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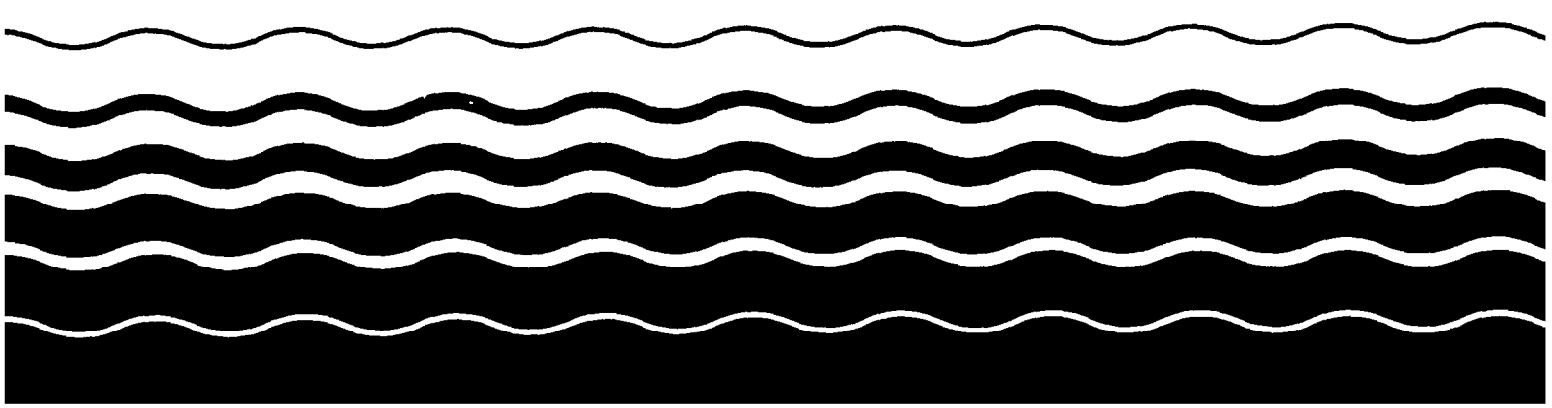

United States
Environmental Protection
Agency

Office of Water
Washington, DC

TECHNICAL SUPPORT DOCUMENT FOR WATER QUALITY-BASED TOXICS CONTROL

This copy represents the second printing of this document.

Changes made to this document reflect corrections of typographical errors and the following update of the interim guidance on criteria for metals: The Agency has issued "Interim Guidance Interpretation and Implementation Aquatic Life Criteria for Metals." The interim guidance supersedes criteria document statements expressing criteria in terms of a acid soluble analytical method and also the metals discussion of Section 5.7.3. The availability of this document appeared in the June 5, 1992 Federal Register (Vol. 57, No. 109, pg. 24401).

March 1991
Office of Water Enforcement and Permits
Office of Water Regulations and Standards
U.S. Environmental Protection Agency
Washington, DC 20460

FOREWORD

The U.S. Environmental Protection Agency (EPA) and the State pollution control agencies have been charged with enforcing the laws regarding pollution of the natural environment. Environmental pollution is an urgent and continuing problem and, consequently, the laws grant considerable discretion to the control authorities to define environmental goals and develop the means to attain them. Establishing environmentally protective levels and incorporating them in a decisionmaking process entails a considerable amount of scientific knowledge and judgment. One area where scientific knowledge is rapidly changing concerns the discharge of toxic pollutants to the Nation's surface waters.

This document provides technical guidance for assessing and regulating the discharge of toxic substances to the waters of the United States. It was issued in support of EPA regulations and policy initiatives involving the application of biological and chemical assessment techniques to control toxic pollution to surface waters. This document is agency guidance only. It does not establish or affect legal rights or obligations. It does not establish a binding norm and is not finally determinative of the issues addressed. Agency decisions in any particular case will be made applying the law and regulations on the basis of specific facts when permits are issued or regulations promulgated.

This document is expected to be revised periodically to reflect advances in this rapidly evolving area. Comments from users will be welcomed. Send comments to U.S. EPA, Office of Water Enforcement and Permits, 401 M Street, SW, Mailcode EN366, Washington, DC 20460.

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ACKNOWLEDGMENT

The preparation of the revised *Technical Support Document for Water Quality-based Toxic Control* began with a 3-day conference held in Williamsburg, Virginia, in December 1988. Representatives of EPA Headquarters and Regions, States, private industry, municipalities, academia, and various interest groups attended this meeting and provided valuable input. The principal authors of this document were Bill Swietlik, James Taft, Jacqueline Romney, Kathryn Smith, John Cannell, Robert Wood, James Pendergast, and Rick Brandes of the Permits Division; Sheila Frace and Margarete Heber of the Enforcement Division, Elizabeth Southerland and Richard Healy of the Assessment and Watershed Protection Division; and Charles Delos, Warren Banks, and Robert April of the Criteria and Standards Division. Listed below are the contributors to specific chapters of this document.

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EXECUTIVE SUMMARY

The revised *Technical Support Document for Water Quality-based Toxics Control* (TSD) provides States and Regions with guidance on procedures for use in the water quality-based control of toxic pollutants. It presents recommendations to regulatory authorities faced with the task of controlling the point source discharge of toxic pollutants to the Nation's waters. The document provides guidance for each step in the water quality-based toxics control process from standards development to compliance monitoring. Both human health and aquatic toxicity issues are incorporated into the discussions throughout the document. The overall approach in this revised document provides additional explanations and rationales based on accumulated experience and data for the various recommendations that were made in the original TSD. The following is a brief synopsis of the guidance provided in the TSD.

Approaches to Water Quality-based Toxics Control

The Environmental Protection Agency's (EPA) surface toxics control regulation, 54 *FR* 23868, June 2, 1989, established specific requirements that the "integrated" approach be used in water quality-based toxics control. The "integrated" approach consists of whole effluent and chemical-specific approaches as a means of protecting aquatic life and human health. As techniques are made available for implementing biocriteria, they too should be integrated into the water quality-based toxics control, thus creating a triad of approaches: whole effluent, chemical-specific, and biological assessments. Each approach has its limitations and thus, exclusive use of one approach alone cannot ensure required protection of aquatic life and human health. The advantages/disadvantages of each approach and how the integrated approach creates an effective toxics control program are discussed in the text.

The whole effluent approach to toxics control involves the use of toxicity tests and water quality criteria for the parameter "toxicity" to assess and control the aggregate toxicity of effluents. New references and information in support of the whole effluent toxicity assessment and control approach have been included in Chapter 1 and associated appendices (e.g., precision data, justifications for acute-to-chronic ratio recommendations, information on analytical variability in toxicity testing). The chemical-specific approach to aquatic life toxics control relies on numeric water quality criteria in State standards and interpretations of State narrative standards to assess and control specific toxicants individually.

Water Quality Standards and Criteria

Where specific numerical criteria for a chemical or biological parameter (such as toxicity) are absent, compliance with water quality standards must be based on the general narrative criteria and on protection of the designated uses. For many pollutants, EPA's recommended criteria may be used, or criteria may be developed using data from the Integrated Risk Information System, or data on the toxicological effects of the pollutant found either in the literature or required of a discharger.

Aquatic impacts occur not only from the magnitude of a pollutant, but also from the duration and frequency with which criteria are exceeded. EPA's recommended aquatic life criteria for both individual toxicants and whole effluent toxicity are specified as two numbers: the criterion continuous concentration is applied as a 4-day average concentration; and the criterion maximum concentration is applied as an 1-hour average concentration. The frequency with which criteria are allowed to be exceeded depends on site-specific factors as explained in the text.

Strictly speaking the term "criteria" means EPA guidance formally published under the authority of Section 304(a) of the Clean Water Act. The toxicity level recommendations have not been so published. However, they represent EPA's carefully developed technical recommendation, and so are referred to in this document in the same manner as other criteria.

EPA's recommended criteria for whole effluent toxicity are as follows: to protect aquatic life against chronic effects, the ambient toxicity should not exceed 1.0 chronic toxic unit (TU_c) to the most sensitive of at least three different test species. For protection against acute effects, the ambient toxicity should not exceed 0.3 acute toxic units (TU_a) to the most sensitive of at least three different test species.

EPA has developed recommended human health criteria, which are called reference ambient concentrations (RACs). In the absence of EPA's recommended criteria, States may calculate RACs based on the equations in the text. In addition, the need for sediment and biological criteria in State water quality standards is discussed.

Effluent Characterization

This chapter contains completely revised effluent characterization discussions and recommendations. It includes streamlined procedures (as compared to the original TSD) for predicting the likely impacts of toxic effluents on aquatic life and human health. Recommendations are provided for determining, either with or without actual effluent data, whether a discharge causes, has the reasonable potential to cause, or contributes to an excursion above a State water quality standard. These effluent characterization procedures can be performed in one step and do not include initial screening followed by definitive data generation as was recommended in the original TSD.

The revised effluent characterization procedures for assessing potential human health impacts now include control of bioaccumulative chemicals.

Exposure and Wasteload Allocation

A goal of permit writers is to determine what effluent composition will protect aquatic organisms and human health. Exposure assessment includes an analysis of how much of the waterbody is subject to the exceedance of criteria, for how long, and how frequently. The first step is to evaluate the effluent plume dispersion. If mixing is not rapid and complete and if State standards allow a mixing zone, the wasteload allocation also must be based

on a mixing zone analysis. Chapter 5 describes the means to assess dilution at the edge of a mixing zone. As with the original TSD, ambient criteria to control acute toxicity to aquatic life may be met within a short distance of the outfall. However, this provision is no longer restricted to outfalls that have a high-rate diffuser.

If mixing is rapid and complete, there are several models that can be used to assess exposure. Steady-state models assume that the effluent concentration is constant and that the duration and frequency with which criteria are exceeded can be reflected entirely by selecting a design flow in the receiving water of appropriate averaging period and frequency.

Another means of modeling exposure is to use computer models that incorporate variability of the individual inputs (such as effluent flow and concentration, receiving water flow, temperature, background concentration, etc.). These models are termed dynamic models and are more accurate than steady-state models in reflecting or predicting exposure provided adequate data exist. The acceptable effluent condition derived using these models is expressed as the effluent long-term average and variance, which greatly simplifies derivation of permit limits. Three dynamic modeling approaches are described along with instructions for their use.

Permit Requirements

The requirements of a wasteload allocation (WLA) must be translated into a permit limit in the wastewater discharge permit. In many cases permit limits will be different than the WLA to reflect different assumptions and means of expressing effluent quality. Three types of WLAs are identified, and recommendations are provided for deriving permit limits to properly enforce each type of WLA. Other permit-related issues such as permit documentation and how to express limitations are discussed. In addition, guidance for requiring and conducting toxicity reduction evaluations is presented.

Compliance Monitoring

The compliance monitoring and enforcement process for water quality-based permits summarized in Chapter 6 is based on existing regulation and guidance. As with technology-based permits, any failure to meet a limit is a violation, and every violation must be reviewed to determine the appropriate response. Whole effluent toxicity monitoring and enforcement concepts embodied in the *Compliance Monitoring and Enforcement Strategy for Toxics Control* (January 19, 1989) have been added to this revision.

LIST OF ABBREVIATIONS

AA	atomic absorption
ACR	acute-to-chronic ratio
ADI	acceptable daily intake
AML	average monthly limit
ATC	acceptable tissue concentration
ATE	acute toxicity endpoint
AVS	acid volatile sulfides
BAF	bioaccumulation factor
BAT	best available technology
BCF	bioconcentration factor
BCT	best conventional technology
BMP	best management practice
BOD	biochemical oxygen demand
BPJ	best professional judgment
BPT	best practicable technology
CCC	criteria continuous concentration
CEAM	Center for Exposure Assessment Modeling (EPA)
CETTP	Complex Effluent Toxicity Testing Program
CFR	Code of Federal Regulations
CHC	chemical of highest concern
CMC	criteria maximum concentration
CTE	chronic toxicity endpoint
CV	coefficient of variation
CWA	Clean Water Act
DF	dilution factor
DMR	discharge monitoring report
DO	dissolved oxygen
EC	effect concentration
ECAO	Environmental Criteria and Assessment Office
EMS	Enforcement Management System
EP	equilibrium partitioning
EPA	Environmental Protection Agency
ERL	Environmental Research Laboratory (EPA)
FAV	final acute value
FDA	Food and Drug Administration
FM	food chain multipliers
GC/MS	gas chromatograph/mass spectrometer
HHC	human health criteria
HPLC	high-pressure liquid chromatography
IC	inhibition concentration
IRIS	Integrated Risk Information System (EPA)
LA	load allocation
LC	lethal concentration
LOAEL	lowest observed adverse effect level
LOEC	lowest observed effect concentration
LTA	long-term average
MCL	maximum contaminant levels
MDL	maximum daily limit
MERS	Monticello Ecological Research Station
ML	minimum level
NOAEL	no observed adverse effect level
NOEC	no observed effect concentration
NPDES	National Pollutant Discharge Elimination System
NTIS	National Technical Information Service
ONRW	outstanding national resource waters
PCS	Permit Compliance System
POTW	publicly owned treatment works
PQL	practical quantitation limit
q_1^*	cancer potency factor
QA/QC	quality assurance/quality control
QNCR	quarterly noncompliance report
QSAR	quantitative structure-activity relationships
RAC	reference ambient concentration
RfD	reference dose
RWC	receiving water concentration
SQC	sediment quality criteria
STORET	storage and retrieval of water quality information
TIE	toxicity identification evaluation
TMDL	total maximum daily load
TRE	toxicity reduction evaluation
TSD	technical support document
TSS	total suspended solids
TTO	total toxic organics
TU	toxic unit
TU_a	acute toxic unit
TU_c	chronic toxic unit
WQS	water quality standard
WLA	wasteload allocation

MODELING ABBREVIATIONS

ARM	agricultural runoff model
CHNTRN	Channel Transport Model
CETIS	Complex Effluent Toxicity Information System
CIS	Chemical Information System
CORMIX 1	Cornell Mixing Zone Expert System
CTAP	Chemical Transport and Analysis Program
DESCON	computer program that estimates design conditions
DFLOW	computer program that calculates biologically based design flows
DYNHYD4	hydrodynamic model
DYNTOX	dynamic toxics model
EXAMS-II	Exposure Analysis Modeling System
FCM2	WASP Food Chain Model
FETRA	Finite Element Transport Model
FGETS	Food and Gill Exchange of Toxic Substances
FLOSTAT	U.S. Geological Survey computer program that estimates the arithmetic mean flow and 7Q10 of rivers and streams
HHDFLOW	historic daily flow program
HSPF	Hydrologic Simulation Program - FORTRAN
MEXAMS	Metals Exposure Analysis Modeling System
MINTEQA2	Equilibrium Metals Speciation Model
MICH	Michigan River Model
NPS	Nonpoint Source Model for Urban and Rural Areas
PSY	steady-state, two-dimensional plume model
SARAH2	surface water assessment model for back calculating reductions in biotic hazardous wastes
SERATRA	Sediment Contaminant Transport Model
SLSA	Simplified Lake/Stream Analysis
TODAM	Transport One-Dimensional Degradation and Migration Model
TOXIWASP	Chemical Transport and Fate Model
TOXI4	a subset of WASP4
TOXIC	Toxic Organic Transport and Bioaccumulation Model
UDKHDEN	three-dimensional model used for single or multiple port diffusers
ULINE	uniform linear density flume model
UMERGE	two-dimensional model used to analyze positively buoyant discharge
UOUTPLM	cooling tower plume model adapted for marine discharges
UPLUME	numerical model that produces flux-average dilutions
WASP4	water quality analysis program
WASTOX	Estuary and Stream Quality Model
WQAB FLOW	water quality analysis system flow data subroutine

GLOSSARY

absolute toxicity is the toxicity of the effluent without considering dilution.

acute means a stimulus severe enough to rapidly induce an effect; in aquatic toxicity tests, an effect observed in 96 hours or less typically is considered acute. When referring to aquatic toxicology or human health, an acute affect is not always measured in terms of lethality.

acute-to-chronic ratio (ACR) is the ratio of the acute toxicity of an effluent or a toxicant to its chronic toxicity. It is used as a factor for estimating chronic toxicity on the basis of acute toxicity data, or for estimating acute toxicity on the basis of chronic toxicity data.

acutely toxic conditions are those acutely toxic to aquatic organisms following their short-term exposure within an affected area.

acute toxicity endpoints (ATE) are toxicity test results, such as an LC_{50} (96 hours) and EC_{50} (48 hours), which describe a stimulus severe enough to rapidly induce an effect on aquatic organisms.

additivity is the characteristic property of a mixture of toxicants that exhibits a total toxic effect equal to the arithmetic sum of the effects of the individual toxicants.

ambient toxicity is measured by a toxicity test on a sample collected from a waterbody.

antagonism is the characteristic property of a mixture of toxicants that exhibits a less-than-additive total toxic effect.

antidegradation policies are part of each State's water quality standards. These policies are designed to protect water quality and provide a method of assessing activities that may impact the integrity of the waterbody.

aquatic community is an association of interacting populations of aquatic organisms in a given waterbody or habitat.

averaging period is the period of time over which the receiving water concentration is averaged for comparison with criteria concentrations. This specification limits the duration of concentrations above the criteria.

bioaccumulation is the process by which a compound is taken up by an aquatic organism, both from water and through food.

bioaccumulation factor (BAF) is the ratio of a substance's concentration in tissue versus its concentration in ambient water, in situations where the organism and the food chain are exposed.

bioassay is a test used to evaluate the relative potency of a chemical or a mixture of chemicals by comparing its effect on a living organism with the effect of a standard preparation on the same type of organism. Bioassays frequently are used in the pharmaceutical industry to evaluate the potency of vitamins and drugs.

bioavailability is a measure of the physicochemical access that a toxicant has to the biological processes of an organism. The less the bioavailability of a toxicant, the less its toxic effect on an organism.

bioconcentration is the process by which a compound is absorbed from water through gills or epithelial tissues and is concentrated in the body.

bioconcentration factor (BCF) is the ratio of a substance's concentration in tissue versus its concentration in water, in situations where the food chain is not exposed or contaminated. For nonmetabolized substances, it represents equilibrium partitioning between water and organisms.

biological assessment is an evaluation of the biological condition of a waterbody using biological surveys and other direct measurements of resident biota in surface waters.

biological criteria, also known as **biocriteria**, are narrative expressions or numeric values of the biological characteristics of aquatic communities based on appropriate reference conditions. Biological criteria serve as an index of aquatic community health.

biological integrity is the condition of the aquatic community inhabiting unimpaired waterbodies of a specified habitat as measured by community structure and function.

biological monitoring, also known as **biomonitoring**, describes the living organisms in water quality surveillance used to indicate compliance with water quality standards or effluent limits and to document water quality trends. Methods of biological monitoring may include, but are not limited to, toxicity testing such as ambient toxicity testing or whole effluent toxicity testing.

biological survey or biosurvey is the collecting, processing, and analyzing of a representative portion of the resident aquatic community to determine its structural and/or functional characteristics.

biomagnification is the process by which the concentration of a compound increases in species occupying successive trophic levels.

cancer potency slope factor (q1*) is an indication of a chemical's human cancer-causing potential derived using animal studies or epidemiological data on human exposure. It is based on extrapolating high-dose levels over short periods of time to low-dose levels and a lifetime exposure period through the use of a linear model.

chronic means a stimulus that lingers or continues for a relatively long period of time, often one-tenth of the life span or more. Chronic should be considered a relative term depending on the life span of an organism. The measurement of a chronic effect can be reduced growth, reduced reproduction, etc., in addition to lethality.

chronic toxicity endpoints (CTE) are results, such as a no observed effect concentration, lowest observed effect concentration, effect concentration, and inhibition concentration based on observations of reduced reproduction, growth, and/or survival from life cycle, partial life cycle, and early life stage tests with aquatic animal species.

coefficient of variation (CV) is a standard statistical measure of the relative variation of a distribution or set of data, defined as the standard deviation divided by the mean.

community component is a general term that may pertain to the biotic guild (fish, invertebrates, algae), the taxonomic category (order, family, genus, species), the feeding strategy (herbivore, omnivore, predator), or the organizational level (individual, population, assemblage) of a biological entity within the aquatic community.

completely mixed condition means no measurable difference in the concentration of a pollutant exists across a transect of the waterbody (e.g., does not vary by 5 percent).

continuous simulation model is a fate and transport model that uses time series input data to predict receiving water quality concentrations in the same chronological order as that of the input variables.

criteria continuous concentration (CCC) is the EPA national water quality criteria recommendation for the highest instream concentration of a toxicant or an effluent to which organisms can be exposed indefinitely without causing unacceptable effect.

criteria maximum concentration (CMC) is the EPA national water quality criteria recommendation for the highest instream concentration of a toxicant or an effluent to which organisms can be exposed for a brief period of time without causing an acute effect.

critical life stage is the period of time in an organism's lifespan in which it is the most susceptible to adverse effects caused by exposure to toxicants, usually during early development (egg, embryo, larvae). Chronic toxicity tests are often run on critical life stages to replace long duration, life-cycle tests since the most toxic effect usually occurs during the critical life stage.

design flow is the flow used for steady-state wasteload allocation modeling.

designated uses are those uses specified in water quality standards for each waterbody or segment whether or not they are being attained.

discharge length scale is the square root of the cross-sectional area of any discharge outlet.

diversity is the number and abundance of biological taxa in a specified location.

effect concentration (EC) is a point estimate of the toxicant concentration that would cause an observable adverse effect (such as death, immobilization, or serious incapacitation) in a given percentage of the test organisms.

equilibrium partitioning (EP) is a method for generating sediment criteria that focuses on the chemical interaction between sediments and contaminants.

final acute value (FAV) is an estimate of the concentration of the toxicant corresponding to a cumulative probability of 0.05 in the acute toxicity values for all genera for which acceptable acute tests have been conducted on the toxicant.

frequency is how often criteria can be exceeded without unacceptably affecting the community.

genotoxic is the ability of a substance to damage an organism's genetic material (DNA).

harmonic mean flow is the number of daily flow measurements divided by the sum of the reciprocals of the flows. That is, it is the reciprocal of the mean of reciprocals.

inhibition concentration (IC) is a point estimate of the toxicant concentration that would cause a given percent reduction (e.g., IC_{25}) in a nonlethal biological measurement of the test organisms, such as reproduction or growth.

lethal concentration is the point estimate of the toxicant concentration that would be lethal to a given percentage of the test organisms during a specific period.

lipophilic is a high affinity for lipids (fats).

load allocations (LA) are the portion of a receiving water's total maximum daily load that is attributed either to one of its existing or future nonpoint sources of pollution or to natural background sources.

lognormal probabilistic dilution model calculates the probability distribution of receiving water quality concentrations from the lognormal probability distributions of the input variables.

log P (also expressed as log k_{ow} or as n-octanal/water partition coefficient) is the ratio, in a two-phase system of n-octanol and water at equilibrium, of the concentration of a chemical in the n-octanol phase to that in the water phase.

lowest observed adverse effect level (LOAEL) is the lowest concentration of an effluent or toxicant that results in statistically significant adverse health effects as observed in chronic or subchronic human epidemiology studies or animal exposure.

magnitude is how much af a pollutant (or pollutant parameter such as toxicity), expressed as a concentration or toxic unit is allowable.

minimum level (ML) refers to the level at which the entire analytical system gives recognizable mass spectra and acceptable calibration points when analyzing for pollutants of concern. This level corresponds to the lowest point at which the calibration curve is determined.

mixing zone is an area where an effluent discharge undergoes initial dilution and is extended to cover the secondary mixing in the ambient waterbody. A mixing zone is an allocated impact zone where water quality criteria can be exceeded as long as acutely toxic conditions are prevented.

Monte Carlo simulation is a stochastic modeling technique that involves the random selection of sets of input data for use in repetitive model runs in order to predict the probability distributions of receiving water quality concentrations.

no observed adverse effect level (NOAEL) is a tested dose of an effluent or a toxicant below which no adverse biological effects are observed, as identified from chronic or subchronic human epidemiology studies or animal exposure studies.

no observed effect concentration (NOEC) is the highest tested concentration of an effluent or a toxicant at which no adverse effects are observed on the aquatic test organisms at a specific time of observation. Determined using hypothesis testing.

nonthreshold effects are associated with exposure to chemicals that have no safe exposure levels (i.e., cancer).

permit averaging period is the duration of time over which a permit limit is calculated (days, weeks, or months).

persistent pollutant is not subject to decay, degradation, transformation, volatilization, hydrolysis, or photolysis.

priority pollutants are those pollutants listed by the Administrator under CWA Section 307(a).

probability is a number expressing the likelihood of occurrence of a specific event, such as the ratio of the number of outcomes that will produce a given event to the total number of possible outcomes.

probability distribution is a mathematical representation of the probabilities that a given variable will have various values.

practical quantitation limit (PQL) is a correction factor, sometimes arbitrarily defined, used to account for uncertainty in measurement precision.

reasonable potential is where an effluent is projected or calculated to cause an excursion above a water quality standard based on a number of factors including, as a minimum, the four factors listed in 40 *CFR* 122.44(d)(1)(ii).

receiving water concentration (RWC) is the concentration of a toxicant or the parameter toxicity in the receiving water after mixing (formerly termed "instream waste concentration" [IWC]).

recurrence interval is the average number of years within that a variable will be less than or equal to a specified value. This term is synonymous with return period.

reference ambient concentration (RAC) is the concentration of a chemical in water that will not cause adverse impacts to human health. RAC is expressed in units of mg/l.

reference tissue concentration (RTC) is the concentration of a chemical in edible fish or shellfish tissue that will not cause adverse impacts to human health when ingested. RTC is expressed in units of mg/kg.

reference dose (RfD) is an estimate of the daily exposure to human population that is likely to be without an appreciable risk of deleterious effect during a lifetime; derived from nonobserved adverse effect level or lowest observed adverse effect level.

relative toxicity is the toxicity of the effluent when it is mixed with the receiving water, or a dilution water of similar composition for toxicity testing.

slug flow sampling is a monitoring procedure that follows the same slug of wastewater throughout its transport in the receiving water. Water quality samples are collected at receiving water stations, tributary inflows, and point source discharges only when a dye slug or tracer passes that point.

steady-state model is a fate and transport model that uses constant values of input variables to predict constant values of receiving water quality concentrations.

STORET is EPA's computerized water quality data base that includes physical, chemical, and biological data measured in waterbodies throughout the United States.

sublethal means a stimulus below the level that causes death.

synergism is the characteristic property of a mixture of toxicants that exhibits a greater-than-additive total toxic effect.

threshold effects result from chemicals that have a safe level (i.e., acute, subacute, or chronic human health effects).

total maximum daily load (TMDL) is the sum of the individual wasteload allocations and load allocations. A margin of safety is included with the two types of allocations so that any additional loading, regardless of source, would not produce a violation of water quality standards.

toxicity identification evaluation (TIE) is a set of procedures to identify the specific chemicals responsible for effluent toxicity.

toxicity reduction evaluation (TRE) is a site-specific study conducted in a stepwise process designed to identify the causative agents of effluent toxicity, isolate the sources of toxicity, evaluate the effectiveness of toxicity control options, and then confirm the reduction in effluent toxicity.

toxicity test is a procedure to determine the toxicity of a chemical or an effluent using living organisms. A toxicity test measures the degree of effect on exposed test organisms of a specific chemical or effluent.

toxics are those pollutants that have a toxic effect on living organisms. The CWA Section 307(a) "priority" pollutants are a subset of this group of pollutants.

toxic pollutants are those pollutants listed by the Administrator under CWA Section 307(a).

toxic units (TUs) are a measure of toxicity in an effluent as determined by the acute toxicity units or chronic toxicity units measured.

toxic unit acute (TU_a) is the reciprocal of the effluent concentration that causes 50 percent of the organisms to die by the end of the acute exposure period (i.e., 100 LC_{50}).

toxic unit chronic (TU_c) is the reciprocal of the effluent concentration that causes no observable effect on the test organisms by the end of the chronic exposure period (i.e., 100/NOEC).

water quality assessment is an evaluation of the condition of a waterbody using biological surveys, chemical-specific analyses of pollutants in waterbodies, and toxicity tests.

wasteload allocation (WLA) is the portion of a receiving water's total maximum daily load that is allocated to one of its existing or future point sources of pollution.

water quality criteria are comprised of numeric and narrative criteria. Numeric criteria are scientifically derived ambient concentrations developed by EPA or States for various pollutants of concern to protect human health and aquatic life. Narrative criteria are statements that describe the desired water quality goal.

water quality limited characterizes a stream segment in which it is known that water does not meet applicable water quality standards, and/or is not expected to meet applicable water quality standards even after application of technology-based effluent limitations.

water quality standard is a law or regulation that consists of the beneficial designated use or uses of a waterbody, the numeric and narrative water quality criteria that are necessary to protect the use or uses of that particular waterbody, and an antidegradation statement.

whole effluent toxicity is the total toxic effect of an effluent measured directly with a toxicity test.

2. WATER QUALITY CRITERIA AND STANDARDS

2.1 INTRODUCTION

The foundation of a water quality-based toxics control program consists of the State water quality standards applicable to the waterbody. The following discussion describes the regulatory and technical considerations for application of water quality standards.

2.1.1 Overview of Water Quality Standards

A water quality standard defines the water quality goals of a water body, or portion thereof, by designating the use or uses to be made of the water, by setting criteria necessary to protect the uses, and by establishing antidegradation policies and implementation procedures that serve to maintain and protect water quality. States adopt water quality standards to protect public health or welfare, enhance the quality of water, and serve the purposes of the Clean Water Act (CWA). "Serve the purposes of the Act" (as defined in Sections 101(a), 101(a)(2), and 303(c) of the Act) means that water quality standards should (1) include provisions for restoring and maintaining chemical, physical, and biological integrity of State waters; (2) provide, wherever attainable, water quality for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water ("fishable/swimmable"); and (3) consider the use and value of State waters for public water supplies, propagation of fish and wildlife, recreation, agriculture and industrial purposes, and navigation.

The CWA describes various uses of waters that are considered desirable and should be protected. These uses include public water supply, recreation, and propagation of fish and wildlife. The States are free to designate more specific uses (e.g., cold water and warm water aquatic life), or to designate uses not mentioned in the CWA, with the exception that waste transport and assimilation is not an acceptable designated use (see 40 *CFR* 131.10(a)). EPA's regulations emphasize the uses specified in CWA Section 101(a)(2), but do not preclude other beneficial uses and subcategories of uses as determined by the State.

When designating uses, States should give careful consideration to whether uses that will support the "fishable and swimmable" goal of Section 101(a)(2) are attainable. If the State does not designate uses in support of this goal, the State must perform a use attainability analysis under Section 131.10(j) of the standards regulation. States should designate uses for the waterbody that the State determines can be attained in the future. "Attainable uses" are those uses (based on the State's system of water use classification) that can be achieved when effluent limits under CWA Section 301(b)(1)(A) and (B) and Section 306 are implemented for point source discharges and when cost-effective and reasonable best management practices are implemented for nonpoint sources. The Water Quality Standards regulation specifies the conditions under which States may remove uses or establish subcategories of uses. Among these are that the State must provide opportunity for public hearing. In addition, uses that have been attained in the waterbody on or after November 28, 1975, whether or not they are included in the water quality standards, may not be removed unless a use requiring more stringent criteria is added. These uses are the "existing uses" as defined in 40 *CFR* 131.3(e). Also, uses that are attainable, as defined above, may not be removed. Removal of a "fishable/swimmable" use, or adoption of a subcategory of a "fishable/swimmable" use that requires less stringent criteria, requires the State to conduct a use attainability analysis. Technical guidance on conducting use attainability analyses is available from EPA (e.g., Chapter 3 of the *Water Quality Standards Handbook* (1983) [1], and *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses* (1983) [2].

In the Water Quality Standards regulation, Section 131.11 encourages States to adopt both numeric and narrative criteria. Aquatic life criteria should protect against both short-term (acute) and long-term (chronic) effects. Numeric criteria particularly are important where the cause of toxicity is known or for protection against pollutants with potential human health impacts or bioaccumulation potential. Numeric water quality criteria also may be the best way to address nonpoint source pollution problems. Narrative criteria can be the basis for limiting toxicity in waste discharges where a specific pollutant can be identified as causing or contributing to the toxicity but there are no numeric criteria in the State standards or where toxicity cannot be traced to a particular pollutant. Section 131.11(a)(2) requires States to develop implementation procedures that explain how the State will ensure that narrative toxics criteria are met.

EPA's water quality standards regulation requires each State to adopt, as part of its water quality standards, an antidegradation policy consistent with 40 *CFR* 131.12 and to identify the methods it will use for implementing the policy. Activities covered by the antidegradation policy and implementation methods include both point and nonpoint sources of pollution. Section 131.12 effectively sets out a three-tiered approach for the protection of water quality.

"Tier I" (40 *CFR* 131.12(a)(1)) of antidegradation maintains and protects existing uses and the water quality necessary to protect these uses. An existing use can be established by demonstrating that fishing, swimming, or other uses have actually occurred since November 28, 1975, or that the water quality is suitable to allow such uses to occur, whether or not such uses are designated uses for the waterbody in question. (Compare Sections 131.3(e) and 131.3(f) of the existing regulation.) For example, in an area where shellfish are propagating and surviving in a biologically suitable habitat, the shellfish use is existing, whether or not people are harvesting the shellfish. The aquatic life protection use is a broad category requiring further explanation, which may be found in the *Water Quality Standards Handbook*.

"Tier II" (Section 131.12(a)(2)) protects the water quality in waters whose quality is better than that necessary to protect "fishable/swimmable" uses of the waterbody. 40 *CFR* 131.12(a)(2) requires that certain procedures be followed and certain showings be made before lowering water quality in high-quality waters. These showings may be called an "antidegradation review." In no case may water quality on a Tier II waterbody be lowered to the level at which existing uses are impaired. The Tier II protection usually is applied on a parameter-by-parameter basis (called the definitional approach to Tier II). This approach is applied on a case-by-case basis so that, if the level of any parameter is better than water quality standards for that waterbody, then an antidegradation review will be performed for any activity that could reduce the level of that parameter.

Outstanding national resource waters (ONRWs) are provided the highest level of protection under the antidegradation policy (Tier III); no degradation is allowed. ONRWs include the highest-quality waters of the United States. However, the ONRW antidegradation classification also offers special protection for waters of "exceptional ecological significance," i.e., those waterbodies that are important, unique, or sensitive ecologically, but whose water quality, as measured by the traditional parameters such as dissolved oxygen or pH, may not be particularly high. Waters of exceptional ecological significance may also include waters whose characteristics cannot be described adequately by traditional parameters (such as wetlands and estuaries).

States may, at their discretion, adopt certain policies in their standards affecting the application and implementation of standards. For example, policies concerning mixing zones, variances, low-flow exemptions, and schedules of compliance for water quality-based permit limits may be adopted. Although these are areas of State discretion, EPA retains authority to review and approve or disapprove such policies (see 40 *CFR* 131.13). Guidance on these subjects is available from EPA's Office of Water Regulations and Standards, Criteria and Standards Division.

2.1.2 Water Quality Standards and State Toxics Control Programs

Applicable requirements for State adoption of water quality criteria for toxicants vary depending upon the toxicant. The reason for this is that the 1983 water quality standards regulation and the 1987 amendments to the CWA (Pub. L. 100-4) include more specific requirements for the particular toxicants listed in CWA Section 307(a). For regulatory purposes, EPA has translated the 65 compounds and families of compounds listed in Section 307(a) into 126 specific substances that EPA refers to as priority toxic pollutants. The 126 priority toxic pollutants are listed in Appendix A of 40 *CFR* Part 423. Because of the more specific requirements for priority toxic pollutants, it is convenient to organize the requirements applicable to State adoption of criteria for toxicants into three categories:

- Requirements applicable to priority toxic pollutants that have been the subject of CWA Section 304(a)(1) criteria guidance
- Requirements applicable to priority toxic pollutants that have not been the subject of CWA Section 304(a)(1) criteria guidance and
- Requirements applicable to all other toxicants (i.e., nonpriority toxic pollutants).

