \frac{(z_{1-\beta} + z_{1-\alpha})^2}{\tau^2} = \frac{(.842 + 1.645)^2}{.465^2} = 28.6$$

and

$$n_f = \frac{n_d}{R} = \frac{28.6}{.99} = 28.9.$$

Rounding up, the sample size is 29.

Monitoring program designs should be clearly documentation. The importance of this cannot be overemphasized. Without such documentation, changes in key personnel can disrupt sample collection, resulting in failure to acquire needed information or changes in collection or analyses methods leading to an additional source of variation in the collected data. Also, during times of limited financial resources, budget cuts can often fall on programs that lack clear, documented reasons for their existence. Table 5-4 presents a checklist of topics under each of six general monitoring components useful when developing and documenting monitoring program designs.

5.6 Special Waterbody Considerations

Receiving waters potentially requiring TMDLs include a diversity of environmental types. Streams and rivers are linearly dynamic systems. Lakes are primarily cyclic dynamic systems. Reservoirs and estuaries are a mix of both system types, and are often influenced strongly by density gradients and currents. And wetlands and riparian areas merge aquatic and terrestrial components. Some of the key environmental characteristics for each of these types of environments, which must be generally considered when designing monitoring programs and assessing impacts, are briefly reviewed in this subsection. The key characteristics discussed are summarized in Table 5-5.

5.6.1 Flowing waters - streams and rivers

Stream waters are intrinsically interconnected to their surrounding watersheds (Motten and Hall, 1972; Hynes, 1975; Likens, 1984). Stream waters mostly originate as runoff from overland flows or as groundwater seepage from the watershed. Hence, the quality and quantity of water naturally reaching streams depend upon the infiltration capacity of the soil and on other physical and chemical attributes of the watershed's geology and soil lithology.

As water percolates over and through the soils, chemical constituents are dissolved and other chemical reactions occur that define the natural water quality characteristics in the stream. Within the stream, other mechanisms further affect the composition of dissolved substances. For example, dissolved ions can be rapidly adsorbed onto inorganic particles or absorbed by living organisms. These process also the same properties that generally affect the abilities of flowing water systems to assimilate waste chemicals and sediment discharged into them. Important considerations involved in applying the TMDL framework to flowing water systems include stream flow velocities, sediment transport, water temperatures, and material spiralling.

Table 5.5. Key considerations by environment type important to the design of water quality monitoring programs

- **Flowing waters - streams and rivers**
    - o Watershed geology and soil lithology
    - o Flow characteristics
    - o Sediment transport
    - o Water temperatures
    - o Nutrient and material spiralling
- **Lakes**
    - o Water temperatures
    - o Thermal stratification
    - o Potential anaerobic processes in sediments and near bottom waters
    - o Seasonal vertical mixing of lake waters
- **Reservoirs**
    - o Retention times for resident water
    - o Water temperatures
    - o Thermal stratification
    - o Intrareservoir water throughflow patterns
    - o Aging characteristics of reservoir
    - o Character of reservoir discharge structure
    - o Influences by sources/sinks characteristic on water qualities
    - o Downstream influences on thermal and chemical water qualities
- **Estuaries, harbors, and related near-shore brackish/marine waters**
    - o Geomorphology characteristics, e.g, depth and mouth characteristics
    - o Upstream drift of marine waters
    - o Tidal waves
    - o Stratification and vertical mixing patterns
    - o Sediment transport and sediment-water interactions
    - o Mixing processes of fresh and sea waters
- **Wetlands and riparian areas**
    - o "Nine" principal wetland functions
    - o Opportunities and effective in performing these functions

Stream flow velocities may be thought of in terms of *subcritical* and *supercritical* flows (Heede, 1980). Subcritical flows exert relatively low energies on banks and beds, while supercritical flows can produce highly erosive forces and cause channel damage. Standing waves are commonly associated with supercritical flows.

Sediment transport in streams is a very complex relationship involving at least 30 variables (Heede, 1980). But, in general, the mean particle size transported and sediment mass discharged is proportional to the water volume discharged and the gradient of the streamed. Also, water temperature can influence sediment transport. For example, 40°F water is able to carry two to three times more sediment than 80°F waters (Heede, 1980).

Flowing waters pose potentials for high erosion and production of deeply incised channels and steepened valley slopes (Heede, 1980). To counter these potentials, natural mechanisms exist that allow streams to adjust channel slopes, which helps to protect streambeds. These mechanisms include (1) *bed armoring* by gravel and boulders, (2) *gravel bars* that form transverse to stream flows, and (3) *log steps* that incorporate fallen timber and associated debris into the streamed. Through such mechanisms streams can reach a *dynamic equilibrium* with their channels (Heede, 1981).

Water temperatures in streams vary with air temperatures. This means that stream temperatures have daily and seasonal cyclic patterns paralleling air temperatures, but because of the greater density of water and the fact that water freezes at 0°C, temperature extremes are less. Thus, in the northern temperate zone, stream temperatures tend to be cooler than air temperatures during the summer and warmer during the winter. Also, water temperatures tend to warm with distance downstream in response to solar radiation and warm air; the rate of the warming is approximately proportional to the distance traveled (Hynes, 1970). In stream reaches predominated by groundwater contributions, stream temperatures tend to reflect the often cooler or warmer groundwater temperatures.

Both cooling and warming trends can be important in triggering either developmental or reproductive changes in aquatic organisms. Thus, daily and seasonal cycles in stream temperature are frequently important in the development and growth of aquatic invertebrates and fish.

In lakes and terrestrial ecosystems, nutrients move in *cycles* from organisms to soils or sediment and then back to organisms again. Often these cycles can be repeatedly completed within close proximity of each other. But in rivers and streams, the flow of water causes the cycles to be completed at progressive intervals downstream. This downstream movement of these cycles, which include nutrient processing and transport, is termed nutrient spiralling (Webster and Patten, 1979).

While moving downstream, materials can be transferred among environmental compartments where nutrients can be "stored" for varying periods, effectively altering rates for downstream movements. Storage times for different compartments can be relatively long or short. For example, some nutrients incorporate mostly into the tissue of organisms and are released back into the water at relatively slow rates; while other nutrients can be rapidly excreted by organisms, soon becoming available again for use by other organisms. Such differences produce *nutrient spirals* of different spatial lengths for different nutrients.

It has been suggested that shorter nutrient spirals help to establish constancy in stream ecosystems, leading to potentially increased biomasses, spatial heterogeneity through the stream continuum, resistance to external stresses, and ability to rapidly recover from perturbations (O'Neill et al., 1979). Disturbances will tend to disrupt storage mechanisms, increase losses of dissolved nutrients and/or nutrients in sediment, and to increase spiral lengths within streams (Webster and Patten, 1979; Newbold et al., 1983; Mulholland et al., 1985).

This valuable concept of *material spiralling* can also be applied to the transport of other materials, including toxicants. As such, this recurring pattern of temporary storage and repeated release of stored chemical should be carefully considered as a part of any TMDL in flowing water systems.

5.6.2 Lakes

Thermal characteristics are, perhaps, the single most important physical influence in lake environments. Water temperatures not only affect rates biological process at which biological processes occur in lakes and other surfaces waters, but also heat contents of the water also drive the primary processes mixing lake water and their constituent chemicals.

Typically, temperate lakes are warmed by solar radiation during the spring. As the cold (about 0°C) surface waters from winter are heated to 4°C (the temperature at which water has its maximum density), these dense surface waters sink to the bottom, pushing the colder, less dense bottom waters toward the surface. Following this spring turnover of lake water, the lake first develops uniform temperatures extending through the vertical profile of the water column. Then, as surface waters are gradually warmed, lakes often become thermally *stratified* with a layer of warm water nearest the surface (the *epilimnion*), a middle layer where temperatures decrease at about 1°C per meter of water depth (the *metalimnion* or *thermocline*), and a layer of cool water nearest the bottom (the *hypolimnion*). Warming of surface waters tends to cause the metalimnion to sink through the summer.

With the onset of autumnal cooling, the reverse of the above process occurs. Surface waters cool until 4°C is again reached and these dense waters sink to the bottom, displacing the warmer water upwards toward the surface where it cools. The fall mixing of the lake leads again to uniform lake temperatures, i.e., *isothermal* conditions.

Through the winter, as ice cover forms over the lake, lake waters often tend to stagnate with relatively warm, dense (3-4°C) water on the bottom and cold, lighter (ca. 0°C) water near the ice-covered surface. During this winter stagnation period a chemical stratification, of sorts, can develop, as will be discussed below.

Lakes that turnover twice yearly, such as just discussed, are termed *dimictic* lakes. Lakes also exist that do not mix (*amictic*), mix once (*monomictic*), or multiple times (*polymictic*). But dimictic lakes are the most common and are the ones most studied in the north temperate zone. A comprehensive discussion of the thermal properties of lakes is presented in Hutchinson (1957); good discussions are also found in most limnology texts.

Most nutrients dissolved in the upper illuminated layers of a lake (the *euphotic zone*) are incorporated into plant tissue through the process of photosynthesis by small, free-floating algae (*phytoplankton*). Concurrently, many organic and inorganic pollutants can be assimilated or absorbed into these growing cells or adsorbed onto them or adjacent nonliving particles suspended in the lake water. Subsequently, many of these algae are consumed by small floating animals (*zooplankton*) or fish, and many of the zooplankton are consumed by larger animals, including fish.

Other phytoplankton and zooplankton die without being consumed. Carcasses of these dead organisms and fecal wastes from living organisms slowly sink toward the lake's bottom. As these materials pass through the water column, chemical and bacterial decomposition release some of the bound chemicals into the water column, where they become readily available for reuse. The bulk of the settling material, however, passes through the euphotic zone into the deep, often dark (*aphotic*), waters of the hypolimnion. Eventually, much of the material settles in organic layers (*detritus*) on the bottom, where it can become permanently stored.

Decomposition of detritus continues in the deep waters and on the bottom. Because of low light intensities in these deep waters, photosynthesis usually cannot occur. Without the continual input of oxygen through photosynthesis and with the continual use of oxygen, through both decomposition of settled organic materials and respiration by the resident organisms, concentrations of oxygen continually decrease in the deep-water environment.

During periods of summer and winter stratification, oxygen can be totally depleted in the deep waters. This depletion produces reducing conditions for chemicals in the waters and in the sediment. Under such conditions, chemically bound oxygen present in the settled organic materials can be used during bacterial metabolism. For example, bacteria metabolize the oxygen contained in nitrate (NO_3) and nitrite (NO_2) compounds, which chemically reduces these substances into ammonia (NH_3) and ammonium (NH_4) containing compounds (Wetzel, 1975). In this form the dissolved nitrogen containing compounds readily move from sediment into the water column. Similar processes can occur for other to: : and non-toxic pollutants.

Phosphorus is present in aerobic sediment as both organic and inorganic compounds, including apatite and orthophosphate ions covalently bonded to hydrated iron oxides. In the reducing conditions produced by oxygen depletion, this organic bound phosphorus is decomposed and phosphorus in ferric hydroxides and complexes are reduced. Consequently, ferrous iron and absorbed phosphate are mobilized and released into the overlying water. In contrast to the importance of bacteria in recycling nitrogen compounds, they are relatively unimportant in the movement of phosphorus compounds from sediment into the water column (Wetzel, 1975).

Once nutrients and other discharge pollutants are dissolved into the deeper layers of the water column, the spring and fall turnovers can mix the redissolved nutrients up through the entire column, where these chemicals again become available to affect the biota inhabiting these upper water layers. Once exposed to oxygenated waters, however, phosphates and some other metals rapidly react to form insoluble compounds and complexes, in which form they again precipitate to the sediment. Thus, the process of recycling chemicals between the sediment and the biota in the upper water column is repeated twice annually in dimictic lakes.

When applying the TMDL process to lakes, it critical to consider the seasonal important of lake mixing, temperature, and chemical processed in the bottom sediment. These processes are key to defining both the ultimate availability, fate, and effects of many pollutants in lake ecosystems.

5.6.3 Reservoirs

Most simply, reservoirs are dammed rivers that form lakes. Consequently, reservoirs include characteristics that are similar to those in both lakes and rivers. The natural water quality in reservoirs, as in any surface-water body, is related to

(1) climate, especially the quantity and quality of incoming precipitation;

(2) the chemical nature of the geologic formations within the watershed basin

through which waters drain before entering surface waters;

(3) vegetation within the basin and its influences, along with influences of climate, on the structure and chemistry of basin soils; and

(4) influences of man within the watershed, i.e., contributions to surface runoff produced by surface disturbances or effluents discharged directly into the reservoir or its tributaries.

Construction and operation of reservoirs variously alter the natural quality of water. Factors within reservoirs that can influence the extent of these alterations include its shape (*morphometry*), *retention time* for waters within the reservoir, its age, thermal or chemical stratification of the waters, biological activities, and discharge depth(s) and timing.

A number of downstream hydrological changes are produced by all impoundments, but the extent and timing of these changes vary among reservoirs:

(1) evaporation from reservoirs reduces the total average annual runoff from watersheds;

(2) seasonal flows become less variable;

(3) annual extremes in flow are altered;

(4) magnitudes for floods attenuate; and

(5) pulses occur that are unnatural in terms of timing and duration (Petts, 1984).

Additional downstream effects often resulting from reservoir construction and operation include alteration of

(1) daily and annual thermal patterns;

(2) nutrient, dissolved gas, suspended sediment, and salinity levels;

(3) composition of bottom sediment;

(4) shoreline stability; and

(5) composition of the biological community (Canter, 1985; Petts, 1984).

The multiple effects on the downstream biological communities include alterations to the riparian vegetation, instream aquatic vegetation, composition and diversity of resident organisms, migration routes for the stream's inhabitants, biotic productivity, life cycles for the resident organisms, and trophic relationships among these organisms.

Dams also can cause permanent physical changes by increasing seismic tendencies; altering groundwater flows and water tables; inundating settled areas; destroying wildlife habitat; interfering with migrations of fish and other aquatic organisms; increasing extensive aquatic weed growths; and potentially increasing the spread of communicable diseases, particularly in tropical areas (Canter, 1985).

While thermal stratification is often the dominant physical force in lakes, through reservoir water flow is often the dominant physical force within reservoirs. This is not to say that thermal characteristics of reservoirs are not important. In fact, thermal patterns in reservoirs can be very similar to those seen in natural lakes; and the flow of water through reservoirs is often guided by thermal patterns within reservoirs. Also similar to lakes, reservoir waters are primarily heated both by solar radiation and through heat advection from inflowing waters; about 20% of the heat within a reservoir can result from the latter mechanism (Whalen et al., 1982).

One approach to estimating whether a reservoir will stratify is to use the densimetric Froude number (F). When F is less than $1/\pi$, no stratification is expected (Canter, 1985):

$$F = \frac{320\,LQ}{DV}$$

where L = reservoir length (m),
D = mean depth of reservoir (m),
Q = volumetric discharge through the reservoir (m^3/sec), and
V = volume of reservoir (m^3).

Thermal stratification is common in reservoir waters (e.g., Neel, 1963), particularly during the summer. Thermal patterns in reservoirs and lakes with surface outlets, however, can contrast markedly with those found in reservoirs with deep-water (or hypolimnetic) outlets (Wright, 1967; Martin and Arneson, 1978). In systems with surface-level outlets, the heat contained in surface water layers, which are heated by solar radiation, is relatively rapidly discharged downstream. In contrast, those systems with deep-water outlets act as *heat traps*. Here, solar radiation again heats the surface waters, but these heated waters are stored as the deeper, cooler waters are discharged. The process can cause greater masses of warmer waters to accumulate in the upper layers of deep-release reservoirs than occur in similar reservoirs or lakes having only surface outlets. One consequence of these different patterns is the relatively more rapid downward movement of the thermocline during the summer in those reservoirs with deep-water outlets.

Perhaps the most significant aspect of summer and winter thermal stratification in reservoirs is its influence on water flow through reservoirs (Neel, 1963; Wright, 1967; Marcus, 1989). Upon entering a reservoir, tributary waters may flow through the reservoir in a "river" near the surface as *overflows,* near the bottom as *underflows,* or midway through the water column as *interflows;* or they may move down the reservoir as discrete bodies of slowly moving water following prior inflows, while mingling somewhat with waters both ahead and behind (Neel, 1963; Wunderlich, 1971).

Which path inflows follow through reservoirs depends on the relative densities of both the inflow waters and the existing reservoir waters. These densities are, in turn, mostly defined by their relative temperatures. But factors other than temperature, including total dissolved solids (*salinity*), can also influence water density and flow patterns (Wunderlich, 1971). Also other factors, in particular wind, can have considerable periodic influences on the movement of water in lakes and reservoirs (Hutchinson, 1957).

Cycles for nutrients in reservoirs can be quite similar to those in lakes, discussed above. Oxygen contents in the hypolimnion of some productive reservoirs can be depleted by the decomposition of organic materials settling from the epilimnion (Neel, 1963). Differences do exist, however, between reservoir and lake chemistries. First, construction and filling of reservoirs flood the former terrestrial environments. Subsequent leaching of chemicals from the flooded soils and from rotting organic forest debris can profoundly affect water quality in the reservoir. Decaying forest materials consume dissolved oxygen and elevate carbon dioxide concentrations, while leaching can extract dissolved nutrients and organic compounds from the flooded plants and soils. As a result, heavy algae growths can be supported, undesirable levels of color and odorous substances may be produced, and conditions that enhance aquatic productivity or that can even be toxic to aquatic life may result (Sylvester, 1965; Canter, 1985).

Comparisons between areas in a reservoir where forest vegetation was left standing to neighboring areas where it was removed showed the former to be more eutrophic than the latter (Hendricks and Silvey, 1977). These researchers also noted that increases in the overlying water depths tended to decrease the influences of flooded terrestrial materials on reservoir water quality. Over time, influences from the flooded terrestrial environment on the overlying water quality decrease (Sylvester, 1965).

Nutrient regimes in reservoirs also can differ markedly from those found in lakes, depending on the relative vertical location of the outlet through which reservoir waters are discharged (Wright, 1967; Martin and Arneson, 1978). In effect, reservoirs with deep-water outlets act as nutrient sources, while those with surface outlets act as nutrient sinks, with respect to the downstream waters. Recalling the discussion for lakes, nutrients contained in organisms and their wastes mineralize as they sink through the water column, with the bulk of the mineralized nutrients becoming stored in the deeper waters and sediment. Then with spring and fall mixing, these nutrient are recycled up to water layers nearer the surface, and again become available for organic production. In contrast, reservoirs having deep-water outlets continually discharge deeper waters and the nutrients they contain (Wright, 1967). This continually removes nutrients from the reservoir, preventing them from being recycled to producers in the upper water layers.

Passage of water through reservoirs tends benefit downstream water quality by reducing turbidity, hardness, and coliform bacteria levels; and by oxidizing organic materials within the reservoir, potentially reduces downstream biochemical oxygen demand (Canter, 1985). But reservoir passage also can be detrimental to water quality by lowering reaeration rate for the water during storage, allowing buildup of inorganic chemicals in the hypolimnion that can be released to enrich downstream waters, and enhancing potentials for algae blooms (Baxter, 1977; Canter, 1985).

Perhaps the worst potential consequences of reservoir storage occur in the hypolimnion during thermal stratification when dissolved oxygen decreases, anaerobic waters develop, and iron, manganese and hydrogen sulfides can dissolve from the bottom deposits (Neel, 1963; Canter, 1985). Also, by allowing surface discharges from reservoirs to drop considerable distances into plunge pools, reservoir effluents can develop supersaturated concentrations of dissolved gases, which can have substantial adverse effects on resident stream biota (Petts, 1984).

5.6.4 Estuaries, harbors, and related near-shore brackish/marine waters

Estuaries are coastal water areas where fresh water, sea water, the atmosphere, and sediment interact. Traditionally, estuaries are defined as semienclosed water bodies that have a free connections to the open sea and, within which, sea waters are diluted with fresh waters from land drainages (Pritchard, 1967). Classical estuary systems include the lower reaches of rivers where saline and fresh waters meet and mix due to tidal action. More recently this term has been extended to include additional marine coastal waters, including saline bays and sounds receiving riverine discharges, and to include backwaters of rivers discharging into the Great Lakes (Ambrose and Martin, 1990).

Estuaries and related near-shore brackish/marine waters receive all pollutant loads contained in their tributary inputs. Also, many of these waters are commonly associated with municipal areas and ports, leading to additional direct loadings of pollutants from adjacent point and nonpoint sources and from discharged shipping wastes. Circulation patterns in estuaries often trap these nutrients, toxicants, and other pollutants. Also, bottom sediment can store these pollutants, where they can be transformed, again released to the water column, or buried. Other near-shore waters can be affected by coastal currents.

The complex loading, circulation, and sedimentation processes occurring in these waters present many difficulties for modeling water quality under waste load allocations and the TMDL framework. Water transport and circulation processes in estuaries are driven primarily by both river flow and tidal action. Upstream drift of heavier sea water can produce longitudinal salinity gradients through the estuary. Where rivers flows are strong and tidal mixing weak, relatively fresh waters can stratify over the denser saline bottom layer. When this occurs, flow of bottom waters can transport saline waters and pollutants upstream (much like the underflow pattern discussed above for reservoirs), while surface layers transport fresh water and other pollutants downstream. In some larger estuaries, coriolis acceleration, due to the rotation of the Earth, can significantly deflect currents to the right in the northern hemisphere. Also, particularly in shallow estuaries, wind can dominate estuarine mixing processes.

The geomorphology of estuaries strongly affects the transport of pollutants and, consequently, their water qualities (Ambrose and Martin, 1990). For example, estuarine depth controls propagation of tidal waves; shallow channels and sills increase vertical mixing, while deeper channels promote stratification and greater upstream salinity intrusions. Shallow sills near the mouth of estuaries can limit circulation and flushing of bottom waters. Since each estuary is unique, the fundamental processes controlling water quality in each must be evaluated individually. That is, to determine the fate and effects of water quality constituents it is necessary to first determine processes controlling their transport.

Often, one the most important aspect of water quality modeling in estuaries is the ability to successfully simulate sediment transport and sediment-water interactions, since sediment can affect transparency and carry nutrients and toxicants into the water column. Unlike fresh waters, which maintain reasonably constant water quality conditions over extended periods, the frequent large changes in salinity and pH in an estuary directly affect the transport behavior of many suspended solids. Many colloidal particles agglomerate and settle in areas of significant salinity.

The mixing of fresh and sea waters in estuaries produces a dynamic continuous variations in both space and time, with shifting patterns of potential impacts. For example, nitrogen commonly limit algal and plant growth in sea waters whereas phosphorus usually limits their productivities in fresh water; either nutrient may be limiting in estuaries. Further, the interaction between nutrient limitations from these sources may be additionally altered by nutrient from atmospheric inputs. Nearly 40% of the nitrogen in Chesapeake Bay, for example, can come from wet and dry atmospheric sources (Fisher et al., 1988).

Since pollutants entering estuaries can affect water quality both directly and indirectly, evaluating potential effects of pollutant inputs should include both quantitative and qualitative considerations. The principal concerns for water quality in estuaries include:

- **Salinity** is an important determinate of available habitat and habitat quality for estuarine organisms. High volume discharges can adversely impact estuaries by decreasing salinities in marine estuaries or increasing salinities in Great Lake estuaries.

- **Sediment** entering estuaries from various sources (e.g., tributary and nonpoint source inflows) can alter benthic habitats and carry hydrophobic organic chemicals, metals, and nutrients into an estuary. Also, upstream movement of saline waters can carry contaminated sediment upstream. Other transport process can mix pollutants from the bottom into the upper water layers.

- **Bacteria and viruses** can enter estuaries with discharges from municipal, industrial, and marina discharges and runoff from farms, feedlots, and municipal areas. These pathogens are particular concerns for impact to beneficial uses of beaches and other recreational areas, and of shellfisheries, where accumulations of pathogens in shellfish can produce health threats to humans.

- **Dissolved oxygen depletion** caused by oxidation of organic materials, nitrification, digenesis of benthic sediment, and respiration excessive growths of algae and higher aquatic plants can stress many aquatic organisms. These problems can be aggravated by excessive loading inputs of nutrients and organics from various sources, and waste heat discharges from power plants.

- **Nutrient enrichment, eutrophication, and overproduction** caused by excessive loading of nitrogen and phosphorus can produce problem blooms of phytoplankton, excessive growths by other plant species, and other problems discussed above, leading to disruption of the natural communities.

- **Aquatic toxicity** by excessive ammonia, many organic chemicals, and metal loads, often even at very low receiving water concentrations, can produce widespread problems in estuaries that disrupt natural communities. Short-term acute and long-term chronic toxicities can be affected by pH, temperature, sediment concentrations, and various interactions among the toxicants themselves.

- **Bioaccumulation and effects to humans** can occur as low concentrations in waters are taken up and concentrated leading to concentrations in organisms that are potentially toxic to fish predators, including humans. Resuspension of these toxicants stored in bottom sediment can cause such food chain effects to persist long after the time where problem causing discharges are eliminated.

Because of these complex interaction among the possible mixing processes, estuaries cannot be treated as simple advective systems, as is possible for most river systems. Both advection and dispersion must be considered is selection appropriate qualitative and quantitative models of estuarine processes (Ambrose and Martin, 1990).

5.6.5 Wetlands and riparian areas

In general, wetlands include lands where saturation by water is the primary determinant affecting soil development and the composition of plant and animal communities within the affected area (Cowardin et al., 1979). Water at least periodically saturates or covers soils and substrates of most wetlands. This event severely stresses all plants and animals not physiologically adapted to living in water or saturated soils.

Maintaining the physical, chemical, and biological integrity of wetlands, which are counted among the Waters of the United States under the Clean Water Act, is often basic to maintaining these qualities within their adjacent surface waters. Wetlands provide important hydrologic functions in purifying waters and regulating water levels within watersheds. For example, wetlands can remove and retain sediment and pollutants from the water through the combined actions of sedimentation, biological assimilation, and chemical decomposition (Mitsch and Gosselink, 1986). Sedimentation can occur as water flow is restricted by dense vegetation. At these places much of the suspended load of the stream is deposited on the wetland floor. Wetland plants can then take up (biologically assimilate) the nutrients and some pollutants. This often occurs very quickly because the high productivity of most wetlands allows many dissolved chemicals to be rapidly incorporated into biomass. While nutrients and many pollutants can be released back into the water when the plant dies, much also is retained in wetlands within peat accumulations. Chemical alterations of pollutants, including denitrification, metabolic breakdown by microbes, chemical precipitation, and physical binding to organic debris, are also important for pollution attenuation.

Mitigating effects of floods and storms in the watershed is another significant hydrologic function of wetlands. Wetlands can mitigate flood effects by decreasing water velocity and increasing the area over which the water flows, thereby increasing the rate of absorption of water into the soil. As flood waters subside, wetland soils release the stored water over a period of days to weeks, effectively "desynchronizing" flood water flows. Maintaining riparian wetlands for flood mitigation is often less expensive and more effective than building dams and other flood control projects. Coastal wetlands, with their water-absorbing ability and wind-buffering vegetation, can act as a barrier to storm surges that could otherwise flood the river basin farther upstream.

Coastal and riparian wetlands are increasingly recognized as essential to the survival of many commercial fish stocks because of their function as hatcheries. For example, the precipitous decline of the Pacific Salmon populations are thought to be partially due to the destruction of riparian habitats in the Pacific Northwest and northern California.

As the number and size of wetlands decreases, their importance to endangered flora and fauna increases dramatically. At present, wetlands are the primary habitat for disproportionally large numbers of endangered and threatened species (Mitsch and Gosselink,, 1986). This important function is increasingly making wetlands a conservation imperative.

In all, wetlands have nine *functions* that potentially benefit water qualities and beneficial used in receiving water ecosystems (Adamus et al., 1991):

(1) Ground water recharge,

(2) Ground water discharge,

(3) Flood flow alteration,

(4) Sediment stabilization,

(5) Sediment/toxicant reduction,

(6) Nutrient removal and transformation,

(7) Production export,

(8) Aquatic diversity and abundance, and

(9) Wildlife diversity and abundance.

While all wetlands tend to inherently possess all of these functions to some degree, not all wetlands have equal *opportunities* to perform these functions nor are they equally *effective* in their capability to perform these functions due to physical, chemical, and biological differences in individual wetlands and in their surrounding environments. These functions and their different opportunities and effectiveness that individual wetlands have to provide these functions, in fact, form the basis for one common technique for quantitatively evaluating wetlands (Adamus et al., 1987).

Overall, the major potential stressors that can affect the viability of wetlands are increasing or decreasing water flows; dredging, filling, or other physical alterations; and discharging nutrients or toxicants into the waters flowing into wetlands. Threats to the condition and preservation of wetlands can come from any combination of these stressors.

Riparian ecosystems occupy the interface surface-water and terrestrial ecosystems, which contain distinct soil, vegetation, and, often, wildlife characteristics (cf., Mitsch and Gosselink, 1986). They are associated with surface and subsurface drainage systems that include perennial, intermittent, and ephemeral stream channels, ponds, lakes, reservoirs, seeps, springs, and sinks, and are generally associated with high-water tables. They are also characterized by structural and functional properties of, and interactions between, both aquatic and terrestrial components. These systems are generally physically bounded at their terrestrial edge by the uppermost floodplain or by the up gradient extent of water available to vegetation through the lateral movement of groundwater from the adjacent surface waters. Riparian areas are also bounded at the water's edge, as defined by the low-flow surface water level, or the channel bottom in ephemeral and intermittent streams. Their substrates characteristically have hydric or aquic properties, and have the potential to support hydrophytic or phreatophytic vegetation. Energy, nutrients, and species typically exchange continuously among riparian, aquatic, and upland terrestrial ecosystems.

Riparian areas share the functional attributes listed above for wetlands. In fact, *palustrine wetlands*, as defined by Cowardin et al. (1979) in the classification system for wetlands used by the U.S. Fish and Wildlife Service, are a subset of riparian ecosystems. When evaluating the assimilative capacities of receiving water systems, it is generally valuable to evaluate the potential importance of wetlands and riparian ares in contributing to this process. Ignoring the roles of these systems can lead to significant underestimates of the potential assimilative capacities for the receiving water-wetland-riparian systems. The methods presented by Adamus et al. (1987,, 1991) can be used in completing assessments to evaluation these relationships.

5.7 Special Land-Use Considerations

Differing land uses also can provide unique challenges for monitoring plan designs. Table 5-6 summarizes monitoring parameters of primary concern when assessing impacts related to ten representative land uses. In addition to differences in monitored parameters, monitoring designs also can change to address different data requirements. For example, monitoring programs for urban areas typically must address concerns regarding potential individual and cumulative effects caused by multiple point and nonpoint pollution sources. Similarly, generally a broad array of pollutants are of potential concern for urban monitoring programs, including sediment, nutrients, pesticides, metals, oil & grease, other toxicants, temperature, and BOD/COD (Woodward-Clyde Consultants, 1990; U.S. EPA, 1993). Depending on the goals and objectives of the monitoring program, individual monitoring stations may be located to characterize:

(1) water quality and quantity contributions from each individual source;

(2) relative concentrations and changing characteristics along separate reaches of the stream or river, or across individual section of the lake, reservoir, or estuary;

(3) net changes in concentrations between upstream and downstream sampling river stations, or between inflow and outflows for the lake, reservoir, or estuary; or

(4) changing conditions through time at individual sampling stations within any of the receiving water types.

Monitoring programs in agricultural area are generally concerned primarily with investigating potential aquatic impacts from sediment, pesticides, and nutrients. These pollutants enter aquatic ecosystems in these areas primarily through surface runoff, although aerial application of either pesticides or, less often, nutrients can result in airborne drift resulting in direct deposition of these chemical in surface waters. Additional pollutants of concern in agricultural areas include organic (animal) wastes and total salt loads (U.S. EPA, 1993).

Table 5-6. Examples of monitoring parameters of primary concern to assess impacts related to selected land uses (modified and expanded from U.S. EPA, 1993)

Land Use	Chemical and Physical	Biological	Habitat
Cropland	Sediment, nutrients, pesticides, temperature	Benthic macroinvertebrates, algae	Substrate composition, cover, eutrophication
Grazing land	Sediment, nutrients, temperature	Benthic macroinvertebrates, fish, bacteria	Streambank stability, substrate composition, cover, channel characteristics
Feed lots	Sediment, nutrients, temperature, BOD/COD	Benthic macroinvertebrates, bacteria	Streambank stability, substrate composition, eutrophication
Urban areas	Sediment, nutrients, pesticides, metals, oil & grease, other toxicants, temperature, BOD/COD	Benthic macroinvertebrates, algae, fish, bacteria	Streambank stability, substrate composition, cover, channel characteristics
Urban construction sites	Total suspended solids, temperature	Benthic macroinvertebrates	Streambank stability, substrate composition, cover, channel characteristics
Highways	Metals, toxics, flow, sediment, oil & grease	Benthic macroinvertebrates, fish	Streambank stability, substrate composition, cover, channel characteristics
Forest harvest	Sediment, intergravel dissolved oxygen, temperature	Benthic macroinvertebrates, fish	Streambank stability, substrate composition, cover, channel characteristics
Forest road construction and maintenance	Sediment, intergravel dissolved oxygen, temperature	Fish, benthic macroinvertebrates, fish	Channel characteristics, substrate composition, streambank stability, cover
Marinas	Metals, dissolved oxygen, temperature, oil & grease, BOD/COD, bacteria	Fecal coliform	Wetland vegetation, substrate composition, cover
Channelization	Flow, temperature, sediment	Fish, benthic macroinvertebrates	Aquatic vegetation, substrate composition, cover

In areas where the agricultural practices include provisions for irrigation return flows, the qualities and quantities of a sampling of these discharge flows can be monitoring to determine contributions to receiving waters. Where such practices are lacking, contributions to flowing water may be estimated through upstream-downstream study designs to compute possible concentration differences and loading through the reach of receiving water defined by the locations of the up- and downstream monitoring stations. Non-point runoff to lakes, reservoirs, and estuaries are a more difficult monitoring problem. Often the best that may be estimate the quality and quantity of runoff by sampling receiving waters near representative location where runoff waters enter the receiving water. Also, it also can be instructive to sample representative surface-flow waters shortly prior to their entry into the receiving water, when such surface waters are available for sampling. Total contributions can be roughly estimated by extrapolation with total runoff volur s calculated using precipitation volumes and appropriate hydrologic models. Brakensiek et al. (1979) provide detailed descriptions of methods and equipment needed for discharge monitoring use in both field and watershed studies.

Timber harvest, silviculture, grazing, mining, oil and gas development, and road and highway construction are the principal sources of impact to aquatic resources in forests, rangelands, and other wildland areas. Potential aquatic contaminants of concern include sediment, pesticides, animal wastes, and, generally to a lesser extent, petroleum products. Often, the greatest concern regarding aquatic impacts in these wildland areas involve physical impacts to the habitat and food resources of fisheries caused altered flow patterns, breakdown of stable stream banks, loss of protective habitats (cover), and excess deposition of fine sediment on stream bottoms causing loss of spawning habitat and possible death or loss of benthic invertebrate food resources. MacDonald et al. (1991) provide excellent monitoring guidance useful for assessing such impacts in forest streams for the Pacific Northwest. Much of their guidance is useful in designing monitoring programs to assess impacts to potential wildland aquatic resource from other sources.

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Chapter VI. Sampling and Data Collection

Purpose: This chapter discusses in greater detail the types of data likely to be required for TMDLs, and methods of collection. Both the assembly of existing data and sampling methods for the collection of new data are summarized.

6.1 Data Requirements

The guidance document for the TMDL process (U.S. EPA, 1991a) lists five steps for a water quality-based approach to water resource protection. The first three of those steps apply directly to TMDLs:

1. Identification of water quality-limited waters
   - Review water quality standards
   - Evaluate monitoring data
   - Determine if adequate controls are in place

2. Priority ranking and targeting
   - Integrate priority ranking with other water quality planning ad management activities
   - Use priority ranking to target waterbodies for TMDLs

3. Development of TMDLs
   - Apply geographic approach where applicable
   - Establish schedule for phased approach, if necessary
   - Complete TMDL development

These three data steps are accomplished using a combination of existing data, original data collected through monitoring programs, and modeling.

6.1.1 Identification of Water Quality-limited Waters

States need to examine the condition of water bodies and evaluate whether three elements of water quality standards are supported:

- Designated uses (i.e. recreation, water supply)
- Chemical, physical, and biological criteria
- Anti-degradation requirement

Water quality data is collected and summarized under several programs. These data sets will be useful when determining which water bodes are listed under Section 303.