The criteria requirements applicable to priority toxic pollutants (i.e., the first two categories above), are specified in CWA Section 303(c)(2)(B). On December 2, 1988, EPA sent "Guidance for State Implementation of Water Quality Standards for CWA Section 303(c)(2)(B)" to each of its Regions and to each State water pollution control agency. The guidance contained three options for implementing the new numeric criteria requirements of the Act: (1) adopt Statewide numeric criteria in standards for all those priority toxic pollutants for which EPA has published national criteria; (2) adopt numeric criteria for only those priority toxic pollutants and those stream segments where the discharge or presence of the pollutant could reasonably be expected to interfere with designated uses; or (3) adopt a specific procedure in the standards to "translate" the State's narrative "free from toxics" standard to derived numeric criteria.

The transmittal memorandum for the Section 303(c)(2)(B) national guidance expresses the Office of Water position regarding priority toxic pollutants that may "reasonably be expected" to interfere with designated uses. That memorandum and guidance established a rebuttable presumption that <u>any</u> information indicating that such pollutants are discharged or present in surface waters (now or in the future) is sufficient justification to require adoption or derivation of numerical criteria. The goal is not just to identify pollutants that are already impacting surface waters, but rather to identify pollutants that may be impacting surface waters now, or have the potential to do so in the future. Lack of detailed or widespread monitoring data is not an acceptable basis to omit numerical (or derived numerical) criteria from water quality standards under Options 2 and 3. Even a limited amount of monitoring data indicating the discharge or presence of priority toxic pollutants in surface waters is sufficient basis to conclude that numerical (or derived numerical) criteria are necessary.

Where States select an Option 2 or 3 approach, States must include, as part of the rationale supporting the adopted standards, the information used in determining which priority toxic pollutants require criteria. Where there is uncertainty about the need for criteria for specific priority toxic pollutants, the State should adopt (or derive) criteria for such pollutants so as to err on the side of environmental protection and pollution prevention. This approach is appropriate given the general lack of monitoring data for priority toxic pollutants; it will provide maximum protection to the environment by anticipating, rather than reacting to, water quality problems.

For priority toxic pollutants for which EPA has not issued Section 304(a)(1) criteria guidance, CWA Section 303(c)(2)(B) requires States to adopt criteria based on biological monitoring or assessment methods. The phrase "biological monitoring or assessment methods" includes (1) whole effluent toxicity control methods, (2) biological criteria methods, or (3) other methods based on biological monitoring or assessment. The phrase "biological monitoring or assessment methods" in its broadest sense also includes criteria developed through translator procedures. This broad interpretation of that phrase is consistent with EPA's policy of applying chemical-specific, biological, and whole effluent toxicity methods independently in an integrated toxics control program. It also is consistent with the intent of Congress to expand

State standards programs beyond chemical-specific approaches.

Where EPA has not issued Section 304(a) criteria guidance, but available laboratory toxicity (bioassay) data are sufficient to support derivation of chemical-specific criteria, States should consider deriving and adopting numeric criteria for such priority toxic pollutants. This is particularly important where other components of a State's narrative criterion implementation procedure (e.g., whole effluent toxicity controls or biological criteria) may not ensure full protection of designated uses. For some pollutants, a combination of chemical-specific and other approaches is necessary (e.g., pollutants where bioaccumulation in fish tissue or water consumption by humans is a primary concern).

Criteria requirements applicable to toxicants that are not priority toxic pollutants (i.e., the third category above), are specified in the 1983 water quality standards regulation (see 40 *CFR* 131.11). Under these requirements, States must adopt criteria based on sound scientific rationale that cover sufficient parameters to protect designated uses. Both numeric and narrative criteria are addressed by these requirements.

Numeric criteria are required where such criteria are necessary to protect designated uses. Numeric criteria to protect aquatic life should be developed to address both short-term (acute) and long-term (chronic) effects. Saltwater species, as well as freshwater species, must adequately be protected. Adoption of numeric criteria is particularly important for toxicants known to be impairing surface waters and for toxicants with potential human health impacts (e.g., those with high bioaccumulation potential). Human health should be protected from exposure resulting from consumption of water and fish or other aquatic life (e.g., mussels, crayfish). Numeric water quality criteria also are useful in addressing nonpoint source pollution problems.

In evaluating whether chemical-specific numeric criteria for toxicants are required, States should consider whether other approaches (such as whole effluent toxicity criteria or biological controls) will ensure full protection of designated uses. As mentioned above, a combination of independent approaches may be required in some cases to support the designated uses and comply with the requirements of the water quality standards regulation (e.g., pollutants where bioaccumulation in fish tissue or water consumption by humans is a primary concern).

To supplement numeric criteria for toxicants, all States also have adopted narrative criteria for toxicants. Such narrative criteria are statements that describe the desired water quality goal, such as the following:

> All State waters must, at all times and flows, be free from substances that are toxic to humans or aquatic life.

EPA considers that the narrative criteria apply to all designated uses at all flows unless specified otherwise in a State's water quality standards. EPA also believes that no acutely toxic condition may exist in any State waters regardless of designated use (54 *FR* 23875).

Narrative criteria can be the basis for establishing chemical-specific limits for waste discharges where a specific pollutant can be identified as causing or contributing to the toxicity and the State has not adopted chemical-specific numeric criteria. Narrative criteria also can be the basis for establishing whole effluent toxicity controls required by EPA regulations at 40 *CFR* 122.44(d)(1)(v).

To ensure that narrative criteria for toxicants are attained, the water quality standards regulation requires States to develop implementation procedures (see 40 *CFR* 131.11(a)(2)). Such implementation procedures (Box 2-1) should address all mechanisms used by the State to ensure that narrative criteria are attained. Because implementation of chemical-specific numeric criteria is a key component of State toxics control programs, narrative criteria implementation procedures must describe or reference the State's procedures to implement such chemical-specific numeric criteria (e.g., procedures for establishing chemical-specific permits limits under the NPDES permitting program). Implementation procedures also must address State programs to control whole effluent toxicity and may address programs to implement biological criteria, where such programs have been developed by the State. Implementation procedures therefore serve as umbrella documents that describe how the State's various toxics control programs are integrated to ensure adequate protection for aquatic life and human health and attainment of the narrative toxics criterion. In essence, the procedure should apply the "independent application" principle, which provides for independent evaluations of attainment of a designated use based on chemical-specific, whole effluent toxicity, and biological criteria methods (see Chapter 1, Reference 56).

EPA encourages, and may ultimately require, State implementation procedures to provide for implementation of biological criteria. However, the regulatory basis for requiring whole effluent toxicity controls is clear. EPA regulations at 40 *CFR* 122.44(d)(1)(v) require NPDES permits to contain whole effluent toxicity limits where a permittee has been shown to cause, have the reasonable potential to cause, or contribute to an in-stream excursion of a narrative criterion. Implementation of chemical-specific controls also is required by EPA regulations at 40 *CFR* 122.44(d)(1). State implementation procedures should, at a minimum, specify or reference methods to be used in implementing chemical-specific and whole effluent toxicity-based controls, explain how these methods are integrated, and specify needed application criteria.

In addition to EPA's regulation at 40 *CFR* Part 131, EPA has regulations at 40 *CFR* 122.44 that cover the National Surface Water Toxics Control Program. These regulations intrinsically are linked to the requirements to achieve water quality standards, and specifically address the control of pollutants both with and without numeric criteria. For example, Section 122.44(d)(1)(vi) provides the permitting authority with several options for establishing effluent limits when a State does not have a chemical-specific numeric criteria for a pollutant present in an effluent at a concentration that causes or contributes to a violation of the State's narrative criteria.

2.2 GENERAL CONSIDERATIONS

2.2.1 Magnitude, Duration, and Frequency

As stated earlier, criteria are specifications of water quality designed to ensure protection of the designated use. EPA criteria are

Box 2-1. Components of an Ideal State Implementation Procedure

- Specific, scientifically defensible methods by which the State will implement its narrative toxics standard for all toxicants, including:
  - Methods for chemical-specific criteria, including methods for applying chemical-specific criteria in permits, developing or modifying chemical-specific criteria via a "translator procedure" (defined and discussed below), and calculating site-specific criteria based on local water chemistry or biology
  - Methods for developing and implementing whole effluent toxicity criteria and/or controls
  - Methods for developing and implementing biological criteria.
- Integration of these methods in the State's toxics control program (i.e., how the State will proceed when the specified methods produce conflicting or inconsistent results).
- Application criteria and information that are needed to apply numerical criteria, for example:
  - Methods the State will use to identify thosepollutants to be regulated in a specific discharge
  - An incremental cancer risk level for carcinogens
  - Methods for identifying compliance thresholds inpermits where calculated limits are below detection
  - Methods for selecting appropriate hardness, pH, and temperature variables for criteria expressed as functions
  - Methods or policies controlling the size and in-zone quality of mixing zones
  - Design flows to be used in translating chemical-specific numeric criteria for aquatic life and human health into permit limits
  - Other methods and information that will be needed to apply standards on a case-by-case basis.

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 of 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.

A typical aquatic life water quality criteria statement contains a concentration, averaging period, and return frequency, stated in the following format:

> The procedures described in the *Guidelines for Deriving National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* indicate that, except possibly where a locally important species is very sensitive, (1) aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of (2) does not exceed (3) µg/L more than once every three years on the average and if the one-hour average concentration does not exceed (4) µg/L more than once every three years on the average.

In this example generic statement, the following terms are inserted at:

(1) — either "freshwater" or "saltwater"

(2) — the name of the pollutant

(3) — the lower of the chronic-effect or residue-based concentrations as the criterion continuous concentration (CCC)

(4) — the acute effect-based criterion maximum concentration (CMC).

Defining water quality criteria with an appropriate duration and frequency of excursions helps to ensure that criteria appropriately are considered in developing wasteload allocations (WLAs), which are then translated into permit requirements. Duration and frequency may be defined in the design stream flow appropriate to the criterion. However, in these cases, the State should provide an evaluation that the selected design stream flow approximates the recommended duration and frequency.

2.2.2 Mixing Zones

It is not always necessary to meet all water quality criteria within the discharge pipe to protect the integrity of the waterbody as a whole. Sometimes it is appropriate to allow for ambient concentrations above the criteria in small areas near outfalls. These areas are called mixing zones. Since these areas of impact, if disproportionately large, could potentially adversely impact the productivity of the waterbody, and have unanticipated ecological consequences, they should be carefully evaluated and appropriately limited in size. As our understanding of pollutant impacts on ecological systems evolves, there may be cases identified where no mixing zone is appropriate.

To ensure mixing zones do not impair the integrity of the waterbody, it should be determined that the mixing zone will not cause lethality to passing organisms and, considering likely pathways of exposure, that there are no significant human health risks. One means to achieve these objectives is to limit the size of the area affected by the mixing zones.

For application of two-number aquatic life criteria, there may be up to two types of mixing zones (Figure 2-1). In the zone immediately surrounding the outfall, neither the acute nor the chronic criterion is met. The acute criterion is met at the edge of this zone. In the next mixing zone, the acute, but not the chronic, criterion is met. The chronic criterion is met at the edge of the second mixing zone.

In the general case, where a State has both acute and chronic aquatic life criteria, as well as human health criteria, independently established mixing zone specifications may apply to each of the three types of criteria. The acute mixing zone may be sized to prevent lethality to passing organisms, the chronic mixing zone sized to protect the ecology of the waterbody as a whole, and the health criteria mixing zone sized to prevent significant human risks. For any particular pollutant from any particular discharge, the magnitude, duration, frequency, and mixing zone associated with each of the three types of criteria will determine which one most limits the allowable discharge.

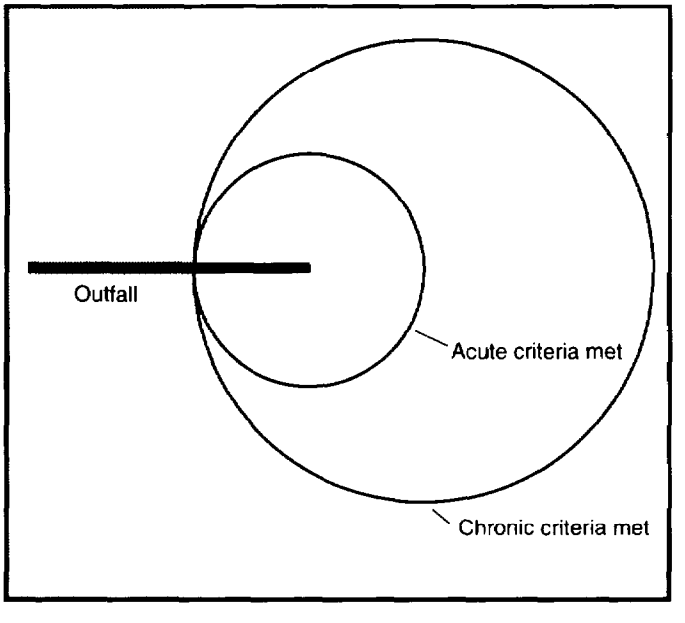

Figure 2-1. Diagram of the Two Parts of the Mixing Zone

Mixing zone allowances will increase the mass loadings of the pollutant to the waterbody, and decrease treatment requirements. They adversely impact immobile species, such as benthic communities, in the immediate vicinity of the outfall. Because of these and other factors, mixing zones must be applied carefully, so as not to impede progress toward the CWA goals of maintaining and improving water quality. EPA recommendations for allowances for mixing zones, and appropriate cautions about their use, are contained in this section.

The CWA allows mixing zones at the discretion of the State [1]. **EPA recommends that States have a definitive statement in their standards on whether or not mixing zones are allowed.** Where mixing zones provisions are part of the State standards, the State should describe the procedures for defining mixing zones.

To determine that a mixing zone is sized appropriately for aquatic life protection, water quality conditions within the mixing zone may be compared to laboratory-measured or predicted toxicity bench marks as follows:

> It is not necessary to meet chronic criteria within the mixing zone, only at the edge of the mixing zone. Conditions within the mixing zone would thus not be adequate to ensure survival, growth, and reproduction of all organisms that might otherwise attempt to reside continuously within the mixing zone.
>
> If acute criteria (CMC derived from 48- to 96-hour exposure tests) are met throughout the mixing zone, no lethality should result from temporary passage through the mixing zone. If acute criteria are exceeded no more than a few minutes in a parcel of water leaving an outfall (as assumed in deriving the Section 4.3.3 options for an outfall velocity of 3 m/sec, and a size of 50 times the discharge length scale), this likewise assures no lethality to passing organisms.
>
> If a full analysis of concentrations and hydraulic residence times within the mixing zone indicates that organisms drifting through the plume along the path of maximum exposure would not be exposed to concentrations exceeding the acute criteria when averaged over the 1-hour (or appropriate site-specific) averaging period for acute criteria, then lethality to swimming or drifting organisms ordinarily should not be expected, even for rather fast-acting toxicants. In many situations, travel time through the acute mixing zone must be less than roughly 15 minutes if a 1-hour average exposure is not to exceed the acute criterion.
>
> Where mixing zone toxicity is evaluated using the probit approach described in the water quality criteria "Bluebook" [3], or using models of toxicant accumula-

tion and action in organisms (described by Mancini [4] or Erickson et al. [5]), the phenomenon of delayed mortality should be taken into account before judging the mixing zone concentrations to be safe.

The above recommendations assume that the effluent is repulsive, such that free-swimming organisms would avoid the mixing zones. While most toxic effluents are repulsive, caution is necessary in evaluating attractive mixing zones of known effluent toxicity, and denial of such mixing zones may well be appropriate. It also is important to ensure that concentration isopleths within any plume will not extend to restrict passage of swimming organisms into tributary streams.

In all cases, the size of the mixing zone and the area within certain concentration isopleths should be evaluated for their effect on the overall biological integrity of the waterbody. If the total area affected by elevated concentrations within all mixing zones combined is small compared to the total area of a waterbody (such as a river segment), then mixing zones are likely to have little effect on the integrity of the waterbody as a whole, provided that they do not impinge on unique or critical habitats. EPA has developed a multistep procedure for evaluating the overall acceptability of mixing zones [6].

For protection of human health, the presence of mixing zones should not result in significant health risks, when evaluated using reasonable assumptions about exposure pathways. Thus, where drinking water contaminants are a concern, mixing zones should not encroach on drinking water intakes. Where fish tissue residues are a concern (either because of measured or predicted residues), mixing zones should not be projected to result in significant health risks to average consumers of fish and shellfish, after considering exposure duration of the affected aquatic organisms in the mixing zone, and the patterns of fisheries use in the area.

While fish tissue contamination tends to be a far-field problem affecting entire waterbodies rather than a narrow-scale problem confined to mixing zones, restricting or eliminating mixing zones for bioaccumulative pollutants may be appropriate under conditions such as the following:

- Mixing zones should be restricted such that they do not encroach on areas often used for fish harvesting particularly of stationary species such as shellfish.
- Mixing zones might be denied where such denial is used as a device to compensate for uncertainties in the protectiveness of the water quality criteria or uncertainties in the assimilative capacity of the waterbody.

2.3 WATER QUALITY CRITERIA FOR AQUATIC LIFE PROTECTION

2.3.1 Development Process for Criteria

The development of national numerical water quality criteria for the protection of aquatic organisms is a complex process that uses information from many areas of aquatic toxicology. (See Reference 7 for a detailed discussion of this process.) After a decision is made that a national criterion is needed for a particular material, all available information concerning toxicity to, and bioaccumulation by, aquatic organisms is collected and reviewed for acceptability. If enough acceptable data for 48- to 96-hour toxicity tests on aquatic animals are available, they are used to derive the acute criterion. If sufficient data on the ratio of acute to chronic toxicity concentrations are available, they are used to derive the chronic or long-term exposure criteria. If justified, one or both of the criteria may be related to another water quality characteristic, such as pH, temperature, or hardness. Separate criteria are developed for freshwaters and saltwaters.

The water quality standards regulation allows States to develop numerical criteria or modify EPA's recommended criteria to account for site-specific or other scientifically defensible factors. In cases where additional toxicological data are needed to modify or develop criteria, the discharger may be required to generate the data. Guidance on modifying national criteria is found in the handbook [1]. When a criterion must be developed for a chemical for which a national criterion has not been established, the regulatory authority should refer to the *Guidelines for Deriving Criteria for Aquatic Life and Human Health* (see 45 *FR* 79341, November 28, 1980, and 50 *FR* 30784, July 29, 1985).

2.3.2 Magnitude for Single Chemicals

Water quality criteria for aquatic life contain two expressions of allowable magnitude: a CMC to protect against acute (short-term) effects and a CCC to protect against chronic (long-term) effects. EPA derives acute criteria from 48- to 96-hour tests of lethality or immobilization. EPA derives chronic criteria from longer-term (often greater than 28-day) tests that measure survival, growth, reproduction, or in some cases, bioconcentration.

Most State standards include numerical criteria for a limited number of individual toxic chemicals. Therefore, evaluation and control of toxic pollutants is based on maintenance of the designated use and often relies on the narrative criterion prohibiting toxic substances in toxic amounts. The adverse effects of concern will depend on the designated use and the chemical. Bioaccumulation of chemicals in aquatic organisms, toxicity to these organisms, the potential for additivity, antagonism, synergism, and persistence of the chemicals may be important. Available information on the toxic effects of the chemical is used when standards do not include specific numerical criteria. Such information can include EPA criteria documents, published literature reports, or studies conducted by the discharger.

As mentioned in Section 2.1.2, water quality-based controls may be based directly on the State's technical determination of what concentration of a specific pollutant meets the State's narrative "free from" toxics criterion. Although EPA water quality standards regulation requires that the State's process for implementing its narrative criterion be described in the State standards, there is no requirement that this concentration be adopted as a numerical criterion in State water quality standards prior to use in developing water quality-based controls and therefore a case-by-case interpretation of the narrative criterion may be necessary.

2.3.3 Magnitude for Whole Effluent Toxicity

Criteria for toxicity in current State standards range from the narrative prohibition (e.g., no discharge of toxic chemicals in toxic amounts) to detailed requirements that specify the test species and the allowable toxicity level. At present, there are no national criteria developed under CWA Section 304(a) for whole effluent toxicity. Acute and chronic toxicity units (TUs) are a mechanism for quantifying instream toxicity using the whole effluent approach. The procedure to implement the narrative criteria using a whole effluent approach should specify the testing procedure, the duration of the tests (acute or chronic), the test species, and the frequency of testing required.

EPA's recommended magnitudes for whole effluent toxicity are as follows (again, two expressions of allowable magnitude are used): a CMC to protect against acute (short-term) effects and a CCC to protect against chronic (long-term) effects. **For acute protection, the CMC should be set at 0.3 acute toxic unit (TU_a) to the most sensitive of at least three test species.**

The selection of test species for testing the effluent is not critical provided species from ecologically diverse taxa are used (e.g., a fish, an invertebrate, and a plant). The factor of 0.3 is used to adjust the typical LC_{50} endpoint of an acute toxicity test (50 percent mortality) to an LC_1 value (virtually no mortality). Specifically, a factor of 0.3 was found to include 91 percent of observed LC_1 to LC_{50} ratios in 496 effluent toxicity tests as illustrated in Figure 2-2. This figure presents effluent toxicity data from many years of toxicity testing of both industrial and municipal effluents by the Environmental Services Division, U.S. EPA Region IV, Athens, Georgia.

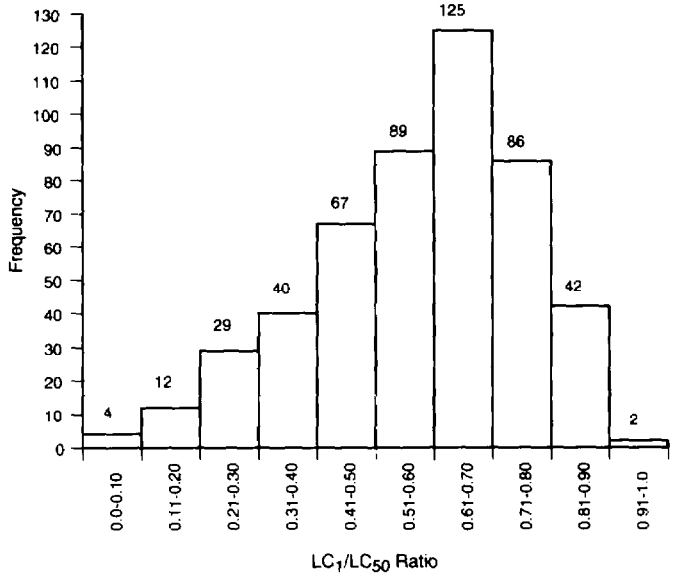

Figure 2-2. LC_1 to LC_{50} Ratios for Effluent Toxicity Tests

For chronic protection, the CCC should be set at 1.0 chronic toxic unit (TU_c) to the most sensitive of at least three test species. The selection of test organisms is as described above. A 1.0 TU_c is applied at the edge of the mixing zone to prevent any chronic toxicity in the receiving water outside the mixing zone.

2.3.4 Duration for Single Chemicals and Whole Effluent Toxicity

The quality of an ambient water typically varies in response to variations of effluent quality, stream flow, and other factors. Organisms in the receiving water are not experiencing constant, steady exposure but rather are experiencing fluctuating exposures, including periods of high concentrations, which may have adverse effects. Thus, EPA's criteria indicate a time period over which exposure is to be averaged, as well as a maximum concentration, thereby limiting the duration of exposure to elevated concentrations.

For acute criteria, EPA recommends an averaging period of 1 hour. That is, to protect against acute effects, the 1-hour average exposure should not exceed the CMC. The 1-hour acute averaging period was derived primarily from data on response time for toxicity to ammonia, a fast-acting toxicant. The 1-hour averaging period is expected to be fully protective for the fastest-acting toxicants, and even more protective for slower-acting toxicants. Scientifically justifiable alternative (site-specific) averaging periods can be derived from (1) data relating toxic response to exposure time, if coupled with considerations of delayed mortality (mortality occurring after exposure has ended), or (2) models of toxicant uptake and action, such as presented by Erickson [5] and Mancini et al. [4].

In practice, 1-day periods are the shortest periods for which WLA modelers and enforcement personnel have adequate data. Attainment of the duration criterion can be ensured by paying particular attention to short-term effluent variability and requiring measures to control variability (e.g., installation of equalization basins) when needed.

For chronic criteria, EPA recommends an averaging period of 4 days. That is, the 4-day average exposure should not exceed the CCC. Different chronic averaging periods could be derived, depending on the nature of the pollutant and the toxic endpoint of concern (e.g., the rate of uptake and accumulation, and the mode of action).

The toxicity tests used to establish the national criteria are conducted using steady exposure to toxicants usually for at least 28 days. The test concentrations do not fluctuate as much as typically occurs instream. As the period of averaging increases, so too does the period of time the exposure concentrations can be above the criterion concentration without exceeding the average. The significant consideration involved in setting duration criteria is how long the exposure concentration can be above the criterion concentration without unacceptably affecting the endpoint of the test (e.g., survival, growth, or reproduction). EPA selected the 4-day averaging period based on the shortest duration in which chronic effects are sometimes observed for certain species and toxicants, and thus should be fully protective even for the fastest-acting toxicants.

2.3.5 Frequency for Single Chemicals and Whole Effluent Toxicity

To predict or ascertain the attainment of criteria it is necessary to specify the allowable frequency for exceeding the criteria. This is because it is statistically impossible to project that criteria will never be exceeded. As ecological communities are naturally subjected to a series of stresses, the allowable frequency of pollutant stress may be set at a value that does not significantly increase the frequency or severity of all stresses combined.

EPA recommends a once in 3-year average frequency for excursions of both acute and chronic criteria. These recommendations apply to both chemical-specific and whole effluent approaches. However, the allowable frequency depends on site-specific factors. To implement alternative frequencies, site-specific factors (see Appendix D) or other data or analyses should be taken into account. In all cases, the recommended frequency applies to actual ambient concentrations, and excludes the influence of measurement imprecision.

EPA established its recommended frequency as part of its *Guidelines for Deriving Criteria,* last issued in 1985 [8]. EPA selected the 3-year return interval with the intent of providing a degree of protection roughly equivalent to a 7Q10 design flow condition, and with some consideration of rates of ecological recovery from a variety of severe stresses. Because of the nature of the ecological recovery studies available, the severity of criteria excursions could not be related rigorously to the resulting ecological impacts. Nevertheless, EPA derives its criteria intending that a single marginal criteria excursion (i.e., a slight excursion over a 1-hour period for acute or over a 4-day period for chronic) would result in little or no ecological effect and require little or no time for recovery. If the frequency of marginal criteria excursions is not high, it can be shown that the frequency of severe stresses, requiring measurable recovery periods, would be extremely small. EPA thus expects the 3-year return interval to provide a very high degree of protection.

Field studies indicate that many discharge situations are affected both by predictable and measurable discharges of toxicants and by unpredictable spills of toxic substances. In most cases, the dischargers were unaware that spills were occurring. These spills are a second source of stress for the community and decrease recovery potential. An aggressive program to minimize, contain, and treat spills should be in place at any plant where the potential for spills exists.

The concentration, duration, and frequency provisions of the criteria are implemented through the development of WLAs and water quality-based effluent limits. As discussed in Chapter 4, the duration and frequency recommendations are implemented directly if a dynamic modeling approach is used to develop WLAs and permit limits. However, if a steady-state approach is used, a design condition is needed for the calculations.

For the protection of aquatic life, the duration and frequency recommendations provided above have been used to develop recommended design flows for steady-state modeling. Chapter 4 discusses these recommended design flows.

Traditionally, most water quality-based permits for point source discharges had been tied to the 7-day, once in 10-year, low-flow conditions. The reason for this is that critical conditions for perennial point source discharges occur, in general, during the low-flow period. Currently, State laws and regulations generally state that water quality standards are applicable to the 7-day, 10-year low-flow or higher flow conditions.

It should be noted that EPA's water quality criteria for aquatic life protection are applicable at all flow conditions, low as well as high. These criteria and their specified duration and frequency, if adopted into or used to interpret State water quality standards, may be used as the basis for total maximum daily load (TMDL) after considering seasonal flow and loading scenarios. The concentration, duration, and frequency provisions of EPA's water quality criteria can be modified to account for site-specific conditions. As States have started using the new two-number water quality criteria for perennial as well as intermittent discharges such as combined sewer overflows, urban runoff, etc., their proper use in the context of the TMDL/WLA process needs to be emphasized.

2.4 WATER QUALITY CRITERIA FOR HUMAN HEALTH PROTECTION

2.4.1 Overview

There are a number of key elements of State water quality standards and implementation procedures relevant to human health protection. States must determine ambient standards for the two primary human exposure routes, fish consumption and drinking water. States must then establish whether mixing zones will apply, and, if so, determine the design conditions.

State standards or their implementation procedures often specify the risk level for carcinogens; methods for identifying compliance thresholds in permits where calculated limits are below detection; and methods for selecting appropriate hardness, pH, and temperature variables for criteria. However, if State standards do not specify these items, then the permitting authority must develop water quality-based effluent limits based upon either an interpretation of the State's water quality standards or EPA's criteria and procedures.

The purpose of the following section is to provide a review of EPA's procedures used to develop assessments of human health effects in developing water quality criteria and reference ambient concentrations. A complete human health effects discussion is included in the (draft) *Guidelines and Methodology Used in the Preparation of Health Effects Assessment Chapters of the Consent Decree Water Documents* by EPA's Environmental Criteria and Assessment Office (ECAO). The procedures contained in the ECAO document are used in the development and updating of EPA water quality criteria and may be used in developing reference ambient concentrations (RACs) for those pollutants lacking EPA human health criteria. Although the same procedures are used to develop criteria and RACs, only those values that are subjected to the regulatory process of regional, State, and public comment can be considered "criteria." RACs may be applied as site-specific interpretations of narrative standards and as a basis for permit limits under 40 *CFR* 122.44 (d)(1)(vi).

Procedures also are provided in this chapter to develop values called reference tissue concentrations (RTCs) that can be used in assessing or monitoring fish tissues for unacceptable residues.

2.4.2 Magnitude and Duration

Water quality criteria for human health contain only a single expression of allowable magnitude; a criterion concentration generally to protect against long-term (chronic) human health effects. Currently, national policy and prevailing opinion in the expert community dictate that the duration for human health criteria for carcinogens be derived assuming lifetime exposure, taken to be a 70-year time period. The duration of exposure assumed in deriving criteria for noncarcinogens is more complicated due to a wide variety of endpoints: some developmental (and thus age-specific and perhaps sex-specific), some lifetime, and some, such as organoleptic effects, not duration-related at all. Thus, appropriate durations depend on the individual noncarcinogenic pollutants and the endpoints or adverse effects being considered.

2.4.3 Human Exposure Considerations

A complete human exposure evaluation for toxic pollutants of concern for bioaccumulation would not only encompass estimates of exposures due to fish consumption, but also exposure due to background concentrations and other exposure routes, including recreational and occupational contact, dietary intake from other than fish, inhalation of air, and drinking water. However, the focus of this document is on ingestion of contaminated fish tissue, a direct human exposure route of potentially significant risk. (For the human health sections in this document the term "fish" generally is used to mean both fish and shellfish.) The consumption of contaminated fish tissue is of serious concern since the presence of even extremely low ambient concentrations of bioaccu-mulative pollutants (sublethal to aquatic life) in surface waters, can result in residue concentrations in fish tissue that can pose a human health risk. Other exposure route information should be considered and incorporated in human exposure evaluations to the extent it is available.

Levels of actual human exposures from consuming contaminated fish vary depending upon a number of case-specific consumption factors. These factors include type of fish species consumed, type of fish tissue consumed, tissue lipid content, consumption rate and pattern, and food preparation practices. In addition, depending on the spatial variability in the fishery area, the behavior of the fish species, and the point of application of the RAC or criterion, the average exposure of fish may be only a small fraction of the expected exposure at the point of application of the criterion. If an effluent attracts fish, the average exposure might be greater than the expected exposure.

With shellfish, such as oysters, snails, and mussels, whole body tissue consumption commonly occurs, whereas with fish, muscle tissue and roe are most commonly eaten. This difference in the types of tissues consumed has implications for the amount of available bioaccumulative contaminants likely to be ingested. Whole body shellfish consumption presumably means ingestion of the entire burden of bioaccumulative contaminants. However, with most fish, selective cleaning and removal of internal organs, and sometimes body fat as well, from edible tissues, may result in removal of much of the lipid material in which bioaccumulative contaminants tend to concentrate.

2.4.4 Fish Consumption Values

EPA's human health criteria have assumed a human body weight of 70 kg and the consumption of 0.0065 kg of fish and shellfish per day. Based on data collected in 1973-1974, the national per capita consumption of freshwater and estuarine fish was estimated to average 6.5 g/day. Per capita consumption of all seafood (including marine species) was estimated to average 14.3 g/day. The 95th percentile for consumption of all seafood by individuals over a period of 1 month was estimated to be 42 g/day [9]. The mean lipid content of fish tissue consumed in this study was estimated to be 3.0 percent [10].

Currently, four levels of fish consumption are provided in the EPA guidance manual, *Assessing Human Health Risk from Chemically Contaminated Fish and Shellfish.* These are:

- 6.5 g/day to represent an **estimate of average consumption of fish and shellfish from estuarine and freshwaters** by the entire U.S. population [9]. This fish consumption level is based on the average of both consumers and nonconsumers of fish.

- 20 g/day to represent an **estimate of the average consumption of fish and shellfish from marine, estuarine, and freshwaters** by the U.S. population [11]. This average fish consumption level also includes both consumers and nonconsumers of fish.

- 165 g/day to represent **consumption of fish and shellfish from marine, estuarine, and freshwaters by the 99.9th percentile of the U.S. population** consuming the most fish or seafood [12].

- 180 g/day to represent a "**reasonable worst case**" based on the assumption that some individuals would consume fish at a rate equal to the combined consumption of red meat, poultry, fish, and shellfish in the United States (EPA Risk Assessment Council assumption based on data from the U.S. Department of Agriculture Nationwide Food Consumption Survey of 1977-1978).

EPA currently is updating the national estuarine and freshwater fish and shellfish consumption default values and will provide a range of recommended national consumption values. This range will include mean values appropriate to the population at large, and values appropriate for those individuals who consume a relatively large proportion of fish in their diets (maximally exposed individuals).

Many States use the EPA's 6.5 g/day consumption value. However, some States (e.g., Wisconsin, Louisiana, Illinois, and Arizona) use the above mentioned 20 g/day value. For salt waters Delaware uses another EPA value, 37 g/day [13]. In general, EPA recommends that the consumption values used in deriving RACs from the formulas in this chapter reflect the most current relevant and/or site-specific information available.

2.4.5 Bioaccumulation Considerations for Reference Ambient Concentration Development

The ratio of the contaminant concentrations in fish tissue versus water is termed either the bioconcentration factor (BCF) or the bioaccumulation factor (BAF). Bioconcentration is defined as involving contaminant uptake from water only (not from food). Bioaccumulation is defined as involving contaminant uptake from both water and food. Under laboratory conditions, measurements of tissue/water partitioning generally are considered to involve uptake from water only. On the other hand, both process are likely to apply in the field since the entire food chain is exposed.

Table 2-1 shows the ratio of the BAF to the BCF as a function of the trophic level of the aquatic organism, and the log P (log octanol-water partition coefficient) of the chemical [14]. The BAF/BCF ratio ranges from 1 to 100, with the highest ratios applying to organisms in higher trophic levels, and to chemicals with log P close to 6.5. For chemicals with log P values greater than about 7, there is some uncertainty regarding the degree of bioaccumulation, but generally, trophic level effects appear to decrease due to slow transport kinetics of these chemicals in fish, the growth rate of the fish, and the chemical's relatively low bioavailability.

Care must be taken in assigning the trophic level since certain fish species may inhabit one source area of contaminated food for only a portion of their life. Under such conditions of migration, fish would only receive a small portion of the chemical and never come into equilibrium. In addition, trophic level for a given fish species will vary with life stage and structure of the food chain.

In this document, bioaccumulation considerations are integrated into the RAC equations in Sections 2.4.7 and 2.4.8 by using food chain multipliers (FMs) with the BCF. The bioaccumulation and bioconcentration factors for a chemical are related as follows:

$$BAF = FM \times BCF$$

By incorporating the FM and BCF terms into the RAC equations, bioaccumulation is addressed.

In this process, bioaccumulation considerations are included by incorporating the FM term with the BCF in calculating the RTCs and RACs. In Table 2-1, FM values derived from the work of Thomann [14, 15] are listed according to log P value and trophic level of the organism. **Trophic level 4 organisms are typically the most desirable species for sport fishing and therefore, FMs for trophic level 4 generally should be used in the equations for calculating RTCs and RACs.** In those very rare situations where only lower trophic level organisms are found, e.g., possibly oyster beds, an FM for a lower trophic level may be used in calculating the RTCs and RACs.

Measured BAFs (especially for those chemicals with log P values above 6.5) reported in the literature should be used when available. To use experimentally measured BAFs in calculating the RAC or RTC, the (FM x BCF) term, is replaced by the BAF in the equations in Sections 2.4.7 and 2.4.8. Relatively few BAFs have been measured accurately and reported, and their application to sites other than the specific ecosystem where they were developed is problematic and subject to uncertainty. The option also is available to develop BAFs experimentally, but this will be extremely resource intensive if done on a site-specific basis with all the necessary experimental and quality controls.

Table 2-1. Estimated Food Chain Multipliers

	Trophic Levels		
Log P	2	3	4
3.5	1.0	1.0	1.0
3.6	1.0	1.0	1.0
3.7	1.0	1.0	1.0
3.8	1.0	1.0	1.0
3.9	1.0	1.0	1.0
4.0	1.1	1.0	1.0
4.1	1.1	1.1	1.1
4.2	1.1	1.1	1.1
4.3	1.1	1.1	1.1
4.4	1.2	1.1	1.1
4.5	1.2	1.2	1.2
4.6	1.2	1.3	1.3
4.7	1.3	1.4	1.4
4.8	1.4	1.5	1.6
4.9	1.5	1.8	2.0
5.0	1.6	2.1	2.6
5.1	1.7	2.5	3.2
5.2	1.9	3.0	4.3
5.3	2.2	3.7	5.8
5.4	2.4	4.6	8.0
5.5	2.8	5.9	11
5.6	3.3	7.5	16
5.7	3.9	9.8	23
5.8	4.6	13	33
5.9	5.6	17	47
6.0	6.8	21	67
6.1	8.2	25	75
6.2	10	29	84
6.3	13	34	92
6.4	15	39	98
6.5	19	45	100
≥6.5	19.2*	45*	100*

* These recommended FMs are conservative estimates; FMs for log P values greater than 6.5 may range from the values given to as low as 0.1 for contaminants with very low bioavailability.

2.4.6 Updating Human Health Criteria and Generating RACs Using IRIS

EPA recommends using the most current risk information when updating criteria and generating RACs. The Integrated Risk Information System (IRIS) is an electronic online data base of the U.S. EPA that provides chemical-specific risk information on the relationship between chemical exposure and estimated human health effects [16]. Risk assessment information contained in the IRIS, except as specifically noted, has been reviewed and agreed upon by an interdisciplinary group of scientists representing various program offices within the Agency and represent an Agencywide consensus. Risk assessment information and values are updated monthly and are approved for Agencywide use.

The IRIS is intended to make risk assessment information readily available to those individuals who must perform risk assessments and also to increase consistency among risk assessment/risk management decisions. The IRIS is available to Federal and some State and local environmental agencies through the EPA's electronic MAIL system and also is available to the public through the Public Health Network and TOXNET. Since IRIS is designed to be a publicly available data base, interested parties may submit studies or documents for consideration by the appropriate interdisciplinary review group for chemicals currently on the IRIS or scheduled for review. Information regarding the submission of studies of chemicals may be obtained from the IRIS Information Submission Desk. In addition to chemical-specific summaries of hazard and dose-response assessments, the IRIS contains a series of sections identified by service codes that serve as a user's guide as well as provide background documentation on methodology. Additional information is available from IRIS Users Support: 513/FTS 684-7254.

The IRIS contains two types of quantitative risks values: reference dose (RfD) and the carcinogenic potency estimate or slope factor. The RfD (formerly known as the acceptable daily intake or ADI) is the human health hazard assessment for noncarcinogenic (target organ) effects. The carcinogenic potency estimate (formerly known as $q1^*$) represents the upper bound cancer causing potential resulting from lifetime exposure to a substance. The RfD or the oral carcinogenic potency estimate are used in the derivation of an RAC. Appendix H contains the supporting information for derivation of RfDs.

EPA periodically updates risk assessment information including RfDs, cancer potency estimates, and related information on contaminant effects, and reports the current information on IRIS. Since the IRIS contains the Agency's most recent quantitative risk assessment values, current IRIS values should be used in developing new RACs. **This means that the 1980 human health criteria should be updated with the latest IRIS values.** The procedure for deriving an updated human health water quality criterion would require inserting the current RfD or carcinogenic potency estimate on the IRIS into the appropriate equation in Section 2.4.7 or 2.4.8.

Figure 2-3 shows the procedure for determining an updated criterion or RAC using IRIS data. **If a chemical has both carcinogenic and noncarcinogenic effects, i.e., both a cancer potency estimate and RfD, the carcinogen RAC formula in Section 2.4.8 should be used as it will result in the more stringent RAC of the two.**

2.4.7 Calculating RACs for Noncarcinogens

The RfD is an estimate of the daily exposure to the human population that is likely to be without appreciable risk of causing

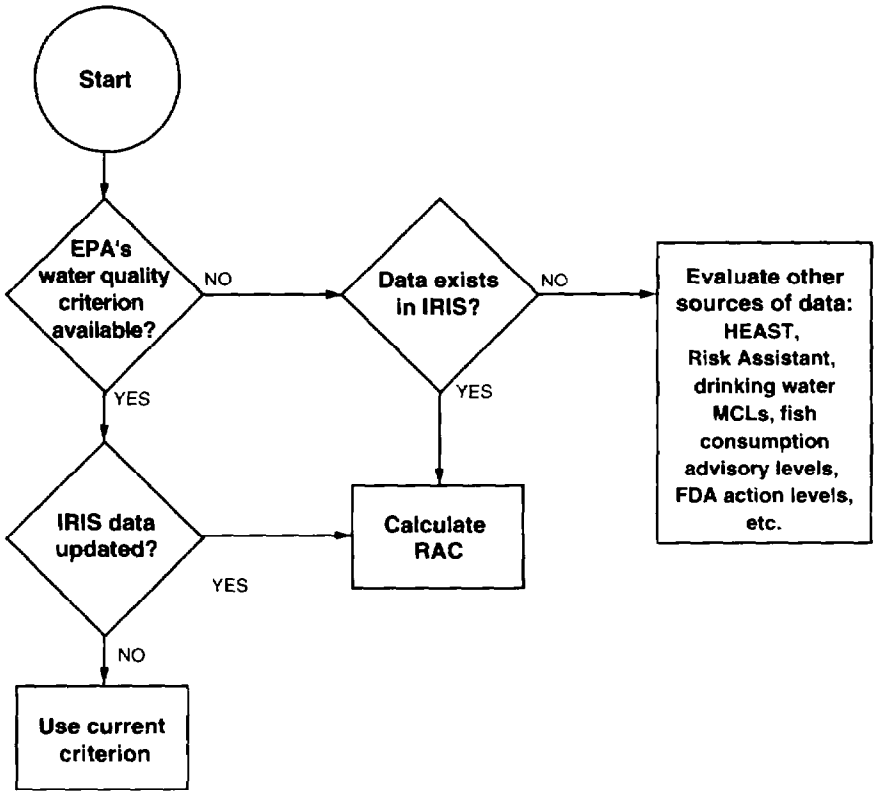

Figure 2-3. Procedure for Revising an EPA Human Health Criterion or Developing a Reference Ambient Concentration

deleterious effects during a lifetime. The RfD is expressed in units of mg toxicant per kg human body weight per day.

RfDs are derived from the "no observed adverse effect level" (NOAEL) or the "lowest observed adverse effect level" (LOAEL) identified from chronic or subchronic human epidemiology studies or animal exposure (mammal LD_{50}) studies. [Note: LOAEL and NOAEL refer to animal and human toxicology and are there fore distinct from the aquatic toxicity terms "no observed effect concentration" (NOEC) and the "lowest observed effect concentration" (LOEC)]. Uncertainty factors are then applied to the NOAEL or LOAEL to account for uncertainties in the data associated with variability among individuals, extrapolation from nonhuman test species to humans, data on other than long-term exposures, and the use of an LOAEL [17]. An additional uncertainty may be applied to account for significant weakness or gaps in the data base.

The RfD is a threshold below which effects are unlikely to occur. While exposures above the RfD increase the probability of adverse effects, they do not produce a certainty of adverse effects. Similarly, while exposure at or below the RfD reduces the probability, it does not guarantee the absence of effects in all persons. The RfDs contained in the IRIS are values that represent EPA's consensus (and have uncertainty spanning perhaps an order of magnitude).

For noncarcinogenic effects, an updated criterion or an RAC can be derived using the following equation:

$$\text{C or RAC (mg/l)} = \frac{(\text{RfD} \times \text{WT}) - (\text{DT} + \text{IN}) \times \text{WT}}{\text{WI} + [\text{FC} \times \text{L} \times \text{FM} \times \text{BCF}]}$$

where

- C = updated water quality criterion (mg/l)
- RAC = reference ambient concentration (mg/l)
- RfD = reference dose (mg toxicant/kg human body weight/day)
- WT = weight of an average human adult (70 kg)
- DT = dietary exposure (other than fish) (mg toxicant/kg body human weight/day)
- IN = inhalation exposure (mg toxicant/kg body human weight/day)
- WI = average human adult water intake (2 liters/day)
- FC = daily fish consumption (kg fish/day)
- L = ratio of lipid fraction of fish tissue consumed to 3 percent
- FM = food chain multiplier (from Table 3-1)
- BCF = bioconcentration factor (mg toxicant/kg fish divided by mg toxicant/l water) for fish with 3 percent lipid.

If the receiving waterbody is not used as a drinking water source, the factor WI can be deleted. Where dietary and/or inhalation exposure values are unknown, these factors may be deleted from the above calculation. For identified noncarcinogenic chemicals without known RfDs, extrapolation procedures can be used to estimate the RfD (see Appendix H).

2.4.8 Calculating RACs for Carcinogens

Any human health criterion for a carcinogen is based on at least three interrelated considerations: potency, exposure, and risk characterization. States may make their own judgments on each of these factors within reasonable scientific bounds, but documentation to support their judgments must be clear and in the public record.

Maximum protection of human health from the potential effects of exposure to carcinogens via contaminated fish would require an RAC of zero. The zero level is based upon the assumption of nonthreshold effects (i.e., no safe level exists below which any increase in exposure does not result in an increase in the risk of cancer) for carcinogens. However, because safety does not require the absence of all risk, a numerical estimate of risk (in µg/l) that corresponds to a given level of risk for a population of a specified size is selected instead. A cancer risk level is defined as the number of new cancers that may result in a population of specified size due to an increase in exposure (e.g., 10^{-6} risk level = 1 additional cancer in a population of 1,000,000). Cancer risk is calculated by multiplying the experimentally derived cancer potency estimate by the concentration of the chemical in the fish and the average daily human consumption of contaminated fish. The risk for a specified population (e.g., 1,000,000 people or 10^{-6}) is then calculated by dividing the risk level by the specific cancer risk. EPA's ambient water quality criteria documents provide risk levels ranging from 10^{-5} to 10^{-7} as examples.

When the cancer potency estimate, or slope factor (formerly known as the q1*), is derived using animal studies, high-dose exposures are extrapolated to low-dose concentrations and adjusted to a lifetime exposure period through the use of a linearized multistage model. The model calculates the upper 95 percent confidence limit of the slope of a straight line that the model postulates to occur at low doses. When based on human (epidemiological) data, the slope factor is based on the observed increase in cancer risk, and is not extrapolated. For deriving RACs for carcinogens, the oral cancer potency estimates or slope factors from the IRIS are used.

It is important to note that cancer potency factors may overestimate actual risk. Such potency estimates are subject to great uncertainty due to two primary factors: (1) adequacy of the cancer data base (i.e., human versus animal data) and (2) limited information regarding the mechanism of cancer causation. The actual risk may be much lower, perhaps as low as zero, particularly for those chemicals for which human carcinogenicity information is lacking. Risk levels of 10^{-5}, 10^{-6}, and 10^{-7} are often used by States as minimal risk levels in interpreting their standards. EPA considers risks to be additive, i.e., the risk from individual chemicals is not necessarily the overall risk from exposure to water. For example, an individual risk level of 10^{-6} may yield a higher overall risk level if multiple carcinogenic chemicals are present.

For carcinogenic effects, the RAC can be determined by using the following equation:

$$\text{C or RAC (mg/l)} = \frac{(\text{RL} \times \text{WT})}{\text{q1}^* [\text{WI} + \text{FC} \times \text{L} \times (\text{FM} \times \text{BCF})]}$$

where

C = updated water quality criterion (mg/l)
RAC = reference ambient concentration (mg/l)
RL = risk level (10^{-X})
WT = weight of an average human adult (70 kg)
q1* = carcinogenic potency factor (kg day/mg)
WI = average human adult water intake (2 liters/day)
FC = daily fish consumption (kg fish/day)
L = ratio of lipid fraction of fish tissue consumed to 3 percent
FM = food chain multiplier (from Table 3-2)
BCF = bioconcentration factor (mg toxicant/kg fish divided by mg toxicant/l water) for fish with 3 percent lipid.

If the receiving waterbody is not used as a drinking water source, the factor WI can be deleted. For identified carcinogenic chemicals without known cancer potency estimate values, extrapolation procedures can be used to estimate the cancer potency.

2.4.9 Deriving Quantitative Risk Assessments in the Absence of IRIS Values

The RfDs or cancer potency estimates comprise the existing dose factors for developing RACs. When IRIS data are unavailable, quantitative risk level information may be developed according to a State's own procedures. Some States have established their own procedures whereby dose factors can be developed based upon extrapolation of acute and/or chronic animal data to concentrations of exposure protective of fish consumption by humans. Where no procedure exists, factors may be based upon extrapolation from mammalian or other data using IRIS documentation or information available from other EPA risk data bases. Also, where no other information or procedure exists, drinking water maximum contaminant levels (MCLs) or Food and Drug Administration (FDA) action levels may be used as guidance in developing numerical estimates.

2.4.10 Deriving Reference Tissue Concentrations for Monitoring Fish Tissue

Where fish tissue evaluations have been used for assessing human health risks, or, perhaps, used for additional routine monitoring where a chemical is below analytical detection limits, the following formulas may be used to calculate an RTC. Readers also should consult EPA's *Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish* [17].

The basic equations for deriving RTC (in mg/kg) use the same parameters as in equations 2.1 and 2.2, where BCF is normalized at 3.0 percent lipid:

For noncarcinogens:

$$RTC\ (mg/kg) = \frac{(RFD \times WT) - (DT + IN) \times WT}{[WI/(BCF \times FM \times L)] + FC}$$

For carcinogens:

$$RTC\ (mg/kg) = \frac{RL \times WT}{q1^* [WI/(BCF \times FM \times L) + FC]}$$

The above equations should be corrected for site-specific lipid content and bioaccumulation factors where data are available.

Again, some States have established their own procedures whereby RTCs can be developed based upon extrapolation of acute and/or chronic animal data to safe concentrations protective of fish consumption by humans. Where additional risk information is needed, an RTC could be based upon other information such as drinking water MCLs or FDA action levels.

2.5 BIOLOGICAL CRITERIA

As discussed in Chapter 1, to fully protect aquatic habitats and provide more comprehensive assessments of aquatic life use attainment/nonattainment, States are to fully integrate chemical-specific techniques, toxicity testing, biological surveys, and biocriteria into their water quality programs. In particular, the Agency's policy is that States should develop and implement biological criteria in their water quality standards (see Chapter 1, Reference 55).

2.5.1 Regulatory Bases for Biocriteria

The primary statutory basis for EPA's policy that States should develop biocriteria is found in Sections 101(a) and 303(c)(2)(B) of the Water Quality Act of 1987. Section 101(a) of the CWA gives the general authority for biological criteria. It establishes as the objective of the Act the restoration and maintenance of the chemical, physical, and biological integrity of the Nation's waters. To meet this objective, water quality criteria should address biological integrity. Section 101(a) includes the interim water quality goal for the protection and propagation of fish, shellfish, and wildlife.

Section 304 of the Act provides the legal basis for the development of informational criteria, including biological criteria. Specific directives for the development of regulatory biocriteria can be found in Section 303, which requires EPA to develop criteria based on biological assessment methods when numerical criteria are not established.

Once biocriteria formally are adopted into State standards, biocriteria and aquatic life use designations serve as direct, legal endpoints for determining a quality life use attainment/nonattainment. As stated in Section 131.11(b)(2) of the Water Quality Standards Regulation (40 *CFR* Part 131), biocriteria should be used as a supplement to existing chemical-specific criteria and as criteria where such chemical-specific criteria have not been established. States are encouraged to implement and integrate all three approaches (biosurvey, chemical-specific, and toxicity testing methods) into their water quality programs, applying them in combination or independently (providing the most protective of the three methods is used) as site-specific conditions and assessment objectives dictate.

Section 304(a) directs EPA to develop and publish water quality criteria and information on methods for measuring water quality and establishing water quality criteria for toxic pollutants on bases other than pollutant-by-pollutant, including biological monitoring and assessment methods that assess:

- The effects of pollutants on aquatic community components (". . . plankton, fish, shellfish, wildlife, plant life . . .")

and community attributes (". . . biological community diversity, productivity, and stability . . ."); in any body of water.

- Factors necessary " . . . to restore and maintain the chemical, physical, and biological integrity of all navigable waters . . ." for " . . . the protection of shellfish, fish, and wildlife for classes and categories of receiving waters . . . ."

2.5.2 Development and Implementation of Biocriteria

Biocriteria are numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting unimpaired waters of a designated aquatic life use. The biological communities in these waters represent the best attainable conditions. The reference site conditions then become the basis for developing biocriteria for major surface water types (streams, rivers, lakes, wetlands, estuaries, or marine waters).

Biological criteria support designated aquatic life use classifications for application in State standards. Each State develops its own designated use classification system based on the generic uses cited in the Act (e.g., protection and propagation of fish, shellfish, and wildlife). Designated uses are intentionally general. However, States may develop subcategories within use designations to refine and clarify the use class. Clarification of the use class is particularly helpful when a variety of surface waters with distinct characteristics fit within the same use class, or do not fit well into any category.

For example, subcategories of aquatic life uses may be on the basis of attainable habitat (e.g., cold versus warmwater communities dominates by bass versus catfish). Special uses also may be designated to protect particularly unique, sensitive, or valuable aquatic species, communities, or habitats.

Resident biota integrate multiple impacts over time and can detect impairment from known and unknown causes. Biocriteria can be used to verify improvement in water quality in response to regulatory efforts and detect continuing degradation of waters. They provide a framework for developing improved best management practices for nonpoint source impacts. Numeric criteria can provide effective monitoring criteria for inclusion in permits.

The assessment of the biological integrity should include measures of the structure and function of an aquatic community of species within a specified habitat. Expert knowledge of the system is required for the selection of appropriate biological components and measurement indices. The development and implementation of biological criteria requires:

- Selecting unimpaired (minimal impact) surface waters to use as the reference condition for each designated use
- Measuring the structure and function of aquatic communities in reference surface waters to establish biological criteria
- Establishing a protocol to compare the biological criteria to biota in impacted waters to determine whether impairment has occurred.

These elements serve as an interactive network that is particularly important during early development of biological criteria where rapid accumulation of information is effective for refining both designated uses and developing biological criteria values.

2.6 SEDIMENT CRITERIA

2.6.1 Current Developments in Sediment Criteria

While ambient water quality criteria are playing an important role in assuring a healthy aquatic environment, they alone have not been sufficient to ensure appropriate levels of environmental protection. Sediment contamination, which can involve deposition of toxicants over long periods of time, is responsible for water quality impacts in some areas.

EPA has authority to pursue the development of sediment criteria in streams, lakes, and other waters of the United States under CWA Sections 104 and 304(a)(1) and (2) as follows:

- Section 104(n)(1) authorizes the Administrator to establish national programs that study the effects of pollution, including sedimentation, in estuaries on aquatic life.
- Section 304(a)(1) directs the Administrator to develop and publish criteria for water quality, including information on the factors affecting rates of organic and inorganic sedimentation for varying types of receiving waters.
- Section 304(a)(2) directs the Administrator to develop and publish information on, among other things, "the factors necessary for the protection and propagation of shellfish, fish, and wildlife for classes and categories of receiving waters..."

To the extent that sediment criteria could be developed that address the concerns of the Section 404(b)(1) guidelines for discharges of dredged or fill material under the CWA or the Marine Protection Research, and Sanctuaries Act, they also could be incorporated into those regulations.

2.6.2 Approach to Sediment Criteria Development

Over the past several years, sediment criteria development activities have centered on evaluating and developing the equilibrium partitioning approach for generating sediment criteria. The equilibrium partitioning approach focuses on predicting the chemical interaction between sediments and contaminants. Developing an understanding of the principal factors that influence the sediment/contaminant interactions will allow for predictions to be made as to what concentration of a contaminant benthic and other organisms may be exposed to. Chronic water quality criteria, or possibly other toxicological endpoints can then be used to predict potential biological effects. In addition to the development of sediment criteria, EPA also is working to develop a standardized sediment toxicity test that could be used with or independently of sediment criteria and could be used to assess chronic effects in freshwater and marine water.

Equilibrium partitioning (EqP) sediment quality criteria (SQC) are the EPA's best recommendation of the concentration of a substance in sediment that will not unacceptably affect benthic organisms or their uses.

Methodologies for deriving effects based SQC vary for different classes of compounds. For non-ionic organic chemicals the methodology requires normalization to organic carbon. A methodology for deriving effects based sediment criteria for metal contaminants is under development and is expected to require normalization to acid volatile sulfide. EqP SQC values can be derived for varying degrees of uncertainty and levels of protection thus permitting use for ecosystem protection and remedial programs.

2.6.3 Application of Sediment Criteria

SQC would provide a basis for making more informed decisions on the environmental impacts of contaminated sediments. Existing sediment assessment methodologies are limited in their ability to identify chemicals of concern, responsible parties, degree of contamination, and zones of impact. EPA believes that a comprehensive approach using SQC and biological test methods is preferred in order to make the most informed decisions.

Sediment criteria will be particularly valuable in site monitoring applications where sediment contaminant concentrations are gradually approaching a criteria over time. Sediment criteria also are valuable as a preventative tool to ensure that point and nonpoint sources of contamination are controlled to ensure uncontaminated sediments remain uncontaminated. Also, comparison of field measurements to sediment criteria will be a reliable method for providing early warning of a potential problem. An early warning would provide an opportunity to take corrective action before adverse impacts occur. For the reasons mentioned above it has been identified that SQC are essential to resolving key contaminated sediment and source control issues in the Great Lakes.

Specific Applications

Specific applications of sediment criteria are under development. The primary use of EqP-based sediment criteria will be to assess risks associated with contaminants in sediments. The various offices and programs concerned with contaminated sediment have different regulatory mandates and thus, have different needs and areas for potential application of sediment criteria. Because each regulatory need is different, EqP-based sediment quality criteria designed specifically to meet the needs of one office or program may have to be implemented in different ways to meet the needs of another office or program.

One mode of application of EqP-based numerical SQC would be in a tiered approach. In such an application, when contaminants in sediments exceed the SQC, the sediments would be considered as causing unacceptable impacts. Further testing may or may not be required depending on site-specific conditions and the degree in which a criteria has been violated. (No additional testing would be required in locations where contamination significantly exceeds a criterion. Where sediment contaminant levels are close to a criteria, additional testing may be necessary.) Contaminants in a sediment at concentrations less than the sediment criteria would not be of concern. However, in some cases the sediment could not be considered safe because they may contain other contaminants above safe levels for which no sediment criteria exist. In addition, the synergistic, antagonistic, or additive effects of several contaminants in the sediments may be of concern.

Additional testing in other tiers of an evaluation approach, such as bioassays, could be required to determine if the sediment is safe. It is likely that such testing would incorporate site-specific considerations. Examples of specific applications of sediment criteria after they are developed are as follows:

- Establish permit limits to ensure that uncontaminated sediments remain uncontaminated or sediments already contaminated have an opportunity to cleanse themselves. This would occur only after criteria and the means to tie point sources to sediment deposition are developed.
- Establish target levels for nonpoint source causes of sediment contamination.
- For remediation activities, SQC would be valuable in identifying:
  - Remediation need
  - Spatial extent of remediation area
  - Benefits derived from remediation activities
  - Responsible parties
  - Impacts of depositing contaminated sediments in water environments
  - Success of remediation activities.
- In tiered testing sediment evaluation processes, sediment criteria and biological testing procedures work very well together.

2.6.4 Sediment Criteria Status

Science Advisory Board Review

The Science Advisory Board has completed its review and issued a favorable report on the EqP for assessing sediment quality. The Subcommittee found the EqP "to have major strengths in its foundation in chemical theory, its ease of calculation, and its ability to make use of existing data... The conceptual basis of the approach is supported by the Subcommittee; however, its application at this time is limited."

The Science Advisory Board also identified the need for "a better understanding of the uncertainty around the assumptions inherent in the approach, including assumptions of equilibrium, bioavailability, and kinetics, all critical to the application of the EqP." An uncertainty analysis and a guidance document to assist in the regulatory application of developed criteria are under development and expected to be completed in 1991.

Sediment Criteria Documents and Application Guidance

EPA efforts at producing sediment criteria documents are being directed first toward phenanthrene, fluoranthene, DDT, dieldrin, acenaphthene and endrin. Efforts also are being directed to produce a guidance document, *Application of Sediment Quality Criteria for the Protection of Aquatic Life*, scheduled for release in 1991.

Methodology for Developing Sediment Criteria for Metal Contaminants

EPA is proceeding with a methodology for developing sediment criteria for metal contaminants, with key work focused on identifying and understanding the role of acid volatile sulfides (AVS) in controlling the bioavailability of metal contaminants. A variety of field and laboratory verification studies are underway to add additional support to the methodology. Standard AVS sampling and analytical procedures are under development [18]. Presentation of the metals methodology to the Science Advisory Board for review is scheduled for 1991.

CHAPTER 2
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3. EFFLUENT CHARACTERIZATION

3.1 INTRODUCTION

Once the applicable designated uses and water quality criteria for a waterbody are determined, the effluent must be characterized and the permitting authority must determine the need for permit limits to control the discharge. The purpose of effluent characterization is to determine whether the discharge causes, has the reasonable potential to cause, or contributes to an excursion of numeric or narrative water quality criteria. **Once the permitting authority determines that a discharge causes, has the reasonable potential to cause, or contributes to the excursion of water quality criteria, the permitting authority must develop permit limits that will control the discharge.** At a minimum, the permitting authority must make this determination at each permit reissuance. The effluent characterization procedures described in the following sections apply only to the water quality-based approach, not to end-of-the-pipe technology-based controls.

Although many waterbodies receive discharges from only single point sources, permitting authorities will also occasionally encounter receiving waters where several dischargers are in close proximity. In such situations, the permitting authority may find that each discharger alone does not cause, have the reasonable potential to cause, or contribute to an excursion above water quality criteria. Yet, the dischargers may collectively cause, have the reasonable potential to cause, or contribute to an excursion. **Under these circumstances, limits must be developed for each discharger to protect against collective excursions of applicable water quality standards consistent with the Environmental Protection Agency's (EPA) existing regulations in 40 *CFR* 122.44(d)(1)(ii) for controlling multiple discharges.** The terms "cause," "reasonable potential to cause," and "contribute to" are the terms used in the National Pollutant Discharge Elimination System (NPDES) regulations for conditions under which water quality-based limits are required. Permitting authorities are required to consider each of these concepts when performing effluent characterizations.

This chapter is divided into two parts: Section 3.2, Determining the Need for Permit Limits Without Effluent Data, and Section 3.3, Determining the Need for Permit Limits With Effluent Data. Section 3.3 includes effluent characterization for whole effluent toxicity and for specific chemicals (including those for human health protection) and is based on the cumulative experience gained by EPA, States, publicly owned treatment works (POTWs), and industry when implementing the water quality-based approach to toxics control. The effluent bioconcentration evaluation procedures described in the section on human health are currently draft and are subject to further validation before being used. Until the procedures are fully developed, reviewed, and finalized, permitting authorities should not use them to characterize effluents.

3.1.1 NPDES Regulation Requirements

Effluent characterization is an essential step in determining the need for an NPDES permit limit. NPDES regulations under 40 *CFR* 122.44(d)(1) specify the minimum requirements and general types of analyses necessary for establishing permit limits. Each of these regulations is described below.

40 *CFR* 122.44(d)(1)(ii)

> When determining whether a discharge causes, has the reasonable potential to cause, or contributes to an in-stream excursion above a narrative or numeric criteria within a State water quality standard, the permitting authority shall use procedures which account for existing controls on point and nonpoint sources of pollution, the variability of the pollutant or pollutant parameter in the effluent, the sensitivity of the species to toxicity testing (when evaluating whole effluent toxicity), and where appropriate, the dilution of the effluent in the receiving water.

This regulation requires at a minimum the consideration of each of these elements in determining the need for a limit.

40 *CFR* 122.44(d)(1)(iii)

> When the permitting authority determines, using the procedures in paragraph (d)(1)(ii) of this section, that a discharge causes, has the reasonable potential to cause, or contributes to an in-stream excursion above the allowable ambient concentration of a State numeric criteria within a State water quality standard for an individual pollutant, the permit must contain effluent limits for that pollutant.

Under this regulation, permitting authorities need to investigate for the existence of pollutants in effluents if there is a numeric water quality criterion for that pollutant and to implement limits for those pollutants where necessary.

40 *CFR* 122.44(d)(1)(iv)

> When the permitting authority determines, using the procedures in paragraph (d)(1)(ii) of this section, that a discharge causes, has the reasonable potential to cause, or contributes to an in-stream excursion above the numeric criterion for whole effluent toxicity, the permit must contain effluent limits for whole effluent toxicity.

Under this regulation, permitting authorities need to investigate for the existence of whole effluent toxicity in effluents if there is a numeric water quality criterion for that parameter and to implement whole effluent toxicity limits where necessary.

40 *CFR* 122.44(d)(1)(v)

> Except as provided in this subparagraph, when the permitting authority determines, using the procedures in paragraph (d)(1)(ii) of this section, toxicity testing data, or other information, that a discharge causes, has the reasonable potential to cause, or contributes to an in-stream excursion above a narrative criterion within an applicable State water quality standard, the permit must contain effluent limits for whole effluent toxicity. Limits on whole effluent toxicity are not necessary where the permitting authority demonstrates in the fact sheet or statement of basis of the NPDES permit, using the procedures in paragraph (d)(1)(ii) of this section, that chemical-specific limits for the effluent are sufficient to attain and maintain applicable numeric and narrative State water quality standards.

Under this regulation, permitting authorities need to investigate for the existence of whole effluent toxicity in effluents. If the permitting authority can demonstrate that control of specific chemicals is sufficient to control toxicity to the point of achieving compliance with the water quality criteria, then chemical-specific permit limits alone will be sufficient to comply with the regulation.

40 *CFR* 122.44(d)(1)(vi)

> Where a State has not established a water quality criterion for a specific chemical pollutant that is present in an effluent at a concentration that causes, has the reasonable potential to cause, or contributes to an excursion above a narrative criterion within an applicable State water quality standard, the permitting authority must establish effluent limits using one or more of the following [three] options: ....

Under this regulation, permitting authorities need to investigate for the existence of specific chemicals in effluents for which the State has not adopted numeric criteria, but which may be contributing to aquatic toxicity or impairment of human health. Narrative criteria apply when numeric criteria do not protect all the designated or existing uses. For example, the narrative criteria need to be used to protect human health if a State has only adopted a numeric criteria for protecting aquatic life. Conversely, the narrative criteria need to be used to protect aquatic life if a State has only adopted a numeric criteria for protecting human health. Once the permitting authority determines that one or more specific chemicals in an effluent must be controlled, the authorities can use EPA's national criteria, develop their own criteria, or control the pollutant through use of an indicator pollutant, as provided in subparagraph (d)(1)(vi). In any case, the permitting authority will need to characterize the effluent in a manner consistent with the selected approach for controlling the pollutant.

3.1.2 Background for Toxic Effects Assessments on Aquatic Life and Human Health

Aquatic toxicity effects can be characterized by conducting a general assessment of the effluent, or by measuring effluent toxicity or concentrations of individual chemicals and comparing these measurements to the expected exposure concentrations in the receiving water. The "receiving water concentration" (RWC) is the measured or projected exposure concentration of a toxicant or the parameter toxicity (when dealing with the whole effluent toxicity) in the receiving water after mixing. The RWC is calculated at the edge of a mixing zone if such a zone is allowed by a State's water quality standards.

As with aquatic life protection, there are two possible approaches to characterizing effluents for human health effects: chemical-by-chemical and whole effluent. However, only the chemical-by-chemical approach currently is practical for assessing and controlling human health impacts. Appendix G discusses developing procedures for assessing human health impacts from whole effluents.

A fundamental principle in the development of water quality-based controls is that the RWC must be less than the criteria that comprise or characterize the water quality standards. With individual toxicants (or the parameter toxicity), the potential for toxicity in the receiving water is minimized where the RWC is less than the criterion continuous concentration (CCC), the criterion maximum concentration (CMC), and the reference ambient concentration (RAC). Toxicity becomes maximized where the RWC exceeds these criteria. **Therefore, to prevent impacts to aquatic life or human health, the RWC of the parameter effluent toxicity or an individual toxicant (based on allowable dilution for the criterion) must be less than the most limiting of the applicable criterion, as indicated below.** (The RAC as used throughout this chapter incorporates EPA human health criteria and State standards as well.)

RWC < CCC (chronic aquatic life)
RWC < CMC (acute aquatic life)
RWC < RAC (human health)

The water quality analyst will use the same basic components in the above-described relationship (i.e., critical receiving water flows, ambient criteria values, measures of effluent quality) for both effluent characterization and wasteload allocation (WLA) development, albeit from different perspectives. In the case of effluent characterization, the objective is to project receiving water concentrations based upon existing effluent quality to determine whether or not an excursion above ambient criteria occurs, or has the reasonable potential to occur. In developing WLAs, on the other hand, the objective is to fix the RWC at the desired criteria level and determine an allowable effluent loading that will not cause excursions above the criteria.

Recommendations for projecting the RWC are described within this chapter. Chapter 4, Exposure Assessment and Wasteload Allocation, provides recommendations for determining allowable effluent loadings to achieve established ambient criteria and for calculating WLAs for establishing permit limits. The procedures described within Chapter 4 can also be used to calculate the dilution for analyses within Chapter 3. Chapter 5, Permit Requirements, describes the actual calculation of permit limits after effluent characterization and loadings, as well as WLAs, are complete.

3.1.3 General Considerations in Effluent Characterization

There are two possible ways to characterize an effluent to determine the need for effluent limits for the protection of aquatic life and human health. First, an assessment may be made without generating effluent data; second, an assessment may be conducted after effluent data have been generated. Regulatory authorities must determine whether a discharge causes, has the "reasonable potential" to cause, or contributes to an excursion above an applicable narrative or numeric water quality criterion. An analysis of "reasonable potential" determines an effluent's capability to cause such excursions.

In determining the need for a permit limit for whole effluent toxicity or for an individual toxicant, the regulatory authority is required to consider, at a minimum, existing controls on point and nonpoint sources of pollution, the variability of the pollutant or pollutant parameter in the effluent, the sensitivity of the involved species to toxicity testing (for whole effluent), and, where appropriate, the dilution of the effluent in the receiving water (40 *CFR* 122.44(d)(ii)).

The regulatory authority is also required by NPDES regulations to consider whether technology-based limits are sufficient to maintain State water quality standards. There are two possibilities that will need to be assessed. First, if the limits based on appropriate treatment technology have already been specified in a previous permit, and if the facility is operating at the required level, then historical effluent and receiving water information can be used. Second, if the facility has yet to achieve the required technology performance (best available technology or best conventional technology), the regulatory authority will need to assess the technology-based limit for reasonable potential for causing or contributing to an excursion above the water quality standard.