Section 305 (b) Water quality assessment
Section 304 (l) Impaired waters
Section 319 Nonpoint source program
Section 314 Clean lakes program

Table 6-1 lists suggested screening analyses and data sources for determining whether water bodies are impaired. Data sources are described in Section 6.2.

Table 6-1 **Screening Data for Listing of Impaired Waters**

Impairment Category	Data Sources
Waters where fishing or shellfish bans are in effect or are anticipated.	National Marine Fishery Service (NMFS) State Fish and Wildlife Agencies State Environmental Agencies State Health Departments Local Health Departments
Waters where repeated fishkills or physical abnormalities in aquatic life have been observed during the last ten years.	National Marine Fishery Service (NMFS) State Fish and Wildlife Agencies State Environmental Agencies State Health Departments Local Health Departments Recreational Fishing Groups Commercial Fishing Groups
Waters where there are restrictions on water sports or recreational contact	State Environmental Agencies State and Local Health Departments Local and Regional Park Departments
Waters identified by the State in its most recent section 305(b) report as either "partially achieving" or "not achieving" designated uses.	305(b) reports Waterbody System (WBS) Database
Waters listed under sections 304(l), 314, and 319 of the CWA.	Waterbody System (WBS) Database
Waters listed by State as priority waterbodies	State Water Quality Management Plans
Waters where ambient data, dilution analyses, or effluent toxicity tests indicate potential or actual exceedances of WQ criteria	NPDES permit applications Discharge monitoring reports Pretreatment program data
Waters classified for uses that will not support the fishable/swimmable goals of the CWA.	Section 305 (b) Water quality assessment

Impairment Category	Data Sources
Waters where ambient toxicity or adverse water quality conditions have been reported by local, State, EPA, or other federal agencies, the private sector, public interest groups, or universities.	University researchers NOAA USGS US Fish and Wildlife NMFS
Waters listed as impaired by pollutants from hazardous waste sites.	National Priority List prepared under section 105(8)(A) of CERCLA.

Adapted from U.S. EPA, 1991a.

Section 305 (b) Water quality assessment: States monitor the quality of their waters and report the status biennially to the EPA. This information is compiled into a biennial report to Congress. Information in the reports includes water-quality limited water bodies, use nonattainment causes and sources, the magnitude of the cause, and the source of the cause.

Section 304 (l) Impaired waters: Three lists of impaired waters (the mini, short, and long) are submitted to the EPA by states. The mini list contains waters that the state does not expect to achieve numeric water quality standards for priority pollutants after technology-based controls are in place. The short list contains waters that are not expected to meet standards because of toxic point sources, and the long list contains waters that are not meeting the fishable/ swimmable goals of the CWA due to any source of pollutants.

Section 319 Nonpoint source program: Under Section 319 states assess nonpoint source pollution and report to the EPA. The report lists waters that will not meet WQS without additional control action. The report also lists categories of nonpoint source pollutants which contribute to the impairment of waters, procedures for implementing BMPs, and control measures for reducing NPS.

Section 314 Clean lakes program: Under this program states were awarded grants for studying publicly owned lakes. Although the extent of the studies vary between states and lakes, many lakes were extensively studied under this program. Water quality, biologic and morphological data were typically collected.

6.1.2 Priority Ranking and Targeting

Once impaired waters are listed under step 1, they are prioritized for further action using a ranking procedure. The following ranking criteria are listed in Guidance for Water Quality-Based Decisions: The TMDL Process (USEPA, 1991a):

- Risk to human health and aquatic life
- Degree of public interest a d support
- Recreational, economic, and aesthetic importance of a particular waterbody
- Vulnerability or fragility of a particular waterbody as an aquatic habitat.
- Immediate programmatic needs such as wasteload allocations needed for permits that are coming up for revisions or for new or expanding discharges, or load allocations for needed BMPs.
- Water pollution problems identified during the development of the section 304(l) long list.
- Court orders and decisions relating to water quality
- National policies and priorities such as those identified in EPA's Annual operating Guidance.

A broad range of potential data sources will apply to this rather inclusive list of ranking criteria. Studies conducted by federal and state agencies, universities, and public interest groups should be examined.

6.1.3 Modeling Data Requirements for TMDLs

In the context of this section model is used to refer broadly to mathematical representations of pollutant sources. These representations may vary from a simple spreadsheet computation to a sophisticated computer model.

The data requirements of wet weather nonpoint source modeling efforts will vary greatly depending upon what is simulated, which model is used, and the level of accuracy required. Simple models require limited data collection and often rely on assumed or user supplied transport coefficients or concentrations. For example, the Federal Highway Administration (FHWA, 1990) predicts the pollutant load and concentration washed from roadways to adjacent bodies of water. The model requires measurements of right-of-way and pavement areas, the area of the watershed, general traffic volume data, and basic rainfall recurrence data. The model predicts a probability curve for pollutant washoff based upon either an assumed or user supplied event mean concentration (EMC). This model requires relatively little data but produces a result that is probably accurate only to an order of magnitude.

In contrast, a loading function model such as GWLF (Haith and Shoemaker, 1987) requires a larger set of data. Daily precipitation and temperature data are required, as are soil condition, land use, and agricultural practice data. The model has been successfully applied to agricultural watersheds, and provides reasonable estimates of monthly stream discharge and monthly export of nitrogen, phosphorus, and sediment.

More complex watershed scale models require much greater data collection efforts and produce continuous results in contrast to the probability curve and the monthly values produced by the two previous models. For example, DR3M-QUAL requires sub-catchment areas, impervious area, channel slopes, lengths, and roughness, infiltration parameters, channel dimensions, kinematic wave parameters, water quality parameters including wash-off and build-up coefficients. The model will produce hydrographs and pollutographs.

The Compendium of Watershed-scale Models for TMDL Development (USEPA, 1992c) lists the data inputs required by 21 models useful for TMDLs. For specific data requirements, consult model literature listed in the Compendium.

Table 6-2 lists meteorological and land data requirements used by many watershed scale water quality models. Many of the data requirements can be met, in many locations, by existing data. Typically, data collected in large scale data bases or efforts will need to be cross-referenced with other available data and a portion of the data should be field checked to assess its accuracy. As the spatially scale of data requirements becomes finer (i.e., field scale versus sub-watershed scale), existing, broadly collected data sets are typically less satisfactory and the need for field verification and data collection increases.

Water and Sediment Quality Data: Useful water and sediment quality data are shown in Table 6-3. These data may be available from prior studies, however, in many cases, additional data collection will be required to adequately classify the water quality of a water body. Most models will require a subset of this extensive list of parameters. Reaction rates are used to tune a model to represent reality. They are typically arrived at during the calibration process, although some of them such as dispersion can be measured directly. Discussions of transformation rates useful in surface water modeling are contained in *Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling* (U.S. EPA, 1985). This manual also contains numerous references for further reading.[1]

[1] In this table, three types of biochemical oxygen demand (BOD) are listed: total biochemical oxygen demand (BOD), carbonaceous BOD (CBOD), and nitrogenous BOD (NBOD). Carbonaceous BOD is caused by heterotrophic bacteria which are capable of using organic carbon as an energy source for oxidation. NBOD is exerted by nitrifying bacteria which oxidise forms of nitrogen. In a waste that contains nitrogen, BOD is initially the result of CBOD, but after several days, BOD is largely the result of the oxidation of nitrogen forms or NBOD. The reason for the lag is the time required by the nitrifying bacteria for growth. Some models will only require BOD data, while others use CBOD and NBOD to examine oxygen dynamics on a finer scale. The techniques for sampling for BOD, NBOD, and CBOD are identical, however the laboratory analytic techniques differ.

Table 6-2 General Data Requirements of Watershed Models

Topography	**Hydrology**
Watershed area	Low-flow data
Sub-watershed areas	Stage-discharge data
Channel lengths and slopes	Mean flow data
Channel roughness (Manning coefficient)	High flow data
Stream width and depth	
	Land use practices
Meteorological data	Area of land use by type
Temperature	Impervious areas
Precipitation	Street cleaning practices
Evaporation (ET)	Agricultural practices
Tidal height (for coastal systems)	Population data
Wind Data	Housing data
Solar Insolation	Industrial sites
Land Condition	**Waste Disposal Data**
Soil classifications	Wastewater point sources
Soil conditions	POTW hydraulic capacities
Slopes	Septic tank data
Vegetation cover	Industrial point and nonpoint source data
	Other significant nonpoint sources

Table 6-3 Water quality data model requirements

Water Quality Data	Sediment Data
TDS	Porosity
Temperature	Grain size
TSS	Percent solids
Light extinction	Sediment oxygen demand
DO	
BOD	**Reaction Rates, Constants**
COD	Physical dispersion
CBOD	Dissolved oxygen reaction rates
NBOD	Ph and alkalinity
Nutrients (P, N)	Nutrient dynamics
Chlorophyll	Algal dynamics
Periphyton	Coliform dynamics
Phytoplankton	
Macrophyte growth	
Alkalinity	
Ph	
TOC	
Inorganic carbon	
E_h	
Bacteria	
Metals	
Organic Compounds	

6.2 Assembling Existing Data

There are a wide variety of data sources available for examining watersheds, quantifying pollutant loads, and examining water quality. For example, a single watershed may have been sampled by university researchers, the USGS, and the State environmental agency. Care should be taken when using existing data. Whenever possible collection techniques, analytic techniques, hydrologic condition, and season should be taken into account when usin and comparing data sets. Often monitoring efforts several years apart will have used different collection or analytic techniques that will render comparisons between the data meaningless.

6.2.1 General Availability of Data

Meteorological Data: Rainfall data are available from two federal sources. The U.S. Department of Commerce National Weather Service operates thousands of weather monitoring stations throughout the country. The National Climatic Data Center (NCDC) provides weather data in computer-readable format and will supply a computer program called SYNOP for analyzing and summarizing rainfall data. The NCDC can be reached at (704) 259-0682.

Limitations of meteorological data: Precipitation and temperature values can vary markedly over very short distances. As a result, regional weather data collected even short distances from a watershed of interest can poorly represent actual precipitation and temperature values. This can be tolerable for models such as GWLF which produce monthly loading values, but can cause difficulty for modeling wet weather phenomena such as CSOs and urban runoff.

Agricultural Data: Agricultural activities can result in significant nonpoint source loads of sediment, nutrients, and pesticides. Land use data and cultivation practices are required inputs for several watershed scale models that simulate the effects of these activities. The regional Soil Conservation Service may keep records of land use for the region. The local cooperative extension office will have information on general agricultural practices for the region. The agriculture census published by the USDA is also a general source of agricultural land use information. It provides information on crops and livestock including land area planted and number of head, on a county basis. The data is usually collected on a county basis, and must be converted to watershed boundaries.

Hydrologic Data: The U.S. Geological Survey (USGS) maintains a nationwide network of stream and river gauge stations that continuously monitor flow. Extensive hydrologic data is available from regional offices. Statistics include high and low-flow events, mean flow statistics, and flow event recurrence. Some gages can supply stream discharge values on a 15 minute interval. The USGS can also provide assistance in transferring data from monitored to unmonitored watersheds. The USGS also collects data on geology, water quality and land use.

The complexity of hydrologic data required will vary from model to model. For mid-level watershed models, daily discharge values are sufficient to examine the hydrologic performance of the model. Modeling of storm flows will require a much finer time scale, since storm flows are short-term, dynamic events. Most dry weather pollutant investigations use low flow event statistics such as the 7Q10 to examine peak stresses to aquatic environments. The 7Q10 is the ten year recurring seven day low flow value. Wet-weather investigations such as storm water and CSOs will use storm-related flow statistics.

Other sources of hydrologic data include the SCS Hydrologic Bulletin, the U.S. Agricultural Research Service, the U.S. Forest Service, and the U.S. Bureau of Reclamation.

Soil Condition: The Soil Conservation Service publishes maps of soil types and condition. These data are required by many watershed scale models to determine erosion dynamics. Maps of soil classification are available for most locations. The SCS supports local soil conservation districts. Some district offices also collect agricultural practice information.

Topography: The USGS publishes 7.5 minute topographical maps for most regions of the country, and has digital information files for use in GISs for many regions.

Population: The U.S. Census Bureau Data Publishes population and housing data, by county and by census tract. Census data can be linked to GIS (Geographic Information System) software using TIGER software.

Land Use Data: Data from state, regional, and city planning agencies are useful for developing land use estimates for watersheds. Land use data are also available from the Land Use Data Analysis (LUDA) program of the USGS, the National Resources Inventory of the U.S. Soil Conservation Service, the Bureau of the Census, and the Census of Agriculture.

6.2.2 Databases

The majority of the following databases are national-scale data collection efforts. Many of them contain valuable information, however, the quality and completeness of the data varies, and they should be cross-referenced with local data sources to ensure data correctness and completeness. Detailed references for the following databases are contained in the *Inventory of Exposure-Related Data Systems Sponsored by Federal Agencies* (U.S. EPA, May 1992d) and *Office of Water Environmental and Program Information Systems Compendium*. (U.S. EPA, 1992e). While data extracted from these existing data sources may not be sufficient to calibrate a complex water quality model, it is useful for simpler screening models and may be useful both to scope and to supplement monitoring efforts. There is significant overlap between several of the databases and the ability to link information by stream reach number.

City and County Files: These databases contain city and county information including city and county names, latitude and longitude, census populations, and Federal Information Processing Standards (FIPS) state and county numbers. Using these databases, the user can link political boundaries (cities, counties) to river reaches. The user can access the names of cities and counties by stream reach, and extract information from STORET and Reach Files databases.

Contact: Bob King, Office of wetlands, oceans, and watersheds, Assessment and watershed protection division, 202-260-7028.

Effluent Guidelines Studies: This database is collected by the U.S. EPA Office of Science and Technology to support industry specific technology-based effluent guidelines. Effluent samples in the database are used to develop technology based effluent limits for classes of industries and classes of wastewaters. The user can access industrial sampling data by NPDES permit number, so the database may of use for some location-specific studies. While this database will be of limited use to TMDL efforts, it may provide supplemental data for screening efforts.

Contact: Eric Strassler, Office of science and technology, Engineering and analysis division, 202-260-7120.

Gage Files: This database contains hydrologic data for each reach in the EPA Reach file. The data is collected at USGS stream gaging stations and modeled for reaches where gages are not present. Data available for lited stream reaches includes gage station ID, type of data collected, collection frequency, mean and annual flow, location, reach number, and 7Q10 low flow. The user can link to Reach files, STORET data, and Reach file statistical and mapping tools (see separate descriptions).

Contact: Bob King, Office of wetlands, oceans, and watersheds, Assessment and watershed protection division, 202-260-7028.

Lake Analysis Management System: This is a database of Great Lakes projects. It contains physical and biological information on water, sediment, fish, and phytoplankton from the Great Lakes Basin. The database contains data from 12 projects including the Green Bay Mass Balance Project, and the Upper Great Lakes Channel Study.

Contact: Mr. William L. Richardson, U.S. EPA, 9311 Groh Road, Grosse Ile, MI, 43138-1697, 313-692-7611.

Reports: NTIS, 5285 Port Royal Road, Springfield, VA, 22161, 703-487-4650; Debra Caudell, U.S. EPA, Environmental Research Laboratory, 313-697-7600.

Hydrologically Linked Data Files (HLDF): This system links the following databases:

- Reach File
- Stream Gaging Inventory (GAGE)
- Industrial Facility Discharge (IFD)
- Water Supply Data Base (WSDB)

Industrial Facilities Discharge File: This database tracks industrial point source dischargers. There are approximately 120,000 records in the file. The database can be cross-referenced to STORET and Reach File data by either reach number or NPDES permit number.
The user can obtain NPDES permit information and link to the Gage File, PCS and STORET (see separate descriptions).

Contact: Bob King, Office of wetlands, oceans, and watersheds, Assessment and watershed protection division, 202-260-7028.

National Oceanic and Atmospheric Administration (NOAA): NOAA has several environmental monitoring programs which store data on marine and estuarine waters. The four databases and data compilations described below contain water quality, biotic, and sediment quality data for major estuarine areas throughout the country. Several pollution estimation efforts are part of the data compilation efforts. These estimates are probably most useful for screening waters for impacts and for determining which waters to include in TMDL programs.

National Coastal Pollutant Discharge Inventory Program (NCPDI): This database contains loading estimates for point, nonpoint, and riverine sources that discharge to estuarine and coastal waters. Discharge loading estimates for nine classes of pollutants are included in the database. At present the estimates represent the period 1980 - 1085. The source categories for the estimates include point sources, urban and non-urban nonpoint sources, upstream sources, irrigation return waters, oil and gas operations, marine transportation operations, accidental spills and dredging operations. Sources of land use data for the estimates include the Land Use Data Analysis (LUDA) program of the USGS, the National Resources Inventory of the U.S. Soil Conservation Service, the Bureau of the Census, and the Census of Agriculture. Data from state, regional, and city planning agencies was also used.

Reports: Dan Farrow, SEA, ORCA, NOAA, 1305 East-West Highway, SSMC4 9th Floor, Silver Spring, MD, 20910 (301) 713 - 30C

National Estuarine Inventory: The National Estuarine Inventory (NEI) is a database which combines information from a variety of NOAA sources including the NCPDI, the NS&T, and the National Shellfish Register. Data from the NEI is periodically published as a series of data atlases. Data collected through 1990 is in the process of being published. As part of the NEI estimates of pollutant loading are made for estuaries. Point source information is extracted from PCS, nonpoint sources are estimated using the Simulator for Watershed Resources- Rural Basins (SWRRB) and Natural Resource Inventory (NRI) data, upstream sources are estimated using USGS data.

Contact: Dan Farrow, Strategic Environmental Assessment Division (SEA), Ocean Resource Conservation and Assessment (ORCA), NOAA, 1305 East-West Highway, SSMC4 9th Floor, Silver Spring, MD, 20910 (301) 713 - 3000.

National Shellfish Register of Classified Estuarine Waters (Register): The Register database contains a variety of information on estuaries classified for shellfishing. The Registry was originally established to inventory the area and classification status of shellfishing waters. Recently water quality data has also been included in the Registry. The 1990 Register contains information on point and nonpoint pollution sources, shellfish productivity data, reasons for classification changes, and discusses of relationships between shellfish productivity, pollution sources, classifications, and public expenditures.

Contact: Eric Slaughter, NOAA, ORCA, 1305 East-West Highway, SSMC4 9th Floor, Silver Spring, MD, 20910 (301) 713 - 3000.

National Status and Trends for Marine Environmental Quality (NS&T): The NS&T is a combination of several programs aimed at documenting chemical concentrations reported in marine organisms and sediments. It consists of the Mussel Watch Program, the Benthic Surveillance Project, Biological Effects Surveys and Research, Historical Trends Assessments, Specimen Banking, Regional Assessments, and a Quality Assurance Program.

Contact: Thomas O'Conner, NOAA, ORCA, 1305 East-West Highway, SSMC4 9th Floor, Silver Spring, MD, 20910 (301) 713 - 3000.

National Water Quality Networks Program: This program includes the National Stream Quality Accounting Network (NASQUAN) which is intended to detect trends in water quality of surface waters and to relate those trends to upstream land and water use. Data are stored in the USGS WATSTORE and EPA's STORET data bases (see separate entries). Data can also be accessed through the USGS National Water Data Exchange (NAWDEX).

Contacts: NAWDEX, 703-648-5664; Information guide, call USGS Chief Hydrologist for Operations, 703-648-5031.

Ocean Data Evaluation System: The Office of Wetlands, Oceans, and Watersheds (OWOW) maintains this database on marine water quality. It contains data from the National Estuary Program, the Great Lakes National Program Office, the Ocean Disposal Program, the 301(h) sewage discharge program, the NPDES program, and the 403(c) (ocean discharge) program. Data is available in electronic or hard copy format.

Contact: Bob King, Office of wetlands, oceans, and watersheds, Assessment and watershed protection division, 202-260-7050; Tad Deshler, Tetra Tech, 206-822-99596. For user ID contact Kim Stahlman 703-841-6005.

Permit Compliance System: PCS was created by the USEPA to meet the informational needs of the National Pollution Discharge Elimination System (NPDES) program under the Clean Water Act. The Office of Wastewater Enforcement and Compliance (OWEC) maintains this database to track the permit, compliance, and enforcement status of facilities regulated under the NPDES program. Only major permittees (approximately 7,100 of 63,000 nationwide) are included in the database.

Regional USEPA offices and state users of PCS are responsible for the entry and quality of the data in the system. The information contained in each record includes the identity of the permitted pollutant, the discharge limits for that pollutant, the name, address, and description of the discharging facility, and the measurements of the permitted pollutant from discharge monitoring reports.

PCS contains only those chemicals which are regulated under the Clean Water Act and permitted in waste water discharges. Inclusion in PCS indicates that the particular chemical has been discharged into surface water, and is a possible source of drinking water contamination. The pollutant measurements from the 1990 discharge monitoring reports, given as concentrations, when multiplied by the flow, yield values for loading to the nation's water.

Contact: Dela Ng, OWEC (EN-338), U.S. EPA, 202-475-8323. For reports contact: U.S. EPA Library, Washington, DC--Freedom of Information Requests, or George Gray, 202-475-8313.

Reach Files: This database contains information about segments or *reaches* of waterbodies. The data include latitude and longitude of reaches, reach length, reaches connected to the reach, and descriptions of water bodies. The Reach File system includes mapping and analytic tools that allow linking to ARC/INFO based GISs.

Contact: Bob King, Office of wetlands, oceans, and watersheds, Assessment and watershed protection division, 202-260-7028.

STORET: STORET is a database utility maintained by the USEPA for the STOrage and RETrieval of parametric ground and surface-water quality data. STORET contains water quality information for waterways throughout the country. The majority of the data is from non-EPA sources including: the USGS, U.S. Army Corps of Engineers, U.S. Forest Service, Bureau of Reclamation, and states. The data in STORET are stored in several separate, but related systems. The systems pertinent for TMDLs are (1) the Water Quality System, which records geographical, political, and descriptive information concerning the sampling sites; and (2) the Biological System (BIOS) contains biological information on aquatic organisms including distribution, abundance, condition, ad habitat descriptions; (3) the Daily Flow System which contains daily flow information from USGS gaging stations as well as some water quality information; and (4) the Fish Kill File which contains information on fish kills resulting from pollution sources from 1960 through 1990.

STORET contains only those chemicals which are actively tested for in water; for the most part chemicals which are already regulated or recognized as health and environmental concerns. Appearance in STORET indicates that the particular chemical was found in surface or ground water and is a possible contaminant source for drinking water. Software links are being developed outside of the EPA to link STORET files to GISs.

Contact: Thomas Pandolfi, U.S. EPA, Office of Water, Assessment, and Watershed Protection, 202-260-7030. User registration Dora Craig, 202-260-7031.

Toxic Release Inventory (TRI): Section 313 of the Emergency Planning and Community Right to Know Act, under SARA Title III, required EPA to establish a computerized national database of toxic chemical emissions from manufacturing facilities. This database includes information supplied by industries on releases of toxic materials that occur during manufacturing and processing of materials. TRI is a composite of more than 70,000 submissions of release reports filed on Section 313 chemicals. Release reports are required annually, and data are available through 1990. Chemical release information includes quantitative estimates of the number of facilities and the associated individual releases of chemicals to air, water, and land (data can be separated on these three release medium categories). Facility information includes location, industry by SIC code, storage quantity, and waste treatment data.

TRI is a voluntary reporting system for manufacturers and the quantities of chemical released are estimated. The accuracy of these estimates probably varies greatly. The number of facilities reporting may not be an exact indicator of the number of releases because a facility may have multiple releases and companies with ten or fewer employees are not required to report under this program. Since release quantities are annual estimates, peak releases are not indicated. Inclusion in TRI shows that the specific chemical has been released into surface or ground water or via land application. The use of this database is probably limited to TMDL screening analyses.

Contact: Steve Newburg-Rinn, U.S. EPA, Office of Toxic Substances, 202-382-3757.

Waterbody System: The Waterbody System (U.S. EPA, 1991b) contains state water quality assessment information collected to meet reporting requirements under section 305(b) of the Clean Water Act.

WATSTORE: The Water Data Storage and Retrieval System (WATSTORE) is the USGS's repository for all of its water data including location, chemical and flow information on surface and ground water. It is grouped into files:

Water Quality File
Groundwater Site Inventory File
Daily Values File
Peak Flow File
Water Use File
Station Header File

This database will be incorporated into the National Water Information System II which is being developed by the USGS.

Contact: John Briggs, USGS, WRD, National Center, MS-437. 12201 Sunrise Valley Drive, Reston, VA, 22092, 703-648-5624.

6.2.3 Database Tools

The following database tools are used for linking data contained in several environmental databases.

Graphic Exposure Modeling System (GEMS): The GEMS system is an integrated modeling tool for assessing surface and ground waters. It is linked to the EXAMS model (a risk-exposure model) and to several dilution based surface water models including PROUTE and REACHSCAN.

Contact: Sondra Hollister, U.S. EPA, Office of Prevention and Toxic Substances, 202-260-3929.

Reach Pollution Assessment: This is a database tool that accesses and links data from the Toxic Release Inventory (TRI), Permit Compliance System (PCS), the section 304(l) short list, STORET, the river Reach file, and the Industrial Facilities Discharge File. The RPA can be used a screening tool to determine which pollutants are likely to be found in a given area or river reach. It can help the user generate a map of water quality sampling stations, industrial dischargers, POTWs, and released chemicals for a given area. It does not however calculate or model the fate or transport of pollutants.

The RPA is accessed through the National Computer Center, through federal, state, university and private communication networks. Users must be registered to access the IBM 3090S computer at the NCC.

References: *The Reach Pollutant Assessment User's Guide.*

Contact: Thomas Pandolfi, U.S. EPA, Office of Water, Assessment, and Watershed Protection, 202-260-7030. User registration Dora Craig, 202-260-7031.

Environmental Display Manager: This display tool can be used to link STORET and PCS databases, and can generate maps of permitted discharges, water bodies, and population centers.

6.3 Sampling Techniques

This chapter provides general guidance on appropriate uses of alternative sampling and data collection schemes. It reviews the mechanics and potential problems of sampling. Selection and use of continuous monitoring devices, data recorders, and data loggers is reviewed. Special considerations regarding collection and handling of chemical and biological samples are summarized with appropriate guidance to additional sources for more detailed information. Conventional methods currently used to assess habitat qualities and capacities are introduced. Unique safety concerns for sampling during wet-weather conditions are reviewed.

6.3.1 Benefits and Limitations of Alternative Sampling Protocols

Monitoring samples are collected to gather physical, chemical or biological information about a system. Most often, water or sediment samples or both are collected from locations chosen during design of the sampling plan. They may be collected by hand, *manual samples*; collected using by an automated sampler, *automatic samples*; collected as individual point-in-time samples, *grab* or *discrete samples*; or combined with other samples, *composite samples*. Samples collected and mixed in relation to the measured volume within or flow through a system are commonly termed *volume-* or *flow-weighted composite samples*, whereas equal volume samples collected at regular vertical intervals through a portion or all of the water column may be mixed to provide a *water-column composite sample*.

There are various purposes for collecting and analyzing samples. Concentrations of contaminants in samples may be of interest for a variety of reasons including comparison with a water quality standard or comparison with the known toxicant tolerances for resident species. Mass loads of contaminants computed by multiplying concentrations and flow rates may be used in assessing potential impacts from an effluent discharge. The types, varieties, concentrations, and loadings to surface-waters by contaminants of concern are needed when developing TMDL models and when evaluating effectiveness and benefits from implementing treatment, control, and management mitigation options.

Manual Grab Sampling. Samples collected by hand using various types of containers or devices to collect water or sediment from a receiving water or discharge are often termed *manual grab samples*. These samples can require little equipment and allow recording miscellaneous additional field observations during each sampling visit.

Several advantages of manual sampling exist. They are generally uncomplicated approaches, often inexpensive—particularly when labor is already available, and are required for sampling some pollutants. For example, according to *Standard Methods* (APHA, 1992), oil and grease, volatile compounds, and bacteria, must be analyzed from samples collected using manual methods. (Oil, grease, and bacteria can adhere to hoses and jars used in automated sampling equipment, causing inaccurate results; volatile compounds can vaporize during automated sampling procedures or be lost from weakly sealed sample containers; and bacteria populations can grow and community compositions change during sample storage.) Disadvantages of grab sampling include possible needs for personnel to be available around-the-clock to sample storm events and possible exposure of personal to hazardous conditions during sampling. Also, long-term sampling programs involving many sampling locations can be relatively more expensive.

Grab sampling is often used to collect discrete samples that are not combined with other samples. Grab samples also can be used to collect volume- or flow-weighted composite samples, where several discrete samples are combined proportion to measured volume or flow rates, however this often is more easily accomplished using automated samplers and flow meters. Several examples of manual methods for flow weighting are presented in the NPDES Storm Water Sampling Guidance Document (U.S. EPA, 1992a). Grab sampling also may be used to composite vertical water-column or aerial composite samples of water or sediment from various water kinds of bodies.

Automatic Sampling. Automated samplers have been improved greatly in the last ten years and now possess features that are useful for many sampling purposes. Generally, such sampling devices require larger initial capital investments or the payment of rental fees, but they can reduce overall labor requirements costs (especially for long running sampling programs), and increase the reliability of flow-weighted compositing. Some automatic samplers include an upper part consisting of a microprocessor based controller, a pump assembly and a filling mechanism, and a lower part containing a set of glass or plastic sample containers and a well that can be filled with ice to cool the collected samples. More expensive automatic samplers can include refrigeration equipment in place of the ice well; such devices, however, require using a 120 volt power supply instead of a battery to operate. Also, many automatic samplers can accept input signals from a flow meter to activate the sampler and initiate a flow weighting compositing programs. Some samplers also can accept input from a rain gage to activate a sampling program.

Most automatic samplers allow either the collection of multiple discrete samples or the collection of single or multiple composited samples. Also, samples can be split between sample bottles or composited into a single bottle. Samples can be collected on a predetermined time basis or in proportion to flow measurement signals sent to the sampler.

When an automatic sampler is activated, the following sequence generally occurs. The sampler pumps air through the collection line to displace any remains from a previous sample and to clear debris from the end of the line. The sampler then draws a sample according to the user defined program or flow meter signals. Finally, the meter again pumps air through the line to expel any remaining water.

In spite of the obvious advantages of automated samplers, they include some disadvantages and limitations. Some pollutants cannot be sampled by automated equipment, unless only qualitative results are desired. While the cleaning sequence provided by most such samplers provide reasonably separate samples, there is some cross-contamination of the samples since water droplets usually remain in the tubing. Often, debris in the sampled receiving water (e.g., plastic bags) can block the sampling line and prevent sample collection. If the sampling line is located in the vicinity of a flow meter, debris caught on the sampling line can also lead to erroneous flow measurements. Thus, while automatic samplers can reduce manpower needs during storm and runoff events, these devices must be checked for accuracy during these events, and they must be regularly tested and serviced. If no field checks are made during a storm event, data for the entire event may be lost. Thus, automatic samplers do not eliminate the need for field personnel, but they can reduce these needs and can produce flow weighted composite sample that may be either tedious or impossible using manual methods.

Discrete versus Composite Sampling. Flow rates, physical conditions, and chemical constituents in surface waters often vary continuously and simultaneously. This presents a difficulty when attempting to determine water volumes, pollutant concentrations, and masses of pollutants or their *loads* in the waste discharge flows and in receiving waters. Using automatic or continuous recording flow meters allow obtaining reasonable and continuous measurement of flow rates for these waters. Pollutant loads can be then be computed by multiplying these flow volumes over the period of concern by the average pollutant concentration determined from the discrete or flow composited samples. When manual (i.e., instantaneous) flow measurements are used, actual volume flows over time can only be estimated for loading calculations, but this approach adds additional uncertainty into the loading estimates.

Where monitoring budget are unlimited, samples could be collected and analyzed with a frequency approaching continuously. This would allow clearly establishing patterns of changing chemical concentrations and physical conditions through time. But, monitoring budgets are not unlimited—realistic sampling and analysis efforts must be allocated to provide the maximum possible information for the minimum possible cost.

Analyzing constituents of concern in a single grab sample collection would provide the minimum information at the minimum cost. Such an approach could, however, be appropriate where conditions are relatively unchanging, for example, during periods without rainfall or other potential causes of significant runoff events. Most often, the usual method for collecting samples would be to collect a random or regular series of grab samples at predefined intervals during storm or runoff events. When samples are collected often enough, such that concentration changes between samples are minimized, then a clear pattern or time series for the pollutant's concentration dynamics can be obtained. When, however, sampling intervals are spaced too long apart in relation to changes in the pollutant concentration, less clear understanding of these relationships are obtained.

Mixing samples from adjacent sampling events or regions (i.e., compositing) allows analysis of fewer samples and, for some assessments, this is a reasonable approach. Sample compositing provides a cost savings, especially related to costs for water quality analyses, but it also results in loss of information. For example, information on maximum and minimum concentrations during an runoff event is generally lost. But, compositing many samples collected through multiple periods during the events can help ensure that the samples analyzed do not include only extreme conditions, not entirely representative of the event.

While analytical results from co⁻ ıposited samples will rarely equal "average" conditions for the event, they still can be used, when a sufficient distribution of samples are included, to provide "reasonably representative" conditions for computing loading estimates. In some analyses, however, considerable error can be associated with using analytical results from composited samples in completing loading analyses. For example, when maximum pollutant concentrations accompany the maximum flow rates, yet concentrations in high and low flows are treated equally, true loadings can be underestimated. Consequently, when relationships between flow and pollutant concentrations are unknown, it is often preferable to initially include in the monitoring plan at least three discrete or multiple composite sample collections during the initial period of increasing flow, during the period of the peak or plateau flow, and during the period of declining flow.

The most useful method for sample compositing for flowing waters to assess loadings is to combine samples in relation to the flow volume occurring during study period intervals. There are two variations for accomplishing flow weighted compositing:

1) Collecting samples at equal time intervals at a volume proportional to the flow rate (e.g., collect 100 ml of sample for every 100 gallons of flow that passed during a 10 minute interval), and

2) Collecting equal volume samples at varying times proportional to the flow (e.g., collect a 100 ml sample for each 100 gallons of flow, irrespective of time).

The second method is preferable for sampling to assess loadings accompanying wet-weather flows, since it results in samples being collected most often when the flow rate is highest.

A third method is time-composited sampling, where equal sample volumes are collected at equally spaced time intervals (e.g., collect 100 ml of sample every 10 minutes during the monitored event). This approach provides information on the average conditions at the sampling point during the sampling period. This approach should be used, for example, to determine the average toxic concentrations that resident aquatic biota are exposed during the monitored event. Also, EPA recommends that to protect against acute effects, the 1-hour average exposure should not exceed the critical maximum concentration (CMC) more often than once every three years (U.S. EPA 1991c). One-hour, time-composited samples will allow direct evaluation of this recommendation for individual wet-weather events.

6.3.2 Precipitation Estimates

Rainfall data are available from two federal sources. The U.S. Department of Commerce National Weather Service operates thousands of weather monitoring stations throughout the country. The National Climatic Data Center (NCDC) provides weather data in computer-readable format (U.S. EPA, 1992a) and will supply a computer program called SYNOP for analyzing and summarizing rainfall data. The NCDC can be contacted at (704) 259-0682.

It should be noted, however, that local rainfall can vary significantly from nearby weather stations because the average storm cell is 4 to 5 miles in diameter, resulting in large variations of intensity over short distances. Consequently, a long-term monitoring plans often should include local rain gauges. Three types of recording rain gauges are generally available: the tipping bucket gauge, the weighing gauge, and the float-recording gauge. A thorough discussion of rainfall characteristics is presented in *Evaluation of Wet Weather Design Standards for Controlling Pollution from Combined Sewer Overflows* (U.S. EPA, 1992b).

6.3.3 Flow Rates and Receiving Water Volumes Estimates

Accurate flow monitoring is critical to compute dilution rates, mass loadings, and predicting potential impacts in receiving waters. Selecting the most appropriate monitoring technique depends on site characteristics, budgetary constraints, and personnel availability. The following subsections introduce general options and considerations useful when estimating flow rates and receiving water volume estimates for receiving waters. The discussion conclude with list of several useful references where more detail information can be obtained.

6.3.3.1 Existing Monitoring Stations

The U.S. Geological Survey (USGS) maintains a nationwide network of stream and river gauge stations that continuously monitor flow. Various state and regional organizations maintain similar networks of more limited geographical distribution.

When an existing gauge station is sufficiently near the points of interest in a stream or river, additional flow monitoring in the stream may not be necessary. If the gauge station is located in the same watershed, but up- or down-stream from the discharge, the station's data can be converted to information relevant to the discharge point using an appropriate transfer equation of the following form:

$$Q_a = Q_b\left(\frac{A_a}{A_b}\right)^n$$

where:

Q_a	=	Flow statistic desired for un-monitored watershed or sub-watershed upstream of the CSO discharge.
Q_b	=	Flow statistic for a monitored regional, similar watershed, or sub-watershed.
Aa	=	Area of un-monitored watershed or sub-watershed.
A_b	=	Area of monitored watershed.
n	=	Regional flow exponent, derived for a regional network of gauges.

A network of gauges can be similarly used to develop coefficients for the following equation:

$$Q = CA^p$$

where C and p are regional coefficients developed by regression analysis.

Stream monitoring data collected in other regional watersheds can sometimes be similarly transferred to other nearby watersheds when other information for the watershed of interest is not available. It is important to remember, however, that additional error is introduced into flow estimates when using these regional approaches. For example, these equations are based on the assumption that the watersheds used to develop regional coefficients have similar hydrologic responses. Local USGS offices and USDA Soil Conservation Service (SCS) often can be consulted for help in appropriately converting such data.

6.3.3.2 Flow Measuring Devices

When sufficiently information from an existing gauge station is not available, it may be necessary to install new flow monitoring stations to address the needs of the monitoring program. Flow monitoring requirements can vary from measuring flows during individual sampling visits at each monitoring stations, to installing depth stage gauges (also called, "staff gauges") affixed to bridges or posts near the waters edge, to installing complex and expensive stilling wells and continuous monitoring equipment, or to a combination of several techniques.

Often it is useful to collect manual flow measurements using portable meters to measure flow velocities. Although these methods can be time consuming, these devices can provide valuable "snapshots" of the receiving water flows each time the monitoring station is visited, can be used to characterize microhabitat conditions in the flowing water that may be important to some monitoring objectives, and can be used to calibrate stationary devices installed to monitor flow.

The primary stationary devices available include stage gauges, weirs, and flumes. The main advantage of these devices is that the flow rate can be determined by reading only the water depth, after water depth to flow volume relationships have been developed. Flow measurements taken with these devices are significantly more accurate than estimates based on flow-depth equations. Their primary disadvantage is that they cannot measure flow where surcharging or backflow occur or where pooling occurs immediately downstream of the gauging device. Also, the use of the last two devices are primarily limited to moderate to small streams.

Flow (Velocity) Meters. Rotating, mechanical current meters, such as the Pygmy or Gurley meter, are frequently used to measure water velocity in streams and rivers. Other more recently developed velocity meters use ultrasonic or electromagnetic technology to sense flow velocity at a point, or in a cross section of a flow. Many of these electronic devices are available as portable models or for permanent installation.

Velocity measurements must be combined with a depth values to compute flow volume. Often the width of the stream is broken up into segments. Then the average velocity in each segment of a stream or rivers is determined by measuring the flow velocity at 0.2, 0.4, and 0.8 of the maximum flow depth. The average stream velocity is determined by multiplying each segment's velocity by each segment's area. These products are summed and divided by the total cross-sectional area. Small shallow streams are often measured by using a hand held gauge while wading and the velocity of larger streams is measured from a bridge or a tethered boat. **Extreme care should be taken when working in or near moving waters.**

Many of the newer meters combines velocity sensor with a depth sensors in a single probe. For example, Ultrasonic Doppler sensors detect a phase shift between the source and reflected signals, and electromagnetic meters measure the current generated when the flow moves through a magnetic field. Velocity meters can measure flows in a wider range of locations and flow regimes than can mechanical flow devices, and they are not as prone to clogging as are the mechanical devices. They are, however, more expensive equipment and can be inaccurate at low flows and when suspended-solid loads vary rapidly. Installations using ultrasonic velocity sensors automatically switch to a Manning Equation (see Section 6.3.3.3) when flow depth is too low for the sensors to function properly.

Many flow meters can also be used to send signals to automated water quality samplers, allowing the collection of water samples on a flow-weighted basis. Typically, the user can calibrate these devices to determine how much flow passes through the flow meter, before the automated flow sampler collects draws the next water sample.

Velocity Measurement With Floats. Although less accurate than using current meters, this technique is useful for rough flow estimates when current meters are not available. Surface flow velocities can be estimated by using a slightly buoyant float, such as an orange. While pieces of wood or grass at times also may be used, their use should be limited and care must be employed to distinguish between movements of these light objects by water versus wind. In use, the surface velocity is determined by placing one or more floats in a stream and measuring the time needed to travel a measured distance. Mean velocity for the stream can then be estimated by multiplying the measures surface velocity by a coefficient, usually 0.85. Flow is the product of the mean velocity and the cross-sectional area of the stream.

Stage-Discharge Gauging. Water flows are related to the stage, or the height of the water in the stream channel: the higher the stage, the greater the flow volume. A rating curve can be developed for each site where the stage (staff) gauge is installed by making instantaneous stream flow measurements using portable methods and plotting them against the stage of the stream at the time of measurement. A minimum of five direct stream measurements of flow volume to stage should be made, with the measurements describing the full range of stream flow conditions. Often, measurements during the higher flows are the most difficult to obtain. If they are missing from the stage-discharge plot, flow estimates during high flows may be inaccurate because they require extrapolations beyond the range of the data used to calibrate the stage gauge.

Once a stage-discharge relationship has been developed, flow rates can be accurately estimated by reading the stage gauge periodic intervals or by recording water depths using continuous recording equipment. Automatic water-level recorders can provide a continuous record of flow and do not require tending by field personnel during storm events. They are, however, expensive and require regular maintenance (see additional discussion below).

Weirs. Weirs are permanent or temporary devices placed across the flow channel. They generally consist of a plate or wall structure with a notch, usually with a rectangular or v-notched cut into their face. There are two main types of weirs—broad-crested, and sharp-crested. Broad crested weirs are dam like structures which pass flow in a predictable manner. Masonry weirs can appear like brick walls and can behave in a manner similar to broad-crested weirs. Sharp crested weirs can be fabricated from sheet steel or plastic, and have a sharp top edge. Flow rate can be determined by measuring flow depth behind the weir, often by using a stage gauge or a continuous depth (stage) recorded, and using the depth measurement in an appropriate formula, such as the following:

$$Q = kH^a$$

where:

where:	Q	=	Flow rate
	k	=	Determined by unit conversions and weir dimension (usually supplied by the manufacturer).
	H	=	Height of flow surface above the bottom of the weir notch.
	a	=	1.5 for square notch weir, or 2.5 for v-notch weir.

Weirs can provide more accurate flow measurements than can flow-depth equations. On the other hand, weirs cannot measure flow when they are flooded on the downstream side, or when they are placed in pipes that are full, nearly full or flowing backward. Weirs also are prone to fill with sediment or solids because flow velocity drops to near zero upstream of the weir.

Flumes. Flumes are the most expensive flow gauging device. They are chute-like structures inserted into a channel that force the flow into a known, uniform dimension, allowing a better quantification of flow volumes. These structures include stage gauges for manual calibration of depth-flow volume relationships. Also, the are often associated with stilling wells to allow installation of continuous recorders for water depth/flow volumes and, sometime, continuous sampling devices to assess water quality.

Flumes do not slow flows as much as do weirs and they are not as prone to clogging and silting. Flumes can operate where some downstream flooding occurs, when two depth measurements are collected. They do not work when completely flooded or in backflow conditions. Two types of flumes are most commonly used, the Parshall Flume and the Palmer-Bowlus Flume. The Palmer-Bowlus Flume is easier to install than the Parshall Flume which requires that the floor of the existing pipe or channel be dropped. For this reason the Palmer-Bowlus is commonly used for retrofit and temporary applications.

6.3.3.3 Manual and Automatic Techniques

Manual methods tend to cost less than automatic methods, and they allow additional field observations during storm events. However, they rely more on using well trained field personnel, particularly during storm events and they cannot provide an accurate, continuous flow record. It is often useful to use manual flow measurement as a screening tool when designing a monitoring plan. Monitoring plan designers can use data generated by manual methods to allocate sampling equipment for maximum benefit. Manual techniques are also valuable for checking the operation of automatic flow meters during storm events. Personnel should be extremely careful when using manual methods because conditions during storm events are often dangerous.

In general, manual methods are most useful for instantaneous flow measurement, calibration of other flow measurements, and measurement of flow in small pipes. They are of limited use in monitoring rapidly changing flows because an instantaneous measure can be a poor indicator of the overall flow rate. Because wet weather events occur during storms, use of manual devices can be hazardous to field personnel.

Automatic flow meters can efficiently collect continuous flow rate and volume data, thus reducing the need for direct measurement by field personnel. Data can be transmitted automatically over telephone lines, or accumulate on recording charts or data loggers to be collected later by field personnel. Although automatic flow recorders reduce the need for field personnel during storm events, field work, even during storm events, is required to verify operation of the devices. In addition to retrieving flow data, field personnel must also calibrate meters regularly. Debris can clog or alter meter performance. Debris also can modify flow characteristics near the meter, rendering its readings inaccurate. It is therefore important to operate redundant equipment, or perform periodic manual measurements to verify the performance of automated meters. In fact, when sampling resources are limited, using automated meters in critical areas and manual techniques in less important locations may be cost-effective.

For recent information on continuous monitoring hardware, contact the USGS Hydrologic Instrumentation Facility, Building 2101, Stennis Space Center, MS 39529. Information on automatic gauging stations can be found in the U.S. Geological Survey Water Supply Paper No. 2175 (Rantz, 1982).

Timed Flow. One of the most basic methods of flow measurement is the timing of liquid flow into a container. This technique is accurate if performed carefully; however, it is labor intensive and suitable only for small flows at discharge points. In general, this method has limited use.

Dilution Method. A tracer, such as a fluorescent dye or salt solution, is injected into the flowing water at a known rate. Field personnel measure the diluted tracer in downstream samples using a fluoroscope (for fluorescent dye) or a conductivity meter (for salt concentrations, although laboratory analysis gives more accurate results). The flow rate is calculated from the measured dilution rate.

This method provides reasonable results for instantaneous measurements or measurements during limited periods, but it is not appropriate for the continuous monitoring of varying flows. There are additional drawbacks to this method. Dyes may be adsorbed onto suspended solids or decay with time or because of sunlight, rendering analysis results inaccurate. Industrial waste containing a high conductivity or salt load can confound salt tracer results. Also, laboratory analysis of the salt concentration can delay results of the flow measurement by several days.

Direct Measurement of Velocity and Depth. Flow rates (measured in volume of flow per unit time) can be computed using a current meter to measure flow velocity and a surveying rod to measure depth. Flow velocity (measured in flow per unit time) varies considerably within a channel or pipe, with the slowest flow occurring next to the banks, bottoms, and walls and the fastest flow occurring in the middle, as discussed in Section 6.3.3.2. Depending on the width of the channel, one or more measurements must be taken and adjusted to approximate the average cross-section velocity. Flow rate is calculated by multiplying the computed average velocity by the cross sectional area. While this type of measurement can be dangerous to perform during periods of high flow, it can be useful for measuring flow in open channels and streams and for calibrating other flow monitoring methods.

Flow-Depth Equations. Several equations, including the Manning Equation, allow the user to calculate flow in an open channel or partially full pipe by measuring the depth of flow. The basic parameters used in these equations include surface roughness, diameter, and slope of the pipe. The Manning Equation is

$$V = \left(\frac{1}{n}\right) R^{2/3} S^{1/2}$$

Table 6-4. Manning roughness coefficients for streams (modified from Dunne and Leopold, 1978)

Streambed Characteristics	Coefficient
Winding natural Stream and canals in poor condition, considerable moss growth	0.035
Mountain stream with rocky beds; rivers with variables sections and some vegetation along banks	0.040 - 0.050
Alluvial channels, sand bed, no vegetation	
1. Lower reach	0.025
2. Upper reach	
a) Plane bed	0.011 - 0.015
b) Standing wave	0.012 - 0.016

where:
V = mean flow velocity (m/sec)
n = Manning roughness coefficient: Based on type and condition of channel, typically 0.02 to 0.05 (c.f., Table 6-4)
R = hydraulic radius (m)
S = slope

(With V in feet/second and R in feet, the right side of this equation is multiplied by 1.486.)

Then, flow is computed as follows:

$$Q = VA$$

where:
Q = volumetric flow rate
A = cross-sectional area

This technique can be used with either manual depth measurements or with automated depth-sensing equipment. The automated equipment can compile continuous flow records using the Manning Equation.

The advantage of a flow equation is that it is simple to apply, requires that only depth be measured, and needs limited equipment; however, the method is labor intensive if performed manually. The main disadvantage of these equations is that they rely on average characteristics of the channel network and, thus, are not very accurate. Anomalies in slope, channel shape, or surface roughness often result in errors of 25 percent or greater.

Calibrating the equation to a particular location by using a second more accurate form of flow metering can improve this method's accuracy. First a series of flow velocities and depths are collected using a portable velocity meter and manual depth measurement. Instantaneous flow rates are computed by multiplying the velocities and the depths and by using the flow-depth equation. If results from the flow-depth equation do not agree with the direct measurements, a corrected equation can be developed.

6.3.3.4 Rainfall-Runoff Flow Modeling

When field flow measurements are impractical or impossible, hydrological models can be used to project stream flow volumes. These models require specific information about the watershed upstream from section of channel of concern, including land use, soil classification, slope, precipitation, and temperature. When these data are available, watershed-scale models can do a reasonable job of predicting average stream flows. It is more difficult, however, to predict periods of low and high flow without direct streamflow information.

6.3.3.5 Dispersion Coefficient Measurement

As substances (for example, a pollutant plume) move downstream in a stream or river their concentration decreases through two processes. It is diluted by inflow from tributaries and groundwater, and it disperses or spreads in the direction of the river flow (longitudinally). The longitudinal dispersion coefficient quantifies that spreading and is a necessary input to many river models. The coefficient can be estimated by using general river characteristics, or it can be determined using a dye study. The dispersion coefficient can be estimated using the following equation:

$$E_x = \frac{3.4\times10^{-5}\, U^2 B^2}{H U^*}$$

where: E_x = Dispersion coefficient–mi²/day

U	=	mean river velocity--fps
B	=	mean width--feet
H	=	mean depth--feet
U^*	=	river sheer velocity--fps

and:

$$U^* = \sqrt{gHS}$$

where:

g	=	gravitation acceleration--32 ft/sec^2
S	=	river bottom slope--ft/ft

The dispersion coefficient can be found by releasing a quantity of dye into a river. The progress of the dye is monitored downstream with a fluorometer, and concentrations and times of travel are recorded. The following equation is used to solve for E_x:

$$s_p = \frac{M}{2A\sqrt{\pi E_x t_p}}$$

where:

s_p	=	peak dye concentration
M	=	mass of dye
A	=	cross-sectional area of the stream or river
t_p	=	time of travel for the dye peak

The peak concentration is plotted against the quantity $1/\sqrt{t_p}$. The slope of the plotted line is found from the line. The quantity E_x is then solved for by the following equations (Thomann and Meuller, 1987):

$$S = \frac{M}{2A\sqrt{\pi E_x}} \qquad E_x = \frac{1}{\pi}\left(\frac{M}{2AS}\right)^2$$

6.3.3.6 Flow Monitoring References

Considerable information on flow monitoring is available from additional sources. A brief list of some of the available resources follows:

- ISCO. 1985. Open Channel Flow Measurement Handbook. Lincoln Nebraska. (thorough discussion of the application of primary flow monitoring devices; includes discharge charts for weirs and flumes)

- Parmley, R.O. 1992. Hydraulics Field Manual. McGraw-Hill, New York. (contains useful hydraulic tables and charts; includes weirs, flumes, and pipe flow)

- U.S. EPA. 1981. NPDES Compliance Flow Measurement Manual. Report No. MCD-77. Office of Water Enforcement and Permits Enforcement Division, U.S. Environmental Protection Agency, Washington, DC. (discusses the fundamentals of open channel flow monitoring)

- U.S. EPA. 1988. NPDES Compliance Inspection Manual. Contract No. 68-01-6514, 68-01-7050. Office of Water Enforcement and Permits Enforcement Division, U.S. Environmental Protection Agency, Washington, DC. (contains checklists for the correct application of flow monitoring techniques and equipment)

- U.S. EPA. 1992. NPDES Storm Water Sampling Guidance Document. Report No. EPA 833-B-92-001. Office of Water, U.S. Environmental Protection Agency, Washington, DC. (discusses stormwater sampling, monitoring techniques, and manual methods for flow compositing)

- Water Pollution Control Federation, 1983, Existing Sewer Evaluation and Rehabilitation, Second Edition, Alexandria VA., Manual of Practice No. FD-6. (discusses the fundamentals of open channel flow monitoring)

- Wedepohl, R.E., D.R. Knauer, G.B. Wolbert, H. Olem, P.J. Garrison, and K. Kepford. 1990. Monitoring Lake and Reservoir Restoration. EPA 440/4-90-007. Office of Water, U.S. Environmental Protection Agency, Washington, DC. (contains useful descriptions of many of techniques for assessing flows in of shallow streams and rivers)

6.3.4 Collection and Handling of Water Quality Samples

Samples are analyzed for the parameters identified in the monitoring plan. In most cases, choosing an analytic method is a direct process. Samples collected for compliance with NPDES programs, including TMDL, must be analyzed by a laboratory certified by the appropriate authority, either the State or the U.S. EPA. The laboratories must use analytic techniques listed in the Code of Federal Regulations (CFR), Title 40

Part 136, "Guidelines Establishing Test Procedures for Analysis of Pollutants Under the Clean Water Act." These methods are not, however, required for screening analyses to determine merely the presence of a pollutant. Also, a number of simplified methods may be used to screen samples in the field. It is important to note that data generated by non-approved methods may not be acceptable to EPA. Needs for EPA approval can provide an important determinant in selecting methods for the collection, analysis, and evaluation of water quality samples.

The balance of this subsection notes special considerations regarding those parameters typically sampled and analyzed in the field, including pH, temperature, and dissolved oxygen (DO). Samples collected for bacteria, cyanide, volatile organic compounds (VOC), and oil and grease also require special collection techniques. For a full description of collection techniques other sampling and analyses references should be consulted (e.g., APHA, 1992; U.S. EPA, 1979a).

pH. Levels of pH can change rapidly in samples after collection. Consequently, pH is often measured in the field using a hand-held pH electrode and meter. Electrodes are easily damaged and contaminated, and must be calibrated with a standard solution prior to each use. Field instruments also should be at thermal equilibrium with during calibrations with the solutions being measured and when site measurements are conducted.

Dissolved Oxygen. When multiple dissolved oxygen (DO) readings are required, a dissolved oxygen electrode and meter (EPA method 360.1) is typically used. To obtain accurate measurements, the Winkler titration method should be used to calibrate the meter before and after each day's use. If often is valuable to recheck the calibration during the days of intensive use, particularly when the measurement are of critical importance.

Oxygen electrodes are fragile, subject to contamination, and need frequent maintenance. Membranes covering these probes should be replaced regularly, especially when bubbles form under the membrane, and the electrode should be kept full of fresh electrolyte solution. If the meter has temperature and salinity compensation controls, these should be used carefully according to the manufacturer's instructions.

Several manufacturers offer field kits to measure DO based on a modified Winkler titration. In one method, the user collects a sample, adds three reagent packets, and performs a simple titration. These kits are available for less than $50, while the cost of a DO meter and its probe can range between $1,000 and $2,000.

Bacteria. The maximum recommended holding time for most bacterial tests is 24 hours, although most laboratories recommend that the sample be submitted for laboratory analysis within 6- hours of collections. Bacteria samples are collected using sterile containers or plastic bags, and care must be taken not to let hands contaminate the sample by touching the opening to the sample container or the sample as it enters the container. Samples should generally be held on ice until processed.

Cyanide. Cyanide is reactive and unstable and must be carefully preserved. Any residual chlorine or sulfide present must be removed when the sample is collected. The sample pH is then adjusted to a pH greater than 12.0 using a caustic solution.

VOC. Volatile organic compounds are collected in special glass vials that facilitate analysis. Each vial must be filled so that there is no air space into which the VOCs can volatilize and be lost. A sample is collected by filling both the vial and its lid with water so that a meniscus forms at the water surface. The lid is placed on the vial and screwed tight so no air space forms. If an air bubble is seen in the vial, the sample must be recollected. VOC samples must not be composited in the field, but they can be composited in a laboratory.

Oil and Grease. Oil and grease must be collected by grab sample using a glass jar with a Teflon-coated lid. Samples are preserved by lowering the pH to below 2.0 using a strong acid.

6.3.5 Collection and Handling of Sediment Quality Samples

Receiving water sediments serve as sinks for a wide variety of materials. Nonpoint source discharges typically include large quantities of suspended material that then settle out in sections of receiving waters having low water velocities. Nutrients, metals, and organic compounds can bind to suspended solids and settle to the bottom of a water body when flow velocity is insufficient to keep them in suspension. Contaminants bound to sediments may remain separated from the water column, or they may be resuspended in the water column. Flood scouring, bioturbation (mixing by biological organisms), desorption, and biological uptake all promote the release adsorbed pollutants. Contaminants in sediments are especially available to enter organisms that live and feed in sediment. Having entered the food chain, contaminants can pass to feeders at higher food (trophic) levels and can accumulate or concentrate in these organisms. Humans also can ingest this contaminants by eating fish from waters with contaminated sediments. Sediment depositions also can physically alter benthic (bottom dwelling) habitats to affect habitat and reproductive potentials for many fish and invertebrates. Sediment sampling should allow all these impact potentials to be assessed.

6.3.5.1 Collection Techniques

Sediments samples are collected using hand or winch-operated dredges. Although a wide variety of dredges are available, most operate in a similar fashion:

(1) The device is lowered through the water column by a hand-line or winch.

(2) The device is activated (e.g., released to allow closure) either by the attached line or by a weighted messenger that is dropped down the line.

(3) The scoops or jaws of the device close either by weight or spring action.

(4) The device is retrieved to the surface.

Ideally the device disturbs the bottom as little as possible and closes fully so that fine particles are not lost. Common benthic sampling devices include the Ponar, Eckman, Peterson, Orange-peel, and Van Veen dredges. When information about how chemical depositions and accumulations have varied through time is needed, sediment cores can be collected with a core sampling device. A thorough description of sediment samplers is included in *Macroinvertebrate Field and Laboratory Methods for Evaluating the Biological Integrity of Surface Waters* (Klemm, 1990).

Sediment sampling techniques are useful for two types of investigations related to TMDL assessments: chemical analysis of sediments and investigation of benthic macroinvertebrates communities. In either type of investigation, sediments from reference stations should be sampled so that they can be compared with sediments in the affected receiving waters. Sediments used for chemical analyses should be removed from the dredge or core samples by scraping back the surface layers of the collected sediment and extracting sediments from the central mass of the collected sample. This helps to avoid possible contamination of the sample by the dredge. Sediment samples for toxicological and chemical examination should be collected following method E 1391 detailed in ASTM (1991). Sediments for population analyses may be returned in total for cleaning and analysis, or may receive a preliminary cleaning in the field using a No. 30 sieve (see Section 6.3.7.4 and Klemm, 1990).

6.3.5.2 Sediment Analyses

There are a variety of sediment analysis techniques, each designed with inherent assumptions about the behavior of sediments and sediment-bound contaminants. An overview of developing techniques is presented in *Sediment Quality and Aquatic Life Assessment* (Adams et al., 1992). EPA has evaluated 11 of the methods currently available for assessing sediment quality (U.S. EPA, 1989). Some of the techniques may be helpful demonstrating attainment of narrative requirements of some water quality standards. Two of these common analyses are briefly introduced in the following paragraphs.

Bulk sediment analysis tests for the presence of compounds in sediment material. These tests typically analyze the total concentration of contaminant that are either bound to sediments or present in pore water. Results are reported in milligrams or micrograms per kilogram of sediment material. This type of testing often serves as a screening analysis to classify dredged material. Results of bulk testing tend to over-estimate the mass of contaminants that will be available for release or for biological uptake, because a portion of the contaminants are not biologically available or likely to dissolve.

Elutriate testing estimates the amount of contaminants likely to be released from sediments when mixed with water. In an elutriate test, sediment is mixed with water and then agitated. The standard elutriate test for dredge material mixes 4 parts water from the receiving water body with 1 part sediment (U.S. EPA , 1990). After vigorous mixing, the sample is allowed to settle before the supernate is filtered and analyzed for contaminants. This test was designed to estimate the amount of material likely to enter the dissolved phase during dredging; however, it is also useful as a screening test for determining whether further testing should be performed, and as a tool for comparing sediments upstream and downstream of potential pollutant sources.

6.3.6 Water Quality Sample Preparation and Handling

Sample collection, preparation, preservation, and storage should minimize altering sample constituents. Containers must be made of materials that will not interact with pollutants in the sample and they should be cleaned in such a way that neither the container nor the cleaning agents interfere with sample analysis. Sometimes, sample constituents must be preserved before they degrade or transform prior to analysis. Also, specified holding times for the sample must not be exceeded. Standard procedures for collecting, preserving, and storing samples are presented in 40 CFR Part 136. Useful material is also contained in the *NPDES Storm Water Sampling Guidance Document* (U.S. EPA, 1992a).

Most commercial laboratories provide properly cleaned sampling containers with appropriate preservatives. The laboratories also usually indicate the maximum allowed holding periods for each analysis. Acceptable procedures for cleaning sample bottles, preserving their contents, and analyzing for appropriate chemicals are detailed in various methods manuals, including APHA (1992) and U.S. EPA (1979a). Water samplers, sampling hoses, and sample storage bottles should always be made of materials compatible with the goals of the study. For example, when heavy metals are the concern, bottles should not have metal components that can contaminate the collected water samples. Similarly, when organic contaminants are the concern, bottles and caps should be made of materials not likely to leach into the sample.

6.3.6.1 Sample Preservation, Handling, and Storage

Required sample preservation technique, and maximum holding times are presented in 40 CFR Part 136. Cooling of samples to a temperature of ≤ 4°C is required for most water quality variables. To accomplish this, samples are usually placed in a cooler containing ice or an ice substitute. Many automated samplers have a well next to the sample bottles to hold either ice or ice substitutes. Some more expensive automated samplers have refrigeration equipment requiring a source of electricity. Other preservation techniques include pH adjustment and chemical fixation. When needed, pH adjustments are usually made using strong acids and bases. Extreme care should be exercised when handing these substances.

Bacteria have a short holding time and are not collected by automated sampler. Similarly, volatile compounds must be collected by grab sample as they are lost through volatilization in automatic sampling equipment.

6.3.6.2 Sample Labeling

Samples should be labeled with waterproof labels. Enough information should be recorded to ensure that each sample label is unique. The information recorded on sample container labels should also be recorded in a sampling notebook kept by field personnel. The label typically includes the following information:

- Name of project,
- Location of monitoring,
- Specific sample location,
- Date and time of sample collection,
- Name or initials of sampler,
- Analysis to be performed,

- Sample ID number,
- Preservative used, and
- Type of sample (grab, composite).

6.3.6.3 Sample Packaging and Shipping

It is sometimes necessary to ship samples to the laboratory. Holding times should be checked prior to shipment to ensure that they will not be exceeded. While waste water samples generally are not considered hazardous, some samples, such as those with extreme pH, will require special procedures. If the sample is shipped through a common carrier or the U.S. Mail, it must comply with Department of Transportation Hazardous Material Regulations (49 CFR Parts 171-177). Air shipment of samples defined as hazardous may be covered by the requirements of the International Air Transport Association.

Samples should be sealed in leak-proof bags and padded against breakage. Many samples must be packed with an ice substitute to maintain a temperature of 4°C during shipment. Plastic or metal recreational coolers make ideal shipping containers because they protect and insulate the samples. Accompanying paperwork such as the chain-of-custody documentation should be sealed in a waterproof bag in the shipping container.

6.3.6.4 Chain of Custody

Chain-of-custody forms document each change in possession of a sample, starting at its collection and ending when it is analyzed. At each transfer of possession, both the relinquisher and the receiver of the samples are required to sign and date the form. The form and the procedure document possession of the samples and help prevent tampering with the samples. The container holding samples can also be sealed with a signed tape or seal to help ensure that samples are uncompromised.

Copies of the chain-of-custody form should be retained by the sampler and by the laboratory. Contract laboratories often supply chain-of-custody forms with sample containers. The form is also useful for documenting which analyses will be performed on the samples. These forms typically contain the following information:

- Name of project and sampling locations;
- Date and time that each sample is collected;

- Names of sampling personnel;
- Sample identification names and numbers;
- Types of sample containers;
- Analyses performed on each sample;
- Additional comments on each sample; and
- Names of all those transporting the samples.

6.3.7 Collection and Handling of Biological Samples

Biological assessment and toxicity testing are valuable tools for assessing toxic impacts and may be required by some monitoring programs. This section introduces biological sampling and laboratory toxicity testing.

Aquatic population evaluations used as a monitoring tool provide essential effects information that no other evaluations may provide. This is because analysis of water and sediment only provides concentration information for the analyzed sample(s), and toxicity testing provides only effects information to particular species tested, which may be more or less sensitive than the resident lake fauna. Resident populations and communities of aquatic organisms, in effect, integrates over time all environmental changes affecting these organisms. Thus, the biological community can reveal the consequences of possible cumulative sources of impacts or short-term toxic discharges not represented in the discrete collections of water samples.

Monitoring aquatic communities to assess effects generally requires that the collected data be somehow direction comparable to previous data collected from the same waterbody or from a similar waterbody, i.e., comparable to some reference conditions (cf., Section 7.3). In fact, collecting data that is comparable between systems is essential. Table 6-5 summarizes collection methods and types of information potentially gained through monitoring. The balance of this subsection provides additional introductory information on the four biological groups that are primarily emphasized in monitoring programs. For additional detail beyond the information provided here, the reference documents cited in Table 6-5 should be consulted.

Table 6-5. Introduction to field sampling methods, and sources for more detailed instructions on the proper use of each method

Sample parameter	Information Gained	Methods of collection	References
Phytoplankton Algae	• Chlorophyll a • Community structure • Primary productivity • Biomass • Density	• Plankton buckets attached to a vertical or horizontal tow net (e.g. Wisconsin style net) • Discreet depth samples using VanDorn or Kemmer bottles • Periphytometer	American Public Health Association--APHA, 1992; American Society for Testing and Materials--ASTM, 1991; Lind, 1985; Vollenweider, 1969; Weber, 1973; Wetzel and Likens, 1979
Limitations:	Small organisms can pass through the net, and periphytometers are only good for algae that attach to a substrate.		
Riparian and aquatic macrophytes	• Community structure • Distributions, depth & basin wide • Biomass • Density • Tissue analysis	• Usually qualitative visual assessments • Quantitative assessments use quadrant or line point methods	American Public Health Association--APHA , 1992; American Society for Testing and Materials--ASTM, 1991; Dennis and Isom, 1984; Vollenweider, 1969; Weber, 1973; Wetzel and Likens, 1979
Limitations:	Limited to the growing season for many species.		
Zooplankton	• Community structure • Distributions • Biomass • Sensitivity • Density	• Plankton buckets attached to a vertical or horizontal tow net (e.g. Wisconsin style net) • Discreet depth samples using VanDorn or Kemmer bottles	American Public Health Association--APHA 1992; American Society for Testing and Materials--ASTM, 1991; Lind, 1985; Pennak, 1989; Weber, 1973; Wetzel and Likens, 1979
Limitations:	Small organisms can pass through the net, some zooplankton migrate vertically in the water column, therefore it is possible to miss some species.		

Table 6-5. (continued)

Sample parameter	Information Gained	Methods of collection	References
Benthic invertebrates	• Community structure • Biomass • Density • Distribution • Tissue analysis	• Ponar grab sampler • Eckman dredge sampler Surber • Hess • Kick net or D-ring net • Artificial substrates	American Public Health Association–APHA, 1992; American Society for Testing and Materials–ASTM, 1991; Lind, 1985; Merritt and Cummins, 1984; Pennak, 1989; Weber, 1973; Klemm et al., 1990; Wetzel and Likens, 1979
Limitations:	Some methods are time consuming and labor intensive, some methods are depth restrictive (e.g. can only be used in shallow waters).		
Fish	• Community structure • Distributions, depth & basin wide • Biomass • Density • Bioconcentration • Fecundity	• Electroshocking • Seines • Gill nets • Trawls • Angling • Traps	American Public Health Association–APHA, 1992; American Society for Testing and Materials–ASTM, 1991; Everhart et al., 1975; Lagler, 1956; Nielsen and Johnson, 1983; Schreck and Moyle, 1990; Ricker, 1975; Weber, 1973
Limitations:	Each method is biased to some degree as to the kind and size of fish collected. Some methods are designed for use in relatively shallow water.		

6.3.7.1 Fish

Of all the aquatic organisms, fish generate the greatest public concern over potential impacts. Consequently, fish can be the most important organisms to monitor. Other groups of organisms may be more sensitive, but the effects of toxic chemicals on fish are more readily seen. They include declines of populations and tumor growth on individuals. For fish monitoring programs the principal characteristics of interest include, (1) identification of species present, (2) relative and absolute numbers of individuals of each species, (3) size distributions within species, (4) growth rates, (5) reproduction or recruitment success, (6) incidence of disease, parasitism, and tumors, (7) changes in behavior, (8) taste of fish, and (9) bioaccumulation of toxic constituents.

Common methods of sampling fish include angling, seines, gill and trap nets, and electrofishing. The references shown in Table 6-5 provide guidance on methods used for collection, measurement, preservation, and analyses of fish samples.

6.3.7.2 Phytoplankton

Phytoplankton, free-floating algae, are the principal primary producers, i.e., the primary source of photosynthesis, in most lakes and reservoirs. (Photosynthesis in some shallow lakes is dominated by growths of algae on their bottoms or by larger aquatic plants.) Phytoplankton are useful in monitoring and assessing aquatic conditions because they include many species that are useful indicators of specific water quality conditions, including the presence of some chemicals at concentrations not toxic to other organisms. Phytoplankton also have relatively rapid rates of growth and population-turnover (ca, 3- to 5-days during the summer season).

Laboratory analyses can provide information on the total numerical density (number per water volume) of each phytoplankton taxa identified, the relative abundance by numbers or biomass of these taxa, the presence of or changes in indicator species populations, and, less often, the total biomass of phytoplankton present. Lowe (1974) and VanLandingham (1982) provide useful guides to the environmental requirements and pollution tolerances of two key phytoplankton groups, respectively, diatoms and blue-green algae.

6.3.7.3 Zooplankton

Zooplankton are microscopic animals that have only feeble swimming abilities. As with phytoplankton, zooplankton are sensitive indicators of pollution and can provide useful information about specific toxicants, particularly, in lakes and reservoirs. Zooplankton are often collected by towing a plankton net though a measured or estimated volume of water. Water volume can be measured by a flow meter set into the mouth of the towed net. Water volume also can be estimated using the diameter of the mouth of the net and the measured, or estimated, distance that the net is towed vertically through the water column or horizontally behind a boat.

Laboratory analyses can provide information on taxa present, total numerical density (number per water volume) of each taxon and their abundance by numbers or reproduction or recruitment success, (6) incidence of disease, parasitism, and tumors, (7) changes in behavior, (8) taste of fish, and (9) bioaccumulation of toxic constituents.

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Laboratory analyses can provide information on taxa present, total numerical density (number per water volume) of each taxon and their abundance by numbers or biomass, changes in major populations and indicator species populations, and, less often, the total biomass of zooplankton present.

6.3.7.4 Benthic Macroinvertebrates

Benthic macroinvertebrates, particularly aquatic insects, are an important food source for fish and are widely recognized as useful indicators in aquatic monitoring programs. These organisms can provide valuable information about the presence and nature of toxics in the sediments of lakes and reservoirs. They live on and in sediments where potentially toxic materials can accumulate and, like plankton, include many important indicator species.

The benthic macroinvertebrate community is usually sampled using various dredges, as discussed in Section 6.3.5. Samples are brought to the surface and either preserved in their entirety in polyethylene bags or other suitable containers, or are washed through a fine sieve (e.g., 500-µm screening or a No. 30 sieve) and then preserved in a suitable container (Klemm, 1990).