In addition, the regulatory authority should consider all other available data and information pertaining to the discharger to assist in making an informed judgment. Where both effluent testing data and important other factors exist, the regulatory authority will need to exercise discretion in the determination of the need for a limit. **The authority should employ the principle of "independent application" of the data and information that characterizes the effluent.** In other words, effluent data alone, showing toxicity at the RWC, may be adequate to demonstrate the need for a limit for toxicity or for individual toxicants. Likewise, other factors may form an adequate basis for determining that limits are necessary. For example, where available dilution is low and monitoring information shows that toxic pollutants are frequently discharged at concentrations that have caused toxicity when discharged from similar facilities, the permitting authority may reason that a whole effluent toxicity limit is necessary even without whole effluent toxicity data from the specific facility. In all cases, the decision must be based upon consideration of factors cited in 40 *CFR* 122.44(d)(1)(ii). The regulatory authority will need to prioritize, on a case-by-case basis, the importance of all data and information used in making a determination. To assist in case-by-case determinations, recommended guidelines for characterizing an effluent for the need for a permit limit for whole effluent toxicity or individual toxicants are discussed below and summarized in Boxes 3-1 through 3-3.

Box 3-1. Determining "Reasonable Potential" for Excursions Above Ambient Criteria Using Factors Other than Facility-specific Effluent Monitoring Data

When determining the "reasonable potential" of a discharge to cause an excursion above a State water quality standard, the regulatory authority must consider all the factors listed in 40 *CFR* 122.44(d)(1)(ii). Examples of the types of information relating to these factors are listed below.

Existing controls on point and nonpoint sources of pollution

- Industry type: Primary, secondary, raw materials used, products produced, best management practices, control equipment, treatment efficiency, etc.
- Publicly owned treatment work type: Pretreatment, industrial loadings, number of taps, unit processes, treatment efficiencies, chlorination/ammonia problems, etc.

Variability of the pollutant or pollutant parameter in the effluent

- Compliance history
- Existing chemical data from discharge monitoring reports and applications.

Sensitivity of the species to toxicity testing

- Adopted State water quality criteria, or EPA criteria
- Any available in-stream survey data **applied under independent application of water quality standards**
- Receiving water type and designated/existing uses

Dilution of the effluent in the receiving water

- Dilution calculations

3.2 DETERMINING THE NEED FOR PERMIT LIMITS WITHOUT EFFLUENT MONITORING DATA FOR A SPECIFIC FACILITY

If the regulatory authority so chooses, or if the circumstances dictate, the authority may decide to develop and impose a permit limit for whole effluent toxicity or for individual toxicants without facility-specific effluent monitoring data, or prior to the generation of effluent data. Water quality-based permit limits can be set for a single toxicant or for whole effluent toxicity based on the available dilution and the water quality criterion or the State standard in the absence of facility specific effluent monitoring data. However, in doing so, the regulatory authority must satisfy all the requirements of 40 *CFR* 122.44(d)(1)(ii).

When determining whether or not a discharge causes, has the reasonable potential to cause, or contributes to an excursion of a numeric or narrative water quality criterion for individual toxicants or for toxicity, the regulatory authority can use a variety of factors and information where facility-specific effluent monitoring data are unavailable. These factors also should be considered with available effluent monitoring data. Some of these factors are the following:

- **Dilution**—Toxic impact is directly related to available dilution for the effluent. Dilution is related to the receiving stream flow and the size of the discharge. The lower the available dilution, the higher the potential for toxic effect. If an effluent's concentration at the edge of a mixing zone in a receiving water is expected to reach 1 percent or higher during critical or worst-case design periods, then such an effluent may require a toxicity limit (see discussion in Section 3.3.3). Assessment of the amount of stream dilution available should be made at the conditions required by the water quality standards or, if not specified in the standards, at the harmonic mean flow and the 7Q10 flow. Figure 3-3 (Pg. 57) shows that, whereas a majority of NPDES permittees nationwide discharge to areas during annual mean flow ranging in dilution from 100 to 1,000, the majority of dischargers fall into the 1 to 10 dilution range during low-flow conditions.

- **Type of industry**—Although dischargers should be individually characterized because toxicity problems are site-specific, the primary industrial categories should be of principal toxicity concern. EPA's treatment technology data base generally suggests that secondary industrial categories may have less potential for toxicity than primary industries. However, based on experience, it is virtually impossible to generalize the toxicity of effluents with any certainty. If two plants produce the same type of product, one effluent may be toxic while the other may not be toxic due to the type and efficiency of the treatment applied, general materials handling practices, and the functional target of the compound(s) being produced.

- **Type of POTW**—POTWs with loadings from indirect dischargers (particularly primary industries) may be candidates for toxicity limits. However, absence of industrial input does not guarantee an absence of POTW discharge toxicity problems. For example, commercial pesticide applicators often discharge to POTWs, resulting in pesticide concentrations in the POTW's effluent. Household disposal of pesticides, detergents, or other toxics may have a similar effect. The types of industrial users, their product lines, their raw materials, their potential and actual discharges, and their control equipment should be evaluated. POTWs should also be characterized for the possibility of chlorine and ammonia problems.

- **Existing data on toxic pollutants**—Discharge monitoring reports (DMRs) and data from NPDES permit application forms 2C and 2A may provide some indication of the presence of toxicants. The presence or absence of the 126 "priority pollutants" may or may not be an indication of the presence or absence of toxicity. There are thousands of "nonpriority" toxicants that may cause effluent toxicity. Also, combinations of several toxicants can produce ambient toxicity where the individual toxicants would not. EPA regulations at 40 *CFR* 122.21(j) require POTWs with design flows equal to or greater than 1 MGD and POTWs with approved pretreatment programs, or POTWs required to develop a pretreatment program, to submit the results of whole effluent toxicity tests with their permit applications. These regulations also provide discretion to the permitting authority to request such data from other POTWs at the time of permit application.

- **History of compliance problems and toxic impact**—Regulatory authorities may consider particular dischargers that have had difficulty complying with limits on toxicants or that have a history of known toxicity impacts as probable priority candidates for effluent toxicity limits.

- **Type of receiving water and designated use**—Regulatory authorities may compile data on water quality. Examples of available data include fish advisories or bans, reports of fish kills, State lists of priority waterbodies, and State lists of waters that are not meeting water quality standards. Regulatory authorities should use this information as a means of identifying point sources that discharge to impaired waterbodies and that thus may be contributing to this impairment. One source of this information is the lists of waters generated by states to comply with Section 304(l) regulations at 40 *CFR* 130.10(d)(6); 50 *FR* 23897-98, June 2, 1989:

    1) Waters where fishing or shellfish bans and/or advisories are currently in effect or are anticipated;

    2) Waters where there have been repeated fish kills or where abnormalities (cancers, lesions, tumors, etc.) have been observed in fish or other aquatic life during the last ten years;

    3) Waters where there are restrictions on water sports or recreational contact;

    4) Waters identified by the state in its most recent state section 305(b) report as either "partially achieving" or "not achieving" designated uses;

5) Waters identified by the states under section 303(d) of the Clean Water Act as waters needing water quality-based controls;

6) Waters identified by the state as priority water bodies;

7) Waters where ambient data indicate potential or actual excursions of water quality criteria due to toxic pollutants from an industry classified as a primary industry in Appendix A of 40 CFR Part 122;

8) Waters for which effluent toxicity test results indicate possible or actual excursions of state water quality standards, including narrative "free from" water quality criteria or EPA water quality criteria where state criteria are not available;

9) Waters with primary industrial major dischargers where dilution analyses indicate exceedances of state narrative or numeric water quality criteria (or EPA water quality criteria where state standards are not available) fortoxic pollutants, ammonia, or chlorine;

10) Waters with POTW dischargers requiring local pretreatment programs where dilution analyses indicate exceedances of state water quality criteria (or EPA water quality criteria where state water quality criteria are not available) for toxic pollutants, ammonia, or chlorine;

11) Waters with facilities not included in the previous two categories such as major POTWs, and industrial minor dischargers where dilution analyses indicate exceedances of numeric or narrative state water quality criteria (or EPA water quality criteria where state water quality criteria are not available) for toxic pollutants, ammonia, or chlorine;

12) Water classified for uses that will not support the "fishable/swimmable" goals of the Clean Water Act;

13) 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;

14) Waters identified by the state as impaired in its most recent Clean Lake Assessments conducted under 314 of the Clean Water Act; and

15) Surface waters impaired by pollutants from hazardous waste sites on the National Priority List prepared under section 105(8)(A) of CERCLA.

16) Waters judged to be impaired as a result of a bioassessment/biosurvey.

The presence of a combination of these factors, such as low available dilution, high-quality receiving water, poor compliance record, and clustered industrial and municipal discharges, could constitute a high priority for effluent limits.

Regardless, the regulatory authority, if it chooses to impose an effluent limit after conducting an effluent assessment without facility-specific monitoring data, will need to provide adequate justification for the limit in its permit development rationale or in its permit fact sheet. A clear and logical rationale for the need for the limit covering all of the regulatory points will be necessary to defend the limit should it be challenged. In justification of a limit, **EPA recommends that the more information the authority can acquire to support the limit, the better a position the authority will be in to defend the limit if necessary.** In such a case, the regulatory authority may well benefit from the collection of effluent monitoring data prior to establishing the limit.

If the regulatory authority, after evaluating all available information on the effluent, in the absence of effluent monitoring data, is not able to decide whether the discharge causes, has the reasonable potential to cause, or contributes to, an excursion above a numeric or narrative criterion for whole effluent toxicity or for individual toxicants, the authority should require whole effluent toxicity or chemical-specific testing to gather further evidence. In such a case, the regulatory authority can require the monitoring prior to permit issuance, if sufficient time exists, or it may require the testing as a condition of the issued/reissued permit.

Under these circumstances, the regulatory authority may find it protective of water quality to include a permit reopener for the imposition of an effluent limit should the effluent testing establish that the discharge causes, has the reasonable potential to cause, or contributes to excursion above a water quality criteria. A discussion of these options is provided later in this chapter.

3.3 DETERMINING THE NEED FOR PERMIT LIMITS WITH EFFLUENT MONITORING DATA

3.3.1 General Considerations

When characterizing an effluent for the need for a whole effluent toxicity limit, and/or an individual toxicant limit, the regulatory authority should use any available effluent monitoring data, together with any information like that discussed under Section 3.2 above, as the basis for a decision. The regulatory authority may already have effluent toxicity data available from previous monitoring, or it may decide to require the permittee to generate effluent monitoring data prior to permit issuance or as a condition of the issued permit. EPA regulations at 40 *CFR* 122.21(j) require POTWs with design flows equal to or greater than 1 MGD and POTWs with approved pretreatment programs, or POTWs required to develop a pretreatment program, to submit the results of whole effluent toxicity tests with their permit applications. These regulations also provide discretion to the permitting authority to request such data from additional POTWs at the time of permit application.

In the instance where the permittee is required to generate data in advance, data collection should begin 12 to 18 months in advance of permit development to allow adequate time for conducting toxicity tests and chemical analyses. The type of data, including toxicity testing data, should be specified by the regulatory authority at the outset so that decisions on permit actions will not be delayed. **EPA recommends monitoring data be generated on effluent toxicity prior to permit limit development for the following reasons: (1) the presence or absence of effluent toxicity can be more clearly established or refuted and (2) where toxicity is shown, effluent variability can be more clearly defined.** Several basic factors that should be considered in generating effluent monitoring data are discussed below.

3.3.2 Addressing Uncertainty in Effluent Characterization by Generating Effluent Monitoring Data

All toxic effects testing and exposure assessment parameters, for both effluent toxicity and individual chemicals, have some degree of uncertainty associated with them. The more limited the amount of test data available, the larger the uncertainty. The least amount of uncertainty of an effluent's impact on the receiving water exists where (1) a complete data base is available on the effects of acute and chronic toxicity on many indigenous species, (2) there is a clear understanding of ecosystem species composition and functional processes, and (3) actual measured exposure concentrations are available for all chemicals during seasonal changes and dilution situations. The uncertainty associated with such an ideal situation would be minimal. However, generation of these data can be very resource intensive.

An example of uncertainty that results from limited monitoring data is if a regulatory authority has only one piece of effluent data (e.g., an LC_{50} of 50 percent) for a facility. Effluent variability in such a case, given the range of effluent toxicity variability seen in other effluents, may range between 20 percent and 100 percent (see Appendix A). It is impossible to determine from one piece of monitoring data where in this range the effluent variability really falls. More monitoring data would need to be generated to determine the actual variability of this effluent and reduce this source of uncertainty.

To better characterize the effects of effluent variability and reduce uncertainty in the process of deciding whether to require an effluent limit, EPA has developed the statistical approach described below and in Box 3-2. This approach combines knowledge of effluent variability as estimated by a coefficient of variation with the uncertainty due to a limited number of data to project an estimated maximum concentration for the effluent. The estimated maximum concentration is calculated as the upper bound of the expected lognormal distribution of effluent concentrations at a high confidence level. The projected effluent concentration after consideration of dilution can then be compared to an appropriate water quality criterion to determine the potential for exceeding that criterion and the need for an effluent limit.

The statistical approach has two parts. The first is a characterization of the highest measured effluent concentration based on the desired confidence level. The relationship that describes this is the following:

$$p_n = (1 - \text{confidence level})^{1/n}$$

where p_n is the percentile represented by the highest concentration in the data and n is the number of samples. The following are some examples of this relationship at a 99 percent confidence level:

- The largest value of 5 samples is greater than the 40 percentile
- The largest value of 10 samples is greater than the 63 percentile
- The largest value of 20 samples is greater than the 79 percentile
- The largest value of 100 samples is greater than the 96 percentile.

The second part of the statistical approach is a relationship between the percentile described above and the selected upper bound of the lognormal effluent distribution. EPA's effluent data base suggests that the lognormal distribution well characterizes effluent concentrations (see Appendix E). For example, if five samples were collected (which represents a 40th percentile), the coefficient of variation is 0.6, and the desired upper bound of the effluent distribution is the 99th percentile, then the two percentiles can be related using the coefficient of variation (CV) as shown below:

$$\frac{C_{99}}{C_{40}} = \frac{\exp(2.326\sigma - 0.5\sigma^2)}{\exp(-0.258\sigma - 0.5\sigma^2)} = 4.2$$

where $\sigma^2 = \ln(CV^2+1)$ and 2.326 and -0.258 are the normal distribution values for the 99th and 40th percentiles, respectively. The use of the 99th percentile is for illustrative purposes here. Although it does represent a measure of the upper bound of an effluent distribution, other percentiles could be selected by a regulatory agency. The relationship shown above can be calculated for other percentiles and CVs by replacing the values in the equation.

Tables 3-1 and 3-2 show the combined effects of both parts for a 99-percent confidence level and upper bounds of the 99th and 95th percentiles, respectively. The factors shown in the tables are multiplied by the highest concentration in an effluent sample to estimate the maximum expected concentration.

This procedure can be used for both single and multiple discharges to the same receiving waterbody. This is accomplished for multiple dischargers by summing the projected RWCs for the pollutant or pollutant parameter of concern from each individual discharger, and comparing it to the water quality standard. This involves an assumption of conservative additivity of the pollutant after discharge, which may not accurately reflect the true behavior of the toxicant. To overcome this, and to further refine the proportional contribution of each discharger and the resultant limits, the permitting authority should supplement this evaluation with multiple source WLA modeling and/or ambient water concentration monitoring.

Box 3-2. Determining "Reasonable Potential" for Excursions Above Ambient Criteria Using Effluent Data Only

EPA recommends finding that a permittee has "reasonable potential" to exceed a receiving water quality standard if it cannot be demonstrated with a high confidence level that the upper bound of the lognormal distribution of effluent concentrations is below the receiving water criteria at specified low-flow conditions.

Step 1 Determine the number of total observations ("*n*") for a particular set of effluent data (concentrations or toxic units [TUs]), and determine the highest value from that data set.

Step 2 Determine the coefficient of variation for the data set. For a data set where $n<10$, the coefficient of variation (CV) is estimated to equal 0.6, or the CV is calculated from data obtained from a discharger. For a data set where $n>10$, the CV is calculated as standard deviation/mean (see Figure 3-1). For less than 10 items of data, the uncertainty in the CV is too large to calculate a standard deviation or mean with sufficient confidence.

Step 3 Determine the appropriate ratio from Table 3-1 or 3-2.

Step 4 Multiply the highest value from a data set by the value from Table 3-1 or 3-2. Use this value with the appropriate dilution to project a maximum receiving water concentration (RWC).

Step 5 Compare the projected maximum RWC to the applicable standard (criteria maximum concentration, criteria continuous concentration [CCC], or reference ambient concentration). EPA recommends that permitting authorities find reasonable potential when the projected RWC is greater than an ambient criterion.

Example

Consider the following results of toxicity measurements of an effluent that is being characterized: 5 TU_c, 2 TU_c, 9 TU_c, and 6 TU_c. Assume that the effluent is diluted to 2 percent at the edge of the mixing zone. Further assume that the CV is 0.6, the upper bound of the effluent distribution is the 99th percentile, and the confidence level is 99 percent.

Step 1 There are four samples, and the maximum value of the sample results is 9 TU_c.

Step 2 The value of the CV is 0.6.

Step 3 The value of the ratio for four pieces of data and a CV of 0.6 is 4.7.

Step 4 The value that exceeds the 99th percentile of the distribution (ratio times x_{max}) after dilution is calculated as:

$$[9\ TU_c \times 4.7 \times 0.02] = 0.85\ TU_c.$$

Step 5 0.85 TU_c is less than the ambient criteria concentration of 1.0 TU_c. There is no reasonable potential for this effluent to cause an excursion above the CCC.

3.3.3 Effluent Characterization for Whole Effluent Toxicity

Once an effluent has been selected for whole effluent toxicity characterization after consideration of the factors discussed above, the regulatory authority should require toxicity testing in accordance with appropriate site-specific considerations and the recommendations discussed below. In the past 5 years, significant additional experience has been gained in generating effluent toxicity data upon which to make decisions as to whether or not an effluent will cause toxic effects in the receiving water in both freshwater and marine environments.

General Considerations and Assumptions

EPA has revised its initial effluent toxicity data generation recommendations based on three observations made over the last 5 years:

1) Only rarely have effluents discharged by NPDES permittees been observed to have LC_{50}s less than 1.0 percent or no observed effect concentrations (NOECs) less than 0.1 percent. However, there is always a chance that an effluent could be toxic at such low effluent concentrations.

Table 3-1. Reasonable Potential Multiplying Factors: 99% Confidence Level and 99% Probability Basis

Number of	Coefficient of Variation																			
Samples	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
1	1.6	2.5	3.9	6.0	9.0	13.2	18.9	26.5	36.2	48.3	63.3	81.4	102.8	128.0	157.1	190.3	227.8	269.9	316.7	368.3
2	1.4	2.0	2.9	4.0	5.5	7.4	9.8	12.7	16.1	20.2	24.9	30.3	36.3	43.0	50.4	58.4	67.2	76.6	86.7	97.5
3	1.4	1.9	2.5	3.3	4.4	5.6	7.2	8.9	11.0	13.4	16.0	19.0	22.2	25.7	29.4	33.5	37.7	42.3	47.0	52.0
4	1.3	1.7	2.3	2.9	3.8	4.7	5.9	7.2	8.7	10.3	12.2	14.2	16.3	18.6	21.0	23.6	26.3	29.1	32.1	35.1
5	1.3	1.7	2.1	2.7	3.4	4.2	5.1	6.2	7.3	8.6	10.0	11.5	13.1	14.8	16.6	18.4	20.4	22.4	24.5	26.6
6	1.3	1.6	2.0	2.5	3.1	3.8	4.6	5.5	6.4	7.5	8.6	9.8	11.1	12.4	13.8	15.3	16.8	18.3	19.9	21.5
7	1.3	1.6	2.0	2.4	2.9	3.6	4.2	5.0	5.8	6.7	7.7	8.7	9.7	10.8	12.0	13.1	14.4	15.6	16.9	18.2
8	1.2	1.5	1.9	2.3	2.8	3.3	3.9	4.6	5.3	6.1	6.9	7.8	8.7	9.6	10.6	11.6	12.6	13.6	14.7	15.8
9	1.2	1.5	1.8	2.2	2.7	3.2	3.7	4.3	5.0	5.7	6.4	7.1	7.9	8.7	9.6	10.4	11.3	12.2	13.1	14.0
10	1.2	1.5	1.8	2.2	2.6	3.0	3.5	4.1	4.7	5.3	5.9	6.6	7.3	8.0	8.8	9.5	10.3	11.0	11.8	12.6
11	1.2	1.5	1.8	2.1	2.5	2.9	3.4	3.9	4.4	5.0	5.6	6.2	6.8	7.4	8.1	8.8	9.4	10.1	10.8	11.5
12	1.2	1.4	1.7	2.0	2.4	2.8	3.2	3.7	4.2	4.7	5.2	5.8	6.4	7.0	7.5	8.1	8.8	9.4	10.0	10.6
13	1.2	1.4	1.7	2.0	2.3	2.7	3.1	3.6	4.0	4.5	5.0	5.5	6.0	6.5	7.1	7.6	8.2	8.7	9.3	9.9
14	1.2	1.4	1.7	2.0	2.3	2.6	3.0	3.4	3.9	4.3	4.8	5.2	5.7	6.2	6.7	7.2	7.7	8.2	8.7	9.2
15	1.2	1.4	1.6	1.9	2.2	2.6	2.9	3.3	3.7	4.1	4.6	5.0	5.4	5.9	6.4	6.8	7.3	7.7	8.2	8.7
16	1.2	1.4	1.6	1.9	2.2	2.5	2.9	3.2	3.6	4.0	4.4	4.8	5.2	5.6	6.1	6.5	6.9	7.3	7.8	8.2
17	1.2	1.4	1.6	1.9	2.1	2.5	2.8	3.1	3.5	3.8	4.2	4.6	5.0	5.4	5.8	6.2	6.6	7.0	7.4	7.8
18	1.2	1.4	1.6	1.8	2.1	2.4	2.7	3.0	3.4	3.7	4.1	4.4	4.8	5.2	5.6	5.9	6.3	6.7	7.0	7.4
19	1.2	1.4	1.6	1.8	2.1	2.4	2.7	3.0	3.3	3.6	4.0	4.3	4.6	5.0	5.3	5.7	6.0	6.4	6.7	7.1
20	1.2	1.3	1.6	1.8	2.0	2.3	2.6	2.9	3.2	3.5	3.8	4.2	4.5	4.8	5.2	5.5	5.8	6.1	6.5	6.8

Table 3-2. Reasonable Potential Multiplying Factors: 95% Confidence Level and 95% Probability Basis

Number of	Coefficient of Variation																			
Samples	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	1.0	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0
1	1.4	1.9	2.6	3.6	4.7	6.2	8.0	10.1	12.6	15.5	18.7	22.3	26.4	30.8	35.6	40.7	46.2	52.1	58.4	64.9
2	1.3	1.6	2.0	2.5	3.1	3.8	4.6	5.4	6.4	7.4	8.5	9.7	10.9	12.2	13.6	15.0	16.4	17.9	19.5	21.1
3	1.2	1.5	1.8	2.1	2.5	3.0	3.5	4.0	4.6	5.2	5.8	6.5	7.2	7.9	8.6	9.3	10.0	10.8	11.5	12.3
4	1.2	1.4	1.7	1.9	2.2	2.6	2.9	3.3	3.7	4.2	4.6	5.0	5.5	6.0	6.4	6.9	7.4	7.8	8.3	8.8
5	1.2	1.4	1.6	1.8	2.1	2.3	2.6	2.9	3.2	3.6	3.9	4.2	4.5	4.9	5.2	5.6	5.9	6.2	6.6	6.9
6	1.1	1.3	1.5	1.7	1.9	2.1	2.4	2.6	2.9	3.1	3.4	3.7	3.9	4.2	4.5	4.7	5.0	5.2	5.5	5.7
7	1.1	1.3	1.4	1.6	1.8	2.0	2.2	2.4	2.6	2.8	3.1	3.3	3.5	3.7	3.9	4.1	4.3	4.5	4.7	4.9
8	1.1	1.3	1.4	1.6	1.7	1.9	2.1	2.3	2.4	2.6	2.8	3.0	3.2	3.3	3.5	3.7	3.9	4.0	4.2	4.3
9	1.1	1.2	1.4	1.5	1.7	1.8	2.0	2.1	2.3	2.4	2.6	2.8	2.9	3.1	3.2	3.4	3.5	3.6	3.8	3.9
10	1.1	1.2	1.3	1.5	1.6	1.7	1.9	2.0	2.2	2.3	2.4	2.6	2.7	2.8	3.0	3.1	3.2	3.3	3.4	3.6
11	1.1	1.2	1.3	1.4	1.6	1.7	1.8	1.9	2.1	2.2	2.3	2.4	2.5	2.7	2.8	2.9	3.0	3.1	3.2	3.3
12	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.9	2.0	2.1	2.2	2.3	2.4	2.5	2.6	2.7	2.8	2.9	3.0	3.0
13	1.1	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.9	2.0	2.1	2.2	2.3	2.4	2.5	2.5	2.6	2.7	2.8	2.9
14	1.1	1.2	1.3	1.4	1.4	1.5	1.6	1.7	1.8	1.9	2.0	2.1	2.2	2.3	2.3	2.4	2.5	2.6	2.6	2.7
15	1.1	1.2	1.2	1.3	1.4	1.5	1.6	1.7	1.8	1.8	1.9	2.0	2.1	2.2	2.2	2.3	2.4	2.4	2.5	2.5
16	1.1	1.1	1.2	1.3	1.4	1.5	1.6	1.6	1.7	1.8	1.9	1.9	2.0	2.1	2.1	2.2	2.3	2.3	2.4	2.4
17	1.1	1.1	1.2	1.3	1.4	1.4	1.5	1.6	1.7	1.7	1.8	1.9	1.9	2.0	2.0	2.1	2.2	2.2	2.3	2.3
18	1.1	1.1	1.2	1.3	1.3	1.4	1.5	1.6	1.6	1.7	1.7	1.8	1.9	1.9	2.0	2.0	2.1	2.1	2.2	2.2
19	1.1	1.1	1.2	1.3	1.3	1.4	1.5	1.5	1.6	1.6	1.7	1.8	1.8	1.9	1.9	2.0	2.0	2.0	2.1	2.1
20	1.1	1.1	1.2	1.2	1.3	1.4	1.4	1.5	1.5	1.6	1.7	1.7	1.8	1.8	1.8	1.9	1.9	2.0	2.0	2.0

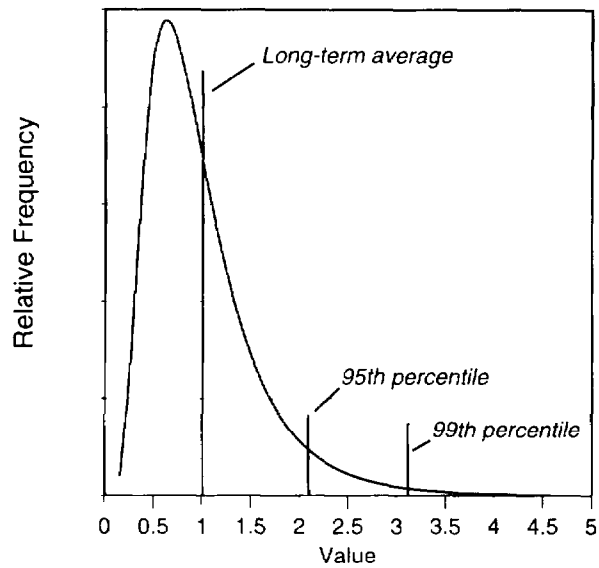


Figure 3-1a. Frequency Distribution of Values for a Lognormal Distribution with a Mean of 1.0 and a Coefficient of Variation of 0.6

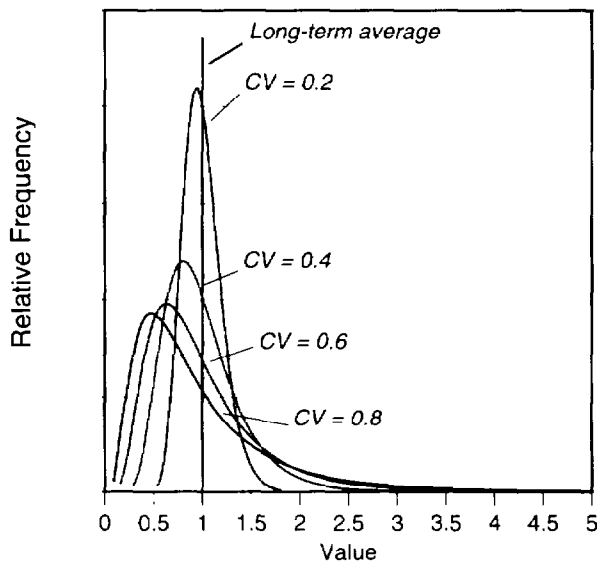


Figure 3-1b. Comparison of Relative Frequencies of Lognormal Distributions with a Mean of 1.0 for Different Coefficients of Variation

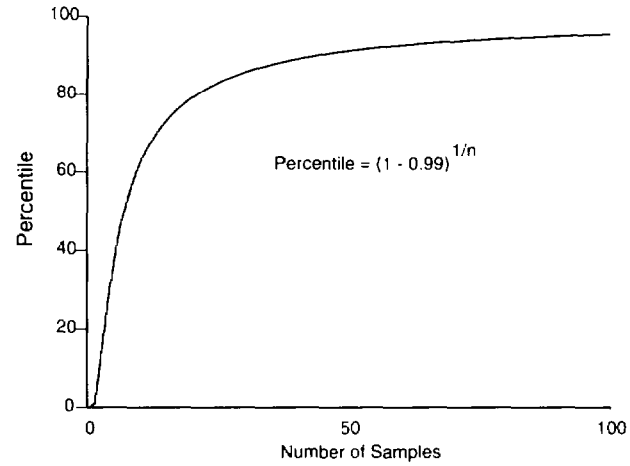


Figure 3-1c. Relationship Between the Largest Value of n Samples and the Percentile It Exceeds with 99 Percent Confidence

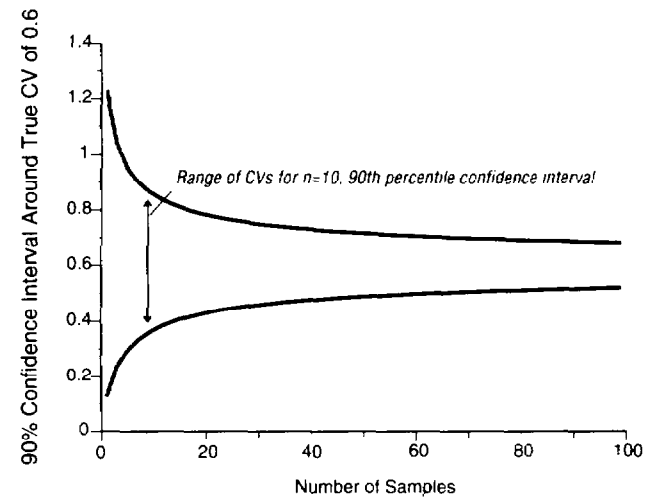
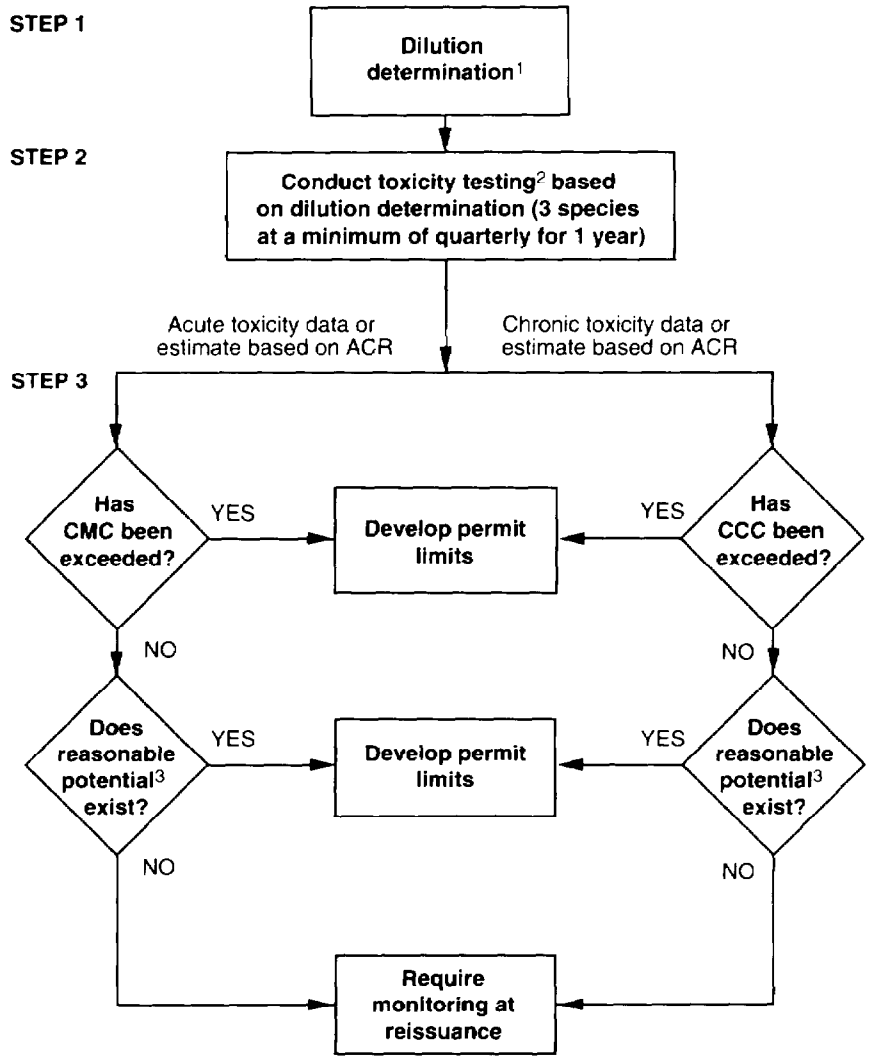


Figure 3-1d. Example of 90 Percent Confidence Intervals Around Coefficient of Variation Estimates for Numbers of Samples

2) With the exception of a small number of "outliers" for which confirmation is not possible, acute-to-chronic ratios (ACRs) above 20 for effluents discharged by NPDES permittees have not been observed by EPA. The majority of observed ACRs are very seldom above 10. However, higher ACRs may be found for selected facilities.

3) The use of the three commonly used freshwater species and of three of the five commonly used marine organisms has generally been sufficient to measure any effluent's toxicity for the purposes of projecting effluent toxicity impact and making regulatory decisions.

Figure 3-2 is a flow chart of EPA's recommendations for data generation for three different dilution scenarios. It is divided into three basic steps: determining initial dilution, developing toxicity testing procedures, and developing decision criteria for permit limit. There are certain basic assumptions built into this flow chart. The basic principle used in making decisions is to compare available dilution to known or projected toxic effect concentrations in order to place an effluent into one of three categories:



Notes:

[1]Dilution determinations should be performed for critical flows and any applicable mixing zones.

[2]Toxicity testing recommendations

a. Dilution > 1000:1: acute testing, check CMC only.

b. 100:1 < Dilution < 1000:1: acute or chronic testing, check CMC and CCC with data or ACR.

c. Dilution < 100:1: conduct chronic testing, check CCC with data and CMC using acute data or ACR.

[3]Reasonable potential: Use procedures in Box 3-3.

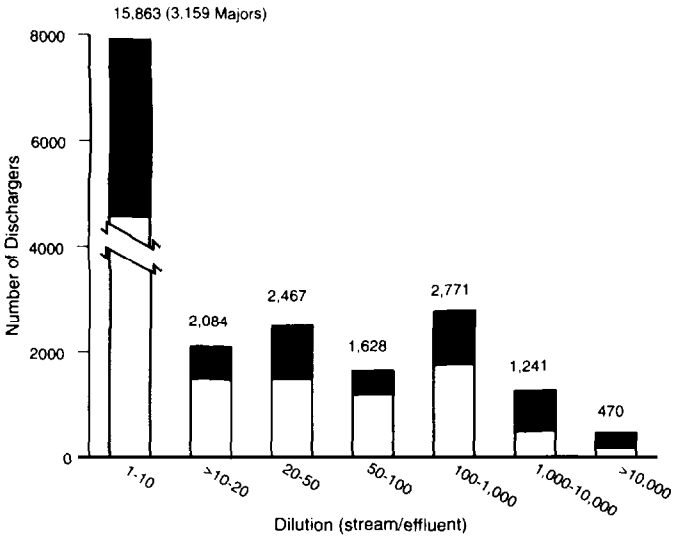
Figure 3-2. Effluent Characterization for Whole Effluent Toxicity

1) The effluent causes or contributes to an excursion of a numeric or narrative water quality criterion and the permit requires a limit on toxicity.