Laboratory analysis can provide information on taxa present, the total numerical density (numbers per sediment area) of each taxon, relative abundance by numbers or biomass of these taxa, changes in major and indicator species populations, and the total biomass of benthic macroinvertebrates present. A thorough discussion of the use of macroinvertebrates in impact assessment is contained in *Rapid Bioassessment Protocols for Use in Streams and Rivers* (Plafkin, 1989) and *Macroinvertebrate Field and Laboratory Methods for Evaluating the Biological Integrity of Surface Waters* (Klemm, 1990).

6.3.8 Special Considerations for Collecting Wet Weather Data

6.3.8.1 Personnel Health And Safety

Hazardous conditions associated with sampling include:

- Hazardous weather conditions,
- Possible activity in confined spaces,
- Chemical hazards,
- Biological hazards,

- Receiving-water hazards,
- Traffic-related hazards, and
- Physical hazards.

Because the hazards listed above can be life threatening, safety should be of the highest importance. The safety policies developed for a monitoring program should be followed every time that personnel are in the field and emergency procedures should be practiced ahead of time.

Some monitoring programs can require sampling in sewer or other enclosed waste water discharges. Sampling such conditions can be hazardous even in dry conditions. A sewer has limited access, little or no ventilation, little oxygen, and can contain toxic or explosive gases. Rats and snakes may be present, and wastewater flows can contain disease vectors and hazardous chemicals. Physical hazards are often present, including high water flows, slippery surfaces, falling objects, and low overhead clearance.

Sampling storm-related flows presents hazards not encountered during dry weather sampling. Visibility is often poor, and traffic hazards are increased, both in driving to the site and working in street environments. Receiving waters, such as streams and rivers, can be much higher than normal and present a drowning hazard.

Confined spaces. Confined spaces have poor ventilation and limited access. Anytime personnel are below ground, a confined space plan should be in place and followed. A written procedure should be developed and at a minimum, include the following:

- Description of the type of work to be performed;
- Hazards that might be encountered;
- Location and description of the confined space;
- Information on atmospheric conditions in the confined space;
- Personnel training and emergency procedures; and
- Names of sampling personnel.

In general, the atmosphere of a confined space should be checked for oxygen and hazardous gases <u>every time</u> that it is entered. Ventilation equipment should be used and, in some cases, harnesses with lifting winches and self-contained breathing apparatus are necessary. Personnel should be trained in confined entry procedures and should practice confined space rescues at least once per year. In general, sewer systems should not be entered during storm events.

When sampling in such spaces are required, the National Institute of Safety and Health (NIOSH) manual, *Working in Confined Spaces* must be followed. The Occupational Safety and Health Administration (OSH is in the process of finalizing a confined space entry permit system. This should be consulted, when it becomes available.

Traffic Hazards. Since sampling wet weather monitoring crews will be mobilized during storm events, driving conditions will be hazardous and should be taken into account. Low-lying and coastal areas may present additional hazards, especially during extreme storm events. If sampling will take place in traffic areas, warning signs and, possibly, barricades should be used. When appropriate, a police officer or other authorized traffic control personnel should be assigned to control traffic.

Chemical Hazards. Sampling personnel can be at risk from exposure to chemical hazards in wastewater and in sample preservatives. Wastewater may be acidic or caustic, or contain high concentrations of metals and organic compounds. Although wastewater hazards are typically associated with industrial flows, hazardous household wastes including herbicides and pesticides can pose a threat in areas without industrial dischargers. Sampling personnel should wear long-sleeve clothing, gloves and safety glasses. They should avoid skin contact with wastewater and preservatives. Personnel should be trained in first-aid procedures for chemical burns and toxic exposure.

Biological Hazards. Biological hazards in sewers include vermin and disease pathogens. Sampling personnel should be up-to-date with all appropriate vaccinations. They should also avoid skin contact with waste water by wearing gloves and coveralls.

6.3.8.2 Placement of continuous monitoring equipment

Placing continuous monitoring equipment in the field prompt what can be aptly described as a "extraordinary attraction for vandals and the preeminent destructive forces of nature." It often seems that 100-year (greater) flood events have an unexplainable and uncanny nature of accompanying new monitoring programs. Consequently and whenever possible, monitoring equipment should be installed in inconspicuous locations, painted and otherwise treated to camouflage their appearance, placed outside of 100-year flood channels, and anchored or attached (i.e., chained and locked) to immovable objects (e.g., bridge pilings, <u>large</u> trees, <u>heavy</u> anchors). Reenforcing rod driven into and cut near the stream bottom can be used to help anchor submerged monitoring devices that must be placed, for example, in the main channel of streams. (Note: such rods should be removed at the completion of the study.) There should be minimum use of cables or rods placed crosswise of water flows to minimize trapping and accumulating floating debris. Use of deflectors on sampling equipments and anchor devices also can help protect installed equipment from the effects of destructive currents and debris. Often, long-term monitoring programs, depending on the continuous collection of data for their success, should install permanent structures made of concrete and steel to house and protect the monitoring equipment.

6.4 Toxicological and Habitat Evaluation

6.4.1 Toxicological Evaluation Methods

Aquatic toxicity testing has been used widely to assess the toxic effects of chemicals and effluents on aquatic species. The role of toxicity testing in monitoring is, most simply, to determine whether toxic chemicals are present in toxic concentrations. A sufficient number of standardized tests are presently available to accommodate most needs to assess potential problems from toxicants in aquatic environments. For example, toxicity tests developed to evaluating chemically complex mixtures, such as effluents, hazardous wastes, and sediments, are suitable for testing in TMDL studies.

In use, toxicity tests expose selected organisms to a range of concentrations for the test chemical or effluent in solution. Generally, test concentrations bracket those expected to occur in the receiving water. This range of concentrations is often prepared by diluting the solution tested with either ambient receiving water or with standardized laboratory dilution water. Effluent toxicity is assessed by comparing result obtained for acute or chronic effects to test organisms in the various diluted concentrations of test solution to results obtained in reference waters, which do not contain concentrations of the test solution.

The preferred dilution water for the tests is ambient receiving water because it contains chemical substances (e.g., suspended particles, dissolved organic carbon, and various anions and cations) that may increase or decrease the toxicity and bioavailability of test chemicals naturally in the environment. In contrast, use of standardized laboratory dilution water may result in overestimates or underestimates of the toxicity posed by the test chemicals in solution. Sometimes, contaminants or other water quality conditions (e.g., very low ion concentrations) found in these waters may sometimes prevent use of receiving waters. Then, use of uncontaminated surface waters from neighboring lakes or streams that have otherwise similar chemistries may be appropriate. Alternatively, the ambient dilution water's toxicity can sometimes be factored into the assessment of the effluent's toxicity. This can be done by using toxic units to subtract the toxicity of the dilution water from the toxicity of the effluent (U.S. EPA, 1991c).

Acute and chronic tests are the primary methods use to determine aquatic toxicities. Acute tests are "short-term" tests. When sufficient concentrations of toxic chemicals are present, effects occur within a relatively short period of time (within 24 to 96 hours). Some chronic toxicity tests also can be relatively short-term tests (7-10 days, depending on the test species), or relatively lengthy tests as is the case for early life stage tests or life cycle tests (30 days to one year or more, depending on the test species). For acute tests, the endpoint is usually mortality, while endpoints for chronic tests may include mortality, reduced reproduction, or reduced growth. Sources for accepted standardized methods applicable to assessing toxic problems in lakes and reservoirs include the methods developed for whole effluent toxicity testing by EPA (Peltier and Weber, 1985, Weber et al., 1988, 1989) and the methods developed the American Society for Testing and Materials for testing single chemicals, effluents, and sediments (ASTM, 1991).

In general, toxicity tests to evaluate the potential toxicity of ambient water are similar to effluent toxicity tests. Test organisms are placed in undiluted receiving water for a specified period of time. Toxicity is then assessed by comparing the acute or chronic effects observed in test organisms in the ambient water with effects observed in organisms in water collected from reference sites or in laboratory control water. An alternative method is to expose test organisms *in situ* in chambers placed in the receiving water.

One key to successfully using toxicity tests to evaluate possibly contaminated waters or sediments is by aptly defining the appropriate endpoints for the tests. For example, selecting endpoints that measure sublethal effects may tend to give "false positives" because they are extremely sensitive to environmental perturbations. If test endpoints are overly sensitive or not sensitive enough then a true prediction of toxicity based on the test results will likely be incorrect. Some criteria for selecting measurement endpoints for toxicity testing in both the laboratory and *in situ* are presented in Table 6-6. Common toxicity test species are listed in Table 6-7.

Table 6-6. General criteria for measurement endpoints

- Readily measured
- Appropriate to the exposure pathway
- Appropriate temporal dynamics
- Low natural variability
- Diagnostic
- Broadly applicable
- Standard
- Existing data series

6.4.1.1 Good Laboratory Practices

Variability within result produced by standardized toxicity tests is inevitable. Inherently, tests that involve living organisms and, often, complex mixtures of chemicals that have some degree of variability. Explicitly defining specifications for completing toxicity tests can help reduce the variability that may occur through differences in applying procedures for each test method used. Although variability will never be completely eliminated, consistent use of *Good Laboratory Practices* (GLP) by competent personnel, will help to reduce variability.

GLPs are defined standards for laboratories. An integral part of GLPs is Quality Assurance and Quality Control (QA/QC). QA/QC insures that the facilities, personnel, equipment, methods, practices, records and controls conform to the rules and regulations established by the US EPA for toxicity testing. Descriptions of QA/QC requirements for aquatic toxicity testing are found in Peltier and Weber (1985) and Weber et al. (1988, 1989).

6.4.1.2 Toxicity Identification Evaluations

While toxicity tests can identify whether toxicity is present, these procedures do not identify the cause of the toxicity. Procedures developed by the U.S. EPA, called *Toxicity Identification Evaluations* (TIEs), can aid in identifying unknown toxicants(s). TIE procedures combine short term toxicity tests with chemical and physical manipulations of the solution, and analytical chemistry methods to identify toxics. These tests are

Table 6-7. Common species used in aquatic toxicity testing

Common Name	Scientific Classification
Vertebrates	
Frog	*Rana* sp.
Toad	*Bufo* sp.
Coho salmon	*Oncorhynchus kisutch*
Rainbow trout	*Oncorhynchus mysis*
Brook trout	*Salvelinus fontinalis*
Goldfish	*Carassius auratus*
Fathead minnow	*Pimephales promelas*
Channel catfish	*Ictalurus punctatus*
Bluegill	*Lepomis macrochirus*
Green sunfish	*Lepomis cyanellus*
Invertebrates	
Daphnids	*Daphnia magna, D. pulex, D. pulicaria, Ceriodaphnia dubia*
Amphipods	*Gammarus lacustris, G. fasciatus, G. pseudolimnaeus*
Crayfish	*Oronectes* sp., *Cambarus* sp., *Procambarus* sp., *Pasifasticus leniusculus*
Mayflies	*Hexagenia limbata, H. bilineata*
Midges	*Chironomus* sp.
Snails	*Physa integra, P. heterostropha, Amnicola limosa*
Planaria	*Dugesia tigrina*

probably beyond the scope of most TMDL investigations; however, they may prove useful in some studies to help separate impacts from individual sources from other upstream sources of toxicity.

In application, the procedures involve up to three phases, characterization, identification, and conformation. Phase I characterizes potential toxicants in the solution through physical and chemical manipulations. These manipulations are designed to alter or reduce the biological availability of chemicals such as oxidants, cationic metals, volatile organics, non-polar organics, or metal chelates. Successful elimination of these groups in each manipulation or fraction is subsequently tested by additional toxicity tests. For more information refer to the *Methods for Aquatic Toxicity Identification Evaluations: Phase I Toxicity Characterization Procedures* (Mount and Anderson-Carnahan, 1988).

The Phase II TIE procedures are designed to identify the suspected toxicants. Separation and concentration steps are used to distinguish toxic from non-toxic constituents. Separation methods may include simple filtration to more sophisticated separation techniques such as High Performance Liquid Chromatography (HPLC), while identification techniques may include Atomic Absorption Spectrophotometry (AAS) or Gas Chromatography/Mass Spectrometry (GC/MS). Fractions are again tested in toxicity tests to evaluate the success or failure of the methods. For more information refer to the *Methods for Aquatic Toxicity Identification Evaluations: Phase II Toxicity Identification Procedures* (Mount and Anderson-Carnahan, 1989).

The Phase III procedures are intended to confirm the suspected toxicant as the true cause of toxicity. The final confirmation is needed not only to provide data to prove that the suspected toxicants are the cause of toxicity in a series of samples, but perhaps more importantly, to assure that the cause of toxicity is consistent from sample to sample over time (Mount, 1989). Phase III incorporates several approaches to provide a "weight of evidence" that the suspected cause of toxicity is indeed the actual cause of toxicity. The approaches include correlation, observation of symptoms, relative sensitivity, spiking, mass balance estimates and various water quality adjustments. For more information refer to the *Methods for Aquatic Toxicity Identification Evaluations: Phase III Toxicity Confirmation Procedures* (Mount, 1989).

6.4.1.3 Toxicity Reduction Evaluation

The *Toxicity Reduction Evaluation* (TRE) is a compliance mechanism that allows dischargers to identify causes and develop corrective action for toxics as part of the whole effluent toxicity (WET) requirements. TREs are designed to identify the causative agents of effluent toxicity, isolate the sources of the toxicity, evaluate the effectiveness of toxicity control options, and then confirm the reduction in effluent toxicity (U.S. EPA, 1991c). The TRE methodology, although designed for effluent toxicity, is applicable to ambient receiving water toxicity and involves many of the steps to resolve toxic contaminants in these waters.

6.4.1.4 Sediment Toxicity Testing

Sediment toxicity testing assesses the effects of sediment on the growth, reproduction, and survival of aquatic organisms (Thomas et al., 1989). Many sediment toxicity tests are similar to those used for ambient water toxicity tests methods, except that for these the test sediments are often overlaid with water to assess the toxicity. These tests are usually 10-day partial life cycle tests, which measure effects similar to those presented above.

In bulk sediment toxicity testing, a sample of sediment is added to a known quantity of water and sensitive organisms are added. When testing dredged materials, a reference test of sediments collected outside of the dredging area is also performed. Often, an upstream sample site can serve as a reference. Although the impact of a single discharge is often difficult to assess with a high level of statistical certainty, using upstream and downstream stations, a general comparison of the effect of the sediment material on aquatic life can often be made.

The *Draft Ecological Evaluation of Proposed Discharge of Dredged Material into Ocean Waters* (U.S. EPA,, 1990) presents a tiered approach to sediment testing when evaluating ocean disposal of dredged materials. Some of the information also will be useful for general sediment testing. The American Society for Testing and Materials procedures for testing single chemicals, effluents, and sediments (ASTM, 1991) also provides useful standardized methods. And, additional information on sediment testing and toxicity assessment can be found in the text *Sediment Toxicity Assessment* (Burton, 1992).

6.4.1.5 Bioaccumulation

Bioaccumulation testing assesses whether a contaminant accumulates in the tissues of an organism following exposure to collected water or sediment samples. Results from similar analyses for samples exposure to waters or sediments from a reference site are used for comparison. Test organisms are exposed to sample or reference conditions for a specified length of time. After the exposure period, tissue samples are analyzed for contaminants.

Fish and other organisms collected for bioaccumulation analyses either from bioaccumulation studies or from the environment should be handled minimally and only with clean hands or plastic gloves. The sampled fish should not touch metal boats, buckets, or other containers that might contaminate the fish.

For fish specimens collected in the field, the species, length, weight, specimen number, etc., of each sample should be recorded, the fish should be rinsed with water from the receiving water body, then placed individually into a clean polyethylene bag until analysis. Each sample should be double bagged in a second bag, and immediately stored on ice or frozen.

When analyzing metal accumulation in fish, fillets of muscle (meat) and skin, specific organs (most often gills, liver, kidney), or bone may be used. Wearing clean polyethylene gloves, laboratory technicians should take appropriate tissues samples from each fish. Fish should be dissected on clean polyethylene work surfaces. Methods for analyzing metal and organic contaminants in tissues are detailed by U.S. EPA (1991d).

6.4.2 Aquatic Habitat Assessment Procedures

The quality of aquatic habitats is defined through interactions among a diversity of physical, chemical, and biological variables. Cultural developments have and will continue to affect these natural interactions. Our knowledge of the natural range of variation for these variables, how these variables interact, which of these variables are most important, and how our cultural activities affect these natural relationships and impact the quality of salmonid habitats is continuing to grow through extensive research efforts and experience. Many questions remain.

It is clear, however, that successful survival and reproduction by aquatic organisms is broadly defined by the physical structure of the environment, the quality of the surrounding waters, and interactions with other organisms. The principal characteristics of environmental structure that influence population abundance and structure include riparian vegetation, channel morphology, streamflows, deposited sediment, and winter snow and ice accumulation. Important water quality characteristics include suspended sediment, temperature, pH, nutrients, and potentially toxic chemicals. Biological influences involve nutrient and energy cycles, interactions with invertebrates, competition with and predation by other fish, and predation by birds and mammals. Additional, information to aid interpretation of many of these variables individually is presented in Section 7.3.

Once receiving water conditions for these variables that constitute aquatic habitat have been defined, the next step is to evaluate the quality of the habitat based on some measure of the variables. Following this, the variables can be monitored through time to detect changes in habitat quality. Note however, that the definition of *habitat quality* is difficult and equivocal, and that a single definition of quality does not directly apply to all aquatic species or their habitats.

In practice, aquatic habitats are evaluated by first measuring physical, chemical, and biological conditions in the study waterbody of concern. Often similar studies have previously led to defining quantitative relationships (e.g., models based on regression analysis) associating biological conditions in similar waterbodies (e.g., kinds of species, numbers of species, and densities or biomasses of populations) to measures of physical and chemical habitat conditions. When this information is available, physical and chemical conditions measures in the study waterbody of concern can be compared to the modeled conditions to determine whether the biological conditions found during the study conform to expectations based on the existing model relations.

When appropriate information from previous studies is not available for relevant comparisons, new information on which to complete relative assessments of habitat to biological condition relationships can be developed by conducting physical, chemical, and biological studies in one or more reference waters using the same methods used in the study waterbody. Including three, five, or more reference waters in these efforts allow developing new quantitative relationships of habitat to biological conditions relationships in the references waterbodies. These results can then be used to assess habitat and biological conditions in the study waterbody of concern. Use of fewer reference waterbodies allows less rigorous and sometimes only qualitative comparisons to be completed between the reference and study waterbodies. This section briefly introduces sources for information on the appropriate methods and models available to evaluate aquatic habitat quality.

6.4.2.1 U.S. EPA Use Attainability Analyses

EPA's three *Technical Support Manuals: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses* provide useful guidance on methods to measure and assess individual habitat variables and on approaches to integrate these measures to assess habitats in streams and rivers (U.S. EPA, 1983b), estuaries (U.S. EPA, 1984a), and lake and reservoirs (U.S. EPA, 1984b).

Habitat procedures discussed in the technical support manual for flowing waters (U.S. EPA, 1983b) include the USDI Fish and Wildlife Services Habitat Suitability Index Models (see Section 6.3.9.3), a broad selection of diversity indices and measures of community structure, a community recovery index, intolerant species analyses, omnivore-carnivore (trophic structure) analysis, use of reference sites, and important considerations to aid interpretation of these measures with analyzing attainable uses.

For lake and reservoir, the technical support manual (U.S. EPA, 1984b) includes discussions of routine physical habitat measurement techniques; nutrient models (primarily phosphorus loading models); hydrodynamic circulation models; general environmental relationships for phytoplankton, aquatic plants, zooplankton, benthic invertebrates, and fish; chlorophyll and trophic state models; indicator organisms; use of reference site information; aquatic life rating systems; and remediation and restoration techniques.

Biological conditions in estuaries and related near-shore brackish water environments is principally controlled naturally by salinity and water circulation patterns that affect salinity, as discussed in Section 6.2.5.4. U.S. EPA's (1984a) technical support manual for conducting use attainability analyses in estuaries discusses physical processes affecting estuarine processes; estuarine classification systems; models for assessing the influences by physical and chemical conditions on use attainability; the plants and animals specially adapted to allow inhabiting estuaries; habitat suitability models for estuarine species; and approaches for synthesizing and interpreting monitoring results.

6.4.2.2 U.S. EPA Rapid Bioassessment Procedures

EPA's *Rapid Bioassessment Protocols for Use in Streams and Rivers* (Plafkin et al., 1989) provides a systematic approach, including five individual protocols, for integrating field data for habitat conditions with data for benthic macroinvertebrates and fish to assess the quality of flowing water systems. These protocols were developed under the approach that the overall assessment of ecological conditions must first focus on evaluating habitat quality, then aquatic system conditions represented by biological data can be analyzed in light of the defined habitat quality. In other words, habitat, as the principal determinant of biological potential, sets the context for interpreting biological survey or monitoring results; habitat can be used as a general predictor of biological conditions. Routine water chemistry can also help to characterize certain impacts.

In the Rapid Bioassessment Protocols (RBPs) I and IV, habitat evaluations are emphasized in the final assessments because of minimal efforts spent in these approaches in collecting and analyzing biological data. In RBPs II, III, and V, however, the biological evaluations are more rigorous and appropriately take prominence in the evaluations. Habitat assessments support the evaluations in these three protocols. It can identify obvious constraints on the attainable potential of the assessed waterbodies, help in selecting appropriate sampling stations, and provide basic information for interpreting the results from the biological samples. This document provides detailed guidance on methods appropriate for sampling, measuring, and assessing habitat and biological population in flowing waters; and evaluating the relationships between habitat quality and biological conditions. In discussing how to complete integrated assessments, six main topics are considered:

- Evaluating habitat at a site-specific control relative to that at a regional reference.

- Evaluating water quality effects.
- Evaluating biological impairment due to reversible habitat alterations.
- Evaluating an alternative site-specific control station.
- Bioassessment using a site-specific control station.
- Bioassessment using a regional reference.

6.4.2.3 USDI FWS Habitat Suitability Index (HSI) Models

The Habitat Suitability Index (HSI) Models developed by the U.S. Fish and Wildlife Service provide another source of potentially useful information to help interpret relationships between habitat conditions and resident biota in receiving waters. These models are synthesized from facts, ideas, and concepts obtained from the research literature and expert reviews.

For the most part, HSI models are uncalibrated, unverified, and unvalidated. Consequently, developers and users generally acknowledge the limitations on the predictive accuracy of these models. Nonetheless, HSI models provide valuable compilations of the literature for each species on which HSI reports have been prepared. Further, the models themselves target the habitat variables most generally found to have high relationships with the modeled species and the relative weights given each habitat variable included in each model provide a useful guide to the relative importance of each variable to the fish species. In essence, these models provide well founded hypotheses on specific species-habitat relationships. Table 6-8 lists the available HSI models for various aquatic species and sources for copies of these models.

6.4.2.4 Regression based stream habitat-fish models

Much research has focused on relationships among stream fisheries, habitats, and flows. This research is reviewed by Loar and Sale (1981), Fausch et al. (1988), and others. Fausch et al. (1988), for example, reviewed 99 instream flow and habitat quality models. Most of these models were developed using regressions methods to define the relationship of measured habitat characteristics to fish numbers or densities in streams. The authors and other similar reviewers conclude that there are many potential problems in all of these models and the should be used with caution. In some applications of the TMDL framework, there will arise opportunities to use of these models to help define potential uses of the receiving water by fish. When these occasions arise, it is important

Table 6-8. Habitat Suitability Index Models for Freshwater Fish, Saltwater Fish, Aquatic Invertebrates, and Aquatic Vertebrates

Taxonomic Family Common Name	Scientific Name	NTIS Number[1]	USDI FWS Number[2]
FRESHWATER FISHES			
Acipenseridae			
Shortnose Sturgeon	*Acipenser brevirostrum*		FWS/OBS-82/10.129
Atherinidae			
Inland Silverside	*Menidia beryllina*		FWS/OBS-82-10.80
Polyodontidae			
Paddlefish	*Polydon spathula*		FWS/OBS-82/10.80
Salmonidae			
Arctic Grayling	*Thymallus arcticus*		FWS/OBS-82/10.110
Brook Trout	*Salvelinus fontinalis*	PB83-147041/AS	FWS/OBS-82/10.24
Brown Trout	*Salmo trutta*		FWS/OBS-82/10.124
Chinook Salmon	*Oncorhynchus tshawytscha*		FWS/OBS-82/10.122
Coho Salmon	*Oncorhynchus kisutch*		FWS/OBS-82/10.49
Pink salmon	*Oncorhynchus gorbuscha*		FWS/OBS-82/10.109
Chum salmon	*Oncorhynchus keta*	PB86-126802/AS	FWS/OBS-82/10.108
Cutthroat Trout	*Oncholynchus clarki*	PB82-239922/AS	FWS/OBS-82/10.5
Lake Trout	*Salvelinus namaycush*		FWS/OBS-82/10.84
Rainbow Trout	*Oncorhynchus mykiss*		FWS/OBS-82/10.60
Esocidae			
Northern Pike	*Esox lucius*		FWS/OBS-82/10.17
Muskellunge	*Esox masquinongy*		FWS/OBS-82/10.148
Cyprinidae			
Blacknose Dace	*Rhinichthys atratulus*	PB84-129501/AS	FWS/OBS-82/10.41
Longnose Dace	*Rhinichthys cataractae*		FWS/OBS-82/10.33
Common Carp	*Cyprinus carpio*	PB84-150615/AS	FWS/OBS-82/10.12
Common Shiner	*Notropis cornutus*	PB84-128529/AS	FWS/OBS-82/10.40
Creek Chub	*Semotilus atromaculatus*	PB82-239914/AS	FWS/OBS-82/10.4
Fallfish	*Semotilus corporalis*		FWS/OBS-82/10.48
Catostomidae			
Bigmouth Buffalo	*Ictiobus cyprinellus*		FWS/OBS-82/10.34
Smallmouth Buffalo	*Ictiobus bubalus*		FWS/OBS-82/10.13
Longnose Sucker	*Catostomus catostomus*		FWS/OBS-82/10.35
White Sucker	*Catostomus commersoni*		FWS/OBS-82/10.64

Taxonomic Family Common Name	Scientific Name	NTIS Number[1]	USDI FWS Number[2]
Ictaluridae			
Channel Catfish	*Ictalurus punctatus*		FWS/OBS-82/10.2
Flathead Catfish	*Pylodictis olivaris*		FWS/OBS-82/10.152
Black Bullhead	*Ictalurus melas*	PB83-147025/AS	FWS/OBS-82/10.14
Percichthyidae			
Striped Bass (Inland)	*Morone saxatilis*		FWS/OBS-82/10.85
Striped Bass (Coastal)	*Morone saxatilis*		FWS/OBS-82/10.1
White Bass	*Morone chrysops*		FWS/OBS-82/10.89
Centrarchidae			
Black Crappie	*Pomoxis nigromaculatus*	PB82-239930/AS	FWS/OBS-82/10.6
Bluegill	*Lepomis macrochirus*		FWS/OBS-82/10.8
Green Sunfish	*Lepomis cyanellus*		FWS/OBS-82/10.15
Redbreast Sunfish	*Lepomis auritus*		FWS/OBS-82/10.119
Redear Sunfish	*Lepomis microlophus*		FWS/OBS-82/10.79
Largemouth Bass	*Micropterus salmoides*		FWS/OBS-82/10.16
Smallmouth Bass	*Micropterus dolomieui*		FWS/OBS-82/10.36
Spotted Bass	*Micropterus punctulatus*		FWS/OBS-82/10.72
Warmouth	*Lepomis gulosis*		FWS/OBS-82/10.67
White Crappie	*Pomoxis annularis*		FWS/OBS-82/10.7
Clupeidae			
Alewife Blueback Herring	*Alosa aestivalis*	PB85-222693/AS	FWS/OBS-82/10.58
American Shad	*Alosa sapidissima*		FWS/OBS-82/10.88
Gizzard Shad	*Dorosoma cepedianum*	PB86-113586/AS	FWS/OBS-82/10.112
Percidae			
Slough Darter	*Ethoestoma gracile*		FWS/OBS-82/10.9
Walleye	*Stizostedion vitreum*		FWS/OBS-82/10.56
Yellow Perch	*Perca flavescens*		FWS/OBS-82/10.55
SALTWATER FISHES			
Sciaenidae			
Spotted Seatrout	*Cynoscion nebulosus*		FWS/OBS-82/10.75
Southern Kingfish	*Menticirrhus americanus*	PB85-222602/AS	FWS/OBS-82/10.31
Juvenile Atlantic Croaker (Revised)	*Micropogonias undulatus*		FWS/OBS-82/10.98
Juvenile Spot	*Leiostomus xanthurus*	PB83-148197/AS	FWS/OBS-82/10.20
Red Drum (Larval and Juvenile)	*Sciaenops ocellatus*		FWS/OBS-82/10.74

Taxonomic Family Common Name	Scientific Name	NTIS Number[1]	USDI FWS Number[2]
Bothidae			
Southern and Gulf Flounders	*Paralichthys albigutta* *Paralichthys lethostigma*		FWS/OBS-82/10.92
Clupeidae			
Gulf Menhaden	*Brevoortia patronus*	PB83-142513/AS	FWS/OBS-82/10.23
Pleuronectidae			
Juvenile English Sole	*Parophrys vetulus*		FWS/OBS-82/10.133
AQUATIC INVERTEBRATES			
Pink Shrimp	*Penaeus duorarum*		FWS/OBS-82/10.76
Hard Clam	*Mercenaria mercenaria*		FWS/OBS-82/10.77
Red King Crab	*Paralithodes camtschatica*		FWS/OBS-82/10.153
AQUATIC VERTEBRATES			
Slider Turtle	*Pseudemys scripta*		FWS/OBS-82/10.125
Snapping Turtle	*Chelydra serpentina*		FWS/OBS-82/10.141
American Alligator	*Alligatoridae mississippiensis*		FWS/OBS-82/10.136
Bullfrog	*Rana catesbiesna*		FWS/OBS-82/10.138

[1] USFWS out-of-print publications available from the National Technical Information Service (NTIS), U.S. Department of Commerce, 5285 Port Royal Road, Springfield, Virginia 22161.

[2] Single copies available from the U.S. Fish and Wildlife Service, Publications Unit, Arlington Square Building, Mail Stop 725, 18th and C Streets NW, Washington, DC 20240.

to be aware of potential limitations and the necessary cautions when using these models. This subsection reviews the development of these models, their potential shortcomings, and necessary considerations for their subsequent use. This discussion draws heavily from the reviews of Fausch et al. (1988) and Marcus et al. (1990).

In concept, most habitat investigations endeavor to develop techniques and models through which standing crops and/or other measures of biological productivity, generally as pertaining to fish, can be described or predicted using a set of habitat variables. Underlying all of the resulting models is the premise that for each habitat variable, or set of habitat variables, there are definable limits, beyond which conditions become unsuitable for fish. Somewhere between these upper and lower extremes, optimal conditions exist that grade predictably to the unacceptable conditions.

Ultimately, after appropriate relationships are derived, it is hoped that a few well chosen, easily obtained measurements made for a stream can be entered into a model to predict the stream's potential carrying capacity for fish.

Present models, which have been developed using both qualitative and quantitative approaches, include as few as one to as many as 21 input variables. Some variables included in some models are transformed from or derived (recombined) using the originally measured variables. Overall, the variables include details on basin morphology, channel morphology, flow rates, habitat structure, species present, and other physical and chemical measurements.

Among the assorted problems are associated with all existing habitat models, the potentially most damaging in the long term is that no truly standard methods exist for measuring habitat variables. Without use of consistent measurements, it is impossible to compare or synthesize data from different investigators. Moreover, because methods used in collecting data are reflected in the resulting models developed, single data sets can not be explored using otherwise similar models that are based on alternative sampling methods.

Various problems associated with many of the regression based models have statistical bases. First, presentations for most of these models lack information necessary to critically evaluate how the model was statistically selected or how the model may perform in general application. To evaluate the statistical worth of models, sufficient information should be included to enable evaluation (1) the correlation coefficients (r); (2) coefficients of determination (i.e., r^2, R^2, or adjusted R^2: R^2 is the r^2 for multiple regression relationships and adjusted R^2 is corrected for the number of variables included in the equation); (3) standard errors for the regression coefficients (i.e., is the coefficient for each variable included in the model significantly different from 0); and/or (4) the confidence interval for the presented models.

Another statistical problem in many of the models is small sample sizes. This potentially limits the applicability of the model to the limited ranges in habitat variability use to develop the model. If models are used to extrapolate outside these limits, the resulting predictions can be biased and unreliable. Subsequent model calibration and verification, as discussed in Section 7.1, can help to overcoming this limitation.

Further, errors associated with measuring the various habitat variables have rarely been evaluated during model development. If measurements upon which the model is based are biased, the model will yield similarly biased predictions. Most models have not been tested with data that was not used in developing the models. Thus, we generally know little of the overall realism, precision, or generality of the models.

Various potentially unreasonable assumptions about habitat relationships are also implicit in many of these models. For example, the U.S. Fish and Wildlife Service's Instream Flow Incremental Methodology (IFIM) include potentially erroneous assumptions that (1) fish primarily respond to average water velocities at some defined depth (e.g., 0.6 of the depth below the surface); (2) stream depths, velocities, and substrates are not related to each other (i.e., an underlying assumption for regression analysis is that independent variables are uncorrelated); and (3) large amounts of suboptimal habitat are equivalent to a small amount of optimal habitat. Yet, studies show that fish respond more to flow difference in microhabitats; that stream depths, velocities, and substrates are often highly correlated; and that suboptimal habitats can often be uninhabitable.

Most models include at least one additional and perhaps also unreasonable assumption about the relationship between measured habitat variables and measured fisheries densities or biomasses used to derive the models. That is, it is often assumed that the measured density or biomass for fish is at the carrying capacity for the habitat, and that this carrying capacity is defined by those physical and chemical conditions measured in the habitat. This assumption precludes such effects as predation (including fishing), or competition as having any potential influence on the population.

Finally, while a plethora of models relating stream habitat to fish are indeed available, few models, ir any, currently available has been verified as reliably predicting effects of stream flow alterations on standing crops of fish.

6.5 Data Management Techniques

6.5.1 Data management

All monitoring data should be organized and stored in a form that allows ready access. The voluminous and diverse nature of the data, and the variety of individuals who can be involved in collecting, recording and entering data, can easily lead to the loss of data or the recording of erroneous data. Lost or erroneous data can severely damage the quality of monitoring programs. A sound and efficient data management program for a monitoring program should focus on preventing such problems. This requires that data be managed directly and separately from the activities that use them.

Data management systems comprise technical and managerial components. The technical components involve the selection of appropriate computer equipment and software and the design of the database, including data definition, data standardization, and a data dictionary. The managerial components include data entry, data validation and verification, data access, and methods for users to access the data.

To ensure the integrity of the database, it is imperative that data quality be controlled from the point of collection to the time the information is entered into the database. Field and laboratory personnel must carefully enter data into proper spaces on data sheets and avoid transposing numbers. To avoid transcription errors, entries into a database should be made from original data sheets or photocopies. As a preliminary screen for data quality, the database design should include automatic range-checking of all parameters. Values outside the defined ranges should be flagged by the program and immediately corrected or included in a follow-up review of the entered data. For some parameters, it might be appropriate to include automatic checks to disallow duplicate values. Preliminary database files should be printed and verified against the original data to identify errors.

Additional data validation can include expert review of the verified data to identify possible suspicious values. Sometimes, consultation with the individuals responsible for collecting or entering original data is required to resolve problems. After all data are verified and validated, they can be merged into the monitoring program's master database. To prevent loss of data from computer failure, at least one set of duplicate (backup) database files should be maintained at a location other than where the master database is kept.

6.5.2 Record Keeping Requirements

NPDES permits typically require that all data collected to comply with permit conditions be retained for at least five years from the date of sampling, recording, or permit application. Information that must be retained includes calibration and maintenance records, strip chart recordings from continuous monitoring equipment, reports, and data records. Data records should include the following information:

- The date, location, and time of sampling or measurements;
- The persons who performed the sampling;
- The date analyses were performed
- The persons who performed the analyses;
- The analytic techniques used; and
- The results of the analyses.

6.5.3 GIS and Data Analysis in TMDL Planning

6.5.3.1 Introduction to GIS

TMDL development frequently requires analysis of data over broad spatial scales (e.g., across a watershed), utilizing land attributes such as basin size, land use types, slopes, and vegetation cover. A geographic information system (GIS) can be an integral part of this process, depending on the phase of TMDL development, the type of loading model used, and the resources available. GIS is being used in a growing number of fields, and it is likely that those developing TMDLs will have access to a GIS currently or in the near future. The application of GIS in the TMDL process can include data management, the presentation of data, and the interpretation or analysis of data. The most significant use of GIS is as a sophisticated and powerful decision support tool to analyze and manipulate data. GIS allows the examination of relationships among data that would otherwise be too complex or cumbersome to discern. The TMDL process is, by necessity, one that must take a "big picture" view of water quality problems. GIS is well suited to this scale of analysis, and there are many possible uses for GIS in this process.

In addition to representing data in the form of maps, many GIS provide a set of tools for the analysis of data. Analysis may be in the form of overlay operations (i.e., analogous to using mylar map overlays), aggregating data to a desired level, or statistical functions. It is also possible to implement algorithms and automate procedures in many GIS using a "macro" programming language. The type of GIS analysis being utilized will correspond to the level of detail inherent in the different phases of TMDL development. In following with the rest of this guidance document, this section will address the use of GIS for: 1) gaining an understanding of water quality issues in an area, 2) performing a scoping or screening analysis to prioritize issues, and 3) detailed modelling relevant to TMDL development. While the first two purposes of GIS will be discussed briefly, the focus of this section will be on the use of GIS in relation to modelling for the TMDL process.

6.5.3.2 Use of GIS

At a general level, a GIS is a system for the collection, storage, manipulation, and display of geographically referenced data. It is a set of tools that has evolved from various fields (e.g., cartography, geography, and computer science), and there are several different GIS currently available. There are two basic types of GIS: a raster-based system or a vector-based system. A raster-based system represents data in a grid format, in which each cell of the grid has a single value for the attribute in question. A vector-based GIS maintains the topological relationships between geographical objects, which are stored as points, lines, or polygons. A discussion of the technical details of these two types of GIS is out of the scope of this document, and the reader should refer to a GIS text for greater detail. Historically, vector-based GIS have tended to be used in the planning field, and raster-based GIS have been used for scientific analyses. Over time, this distinction has become much less clear, and there are now GIS that can handle data in both raster and vector formats.

GIS software can run on a variety of hardware platforms, ranging from personal computers to powerful workstations. The more sophisticated GIS packages offer a great deal of functionality, but this is often offset by the level of complexity in using the GIS. Providers of GIS are currently addressing this issue, and more menu-driven, user-friendly GIS are becoming available. Regardless of the GIS in question, it can aid in the first two phases of the TMDL process (i.e., gaining an understanding of water quality issues in an area and performing a scoping or screening analysis to prioritize issues). Using GIS in combination with water quality models, or implementing the model through GIS itself, will depend much more on the specifics of the GIS package and the model chosen.

The TMDL process has evolved from examining individual stretches of rivers or portions of basins to include entire watersheds. Because there can be several water quality problems that require attention in a particular watershed, gaining an understanding of the water quality issues in a watershed is frequently the starting point. Given the appropriate data, or even data of limited detail (e.g., approximations of slope and crude delineation of land uses), GIS can quickly supply information to decision-makers about the nature of the problems at hand. If there is a particular water quality issue being addressed, GIS can be used to gain an understanding of the contributing factors and to indicate the next step in addressing the issue. Used in this way, GIS can provide a context for further decisions and analyses in the TMDL development process.

Performing scoping and screening analyses with a GIS can improve and speed up this step in TMDL development, although the data requirements tend to be greater. In general, data gathering and data input are the most time- and effort-intensive aspect of a project using GIS technology; however, once the necessary data are in a GIS, it is relatively simple to use these data in a meaningful way. For example, by assigning nutrient or sediment export coefficients to different land uses, GIS could be used to screen out priority areas (e.g., those near water bodies) or to estimate the amount of pollutant reduction that is necessary (e.g., across agricultural regions). Based on the spatial characteristics of an area, GIS can help to break a basin up into hydrologic sub-units or to isolate those with specific attributes. Many other analyses, such as buffering riparian areas and accounting for distances between stream gauges, are also possible through GIS. These types of analyses do not require extensive GIS training to accomplish, and their applicability is obvious. Without GIS, many analyses would require too much time or would not be possible without a great deal of experience.

After prioritization of water quality issues, GIS can facilitate detailed modelling relevant to the TMDL process. Loading can be estimated by models that range in complexity from an equation implemented on a hand-held calculator to sophisticated, computer-based systems. As was discussed earlier, a GIS is usually a raster- or vector-based system. The choice of a water quality model, and its corresponding requirements, may largely determine whether a raster- or vector-based GIS is preferred. On the other hand, much of the data required to develop a TMDL may already be in an existing GIS. If this is the case, the availability and format of the data itself may help in choosing among the many water quality models in existence.

6.5.3.3 Linkages between GIS and TMDL Models

There are many ways of incorporating GIS into the modelling of water quality. Some models have "seamless" linkages (i.e., transparent to the user) with certain GIS packages, but most do not. Frequently, a GIS can provide summary statistics and other input parameters, depending on the model used and the data available. Therefore, the link between GIS and TMDL models can be automated to different degrees, or not at all. When compared to providing parameter data manually (e.g., aggregating classes of data or calculating the coincidence of several different attributes), GIS is of great value for this task. GIS is also well-suited to displaying the output of models, provided that the output is geographically referenced in some way, and visualizing various mitigation scenarios can aid in their evaluation.

Table 6-9 summarizes many of the data inputs for watershed-scale models. It is clear that commonly available GIS data, which are stored in "layers" that represent different themes or land attributes (e.g., land use types, soils, vegetation cover, and slope), can be manipulated to meet many of the data needs listed in this table. Again, note that the linkage between a GIS and a particular model can be automated to various degrees, and it may also go the other way (i.e., output from a model into a GIS to display modelling results). Extracting input parameters from a GIS may involve several GIS operations, and the subsequent input of these parameters into a model may require that they be entered manually. In contrast, resources and programming expertise may be available so that this is accomplished through a few commands. Regardless of how sophisticated this link may be, GIS has been able to improve and facilitate the water quality modelling process.

Table 6-9. **Input Data Needs for Watershed Models**

1. System Parameters:
Watershed size
Subdivision of the watershed into homogenous subareas
Imperviousness of each subarea
Slopes
Fraction of Impervious areas directly connected to a channel
Maximum surface storage (depression plus interception storage)
Soil characteristics including texture, permeability, erodibility, and composition
Crop and vegetative cover
Curb density or street gutter length
Sewer system or natural drainage characteristics
2. State Variables
Ambient temperature
Reaction rate coefficients
Adsorption/desorption coefficients
Growth stage of crops
Daily accumulation of rates of litter
Traffic density and speed
Potency factors for pollutants (pollutant strength on sediment)
Solar radiation (for some models)
3. Input Variables
Precipitation
Atmospheric fallout
Evaporation rates

Source: U.S. EPA, 1992a (after Novotny and Chesters, 1981)

The U.S. Army Corps of Engineers Hydrologic Engineering Center (HEC) has pursued the integration of hydrologic modeling and GIS extensively. A recent review of GIS applications in hydrologic modeling is provided by DeVantier and Feldman (1993), ASCE Journal of Water Resources Planning and Management (Vol. 119, No. 2, March/April 1993 contains a special issue focusing on GIS applications and covering topics such as nonpoint pollution and urban stormwater management.

6.5.3.4 Other Uses of GIS in the TMDL Process

As was mentioned above, some water quality modelling has been implemented through GIS alone, without the use of a separate model such as those described above. This use of GIS may require significant programming and GIS expertise, but the resulting models can be specific to the geographic characteristics of the area and the needs of those performing the analysis. Linking GIS to other types of models (e.g., groundwater flow models) can also yield insights to water quality that would not otherwise be possible. These approaches may be most appropriate in areas where water quality analyses cannot be performed using existing water quality models. The following two case studies illustrate these uses of GIS.

"Land Use Change and Impacts on the San Francisco Estuary: A Regional Assessment with National Policy Implications"
Water quality modelling over very large areas, such as that contributing to major estuaries, may require model implementation through GIS alone. McCreary et al. (1992) describe the use of GIS in modelling changes in water quality and losses of different wetland types, given development scenarios in the areas affecting the San Francisco estuary. Runoff was estimated from local precipitation averages and the imperviousness of different land use types. Contaminant concentrations, which included metals, nutrients, biological oxygen demand, and total suspended solids, were derived from national and local studies.

"Analyzing Septic Nitrogen Loading to Receiving Waters: Waquoit Bay, Massachusetts"
This case study by Sham et al. (1993) shows how GIS can be linked to a groundwater model to analyze changes in water quality over time. By incorporating the spatial and temporal characteristics of development in the watershed, the accuracy of nutrient loading estimates can be improved. This study indicates how important a role groundwater can play in water quality problems and how a long-term view must be taken in their solution.

One way to ensure the success of a GIS operation is to integrate it into as many tasks as possible. In addition to water quality modelling, GIS can also play a part in the management of data related to water quality and the TMDL process. Because these data have a spatial component, GIS can be an excellent facility for their management, analysis, and display, as the next two studies indicate.

"Application of a GIS to a Water Regulatory Permit Inventory"
This case study (Crowell, 1989) describes how the Southwest Florida Water Management District has used GIS to inventory and update permit information, instead of performing the process on USGS topographic maps. Regulatory permit

mapping (and the storage of relevant permit data) via GIS has not only aided in the evaluation of permit applications, but it has streamlined data management and limited access to only the most up-to-date mapped permit information.

"Geographic Information Systems as a Tool in Water Use Data Management" Schoolmaster and Marr (1992) discuss some of the possible uses of GIS in the management of water use data, incorporating spatial and temporal dimensions for both surface and groundwater. While this study does not specifically address water quality issues, it does discuss concepts for data storage, use, and display that could be useful for managing data in the TMDL process.

6.6 Quality Assurance and Quality Control—(QA/QC)

QA/QC procedures are essential to ensure that data collected in environmental monitoring programs are useful and reliable. *Quality assurance* refers to programmatic efforts to ensure the quality of monitoring and measurement data. *Quality control,* which is a subset of quality assurance, refers to the routine application of procedures designed to obtain prescribed standards of performance in monitoring and measurement. This section introduces procedures for sample and analytic quality control and for field quality assurance. U.S. EPA (1985) and Plafkin et al. (1989) provide additional information for defining QA/QC program plans as part of the monitoring design process, including the development of QA/QC adequate program descriptions, integration QA/QC programs with monitoring program plans, EPA's responsibility in the QA/QC process, and the importance of QA/QC in the bioassessment process.

6.6.1 Sample and Analytic Quality Control

The following techniques are useful in assessing sampling and analytic performance (see also U.S. EPA, 1979b):

- *Duplicate samples.* Duplicate samples collected at selected locations using two sets of field equipment or by grab sampling provide a check for precision in sampling equipment and techniques.

- *Split samples.* Single samples split and analyzed separately check for variation in laboratory method or between laboratories. Samples can be split and submitted to a single laboratory or split between laboratories.

- *Spiked samples*. Introduced a known quantity of a substance into a volume of distilled water and analyzing for that substance provides a check of the accuracy of laboratory and analytic procedures.

- *Reagent blanks*. Preserving and analyzing a quantity of distilled water in the same manner as environmental water samples can indicate contamination caused by sampling and laboratory procedures. Normally the value of pollutant measured in the blank is subtracted from the values obtained by analyzing environmental samples.

6.6.2 Field Quality Assurance

Errors or a lack of standardization in field procedures can significantly decrease the reliability of environmental monitoring data. A quality assurance plan for field measurement procedures and equipment should at a minimum include the following elements:

- Identification of the analytic method, including special handling procedures. In most cases analytic methods will be specified by regulation.

- Allocation of field and laboratory analyses to quality control.

- Procedures for recording, processing, and reporting data.

- Procedures for and records of maintenance and calibration of field instruments.

- Evaluation of the performance of field personnel.

It is important that quality procedures be followed and regularly examined. For example, field meters can provide erroneous values if they are not regularly calibrated and maintained. Reagent solutions and probe electrolyte solutions have expiation periods and should be refreshed periodically.

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Chapter VII. Model Implementation and Analysis

Interpretation

Purpose: Previous chapters have discussed the identification and selection of simulation models for the estimation of TMDLs, and the development of monitoring plans and data collection in support of modeling. This chapter discusses various issues relating to the implementation of models, including the integration of modeling and monitoring. Chapter 8 then gets to the basics of using modeling results to actually perform TMDLs. Parts of the discussions contained in this chapter were adapted from guidance for modeling and monitoring of combined sewer overflows (CSOs) presently under development for OWEC.

7.1. Model Calibration and Validation

Once a model has been selected, model parameters are generally adjusted or fine-tuned to reproduce observations at a particular site. The performance of the adjusted model should then be checked for ability to predict on a separate set of data. These two processes are known as model *calibration* and *validation*.

While the process in which the calibrated model is tested on an independent set of data is commonly referred to in the modeling literature as "verification", it should be noted that this usage is not really correct from a systems science perspective (see, for instance, Nix et al., 1991). "Verification" has the implication of proving something to be true. For environmental models we do not expect to be able to show that calibrated model parameter values are exactly true and correct; instead, the best we can expect is to show that the calibrated model does an adequate job of prediction. This is more properly a test of the validity (rather than verity) of the calibration. Nix et al. suggest that the term verification should be reserved for another step in the process: ensuring that the computer code performs as expected. Therefore, and in accordance with recent guidance from the Office of Water (Martin et al., 1990), we will refer to the process of testing the calibrated model as *validation*.

7.1.1 Role of Calibration and Validation

The purpose of model calibration and validation is to take a generalized numerical construct or model, and, by specifying the 'right' parameter values (and using the 'right' model subroutines) turn it into a site-specific predictive tool. Calibration and validation are thus integral parts of the model development process: When an established computer model is used, calibration and validation can be thought of as the finishing touches required to create the site-specific model (see Figure 7-1).

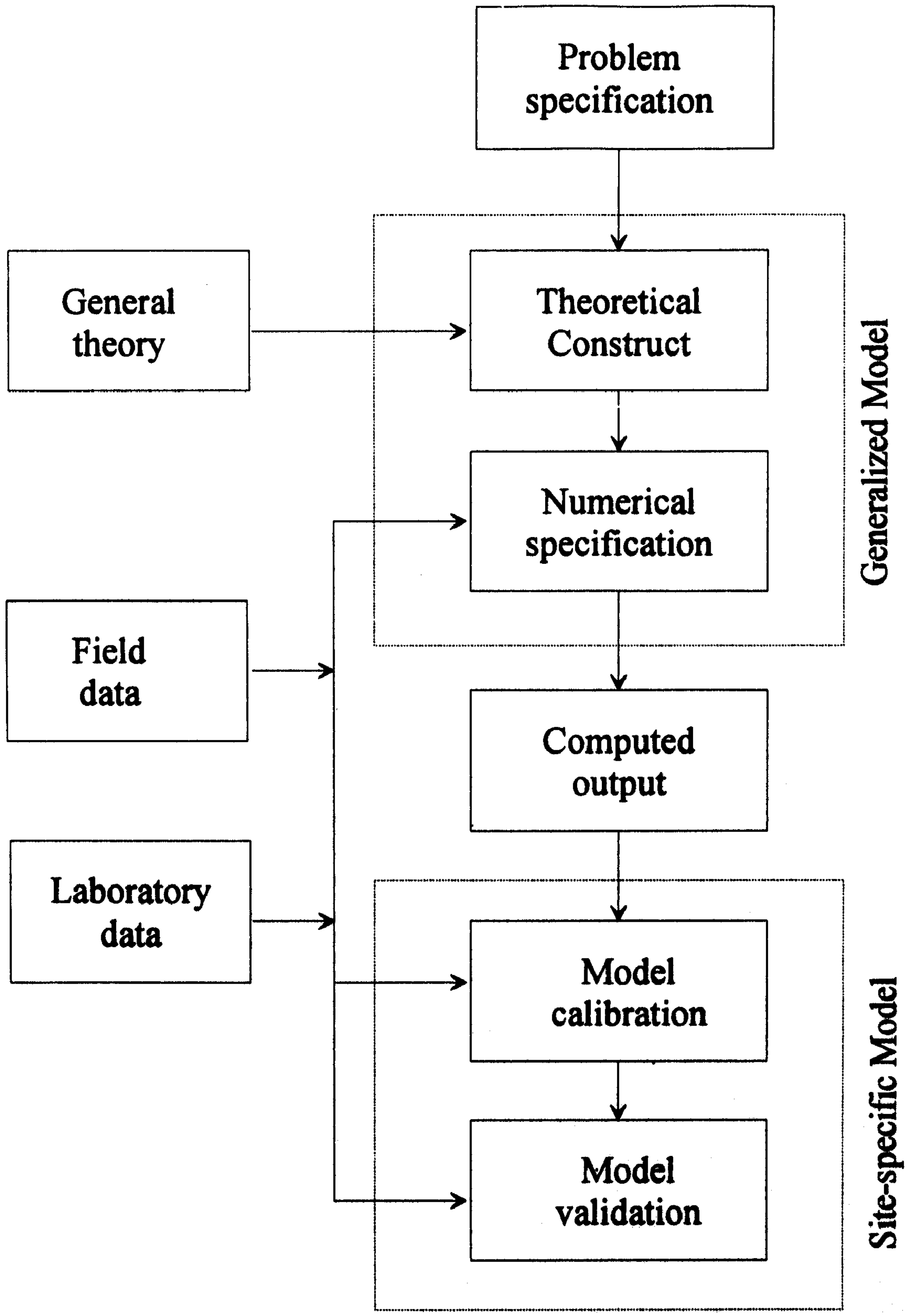


Figure 7-1. Principal Components of Modeling Framework
(after Thomann, 1980)

A TMDL developer could run a loading or receiving water model without calibration for screening purposes, essentially to provide an informed "best guess" of impacts, based in part on generic, rather than site-specific conditions. However, the uncalibrated result is still only a guess, without confirmation. To use model simulation results in support of a particular management alternative it is necessary to provide evidence that the guess is a reasonable one. This is accomplished through the process of model calibration and validation, which consists of fine-tuning the model to the site, and proving that the results are reasonable (and measuring just how reasonable they are). The process can be thought of as establishing the model's credibility as a witness.

The needs for, and functions of model calibration and validation are succinctly summarized in Martin et al. (1990), which also provides extensive guidance on the calibration/validation procedure as applied to WLA modeling of estuaries:

> While models can be run with minimal data, their predictions are subject to large uncertainty. Models are best operated to extrapolate from existing to future conditions, such as in the projection of conditions under anticipated waste loads. The confidence that can be placed on those projections is dependent upon the integrity of the model, and how well the model is calibrated to that particular [waterbody], and how well the model compares when evaluated against an independent data set (to that used for calibration).
>
> Model calibration is necessary because of the semi-empirical nature of present day...water quality models. Although...formulated from the mass balance and, in many cases, from conservation of momentum principles, most of the kinetic descriptions in the models that describe the change in water quality are empirically derived. These empirical derivations contain a number of coefficients and parameters that are usually determined by calibration using data collected in the [waterbody] of interest.
>
> Calibration alone is not adequate to determine the predictive capability of a model... To map out the range of conditions over which the model can be used to determine cause and effect relationships, one or more additional independent sets of data are required to determine whether the model is predictively valid. This testing exercise...defines the limits of usefulness of the calibrated model. Without validation testing, the calibrated model remains a description of the conditions defined by the calibration data set. The uncertainty of any projection or extrapolation of a calibrated model would be unknown unless this is estimated during the validation procedure.

Model calibration and validation interact strongly with both the data-collection and model-selection activities of TMDL development covered in of this guidance. That is, model calibration/validation requires data, and, conversely, the design of effective monitoring plans can be guided by preliminary modeling results. Model development and data collection thus interact in an iterative fashion (Figure 7-2). This will be true for any effort involving application of other than the most simple models. However, it is particularly appropriate to the development of TMDLs using the phased approach, where ample time will be available for the collection of additional data and refinement of modeling approach.

EPA's *Compendium of Watershed-Scale Models for TMDL Development* (Shoemaker et al., 1992) summarizes the calibration and validation process as follows (we have substituted "validation" for "verification" for consistency in usage with the current document):

> Calibration involves minimization of deviation between measured field conditions and model output by adjusting parameters of the model (Jewell et al., 1978). Data required for this step are a set of known input values along with corresponding field observation results. The results of [a] sensitivity analysis provide information as to which parameters have the greatest effect on output. For the best results, CSO models should be calibrated during storm events as opposed to dry flow periods (WPCF, 1989).
>
> [Validation] involves the use of a second set of independent information to check the model calibration. The data used for [validation] should consist of field measurements of the same type as the data output from the model. Specific features such as mean values, variability, extreme values, or all predicted values may be of interest to the modeler and require testing (Reckhow and Chapra, 1983). Models are tested based on the levels of their predictions, whether descriptive or predictive. More accuracy is required of a model designed for absolute versus relative predictions. If the model is calibrated properly, the model predictions will be acceptably close to the field observations.
>
> [M]ost models are more accurate when applied in a relative rather than an absolute manner. Model output data concerning the relative contribution...to overall pollutant loads is more reliable than an absolute prediction of the impacts of one control alternative viewed alone. When examining model output . . . it is important to note three factors that may influence the model output and produce unreasonable data. First, suspect data may result from calibration or [validation] data that are insufficient or inappropriately applied. Second, any given model, including detailed models, may not represent enough detail to adequately describe existing

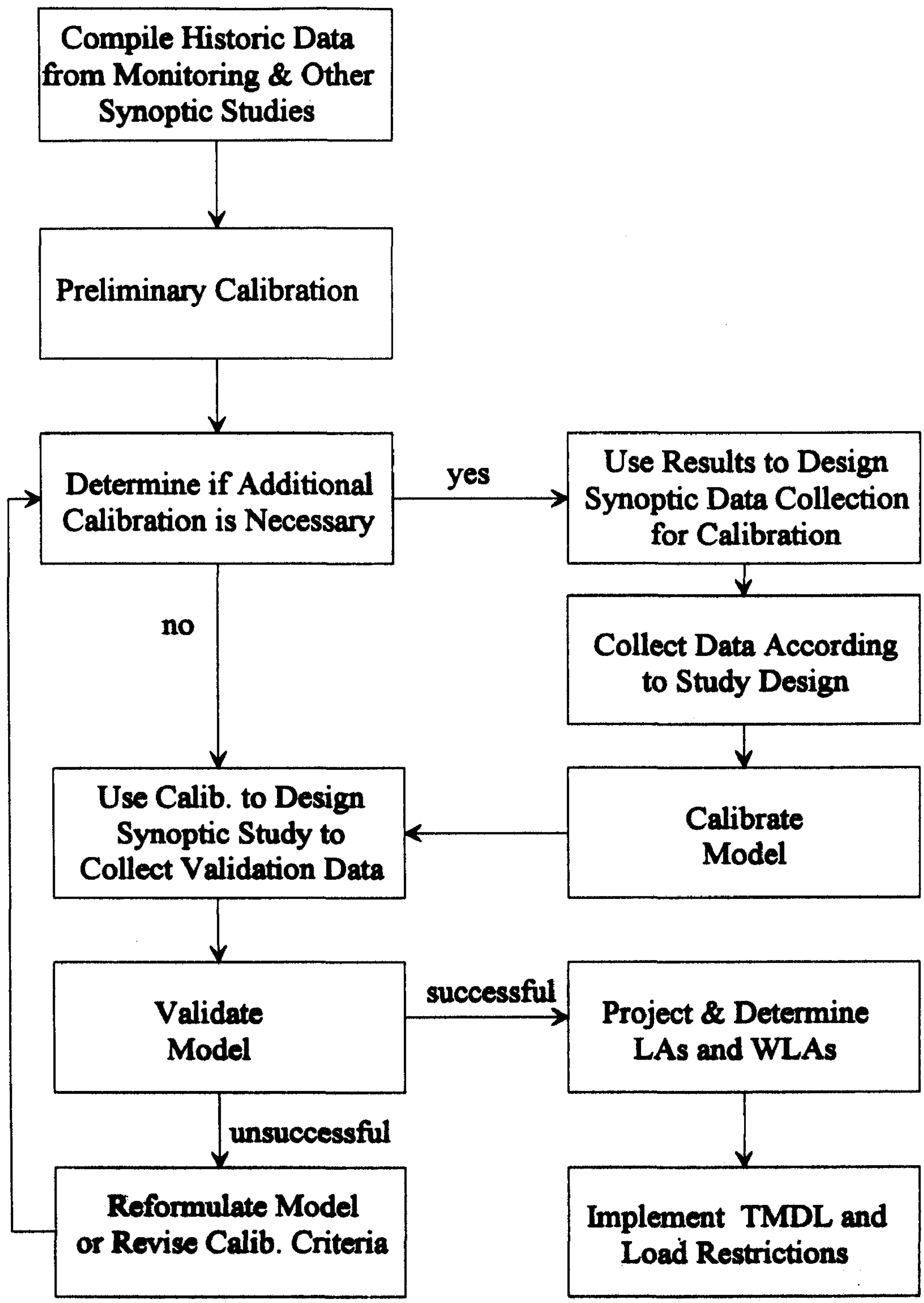


Figure 7-2. Relationship Between Data Collection, Model Calibration, Validation, and TMDL Procedures (adapted from McCutcheon et al., 1990)

conditions and generate reliable output. Finally, modelers should remember that all models have limitations and the selected model may not be capable of simulating desired conditions. Model results must therefore be interpreted within the limitations of their testing and their range of application. Inadequate model calibration and [validation] can result in spurious model results, particularly when used for absolute predictions. Data limitations may require that model results be used only for relative comparisons.

The basic concepts of model calibration on one synoptic data set and validation on a second, independent synoptic data set are summarized in Figure 7-3.

7.1.2 Assessing Model Goodness of Fit

The calibration and validation process requires a series of judgements as to just how well model performance matches observations. The question of how best to estimate the goodness-of-fit of model output to validation data has received considerable attention in the literature. Common practice has been to use a combination of modeler judgment and graphical analysis to assess the adequacy of a model. However, statistical evaluation can provide a more rigorous and less subjective approach to validation. This section discusses several quantitative statistical techniques for assessing goodness of fit for model predictions. Reckhow et al. (1990) provide a more detailed analysis of the relative merits of these procedures.

Assessing Goodness of Fit: *t*-Test

Uses: The t-test is used to compare a sample mean to a specified value. The t-test may be used to:

(1) Compare monitoring sample mean to a model prediction;

(2) Test the magnitude of model residuals against a specified target magnitude.

(3) Compare a distribution of modeling predictions (such as events per year obtained from a continuous simulation, or point estimates from a Monte Carlo simulation) to a specified or observed value.

Implementation: The null hypothesis for the t-test takes the form

$$\bar{x} = m$$

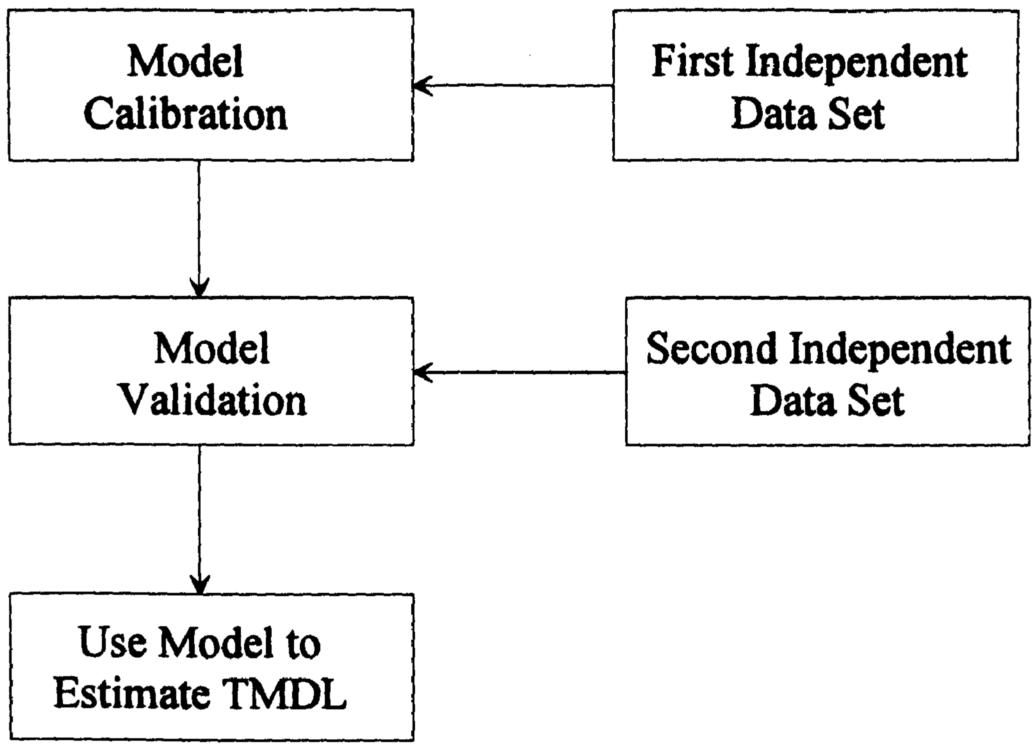


Figure 7-3. Model Calibration and Validation Procedure
(adapted from McCutcheon et al., 1990)

where $\bar{x}$ represents the mean of the sample or model residuals and m is a fixed value. That is, the test is set up to examine if the sample mean, or the average model residual, both given by $\bar{x}$, can be determined to be different from the value m at a given level of statistical significance. It can also be thought of as testing whether the bias is significantly different from zero, where bias is the average value of the residuals.

To test the null hypothesis we require a test statistic. The test statistic in this case is the t-statistic, given by

$$t = \frac{\bar{x} - m}{s/\sqrt{n}}$$

where s is the sample standard deviation and n is the sample size. For use (1) above, m would be the model prediction; for use (2), m would be the pre-specified target absolute error of the model predictions, while for use (3), m would be a specified target value or an observation.

The hypothesis is tested by comparison to the widely tabulated values of the t distribution.

Assumptions and Limitations: The t-test assumes samples come from a Normal distribution, variances are constant across distributions, and observations are independent. The method is relatively robust to violations of the first two assumptions. However, violations of the independence assumption can cause erroneous results.

Extensions: In some cases of model validation, the t-test may be seen as too strict a measure. Instead of a null hypothesis that the model predicts exactly, within a given level of statistical certainty, we might instead wish to ask whether or not the model can predict within a given factor f of the true value. Extensions of the t-test to address this situation are provided by Parrish and Smith (1990).

Assessing Model Goodness of Fit: Regression Analysis

Uses: A natural and intuitive method for quantitative analysis of model goodness of fit would seem to be the formation of a linear regression between model predictions and observations. The basic linear model is

$$y = \alpha + \beta x$$

where y represents model predictions and x represents calibration observations. If the model provides a good fit, we would expect the slope of the regression, β, to be close to 1, while the intercept, α, should be zero. We could then test whether the values of α and β show a significant departure from these values: If the intercept is not equal to zero, this indicates a consistent bias, while if β is not equal to 1 this indicates that model error increases or decreases with the magnitude of the true value, suggesting inadequate calibration. However, as warned by Reckhow et al. (1990), interpretation of the usual regression model statistics (standard error and R^2) can lead to misleading results when applied to the model calibration problem.

Application: An ordinary least squares regression is usually formed between the measured and calculated values. Simple t-tests may then be performed to test whether the regression slope is significantly different from 1 or the regression intercept is significantly different from 0, indicating the presence of bias. Flavelle (1992) discusses the range of validity of this procedure, as well as the use of regression analysis in optimizing the calibration.

Assumptions and Limitations: The usual assumptions regarding linear regression apply to this case. Reckhow et al. (1990) point out a number of potential difficulties in the use of regression analysis to evaluate the behavior of environmental models in this way:

- Often the null hypothesis will be accepted simply because the variation on the x-variable (observations) is too small to allow meaningful hypothesis testing.

- The regression R^2 value and standard error will yield misleading measures of the quality of fit of the model, due to lost degrees of freedom and the unlikelihood that the true parameter values are 0, 1.

- For time series observations there is likely to be autocorrelation in the regression error. This causes bias in the evaluation of regression parameter error. For positive autocorrelation, common in environmental applications, the regression slope t-statistic is inflated as is the model R^2.

Assessing Model Goodness of Fit: Wilcoxon Rank Sum Test

Uses: The Wilcoxon Rank Sum Test, sometimes referred to as the Mann-Whitney test, is a nonparametric test designed to test whether two independent random data sets have a systematic difference. It can thus be used to compare residuals between validation and calibration, or to compare random residuals from different model runs, as long as the residuals have not been constrained to zero mean in the calibration. However, there are other potential limitations to the validity of the test, as discussed below.

Application: The null hypothesis for this test states that the populations from which the two data sets have been drawn (i.e., predictions and observations) have the same mean, while the alternative hypothesis is that the populations have different means.

The procedure for calculating the statistic is as follows:

1. Given two samples of size n_1 and n_2, with $m=n_1+n_2$, combine all data and rank from 1 to m, while preserving the identification as a member of group 1 or 2. If several data have the same value, assign them the average of the individual ranks.

2. Sum the ranks belonging to population 1 as W.

For samples of size less than 10 (either n_1 or n_2) the test of the null hypothesis must be made by comparison to critical values given by Hollander and Wolfe (1973) and other texts on nonparametric methods. For size greater than 10, the distribution of the statistic is approximately normal, given that the null hypothesis is true. The hypothesis may then be evaluated from comparison to a standard normal table. When no ties are present in the ranks, calculate

$$z = \frac{W - E(W)}{[var(W)]^{1/2}}$$

where

$$E(W) = \frac{n_1(n_1+n_2+1)}{2}$$

$$var(W) = \frac{n_2 n_1(n_1+n_2+1)}{12}$$

The calculated z value is compared to a critical z value from a standard normal table to test the hypothesis. A slight modification to the test statistic is needed to account for the presence of tied ranks. For this, see Gilbert (1987).

Assumptions and Limitations: The primary advantage of the Wilcoxon test is that it does not require an assumption of normality, yet retains a relatively high degree of power. It can thus be an attractive alternative to a t-test when the normality assumption is thought to be violated. However, as warned by Reckhow et al. (1990), the Wilcoxon test can be severely effected by violations in the independence assumption, whether due to trends or autocorrelation.

Assessing Model Goodness of Fit: Two-Sample Kolmogorov-Smirnov Test

Uses: The two-sample Kolmogorov-Smirnov test is constructed to compare two empirical Cumulative Density Functions (CDFs), drawn from unknown distributions. A natural application is comparing a set of model predictions to a corresponding set of observations.

Application: Applying the two-sample Kolmogorov-Smirnov test is a simple matter of forming empirical CDFs for the two samples (where the points on the empirical CDF are given as i/n, for the ith ranked sample out of a total of n samples), and evaluating D_2^* as the maximum value of the difference between the CDFs for each data point. For the two-sample test, define the test statistic as (DeGroot, 1986)

$$d = D_2^* \left(\frac{n_1 n_2}{n_1 + n_2} \right)^{1/2}$$

in which case the distribution of the statistic d under the null hypothesis is the same as that of D_2, the statistic for the one-sample Kolmogorov-Smirnov test, which is tabulated in various texts (e.g., DeGroot, 1986). As n becomes large, the critical statistic for α = 0.05 approaches $1.36/\sqrt{n}$. For the two-sample Kolmogorov-Smirnov test, the null hypothesis is that the two samples come from the same distribution (i.e., that the simulation is a good one). This hypothesis should be rejected at the α level if the observed value of the statistic is greater than the critical value.

Assumptions and Limitations: The Kolmogorov-Smirnov test is attractive because it is constructed directly from the observed, empirical CDFs, and makes no assumptions regarding the underlying form of the distribution. However, the test is based on the assumption of independence in the samples, and exact inference in the presence of correlation would require an appropriate adjustment in the effective sample size.