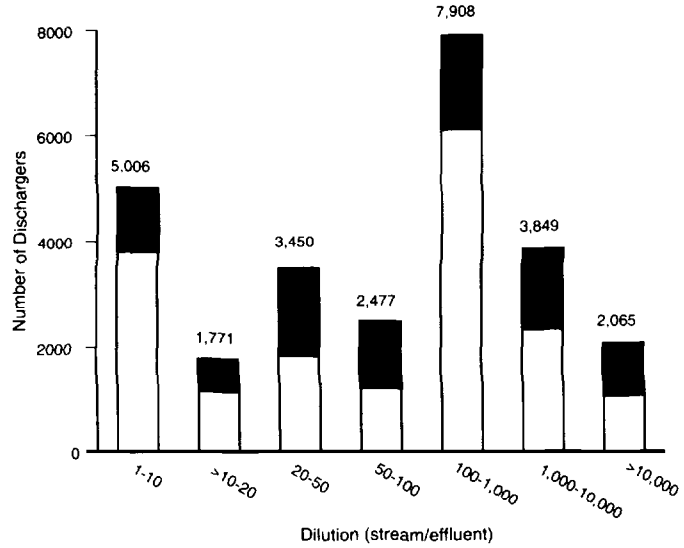
2) The effluent has a reasonable potential of causing or contributing to an excursion of a numeric or narrative water quality criterion and a limit is required.

3) The effluent has a very low probability of causing or contributing to an excursion of a water quality standard and no limit is required.

This categorization is accomplished by using dilution estimates in the first step and the results of the toxicity tests in the next steps. In addition, all these impact estimates assume discharge at critical conditions and imposition of any applicable mixing zone requirements. Therefore, a conservative assumption is used to determine whether or not an impact is projected to occur. Estimates of possible toxic impact are made assuming that the effluent is most toxic to the most sensitive species or lifestage at the time of lowest available dilution.



(a) At Low Flow (7Q10)



(b) At Annual Mean Flow

Figure 3-3. National Distribution of NPDES Dilution Conditions at 7Q10 and at Annual Mean Flow

The changes to the EPA's data generation recommendations eliminate the application of multiple sets of safety margins that was proposed in the 1985 version of this document. Rather, general observations on effluent toxicity described above now allow regulatory authorities to tighten the bounds of the initial dilution categorization, eliminate the species sensitivity uncertainty factor and target LC_{50}s of 1 percent and NOECs of 0.1 percent as the most extreme toxicity measurements that can normally be expected for the vast majority of effluents discharged by NPDES permittees for acute and chronic toxicity, respectively. The observation of toxicity was based on multiple dilution tests. The same observation may not hold for toxicity measured with single dilution tests (pass/fail). As reflected in Chapter 1, single dilution toxicity tests are much more variable than multiple dilution tests. **Therefore, the use of single concentration toxicity tests is strongly discouraged for this data generation process.**

Since the new data generation requirements are much less expensive than the previous requirements, tiered testing (less expensive, single-concentration, initial screening followed by increasingly expensive definitive data generation, using multiconcentration tests, as described in the September 1985 version of the technical support document) is unnecessary. However, **elimination of the requirement to conduct toxicity testing on the basis of projections using dilution alone is not recommended.** Although EPA's data review suggests that an LC_{50} of 1 percent and an NOEC of 0.1 percent are the lower bounds on effluent toxicity, there may be other effluents that are presently unmeasured that are more toxic. Testing data are always desirable for fully characterizing discharges of concern.

Steps in Whole Effluent Characterization Process

The following is a detailed description of the major steps presented in Figure 3-2 and the rationale behind each.

Step 1: Dilution Determination

The initial step is to determine the dilution of the effluent at the edge of the mixing zone, assuming the State allows mixing zones. Figure 3-4 shows a schematic representation of typical mixing zone requirements for both acute and chronic toxicity. Calculating the dilution at the edges of mixing zones for site-specific situations can be complicated. Modeling can be employed using either steady-state or dynamic approaches to calculate the dilution (see Chapter 4). However, for complex situations, such as marine and estuarine waters or lakes, dye studies (or other techniques used to assess mixing zones) may still be required.

Some State water quality standards do not allow the use of mixing in the control of acute toxicity. For these States, acute toxicity is often limited at the end of the pipe. Permit limits derived to enforce such requirements would be considered "water quality-based" because they would be based upon an ambient criterion (as opposed to an arbitrary test endpoint). Regardless, both chronic and acute toxicity must be assessed in these situations.

Step 2: Toxicity Testing Procedures

Where toxicity tests are required in order to make decisions regarding appropriate next steps in a screening protocol, **EPA recommends as a minimum that three species (for example, a vertebrate, an invertebrate, and a plant) be tested quarterly for a minimum of 1 year.** As discussed in Chapter 1, the use of three species is strongly recommended. Experience indicates that marine algae can be a highly sensitive test species for some effluents. Using a surrogate species of the plant kingdom adds another trophic level to the testing regimen. For both freshwater and marine situations, the use of three species is more protective than two species since a wider range of species sensitivity can be measured. EPA is continuing to develop toxicity test methods using additional organisms including plants. In addition, EPA has revised the test for *Selenastnum*, which has improved the test precision.

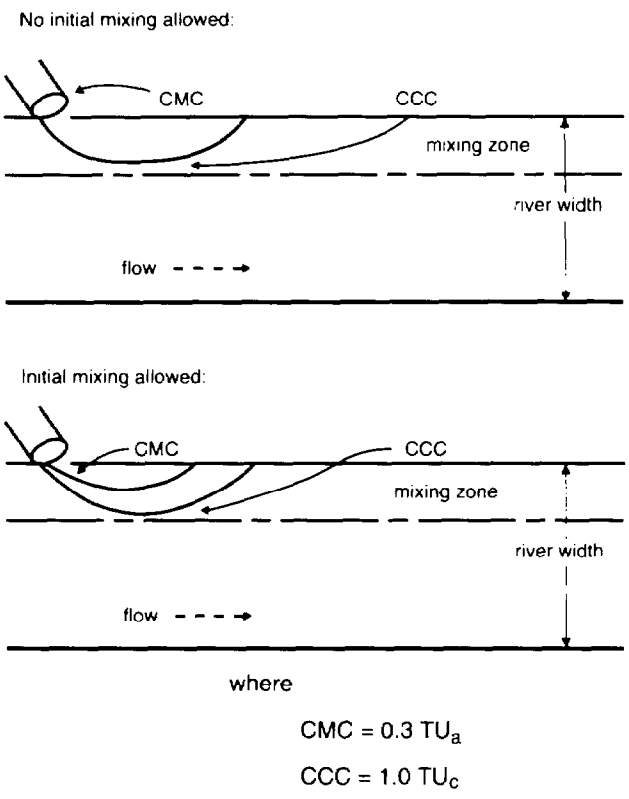


Figure 3-4. Schematic Representation of Mixing Zone Areas Where the CMC and CCC Apply

EPA recommends against selecting a "most sensitive" species for toxicity testing. For one organism to consistently be the most sensitive in a battery of toxicity tests, two conditions must occur: (1) the toxicants causing toxicity must remain the same, and (2) the ratios of the toxicants in the effluent (if more than one) must remain the same. Based on EPA's experience at the Duluth research laboratory, neither of these conditions is likely to occur. For example, the causes of effluent toxicity in POTWs can vary on a seasonal basis. Toxicity in the summer can be caused by pesticides to which invertebrates are most sensitive. However, the winter toxicity could be caused by ammonia to which fathead minnows will respond most sensitively. The most sensitive species for an effluent actually may not exist and at best is difficult to identify.

Conducting toxicity tests using three species quarterly for 1 year is recommended to adequately assess the variability of toxicity observed in effluents. Below this minimum, the chances of missing toxic events increase. The toxicity test result for the most sensitive of the tested species is considered to be the measured toxicity for a particular effluent sample.

The data generation recommendations in Figure 3-2 represent minimum testing requirements. Since uncertainty regarding whether or not an effluent causes toxic impact is reduced with more data, **EPA recommends that this test frequency be increased where necessary to adequately assess effluent variability.** If less frequent testing is required in the permit, it is preferable to use three species tested less frequently than to test the effluent more frequently with only a single species whose sensitivity to the effluent is not well characterized.

EPA recommends that a discharger conduct acute toxicity testing if the dilution of the effluent is greater than 1000:1 at the edge of the mixing zone [3]. Such a discharger would be considered a low priority for chronic toxicity testing. The rationale for this is that the effluent concentration would be below 0.1 percent at the edge of the mixing zone and thus incapable of causing an excursion above the CCC. A worst case NOEC of 0.1 percent translates into 1,000 TU_c, which would result in a concentration of less than 1.0 TU_c at the edge of the mixing zone for this dilution category. The test results would be compared to the CMC after consideration of any allowable mixing.

EPA recommends that a discharger conduct either acute or chronic toxicity testing if the dilution of the effluent falls between 100:1 and 1,000:1 at the edge of the mixing zone. Effluents have been shown to be both acutely and chronically toxic within this range of receiving water dilution. Under worst-case scenarios, LC_{50}s of 1.0 percent and ACRs of 10 will result in excursions above both the CCC and CMC at the edge of the regulatory mixing zone.

Although either acute or chronic testing can be required within this dilution range, acute testing would be more appropriate at the higher end of this dilution range (1,000:1 or 0.1 percent). At the lower end of this dilution range (100:1 or 1.0 percent), chronic tests may be more appropriate. Where other factors are equal, chronic testing may be preferable since the interim results in a chronic test gives data on acute toxicity as well. The acute endpoint data can then be used to compare directly to the CMC without the need for an ACR.

Whichever type of toxicity test (either acute or chronic) is specified, the results from that test should be compared to the criterion associated with that type of test. For example, a chronic test would be compared to the CCC. Comparisons to the other criteria can be made by using the ACR or additional data generated to convert a chronic test result to an acute endpoint and vice versa. For example, a chronic NOEC of 5 percent effluent (or 20 TU_c) represents an acute LC_{50} of 50 percent (or 2 TU_a) at an ACR of 10.

EPA recommends that a discharger conduct chronic toxicity testing if the dilution of the effluent falls below 100:1 at the edge of the mixing zone. The rationale for this recommendation is that chronic toxicity has been observed in some effluents down to the 1.0 percent effect concentration. Therefore, chronic toxicity tests, although somewhat more expensive to conduct, should be used directly in order to make decisions about toxic impact.

There is a potential for acute toxicity within this dilution range, although this is less likely as the 100:1 dilution level is approached. Thus, the recommended screening protocol shown in Figure 3-2 includes a determination of whether excursions above the CMC are projected [4]. This analysis may be performed by assuming an ACR, applying this value to the chronic toxicity testing data, and allowing for any allowable initial mixing. Alternatively, the regulatory authority may use the interim results in the chronic test to calculate the acute toxicity.

Both the chronic and acute toxicity test data would be compared to their respective criterion. The chronic test results would be compared to the CCC, and the acute results, regardless of how calculated, would be compared to the CMC.

Step 3: Decision Criteria for Permit Limit Development

Once the toxicity data have been generated for a discharger, the regulatory authority must decide whether or not the results show that the permittee causes, has the reasonable potential to cause, or contributes to an excursion of an applicable numeric or narrative water quality criterion and therefore needs to limit effluent toxicity. To do this, these data should be used to project receiving water concentrations, which are then compared to the CCC and CMC. One of four outcomes will be reached when following the screening protocol shown in Figure 3-2:

1) Excursion Above CMC or CCC—Where any one data point shows an excursion above the State's numeric or narrative criterion for the parameter toxicity, EPA regulations require a permit limit be set for whole effluent toxicity (40 *CFR* 122.44(d)(1)(iv or v)), unless limits on a specific chemical will allow the narrative water quality criterion to be attained or maintained. In the absence of a State numeric criterion for the parameter toxicity, **EPA recommends that 1.0 TU_c and 0.3 TU_a be used as the CCC and CMC, respectively.** The decision to develop permit limits based upon an excursion above either the CMC or CCC will lead to protection against both acute and chronic toxicity if the permit derivation procedures in Chapter 5 are used to set effluent limits.

2) Reasonable Potential for Excursion Above CMC or CCC—**EPA believes that "reasonable potential" is shown where an effluent is projected to cause an excursion above the CCC or CMC. This projection is based upon a statistical analysis of available data that accounts for limited sample size and effluent variability.** EPA's detailed recommendations for making a statistical determination based upon effluent monitoring data alone are shown in Box 3-2. Where a regulatory authority finds that test results alone indicate a "reasonable potential" to cause an excursion above a State water quality criterion in accordance with 40 *CFR* 122.44(d)(1)(ii), a permit limit must be developed.

   A regulatory authority may select an alternative approach for assessing reasonable potential. For example, an authority may opt to use a stochastic dilution model that incorporates both ambient dilution and effluent variability for determining reasonable potential. Such an approach is analogous to the statistical approach shown in Box 3-2. Whatever approach selected by the authority, it must use all the factors that account for all the factors listed in 40 *CFR* 122.44(d)(1)(ii).

   In some cases the statistical analysis of the effluent data may not actually project an excursion above the CMC or CCC but may be close. Under such conditions, reasonable potential determinations will include an element of judgment on the part of the regulatory authority. Other factors will need to be considered and given appropriate weight in the decisionmaking process, including value of waterbody (e.g., high-use fishery), relative proximity to the CCC or CMC, existing controls on point and nonpoint sources, information on effluent variability, compliance history of the facility, and type of treatment facility. These factors are summarized in Box 3-2 and are discussed in detail in Section 3.1. **EPA recommends regulatory authorities establish a written policy and procedure for making determinations of "reasonable potential" under these circumstances.**

3) No Reasonable Potential for Excursions Above CMC or CCC—In these situations, **EPA recommends that the toxicity tests recommended above be repeated at a frequency of at least once every 5 years as a part of the permit application.** Such testing is required for certain POTWs under 40 *CFR* 122.21(j).

4) Inadequate Information—Where a regulatory authority has inadequate information to determine reasonable potential for an excursion of a numeric or narrative water quality criterion, there may still be a basis for concern on the part of the authority. The permit should contain whole effluent toxicity monitoring requirements and a reopener clause. This clause would require reopening of the permit and establishment of a limit based upon any test results, or other new factors, which substantiate that the effluent causes, has the reasonable potential of causing, or contributes to an excursion above the CCC or CMC.

3.3.4 Use of Toxicity Testing in Multiple-source Discharge Situations

Where more than one discharge to the same receiving waterbody contributes, or has the reasonable potential to contribute to an excursion of water quality standards, permit limits must be developed for each individual discharger on that waterbody. For the regulatory authority to make this assessment, additional testing may be needed to provide the authority with the information necessary to assess the relative impact of each source. For purposes of this discussion, a multiple-source discharge situation is defined as a situation where impact zones overlap, or where ambient receiving water concentrations of a pollutant are elevated due to upstream discharges. In multiple-source discharge situations, additivity, antagonism, and persistence of toxicity can be of concern. To collect additional data, the permit authority should employ the toxicity testing procedures for multiple dischargers described in Box 3-3. In addition, ambient toxicity testing, as described below, could be used.

Assuming that screening has been conducted that reveals the need for permit limits, two options for controlling the discharges exist. The first option is for the permit authority to regulate each source separately using the procedures for individual point sources. In this option, the permitting authority would require use of upstream ambient water as a diluent in the toxicity test so as to be able to evaluate the contributions of upstream sources of toxicity. A second option is to treat each discharge as an interactive component of a whole system. In this option, the permit writer would determine a total maximum daily load for the receiving waterbody and develop individual wasteload allocations for each discharger using the procedures discussed in Chapter 4.

Box 3-3. Recommend Multiple-source Toxicity Testing Procedures

Tests

Where the combined effluents make up 1 percent or greater of the receiving waters, conduct chronic toxicity tests following the testing procedures described in Section 3.3.3.

Where the combined effluents make up less than 1 percent of the receiving waters, conduct acute toxicity tests following the testing procedures described in Section 3.3.3 (see Figure 3-2) to determine if any of the effluents are exhibiting toxicity.

An additional data requirement is the assessment of relative and absolute toxicity of each source so that appropriate permit conditions can be set for individual dischargers. The following procedure is suggested.

1) Conduct one set of toxicity tests on the effluents using a control of reconstituted or uncontaminated dilution water. The set of tests will give an absolute toxicity measurement of the effluent.

2) Run a parallel set of toxicity tests on the effluent using dilution water taken directly upstream from the point of discharge or, for estuarine waters, from an area outside of the immediate discharge impact zone (this will have to be determined by a dye study). This dilution water may be contaminated with upstream effluents or other toxicant sources. The purpose of this test is to project toxic impact of the effluent after it is mixed at its point of discharge. This is a relative effluent toxicity measurement. The relative testing procedure could result in a change in the standard concentration-effect curve generated by the testing. The dilution water for the relative toxicity test may cause significant mortality, growth, or reproductive effects at the lower effluent concentrations (including the 100 percent diluent control concentration) if the diluent from the receiving water is toxic (from an upstream discharge). Such mortality does not invalidate the test. Instead, analysis of toxicity trends resulting from the relative toxicity tests can be used to assess the effluent's toxicity in relation to other sources and ambient receiving water conditions. However, a control dilution water with no toxicity must be used for quality assurance and determination of absolute toxicity of the effluent.

3) Conduct ambient toxicity tests to (a) determine whether or not the effluent has a measurable toxicity after mixing, (b) measure persistence of toxicity from all sources contributing to receiving water toxicity, and (c) determine combined toxicity resulting from the mixing of multiple, point, and nonpoint sources of toxicity. See Appendix C for a discussion of ambient toxicity testing procedures.

The ambient testing can be required of each discharger and conducted during low-flow or worst-case design periods.

Frequency for Ambient Testing

All testing should be conducted simultaneously by each discharger, if possible. At a minimum, the tests should be conducted concurrently starting within a short time period (1 to 2 days). Repeated ambient toxicity analyses will be desirable when variable effluents are involved. Effluent toxicity data showing variability can be used to assess what frequency will be most applicable. The level of repetition for variability analysis should be similar to that used in effluent variability analyses.

Other Considerations

Dye studies of effluent dispersion for rivers, lakes, reservoirs, and estuaries are strongly recommended. This allows analysis of effluent concentration at the selected sampling stations above and below the discharge points.

The procedures suggested in this multiple source section are based on actual multiple source site investigations conducted under the Complex Effluent Toxicity Testing Program. Site reports from that study can be used to obtain further description of the toxicity testing procedures used to analyze multiple source toxic impact [1, 2].

3.3.5 Ambient Toxicity Testing

Ambient toxicity testing also is useful in screening receiving water bodies for existing toxic conditions. The procedure described in Appendix C uses short-term chronic toxicity tests to measure the toxicity of samples of receiving water taken above, at, and below outfalls. It can be used in freshwater, marine, and estuarine systems. The procedure must be conducted during an appropriate low-flow or worst-case design period.

The utility of the ambient toxicity screening approach is that actual receiving water toxicity is directly measured. No extrapolation from exposure or ACR is needed. Further, impact from multiple source discharge situations, which may not be apparent from individual discharger data, is identified. Finally, the technique can provide an assessment of the persistence of effluent toxicity.

3.3.6 Special Considerations for Discharges to Marine and Estuarine Environments

Special problems are encountered when assessing and controlling impacts of toxic pollutants discharged to marine and estuarine waterbodies. These special problems include the following:

- Determining the physical characteristics of estuaries and the complex mixing and effluent dilution situations for RWCs of effluents.
- Generating toxicity data on nonsaline effluents that discharge to brackish or saline waters and establishing cause-effect relationships on that basis.
- Assessing exposure and controlling impacts from persistent toxicants accumulating in fish and shellfish tissues and in sediments. These factors are particularly important in estuaries and near coastal waters because of high use of estuaries as breeding and fishing areas for important commercial seafood supplies and recreational fishing, and because many estuaries and near coastal waters act as sinks for pollutants that accumulate in sediments.

Where these special problems are encountered, additional information may need to be gathered to better quantify dilution, to determine metals partitioning, and to identify potential interferences in whole effluent toxicity tests.

To characterize the type of whole effluent toxicity that is most relevant for a particular discharge to marine and estuarine waters, the following questions should be considered [5]:

- What is the salinity of the receiving water, and is this important in terms of the State standards?
- What is the appropriate test organism to require for toxicity testing under differing salinity conditions?

The answers to these questions will enable the permitting authority to determine what type of toxicity testing is most suitable for effluent characterization and whole effluent toxicity control.

For most marine and estuarine discharges the choice of test species and dilution water should be made based on the characteristics of the receiving water at the critical conditions for flow, mixing, and salinity. Foremost in this determination should be the salinity of the receiving water and, to a lesser extent, the salinity of the effluent itself.

The primary objective of whole effluent toxicity tests is to identify sources of toxicity that can potentially cause an excursion of a State's narrative or numeric water quality criteria. For this reason, the toxicity tests should reflect the natural conditions of the receiving water so to be able to measure any effluent characteristic that could contribute to ambient toxicity. The marine toxicity test methods identify 1,000 mg/l as the point at which salinity begins to exert an effect on freshwater species. **As a general rule, EPA recommends that freshwater organisms be used when the receiving water salinity is less than 1,000 mg/l, and that marine organisms be used when the receiving water salinity equals or exceeds 1,000 mg/l.**

Saline Effluent Discharges to Saltwater

The dissolved salts in the effluent are pollutants. These salts may or may not be the same as those present in the receiving water. Also, the proportion of dissolved salts in the effluent may be different from that of the salts in the receiving water. In this case, the toxicity test needs to be able to determine if these salts contribute to ambient toxicity. For this reason, marine organisms are needed.

Saline Effluent Discharged to Freshwater

In this case, the dissolved salts in the effluent is a pollutant that does not exist in the receiving water. The toxicity test needs to determine whether the dissolved salts can be one of the toxicants that contribute to ambient toxicity. For this reason, freshwater organisms are needed.

Freshwater Effluent Discharged to Saltwater

In this instance, the lack of dissolved salts in the effluent can cause an apparent toxic effect to the marine organisms in the toxicity test. However, in contrast to the instances presented above, the toxicity test does not need to be able to measure this effect because a lack of salts is not a pollutant. The marine toxicity test methods account for this by requiring that the salinity of the effluent be adjusted to approximate the salinity of the receiving water. As an alternative to using a marine organism, a freshwater organism can be used if the test is being conducted only on a 100-percent effluent sample and if State water quality standards do not require that a marine organism be used.

3.3.7 Using a Chemical-specific Limit to Control Toxicity

EPA regulations at 40 *CFR* 122.44(d)(1)(v) provide that limits on whole effluent toxicity are not necessary where the permitting authority demonstrates in the fact sheet or statement of basis of the NPDES permit that chemical-specific limits for the effluent are sufficient to attain and maintain applicable numeric and narrative State water quality criteria. To make this demonstration that chemical-specific limits are sufficient, additional effluent information will be needed. **EPA recommends that the discharger conduct a toxicity identification evaluation to identify the causative agent(s) in the effluent.** Where the permitting authority determines that the demonstration required by 40 *CFR* 122.44(d)(1)(v) has been made, limits on whole effluent toxicity

need not be imposed. Effluent limits on the controlling chemical with concurrent whole effluent monitoring will be sufficient. Where subsequent whole effluent toxicity testing reveals the presence of toxicity in the effluent, the above process will need to be repeated, or alternatively a whole effluent toxicity limit will be needed. If continued toxicity testing shows that additional chemical-specific effluent limits are insufficient to control whole effluent toxicity, then toxicity limits may be the only practical way to control toxicity.

3.3.8 Effluent Characterization for Specific Chemicals

The previous section discussed effluent characterization for whole effluent toxicity. This section will describe EPA's recommendations for data generation to determine whether or not permit limits are needed to control specific chemical pollutants in effluents. While many of the same principles apply when developing chemical-specific limits, there are some differences based upon regulatory and analytical considerations.

Characterization of impacts due to specific chemicals do not require a determination of the type of testing as is required for whole effluent toxicity because there is generally only one type of test for specific chemicals. However, there are some antecedent steps that are unique to effluent characterization for specific chemicals: determination of the chemicals of concern and determination of acceptable ambient levels (RAC, CMC, or CCC) for these pollutants.

Steps for Chemical-specific Effluent Characterization Process

Figure 3-5 illustrates EPA's recommendations for determining whether or not permit limits need to be developed according to an evaluation of a limited data set. The following discussion corresponds to the various activities shown in Figure 3-5. (Refer to the human health discussion in Section 3.3.9 for additional details on procedures to characterize the bioconcentration potential of effluents.)

Step 1: Identify the Pollutants of Concern

This process should begin with an examination of existing data to determine the presence of specific toxicants for which criteria, standards, or other toxicity data are available. Sources of data include the following:

- Permit application forms, DMRs, permit compliance systems (PCS), and permit files
- Pretreatment industrial surveys
- STORET for ambient monitoring data
- SARA Title III Toxic Chemical Release Inventory
- Industrial effluent guidelines development documents
- The Treatability Manual [6]
- Effluent bioconcentration assessment (see Section 3.3.9).

Data on specific chemicals that are typically submitted with NPDES application forms will consist of a limited number of analytical test results for many of the reported parameters. Where the regulatory authority has reason to believe that additional data for key parameters of concern are needed in order to adequately characterize the effluent, this information should be requested as a part of the application or, in some cases, through the use of Section 308 letters. **It is recommended that 8 to 12 samples be analyzed for key parameters of concern.** In some cases, special analytical protocols will need to be specified in order to gather all appropriate information.

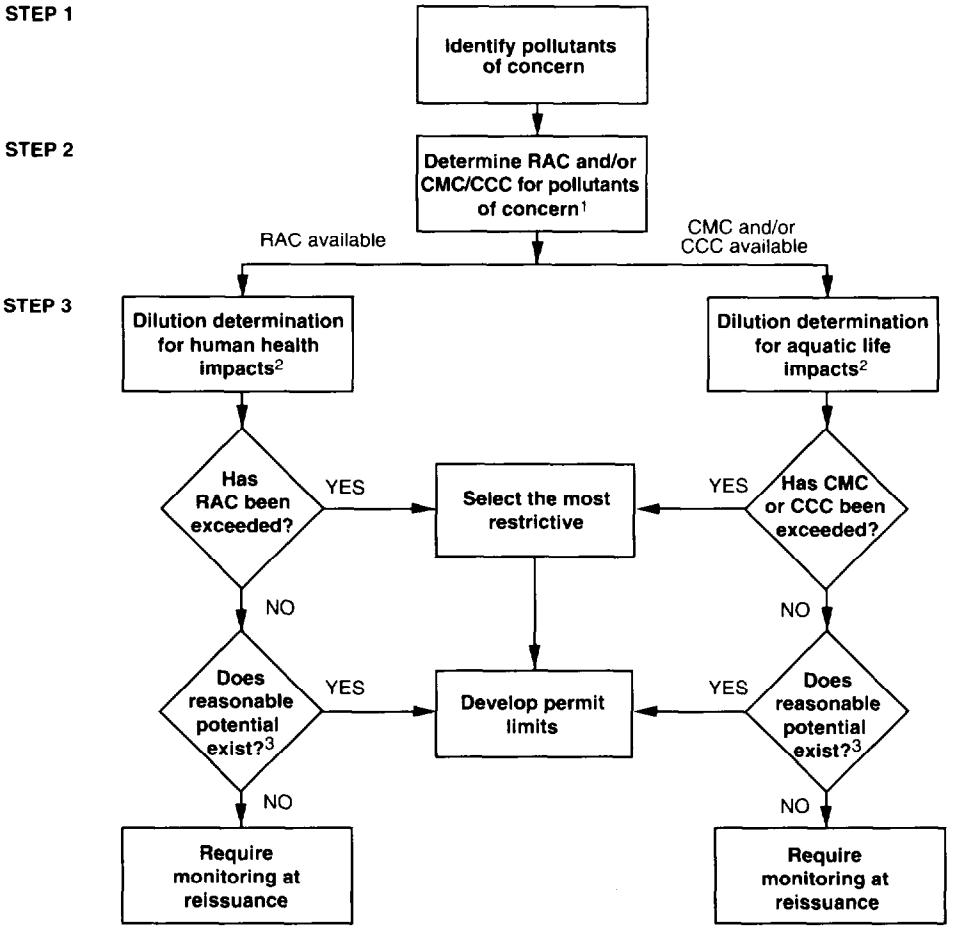
Step 2: Determine the Basis for Establishing RACs, CMCs, and CCCs for the Pollutants of Concern

The second step is to identify the appropriate water quality standard, including designated or existing use, and criteria for use. Ideally, the State water quality standards include aquatic life and human health criteria for the pollutants of concern. If a State does not have a numeric water quality criterion for the pollutant of concern, then one of three options for using the narrative criterion may be used (40 *CFR* 122.44(d)(1)(vi)) to determine whether a discharge causes, has the reasonable potential to cause, or contributes to an excursion above a narrative criteria because of an individual pollutant. Although the provisions of 40 *CFR* 122.44(d)(1)(vi) are presented in the regulation in the context of permit limit development, these same considerations should be applied in characterizing effluents in order to determine whether limits are necessary. The options available are as follows:

- **Option A** allows the regulatory authority to establish limits using a "calculated numeric water quality criterion" that the regulatory authority demonstrates will attain and maintain applicable narrative water quality criteria and fully protect the designated use. This option allows the regulatory authority to use any criterion that protects aquatic life and human health. This option also allows the use of site-specific factors, including local human consumption rates of aquatic foods, the State's determination of an appropriate risk level, and any other current data that may be available.

- **Option B** allows the regulatory authority to establish effluent limits using EPA's Water Quality Criteria guidance documents, if EPA has published a criteria document for the pollutant supplemented where necessary by other relevant information. As discussed earlier, EPA criteria documents provide a comprehensive summary of available data on the effects of a pollutant.

- **Option C** may be used to develop limits for a pollutant of concern based on an indicator parameter under limited circumstances. An example of an indicator parameter is total toxic organics (TTO); effluent limits on TTO are useful where an effluent contains organic compounds. However, use of this option must be justified to show that controls on one pollutant control one or more other pollutants to a level that will attain and maintain applicable State narrative water quality criteria and will protect aquatic life and human health (see 40 *CFR* 122.44(d)(1)(vi)(C)). Use of this option is restricted by regulation to those instances where it can be demonstrated that controls on indicator pollutants serve to control the toxicant of concern. Using Option A or Option B is a more direct and perhaps more defensible approach.



Notes:

[1] RAC and/or CMC/CCC: Use State numeric criterion or interpret State narrative criterion using one of three options specified under 40 *CFR* 122.44(d).

[2] Dilution determination: Perform for critical flow and for any applicable mixing zones for aquatic life and human health protection procedures, respectively.

[3] Reasonable potential: Use procedures in Boxes 3-2 and 3-4.

Figure 3-5. Effluent Characterization for Specific Chemicals

Step 3: Dilution Determination

The third step is to calculate the effluent dilution at the edge of the mixing zone. The pertinent factors for consideration here are the same as were previously presented for whole effluent toxicity with one difference: there are two levels of dilution analysis for chemical data. The first level is to use simple fate models based on a dilution analysis and comparison with the RAC, CMC, or CCC. The second level of analysis is to use more complex fate models, including dynamic models to estimate persistence, and may be applied to lakes, rivers, estuaries, and coastal systems using a desktop calculator or microcomputer. EPA has supported development of a second level of analysis that estimates point source wasteload allocations and nonpoint source allocations and predicts the resulting pollutant concentrations in receiving waters [7].

Step 4: Decision Criteria for Permit Limit Development

After this dilution analysis has been performed, the projected RWC is compared to the RAC, CMC, or CCC (either the State numeric criteria or an interpretation of the narrative criteria as described earlier). Whereas analysis of aquatic impacts should include evaluations with respect to both the CCC and the CMC, analysis of human health impacts will only involve comparisons with the RAC. The four possible outcomes discussed above in the triggers for permit limit development discussion in Section 3.3.3 also apply here:

- Excursion above the RAC, CMC, or CCC
- Reasonable potential for excursion above the RAC, CMC, or CCC

- No reasonable potential for excursion above the RAC, CMC, CCC

- Inadequate information.

If these evaluations project excursions or the reasonable potential to cause or contribute to an excursion above the RAC, CMC, or CCC, then a permit limit is required (40 *CFR* 122.44(d)(1)(iii)). The statistical approach shown in Box 3-2 or an analogous approach developed by a regulatory authority can be used to determine the reasonable potential. Effluents that are shown not to cause or that have a reasonable potential to cause or contribute to an excursion above an RAC, CMC, or CCC should be reevaluated at permit reissuance.

Where chemical-specific test results do not show a reasonable potential but indicate a basis for concern after consideration of the other factors discussed in Section 3.2, or if there were inadequate information to make a decision, the permit should contain chemical testing requirements and a reopener clause. This clause would require reopening of the permit and establishment of a limit based upon any test results that show effluent toxicity at levels that cause or have a reasonable potential to cause or contribute to an excursion above the RAC, CCC, or CMC.

3.3.9 Effluent Characterization for Bioconcentratable Pollutants

The previous section discussed how to characterize effects of specific chemicals, including those that may threaten human health, to determine whether or not a discharge causes, has the reasonable potential to cause, or contributes to excursions above an water quality criterion. The primary disadvantage of this approach is that it does not identify all effluent chemicals of potential concern for human health. To help address this gap, EPA is developing a procedure for identifying pollutants with the propensity to bioconcentrate in fish tissue. This procedure is presently in draft form and should not be used for establishing NPDES permit limits until EPA releases the final document on the procedure. This section describes the outline of this procedure.

The overall approach illustrated in Figure 3-6 is a seven-step procedure that starts with collecting samples and ends with developing permit effluent limits. The effluent characterization step unique to this approach lies in Step 3. There are two alternatives under this step: fish tissue residue and effluent assessment. An analytical chemistry laboratory with residue chemistry and gas chromatograph/mass spectometer (GC/MS) capability is needed to conduct the analytical methods for both alternatives. A summary of the alternatives follows:

- Tissue Residue Alternative: This alternative measures the concentrations of organic bioconcentratable chemicals in tissue samples of indigenous organisms from the receiving water. This analysis involves the collection of fish or shellfish samples, the extraction of the organic chemicals from the tissue and the analysis of these extracts with GC/MS to identify and quantify the bioconcentratable contaminants. The procedure provides recommendations to sort the results of this screening analysis in order to determine which of the contaminants pose a hazard and require regulatory action. The approach recommends that the identity of those contaminants then be confirmed prior to taking subsequent action.