7.2 Model Accuracy and Reliability

One important outcome of the calibration and validation process should be some sort of estimate, either qualitative or quantitative, of the accuracy or reliability of model predictions. This will, of course, be an important factor in deciding how to use the model results in the estimation of the TMDL. The basic point is that models produce only an approximation of reality. Model predictions cannot be any better than the calibration/validation effort, and will always have some uncertainty associated with the output. If model predictions are to be the basis of decisions, it is essential to have some understanding of the uncertainty associated with the model prediction. For instance, suppose a model for a CSO event of a given volume predicts a coliform count of 350 MPN/100 ml, well below the (hypothetical) permit requirement instream of 400 MPN/100 ml. However, the model prediction is not exact, as observation of an event of that volume would readily show. The model must thus provide additional information specifying how much variability to expect around the "most likely" prediction of 350. Obviously, it makes a great deal of difference if the answer is, on the one hand, "likely between 340 and 360", or, on the other hand, "likely between 200 and 2000".

Evaluating these issues involves the closely related concepts of model accuracy and reliability. "Accuracy" can be defined as a measure of the agreement between the model predictions and observations. "Reliability" is a measure of confidence in model predictions for a specific set of conditions and for a specified confidence level. For instance, for a simple mean estimation problem, the accuracy could be measured by the sample standard deviation, while the reliability of the prediction (the sample mean in this case) could be evaluated at the 95% confidence level as plus or minus approximately two standard deviations around the mean.

An assessment of model accuracy and reliability can be an integral part of the modeling strategy for the phased approach to TMDLs. In a phased approach, as more data are collected and understanding of processes in the receiving waterbody increases, it is expected that reliability or accuracy of model predictions should also increase. During each step of the process, the TMDL developer must assess reliability of current results to design the necessary refinements and data collection to increase accuracy.

Assessing the accuracy of individual models is also an integral part of the model selection process. WPCF (1989) summarizes the role of model accuracy in model selection objectives: "Modeling objectives include the need for an assessment of the accuracy of the analysis needed for the work. This analysis will affect the requirements for data collection, the model to be used (if any), the degree of model calibration/ validation required, and ultimately the engineering budget for the work."

Unfortunately, it is not easy to assess relative accuracy among models. The formality and degree to which model reliability must be assessed will vary on a case by case basis, from narrative statements to detailed quantitative analysis. It is suggested that a quantitative analysis is usually advisable when model results are used as the major basis for significant management decisions.

In terms of the probability of excursion of WQSs, there are two separate sources of temporal variability to consider. These are *natural variability* and *model uncertainty*. Natural variability concerns the variability in loading and waterbody response that occurs as a result of precipitation sequences, and so on. Model uncertainty adds an additional layer of "noise": for instance, the simulated response to a precipitation sequence may not be quite right. The probability of WQS excursions due to natural variability alone can be assessed through continuous simulation over a sufficiently long period of precipitation/flow records. However, assessment of the risk of impairment to a waterbody should also take the accuracy of the model into account.

In the following sections we provide a brief review of techniques available to assess the reliability, or uncertainty, associated with simulation model predictions. There are many different techniques used to assess model reliability. This focuses on three of the most commonly used methods: sensitivity analysis, first-order analysis, and Monte Carlo simulation. Listed in increasing order of complexity and detail, each method is useful for specific purposes. While sensitivity and first-order analyses are basically analytical (rather than numerical) techniques, the nature and complexity of most wet-weather episodic loading models and receiving water models more complex than simple analytical methods will likely require the use of a computer to complete an analysis, regardless of the technique chosen. Many published reports document model reliability analysis techniques (IAEA, 1989; Cox and Baybutt, 1981; Freeze, et al., 1990; Inman and Helton, 1988; IAEA, 1989; Marin, et al., 1989).

7.2.1 Sensitivity Analysis

Sensitivity analysis is the least sophisticated and easiest analysis of the three to conduct. However, this ease of use produces only rudimentary results. Consequently, sensitivity analysis is best suited to preliminary reliability analysis and model selection and screening.

The object of a sensitivity analysis is most clearly described by its name. This method is used primarily to assess the sensitivity of model output to perturbations of individual model parameters. The means of conducting such an analysis is fairly straightforward. First, identify one, or more, parameter of interest. In most cases all of the model parameters are chosen for the analysis. Vary each selected parameter through its range of values, while holding all other parameters at their median, or "best-estimate" values, and calculate the model output for each scenario. In many cases it is sufficient to run the model with the selected parameter at only two points, its realistic upper and lower bounds. The analysis is then repeated for each parameter identified earlier. If the model output varies considerably for a given parameter, that parameter is determined to have a large effect on the uncertainty in model output. If the effect is small, the model is determined to be less sensitive to the parameter.

The value of a sensitivity analysis for complex models can be improved by conducting the effort in a formal experimental design format, in which different combinations of parameters are varied on a regular, preplanned scheme. The classical text on experimental design is Box et al. (1978).

The results of a sensitivity analysis should help to identify those parameters that contribute most significantly to the uncertainty in the model output. It is important to note that sensitivity analyses yield only credible *ranges* for model response, under relatively strict conditions (i.e., fixing all other parameters at their median values). No quantitative conclusions about the model accuracy can be drawn from such an analysis. However, undertaking a sensitivity analysis can identify which parameters are likely to provide a significant model response, and should therefore be included as a first step in any more detailed uncertainty analysis.

7.2.2 First-Order Analysis

First-order analysis (also called variance or analytical uncertainty propagation) is a slightly more sophisticated approach to assessing model reliability. It is used to determine the variance of the model output as a function of the variances and covariances of model inputs/parameters. Like sensitivity analysis, variance propagation examines the effects of uncertainty in individual parameters on the model prediction; however, first-order analysis produces a numerical estimate of the additional variability. If the modeler can reasonably assume (and justify) a specific distribution on the predicted values (e.g., a Normal distribution), then this estimated variance can be used to compute confidence intervals for estimated values.

Depending on the nature of the model, the variance associated with one parameter may propagate through the model very differently from the variance of another parameter with the same level of uncertainty. That is, uncertainty in "important" parameters will have a relatively large effect on the uncertainty associated with model prediction; while less important variables will have a smaller impact. Clearly then, the effect of variance propagation depends on both the uncertainty associated with model parameters and the structure of the model itself.

The object of first-order analyses is to determine the variance of the model prediction, based on known or estimated variances of the model parameters. For linear systems the variance of a model prediction can be derived exactly (see an introductory statistics text for more details). As a simple example, assume we have a model of the form

$$Y = P_1 + P_2 + P_3$$

the variance of Y is then calculated from

$$V(Y) = V(P_1) + V(P_2) + V(P_3) + 2[\rho(P_1,P_2)\sqrt{V(P_1)\,V(P_2)} + \rho(P_1,P_3)\sqrt{V(P_1)\,V(P_3)} + \rho(P_2,P_3)\sqrt{V(P_2)\,V(P_3)}],$$

where

$V(P_i)$ is the variance of component P_i, and

$\rho(P_i,P_j)$ is the correlation between P_i and P_j.

Substituting estimates of the parameter variances and correlations into the equation, the modeler can then easily estimate the variance of the predicted value.

While this method does yield an estimated measure of uncertainty in the model, it also has two important limitations. First, it does not provide a probability distribution for the model output. Consequently, without making any assumptions about the distribution of the model parameters or model output, only very broad probability-based statements can be made. For example, the modeler cannot generate specific confidence intervals for the mean value or quantiles. A second drawback of this method is that its applicability is limited to linear or nearly linear systems. Since first-order analysis is based on analytical techniques, we must directly compute the expression for the model output variance. This can only be done exactly for linear systems; for nearly linear systems the modeler can estimate the variance with a linear approximation.

7.2.3 Monte Carlo Simulation Analysis

The third method, Monte Carlo simulation, is a form of probabilistic uncertainty analysis. The objective of this method is to build up an empirical picture of the complete distribution function of model output over the possible range of input parameters. For instance, if a model, denoted f, depended on a parameter, Θ, with distribution function $g(\Theta)$, we would wish to derive the cumulative distribution of the model predictions over the range of Θ:

$$CDF(f) = \int_{\Theta} f(x,y,t,\Theta)\; g(\Theta)\; d\Theta$$

Evaluation of the CDF is accomplished by a "brute force" approach, involving running the model over and over with randomly varied parameter values and collecting the results.

The Monte Carlo method yields not only a variance estimate but also a probability distribution for the model prediction. This distribution is an important piece of information, allowing the modeler to compute interval estimates and draw probability-based conclusions about the model output.

To use the Monte Carlo technique, the modeler first assigns probability distributions to each of the model parameters. These distributions should be based on a solid combination of past experience, preliminary data screening, and expert opinion. No inherent restrictions are placed on the form of these distributions, making Monte Carlo analysis an easily generalizable technique.

After choosing distributions for the parameters, the modeler randomly generates a parameter value from the appropriate distribution and inserts these values into the model equations, yielding a predicted value. This process is repeated many (several hundred or thousand) times, from which a sample probability distribution is generated for the model output. This distribution reflects the overall uncertainty in the inputs to the calculation.

The Monte Carlo technique provides several advantages over the previously discussed approaches to reliability analysis. Most importantly, this method provides the modeler with a probability distribution for model prediction, rather than simply an estimate of its variance. This distribution forms the basis for computing various estimates (e.g., mean, median, 95th percentile) and appropriate confidence intervals for these estimates. As mentioned above, the Monte Carlo method is also applicable to a wide variety of circumstances. For example, its use is not restricted to linear models, wide classes of distributions may be used for input parameters, and the computations are very straightforward. However, these advantages do not come without some cost. Most notably, the modeler must specify distributions for the input parameters. Careful thought must be put into assigning these distributions, as they form the basis for the model output distribution. A frequent criticism of conclusions drawn from a Monte Carlo simulation revolves around the choices of parameter distributions. As a result, sensitivity to the choice of parameter distributions is an important issue to consider; unfortunately, the effect of different distributional choices is difficult to assess. A second potential problem lies in the computer-intensive nature of the analysis. For large, complex models with disperse parameter distributions, Monte Carlo analysis may be computationally infeasible. Stratified sampling techniques (e.g., Latin hypercube sampling) may be used to reduce the effort required to obtain a representative approximation of the CDF.

Several popular environmental fate and transport models are currently available with Monte Carlo analysis capability (such as QUAL2E-UNCAS). Others may be modified to perform such a function, with level of effort dependent on the clarity and structure of the original computer code. IAEA (1989) provide a good introduction to the details of implementing Monte Carlo methods.

7.3. Interpretation of Monitoring Data and Modeling Results

Purpose: This section provides general guidance to aid interpretation and integration of monitoring and modeling results. The information presented aims to provide the basis for assessing receiving water impacts and assessing benefits from implemented control and management practices, which is addressed in Section 7.4.

7.3.1 Data Accuracy, Precision, Bias, and Uncertainty

Accurate environmental assessments requires accurate information. This information may consist of observed data, model predictions, or a combination of the two. The first step in interpreting receiving water monitoring data and model predictions is assessing their level of accuracy. This requires understanding that *accuracy* is a measure of closeness of a measurement to the true value and it has two components—precision and bias.

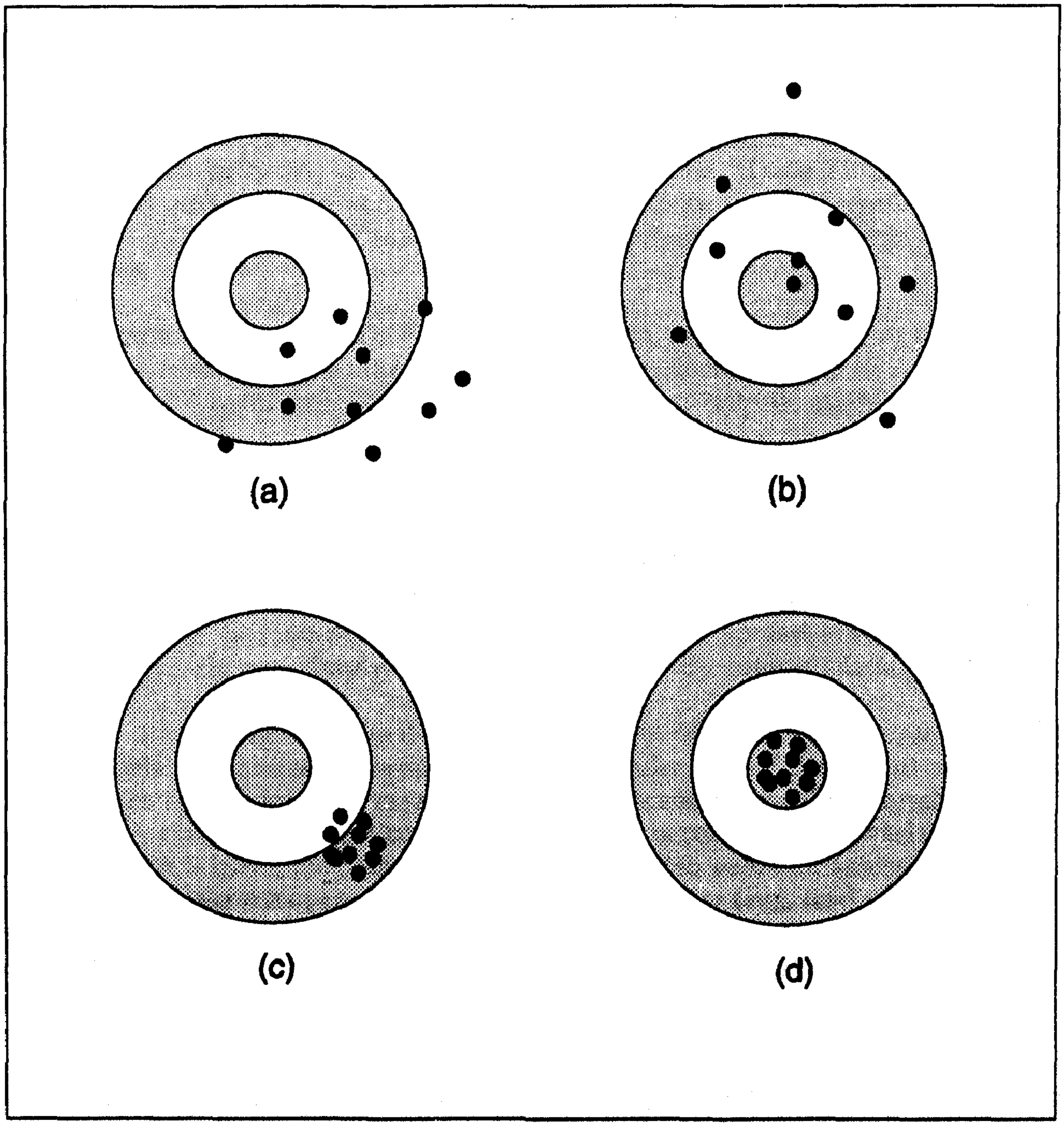
- *Precision* is a measure of the closeness for the collected data to their mean. That is, do the results have a lot of "scatter?"

- *Bias*, most simply, is the tendency for data to be consistently and directionally wrong. Possible examples include regular over- or under-estimates of flow velocities or chemical concentrations.

Statistical techniques are commonly used to measure data precision through calculation of variance, standard deviation, and confidence interval estimates for collected data. Evaluating bias often involves using *measurement standards* to compare results produced by the measuring devices used against results expected from the standards. Such standards can include known volume flows through a calibrated channel or known concentrations for a chemical in the standard solution analyzed. An inexpensive analytical meter, for example, can have a low sensitivity for the variable measured and, consequently, poor precision due to either accuracy or bias problems, or both. But even the finest meters capable of providing the greatest precision can, when they are poorly calibrated, provide biased results and very poor accuracy. Figure 7-4 presents a commonly used example for showing the relationship among precision, accuracy, and bias.

Modeling predictions similarly have characteristics of precision and bias. For models, precision may be thought of as the "scatter" in predictions that could result from the range of possible values of estimates of model parameters, while bias measures the difference between model forecasts and reality. Techniques for assessing the reliability and accuracy of model predictions were discussed in Section 7.2.

Figure 7-4.
Data precision, accuracy, and bias represented by shot patterns in targets (after Gilbert, 1987)



(a) High Bias + Low Precision = Low Accuracy; (b) Low Bias + Low Precision = Low Accuracy; (c) High Bias + High Precision = Low Accuracy; (d) Low Bias + High Precision = High Accuracy.

To develop confidence that data are of sufficient quality for completing an assessment requires three practical "data screening" considerations:

(1) *Are the data reasonable?* This is the common-sense consideration, also sometimes appropriately called "a laugh test." For example, some reported flow data for a creek might be more suggestive of storm flows expected down the Mississippi River, or some reported dissolved metal concentrations might indicate that the water might be suitable as an ore replacement. Such results most commonly indicates an obvious error somewhere in the analysis or data handling. While not all problems in the data are this obvious, many can be nearly as quickly and easily spotted. For example, comparing data to expected ranges for the variable defined as part of the quality assurance program for the project or using some of the simple graphical data plots can aid in identifying such problems.

(2) *Are the data biased?* This question can often be answered by examining the quality assurance and quality control (QA/QC) program documentation (see Section 6.6) and determining whether the monitoring results attain criteria established by that program.

3) *Are the data too imprecise?* After concluding that the data are reasonable and their bias is acceptable, the last step before progressing with the assessment of impact is to determine the level of variability contained within the collected data. Statistical estimators of means, variances, standard deviations, and confidence limits are a common way to help assess this question. Most of the common spreadsheet and database programs can quickly and simply complete these necessary computations.

Before proceeding with final statistical analysis of collected data, one should examine the data summaries for those variables, particularly their means and confidence intervals. When the random scatter (as opposed to inherent non-random variability, e.g., diurnal shifts in dissolved oxygen concentrations) in data collected for a variable at each sampling location is great, the data may have a very limited value for assessing impacts. For example, suppose mean concentrations for a constituent (e.g., lead concentrations) appear markedly reduced in receiving waters after implementing watershed BMPs (e.g., increased intensity of street sweeping) compared to before BMP implementation. Suppose also that the confidence limits from the data sets to be contrasted for these variables both overlap zero (or the lower detection limit for lead). Here, since the analyzed concentrations for the individual sets of analyses are not significantly different from zero (or the lower detection limit) individually, further statistical comparisons between these data would not provide useful information either on differences between data sets for the variable or on potential benefits from the BMPs. Similar analyses could be made for data having underlying trends or cyclic variability, where the scatter can be evaluated around the "true" line or curve for their inherent variability.

Overall, there is little reason to proceed with more complex analysis when confidence intervals for sample test statistics overlap those for other sample sets that are to be contrasted, or when confidence intervals for all contrasted data sets include zero. Box and whisker plots is a useful graphical method for making similar comparisons of data medians for data sets (see Wedepohl et. al., 1990, for an example application).

7.3.2 Combining Monitoring Data and Model Predictions

Monitoring and modeling are complementary processes. As previously discussed in Section 5.1, one is not a replacement for the other and hard data are always preferable to uncertain model predictions. However, even when monitoring data are available, model results are also used to assess impacts and probability of water quality excursions. At its simplest, this is because we rarely have enough monitoring data to provide a complete picture of conditions in the receiving waterbody at all points in space and time. Models can help fill in the gaps in the monitoring record. They can also help predict response to conditions which have not been observed. This is particularly important for wet-weather, episodic loads, which may depend strongly on large and infrequent precipitation events.

The role of modeling relative to monitoring is summarized by Donigian and Huber (1991):

> Computer models allow some types of analysis, such as frequency analysis, to be performed that could rarely be performed otherwise since periods of water quality measurements are seldom very long. It should always be borne in mind, however, that use of measured data is usually preferable to use of simulated data, particularly for objectives 1 and 2 in which accurate concentration values are needed. In general, models are *not* good substitutes for good field sampling programs. On the other hand, models can sometimes be used to extend and extrapolate measured data.

Modeling can also be viewed as a data synthesis tool. That is, the development of a mathematical model gives model developers the ability to compile and synthesize all, or essentially all, information known regarding the system being modeled. This process frequently leads to identifying unknowns and specific additional monitoring needs. In an ideal modeling application, the essence of the information contained in the monitoring data is summarized in the model via the calibration and validation process. The model then constitutes the most convenient summary of the responses of the receiving waterbody.

The interpretation of model predictions depends in large part on the question of whether the spatial and temporal scales of model predictions and desired results match. For instance, using simple steady-state screening models to assess CSO impacts in receiving waters cannot capture the actual time variability of impacts, and may thus be inherently biased for analysis of pollutants with a short response time. However, if set up correctly (i.e., designed to err on the side of safety), such an analysis could provide a worst-case analysis, and thus yield information on the maximum likely impact.

More sophisticated modeling applications will generally attempt to provide predictions at the spatial and temporal scale appropriate to the WQS. When used for continuous (rather than event) simulation, simulation model results may be used to predict the frequency of excursions of WQSs. This can also be accomplished by probabilistic model applications, such as Monte Carlo simulation, in which the simulation is made over the probability distribution of precipitation and other forcing functions (see Section 7.2.3). In either case, model output may be analyzed for WQS excursions, and the frequency of such excursions evaluated.

In interpreting model results, the inherent limitations of modeling must be kept in mind. While modeling is an invaluable tool, it cannot provide exact forecasts of the future, nor can it take the place of good observational data. Modeling of CSOs and their impacts in receiving waters is a relatively inexact science. For instance, with sufficient effort, TMDL developers can often obtain a fairly high degree of accuracy in modeling the hydraulic response of a CSS. In contrast, modeling pollutant buildup/washoff, transport in the CSS, and fate in receiving waters is considerably less exact. However, the predictive ability of even a highly accurate hydraulic response model of a CSS is limited because CSOs are largely the result of essentially random sequences of precipitation events. Because of this, CSO modeling cannot produce deterministic predictions of future events. However, it can be used to predict the expected frequency and duration of such events.

The TMDL developer should keep in mind the various inherent limitations of simulation modeling. The cautions expressed by Nix et al. (1991) provide a useful summary of many of these issues:

> The limitations of all models and the frustrations of simulation are quickly evident to the newly initiated. Sometimes the negative reaction leads to a rejection of the whole notion of modeling. The most healthy response, though, is one that recognizes limits and learns to treat sewer system modeling as a science *and* an art. Said time and time again, it bears repeating anyway—a model is just a tool and not a replacement for sound engineering. Models are vital to the assessment and abatement of combined sewer overflow problems. However, any model must be placed

> in its proper role in the overall analysis and design process, and its output must also be interpreted with a keen awareness of inherent limitations and assumptions . . . .
>
> The use of any model to simulate a combined sewer system and its catchment is inherently limited in a number of ways. First, a computer model cannot improve a database. It can extract information from a database, but it cannot overcome data inadequacies. Second, no model will produce completely accurate results because every model is incomplete and biased in its representation of the system. Third, numbers produced by a computer model are no more accurate than numbers produced by hand calculations, just faster. Placing a model (like the rational method) on a computer does not improve it. The computer model just makes the large number of calculations tractable . . . .

With these caveats in mind, model interpretation should be guided by two principles: (1) model predictions are no better than the quality of the calibration, and (2) all model predictions provide only an estimate, or best guess of future events. In other words, model predictions are uncertain. Ideally, the levels of uncertainty present in the model predictions should be analyzed explicitly. The predictions can then be examined in terms of the probability of excursions of WQSs.

7.4 Evaluation of Effectiveness of Best Management Practices and Other Control Strategies

This guidance focusses on TMDLs involving wet-weather loads. Such loads may discharge as point or nonpoint sources, but typically involve a load component derived from land surface runoff processes. This component of loading can be addressed through management practices

The phased approach to TMDLs allows an iterative approach to TMDL development, particularly where nonpoint source controls are involved. The phased approach is defined (U.S. EPA, 1991a) "as a TMDL that includes monitoring requirements and a schedule for re-assessing TMDL allocations to ensure attainment of water quality standards." Where LAs for nonpoint sources are established as part of the TMDL, the phased approach includes a description of the implementation mechanisms and a process for the evaluation of their effectiveness. The implementation mechanisms are typically best management practices (BMPs) for the reduction or elimination of nonpoint source pollution. Evaluation of their effectiveness usually involves a combination of monitoring and modeling. Because BMP evaluation plays an important role in the estimation of TMDLs where episodic, wet-weather loads are significant, it is emphasized in this section.

7.4.1 Description of Best Management Practices

A Best Management Practice (BMP) is defined in 40 CFR Part 130 as "Methods, measures, or practices [selected by an agency] to meet its nonpoint source control needs. BMPs include but are not limited to structural and non-structural controls and operation and maintenance procedures. BMPs can be applied before, during, and after pollution-producing activities to reduce or eliminate the introduction of pollutants into receiving waters." The broadness of this definition reflects the broad category of nonpoint pollutant sources and the still broader category of potential remedies. BMPs for nonpoint source control are also referred to as management measures. EPA (1993) recently has provided a comprehensive summary of BMPs for most urban and non-urban land uses in its Management Measures guidance. The TMDL developer is referred to this guidance for detailed information on BMPs. Management measures described therein fall into three general categories, delivery reduction, source reduction, and reduction of direct impacts. These have differing degrees of susceptibility to evaluation through monitoring. The Management Measures Guidance discusses these as follows:

Delivery Reduction. Delivery-reduction measures lend themselves to inflow-outflow, or process, monitoring to estimate the effectiveness in reducing loads. The simple experimental approach is to take samples of inflow and outflow at appropriate time intervals to measure differences in the water quality between the two points. An example is the analysis of totals suspended solids (TSS) concentrations at the inflow and outflow of a sediment retention basin to determine the percentage of TSS removed.

Source Reduction. Source-reduction measures generally cannot be monitored using a process design because there are usually no discrete inflow and outflow points. The effectiveness of these measures will generally be determined by applying approaches such as paired-watershed studies and upstream-downstream studies.

Reduction of Direct Impacts. The effectiveness of measures intended to prevent direct impacts cannot be determined through the monitoring of loads since pollutant loads are not generated. Instead, monitoring might include reference site approaches where the conditions (e.g., habitat or macroinvertebrates) at the affected (or potentially affected) area are compared over time (as management measures are implemented) versus conditions at a representative unimpacted site or sites nearby (Ohio EPA, 1988). This approach can be taken to the point of being a paired-watershed study if the monitoring timing and protocols are the same at the impacted and reference sites.

7.4.2 Evaluation of BMPs

The TMDL developer needs to evaluate BMPs, and other control strategies included in a TMDL, in terms of their effect on waterbody impairment. Four concepts are integral to the discussion presented in this section that need to be defined to avoid possible confusion (U.S. EPA, 1991). An *impact* is a change in the physical, chemical, or biological quality or condition of a waterbody. An *impairment* is a detrimental effect on the biological integrity of a waterbody caused by an impact that prevents obtainment of designated uses. *Biological integrity* is functionally defined as the condition of the aquatic community inhabiting unimpaired waterbodies of a specified habitat as measured by community structure and function. And, an *aquatic community* is an association of interacting populations of aquatic species within a waterbody or habitat.

There is a frequent tendency to equate any impact associated with a cultural activity affecting a surface water as an impact resulting in impairment. However, it is possible to have impacts that benefit aquatic communities in some systems. To help reduce possible confusion, the terms "adverse impacts" and "beneficial effects" can be used. Table 7-1 provides guidance on the general scope of considerations that should be given aquatic, wetland, and riparian communities when assessing potential effects associated with point and non-point sources. All potential adverse impacts and beneficial effects shown in the table should be assessed, as possible, through monitoring, modeling, and/or reference ecosystems studies. Evaluation of other special adverse impacts and beneficial effects also may be appropriate when completing the TMDL process.

There are three classes of techniques for quantifying the benefit of BMPs for reducing nonpoint pollution: (1) Paired-Watershed Studies; (2) Upstream-Downstream Studies; and (3) Use a simulation model to compute the change in pollutant concentration and loading resulting from the installation/implementation of BMPs. Each of the techniques has inherent advantages and disadvantages. The following sections outline those relative advantages and disadvantages. Given the general poor reliability of water quality modeling for pollutants, a successful strategy for BMP simulation might involve two or even three of the above classes of efforts.

7.4.2.1 BMP Evaluation by Paired-Watershed Method

The following discussion of paired-watershed design is adapted from U.S. EPA (1993): In the paired-watershed design there is one watershed where the level of implementation (ideally) does not change (the control watershed) and a second watershed where implementation occurs (the study watershed). This design has been shown in agricultural nonpoint source studies to be the most powerful study design for

Table 7-1. Guidelines useful to determine potential impacts and benefits in receiving waters associated with discharges (expanded from Tuden et al., 1992)

Adverse impacts:

Reduced abundance of desirable species
Decreased vigor of desirable species
Loss of productivity by desirable species
Disappearance of desirable species
Increased abundance of undesirable species
Shifts in community dominance to undesirable species
Bioaccumulation of materials in tissues likely to harm human, wildlife, or fish consumers of those tissues
Violation of downstream water quality standards, impacts to actual beneficial uses, or degradation of groundwaters

Beneficial effects:

Improvement of water quality conditions key to ecosystem development and preservation, e.g., pH, dissolved oxygen, ammonia
Preservation of existing, desirable riparian or aquatic species that would not be maintained without discharge of the effluent
Increased diversity of desirable species
Increased productivity by desirable species
Flow augmentation

Desirable species:

Native species reflecting non-degraded conditions in the receiving water.
Species of special concern (e.g., federal and state listed "threatened" or "endangered" species; otherwise listed "sensitive" species)
Species of special cultural interest (e.g., non-native game fish species)

Undesirable species:

Noxious species, not dominant under natural conditions (e.g., blue-green algae that form blooms and cause taste or odor problems; weedy species)
Non-native species that prey on or compete for food or habitat space with desirable species (e.g., green sunfish, carp, salt cedar, zebra mussels)

demonstrating the effectiveness of nonpoint source control practice implementation (Spooner et al., 1985). Paired-watershed designs have a long history of application in forest hydrology studies. The paired-watershed design must be implemented properly, however, to generate useful data sets. Some of the considerations to be made in designing and implementing paired-watershed studies are described below.

In selecting watershed pairs, the watersheds should be as similar as possible in size, shape, aspect, slope, elevation, soil type, climate, and vegetative cover (Striffler, 1965). (Further details on the selection of reference or paired watersheds is provided in Section 7.4.3 of this guidance). The general procedure for paired-watershed studies is to monitor the watersheds long enough to establish a statistical relationship between them. A correlation should be found between the values of the monitored parameters for the two watersheds. For example, the total nitrogen values in the control watershed should be correlated with the total nitrogen values in the study watershed. A pair of watersheds may be considered sufficiently calibrated when a parameter for the control watershed can be used to predict the corresponding value for the study watershed (or vice versa) within an acceptable margin of error.

It is important to note that the calibration period should cover all or the significant portion of the range of conditions for each of the major water quality determinants in the two watersheds. For example, the full range of hydrologic conditions should be covered (or nearly covered) during the calibration period. This may be problematic in areas where rainfall and snowmelt are highly variable from year to year or in areas subject to extended wet periods or drought. Calibration during a dry year is likely to not be adequate for establishing the relationship between the two watersheds, particularly if subsequent years include both wet and dry periods.

Similarly, some agricultural areas of the country use long-term, multiple-crop rotations. The calibration period should cover not only the range of hydrologic conditions but also the range of cropping patterns that can reasonably be expected to have an influence on the measured water quality parameters. This is not to say that the calibration period should take 5 to 10 years, but rather that States should use careful judgment in determining when the calibration period can be safely ended.

After calibration, the study watershed receives implementation of management measures, and monitoring is continued in both watersheds. The effects of the management measures are evaluated by testing for a change in the relationship between the monitored parameters (i.e., a change in the correlation). If treatment is working, then there should be a greater difference over time between the treated study watershed and the untreated (poorly managed) control watershed. Alternatively, the calibration period could be used to establish statistical relationships between a fully treated watershed (control watershed) and an untreated watershed (study watershed). After calibration under this approach, the study watershed would be treated and monitoring continued. The effects of the management measures would be evaluated, however, by testing for a change in the correlation that would indicate that the two watersheds are more similar than before treatment.

It is important to use small watersheds when performing paired-watershed studies since they are more easily managed and more likely to be uniform (Striffler, 1965). EPA recommends that paired watersheds be no larger than 5,000 acres (USEPA, 1991c).

7.4.2.2 BMP Evaluation by Upstream-Downstream Studies

The discussion of upstream-downstream studies is also taken from U.S. EPA (1993): In the upstream-downstream design, there is one station at a point directly upstream from the area where implementation of management measures will occur and a second station directly downstream from that area. Upstream-downstream designs are generally more useful for documenting the magnitude of a nonpoint source than for documenting the effectiveness of nonpoint source control measures (Spooner et al., 1985), but they have been used successfully for the latter. This design provides for the opportunity to account for covariates (e.g., an upstream pollutant concentration that is correlated with a downstream concentration of same pollutant) in statistical analyses and is therefore the design that EPA recommends in cases where paired watersheds cannot be established (U.S. EPA, 1991c).

Upstream-downstream designs are needed in cases where project areas are not located in headwaters or where upstream activities that are expected to confound the analysis of downstream data occur. For example, the effects of upstream point source discharges, uncontrolled nonpoint source discharges, and upstream flow regulation can be isolated with upstream-downstream designs.

It is important to note that background seasonal variation in pollutant concentrations can compound the difficulty of discerning individual pollution sources and BMP effects. For example, monthly mean nitrate concentration in the Fall Creek watershed in New York varies sinusoidally in a seasonal pattern with a relative maximum occurring each spring or late winter and a relative minimum occurring each summer (Bouldin, 1975). A similar pattern has been observed in other watersheds (Likens, et al., 1977; Hill, 1986). This background variation can mask or exaggerate the effect of BMPs unless it is taken into account in designing a sampling plan.

7.4.2.3 BMP Evaluation with Simulation Models

A third technique for estimating the effectiveness of BMPs on wet-weather loading is simulation modeling of the resultant loading in runoff. This is the only technique available for direct estimation of *prospective* BMPs, prior to implementation. Models for wet-weather loads, such as those discussed in Chapter 3, may be used for this purpose. However, it is often difficult to reliably simulate the effects of small scale BMPs with watershed-scale models. The exception would be where relatively large scale BMPs are planned, for example the alteration of cultivation practices for the majority of an agricultural watershed.

A full range of model types, from simple to complex can potentially be used to simulate the effectiveness of changes in management practices in watersheds, although simple models are too coarse in scope to simulate field scale changes. Even moderately complex models will be poor simulators of field-scale changes if they are based on a lumped-parameter method. The selection of appropriate simulation models for BMP evaluation is addressed in Chapter 4.

In most cases, the final evaluation of the effectiveness of BMPs will be based on a combination of methods. Models as the sole basis of evaluation will be applied only *in prospective evaluations, that is, in the first stage of the phased TMDL process.* As the phased approach requires monitoring to establish effectiveness of management measures, subsequent analyses will be based on a combination of monitoring and modeling. However, because wet-weather loading processes are driven by stochastic rainfall events, monitoring alone will usually be insufficient to represent the full range of potential loads that may be generated by precipitation events. Therefore, simulation of the effectiveness of BMPs is usually necessary to complete the analysis based on monitoring results.

7.4.3 Establishing Appropriate Reference Conditions

Evaluating impacts to receiving water environments requires judging observed against expected conditions in the receiving waters. This is typically done by comparing data collected from sites where impact may occur to similarly data from the same sites collected prior to the expected impact and to similarly collected data from one or more appropriate reference sites. Such reference data can often provide definitive benchmark information both for (1) establishing what environmental conditions can be attained and (2) assessing impacts (Green, 1979; U.S. EPA, 1990). Hence, selecting reference sites that appropriately represent reasonable obtainable target conditions for the receiving water is key to successful assessments.

Reference sites may represent the completely unaffected state, a relatively unaffected state, or increasing degrees of existing impact, as deemed appropriate for a study. Appropriateness of a reference site should arise out of a clear understanding of the overall environment in which the receiving water is a part. For example, some interested parties may advocate defining unimpacted conditions as those existing in a waterbody prior to any impacts caused by modern society. Yet no unimpacted waterbody likely exists today according to this restrictive definition. Essentially all waterbodies in North America are at least subtly impacted, for example, by airborne pollutants. But even including such impacts, conditions occurring in wilderness waterbodies are not reasonably obtainable at present for most waterbodies nearer to and affected more directly by modern developments. Consequently, impacts to aquatic communities from discharges into channelized streambeds having upstream inputs from point and nonpoint discharges cannot be directly, and perhaps not usefully, assessed using comparisons to aquatic communities inhabiting unchannelized "wilderness" reference sites.

Sensible assessments of receiving water conditions generally build from data collected from appropriate reference monitoring sites. Selection of sampling sites for assessment and reference monitoring should include the considerations on designing monitoring programs presented in Chapter 5.