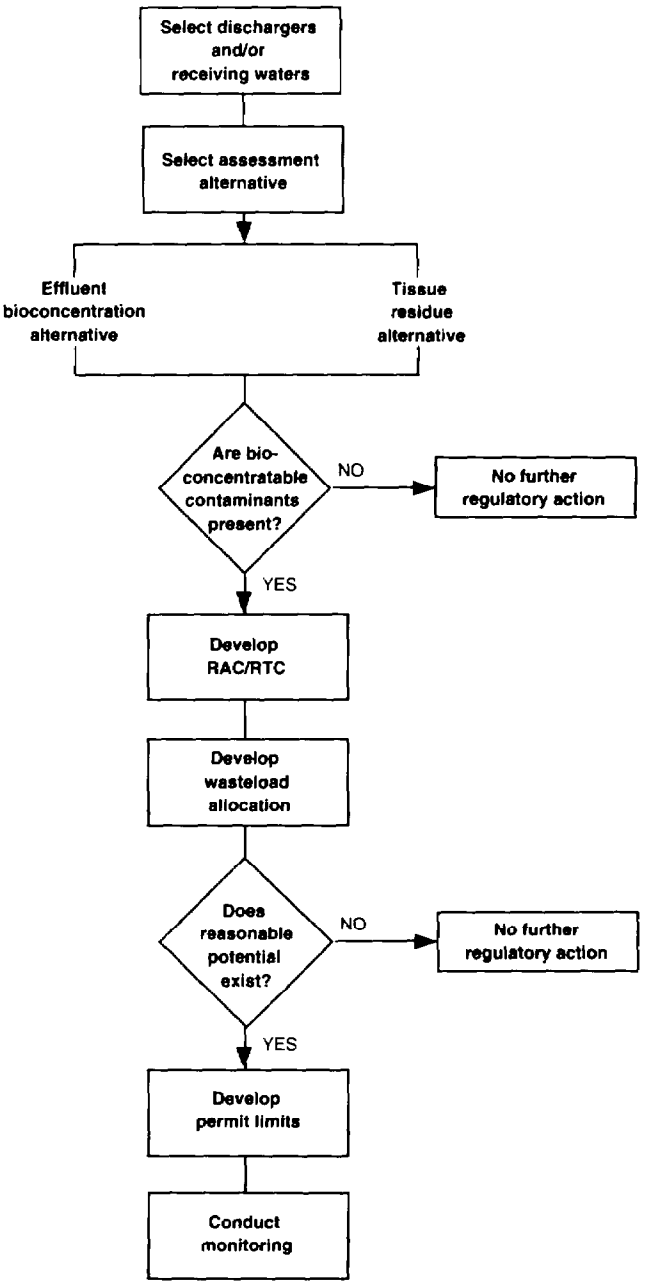


Figure 3-6. Procedure for Assessment and Control of Bioconcentratable Contaminants in Surface Waters

- Effluent Alternative: This alternative measures the concentrations of organic bioconcentratable chemicals in effluent samples from point source dischargers. This analysis involves the collection of effluent samples, the extraction of the organic chemicals from the effluent sample, and the separation of the chemicals that have characteristics known to result in bioconcentration from the other chemical components of the effluent sample. This separation is achieved by way of an analytical chemistry methodology called high-

pressure liquid chromotography (HPLC). The HPLC also separates (fractionates) an effluent sample into three subsamples or "fractions." These three fractions contain chemicals with increasing potential to bioconcentrate, with the third fraction containing those chemicals with the highest bioconcentration rates. Following HPLC fractionation, each fraction is then analyzed with GC/MS to identify and quantify the bioconcentratable contaminants. The effluent procedure also provides recommendations to sort the results of the initial screening analysis to determine which of the contaminants pose a hazard and require subsequent regulatory action. The approach then recommends that the identity of those contaminants then be confirmed prior to taking further regulatory action.

While both of the assessment alternatives described above may be used for a given discharger, generally one of these alternatives may be preferred by the regulatory authority. The regulatory authority would select the assessment approach based on the available site- and facility-specific information and the objectives of the application.

Although the approach provides a means to identify chemicals that can bioconcentrate, it does not identify all bioconcentratable chemicals. Chemicals that bioconcentrate include many organic compounds, and a small number of metals (e.g., mercury and selenium) and organometals (e.g., tributyltin). The new approach is limited to nonpolar organic chemicals that produce measurable chemical residues in aquatic organisms or that have log octanol-water partition coefficients greater than 3.5.

3.3.10 Analytical Considerations for Chemicals

Analysis of discharges for toxic substances requires special quality control procedures beyond those necessary for conventional parameters. Toxicants can occur in trace concentrations and are frequently volatile or otherwise unstable. An EPA publication entitled, *Test Methods—Technical Additions to Methods for Chemical Analysis of Water and Wastes* [8], contains sampling and handling procedures recommended by EPA for a number of toxic and conventional parameters. Additional methods for analyses for toxicants are described in Standard Methods of Water and Wastewater Analyses (ASTM, 17th edition, 1989, or most recent edition) and 40 CFR Part 136. Chapter 5 discusses detection limits and sampling requirements.

CHAPTER 3
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4. EXPOSURE AND WASTELOAD ALLOCATION

4.1 INTRODUCTION

At this point in the toxics control process, a water quality problem has been identified. Screening analyses may have been done to assess the extent of toxicity, or a wasteload allocation (WLA) based on an existing total maximum daily load (TMDL) may already have been established. A TMDL is the sum of the individual WLAs for point sources and load allocations (LAs) for nonpoint sources of pollution and natural background sources, tributaries, or adjacent segments. WLAs represent that portion of a TMDL that is established to limit the amount of pollutants from existing and future point sources so that surface water quality is protected at all flow conditions.

The TMDL process uses water quality analyses to predict water quality conditions and pollutant concentrations. Limits on wastewater pollutant loads are set and nonpoint source allocations are established so that predicted receiving water concentrations do not exceed water quality criteria. TMDLs and WLAs/LAs should be established at levels necessary to attain and maintain the applicable narrative and numerical water quality standards, with seasonal variations and a margin of safety that takes into account any lack of knowledge concerning the relationship between point and nonpoint source loadings and water quality. Determination of WLAs/LAs and TMDLs should take into account critical conditions for stream flow, loading, and water quality parameters. Conditions that will protect the receiving water have been determined from State numeric or narrative water quality criteria.

This chapter is divided into sections that explain the steps that precede establishment of a WLA and then the methods and tools (models) that can be used to determine the WLA. Section 4.2 briefly discusses TMDLs and how they relate to waters identified as requiring a water quality-based approach for toxics control. The section also discusses different WLA schemes. Sections 4.3 and 4.4 discuss mixing zones, areas described as allocated impact zones where acute and chronic water quality criteria may be exceeded. Section 4.3 provides background information on mixing zones and discusses EPA's mixing zone policy and how this policy affects the allowable toxic load that can be discharged from a point source. State mixing zone dimensions and the determination of mixing zone boundaries are also discussed.

Section 4.4 discusses mixing zone analyses for situations in which the discharge does not mix completely with the receiving water within a short distance. Included in Section 4.4 are discussions of outfall designs that maximize initial dilution in the mixing zone, critical design periods for mixing zone analyses, and methods to analyze and model near-field and far-field mixing.

Section 4.5 discusses the calculations of the WLA and LA and the types of EPA-recommended mathematical models available to determine WLAs in completely mixed situations for both aquatic life and human health. The WLA models listed in Section 4.5 can be used to predict ambient concentrations and to calculate the effluent quality required to meet the criteria and protect designated and existing uses of the receiving water. The data requirements of each of these models are also described so that the effluent characterization procedures described in Chapter 3 can be designed to support the specific types of WLA modeling selected by the regulator. Section 4.6 discusses human health considerations and how to determine WLAs for human health toxicants.

EPA is currently working on methods to develop sediment criteria. Once developed, point source discharges could be further limited to prevent accumulation of pollutants in the bed sediment; such accumulation impairs beneficial uses. Although the criteria are not yet available for this document, they will be addressed in future documents. In the meantime, some of the models discussed in Section 4.5 are capable of simulating interactions between the water column and sediment and between toxic transport and transformation in the sediment. EPA is encouraging the States to consider the role of sediments in WLA.

4.2 TOTAL MAXIMUM DAILY LOADS AND WASTELOAD ALLOCATIONS

4.2.1 Total Maximum Daily Loads

The Federal Clean Water Act (CWA), under Section 303(d), requires the establishment of TMDLs for "water quality limited" stream segments. In such segments, water quality does not meet applicable water quality standards and/or is not expected to meet applicable water quality standards even after the application of the technology-based effluent limitations. A TMDL includes a determination of the amount of a pollutant, or property of a pollutant, from point, nonpoint, and natural background sources, including a margin of safety, that may be discharged to a water quality-limited waterbody. Any loading above this loading capacity risks violating water quality standards. TMDLs can be expressed in terms of chemical mass per unit of time, by toxicity, or by other appropriate measures. Permits should be issued based on TMDLs where available.

The establishment of a TMDL for a particular waterbody is dependent on the location of point sources, available dilution, water quality standards, nonpoint source contributions, background conditions, and instream pollutant reactions and effluent toxicity. All of these factors can affect the allowable mass of the pollutant in the waterbody. Thus, two issues must be determined in conjunction with the establishment of the TMDL: (1) the definition of upstream and downstream boundaries of the waterbody for which the TMDL is being determined, and (2) the definition of critical conditions. For the following discussion, the waterbody boundaries are delineated as the portion of the waterbody be-

tween the pollutant source (whether point source or nonpoint source) that is farthest upstream and the downstream point at which water quality has recovered to the background quality found above the pollutant source that is farthest upstream. The delineation of critical conditions for stream flow, loading, and water quality parameters may be specific to the type of waterbody and is discussed in Section 4.4.

TMDLs are established based on water quality criteria pertinent to the designated and existing uses for the waterbody in question. TMDLs are traditionally calculated using State water quality standards as applied to a specific waterbody. Such a fitting of the TMDL to desired water quality criteria requires information concerning the distribution of loadings within the waterbody, namely, the locations and relative contributions of pollutant-specific loadings from point, nonpoint, and background sources during all flow conditions (40 *CFR* 130.2(f)). Low-flow TMDLs, by themselves, will not be adequate in situations where nonpoint source loadings (LAs) during high or intermediate flow conditions cause excursions above water quality standards (40 *CFR* 130.2(f)).

The loading capacity of TMDLs have been determined in many ways, but the most common method is to find the pollutant loading that will attain and maintain applicable water quality criteria. For example, in the Tualatin River Basin in Oregon, loading capacity was determined by multiplying stream flow in critical flow periods by the pollutant water quality standard [1]. Another method of determining a loading capacity is by quantifying instream toxicity. This method was used in developing a TMDL for the Amelia River in Florida [2].

The allowable TMDL is defined as the sum of the individual WLAs and LAs; a margin of safety can be included with the two types of allocations to ensure that allocated loads, regardless of source, would not produce an excursion above water quality standards. The WLAs are those portions of the TMDL assigned to point sources; the LAs are those portions of the TMDL assigned to the sum of all nonpoint sources and background sources (40 *CFR* 130.2(f)). The background sources represent loadings to the specified waterbody or stream segment that come from sources outside the defined segment. For example, loadings from regions upstream of the segment and estimated atmospheric deposition of the pollutant would constitute background sources. Sediments that are highly contaminated from upstream discharges or historical discharges might also act as a source of toxicants and contribute to the background levels; these sediments also may be part of the nonpoint sources.

The TMDL represents a mass loading that may occur over a given time period to attain and maintain water quality standards. As a result, the design flows under which the TMDL is determined can significantly alter its value. This phenomenon results in a somewhat unusual dichotomy. The design flows for aquatic life protection most applicable to point source loadings (WLAs) usually involve low-flow events (e.g., 7Q10) because the volumes associated with the point sources generally do not decrease with decreased stream flow. As a result, the highest concentrations associated with specific point source loads would be expected under low flow conditions. Conversely, elevated nonpoint source pollutant loadings (i.e., urban, agricultural) generally correspond to storm events. In fact, agricultural and urban runoff are often minimal or nonexistent in the absence of precipitation (i.e., nonexistent under low-flow drought conditions).

The TMDL is a composite of the allowable loads associated with point sources and nonpoint sources within the defined boundaries of the waterbody segment and the background loadings to that segment from upstream and from in-place sediments. Therefore, the TMDL should be evaluated under conditions that reflect worst-case (critical) conditions for both point and nonpoint source loadings (i.e., low-flow drought and high flow conditions). Determination of the TMDL under these two scenarios would identify the lower of the two loading capacities of the waterbody. This lower capacity is necessary to protect the waterbody in question.

In the case of design flows for human health protection, the harmonic mean flow is recommended as the basis for TMDLs for carcinogens. Design flows for human health protection should consider worst-case conditions for both point and nonpoint source loadings under this flow condition (see Section 4.6).

In many cases, LAs for nonpoint sources are difficult to assess because the information needed to describe the runoff associated with the high-flow storm events does not exist. This lack of information is due to the high variability of the events. Because of the importance of estimating the nonpoint contributions to the waterbody, site-specific models may be required to estimate nonpoint source loadings. Even then, detailed models are difficult to calibrate with accuracy without intensive monitoring studies, and simplistic correlations between loadings and rainfall can be, by their statistical nature, unreliable for estimating low-frequency events (e.g., worst 10-year storm). The uncertainties associated with nonpoint source loadings and background sources require that the TMDL be determined with a sufficient margin of safety to allow for significant variability in nonpoint source loadings.

CWA Section 303(d) and EPA regulations (40 *CFR* Parts 35 and 130, January 11, 1985) require that TMDLs contain a margin of safety "which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality." The margin of safety is to take into account any uncertainties related to development of the water quality-based control, including any uncertainties in pollutant loadings, ambient conditions, and the model analysis. The size of the required margin of safety can, of course, be reduced by collecting additional information, which reduces the amount of uncertainty. The margin of safety can be provided for in the TMDL process by one of the following:

- Reserving a portion of the loading capacity to a separate margin of safety.
- Including a margin of safety within the individual WLAs for point sources and within the LAs for nonpoint sources and background sources.

Most TMDLs are developed using the second approach, most often through the use of conservative design conditions.

In addition, all WLAs, LAs, and TMDLs must meet the State antidegradation provisions developed prusuant to the Water Quality Standards Regulation (Section 131.12 of 40 *CFR* Part 131,

November 8, 1983). This regulation establishes explicit procedures that must be followed prior to lowering existing water quality to a level that still supports the Section 101(a)(2) "fishable/swimmable" goal of the Act. WLAs, LAs, and TMDLs that allow such a decline in water quality cannot be established unless the applicable public participation and intergovern-mental review requirements of the antigradation provisions have been met and all existing uses are fully maintained and protected.

4.2.2 Wasteload Allocation Schemes

WLAs for water quality-based toxics permits must be set in accordance with EPA regulations [3, 4]. EPA has developed a number of WLA guidance documents to assist regulatory authorities in developing TMDLs and WLAs. The EPA Office of Water Regulations and Standards, Assessment and Watershed Protection Division, maintains the latest listing of all WLA guidance documents. Toxic WLA guidance documents are currently available for rivers and streams [5], lakes and reservoirs [6], and estuaries [7]. Guidance for the determination of critical design conditions for steady-state modeling of rivers and streams also is available [8].

Table 4-1 lists 19 allocation schemes that may be used by the States to develop WLAs. This is not intended to be a complete list of approaches; regulatory authorities may use any reasonable allocation scheme that meets the antidegradation provisions and other requirements of State water quality standards [3].

The most commonly used allocation methods have been equal percent removal, equal effluent concentrations, and a hybrid method. The equal percent removal approach can be applied in two ways: the overall removal efficiencies of each pollutant source must be equal, or the incremental removal efficiencies must be equal. The equal effluent concentration approach also can be applied in two acceptable ways—equal final concentrations or equal incremental concentration reductions. This method is similar to the equal percent removal method if influent concentrations at all sources are approximately the same. However, if one point source has substantially higher influent levels, requiring equal effluent concentrations will result in higher overall treatment levels for that source than the equal percent removal approach.

The final commonly used method of allocating wasteloads is a hybrid method in which the criteria for waste reduction may not be the same for each point source. One facility may be allowed to operate unchanged, while another may be required to provide the entire load reduction. More often, a proportionality rule that requires the percent removal to be proportional to the input loading can be assigned. In these cases, larger sources would be required to achieve higher overall removals.

4.3 INCOMPLETELY MIXED, DISCHARGE RECEIVING WATER SITUATIONS

Mixing zones are areas where an effluent discharge undergoes initial dilution and are extended to cover the secondary mixing in the ambient waterbody. A mixing zone is an allocated impact zone where acute and chronic water quality criteria can be exceeded as long as a number of protections are maintained, including freedom from the following:

- Materials in concentrations that settle to form objectionable deposits
- Floating debris, oil, scum, and other matter in concentrations that form nuisances

Table 4-1. Wasteload Allocation Methods [9]

1. Equal percent removal (equal percent treatment)
2. Equal effluent concentrations
3. Equal total mass discharge per day
4. Equal mass discharge per capita per day
5. Equal reduction of raw load (pounds per day)
6. Equal ambient mean annual quality (mg/l)
7. Equal cost per pound of pollutant removed
8. Equal treatment cost per unit of production
9. Equal mass discharged per unit of raw material used
10. Equal mass discharged per unit of production

11a. Percent removal proportional to raw load per day

11b. Larger facilities to achieve higher removal rates

12. Percent removal proportional to community effective income

13a. Effluent charges (dollars per pound, etc.)

13b. Effluent charge above some load limit

14. Seasonal limits based on cost-effectiveness analysis
15. Minimum total treatment cost
16. Best availability technology (BAT) (industry) plus some level for municipal inputs
17. Assimilative capacity divided to require an "equal effort among all dischargers"

18a. Municipal: treatment level proportional to plant size

18b. Industrial: equal percent between best practicable technology (BPT) and BAT, i.e., Allowable wasteload allocation:

$$(WLA) = BPT - \frac{X}{100}(BPT-BAT)$$

19. Industrial discharges given different treatment levels for different stream flows and seasons. For example, a plant might not be allowed to discharge when stream flow is below a certain value, but below another value, the plant would be required to use a higher level of treatment than BPT. Finally, when stream flow is above an upper value, the plant would be required to treat to a level comparable to BPT.

- Substances in concentrations that produce objectionable color, odor, taste, or turbidity

- Substances in concentrations that produce undesirable aquatic life or result in a dominance of nuisance species.

Acutely toxic conditions are defined as those lethal to aquatic organisms that may pass through the mixing zone. As discussed in Chapter 2, the underlying assumption for allowing a mixing zone is that a small area of concentrations in excess of acute and chronic criteria, but below acutely toxic releases, can exist without causing adverse effects to the overall waterbody. The State regulatory agency can decide to allow or deny a mixing zone on a site-specific basis. For a mixing zone to be permitted, the discharger should prove to the State regulatory agency that all State requirements for a mixing zone are met.

When wastewater is discharged into a waterbody, its transport may be divided into two stages with distinctive mixing characteristics. Mixing and dilution in the first stage are determined by the initial momentum and buoyancy of the discharge. This initial contact with the receiving water is where the concentration of the effluent will be its greatest in the water column. The design of the discharge outfall should provide ample momentum to dilute the concentrations in the immediate contact area as quickly as possible.

The second stage of mixing covers a more extensive area in which the effect of initial momentum and buoyancy is diminished and the waste is mixed primarily by ambient turbulence. In large rivers or estuaries, this second-stage mixing area may extend for miles before uniformly mixed conditions are attained. In some instances, such as larger lakes or coastal bays, completely mixed conditions are never reached in the waterbody. The general definition for a completely mixed condition is when no measurable difference in the concentration of the pollutant (e.g., does not vary by more than 5 percent) exists across any transect of the waterbody.

This section provides background information on the policy of mixing zones and the means to characterize them for use in WLAs (Section 4.5). The first subsection discusses the concerns that must be addressed when the boundaries and restrictions of a mixing zone are determined. The second subsection discusses the guidelines for preventing lethal conditions in the mixing zone.

4.3.1 Determination of Mixing Zone Boundaries

Allowable mixing zone characteristics should be established to ensure the following:

- Mixing zones do not impair the integrity of the waterbody as a whole.

- There is no lethality to organisms passing through the mixing zone.

- There are no significant health risks, considering likely pathways of exposure (see Section 2.2.2).

The Water Quality Criteria—1972 [10] recommends that mixing zone characteristics be defined on a case-by-case basis after it has been determined that the assimilative capacity of the receiving system can safely accommodate the discharge. This assessment should take into consideration the physical, chemical, and biological characteristics of the discharge and the receiving system; the life history and behavior of organisms in the receiving system; and the desired uses of the waters. Nearly all States require such an analysis before they allow a mixing zone [11]. Further, mixing zones should not be permitted where they may endanger critical areas (e.g., drinking water supplies, recreational areas, breeding grounds, areas with sensitive biota).

EPA has developed a holistic approach to determine whether a mixing zone is tolerable [12]. The method considers all the impacts to the waterbody and all the impacts that the drop in water quality will have on the surrounding ecosystem and waterbody uses. It is a multistep data collection and analysis procedure that is particularly sensitive to overlapping mixing zones. It includes the identification of all upstream and downstream waterbodies and the ecological and cultural data pertaining to them; the collection of data on all present and future discharges to the waterbody; the assessment of relative environmental value and level of protection needed for the waterbody; and, finally, the allocation of environmental impact for a discharge applicant. Because of the difficulty in collecting the data necessary for this procedure and the general lack of agreement concerning relative values, this method will be difficult to implement in full. However, the method does serve as a guide on how to proceed in allocating a mixing zone.

Most States allow mixing zones as a policy issue, but provide spatial dimensions to limit the areal extent of the mixing zones. The mixing zones are then allowed (or not allowed) after case-by-case determinations. State regulations dealing with streams and rivers generally limit mixing zone widths, cross-sectional areas, and flow volumes and allow lengths to be determined on a case-by-case basis. For lakes, estuaries, and coastal waters, dimensions are usually specified by surface area, width, cross-sectional area, and volume.

Where a mixing zone is allowed, water quality standards are met at the edge of that regulatory mixing zone during design flow conditions and generally, (1) provide a continuous zone of passage that meets water quality criteria for free-swimming and drifting organisms and (2) prevent impairment of critical resource areas. Individual State mixing zone dimensions are designed to limit the impact of a mixing zone on the waterbody. Furthermore, EPA's review of State WLAs should evaluate whether assumptions of complete or incomplete mixing are appropriate based on available data.

In river systems, reservoirs, lakes, estuaries, and coastal waters, zones of passage are defined as continuous water routes of such volume, area, and quality as to allow passage of free-swimming and drifting organisms so that no significant effects are produced on their populations. Transport of a variety of organisms in river water and by tidal movements in estuaries is biologically important in a number of ways: food is carried to the sessile filter

feeders and other nonmobile organisms, spatial distribution of organisms and reinforcement of weakened populations are enhanced, and embryos and larvae of some fish species develop while drifting [11]. Anadromous and catadromous species must be able to reach suitable spawning areas. Their young (and in some cases the adults) must be assured a return route to their growing and living areas. Many species make migrations for spawning and other purposes. Barriers or blocks that prevent or interfere with these types of essential transport and movement can be created by water with inadequate chemical or physical quality.

As explained above, a State regulatory agency may decide to deny a mixing zone in a site-specific case. For example, denial should be considered when bioaccumulative pollutants are in the discharge. The potential for a pollutant to bioaccumulate in living organisms is measured by (1) the bioconcentration factor (BCF), which is chemical-specific and describes the degree to which an organism or tissue can acquire a higher contaminant concentration than its environment (e.g., surface water); (2) the duration of exposure; and (3) the concentration of the chemical of interest. While any BCF value greater than 1 indicates that bioaccumulation potential exists, bioaccumulation potential is generally not considered to be significant unless the BCF exceeds 100 or more. Thus, a chemical that is discharged to a receiving stream, resulting in low concentrations, and that has a low BCF value will not create a bioaccumulation hazard. Conversely, a chemical that is discharged to a receiving stream, resulting in a low concentration but having a high BCF value, may cause in a bioaccumulation hazard. Also, some chemicals of relatively low toxicity, such as zinc, will bioconcentrate in fish without harmful effects resulting from human consumption.

Another example of when a regulator should consider prohibiting a mixing zone is in situations where an effluent is known to attract biota. In such cases, provision of a continuous zone of passage around the mixing area will not serve the purpose of protecting aquatic life. A review of the technical literature on avoidance/attraction behavior revealed that the majority of toxicants elicited an avoidance or neutral response at low concentrations [13]. However, some chemicals did elicit an attractive response, but the data were not sufficient to support any predictive methods. Temperature can be an attractive force and may counter an avoidance response to a pollutant, resulting in attraction to the toxicant discharge. Innate behavior such as migration may also supersede an avoidance response and cause fish to incur a significant exposure.

4.3.2 Minimizing the Size of Mixing Zones

Concentrations above the chronic criteria are likely to prevent sensitive taxa from taking up long-term residence in the mixing zone. In this regard, benthic organisms and territorial organisms are likely to be of greatest concern. The higher the concentrations occurring within an isopleth, the more taxa are likely to be excluded, thereby affecting the structure and function of the ecological community. It is thus important to minimize the overall size of the mixing zone and the size of elevated concentration isopleths within the mixing zone.

4.3.3 Prevention of Lethality to Passing Organisms

The *Water Quality Standards Handbook* [14] indicates that whether to establish a mixing zone policy is a matter of State discretion, but that any State policy allowing for mixing zones must be consistent with the CWA and is subject to approval of the Regional Administrator. The handbook provides additional discussion regarding the basis for a State mixing zone policy.

Lethality is a function of the magnitude of pollutant concentrations and the duration an organism is exposed to those concentrations. Requirements for wastewater plumes that tend to attract aquatic life should incorporate measures to reduce the toxicity (e.g., via pretreatment, dilution) to minimize lethality or any irreversible toxic effects on aquatic life.

EPA's water quality criteria provide guidance on the magnitude and duration of pollutant concentrations causing lethality. The criterion maximum concentration (CMC) is used as a means to prevent lethality or other acute effects. As explained in Appendix D, the CMC is a toxicity level and should not be confused with an LC_{50} level. The CMC is defined as one-half of the final acute value for specific toxicants and 0.3 acute toxic unit (TU_a) for effluent toxicity (see Chapter 2). The CMC describes the condition under which lethality will not occur if the duration of the exposure to the CMC level is less than 1 hour. The CMC for whole effluent toxicity is intended to prevent lethality or acute effects in the aquatic biota. The CMC for individual toxicants prevents acute effects in all but a small percentage of the tested species. Thus, the areal extent and concentration isopleths of the mixing zone must be such that the 1-hour average exposure of organisms passing through the mixing zone is less than the CMC. The organism must be able to pass through quickly or flee the high-concentration area. The objective of developing water quality recommendations for mixing zones is to provide time-exposure histories that produce negligible or no measurable effects on populations of critical species in the receiving system.

Lethality to passing organisms can be prevented in the mixing zone in one of four ways. The first method is to prohibit concentrations in excess of the CMC in the pipe itself, as measured directly at the end of the pipe. As an example, the CMC should be met in the pipe whenever a continuous discharge is made to an intermittent stream. The second approach is to require that the CMC be met within a very short distance from the outfall during chronic design-flow conditions for receiving waters (see Section 4.4.2).

If the second alternative is selected, hydraulic investigations and calculations indicate that the use of a high-velocity discharge with an initial velocity of 3 meters per second, or more, together with a mixing zone spatial limitation of 50 times the discharge length scale in any direction, should ensure that the CMC is met within a few minutes under practically all conditions. The discharge length scale is defined as the square root of the cross-sectional area of any discharge pipe.

A third alternative (applicable to any waterbody) is not to use a high-velocity discharge. Rather the discharger should provide

data to the State regulatory agency showing that the most restrictive of the following conditions are met for each outfall:

- The CMC should be met within 10 percent of the distance from the edge of the outfall structure to the edge of the regulatory mixing zone in any spatial direction.

- The CMC should be met within a distance of 50 times the discharge length scale in any spatial direction. In the case of a multiport diffuser, this requirement must be met for each port using the appropriate discharge length scale of that port. This restriction will ensure a dilution factor of at least 10 within this distance under all possible circumstances, including situations of severe bottom interaction, surface interaction, or lateral merging.

- The CMC should be met within a distance of five times the local water depth in any horizontal direction from any discharge outlet. The local water depth is defined as the natural water depth (existing prior to the installation of the discharge outlet) prevailing under mixing zone design conditions (e.g., low flow for rivers). This restriction will prevent locating the discharge in very shallow environments or very close to shore, which would result in significant surface and bottom concentrations.

A fourth alternative (applicable to any waterbody) is for the discharger to provide data to the State regulatory agency showing that a drifting organism would not be exposed to 1-hour average concentrations exceeding the CMC, or would not receive harmful exposure when evaluated by other valid toxicological analysis, as discussed in Section 2.2.2. Such data should be collected during environmental conditions that replicate critical conditions.

For the third and fourth alternatives, examples of such data include monitoring studies, except for those situations where collecting chemical samples to develop monitoring data would be impractical, such as at deep outfalls in oceans, lakes, or embayments. Other types of data could include field tracer studies using dye, current meters, other tracer materials, or detailed analytical calculations, such as modeling estimations of concentration or dilution isopleths.

The Water Quality Criteria—1972 [11] outlines a method, applicable to the fourth alternative, to determine whether a mixing zone is tolerable for a free-swimming or drifting organism. The method incorporates mortality rates (based on toxicity studies for the pollutant of concern and a representative organism) along with the concentration isopleths of the mixing zone and the length of time the organism may spend in each isopleth. The intent of the method is to prevent the actual time of exposure from exceeding the exposure time required to elicit an effect [10]:

$$\sum \frac{T(n)}{ET(X) \text{ at } C(n)} \leq 1$$

where T(n) is the exposure time an organism is in isopleth n, and ET(X) is the "effect time." That is, ET(X) is the exposure time required to produce an effect (including a delayed effect) in X percent of organisms exposed to a concentration equal to C(n), the concentration in isopleth n. ET(X) is experimentally determined; the effect is usually mortality. If the summation of ratios of exposure time to effect time is less than 1, then the percent effect will not occur.

4.3.4 Prevention of Bioaccumulation Problems for Human Health

States are not required to allow mixing zones. Where unsafe fish tissue levels or other evidence indicates a lack of assimilative capacity in a particular waterbody for a bioaccumulative pollutant, care should be taken in calculating discharge limits for this pollutant or the additivity of multiple pollutants. In particular, relaxing discharge limits because of the provision of a mixing zone may not be appropriate in this situation.

4.4 MIXING ZONE ANALYSES

Proper design of a mixing zone study for a particular waterbody requires estimation of the distance from the outfall to the point where the effluent mixes completely with the receiving water. The boundary is usually defined as the location where the concentrations across a transect of the waterbody differ by less than 5 percent. The boundary can be determined based on the results of a tracer study or the use of mixing zone models. Both procedures, along with simple order-of-magnitude dilution calculations, are discussed in the following subsections.

If the distance to complete mixing is insignificant, then mixing zone modeling is not necessary and the fate and transport models described in Section 4.5 can be used to perform the WLA. **It is important to remember that the assumption of complete mixing is not a conservative assumption for toxic discharges; an assumption of minimal mixing is the conservative approach.** If completely mixed conditions do not occur within a short distance of the outfall, the WLA study should rely on mixing zone monitoring and modeling. Just as in the case of completely mixed models, mixing zone analysis can be performed using both steady-state and dynamic techniques. State requirements regarding the mixing zone will determine how water quality criteria are used in the TMDL.

This section is divided into five subsections. The first discusses recommendations for outfall designs and means to maximize initial dilution. The second provides a brief description of the four major waterbody types and the critical design period when mixing zone analysis should be performed for each. The third provides a brief description of tracer studies and how they may be used to define a mixing zone. The fourth and fifth subsections discuss simplified methods and sophisticated models to predict the two stages of mixing (i.e., discharge-induced and ambient-induced mixing). For a detailed explanation of the mechanisms involved in estimating both stages of mixing, two references are recommended, Holley and Jirka [15] and Fischer et al. [16]. Although the models presented in Sections 4.4.4 and 4.4.5 simplify the mixing process, the assessor should have an understanding of the basic physical concepts governing mixing to use these

models appropriately. (The U.S. EPA Center for Exposure Assessment Modeling [CEAM] in Athens, Georgia, provides an overview course that teaches the basics of mixing and how the basics should be used for water quality management.)

It is important to note that the mixing zone models presented here attempt to predict the dispersion and dilution of the effluent plume. They do not attempt to predict any removal or transformation of the pollutants. In the near field, dispersion and dilution caused by discharge-induced mixing and then ambient-induced mixing will be the major cause of toxicity reduction. If incomplete mixing persists downstream (such as in the case of shore hugging plumes), then some far-field processes will become important. Some of the models described in Section 4.5 that have sophisticated hydrodynamic simulation routines coupled with fate simulation routines may be used for these far-field, incomplete mixing analyses.

4.4.1 General Recommendations for Outfall Design

An important factor in maximizing the initial dilution of an effluent is the design of the effluent outfall. There are three major types of outfall designs: surface discharge from free flows in a pipe or canal, single-port submerged discharge, and multiport submerged discharge. The last type is often referred to as multiport diffusers. Of the three, the surface discharge type is the least favorable for toxic discharges since it offers the least initial mixing. In particular, surface discharges at the shoreline of a waterbody usually have an impact along the shoreline when there is significant cross-flow and thus yield high surface concentrations.

Submerged discharges offer more flexibility in meeting the design goals for toxic discharges. Submerged discharges may be in the form of a single pipe outlet or of multiport discharges (diffusers) giving rise to one or several submerged discharge jets. A typical diffuser section is illustrated in Figure 4-1. Submerged discharges allow the effluent to be directed at different angles to the ambient flow to maximize the initial dilution. Diffusers are particularly effective in counteracting the buoyancy of the effluent. However, submerged multiport discharges are only feasible in waterbodies that are of sufficient depth and are not subjected to periodic dredging or to considerable scour or deposition.

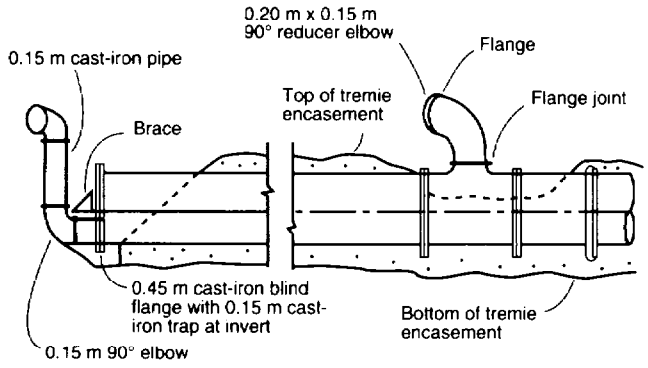


Figure 4-1. A Typical Diffuser Section [17]

Many of the complexities of submerged diffusers have been summarized by Jirka [18], Holley and Jirka [15], and Roberts et al. [19, 20, 21]. Submerged discharges should be designed to avoid direct surface impingement and bottom attachment of the submerged jet or jets. Surface and bottom impacts should be evaluated at critical design conditions (low flow or high stratification) and at off-design conditions (higher flow or lower stratification) to ensure the best placement and design of the diffuser. Diffusers provide more dilution than single outlets, but the alignment of the diffuser with the receiving water flow direction influences how much dilution will be provided. If the outlet structure is directed parallel to the direction of flow, dilution under high ambient velocities (off-design conditions) may be lower than under low velocities (critical design conditions).

In rivers, the preferred arrangement for a submerged discharge is to direct the outlet into the current flow direction or vertically upward. To deal with the reversing currents of estuaries and coastal bays, the preferred arrangements for offshore discharges are parallel diffuser alignment (tee diffuser) and perpendicular diffuser alignment (staged diffuser) [18]. In lakes and reservoirs, the preferred arrangement for a negatively buoyant discharge is to direct the diffuser vertically upward. A positively buoyant, vertically directed jet could penetrate stratification, so the preference for this type of discharge is to orient the diffuser at a slight angle above the horizontal. For ocean outfalls, initial dilution is improved by longer (perpendicular to the shoreline) and deeper diffusers. Further, the ports of the diffuser should be sufficiently separated to minimize merging of the separate plumes [22].

4.4.2 Critical Design Periods for Waterbodies

This section provides a brief description of the four major waterbody types and defines the critical design periods that should be used when performing mixing zone analyses in each of these waterbody types. Appendix D provides a further discussion on the appropriate selection of design periods.

1) Rivers and Run-of-River Reservoirs

Rivers and run-of-river reservoirs are waterbodies that have a persistent throughflow in the downstream direction and do not exhibit significant natural density stratification. Recommendations for hydrologically based and biologically based design flows for completely mixed, steady-state modeling of rivers are described in Appendix D of this document. The biologically based design flows are determined using the averaging periods and frequencies specified in water quality criteria [8]. Also, the hydrologically based flows 1Q10 and 7Q10 for the CMC and CCC, respectively, have been used traditionally and may continue to be used for steady-state modeling. Run-of-river reservoirs with residence times less than 20 days at critical conditions also should be analyzed using biologically or hydrologically based design flows (see below). Regulated rivers may have a minimum flow in excess of these toxicological flows. In such cases, the minimum flow should be used in TMDL modeling.

2) Lakes and Reservoirs

This receiving water category encompasses lakes and reservoirs with residence times in excess of 20 days at critical conditions

[23]. Seasonal variations in the water level, wind speed and direction, and seasonal solar radiation should be determined to define the critical period [23]. In the case of long and narrow reservoirs, areas above the plunge point (i.e., areas where no stream-like flow is present and waters are mixed or stratified by density) can be analyzed as rivers. The areas below can be analyzed as reservoirs. Since effluent density relative to the ambient water can vary over seasons, no one season or stratification condition can be selected as the most critical dilution situation for all cases. In general, all four seasons should be analyzed to determine the most critical periods for mixing zone analyses. All seasonal analyses should assume an ambient velocity of zero unless persistent currents have been documented. Special attention should be given to periods of rising water level since pollutants can move back into coves and accumulate under these conditions. Location of discharges in coves and dead-end embayments should be prevented whenever possible.