In general, the most useful reference sites are located within the receiving water of concern, relatively near the impact-monitoring site(s), but outside of the zone of impact associated with the pollutant source being assessed. Reference sites typically should not include effects attributable to any significant identifiable pollutant source. Often, appropriate reference sites within a receiving water include upstream site(s) above downstream impact monitoring sites, or sites in unimpacted tributary streams of the receiving water stream. Appropriate reference sites in lakes, reservoirs, or estuaries receiving waters should be located at sites having ecologically similar conditions to the assessment monitoring site(s), but located outside of the zone of potential impact.

As a simple example, consider the rare case where a point source discharge is the sole source of potential impact to a receiving water. Here, an appropriate sampling design might include at least one impact monitoring sampling site located inside the zone of expected impacts and one reference monitoring site outside this zone. Including more than one impact monitoring site within the zone of impact would provide data for assessing impact variability. To evaluate the spatial extent of impacts requires spacing sampling sites at regular, or progressively increasing, distances apart through and beyond the zone of expected impacts.

Monitoring designs, and the process of identifying appropriate reference sites to assess impacts from a particular source, become progressively more complex as additional sources of potential impact to the receiving water system occur. In general, using a site-specific reference study approach, investigators should select one or more appropriate reference sites for each source of potential impact being assessed. Ideally, each location used for site-specific reference monitoring should have environmental conditions equal (minus impacts) to the site(s) used to monitor potential impacts for each source of discharge being evaluated. Clearly, "sorting out" impacts caused by any individual source under conditions having multiple impact sources can sometimes present difficult sampling design problems. Similarly difficult problems can occur where significant gradients for environmental variables naturally occur within the monitored system.

Where the multiple sources of potential impacts are widely dispersed and zones of recovery occur between zones of impact, it is often possible to distribute multiple monitoring sites between adjacent sources of potential impact to detect possible gradients of change (trends for either impact or recovery) associated with each source. Here, each set of upstream sites [or those sites most distant from source(s) of impacts on other water types—far-field sites] may provide adequate reference conditions for the next set of downstream impact monitoring sites [or the next set of sites closer to the impact source(s)—near-field sites].

EPA presents more detailed examples of useful sampling designs for monitoring in Klemm et al. (1990). If no other portions of the receiving waterbody exhibit these capacities because of existing impacts, selecting reference sites in another waterbody would be necessary. Here, investigators may select reference sites in ecologically similar tributary streams, neighboring watershed, lakes, or estuaries, or regional reference sites. Considerable guidance is available for using paired watersheds and "ecoregion" reference sites (e.g., U.S. EPA, 1990; Gallent et al., 1989; U.S. EPA SAB, 1991).

7.4.4 Evaluating Environmental Trends

Natural variations in data for water quality and other ecological variables generally complicate abilities to assess the effectiveness of implementing source controls. Water quality and ecological systems naturally vary over daily, seasonal, and annual intervals. Variations in physical patterns for precipitation, wind, temperature, and solar radiation primarily drive these changes. Random variations in these parameters over daily intervals cause generally smaller-scale, irregular fluctuations in many ecosystem variables. Concurrently, regular seasonal changes in these physical parameters can produce other more rhythmic patterns of larger magnitudes.

Using mathematical or statistical methods to identify trends and segregate causes of variability found in environmental data sets can help to define more easily changes associated with source controls. This Section introduces concepts involved and the use of such procedures to detect trends in monitoring data for water quality parameters.

These approaches can help when assessing time-dependent changes in receiving water ecosystems associated with implementing source control alternatives.

Figure 7-5 shows a series of examples simulating how interactions among three different time-dependent influences might affect the distribution of collected data over time. These plots each show what could represent data from 40 samples collected at equal time intervals over 1½ years. The first of these plots [Figure 7-5(a)] shows how collected data for a parameter influenced purely by random variation might appear. Data collected for a chemical parameter in a discharge stream could appear similar to this, for example, as influent concentrations and treatment efficiencies vary within the design range. Figure 7-5(b) shows how a discharge of the substance in the first plot might affect concentrations in a receiving water that already contained seasonally varying background concentrations of this substance.

Next, suppose there had been a gradual and progressive effort to increase the "treatment efficiency" over the period of data collection. Then the generally flat random distribution shown in the first of these four plots might appear as shown in Figure 7-5(c). Here, a general underlying *trend* of increasing treatment efficiency could cause a continued decrease in the measured "discharge" concentrations over time. Figure 7-5(c) shows this with the same random pattern shown in the first plot otherwise maintained. Then, if this substance were again "discharged," receiving water concentrations might appear as shown in Figure 7-5(d). Obviously, if collected data were available only from this last plot, determining natural receiving water patterns or benefits of treatment would not likely be a straightforward process.

Patterns like those shown in the plots of Figure 7-5 could occur in monitoring data for physical, chemical, or biological variables. But, the relationships shown in these plots are simplified from those likely found for real monitoring data. In particular, patterns for data collected in or downstream of source discharge likely would be complicated by the appearance of discharge pulses. At these times, flow volumes and concentrations of various contaminants can show often dramatic short-term increases accompanying runoff events. Also, acutely toxic conditions that can accompany these events can impact aquatic life in receiving waters, causing the death of resident life forms or their migration away from the affected waters to avoid potentially toxic conditions. Either event can cause steep short-term trends for sampled variables.

Various data-analysis techniques are available to help distinguish temporal trends in monitoring data where additional background influences also influence numerical values obtained for the data. When variation in collected data caused by natural variability can be mathematically distinguished and removed, influence of trends attributable to other causes can become increasingly clear. For example, removing seasonal changes from environmental data could leave only background data noise to

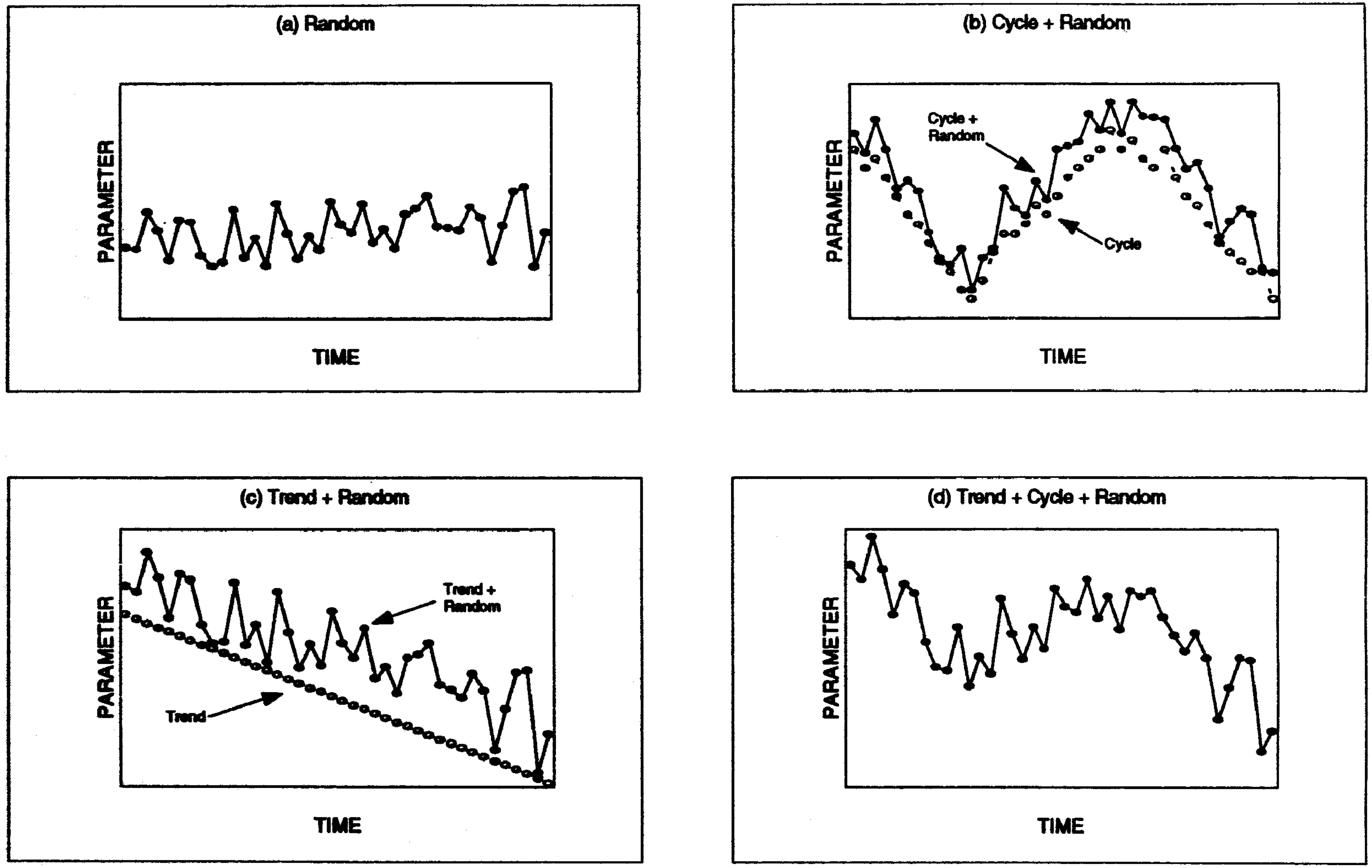


Figure 7-5. Examples of compounded interactions in time series.

complicate assessing the benefits interpretations of control alternatives.

Detailed discussions on the selection and use of trend-analysis techniques are presented by Gilbert (1987), Hipel (1988), Loftis et al. (1989, 1991), and other sources cited later in this Section. Also, the final chapter of the EPA technical guidance document entitled *Monitoring Lake and Reservoir Restoration* (Wedepohl et al., 1990) presents a brief case study that introduces many considerations involved in analyses to detect trends in total phosphorus concentration in the Neuse River, North Carolina. In the remainder of this Section, we briefly summarize the considerations and discussion presented with that example to select the appropriate data-analysis techniques. Similar considerations would be necessary when selecting appropriate trend-analysis techniques and when completing interpretations of possible trends in environmental data collected during many aquatic environmental monitoring programs.

As background for the case example, the Neuse River drains an area of over 6,000 square miles, which includes the City of Raleigh, two upstream water supply reservoirs, and an extensive forested area. The U.S. Geological Survey collected monthly phosphorus data between 1981 and 1988; detected concentrations ranged from 0.13 mg/l to 1.8 mg/l. To determine whether a trend of change in phosphorus concentrations existed in the data collected over this period, investigators considered both parametric models and non-parametric models (also often called distribution-free methods).

In general, several parametric methods can be used to assess trends of change. These methods often relate the assessed changes to some physical parameter, such as flow, depth, or retention time. But parametric methods generally include the assumptions that linear relationships exist between the analyzed variables and that data for these variables have normal, bell-shaped distributions around their means. Sometimes, data transformations can be used to achieve distributions reasonably resembling normal (McLeod et al., 1983). Other transformations can be used to approximate linear relationships between variable data sets. Similarly, most parametric methods also require that the data not include seasonal influences or not be autocorrelated, i.e., the data are not correlated in time with any prior and subsequently collected data for the variable. Other methods are available to transform data to remove seasonal influences (Taylor and Loftis, 1989).

Where the assumptions of parametric methods can be reasonably met and where the trend is believed to be continuous, instead of abrupt or stepwise, it is often appropriate to use ordinary least square regression to determine the possible slope for the trend of change. This approach could be used, for example, to assess the trend shown by the data in Figure 7-5(c).

When evaluating possible effects that a single event (e.g., a pulse discharge) produces in a receiving water, the Student's t-test can sometimes be used, if assumptions underlying this test are met. This test allows comparing data collected prior and after the event to evaluate possible changes. However, more sophisticated parametric trend analysis techniques are more appropriate when the normally distributed data include seasonal influences or when they are autocorrelated. For additional information about using parametric trend analysis techniques, including Box-Jenkins models, the texts of Gilbert (1987) and Pankratz (1983), and articles by Montgomery and Reckhow (1984) and Berryman et al. (1988) can be consulted.

In the analysis for the Neuse River example, the data failed to meet the requirements for using parametric trend analysis methods: the data were significantly skewed and they displayed a seasonal cycle. In fact, parametric techniques often are not appropriate and should always be cautiously applied to environmental data, since many water quality and other environmental parameters commonly have non-normal and seasonal distributions. Therefore, the Neuse River investigators opted for non-parametric methods to assess trends in their data. While non-parametric methods generally are less powerful than parametric methods, they do not require that the data have a generally normal distribution about their mean. But most non-parametric methods still require that the analyzed data contain neither seasonal or autocorrelated relationships.

The Neuse River data were analyzed using the seasonal Kendall's Tau Test. This test was selected also because it can assess seasonally varying data and it is not overly sensitive to extreme values (Gilbert, 1987; Loftis et al., 1989). These are common problems in many environmental data sets. Following data preparation and autocorrelation testing, as discussed by Wedepohl et al. (1990), this test revealed no significant trend of change in phosphorus concentrations over the seven years of monitoring. Consequently, this study indicates that (1) additional measures to control phosphorus entering the Neuse River and (2) additional monitoring of this river are warranted.

When analyzing water quality and ecological data for trends, we often want to assess possible simultaneous trends in several variables at several monitoring stations. Most trend analysis procedures, including those discussed above (excluding the seasonal Kendall's method), are univariate procedures, i.e., they are able to assess trends in single variables at single locations. These techniques are inappropriate for multivariate or multiple location analyses. Additional multivariate tests, which are generally less widely known and used, are available to assess such data. These methods were recently reviewed by Loftis et al. (1991) to assess their performance with both serially independent and serially correlated data. They report the strengths and limitations of the five parametric and nonparametric procedures evaluated.

7.5 Assessing Water Quality Impacts on Fish and Other Aquatic Life

To assess the protection of beneficial uses of a waterbody from a proposed control strategy it is often necessary to assess or predict water quality impacts on aquatic life. This brief overview introduces several common criteria and discusses some of the possible consequences that altered water quality has on aquatic life. Water quality variables discussed include temperature, pH, nutrients, dissolved oxygen, potentially toxic chemicals, and suspended sediment. This discussion draws heavily from U.S. EPA's 1986 *Quality Criteria for Water,* on an earlier review completed by the American Fisheries Society (Thurston et al., 1979), and a review for the USDA Forest Service (Marcus et al., 1990).

7.5.1 Temperature

Most biological and chemical processes in aquatic environments ultimately are regulated by water temperature. Fish and essentially all other aquatic animals are cold blooded (*poikilotherms*); thus, their metabolism, reproduction, development, and scope for activity is largely controlled by environmental temperatures. Similarly, aquatic plant photosynthesis and respiration, chemical reaction rates, gas solubilities, and microbial mediated processes including decomposition and nutrient cycling are also temperature dependent. In fact, the Federal Water Pollution Control Administration in 1967 described temperature as "a catalyst, a depressant, an activator, a restrictor, a stimulator, a controller, a killer, one of the most important and influential water quality characteristics to life in water" (U.S. EPA, 1986). Section 5.6 introduced the role of thermal characteristic in the dynamics of most receiving water environments.

The present criterion to protect freshwater aquatic life is based on "the important sensitive species" resident during the time of concern and consists of two upper temperature limits (U.S. EPA, 1986). The first limit is based on short (i.e., over durations of minutes) exposure, is computed using an equation presented in the EPA criterion, and uses data presented in a National Academy of Sciences document. The second limit is based on a weekly maximum average temperature, which changes with season, with reproductive stage present, to maintain species diversities, or to prevent nuisance growths of organisms. For rainbow and brook trout adults and juveniles, the maximum weekly average temperature for growth during the summer is listed as 19°C, and the short-term maximum temperature limit for survival during summer is 24°C (U.S. EPA, 1986). This report also lists 9°C as the average weekly maximum temperature reported for spawning by these species, and 13°C as the short-term maximum reported for survival of their embryos. These present temperature criterion, however, present numerous interpretational problems with respect to defining "important sensitive species" and "short-term" (Thurston et al., 1979).

Thermal criteria for marine environment are much more straightforward. To protect the characteristic of indigenous marine communities from adverse thermal effects, U.S. EPA (1986) maintains that

a. The maximum acceptable increase in the weekly average temperature resulting from artificial sources is 1°C (1.8°F) during all seasons of the year, providing the summer maxima are not exceeded; and

b. Daily temperature cycles characteristic of the waterbody segment should not be altered in either amplitude or frequency.

7.5.2 pH

In natural waters, pH is primarily regulated by the solution of atmospheric carbon dioxide, which reacts with water to form carbonic acid and then disassociates to hydrogen and bicarbonate ions. Distilled water at equilibrium with atmospheric CO_2 has a pH of 5.6 at sea level. But natural waters contain various dissolved salts and organic chemicals derived from watershed rocks, soils, and organisms that tend to buffer the natural acidities and raise the pH of surface waters. Over approximately the past 15 years it also has become increasing apparent that the pH of weakly buffered aquatic are often influenced by the deposition of atmospheric acids (e.g., Baker et al., 1990).

As with temperature, the concentration of hydrogen ions is an important regulator of many chemical and biological processes in aquatic and marine environments. (Most accurately, it is the chemical activity of hydrogen ions, rather than their concentration, that is reflected by measured pH levels.) For example, pH primarily defines the chemical natures of dissolved ions in waters, the directions of chemical reactions, the adsorption of chemicals onto organic and inorganic particles, plus the availability and toxicity of chemical to organisms. Similarly, uptake and release rates for ions across gills, the primary method of ion regulation for aquatic animals, is at least partly pH dependent. Environmental conditions beyond their natural pH limits can produce stress and cause mortality of organisms.

The criteria range to protect freshwater aquatic life is pH 6.5 to 9.0 and 6.5 to 8.0 for marine life (U.S. EPA, 1986). Many fish and other aquatic life, however, are well able to survive and reproduce at pH levels outside of this range. Also, not all species and not all life stages of most species are equally sensitive to pH changes. For example, as acidity levels increase, brook trout are generally less sensitive than brown trout, which are in turn less sensitive than rainbow trout; hatching and larval stages are the life stages for these species that are most sensitive to acidity (Marcus et al., 1986). Brook trout populations frequently have been found to inhabit waters have pH levels less than pH 5.5 (e.g., Schofield and Trojnar, 1980).

7.5.3 Nutrients

The two nutrients of greatest potential concern in aquatic systems are nitrate and phosphate. These nutrients are the two most related to the eutrophication of surface waters, the associated nuisance growths of algae, and the development of other noxious conditions. The problem of excess nutrient enrichment and eutrophication of surface waters of all types is a long recognized problem, dating back to at least 1907 (Hutchinson, 1969, 1973). According the National Academy of Science (NAS, 1969: pages 3-4),

> "The term 'eutrophic' means well-nourished; thus, 'eutrophication' refers to natural or artificial addition of nutrients to bodies of water and to *effects* of added nutrients. Eutrophication of lakes is a natural process that can be greatly accelerated by man. Eutrophication is an aspect of aging; it increases the rate at which lakes disappear. Some disagreement exist as to the applicability of the term to other bodies of water. Streams do not age in the same sense as lakes, although added nutrients will increased their productivity. . . . . When the effects are undesirable, eutrophication may be considered a form of pollution."

The effects of nutrient enrichment differ between streams and lakes. In streams, the downstream flow of nutrients out of the immediate system typically is more important. Further, the development of excess algal biomass is often reset by removal of attached algae during flood scour events (Lohman et al., 1992). The question of whether the term eutrophication is appropriate for streams was particularly brought to focus by Hynes (1969: page 188) in that NAS volume:

> "The term 'eutrophic,' which was coined for application to lakes, has acquired so many connotations of aging and evolution of the environment that it cannot properly be applied to running water. Unlike a lake, a stream has no allotted life-span; it is an ongoing phenomenon, and if it ages, it does so only in the sense of erosion toward base level."

Thus, Hynes suggested that, with respect to streams, it may be better to just use "nutrient enrichment" rather than "eutrophication" as a term. He also acknowledged that problems related to discharges of excess nutrients to either lakes or streams do, in fact, exhibit many commonalities. While there has been some disagreement regarding the application of the term *eutrophication* to flowing waters, it has continued to be commonly used to describe the processes and problem of nutrient enrichment in streams and rivers.

Certainly, it should also be recognized that not all forms of eutrophication or nutrient enrichment necessarily produce undesirable consequences. For example, considerable effort has been directed at learning approaches for maintaining correct nitrogen and phosphorus balances to apply under controlled conditions to enhance salmon production in nutrient-poor lakes along coastal western Canada (e.g., Barraclough and Robertson, 1971; Stockner, 1981). In fact, Hynes (1969: page 194) ended his consideration of nutrient enrichment of streams with,

> "In summary, then, we can say that enrichment produces fairly obvious effects on small watercourses and that in moderation these may be beneficial from our point of view, since they increase production of such things as game fish. Greater amounts produce definitely deleterious effects, although up to a point this may not be true in countries where the value of fish is reckoned in weight of protein rather than quality. In larger rivers the effect is uncertain, but probably bad, and we may make it worse by constructing impoundments. It should, however, be emphasized here that in contrast to a lake, which can be made eutrophic and then probably remains in that state, a stream or river has to be continuously enriched. It can, therefore, be rescued and restored."

What kinds of adverse impacts accompany excess nutrient loading and eutrophication problems? Often, one of the first responses is by phytoplankton (free-floating microscopic algae) in standing waters and periphyton in flowing waters. Periphyton is the assemblage of organisms that grow on underwater surfaces and is commonly predominated by algae, but also can include bacteria, yeasts, molds, protozoa, and other colony forming organisms. According to EPA's manual *Biological Field and Laboratory Methods for Measuring the Quality of Surface Waters and Effluents* (Weber, 1973),

> "Excessive growth stimulated by increased nutrients can result in large, filamentous streamers [and mats] that are aesthetically unpleasing and interfere with such water uses as swimming, wading, fishing, and boating, and can also affect the quality of the overlying water. Photosynthesis and respiration can affect alkalinity and dissolved oxygen concentrations of lakes and streams. Metabolic byproducts released to the overlying water may impart tastes and odors to drinking waters drawn from the stream or lake, a widespread problem throughout the United States. Large clumps of growth may break from the site of attachment and eventually settle to form accumulations of decomposing, organic, sludge-like materials."

There is no clear-cut definition of exactly what constitutes nuisance levels of periphytic algae. However, it has been suggested that nuisance biomass of filamentous periphytic algae may be represented by a level greater than 100-150 mg-chlorophyll-*a*/m^2 (Horner et al., 1983; Welch et al., 1988).

What are the concentrations of nutrients that can produce problems? No criterion is provided by the EPA for either of these two nutrients with respect to the control of eutrophication. U.S. EPA's (1986) criteria document does discuss the toxic potential for nitrates to fish. This report concludes that nitrate-nitrogen concentrations at or below 90 mg/l should be protective for warmwater fishes, while concentrations at or below 0.06 mg/l should be protective for salmonid fish. This guideline for salmonids is based on very limited data, and <u>many</u> natural salmonid waters have nitrate concentrations exceeding this level.

U.S. EPA (1986) also suggests as a guideline to prevent nuisance algal growths and limit cultural eutrophication that total phosphates as phosphorus should not exceed 0.1 mg/l in any stream or other flowing water, exceed 0.05 mg/l in any stream at the point where it enters a lake or reservoir, or exceed 0.025 mg/l in any lake or reservoir.

Golterman (1975) suggests that, in general, eutrophication may occur in surface waters that have nitrate-nitrogen concentrations above 0.3 mg/l and phosphate-phosphorus concentrations above 0.02 mg/l. Similarly, Wetzel (1975) reports that eutrophication in lakes can begin as total inorganic nitrogen concentrations exceed 0.5 mg/l N, and serious hypereutrophication problems occur at concentrations exceeding 1.5 mg/l N. He also reports that eutrophication in lakes generally occur at total phosphorus concentrations exceeding 0.03 mg/l P, and hypereutrophication generally occurs at concentrations greater than 0.1 mg/l.

These concentrations for lakes are very similar to those long associated with problems in flowing waters. For example, Mackenthun (1969: page 41) summarizing Muller's 1953 review of nutrients in flowing water systems, noted " . . . that excessive growths of plants and algae in polluted waters can be avoided if the concentration of nitrate nitrogen is kept below about 0.3 mg/l and the concentration of total nitrogen is not allow to rise much above 0.6 mg/l." Mackenthun (1969: page 41) further concluded, "A considerable judgement suggests that to prevent biological nuisances, total phosphorus should not exceed 100 µg/l P [= 0.1 mg/l P] at any point within the flowing stream, nor should 50 µg/l be exceeded where waters enter a lake, reservoir, or other standing waterbody. Those waters now containing less phosphorus should not be degraded."

Other experiments in phosphorus-limited flowing systems suggests that very low soluble reactive phosphorus (SRP) concentrations may be required to avoid periphytic biomass at nuisance levels. For example, less than 0.001 mg/l SRP was recommended for the Spokane River (Welch et al., 1989), and less than 0.025 mg/l SRP from experiments in laboratory channels (Horner et al., 1983).

Analyses of various plant tissues (including algae) indicate that total nitrogen and total phosphorus concentrations occur within plants in the atomic ratio of 15:1 to 16:1 (Stumm and Morgan, 1970). Also, because of biological nutrient uptake by plants, the global average of TN:TP in surface waters is also in this same range. This ratio is generally considered the optimal ratio for general plant growth and is commonly cited as an important indicator of relative nutrient limitations by these two nutrients Higher ratios indicate possible phosphorus limitation, while lower ratios indicate possible nitrogen limitation.

A TN:TP ratio of 10:1 by weight (e.g., mg/l) in aquatic ecosystems was found to be the ratio below which nitrogen fixation by blue green algae is common (Fleet et al., 1980). The scientific literature further indicates that at TN:TP ratios of less than 29:1 by weight, blue-green algae maintain significant populations (>10%) within algal communities and blooms by blue-green algae are exceedingly common (Smith, 1983). Part of the reason for this is that blue green algae are generally more efficient than many other taxonomic groups of algae in acquiring nitrogen but less efficient at getting phosphorus from surface water environments. At TN:TP ratios of greater than 29:1 blue green algae generally comprise less that 10% of algal communities. Knowledge of these ratios for surface waters can help to target potential sources of nutrients and to guide corrective management actions aimed at reducing possible eutrophication problems.

It also is sometime useful to recognize that every gram of P in the water can grow 115 grams of algal biomass dry weight or about 500 grams wet weight, assuming algae is about 80 percent water. This estimate is based on the average chemical composition of algae (cf, Stumm and Morgan, 1970).

In conclusion, it is clear that uncontrolled and excessive additions of nutrients to surface waters can and has severely damaged receiving water ecosystems. Excessive algal growth in can cause diurnal depletion of dissolved oxygen to unacceptable levels. The question often becomes one of determining the nutrient concentrations or loads beyond which the ability of the receiving water to assimilate the inputs is overloaded. At what concentration is the limits of the system's natural function exceeded, causing adverse impacts to begin? Determining these loading rates are often a primary objective of the TMDL process. Without a TMDL study, expert opinion backed by scientific literature must be used to project nutrient concentrations that will cause potentially adverse impacts.

7.5.4 Dissolved Oxygen

Determining dissolved oxygen concentrations in surface waters is very informative because its concentration reflects the integrated health of the aquatic community. In fact, maintenance of adequate concentrations of dissolved oxygen in receiving waters is one of the key indicators that the discharge of nutrients and other chemicals causing BOD/COD is within the assimilative capacity of the receiving water. EPA maintains water quality criteria for dissolved oxygen only in freshwaters (Table 7-2). In general, any time dissolved oxygen concentrations dips below 4.5 mg/l in cold waters or below 3.5 mg/l in warm waters, there is reason for concern that the chemical and biological integrity of the system is threatened. Site-specific studies would be necessary, however, to appropriately evaluated the degree of this magnitude and degree of this threat relative to the species in inhabiting and the obtainable use defined for the water.

Table 7-2. Water quality criteria for ambient dissolved oxygen concentrations (U.S. EPA, 1986)

Time	Coldwater Criteria		Warmwater Criteria	
	Early Life Stage[1,2]	Other Live Stage	Early Life Stage[2]	Other Live Stage
30-Day Mean	NA[3]	6.5	NA	5.5
7-Day Mean	9.5 (6.5)	NA	6.0	NA
7-Day Mean Minimum	NA	5.0	NA	4.0
1-Day Minimum[4,5]	8.0 (5.0)	4.0	5.0	3.0

1 These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. The 3 mg/l differential is discussed in the criteria document. For species that have early life stage exposed directly to the water column, the figures in parentheses apply.

2 Includes all embryonic and larval stages and all juvenile forms to 30-days following hatch.

3 NA = not applicable

4 For highly manipulatable discharges, further restrictions may apply.

5 All minima should be considered as instantaneous concentrations to be achieved at all times.

7.5.5 Toxic Chemicals

Aquatic life can be affected by an increasing diversity of potentially toxic chemicals. A review of the toxicity and potential impacts of all of these chemicals is beyond the scope of this report. But, Table 7-3 lists the lowest observed effects levels obtained from short-term acute toxicity and long-term chronic toxicity tests for a variety of toxicants potentially encountered in receiving waters; water concentrations less than the indicated levels may not affect aquatic life. The information provide a guideline to determine potentially hazardous conditions for aquatic life. For a specific water, actual toxic concentrations are often likely to be either greater or lesser than the reported values. For example, the toxicity of ammonia increases as either temperature or pH of the water increases (Thurston et al., 1979).

As another example, the toxicity of many metals and other chemicals is affected by hardness. That is, actual instream toxicities of metals to aquatic organisms does not depend solely on the total concentrations of the metals in the water. A variety of studies, show that water quality characteristics, especially hardness, alkalinity, pH, and chelating organic materials, can affect the aquatic chemistries, toxicities, and bioavailabilities of metals (Black et al., 1975; Parkhurst et al., 1984). Studies also reveal that not only can calcium hardness, in particular, affect the instream chemistry of many metals, but it can affect the physiological susceptibilities of the organisms to the potential toxicities of many metals (Davies and Woodling, 1980; Mount et al., 1988; Parkhurst et al., 1984; Pascoe et al., 1986).

In addition, potentially toxic chemicals are rarely present singularly; toxicities derived from multiple chemical sources may be additive, subtractive, or multiplicative. Therefore, when evaluating the potential toxicities to fish by chemicals in surface waters, evaluations must include careful determination of and potential interactions with other chemicals present.

7.5.6 Suspended Sediment

Turbidity is a measure of the scattering and absorption of light by dissolved and particulate matter in water (Lloyd, 1987). Usually, turbidity and suspended sediment concentration are highly correlated; thus, turbidity can provide an index of suspended sediment concentrations (Lloyd et al., 1987). Because murky water absorbs more heat than clear waters, increased suspended sediment loads can cause water temperatures to increase (Hynes, 1970). Water with a temperature of 5°C is able to carry 2 to 3 times more sediment than 27°C waters (Heede, 1980).

Table 7-3. 1986 Quality Criteria for Water (U.S. EPA, 1986).

Chemical	Freshwater Concentration (μg/l)		Marine concentration (μg/l)	
	Acute Criteria	Chronic Criteria	Acute Criteria	Chronic Criteria
Aldrin	3.0			1.3
Ammonia	Criteria are pH and temperature dependent			
Antimony	9,000[1]	1,600[1]		
Benzene	5,300[1]		5,100[1]	700[1]
Beryllium	130[1]	5.3[1]		
Cadmium	3.9[2]	1.1[2]	43	93
Chlordane	2.4	0.0043	0.09	0.004
Chlorine	19	11	13	7.5
Chloroform	28,900	1,240		
Chromium (Hex)	16	11	1,100	50
Chromium (Tri)	1,700[2]	210[2]	10,300[1]	
Copper	18[2]	12[2]	2.9	2.9
Cyanide	22	5.2	1	1
DDT	1.1	0.0010	0.13	.001
DDT Metabolite (DDE)	1,050[1]		14[1]	
DDT Metabolite (TDE)	0.6[1]		3.6[1]	
Dieldrin	2.5	0.0019	0.71	0.0019
Endrin	0.18	0.0023	0.037	0.0023
Iron		1,000		

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Chemical	Freshwater Concentration (µg/l)		Marine concentration (µg/l)	
	Acute Criteria	Chronic Criteria	Acute Criteria	Chronic Criteria
Lead	82[2]	3.2[2]	140	5.6
Malathion		0.1		0.1
Mercury	2.4	0.012	2.1	0.025
Nickel	1,800[2]	96[2]	75	8.3
Parathion		0.04		
pH		6.5-9.0		6.5-8.5
Phenol	10,200[1]	2,560[1]	5,800	
Selenium	260	35	410	54
Silver	4.1[2]	0.12	2.3	
Hydrogen Sulfide		2		2
Thallium	1,400[1]	40[1]	2,130[1]	
Toluene	17,500[1]		6,300[1]	5,000[1]
Zinc	320[2]	47	95	58

1 Insufficient data to develop criteria. Value presented is the Lowest Observed Effects Level (LOEL).

2 Hardness dependent criteria (100 mg/l hardness used for reported value).

The EPA has narrative criteria for suspended solids and turbidity:

Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent for the seasonally established norm for aquatic life.

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The EPA has narrative criteria for suspended solids and turbidity:

> Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent for the seasonally established norm for aquatic life.

This criterion is based on the depth in the water column at which planktonic photosynthesis equals respiration (i.e., the *compensation point)*, and it provides a generally weak bases for extrapolating possible effects in other groups or in flowing water environments.

The European Inland Fisheries Advisory Commission concluded that suspended sediment can affect aquatic organisms by killing them directly, by reducing growth rates and resistance to disease, by preventing successful development of eggs and larvae, by modifying natural movement or migration patterns, or by reducing the natural availabilities of food (U.S. EPA, 1986). A review completed by the National Academy of Science suggested that a limit of 25 mg/l of suspended sediment would provide high, 80 mg/l moderate, 400 mg/l low, and over 400 mg/l very low levels of protection for aquatic organisms (Thurston et al., 1979). Lloyd (1987) suggested, based on his review of turbidity studies in Alaska, that a water quality standard that permitted an increase of 25 NTUs (nephelometric turbidity units) above ambient would provide moderate protection for clear, coldwater stream habitats.

The effects of fine sediment on fish, particularly trout and related species, in streams has received particular research attention (see Chapman, 1988, and Marcus et al., 1990, for extensive reviews). We present a brief overview of some of the results in the balance of this section.

Though researchers do not agree on the exact size definition of fine sediment, it is generally taken to be less than 6.3 mm in diameter (Chapman, 1988). As discussed in Section 5.6, sediment transport in streams is a very complex relationship involving at least 30 variables (Heede, 1980). But, in general, transport of fine sediment may be via saltation along the stream bottom or suspension in the water column, with discharge and channel slope proportional to the quantity and size of transported sediment (Hasfurther, 1985). Typically, transport of sediment is greater on the ascending limbs of storm hydrographs, but this is due more to the supply of sediment rather than the hydraulics of the flows (Sidle, 1988; Sidle and Campbell, 1985).

Everest et al. (1987) and Chapman (1988) suggested that limited fine sediment may be beneficial to some salmonids by contributing to increased invertebrate productivities, and that the adverse consequences of fine sediment introduction to trout streams have been sometimes has been overstated. Nonetheless, the transport and deposition of fine sediment deleteriously affect survival throughout the life history of salmonids is many stream systems.

Suspended sediment may directly or indirectly influence the survival of aquatic organisms directly by clogging and damaging respiratory organs. Concentrations greater than 100 mg/l have reduced survival of juvenile rainbow trout (Herbert and Merkens, 1961). Reductions in growth or feeding of salmonids were associated with turbidity over 25 nephelometric turbidity units (NTU) (Olson et al., 1973; Sigler et al., 1984; Sykora et al., 1972). Since salmonids are considered to be sight-feeders, the reduction in light transmission caused by high turbidity may result in less feeding and decreased growth (Berg, 1982). In response to turbidity, salmonids may change their use of cover or reduce territoriality (Berg and Northcote, 1985). When given the opportunity, juvenile coho salmon avoided turbid water (Bisson and Bilby, 1982). Despite these impacts, salmonids often successfully inhabit streams with seasonally high turbidities, perhaps due to behavioral modifications and to limited exposure to concentrated suspended sediments.