3) Estuaries and Coastal Bays

This receiving water category encompasses estuaries, which are defined as having a main channel reversing flow, and coastal bays, which are defined as having significant two-dimensional flow in the horizontal directions. For both waterbodies, the critical design conditions recommended here are based on astronomical, not meteorological, tides.

Determining the nature and extent of the discharge plume is complicated in marine systems by such conditions as differences in tides, riverine input, wind intensity and direction, and thermal and saline stratification. Because of the tidal nature of the estuaries and coastal systems and their complex circulation patterns, dilution of discharges cannot be determined simply by calculating the discharge rate and the rate of receiving water flow (i.e., the design flow). For example, tidal frequency and amplitude vary significantly in different coastal regions of the United States. Furthermore, tidal influences at any specific location have daily and monthly cycles. These and additional factors require that direct, empirical steps be taken to ensure that basic dilution characteristics of a discharge to salt water are determined.

In estuaries without stratification, the critical dilution condition includes a combination of low-water slack at spring tide for the estuary and design low flow for riverine inflow. In estuaries with stratification, a site-specific analysis of a period of minimum stratification and a period of maximum stratification, both at low-water slack, should be made to evaluate which one results in the lowest dilution. In general, minimum stratification is associated with low river inflows and large tidal ranges (spring tide), whereas maximum stratification is associated with high river inflows and low tidal ranges (neap tide).

After either stratified or unstratified estuaries are evaluated at critical design conditions, an off-design condition should be checked. The off-design condition (e.g., higher flow or lower stratification) recommended for both cases is the period of maximum velocity during a tidal cycle. This off-design condition results in greater dilution than the design condition, but it causes the maximal extension of the plume. Extension of the plume into critical resource areas may cause more water quality problems than the high-concentration, low-dilution situation.

Recommendations for a critical design for coastal bays are the same as for stratified estuaries. The period of maximum stratification must be compared with the period of minimum stratification in order to select the worst case. The off-design condition of maximum tidal velocity should also be evaluated to predict the worst-case extent of the plume.

4) Oceans

Critical design periods for ocean analyses are described in two separate documents, the Section 301(h) *Technical Support Document* [22] and the Section 301(h) document, *Initial Mixing Characteristics of Municipal Ocean Discharges* [24]. The following subsection contains a summary from these documents. Like discharges to estuaries, discharges to ocean waters are subject to two-dimensional horizontal flows. Oceanic critical design periods must include periods with maximum thermal stratification, or density stratification. These periods shorten the distance of vertical diffusion that occurs in the zone of initial dilution. Thus, during these periods it is difficult to achieve the recommended 100-to-1 dilution that is to occur before the plume begins a predominantly horizontal flow as compared to vertical flow. Periods when discharge characteristics, oceanographic conditions (spring tide and neap tide currents), wet and dry weather periods, biological conditions, or water quality conditions that indicate that water quality standards are likely to be exceeded should also be noted. The 10th percentile value from the cumulative frequency of each parameter should be used to define the period of minimal dilution.

4.4.3 General Recommendations for Tracer Studies

A tracer or dye study can be used to determine the areal extent of mixing in a waterbody, the boundary where the effluent has completely mixed with the ambient water, and the dilution that results from the mixing. Analysis of the mixing zone with a dye study that is supplemented with modeling should be performed at flow conditions that approach critical flow. Some of those design conditions are summarized above in the subsections dealing with specific waterbodies. Once the critical design condition has been selected for a waterbody, dye studies can be performed to provide data on the dimensions and dilution of the wastewater plume during this critical period. Tracer studies other than dye studies (e.g., chloride, lithium) can be performed for cases in which the receiving water is amenable to such tests.

For WLA studies in which a discharge is already in operation, tracer studies can be used to determine specific concentration isopleths in the mixing zone that reflect both discharge-induced and ambient-induced mixing. The isopleth concentrations, with effluent toxic concentrations, should be superimposed over a map of the various resource zones of the waterbody. The map will illustrate whether the State's mixing zone dimensions are exceeded, whether the required zone of passage is provided, and whether the plume avoids critical resource areas. The WLA can then be calculated to provide the appropriate zone of passage and to prevent detrimental impacts on spawning grounds, nurseries, water supply intakes, bathing areas, and other important resource areas.

Obviously, if the outfall is not yet in operation, it is impossible to determine discharge-induced mixing by tracer studies. Tracer

studies can be used in these situations to determine characteristics of the ambient mixing. For ambient mixing studies, the tracer release can be either instantaneous or continuous. Instantaneous releases are used frequently to measure longitudinal dispersion, but can also be used to determine lateral mixing in rivers [15] and lateral and vertical mixing in estuaries, bays, reservoirs, and lakes. For waterbodies with significant flow velocities, continuous releases of tracer are normally used to determine lateral and vertical mixing coefficients. Continuous releases can also be used to determine three-dimensional concentration isopleths for steady-state conditions. The tracer study must be made at critical design conditions in order to use the results directly for WLAs. If a tracer study for ambient mixing is conducted at near-to-design conditions, the observed data can be used to determine dimensionless mixing coefficients. These coefficients can then be extrapolated to critical conditions using hydraulic parameters [15]. A tracer study at near-to-critical conditions also can be used to determine the computer model required to predict critical-condition mixing and provide the coefficients needed for that TMDL model.

A number of references provide information concerning the design, conduct, and analysis of tracer studies for mixing analyses. *Techniques of Water-Resources Investigations of the USGS* provides the best overview of how to conduct tracer studies [25, 26, 27]. The fluorescent dyes (usually Rhodamine WT), measuring equipment, fluorometers, field and laboratory procedures, and calculation methods are all discussed. The procedures essentially consist of adding dye to the waterbody and recording concentrations of the dye at various stations at specific time intervals. Examples of tracer studies for river systems are presented in Fischer [28]; Kisiel [29]; Holley and Jirka [15]; and Yotsukura, Fisher, and Sayre [30]. Examples of tracer studies in tidal systems are presented in Wilson, Cobb, and Yotsukura [31] and Hetling and O'Connell [32], both of which are studies of the Potomac River estuary; Baily [33], a study of Suisun Bay in California; Fischer [34], a study of Bolinas Lagoon, a coastal bay in Marin County, California; and Crocker et al. [35], a study of Corpus Christi Bay, Texas. Methods to perform a tracer study in a reservoir are provided in Johnson [36].

The dye study recommended for obtaining a quick saltwater dilution assessment is one in which Rhodamine WT dye is administered to a discharge and monitored in the receiving waters for not less than 24 hours. The basic goal of this study is to determine the near-field nature of the effluent dilution, not the steady-state or far-field dilution. The environmental and discharge conditions selected for the study should be those that would elicit "worst-case" conditions (i.e., highest ambient concentrations in the receiving water). These include low wind, neap tide (tide of minimum range occurring during the 1st and 3rd quarters of the moon), plume trapping by density stratification, low rainfall and low riverine input, and, if possible, high effluent discharge.

The dye should be administered to the effluent before discharge to the receiving water in proportion to effluent flow rate. Dye should be maintained at a concentration in the effluent sufficient to permit detection of the dilution ratio of interest when the amount and variability of background fluorescence in the receiving water are taken into account. Measurements of dye concentration are made using a fluorometer and should be corrected for water temperature.

A survey of background fluorescence and its variability in the anticipated mixing zone must be conducted just prior to the beginning of the study in order to permit correction of fluorescence data and to determine the dye concentration required in the effluent. Since Rhodamine WT dye is bleached by free chlorine, a preliminary study of the degree of dye bleaching by the effluent should precede the study for chlorinated discharges to avoid underestimation of the extent of the mixing zone. Dye concentrations should be surveyed for two successive slack tides, and for any other conditions that could lead to concentration maxima. Surveys should extend from the point of discharge to a distance at which the effluent dilution ratio of interest is attained. The dye fluorescence at this point should be at least twice the variability in background fluorescence.

EPA has completed two TMDL studies to test the procedures outlined in the previous version of this document. Both studies used dye to determine the mixing zone and the dilution within it. The first study was performed on the Amelia River, an estuarine system in Florida [2]; the second was performed on the Greenwich Cove, an embayment of Narragansett Bay in Rhode Island [37]. In both studies, Rhodamine WT dye was introduced continuously into the effluent and numerous stations were set up to measure the spatial and temporal distribution of the dye. Both studies are good examples of how to perform a dye study in complex tidal systems.

4.4.4 Discharge-induced Mixing

The first stage of mixing is controlled by discharge jet momentum and buoyancy of the effluent (see Figure 4-2). This stage generally covers most of the regulatory or near-field mixing zone. It is particularly important in lakes and reservoirs and slow moving rivers since ambient mixing in those waterbodies is minimal.

In shallow environments, it is important to determine whether near-field instabilities occur. These instabilities, associated with surface and bottom interaction and localized recirculation cells extending over the entire water depth, can cause buildup of effluent concentrations by obstructing the effluent jet flow. There are no simple means to estimate dilution in these cases. Criteria for these instabilities and specialized predictive models have been developed to address these problems [13].

In the absence of near-field instabilities, horizontal or nearly horizontal discharges will create a clearly defined jet in the water column that will initially occupy only a small fraction of the available water depth. The following equations and models are designed to describe mixing under stable near-field conditions.

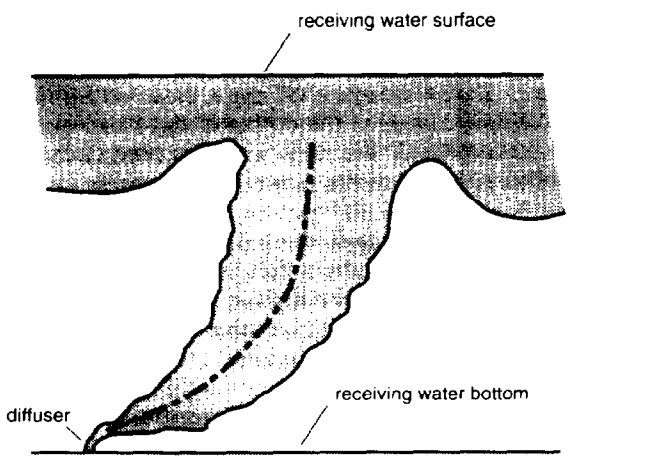
1) Use of a Simplistic Screening Equation

A minimum estimate of the initial dilution available in the vicinity of a discharge can be made using the following equation derived from information in Holley and Jirka (1986) [15]:

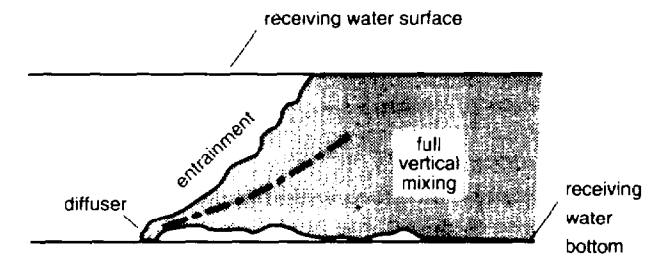
$$S = 0.3 \frac{x}{d}$$

where

- S = flux-averaged dilution
- x = distance from outlet
- d = diameter of outlet.



c) Deep-water, high-buoyancy, nonvertical discharge



d) Shallow-water, low-buoyancy, nonvertical discharge

Figure 4-2. Example of Discharge-Induced Mixing [7]

The coefficient 0.3 represents the average of two values derived from the literature, 0.28 [16] and 0.32 [38].

The equation provides a minimum estimate of mixing because it is based on the assumptions that outlet velocity is zero and the discharge is neutrally buoyant. Dilution may be underestimated for partially full pipes because the equation assumes a fully flowing pipe. The equation can be used in inverse form to solve for the discharge x at which a desired solution—for example, that corresponding to the CMC—has been achieved. The equation is valid only close to the discharge, up to a distance corresponding to several (two to three) water depths. At longer distances, other factors are of increasing importance in jet mixing and must be included.

Mixing graphs that include the effects of discharge buoyancy, ambient velocity, and stratification can be found in Holley and Jirka [15], Fischer et al. [16], and Wright [39]. They are useful to account for these other initial dilution factors and can aid in determining whether criteria will be met at the edge of the regulatory mixing zone.

2) Use of Detailed Computer Models

More detailed design data for the mixing zone can be obtained from the use of computer models based on integral jet techniques. It is important to note that most models represent an idealization of actual field conditions and must be used with caution to ensure that the underlying model assumptions hold for the site-specific situation being modeled. In general, these buoyant jet models require the following input data: discharge depth, effluent flow rates, density of effluent, density gradients in receiving water, ambient current speed and direction, and outfall characteristics (port size, spacing, and orientation). Model output includes the dimensions of the plume at each integration step, time of travel to points along the plume centerline, and the average dilution at each point.

Described below are six mixing zone models that are available through EPA. All of the models require a user who is well versed in mixing concepts and the data necessary to run the models. The first model, CORMIX [40, 41], may be the most useful to regulators since it is an expert system that guides the user in selecting an appropriate modeling strategy for rivers or estuaries. It is available from the National Technical Information Service (NTIS), and user support is available from the U.S. EPA CEAM. The other models were developed and designed for ocean discharges. All but one can be used on rivers, lakes, and estuaries with appropriate input modifications; UPLUME is restricted to stagnant water environments where the ambient water current velocity is zero (e.g., lakes, reservoirs).

These five models were designed for submerged discharges in oceans. They all report dilution, and all terminate execution when the vertical ascent of the plume is zero (e.g., when the plume reaches the surface or when plume density is equal to ambient density in some stratified systems). With the exception of CORMIX1, they all assume that there is a "deep" receiving stream (i.e., no bottom interference). They too are available from NTIS, and user support is provided by the U.S. EPA Hatfield Marine Science Center in Newport, Oregon [24]. These five models have been modified such that the user inputs the data into a universal data format that allows the user to apply any of the five models with only minor input changes.

- CORMIX is a series of software elements for the analysis and design of a submerged buoyant or nonbuoyant discharge containing conventional or toxic pollutants and entering into stratified or unstratified watercourses, with emphasis on the geometry and dilution characteristics of the initial mixing zone. Subsystem CORMIX1 deals with single-port discharges, and subsystem CORMIX2 addresses multiport diffusers. The system operates on microcomputers with the MS-DOS operating system. CORMIX1 can summarize dilution characteristics of the proposed design, flag undesirable designs, give dilution characteristics at specified boundaries (i.e., legal and toxic mixing zones) and recommend design alterations to improve dilution characteristics. The CORMIX1 program guides the user, based on the user's input, to

appropriate analyses of design conditions and mixing zone dimensions.

- UPLUME is an initial dilution model that can be used for stagnant waterbodies, such as lakes and reservoirs, where the ambient currents can be assumed to be zero. The model simulates a submerged single-port discharge. The bouyancy between the effluent and ambient water can be accounted for, and the discharge can be given a vertical angle. UPLUME calculates flux-averaged dilutions and, for one output option, a centerline dilution.

- UOUTPLM can be used in flowing and stagnant waterbodies. The user specifies the current speed of the ambient water, and this speed is assumed to be constant with depth. The model simulates a submerged single-port discharge. Buoyancy between the effluent and ambient water can be modeled, as well as the discharge vertical angle. The ambient current is assumed to be perpendicular to the diffuser.

- UMERGE is a model that can also be used for both flowing and stagnant waters. It has capabilities that UOUTPLM does not have: it considers multiple submerged ports, and the user can specify arbitrary ambient current speed variations with depth. The ports are assumed to be equally spaced. The model accounts for adjacent plume interferences over the course of the plume trajectory and in the subsequent dilution calculation. Positive buoyancy is accounted for, and the discharge vertical angle can be modified. The ambient current is assumed to be perpendicular to the diffuser.

- UDKHDEN is a three-dimensional model that can be used for flowing and stagnant waterbodies. It has all the capabilities of UMERGE plus the ability to simulate instances where the ambient current flow is not perpendicular to the diffuser.

- ULINE models a vertical slot jet discharge into a flowing waterbody. The discharge angle is assumed to be perpendicular to ambient current. The ambient current may vary with depth, and the axis of the diffuser may range from parallel to perpendicular to the ambient current. The buoyancy of the effluent can also be modeled.

An evaluation and comparison of all these models can be found in the *Technical Guidance Manual for Performing Wasteload Allocations—Book 3, Estuaries* [7].

4.4.5 Ambient-Induced Mixing

The equations for discharge-induced mixing can be used to predict concentrations in the regulatory mixing zone where strong jet mixing predominates over ambient mixing. Beyond this point, the mixing is controlled by ambient turbulence. Thus, ambient mixing models must be used to predict the pollutant concentration distributions up to the stage of complete lateral mixing to provide boundary conditions for the completely mixed fate and transport models described in Section 4.5. This information also may be needed to estimate concentrations encountered at important resource areas or at subsequent downstream dischargers.

If there is no discharge-induced vertical mixing associated with the jet action of the discharge, then mixing over the depth of the waterbody must be accomplished by ambient mixing. For a neutrally buoyant, soluble effluent discharged with low velocity at the surface or at the bed of a stream, the flow distance required to achieve complete vertical mixing is on the order of 50 to 100 times the depth of water in that portion of the channel where the effluent is discharged [42]. For a discharge that is either lighter (positively buoyant) or heavier (negatively buoyant) than the ambient water, but still has no excess momentum, the flow distance for mixing over the depth will be greater. In the normal case with a high-velocity jet designed to prevent lethality in the mixing zone, mixing over the depth will be accomplished primarily by jet action, and the distance required for this vertical mixing will be much shorter.

In general, ambient mixing must also accomplish mixing over the width of a waterbody to bring the effluent to the completely mixed condition. For situations where the width of the zone that is mixed by the discharge-induced mixing is much smaller than the width of the river, the flow distance (X_m) required to achieve the completely mixed condition may be estimated from an equation of the form [16]:

$$X_m = \frac{mW^2u}{D_y}$$

where

- W = width of the river
- u = flow velocity for the critical design flow
- D_y = lateral dispersion coefficient as discussed below
- m = a parameter whose value depends on the degree of uniformity used to define "complete mixing" and on the transverse location of the outfall in the stream.

If completely mixed conditions are defined as a 5-percent variation in concentration across the stream width, the value of m would be approximately 0.1 for a discharge near the center of river flow (not the center of river width) and approximately 0.4 for a discharge near the edge of the river. If, because of other uncertainties, a 25-percent variation across the width is accepted as being completely mixed, then the corresponding values for m would be approximately 0.06 for a discharge near the center of river flow and approximately 0.24 for a discharge near the edge of the river. For a very small stream, X_m may be only a few hundred feet; for medium and large streams, X_m is normally several miles to several tens of miles.

The lateral dispersion coefficient (D_y) for most rivers can be calculated with the following equation [16]:

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

where

- d = water depth at design flow
- u^* = shear velocity.

The coefficient (0.6) can vary from 0.3 to above 1.0 depending on the type and degree of irregularity of the channel cross-sections. The more straight and uniform the flow, the lower the

value; the more irregular the flow (resulting from curves, sidewall interference, etc.), the higher the value. Values approaching and exceeding 1.0 are normally associated with significant channel meandering [42]. The following equation for shear velocity should be used [16]:

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

where

- g = acceleration due to gravity
- s = slope of the channel
- d = water depth.

For diffusers that initially spread the discharge across a significant part of the river width or for cases where the discharge-induced mixing causes mixing across a significant part of the river width, the values of m and X_m can be smaller than the ones indicated here. For distances greater than X_m, the models for completely mixed effluents discussed in Section 4.5 can be used to calculate concentrations at these distances. For shorter distances, maximum concentrations can be much greater than those predicted by "completely mixed" models and should be estimated using the following equation:

$$C_x = \frac{C_e Q_e W}{Q_s(\pi D_y X/u)^{1/2}}$$

where

- C_x = maximum pollutant concentration distance x from the outlet
- C_e = effluent concentration
- Q_e = design effluent flow
- Q_s = design stream flow
- D_y = lateral dispersion coefficient
- X = distance from the outlet
- W = stream width
- u = flow velocity for the design flow.

It should be noted that this estimate of C_x is a worst-case prediction since the equation assumes no significant discharge-induced mixing and a neutrally buoyant effluent. A more accurate way to predict concentrations within this second stage of mixing is to use the methods of Yotsukura and Sayre [42]. To use this approach, however, the value of D_y and pollutant concentrations after discharge-induced mixing must be known from tracer studies and/or from the use of one of the discharge-induced models.

The PSY model can be used to predict ambient mixing in shallow, freshwater streams where water depth is small in proportion to the width. PSY is a steady-state, two-dimensional plume model that predicts dilution of a surface discharge into a shallow receiving water where the plume attaches to both bottom and nearshore [43]. Uniform vertical mixing is assumed to occur at the point of discharge.

Ambient mixing is minor for lakes and reservoirs because flow velocity is assumed to be minimal and mixing is accomplished by means of the discharge momentum and buoyancy. For estuaries that are completely mixed with regard to salinity, the equations presented above can be used to estimate concentrations between the outlet and the point of complete mixing with a slight modification of shear velocity. The above equations will be applicable to only unstratified estuaries since the time required to mix across the estuary must be significantly less than the time required for the effluent to pass out of the unstratified part of the estuary, the time required for the effluent to pass into a segment of greatly changed cross-section, or the time required for the substance to decay. When the above equations for estuaries are used, the velocity of the design flow should include the velocity associated with the inflow of freshwater as well as the tidal velocity; thus u_t, which is based on an average total velocity; is substituted for u in the equations and shear velocity becomes

$$u^* = 0.10\ u_t.$$

The CORMIX expert system model can also be used to obtain predictions for the ambient-induced mixing. In addition to the routines for discharge-induced mixing, this model also includes predictive elements that apply to ambient mixing in riverine, lake, or coastal situations.

4.5 COMPLETELY MIXED DISCHARGE RECEIVING WATER SITUATIONS

At the present time, most States and EPA Regions use steady-state models that assume the wastewater is completely mixed with the receiving waters in order to calculate WLAs for contaminants. This approach is appropriate for conventional contaminants where critical environmental effects are expected to occur far downstream from the source. WLAs for toxic chemicals require a different approach, however, because critical environmental conditions occur near the discharge before complete mixing with the receiving water occurs. Consequently, mixing analyses should be performed because many of these toxicants can exert maximal toxicity in a variety of regions spanning from the discharge point to significant distances downstream.

If complete mixing occurs near the discharge point, such as in effluent-dominated receiving streams, then steady-state models may be used to calculate TMDLs. Recent EPA developments in the identification of critical design flows based on toxicological concerns provide for better use of steady-state models in calculating toxic WLAs. However, if complete mixing does not occur near the discharge point and the effluent plume is discernible downriver, then modeling techniques that can simulate and predict mixing conditions are more appropriate. The mixing zone models presented in the previous section may be used to define the mixing zone. However, they only determine the dispersion and dilution of the effluent and do not account for chemical or biological processes in the mixing zone. TMDL models are available that can simulate mixing processes and predict areas of maximal concentrations in the receiving stream based on chemical, biological, and physical processes.

4.5.1 Wasteload Modeling Techniques

1) Steady-State Modeling Techniques

A steady-state model requires single, constant inputs for effluent flow, effluent concentration, background receiving water concentration (RWC), receiving water flow, and meteorological conditions (e.g., temperature). The frequency and duration of ambient concentrations predicted with a steady-state model must be assumed to equal the frequency and duration of the critical receiv-

ing water conditions used in the model. The variability in effluent flows and concentrations also affects RWCs, but these effects cannot be predicted with constant inputs. Steady-state models can be improved for toxic WLAs by means of the following:

- Using design flows that will ensure criteria compliance at the appropriate duration and frequency.
- Calculating both acute and chronic WLAs.

EPA is encouraging the States to adopt two-number aquatic life water quality criteria and is using them in WLA studies. Ambient water quality criteria have been established for numerous toxic pollutants. These criteria specify an acute concentration (CMC) and a chronic concentration (criteria continuous concentration, or CCC) for each toxicant, as well as durations and frequencies of exposure for the two concentration levels. The design flows used in steady-state modeling should be reflective of the CCC and CMC durations and frequencies. The duration of the design flow is based on the maximum exposure time that will prevent acute and chronic effects. The duration of flow is assumed to apply to the duration of the allowable effluent concentration or load. For example, if the flow used is a 7-day average value, the allowable load is considered to be a 7-day average. The return frequency is based on the number of years required for biological population recovery after criteria have been exceeded. Appendix D describes the toxicological basis for selecting receiving stream design flows for steady-state modeling and recommends specific design flows for CCC and CMC calculation of TMDLs for rivers and streams.

In summary, there are two types of design flows, hydrologically based and biologically based. The hydrologically based design flows are those traditionally used by the States, in which the 7Q10 flow is used as the CCC design flow and the 1Q10 is used as the CMC design flow. The biologically based method uses the 1-day, 3-year duration-frequency for determining the CMC design flow and the 4-day, 3-year duration-frequency for determining the CCC design flow. Consequently, the biologically based design flows are based on specific toxicological effects of a pollutant and biological recovery times from localized stresses [6]. The advantages of both types, as well as how they may be calculated, also are described in Appendix D.

A 4-day, 3-year biological design flow does not equate to a 4Q3 hydrological design flow. EPA has determined that a 4Q3 design flow would result in an excessive number of water quality criteria exceedances. As explained in Appendix D, a hydrologically based 7Q10 will, for most streams, be similar to a biologically based 4-day, 3-year design flow.

At the present time, there are no recommended toxicological flows for steady-state modeling of lakes, reservoirs, or estuaries. The design conditions recommended for these waterbodies in Section 4.4.2 are based on hydrological and meteorological conditions rather than on toxicological duration and frequency data. These conditions should be used until further guidance is provided.

Another improvement in steady-state toxics modeling can be realized by performing two separate WLAs, one for the CMC and one for the CCC. Steady-state WLA models should be used to calculate the allowable effluent load that will meet the CMC at the acute design flow and the allowable load that will meet the CCC at the chronic design flow. Calculation of these values will enable the permit writer to calculate the more limiting long-term average (LTA) for the treatment system and develop permit limits protective of both WLAs (see Chapter 5).

In addition to stream design flow, steady-state models require design temperature, pH, alkalinity, and hardness, depending on the pollutants modeled at site-specific conditions. To determine stream design temperature, pH, alkalinity, and hardness, a program called DESCON was developed. (See Appendix D for additional information.) DESCON is a computer program that estimates design conditions for WLA modeling. These conditions are based on maintaining a desired limit on the frequency of water quality excursions in a receiving water. DESCON considers the effect that daily fluctuations in stream flow and water quality conditions, such as temperature and pH, have on the variability of the capability of a receiving water to accept pollutant loadings. It specifically accounts for the within-year correlations observed between such variables as stream flow, temperature, pH, alkalinity, hardness, and dissolved oxygen. DESCON determines design conditions using a four-step process (see Figure 4-3):

1) A long-term record of observed stream flows and pertinent water quality data are assembled or synthesized.
2) The maximum allowable pollutant load that the receiving water can accept without causing a water quality excursion is computed for each day of this record.
3) This synthesized record of allowable loads is searched for the critical load, i.e., the load whose frequency of not being exceeded matches the desired water quality excursion frequency.
4) Design conditions are then derived from receiving water conditions realized during the period of record when the computed allowable load was closest to the critical load.

DESCON provides the same advantages as continuous simulation by considering the joint occurrences of stream flow and other water quality parameters as observed in the historical record. In addition, it is more computationally efficient; it contains a facility for extracting and analyzing flow and water quality data from STORET; it can use both the extreme value and the biologically based methods of calculating of water quality excursions; and it is specifically designed to handle such pollutants as ammonia, heavy metals, pentachlorophenol, and biochemical oxygen demand (BOD) for which water quality criteria are functions of such design condition variables as temperature, pH, alkalinity, hardness, and dissolved oxygen. The main limitations of DESCON are that it requires at least 10 years of historical daily flow data and it can only analyze a single discharger, edge-of-mixing zone situations (or a simplified Streeter-Phelps dissolved oxygen response for BOD).

2) Dynamic Modeling Techniques

Steady-state modeling considers only a single condition; effluent flow and loading are assumed to be constant. The impact of receiving water flow variability on the duration for which and frequency with which criteria are exceeded is implicitly included

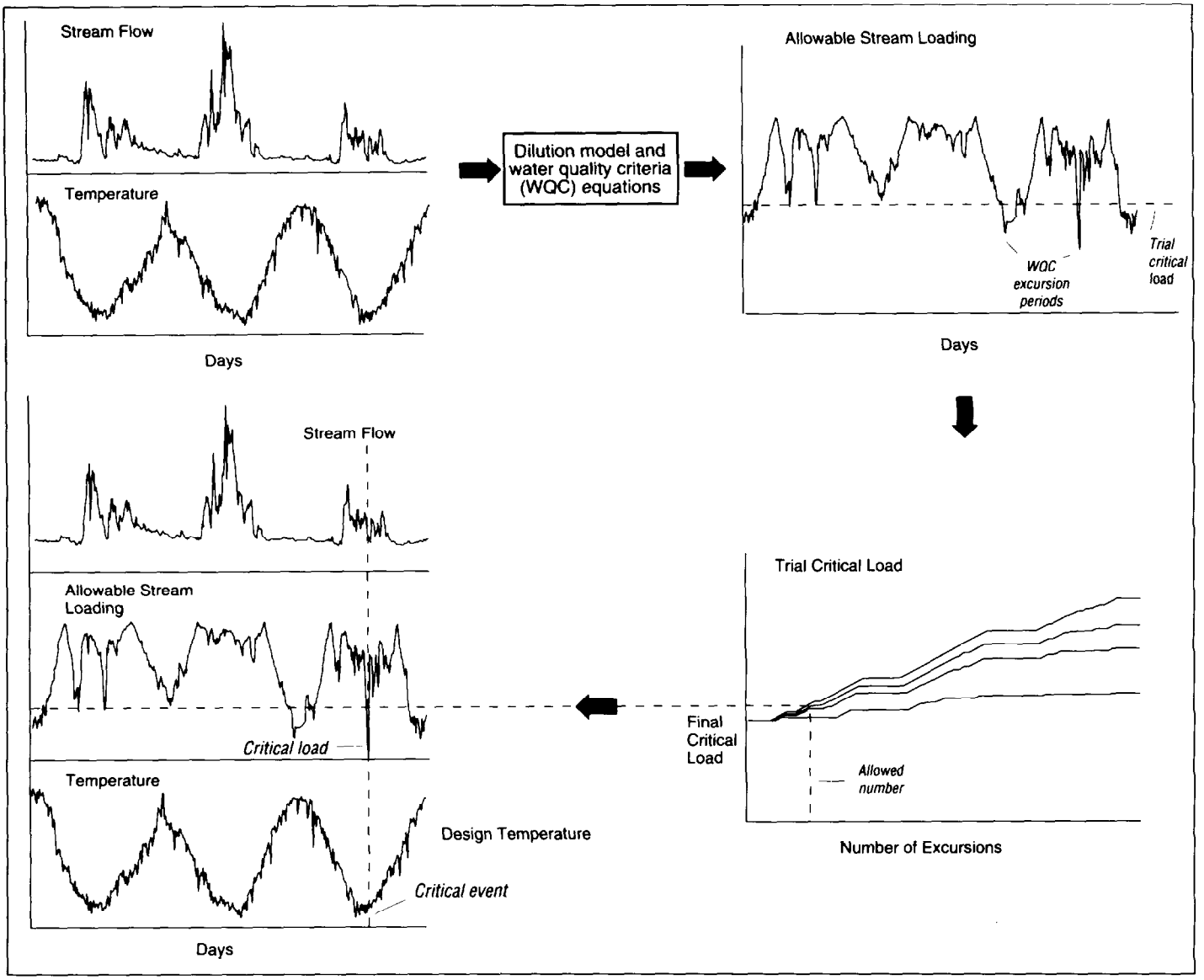


Figure 4-3. Computational Scheme for Deriving Design Conditions

in the design conditions if these conditions reflect the desired toxicological effects regime. Dynamic modeling techniques explicitly predict the effects of receiving water and effluent flow and of concentration variability. The three dynamic modeling techniques recommended by EPA for WLAs are continuous simulation, Monte Carlo simulation, and lognormal probability modeling. These methods calculate a probability distribution for RWCs rather than a single, worst-case concentration based on critical conditions. Prediction of complete probability distributions allows the risk inherent in alternative treatment strategies to be directly quantified.

The use of probability distributions in place of worst-case conditions has been accepted practice for years in water resource engineering, where it was found to produce more cost-effective design of bridge openings, channel capacities, floodplain zoning, and water supply systems. The same cost-effectiveness can be realized for pollution controls if probability analyses are used.

The dynamic modeling techniques have an additional advantage over steady-state modeling in that they determine the entire effluent concentration frequency distribution required to produce the desired frequency of criteria compliance. Maximum daily and monthly average permit limits can be obtained directly from the effluent LTA concentration and coefficient of variation (CV) that characterize this distribution. Generally, steady-state modeling has been used to calculate only a chronic WLA. Steady-state modeling generates a single allowable effluent value and no information about effluent variability. If the steady-state model is used to calculate both acute and chronic wasteloads, limited information will be provided and the entire effluent distribution will not be predicted. Steady-state WLA values can be more difficult to use in permits and enforcement because of the variable nature of the receiving waterbody and the effluent. The outcome of probabilistic modeling can be used to ensure that permit limits are determined based on best probability estimates of RWCs rather than a single, worst-case condition. As a result, maximum daily and monthly average permit limits, based on compliance

with water quality criteria over a 3-year period, can be obtained directly from the probability distribution.

Continuous Simulation Models. As shown in Figure 4-4, a continuous simulation model uses daily effluent flows (Q_e) and concentration data (C_e) with daily receiving water flow (Q_s) and background concentration data (C_s) to calculate downstream RWCs [44]. The model predicts these concentrations in chronological order with the same time sequence as the input variables (C_b versus time). The daily RWCs can then be ranked from the lowest to the highest without regard to time sequence. A probability plot can be constructed from these ranked values, and the occurrence frequency of any 1-day concentration of interest can be determined (C_b versus frequency). Running average concentrations for 4 days (i.e., the chronic design flow), or for any other averaging period, also can be computed from the daily concentrations (Figure 4-5).

The probability plot generated by the continuous simulation model using existing effluent data will indicate whether criteria are predicted to be exceeded more frequently than desired. Appendix D discusses how to select the appropriate allowed frequency of excursions based on the biological recovery period required for a specific waterbody. If recurrence intervals of 10 or 20 years are desired, at least 30 years of flow data should be available to provide a sufficient record to estimate the probability of such rare events. Of the 30 years of required flow data, at least 20 to 25 years should be continuous daily data, with the remaining years represented with only intermittent data. The data should be examined to verify that the receiving stream has not undergone significant hydrological modification. The data also should be examined to determine if there were any long-term changes due to technology-based treatment or periodic changes due to industrial or municipal plant closings or expansions. The same data requirements are also true for the lognormal probabilistic and Monte Carlo methods. However, except for the continuous simulation models, other nonsteady-state models in this section

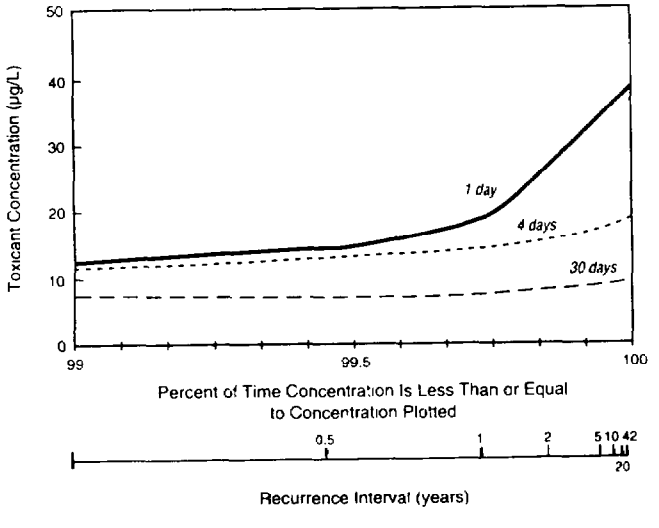


Figure 4-4. Frequency of Occurrence of Concentrations in Receiving Waters and Recurrence Intervals Generated by a Continuous Simulation Model

cannot be used to account for the duration and frequency provision of the two-number water quality criteria. Users are cautioned about the specific limitations of some of the dynamic models included here. Continuous simulation models have the following advantages compared to steady-state formulations:

- The frequency and duration of toxicant concentrations in a receiving water can be predicted.
- The cross-correlation and interaction of time-varying pH, flow, temperature, pollutant discharges, and other parameters are incorporated.
- The effect that the serial correlation of daily flows and other parameters has on the persistence of criteria excursions is incorporated.
- Long-term stream flow records for ungauged rivers using precipitation and evapotranspiration data can be synthesized.
- Long simulation times can prevent the initial conditions used in the model from affecting the calibration of fate and transport processes.