Deposition of fine sediment can acutely affect survival of salmonids (1) during intragravel incubation of eggs and alevins; (2) as fingerlings; and (3) throughout winter (Chapman and McLeod, 1987). Timing, source, and quantity of deposited sediment can affect survival. Increasing proportions of fine sediment in substrates have been associated with reduced intragravel survival of embryonic trout, char, and salmon species. Increases in fine sediment can directly limit survival-to-emergence only by entrapping alevins. The potentially greater influence on survival by increased sediment deposition is the decrease in dissolved oxygen concentration coupled with reduced intragravel water flow (Chapman, 1988). However, most studies evaluating the impacts of fine sediment on embryonic survival have been conducted in the laboratory; few or no field studies have satisfactorily quantified actual impacts (Chapman, 1988).

Fingerling density has often been associated with low concentrations of fine sediment deposited between and on the surface of larger substrate particles, i.e., *embeddedness* (Burns and Edwards, 1985). Chapman and McLeod (1987) reported that the relation between the rearing densities of salmonids and fine sediment was equivocal. Instead, they suggest that changes in stream morphology caused by fine sediment may outweigh the effects of embeddedness on fingerling survival.

Declining water temperatures in winter may cause salmonids to seek refuge within the intersticial spaces of the substrate. Deposition of fine sediment could also restrict winter cover for adult fish by filling in low velocity habitats, e.g. pools, and undercut banks (Bjornn et al., 1977).

7.6 Interpretation of Biological Data

This section provides general guidance and additional reference sources with information that can aid interpretation of monitoring data for aquatic biological communities. The discussion emphasizes impact assessment. It begins with simple evaluation steps to qualitatively evaluated sample data and introduces methods and cautions for conducting quantitative community analysis. The final three subsection reviews additional sources for specific guidance for assessing algae, benthic macroinvertebrates, and fish.

7.6.1 Qualitative Evaluation of Sample Data

After the investigator is satisfied that an adequate monitoring plan was used and the data are of adequate quality (often as defined by the QA/QC protocol), inspection and interpretation of the data can begins. A common and useful first step is to examine the total number of species present (*species richness*), note differences in taxonomic compositions among sampling times and sampling sites, and review the environmental requirements for those species most commonly found in the samples. Various documents, which we note later in this section, provide considerable useful information on the sensitivities and tolerances for many species to various environmental conditions. Some of these documents also list those species that are widely recognized as *key* *indicator species* that have particular sensitivities or tolerances to individual environmental variables. The taxa lists developed from the samples should be scanned to determine whether any of these indicator species occur. When the results from these comparisons are compiled and this information is qualitatively integrated with the physical and chemical monitoring results, a useful "first-cut" qualitative evaluation of the monitored environment often results. (See below for two important cautions on this approach.)

Other key qualitative information to look for when first examining the biological sample data is obvious evidence of possible seasonal cycles in the appearances and disappearances of individual taxa in the collected data. This is especially valuable when examining algae and invertebrate samples results. Are seasonal patterns repeated year to year? Do maximum and minimum densities (e.g., numbers per sample) or biomasses remain relatively constant each year? Is a trend of increase or decrease suggested by the data? While seasonal patterns and year-to-year differences are also important when examining data on fish, it is also important to determine whether some *year classes* (individuals recruited to the population from a single year of spawning) may be missing. Severe environmental stress can cause spawning complete spawning failures in some years, producing gaps in the year-class/age-structure of some fish populations. Often these qualitative evaluations of the collected data are useful in helping to focus, or sometimes to redirect, subsequent statistical analysis of the collected data.

7.6.2 Quantitative Community Analysis

Many community indices are available to aid in the quantitative summarization and interpretation of collected data. Documents noted at the end of this section provide considerable helpful guidance on use and interpretation of many of these indices. One common approach to interpret community structure represented by of monitoring data in terms of *species diversity*. Either or both of two components may be included when characterizing species diversity: species richness (i.e., the numbers of species within the community) and species evenness (i.e., distribution of the relative abundances among these species). As recently summarized by Crowder (1990), the diversity of species in waterbodies can be determined by colonization rates, extinction rates, competition, predation, physical disturbance, pollution, and other factors. A common problem in assessments of monitoring data is to determine how these multiple factors interact to determine species diversity at the monitored sites.

A useful two-step qualitative data assessment approach can help to unravel determinants of species compositions in collected monitoring data. The first step again, as discussed above for the qualitative examination of collected data, is to compare the list of species collected and their relative population sizes against published records of the known sensitivities by these taxa, or their close relatives, to various contaminants known or suspected to be present. The tendency of species to be abundant, present, or absent relative to their individual relative tolerances or sensitivities to, for example, sediments, temperature regimes, or various chemical pollutants should be noted. Frequently, also as suggested above, this comparison will begin pointing toward the most likely determinants of species diversity at the sampled sites.

TMDL developers should note two important cautions with respect such comparisons. First, different strains of the same species sometimes can have widely different sensitivities to a stressor. This is especially relevant for fish species that have undergone extensive hatchery breeding programs. These efforts have affected sensitivities not only for characteristics targeted by the breeding program, but also for nontargeted sensitivities to other potential stressors. Second, when incorporating information from indicator species lists into an assessment, it is important to review and evaluate whether the rankings shown on the list include data collected from the region of interest. When the list includes no species from the area surrounding the receiving water of concern, use of the listed species can represent an over-extension of the credibility of the list. In any assessment using such lists, the TMDL developer should document possible known or expected limitations of the list(s) and note the possible implications for the assessment.

The second step then involves integrating into this assessment implications of two patterns often observed within distributions of species diversity values. Results from two literature reviews indicate that two generalized relationships widely underlie diversity patterns:

(1) Forces that disrupt ecosystems from an equilibrium or stable condition (e.g., wind, irregular chronic toxic input) will tend to cause species diversity to increase, unless the disturbance is too frequent, then diversity will tend to decrease (Huston, 1979).

(2) Species diversities generally first increase to a maximum then decrease again across many environmental gradients, including availabilities of resource supplies such as nutrients (Marcus, 1980).

Knowledge of these two generalized patterns can sometimes help in interpreting species diversity patterns defined through monitoring data for species numbers, abundances, and sensitivities to environmental stressors. Potential causes for relationships appearing in the species diversity data also can be suggested. Such relationships can provide the basis for establishing testable hypotheses for developing and implementing special studies targeted to better define source-pollutant-impact relationships.

A final caution is necessary regarding some assessment approaches using species diversity. A number of indices have been developed to characterize species diversity and evenness. Many of these are presented in the various guidance documents cited below. Computed indices often include various potentially intractable problems for statistical analysis (see Green, 1979; Sokal and Rohlf, 1981). Frequent among these problems for species diversity indices are extremely wide variance estimates and, for some, an inability to calculate variances. Potential deficiencies in the indices used should be noted and, if possible, assessed as part of the overall study and analysis design and assessment effort.

While diversity indices can provide useful general indicators of environmental effects, investigators should generally limit their use to within-study comparisons, where sampling and sample analysis methods are consistent. Additionally, if an assessment uses a diversity index, the TMDL developer should also demonstrate an understanding and (when possible) provide an evaluation of potential limitations caused by using the index. Hurlbert (1971), Peet (1974), and Pielou (1975, 1977) provide additional discussion on the use and misuse of species diversity indices.

7.6.3 Algae

EPA's UAA manuals present instructive introductory guidance for interpreting results from monitoring samples for algal communities collected from streams and rivers, estuaries, and lakes and reservoirs (U.S. EPA, 1983, 1984a, b). Additional useful guidance is available in two additional EPA documents:

- *Environmental Requirements and Pollution Tolerance of Freshwater Diatoms* (Lowe, 1974)
- *Guide to the Identification, Environmental Requirements, and Pollution Tolerance of Blue-Green Algae (Cyanophyta)* (VanLandingham, 1982)

7.6.4 Benthic Macroinvertebrates

Interpretation of biological monitoring data for larger bottom-living invertebrates is greatly aided using information contained in EPA's three UAA manuals (U.S. EPA, 1983, 1984a, b). Their RBP manual also provides additional useful information to help interpret these data collected from flowing water environments (Plafkin et al., 1989). Most recently, EPA's manual for *Macroinvertebrate Field and Laboratory Methods for Evaluating the Biological Integrity of Surface Waters* (Klemm et al., 1990) provides additional useful guidance, including discussions and presentations of:

- Analysis of qualitative and quantitative data
- Community metrics and pollution indicators
- Pollution tolerance of selected macroinvertebrates
- Hilsenhoff's family level pollution tolerance values for aquatic arthropods

7.6.5 Fish

Beyond the various EPA documents noted above that also provide information helpful when interpreting monitoring data for fish samples, two reference works published by the American Fisheries Society contain a wealth of useful information. The first, *Fisheries Techniques* (Nielsen and Johnson, 1983), focuses mainly on considerations important for work in the field. Its chapters include discussions on advantageous and shortcomings of most of the currently practiced sampling techniques. The document discusses many potential problems associated with most field data collection techniques. It also reviews methods and important considerations for length, weight and associated structural indices, age determinations, fish diet analysis, and angler characterizations.

The companion volume, *Methods for Fish Biology* (Schreck and Moyle, 1990), focuses primarily on methods used to analyze and assess collected fish samples in the laboratory and in the office. As stated in that book's Preface, ". . . pros and cons of alternative procedures are treated, as are uses and misuses of the data generated by the techniques." Among the chapters are ones on research method designs, fish growth, stress and acclimation, reproduction, behavior, population ecology, and community ecology.

7.7 Physical Aquatic Habitat Evaluation

Section 101(a) of the Clean Water Act recognizes the importance of preserving the physical integrity of the Nation's waterbodies, as discussed by U.S. EPA (1983a). Physical habitat affects the types and numbers of species inhabiting a waterbody. Characterizing physical habitats can help identify factors not related to water quality that may impair the propagation and protection of aquatic life. Further, this can help determine what uses can be attained in the waterbody given such limitations. In general, physical properties such as temperature, water depth, currents and flow, substrate, suspended solids, and reaeration rates can present critical limitations that preclude attainment of beneficial uses. The physical characteristics of a waterbody also greatly influence its reaction to pollutants and its natural purification processes.

Table 7-4 lists some of the key physical characteristics affecting habitats in the three major categories of aquatic ecosystems.

Point and nonpoint sources can adversely affect physical habitats for aquatic organisms through discharge of high-energy water flows and through discharge of excess fine sediments. High-energy flows can erode bottom and bank materials, destabilize the physical structure of aquatic habitats, kill aquatic resident organisms, and destroy eggs incubating in the benthic environment. Excess deposition of fine sediment can coat bottom materials, smothering organisms and eggs that are still incubating. EPA's UAA guidance for the various water types (U.S. EPA, 1983, 1984a, b) present important considerations to aid in interpreting measurements of these and other physical habitat variables.

Another source of potentially useful information to help interpret impacts to fish associated with differences in physical habitat conditions are the Habitat Suitability Index (HSI) Models developed by the U.S. Fish and Wildlife Service. Available HSI models and sources to obtain copies were listed in Table 6-8.

Table 7-4.
Key physical characteristics affecting aquatic habitats

Ecosystem Type(s)	Important Characteristics	Reference(s)
Streams, rivers	Flow Suspended solids Sediments Pool-riffle ratios Run-bend ratios Substrate composition Channel characteristics (including channelization) Temperature Bank stability Riparian cover	U.S. EPA, 1983, Plafkin et al., 1989
Lakes, impoundments	Size Shape Depth Temperature Flow/current regimes	U.S. EPA, 1984b
Estuaries	Tides Wind shear Momentum and buoyancy of freshwater inflows Topographic frictional resistance Coriolis effect Vertical mixing Horizontal mixing	U.S. EPA, 1984a

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Chapter VIII. Establishing TMDLs/WLAs/LAs and the MOS

Purpose: This chapter discusses the policies, procedures, and nuances of translating data and model results into formally designated TMDLs, WLAs, LAs and an MOS. As was discussed in chapter 1, the statutory mandate for TMDLs is found in CWA § 303(d). However, the procedures for establishing TMDLS are regulated under 40 CFR 130.7. The TMDL Process guidance document (U.S. EPA, 1991a) provides a detailed explanation of the States and EPA responsibilities for development and submittals, but spends less time on the details of translating data and model results into TMDLs. This guidance focuses on the latter.

8.1 Which Comes First, the TMDL or WLAs/LAs?

Logic would have it that the TMDL for a waterbody would be determined first (i.e., determine the "size of the pie") based upon the system's assimilative capacity, and then the TMDL ("pie") would be divided ("sliced") into portions reserved for WLAs and LAs, and possibly even an explicit MOS. In the real world, however, assimilative capacity is imprecisely known and often changes over space and time. It can be dependent on season, variation in meteorological conditions, the distribution of pollutant sources and their inherent characteristics, and a myriad of other ecological factors. Thus, in many cases, it may not be feasible to use data analysis or modeling to arrive at a single allowable load for a given pollutant that adequately and fairly addresses WQSs and loading allocations.

The most practical method may be to use the model to evaluate various combinations of WLAs and LAs that represent existing and potentially future source scenarios. After all, by definition, the TMDL is the sum of all WLAs and LAs that serve to protect WQSs in a given waterbody. Thus, the TMDL developer can use the model in an iterative manner, varying combinations of WLAs and LAs and noting which alternatives protect WQSs throughout the watershed. Combinations of WLAs and LAs that are shown through the modeling analyses to protect WQSs are viable candidates for the TMDL.

In some cases, there may not be a model or the level of uncertainty associated with model results may be high. In these cases, an initial TMDL strategy of WLAs and LAs based on best professional judgment may have to be implemented. Follow up monitoring of the waterbody can demonstrate whether the TMDL is sufficient or needs revision. Additional predictive modeling may be needed to support revision and prevent an endless "trial and error" loop. However, even in the case with sophistocated modeling, performance monitoring may indicate the need to "go back to the drawing board" with regard to the TMDL management strategy.

8.2 Deciding When to Employ the Phased Approach

The concept of the phased approach was discussed in chapter 1. There it was stated that lack of adequate information would be cause to enter into a phased approach that allowed for incremental building of the TMDL. However, most analysts would agree that there is never perfect information on which to act. Thus, the issue arises for the TMDL developer as to how to determine when the lack of information is substantial enough to warrant a phased approach.

8.2.1 How Much is Enough?

Deciding when the lack of information is severe enough to employ a phased approach to TMDL development is usually a judgment call. There are, however, factors that can be taken into consideration in making a sound decision. For one thing, the TMDL developer can review any model reliability analyses that may have been performed (see chapter 7 for more information on reliabilty analyses). If the reliability analyses (formal or informal) reveal a high level of uncertainty in model results, then the TMDL developer should scrutinize the impacts of potential error more carefully. Sensitivity analyses for the model could demonstrate whether substantially different results are predicted for key parameters where uncertainty is high. If these analyses show that important decisions hinge on highly uncertain model assumptions, then the TMDL developer may want to use a phased approach that allows for better information to be gathered to reduce the uncertainty. Important decisions include those where the viability of a designated waterbody use is threatened or where an action may be required that exerts a high or disproportionate cost on one of the stakeholders.

8.2.2 Are Interim Control Measures Needed?

When a phased approach is chosen, the TMDL developer should decide whether existing controls within the watershed are adequate for the intervening period. "Adequate" in this case does not necessarily mean protection of WQSs, since the reason that a TMDL strategy is being adopted will frequently be to restore an impaired waterbody that is not being protected under current controls. Rather, it refers to a level of protection that prevents severe threats to human health or aquatic life, or irreparable damage from occurring in the interim.

In addition, interim measures may be taken to "maintain the status quo" while final goals are established. This can be particularly important for watersheds facing the stress from rapid population growth and land development. An example of this occurred in the Tar-Pamlico Basin in North Carolina. North Carolina environmental officials classified the basin as nutrient sensitive due to severe eutrophication effects. However, not enough information was available to determine the level of nutrient reduction that would be necessary to meet WQSs. Therefore, the State implemented an interim plan with less stringent requirements that were expected to prevent an increase in nutrient loading to the sensitive estuary area while a hydrodynamic water quality model could be developed to help establish final reduction targets (NCDEHNR, 1989).

8.2.3 Scheduling Phased Approach Activities

Section 303(d) of the CWA mandates TMDLs be adopted for water quality limited waterbodies, and 40 CFR 130.7 requires that States identify those priority waters for which TMDLs will be adopted over the next two years. Thus, in delaying adoption of a TMDL for a prioritized waterbody, the administering agency should specify the schedule for when the final TMDL will be established. The TMDL Process document (U.S. EPA, 1991) states that the scheduling should coordinate all various activities (monitoring, modeling, permitting, etc.) and stakeholders (e.g., local, state, and federal authorities, permittees, etc.).

Steps to Scheduling a Phased TMDL

- Identify information gaps
- Determine tasks needed to complete TMDL
- Identify stakeholder roles/responsibilities
- Estimate needed resources
- Compare needs to available resources
- Obtain additional resources or revise plan
- Set milestone dates for completion of tasks

Several steps may need to be gone through in order to arrive at a reasonable schedule. After determining what additional information needs to be gathered and the subsequent tasks that should follow to fill those gaps and complete the TMDL, the parties that are involved (i.e., stakeholders) and their roles/responsibilities should be identified. This would include any agencies/permittees/etc. that will be collecting data, developing models, and/or implementing follow up control measures. Next, an estimate of resources needed to perform the tasks should be developed along with an inventory of available resources. If there is a mismatch between available resources and those needed to perform the tasks, then additional funding sources may need to be sought or the tasks can be revised to work within resource constraints. Where possible, it may be advantageous to spread the costs out over the stakeholders (i.e., pooling resources may lead to a more efficient and effective information gathering effort). Finally, with tasks, stakeholders, roles/responsibilities, and resources identified, a schedule of milestone dates for completion of the tasks can be developed. Where sampling restrictions for particular weather conditions exist, the schedule may need to be kept relatively flexible to allow for time delays caused by inappropriate conditions.

8.3 Splitting Up the Pie

One of the most difficult tasks for the TMDL developer will undoubtedly be how to divide up allocations for a given waterbody. While technical bases such as modeling results play a key role in this task, policy considerations are inextricably intertwined. Issues such as equitability, economics, and "political will" often play a role, and decisions will be made that are not always popular with all of the stakeholders. Nonetheless, adoption of TMDLs requires that this task be performed. This section touches upon some of the options and considerations facing TMDL coordinators in this regard.

8.3.1 Assigning a Margin of Safety (MOS)

Section 303(d) of the CWA requires TMDLs to include "a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality." Given that TMDLs address both point source (WLAs) and nonpoint source (LAs) allocations, this concept may be extended to cover uncertainty in BMP efficacy in addition to effluent limitations. With this in mind, TMDL developers are faced with the issue of how to incorporate a MOS into the TMDL.

Two methods are discussed in the TMDL Process document (U.S. EPA, 1991). The first and most commonly used method is to implicitly incorporate a margin of safety through conservative model assumptions. Therefore, in cases where the TMDL developer and modeler are not the same person, the TMDL developer should sit down with the modeler and evaluate important model assumptions for their conservative nature. This is not an easy task, however, for it requires a criterion for determining how much of a safety factor is adequate. There is no "cut and dried" answer to that question. Model uncertainty and sensitivity analyses may lend some insight; i.e., the greater the uncertainty in sensitive parameters, the more margin of safety should be included in estimating those parameters. If after scrutiny of the level of uncertainty in significant model assumptions and the subsequent implications of error the TMDL developer believes that the implicit MOS is not large enough, then either modeling assumptions can be revised or an explicit MOS can be assigned.

An explicit MOS takes the form of a specified reserve of load. However, since the TMDL is not likely to reflect a single numeric value, this reserve may have to be associated with each component of a TMDL strategy. For instance, the reserve could take a percentage off each WLA and LA. Instead of allocating 100 percent of the allowable allocation, a portion could be withheld as a safety factor (e.g., component is given 80 percent of the allowable allocation). In cases where a single TMDL value is established, it would be simpler just to remove a portion of that load and use the remaining portion to divide out into WLAs and LAs.

Establishing a MOS to account for the level of uncertainty can be problematic for situations with a high level of uncertainty. There may be cases where the range of possible error is large enough to cover an entire spectrum of control needs (i.e., from no additional assimilative capacity available to the need for minimum control devices). In these cases, being absolutely sure that the MOS will protect the waterbody would mean holding the entire amount in reserve. This may be an option if the area is pristine and designated an "outstanding resource water." However, in many cases there will already be point sources and land use activities that require some reasonable level of allocation; holding out the entire allocation to account for uncertainty would be unreasonable. In these cases, a phased approach is recommended where additional information is sought to reduce uncertainty. In addition, interim allocations can be assigned with a MOS that's determined to be reasonable and a follow-up monitoring program can be implemented to document effectiveness.

8.3.2 Achieving a Balance Between WLAs and LAs

During the early years of CWA implementation, EPA and States placed greatest emphasis on the control of point sources through imposition of effluent limitations in NPDES permits that were enforceable. This was a logical step since, in many cases, municipal and industrial discharges existed without any form of treatment or with minimal forms of treatment that did not perform at levels needed to protect WQSs. Also, point sources are easier targets in that their effluent outfalls are readily identifiable and can be monitored relatively easily. Many nonpoint sources, on the other hand, were exempt from CWA permitting requirements and monitoring NPS is not typically an easy task because of their diffuse nature.

With the 1987 amendments to the CWA, however, Congress began to recognize the need to move beyond improving treatment technologies at point sources if the nation's waters are expected to meet water quality objectives. Many States were continuing to report impaired waters despite significant implementation of point source controls. With the addition of section 319 of the Act, Congress set forth a mandate to more aggressively implement best management practices (BMPs) that will reduce NPS loads and aid in the protection of the nation's waters. In response to these actions by Congress, EPA has placed a renewed emphasis on a watershed protection approach (WPA) that attempts to establish a better balance between controls of point and nonpoint sources.

The WPA makes the natural resource (i.e., the waterbody) the "client", rather than a specific EPA or State agency or program. Thus, rather than simply trying to meet specific minimal program or agency requirements, agencies (at all levels: federal, state, and local) are being encouraged to take actions necessary to meet the "client's" needs. Thus, assessments, prioritization, and management actions should strive to find the appropriate balance of controls for both PS and NPS needed to meet WQSs within a given waterbody.

Finding a balance between WLAs and LAs in a TMDL management unit involves evaluation of several factors. First, the manager needs to know how problem parameter loads are apportioned among PS and NPS. Is one source dominating the other? Imposition of controls should reflect the size of the source where possible. For instance, if a pollutant load from NPS was found to be 80 percent of the total loading to a problem area and a 40 percent overall reduction in loading was needed, it would make little sense to focus only on point source controls.

Secondly, the TMDL developer should look at the potential efficacy of controls. What BMP and PS controls are feasible and how effective will they be? TMDL developers are unlikely to have thorough knowledge in this regard and, therefore, will likely need to seek input from the stakeholders. Time constraints may not allow for an indepth review in every case, but efforts to gain an understanding in the efficacy of feasible controls will undoubtedly make for more successful TMDL strategies. Helpful guidance in specifying NPS management measures in coastal waters can be found in U.S. EPA, 1993.

One argument that is often used against balancing WLAs and LAs in TMDL development is that LAs are relatively unenforceable since they involve some activities that are exempt from permitting requirements and the fact that BMP implementation is overseen by nonregulatory agencies. While this poses an implementation challenge, it should not be made an excuse for not assigning allocations that require appropriate reductions from sources of concern. Areas where success cannot be achieved because of the failure of NPS BMPs to be implemented should be highlighted and brought to the attention of those involved. In some cases, new public policy and/or regulations may be the only way that LAs will be complied with in the long run. However, without attempts to include appropriate LAs from the beginning, it is possible that the problems will never be addressed.

Enforcement of LAs

Cost effectiveness should be considered. Since financial resources for controls are limited, emphasis should be placed where possible on achieving the greatest return on the money. For example, environmental officials in North Carolina recently evaluated the needs for additional nutrient reduction controls in the Neuse River Basin due to ongoing use impairment in the lower river and estuary portion of the basin. Point Sources had already been addressed in the basin through imposition of phosphate bans and effluent limitations such that their relative contribution of loading had been reduced from approximately fifty percent down to around twenty percent in less than five years. Rather than immediately considering additional PS controls, the State has chosen to target specific watersheds for closer examination for improved implementation of BMPs. Since reasonable LAs in the Neuse basin are currently unknown by the Water Quality Section incharge of establishing the TMDLs for NC, the State has entered into a phased TMDL approach to focus resources and schedules around the tasks of gathering enough information to establish reliable LAs in addition to their existing WLAs.

As the level of information on targets, costs, removal efficiencies improves, formal optimization analysis or multi-objective decision theory may be options for those familiar with them. These methods would allow for decision criteria to be made explicit and, therefore, might help those agencies looking for methods that appear less subjective to stakeholders scrutinizing the allocation process.

8.3.3 Equitability and Fairness Between Allocations

One issue that arises in splitting up the pie is that of equitability between allocations. While there is no legal mandate that specifies that each allocation will be of equal amount, its probably wise not to develop allocations that disproportionately impact one stakeholder over another. This is not always an easy issue to decipher, however, since there can be so many complicating factors such as differences between sources in size, pollutant characteristics, design or layout, control technologies, economics, and the list goes on....

Chadderton et al.(1981) provide some interesting examination of these issues regarding methods to establish WLAs among interacting discharges. The following five WLA methods were reviewed for a situation involving five interacting discharges of BOD:

- Equal percent removal or equal percent treatment
- Equal effluent concentration
- Equal incremental cost above minimum treatment (normalized on the basis of volumetric flow rate)
- Effluent concentration inversely proportional to pollutant mass inflow rate
- Modified optimization (i.e., least cost solution which includes the minimum treatment requirements of the technology based controls)

A comparison of the methods was made based on cost, equity, efficient use of stream assimilative capacity, and sensitivity to fundamental stream quality data. The authors concluded that the "equal percent treatment" was preferable in the example stream because of the method's insensitivity to data errors and accepted use by several states. However, this is not to say that the other methods may not be preferable under different circumstances or based upon other decision criteria.

8.3.4 Opportunities for Unique Solutions

Attempts to split up the pie often run into conflicts between stakeholders. Everyone wants the largest slice they can have for fear of not having enough now or sometime in the future. As environmental restrictions continue to be added, the competition for allocations seems to increase. Thus, EPA and States are finding that the time is ripe for unique solutions to the allocation dilemma.

Pollutant trading is one solution drawing considerable attention. The basic premise of pollution trading is that it is more cost effective to reach the water quality goals by targeting controls where they will do the most good. Therefore, rather than applying across the board control requirements, participants are allowed to pool resources and achieve targeted pollutant loading levels through the most cost effective means possible.

Pollutant trading can occur between point sources (referred to as "point-point trading") or between PS and NPS ("point-nonpoint trading"). Some examples of pilot pollutant trading programs are provided in a document entitled, "Incentive Analysis for Clean Water Act Reauthorization: Point Source/Nonpoint Source Trading for Nutrient Discharge Reductions," (U.S. EPA, 1992a). In addition, a summary of the "Administator's Point/Nonpoint Source Trading Initiative Meeting" was published by EPA (1992b).

While pollutant trading is not a tool that will work in every case, it represents a concerted effort toward finding acceptable solutions to the issues arising from the mandate to establish TMDLs, WLAs and LAs for water quality limited waterbodies.

8.3.5 Planning for the Future

The process of establishing TMDLs allows agencies a greater opportunity to plan for the future. Via monitoring, data analysis and modeling, EPA and States are left with better knowledge of the assimilative capacity of a system. Thus, when it comes time to assign WLAs/LAs and a MOS, information regarding the current state of water quality and activities within the watershed can be related to future considerations.

How much of the assimilative capacity is remaining? Regardless of the answer, the information will be helpful to planning efforts of all stakeholders. For instance, if assimilative capacity is exhausted in an area (a likelihood for impaired waters) and state-of-the-art controls are already in place, then its known that additional growth in the watershed will depend on either improving the state-of-the-art or source reduction. While it may not be the answer some want to hear, it does give stakeholders valuable information regarding where efforts for the future need to be targeted.

On the other hand, the TMDL process can be used to plan ahead to avoid early exhaustion of the available assimilative capacity. Knowledge of future activities regarding land-disturbing activities, population growth, etc. can be used to estimate future wasteflows and NPS loads. Rather than allocating out all of the available assimilative capacity to existing sources, it may be wiser to hold a reserve that covers the future "build out" within a watershed. Given varying growth rates and land use patterns, it may not always be reasonable or even feasible to carry out future analyses to "build out" levels. But, planning for the future at some level may help to avoid future conflicts caused by reallocations due to the failure to adequately consider future activities. States using a basinwide planning approach have indicated that the basinwide planning framework better allows for this future planning effort to take place (U.S. EPA, 1992c).

8.3.6 Defensibility of TMDL Allocations

Competition for allocations and the dislike of regulatory controls by some will undoubtedly lead to the challenge of some TMDL allocation strategies. Therefore, in order to stand up to such challenges, the TMDL developer should be sure that the TMDL allocation strategy is defensible. Defensibility will be facilitated by three things: sound science, good policy judgment, and good documentation. Given the legal mandates to develop TMDLs, the burden of proof will be on those who challenge the TMDL to show that it is inappropriate. Therefore, if the TMDL developer has used the decision processes outlined in this document and based his/her recommendations in sound science, and if the recommendations are supported in law or sound policy, and the process is well documented to demonstrate these facts, then the risk of overturning a TMDL strategy should be reduced.

8.4 Translating WLAs into NPDES Permit Requirements

The NPDES permit is the mechanism for translating WLAs into enforceable requirements for point sources. The National Pollutant Discharge Elimination System (NPDES) is set forth in section 402 of the CWA. Under the NPDES program, permits are required for the discharge of pollutants from most point source discharges into the waters of the United States (see 40 CFR 122 for applicability). While an NPDES permit authorizes a point source facility to discharge, it also subjects the permittee to legally enforceable requirements set forth in the permit.

8.4.1 Effluent Limitations

One of the ways in which WLAs are translated into permits is through the establishment of effluent limitations. Effluent limitations impose restrictions on the quantities, discharge rates, and/or concentrations of specified pollutants in the point source discharge. Effluent limitations reflect either minimum federal/state technology-based guidelines or levels needed to protect water quality, whichever is more stringent. By definition, TMDLs involve WLAs that are more stringent than technology-based limits in order to protect WQSs and are therefore used to establish appropriate effluent limitions.

Converting WLAs into Permit Limits

- Consider regulatory format requirements (e.g. daily max)
- Consider effluent variability and probability basis for limit
- Convert WLA parameters to compliance parameters where necessary

As is required by 40 CFR 122.45(d), effluent limitations are usually expressed as some combination of daily maximum, weekly average, and monthly average concentrations. The limits chosen to fit these formats often reflect separate modeling analyses for different water quality criteria. For example, daily maximum limitations for toxics often reflect near-field modeling analyses using acute criteria, whereas longer term average limits reflect well-mixed conditions and chronic toxicity criteria.

Effluent limitations will not always be the same as the values for parameters derived for WLAs. As stated in the Toxics TSD (U.S. EPA, 1991), "Direct use of a WLA as a permit limit creates a significant risk that the WLA will be enforced incorrectly, since effluent variability and the probability basis for the limit are not considered specifically. For example, the use of a steady state WLA typically establishes a level of effluent quality with the assumption that it is a value never to be exceeded. The same values used directly as a permit limit could allow the WLA to be exceeded without observing permit violations if compliance monitoring was infrequent." In this regard, permit writers are encouraged to take a statistical approach to establishing limits from WLAs.

In addition, WLAs often need to be converted to parameters that can be used better for compliance judgment. For instance, modeling analyses often produce WLAs for oxygen-demanding wastes in terms of ultimate CBOD and NBOD. However, these parameters are difficult to use for compliance judgment because of the time and complexity involved in measurement. Therefore, CBOD and NBOD model results are converted via ratios to corresponding CBOD5 and NH3-N limitations which are easier to measure and, because of their shorter time for analysis, allow for more rapid response to any compliance problems that may arise.

8.4.2 Special Permit Conditions

In addition to effluent limitations, some WLAs may need to be incorporated through special conditions in the NPDES permit. This may be particularly true where non-numeric criteria are involved in the TMDL. Conditions can require specific modes of operation or actions by the permittee to protect water quality, and these can be made to address dynamic situations. For example, a discharger to a tidal area could be required to discharge only during a specific portion of the tidal cycle to ensure complete flushing or to reduce peak concentrations caused by pooling of the effluent.

Permit conditions can also be used to incorporate schedules for the permittee to come into compliance with new limitations that are applied due to new or revised TMDL/WLA requirements. Similarly, special conditions can be input to the permit to require any special monitoring (including in the receiving waters) that is judged necessary to ensure the adequacy of the TMDL/WLA and to document impacts on water quality from the point source.

8.4.3 The Permit Issuance Process

NPDES permit requirements are not finalized until they have gone completely through the permit issuance process. This includes periods for public review and, in the case of major permits where states are the permit issuing authority, potential review by EPA. Thus, it is possible that certain objections learned through the review process could lead to the need for a change in permit requirements and, potentially, the TMDL or WLA strategy.

8.5 Translating LAs into BMPs

Unlike NPDES permits for point sources, there are no corresponding permit requirements for nonpoint sources. Instead, load allocations are addressed, where necessary, through implementation of best management practices (BMPs). In some cases, states have certain mandatory BMP requirements for specific land use activities. However, by in large, implementation of BMPs occurs through voluntary and incentive programs such as government cost sharing.

While LAs may be used to target BMP implementation within a watershed, translation of LAs into specific BMP implementation programs can be problematic. One reason for this difficulty is that there are often many agencies involved in BMP implementation. Rather than a single oversight agency, as is the case for NPDES permits, BMP implementation can typically include federal, state, and local levels of involvement. Many times the objectives of the varying agencies are different, which makes coordination and enforcement difficult.

In addition, as is reflected by much of the material in this document, it is not always easy to accurately predict the effectiveness of BMPs. Therefore, its not easy to determine how much of an effort to place into BMP implementation in order to comply with LAs. As discussed earlier, TMDL strategies that are heavily dependent on loading reductions through LAs are strong candidates for use of a phased approach because of the inherent uncertainty in the system response to BMPs. A phased approach can provide time for improved interagency coordination and establishment of long-term watershed water quality monitoring programs to evaluate BMP effectiveness and compliance with LAs.

8.6 Communicating the Results

In order for TMDL strategies to be successful, those parties likely to be effected by the TMDL (i.e., the "stakeholders") should be involved in the TMDL development process. Effective communication is a key element to the public participation process. Stakeholders should be made aware of decisions regarding priority status of a waterbody, modeling results or data analyses used to establish TMDLs for the waterbody, and the pollutant control strategies resulting from the TMDL (i.e., WLAs and LAs).

Methods for Communicating TMDLs

- Issue public notices
- Hold public meetings/hearings
- Circulate basin or watershed plans for public review
- Use education and outreach programs to expand general knowledge of TMDL process

Public notices and follow up meetings or hearings are one way in which stakeholders can be kept informed. Federal regulations require public notice of NPDES permits and, therefore, those notices can be used to convey additional information about the TMDL process. However, it's best to try and involve the public earlier on in the process in order to avoid situations in which information is learned by the permit issuing authority that would have changed the way in which the TMDL process occurred. The TMDL Process document (1992) recommends that the stakeholders be involved from the very beginning, when the waters are being considered for listing on the 303(d) list along with an assigned priority. Therefore, public notices of the 303(d) list are encouraged.

In addition, basinwide planning can facilitate communication and public involvement. States that are organizing their program activities around a cyclical period can schedule in public meetings throughout the process to keep interested stakeholders informed. In addition, where a basin or watershed plan is prepared, the document can be circulated for public review prior to adoption and implementation of the plan. Typically, due to their comprehensive nature, basinwide plans also help communicate

TMDLs to the public because the plans provide a "big picture" that sets the context for the TMDL. In this manner, the public learns what monitoring and assessment is being performed, which parameters states or EPA are prioritizing for management, what management alternatives have been considered, and which management strategy has been selected for implementation.

Finally, where success of a TMDL strategy is dependent on other agencies for implementation and enforcement of control mechanisms, it is critical that those agencies be brought into the decision making process and soon as the need for their involvement is realized. This means that communication typically must begin with these agencies before final TMDL strategies are promulgated. The results of data and modeling analyses that are used to begin considering management actions should be effectively communicated to those agencies and the process of determining potential management alternatives should then proceed jointly. Subsequently, TMDL results that are communicated through a unified front of administering agencies more clearly demonstrate to the other stakeholders that the requirements have been well planned and coordinated.

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