Unlike steady-state models, continuous simulation models require significantly more data to apply, to calibrate, and/or to verify a specific problem and require that input information for the application of the model be time-series data. Also, the model results need manipulation to calculate the effluent LTA concentration and CV for use in developing effluent limits.

Monte Carlo Simulation Models. Monte Carlo simulation combines probabilistic and deterministic analyses since it uses a fate and transport mathematical model with statistically described inputs. Monte Carlo simulations have been the most frequently used approach in stochastic water quality studies [45-51]. The probability distributions of effluent flow, effluent concentration, and other model input must be defined using the appropriate duration for comparison to the CMC and CCC. If 1-day average RWCs must be predicted for CMC comparisons, probability distributions of daily model input data are needed for Monte Carlo simulation. If 4-day average concentrations must be predicted for CCC comparisons, the probability distributions of 4-day average input data are required. The computer selects input values from these distributions using a random generating function. The fate and transport model is repetitively run for a large number of randomly selected input data sets. The result is a simulated sequence of RWCs. These concentrations do not follow the temporal sequence that is calculated with the continuous simulation model, but they can be ranked in order of magnitude and used to form a frequency distribution. Monte Carlo analyses can be used with steady-state or continuous simulation models [52].

The approach for calculating the allowable pollutant load distribution using Monte Carlo simulation is the same as that described for the continuous simulation model. The advantages of Monte Carlo simulation are the following:

- It can predict the frequency and duration of toxicant concentrations in a receiving water.

- It can be used with steady-state or continuous simulation models that include fate processes for specific pollutants.
- It can be used with steady-state or continuous simulation models that include transport processes for rivers, lakes, and estuaries.
- It can be used with steady-state or continuous simulation models that are designed for single or multiple pollutant source analyses.
- It does not require time series data.
- It does not require model input data to follow a specific statistical distribution or function.
- It can incorporate the cross-correlation and interaction of time-varying pH, flow, temperature, pollutant discharges, and other parameters if the analysis is developed separately for each season and the results are combined.

The primary disadvantages of Monte Carlo simulation are that it requires more input, calibration, and verification data than do steady-state models, and the model results need manipulation to calculate the effluent LTA concentration and CV to develop effluent limits.

Lognormal Probabilistic Dilution Model. Without resorting to the continuous simulation method of computing RWCs in temporal sequence, this probabilistic method uses the lognormal probability distributions of the input variables to calculate probability distributions of output variables [53]. As a result, the method requires only the relevant statistical parameters of the input variables (medians and coefficients of variation) rather than the actual time series data needed for continuous simulation. If 1-day average RWCs must be predicted for comparisons with the CMC, lognormal probability distributions of daily input data are needed. If 4-day average concentrations must be predicted, the lognormal probability distributions of 4-day average input data are required. Because this probabilistic model cannot, as yet, incorporate fate and transport processes, it can be used to predict the concentration of a substance only after complete mixing and before degradation or transformation significantly alters the concentration.

The lognormal probabilistic dilution model has the following advantages:

- It can predict the frequency and duration of toxicant concentrations in riverine environments.
- It does not require time series data.
- It can incorporate the cross-correlation and interaction of time-varying pH, flow, temperature, pollutant discharges, and other parameters if the analysis is developed separately for each season and the results are combined.

The lognormal probability dilution model has the following disadvantages:

- It requires more input than a steady-state model.

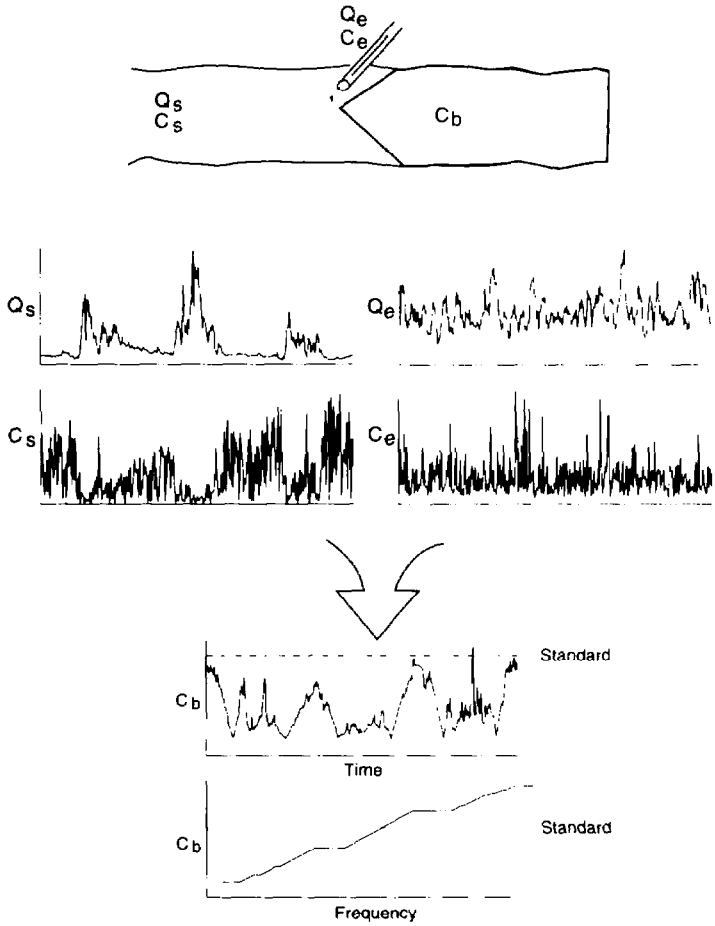


Figure 4-5. Concentration Frequency Curves

- It does not include instream fate processes.
- It applies only to rivers and streams.
- It analyzes multiple pollutant sources inaccurately.
- It requires model input data to be lognormally distributed.

4.5.2 Calculating the Allowable Effluent Concentration Distribution and the Return Period

Information concerning effluent concentration means and variabilities can be obtained from data bases on existing treatment plants and from development documents for specific industrial point source categories. This information is available from the Industrial Technology Division of the Office of Water Regulations and Standards. These effluent data can be used with dynamic models to determine what the effluent concentration distribution must be to meet water quality standards. Two possible approaches can be taken to determine this distribution regardless of the type of dynamic modeling technique (i.e., continuous, Monte Carlo, or lognormal probabilistic). One approach is based on the simplifying assumption that treatment will change only the mag-

nitude of effluent concentrations; no changes are assumed to occur in effluent flows or in the relative variability of effluent concentrations. With these assumptions, no additional model runs are needed to determine the allowable distribution for effluent concentrations. The other approach assumes that the required effluent concentration distribution is the same as the existing distribution except that it is reduced in magnitude by whichever is greater—the percentage necessary for the 1-day average concentrations to meet the CMC, or the 4-day average concentrations to meet the CCC at the desired recurrence interval. Chapter 5 includes details on how permit limits are derived from the mean and coefficient of variation of effluent concentrations determined from this analysis.

The second approach for determining the allowable effluent concentration distribution is based on the assumption that effluent concentrations after treatment will not have the same CV as concentrations before treatment. Studies have documented that advanced secondary treatment increases the CV of BOD and total suspended solids concentrations compared to secondary treatment. Where feasible, investigations should be conducted to evaluate how treatment processes for heavy metals, organic chemicals, and effluent toxicity will change the variability of these constituents. The development documents mentioned above also provide some variability data for treatment processes. To account for a change in variability, an alternative approach should be used to determine the allowable effluent distribution. Iterative model runs can be performed using different concentration means with the effluent "future treatment" variance until a mean is found that meets the criteria at the desired recurrence intervals. These iterative model runs require stochastic generation of effluent input data since daily effluent concentrations will not be available for the hypothetical treatment schemes. The required "future treatment" mean and CV of effluent concentration can then be used to set permit limits (see Chapter 5).

EPA's Office of Water Regulations and Standards developed an interactive preprocessor for DYNTOX that automatically creates input for continuous simulation models, randomly selects the sets of input data required for Monte Carlo simulations, and performs the numerical integration calculation for the lognormal probabilistic model. DYNTOX is available from the EPA CEAM, Environmental Research Laboratory (ERL) [54]. If the observed data base is fairly complete but missing a few points, a linear interpolation scheme is used to fill in the missing data. If data are scarce, a lag-one Markov method is used to generate daily data stochastically. The lag-one Markov method uses the mean, standard deviation, and daily correlation coefficient of the observed data to create random sequences of data having the same statistical properties. The interactive program is written in FORTRAN and is available for use on mainframe or IBM PC-compatible computers.

Two common methods exist to calculate the return period for a given concentration from probabilistic modeling: the **percentile method** and the **extrema method**. The percentile method used by DYNTOX ranks a listing of all individual daily concentrations. The return period for a concentration is then calculated based on the percentile occurrence. In the extrema method, only annual extrema values are used in the ranking. The return periods calculated from these two methods are equally valid statistical representations. When using the percentile method, results express an average return period and multiple occurrences within any year. The extrema method describes the return period for an annual extreme and includes only the extreme of multiple occurrences within a year.

4.5.3 General Recommendations for Model Selection

The reliability of the predictions from any of the modeling techniques depends on the accuracy of the data used in the analysis. The minimum data required for model input include receiving water flow, effluent flow, effluent concentrations, and background concentrations. In many locations, stream flow data should be sufficient for both steady-state and dynamic models. At least 30 years of flow data should be available if excursions of the CMC and CCC must be evaluated at rare frequency of once in 10 or 20 years. Measurements of effluent toxicity or individual toxicity can be much more limited.

If only a few toxicant or effluent toxicity measurements are available, steady-state assessments should be used. Modeling also should be limited to steady-state procedures if a daily receiving water flow record is not available; however, in effluent-dominated situations, critical flow may be used to characterize the receiving stream. Appendix D describes how to select appropriate design flows if State regulations do not require a specific design flow for river WLAs. Fate and transport models or dilution calculations can be used for individual toxicants. At the present time, only dilution calculations or first-order decay equations are recommended for effluent toxicity analyses. Chapter 1 discusses the conservative/additive assumption for toxicity.

If adequate receiving water flow and effluent concentration data are available to estimate frequency distributions, one of the dynamic modeling techniques should be used to develop more cost-effective treatment requirements. If the effluent data exhibit significant seasonal differences or batch process trends, the continuous simulation approach may be the easiest dynamic modeling method to use. The best results will, of course, be obtained if daily effluent flows and concentrations are available for model input for an entire year. The lag-one Markov technique can be used to generate daily effluent data for the entire simulation as long as adequate measurements for the site-specific facility (or a similar one) are available to estimate a day-to-day correlation coefficient and to determine when seasonal or batch process changes in effluent quality occur.

If adequate receiving water flow and effluent concentration data are available and if effluent data exhibit no seasonal or batch process trends, lognormal and Monte Carlo methods may be easier and require less computer time than the continuous simulation approach.

4.5.4 Specific Model Recommendations

The following section recommends models for toxicity and individual toxicants for each type of receiving water—rivers, lakes, and estuaries. Detailed guidelines on the use of fate and transport models of individual toxicants are included in the toxic TMDL guidance available from the Monitoring Branch of EPA's Office of Water Regulations and Standards [5, 6, 7] and Office of Research and Development [55]. These manuals describe in detail the

transport and transformation processes involved in water quality modeling. Transport processes include the dispersion and advection of a contaminant once it enters the receiving stream; its volatilization from the water; and its sorption to suspended sediment, eventual settling, and possible resuspension and diffusion from the sediment. Transformation processes include the oxidation, hydrolysis, photolysis, biodegradation, and bioaccumulation of the chemical.

Most water quality models were developed with an emphasis on the dynamics in the water column and the eventual water column concentrations. Several models, including some of those listed below (EXAMS-II, WASP4) are now capable of simulating water column-sediment interactions (resuspension, settling, and diffusion), however, additional work needs to be completed on the mechanisms of sediment-water column exchange before the models can be validated for predictive applications involving sediments. With the advent of sediment criteria in the next few years, it will be necessary to use models that predict concentrations in both receiving water and bed sediment. This will be of particular importance in areas where the sediments are contaminated to the point at which they act as the source of a pollutant to the water column. Table 4-2 lists and summarizes models that may be used for predicting the fate and transport of toxicants and that are supported by the EPA CEAM [56]. All the models, plus two bioaccumulation models, briefly are described below.

- DYNTOX [54] is a WLA model that uses a probabilistic dilution technique to estimate receiving water chemical concentrations or whole effluent toxicity fractions. The model considers dilution and net first-order loss, but not sorption and benthic exchange. The net loss rate must be determined empirically on a case-by-case basis and cannot be extrapolated to different conditions of flow, temperature, solids, pH, or light.

- EXAMS-II [57] is a compartment model that can be used as either a steady-state or quasi-dynamic model designed for evaluation of the behavior of synthetic organic chemicals in aquatic ecosystems. It simulates a toxic chemical and its transformation products using second-order kinetics for all significant organic chemical reactions. EXAMS-II does not simulate the solids with which the chemical interacts. The concentration of solids must be user-specified for each compartment. The model accounts for sorbed chemical transport based on solids concentrations and specified transport fields. Sediment exchanges with the water column include pore-water advection, pore-water diffusion, and solids mixing. The last describes a net steady-state exchange associated with solids that is proportional to pore-water diffusion.

- WASP4 [58] is a generalized modeling framework for contaminant fate in surface waters. Based on the flexible compartment modeling approach, WASP4 can be applied in one, two, or three dimensions, given the transport of fluxes between segments. WASP4 can read output files from the link-node hydrodynamic model DYNHYD4, which predicts unsteady flow rates in unstratified rivers and estuaries, given variable tides, wind, and inflow. TOXI4, a subset of WASP4, simulates up to three interacting toxic chemicals and up to three sediment size fractions in the bed and overlying waters. First- or second-order kinetics can be used for all significant organic chemical reactions. Sediment exchanges include pore-water advection, pore-water diffusion, and deposition/scour. Net sedimentation and burial rates can be specified or calculated. The output can be used with the two bioaccumulation models FGETS and FCM2, which are described below.

- HSPF [59] simulates watershed hydrology and water quality for both conventional and toxic organic pollutants. HSPF incorporates the watershed-scale ARM and NPS models into a basin-scale analysis framework that includes transport and transformation in one-dimensional stream channels. The simulation provides a time history of the runoff flow rate, sediment load, and nutrient and pesticide concentrations, along with a time history of water quantity and quality at any point in a watershed. HSPF simulates three sediment types (sand, silt, and clay) in addition to specific

Table 4-2. Toxicant Fate and Transport Models

Model	Environment	Time Domain	Spatial Domain	Chemical
DYNTOX	river	dynamic	far field, 1-dimensional	organic, metal
EXAMS-II	lake, river, estuary	steady-state, quasi-dynamic	far field, 3-dimensional	organic
WASP4	lake, river, estuary	steady-state, dynamic	far field, 3-dimensional	organic, metal
HSPF	river	dynamic	far field 1-dimensional	organic, metal
SARAH2	river	steady-state	treatment plant, near field, 2-dimensional	organic
MINTEQA2	lake, river, estuary	steady-state	—	metal

organic chemicals and transformation products of those chemicals. The reaction and transfer processes included are hydrolysis, oxidation, photolysis, biodegradation, volatilization, and sorption. Sorption is modeled as a first-order kinetic process in which a desorption rate and an equilibrium partition coefficient for each of the three solid types must be specified. Resuspension and settling of silts and clays (cohesive solids) are defined in terms of shear stress at the sediment-water interface. For sands, the system's capacity to transport sand at a particular flow is calculated and resuspension or settling is defined by the difference between the sand in suspension and the calculated capacity. Sediment exchanges with surficial benthic sediments are modeled as sorption/desorption and deposition/scour. Underlying sediment and pore water are not modeled.

- SARAH2 [60] is a steady-state, near-field model for calculating acceptable concentrations of hazardous organic chemicals discharged to land disposal or wastewater treatment facilities. Acceptable leachate or treated industrial waste discharge constituent concentrations are estimated by a "back calculation" procedure starting from chemical safety criteria in surface water, drinking water, or fish. For steady or batch waste streams, SARAH2 considers the following concentration reductions: dilution and loss during treatment, initial Gaussian mixing at the edge of a stream, lateral and longitudinal diffusion in the mixing zone, sorption, volatilization, hydrolysis, and bioaccumulation in fish. The user must specify appropriate concentrations for protection of the aquatic community and of humans exposed through consumption of fish and water. The benthic community is not presently considered. Treatment loss is handled empirically. SARAH2 contains data sets for three disposal-watershed scenarios that can be easily modified and employed. The model is designed for screening analysis and contains numerous assumptions that should be verified before the model is used in actual cases.

- MINTEQA2 is an equilibrium metals speciation model for dilute aqueous systems [61]. It does not have any transport and transformation processes and must be run with one of the above models. It can be used to calculate the mass distribution at equilibrium among dissolved, absorbed, and solid phases and the species distribution within each phase. MINTEQA2 contains a chemical component data set for major ions commonly found in aqueous systems (e.g., Ca, Fe, and S), trace metals/metalloids of pollution interest (e.g., Cd, Cr, Ni, Pb, and Zn), and organic ligands of significant affinity for metal complexation. The model can be used to calculate the concentrations of adsorbed metals via any of seven different adsorption algorithms.

- FGETS is a toxicokinetic model that simulates the bioaccumulation of nonpolar organic chemicals by fish from both water and food [62]. Both of these routes of exchange are modeled as diffusion processes that depend upon physicochemical properties of the pollutant and morphological/physiological characteristics of the fish. FGETS contains a moderately sized data base of allometric relationships for gill morphology with which it can simulate the direct gill/water exchange of organic chemicals for essentially any fish species, assuming certain default values. FGETS also contains a limited data base of physiological/morphological relationships that are used to set parameters for food exchange. In addition to simulating bioaccumulation of organic toxicants, FGETS can calculate time to death from chemicals whose mode of action is narcosis. This calculation is based on the existence of a single, lethal, internal chemical activity for such chemicals. The concentrations of toxic chemical to which the food chain is exposed may be specified by the user or may be taken directly from the values calculated by the exposure concentration model WASP4. Thus FGETS may be executed as a separate model or as a postprocessor to WASP4.

- FCM2 is a generalized model of the uptake and elimination of toxic chemicals by aquatic organisms [63]. It generates a mass balance calculation in which the rates of uptake and elimination are related to the bioenergetic parameters of the species. A linear food chain or a food web may be specified. Fish tissue concentrations are calculated as a function of time and age for each species included. Exposure to the toxic chemical in food is based on a consumption rate and predator-prey relationships that are specified as a function of age. Exposure to the toxic chemical in water is functionally related to the respiration rate. Steady-state concentrations also may be calculated. The concentrations of the toxic chemical to which the food chain is exposed may be specified by the user or may be taken directly from the values calculated by the exposure concentration model WASP4. Thus FCM2 may be executed as a separate model or as a postprocessor to WASP4. Migratory species, as well as nonmigratory species, may be considered. Separate nonmigratory food chains may be specified, and the migratory species is exposed sequentially to each food chain based on its seasonal movements.

4.5.5 Effluent Toxicity Modeling

To apply the steady-state, continuous simulation, or probabilistic methods to effluent toxicity modeling, the percent effluent measurements should be converted to toxic units (TUs). As discussed in Chapters 1, 2, and 3, it is necessary to convert toxicity to units that can be directly related to mass. When comparing toxicity among chemicals, the relationship between toxicity and concentration is inverse; chemicals that have toxic effects at low concentrations have a greater "toxicity" than chemicals that have toxic effects at higher concentrations. The modeling of toxic effluents is based on mass balance principles; therefore, toxicity needs to be in units that increase when the percent of the effluent of the receiving stream increases. Thus, a TU is the reciprocal of the dilution that produces the test endpoint, i.e., acute toxicity endpoint (ATE) or chronic toxicity endpoint (CTE). An acute toxic unit (TU_a) is the reciprocal of an ATE. A chronic toxic unit (TU_c) is the reciprocal of a CTE. The TMDL must ensure that the CMC and the CCC are met in the receiving water at the desired duration and frequency. The CMC for toxicity is recommended as 0.3 TU_a. This is a value that should prevent lethality unless the duration of exposure exceeds 1 hour.

The CCC for toxicity measured with chronic tests is recommended as the following:

$$CCC = 1.0\ TU_c.$$

The first step in the TMDL process is to calculate the allowable acute effluent toxicity that meets the CMC in the receiving water at the duration and frequency discussed in Appendix D.

The next step in the TMDL process is to calculate the allowable chronic effluent toxicity that meets the CCC in the receiving water at the duration and frequency discussed in Appendix D. To compare the allowable acute toxicity value to the allowable chronic toxicity value, the numbers must be converted to the same units as follows:

$$TU_a = (ACR)(TU_c)$$

where the acute-to-chronic ratio (ACR) is determined from tests on the effluent. It is important that the ACR used for TMDL purposes be based on actual data and not be assumed to be 10 or 20, as in the screening procedure (Chapter 3). The value of this ratio will influence whether the acute or chronic TMDL is more stringent and is used to calculate the permit limit using the methods described in Chapter 5.

At the present time, the fate of effluent toxicity in a receiving water is not fully understood. Even if a decay rate for toxicity can be measured on a given day in a site-specific situation, there is no way as yet to know how this rate is affected by temperature, pH, or other environmental conditions. There is also no way to know how this rate may change when new treatment is installed. Instream measurements of toxicity should be made at least once per season to identify any time-varying trends in site-specific fate processes. These monitored decay rates can then be used in steady-state or continuous simulation fate and transport models to predict receiving water toxicity, assuming that the rates will not change with future treatment.

Without specific information concerning the persistence of toxicity, it is recommended that effluent toxicity be limited to dilution estimates and that toxicity be assumed to be additive and conservative. Toxicity is expected to be additive even when the toxicity of one effluent affects selected biota while the toxicity of a downstream discharge affects different biota. For rivers and run-of-river reservoirs with a detention time of less than 20 days, the following dilution equation should be used, assuming completely mixed conditions:

$$C = \frac{C_s Q_s + C_e Q_e}{Q_e + Q_s}$$

where

- C = downstream concentration (TU_c or TU_a)
- C_s = upstream concentration (TU_c or TU_a)
- Q_s = upstream flow (cfs)
- C_e = effluent concentration (TU_c or TU_a) and
- Q_e = effluent flow (cfs).

For multiple dischargers, this equation must be applied sequentially to find the concentration as a function of distance downstream. The equation can be used for a steady-state analysis if Q_s is set equal to the design flow, Q_e is set equal to the historical plant flow, and C_e is calculated to meet the CMC and CCC. This equation can also be used with the continuous simulation, log-normal probabilistic, or Monte Carlo methods. For these dynamic analyses, a series of C_e, Q_e, C_s, and Q_s values would be used.

If instream toxicity measurements are available and a first-order decay rate for toxicity can be estimated, the following equation should be used:

$$C = C_o e^{-K(x/u)}$$

where

- C = downstream concentration (TU_c or TU_a)
- C_o = concentration after the point source discharge has mixed completely with the river (TU_c or TU_a)
- x = distance downstream of complete mix point
- u = velocity of river
- K = measured decay rate.

Additional statistical approaches are available that might provide better statistical fits to the available data. However, these models are somewhat more limited than the example provided above.

The same equations used for toxicity analyses in **rivers** can also be used in steady-state, continuous simulation, or probabilistic analysis of long, narrow, shallow impoundments with high inflow velocities. **Wider, deeper lakes** require more complicated analyses since prolonged detention times (>20 days) and stratification exert a significant impact on water quality. The prolonged detention times make it essential that receiving water measurements of toxicity be available to estimate decay factors. These measurements should be made at least once per season to identify any time-varying trends in toxicity fate processes. Steady-state or continuous simulation fate and transport models for lakes can then be run with monitored decay rates for toxicity. A simple steady-state analysis can be performed using the following equations [64]:

$$T_w = V/Q$$
$$C = C_{in}/(1+T_w K)$$

where

- T_w = mean hydraulic residence time
- V = lake volume at design conditions
- Q = mean total inflow rate at design conditions
- C = steady-state lake concentration (TU_c or TU_a)
- C_{in} = steady-state inflow concentration (TU_c or TU_a)
- K = first-order decay rate.

If effluent is discharged into a stratified lake and mixes only with the hypolimnion or epilimnion, the volume of the layer should be used only to calculate mean hydraulic residence time (T_w). The mean total inflow rate (Q) and the inflow concentration (C_{in}) should be calculated as the sum of all sources to the lake, including point source, nonpoint source, and tributary inputs.

Dilution calculations for effluent toxicity discharges to an **estuary** are complicated by the oscillatory motion of the tides and possible stratification of the estuary. The prolonged detention times make it essential that field measurements of toxicity be available to estimate decay factors. These measurements should be made at least once per season to identify any time-varying trends in toxicity rate processes. Steady-state or continuous simulation fate and transport models for estuaries can then be run with monitored decay rates for toxicity. A simple steady-state analysis can be performed using the following equations for each nonconservative pollutant entering from the river at the head of an estuary [64]:

$$C_i = C_{i-1} \frac{(f_i)}{(f_{i-1})} B_i$$

where

$$B_i = \frac{r_i}{1 - (1-r_i)e^{-kt}}$$

- r_i = exchange ratio for segment i as defined by modified tidal prism method
- t = flushing time
- f_i = fraction of freshwater in segment i
- C_i = nonconservative pollutant concentration in segment i (TU_a or TU_c)
- k = decay rate of pollutant.

The following equations should be used for each nonconservative pollutant entering along the side of an estuary:

For segments downstream of outfall:

$$C_i = C_o \prod_{i=1}^{n} \frac{f_i}{f_o} \frac{r_i}{1 - (1-r_i)e^{-kt}}$$

For segments upstream of outfall:

$$C_i = C_o \prod_{i=1}^{n} \frac{S_i}{S_o} \frac{r_i}{1 - (1-r_i)e^{-kt}}$$

where

- C_i = nonconservative pollutant mean concentration in segment i (TU_c or TU_a)
- C_o = nonconservative pollutant mean concentration in segment of discharge
- r_i = exchange ratio for segment i as defined by the modified tidal prism method
- n = number of segment away from outfall
- f_i = fraction of freshwater in segment i
- f_o = fraction of freshwater in segment with discharge
- S_i = salinity in segment i
- S_o = salinity in segment of discharge
- k = decay rate
- t = flushing time.

The details of how to calculate exchange ratios and flushing times for estuaries are included in Part 2 of EPA's water quality assessment manual [64]. This manual also describes how to perform these calculations for stratified estuaries using a two-dimensional box model analysis.

4.6 HUMAN HEALTH

4.6.1 Human Health Considerations

Human exposure to pollutants should be evaluated as completely as available information will allow. Exposure information is used in calculating the human health reference ambient concentration (RAC) from the formulas in Chapter 2, Water Quality Standards. This information should be used to estimate exposures due to fish consumption and drinking water ingestion, background concentrations, and other exposure routes, such as recreational, occupational, drinking water, dietary (other than fish), and inhalation. Factors in the formulas for which information is not available can be omitted from the calculation. If States choose, bioaccumulation factors also can be modified.

4.6.2 Determining the TMDL Based on Human Health Toxicants

TMDLs are typically necessary only where mixing is allowed. Mixing zones are used at the discretion of the States. If a State does not allow a mixing zone or the assumption of complete mixing, then the RAC is applied at the end of pipe and no TMDL determination is typically necessary.

With persistent or bioconcentratable pollutants, special mixing zone considerations apply. Bioconcentratable pollutant criteria exceedances within the mixing zone can potentially result in tissue contamination of organisms directly or indirectly through contamination of bed sediments with subsequent incorporation into the food chain. For discharge situations with incomplete mixing (e.g., large rivers, lakes, estuaries, oceans), States need to carefully consider whether mixing zones for persistent or bioconcentratable pollutants are appropriate. Where a mixing zone is allowed, one TMDL should be calculated to achieve the RAC or criterion selected above [65]. Because most human health criteria are chronic only, a TMDL to protect against acute effects will usually not be needed, although EPA's Office of Drinking Water does have acute criteria for some pollutants.

For the purpose of the following discussion, use of simple, steady-state dilution models is assumed. However, these models may be inappropriate for certain situations where sediments serve as a sink for bioconcentratable pollutants and where additional factors need to be considered. Dynamic models, where available, are useful tools for accounting for an array of variables that may have an impact on the fate of bioconcentratable pollutants in the food chain. These models may be used by States for surface waters in appropriate instances.

In simple situations, the TMDL is determined from the RAC and the design flow of the receiving water. In more complicated situations, e.g., where mixing is not rapid or where lakes or estuaries are involved, a spatial averaging scale must be chosen. Selection of the spatial scale must be consistent with reasonable assumptions about the behavior of aquatic organisms and the target human population.

In some cases, it may be necessary to apply the chronic human health criterion within a mixing zone if it is reasonable to assume that the bioconcentrating aquatic organisms have little mobility, thus spending most of their time within the mixing zone; and the target human population consistently consumes fish from the mixing zone (over a 70-year lifetime, for carcinogenic risks).

The procedure for developing TMDLs/WLAs generally requires determining values for the following parameters, based upon water quality considerations: (1) the duration of the averaging period applicable to the WLA; (2) design considerations, e.g., flow; (3) the discharge (WLA) concentration that will result in meeting the ambient water quality criterion during the design condition; and (4) the allowable probability (or frequency) of the discharge's exceeding the WLA, averaged over the appropriate

duration. The technical basis for setting these values is discussed in the following sections.

1) Averaging Periods

The duration of the averaging period for the WLA should be selected to be consistent with the assumptions used to derive the water quality criteria. Two categories of pollutants should be recognized: carcinogens and noncarcinogens.

The human health criteria for carcinogens are derived assuming lifetime exposure. The upper-bound risk is directly proportional to the lifetime arithmetic mean dose. The criteria thus apply to the ambient water concentrations averaged over a 70-year period.

The duration of exposure assumed in deriving criteria for noncarcinogens may be ambiguous, particularly where a criterion is derived from animal studies. Furthermore, the duration may be highly variable, ranging as high as 20 to 30 years for cadmium.

2) Dilution Design Conditions

a) Carcinogens: River and Stream Discharge Situations

In well-mixed situations, the RWC, C, is determined by the pollutant load, W (mass/time), and the combined receiving water plus effluent flow, Q, such that, C = W/Q.

The long-term harmonic mean flow is recommended as the design flow for carcinogens. The recommendation of long-term harmonic mean flow has been derived from the definition of the human health criteria (HHC) for carcinogenic pollutants. The adverse impact of carcinogenic pollutants is estimated in terms of receptors (human) lifetime intakes. To be within the acceptable level of life-time body-burden of any carcinogen, such intakes should not exceed the HHC during the average life-time of the receptor. A life-time for exposure to carcinogenic pollutants is defined as 70 years, or approximately 365 (days/year) multiplied by 70 years.

The HHC for carcinogenic pollutants can be numerically expressed as:

$$HHC = C\ (design) = (C_1 + C_2 + C_3 + \text{-----} + C_n)/n$$

where

n = (365 days/year) x 70 years
C = concentrations

Based on an assumption of a constant daily load from a treatment facility, the fully mixed instream concentration will go up or down inversely with the ups and downs of receiving water flows. Therefore, instream concentration is a function of, and inversely proportional to, the streamflow downstream of the discharge. Using this concept, 1/Q can be substituted for C, as follows:

$$1/Q\ (design) = (1/Q_1 + 1/Q_2 + 1/Q_3 + \text{-----} + 1/Q_n)/n.$$

The stream design flow (Q design) can then be shown as follows:

$$Q\ (design) = n/(1/Q_1 + 1/Q_2 + 1/Q_3 + \text{-----} + 1/Q_n)$$

The harmonic mean is expressed as follows:

$$Q\ (design) = n/\sum_{i=1}^{n} (1/Q_i)$$

where

n = the number of recorded flows.

The harmonic mean is always less than the arithmetic mean. The harmonic mean is the appropriate design flow for determining long-term exposures using steady-state modeling of effluents. The arithmetic mean flow is not appropriate as the design flow since it overstates the dilution available. Extreme value statistics (such as 7Q10 or 30Q5) are also not appropriate since they have no consistent relationship with the long-term mean dilution. However, for situations involving seasonably variable effluent discharge rates, hold-and-release treatment systems, and effluent-dominated sites, the harmonic mean may not be appropriate. In these cases, the effluent load and downstream flow are not independent (i.e., they are correlated). Modeling techniques that can calculate an average daily concentration over a long period of time are more appropriate to determine the long-term exposure in these cases.

The harmonic mean flow may be estimated by any of several methods [8], assuming that flows are approximately lognormally distributed:

$$Q_{hm} = \frac{Q_{gm}^2}{Q_{am}}$$

where

Q_{gm} is the geometric mean flow
Q_{am} is the arithmetic mean flow.

For U.S. Geological Survey flow records, summaries of the statistical parameters needed to estimate the harmonic mean can be quickly obtained from STORET, through a user-friendly procedure for permit writers, as described in Appendix D.

WQAB DFLOW is a software package available for computation of harmonic mean flow. The DFLOW program (as discussed below and described in Appendix D) should be used with data that are not lognormally distributed.

To develop some quantitative sense of how a long-term harmonic mean flow of any stream compares with its 7Q10 flow, the Assessment and Watershed Protection Division and the Risk Reduction Engineering Laboratory at Cincinnati, Ohio, analyzed flow records of 60 streams selected at random throughout the United States. These are the same stream flow records that had been analyzed for stream design flow condition for aquatic life protection as listed in EPA guidance [8]. Based on the long-term harmonic flow and 7-day, 10-year low-flow estimates for these 60 streams, the long-term harmonic mean flows of all 60 streams were equal to or greater than two times the 7Q10 low flow. Fifty-four of the streams' harmonic mean flows were equal to or greater than 2.5 times their 7Q10 low flows. Finally, 40 of the 60 streams' harmonic mean flows were equal to or greater than 3.5 times the 7Q10.

Based on the above observations, permit authorities may choose a multiplication factor of 3 x 7Q10 to estimate stream design flow for human health protection for carcinogenic pollutants. How-

ever, it is recommended that the harmonic mean flow be calculated directly from the historical daily flow record, if possible. Alternatively, the following equation might be used to estimate harmonic mean flow [66]:

$$Q_{hm} = [1.194 * (Q_{am})^{0.473}] * [(7Q10)^{0.552}], r^2 = 0.99.$$

In this equation, Q_{am} and 7Q10 are estimated using the U.S. Geological Survey computer program, FLOSTAT.

b) Noncarcinogens: River and Stream Discharge Situations

The choice of average period represents a level-of-protection consideration inherent in the risk management decision to be made by the permitting agency. If a short-term duration of exposure is chosen (i.e., 90 days or less), design flows may be appropriately based on extreme value statistics. Because the effects from noncarcinogens are more often associated with shortened exposures, EPA suggests the use of 30Q5. However, in the comparisons of flows for smaller rivers (i.e., low flow of 50 cfs), the 30Q5 flow was, on the average, only 1.1 times that of the 7Q10. For larger rivers (i.e., low flow of 600 cfs), the factor was, on the average, 1.4 times. **If the effects from certain noncarcinogens are manifested after a lifetime of exposure, then a harmonic mean flow may be appropriate.**

3) Point of Application of the Criteria

The point at which the chronic criteria are to be met in the receiving water may be fixed by existing State standards or may be determined by considerations for managing individual and aggregate risks. The several possibilities include the following:

- Where State standards allow no mixing zone and no spatial averaging, the criterion would be met at the end of the pipe.
- Where State standards specify that the criterion must be met at the end of the mixing zone, the criterion would be applied at that point.
- Where State standards allow consideration of spatial averaging, the criterion may be met as an average within a specified area, as appropriate for the individual and aggregate risk scenarios underlying the application.

CHAPTER 4

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