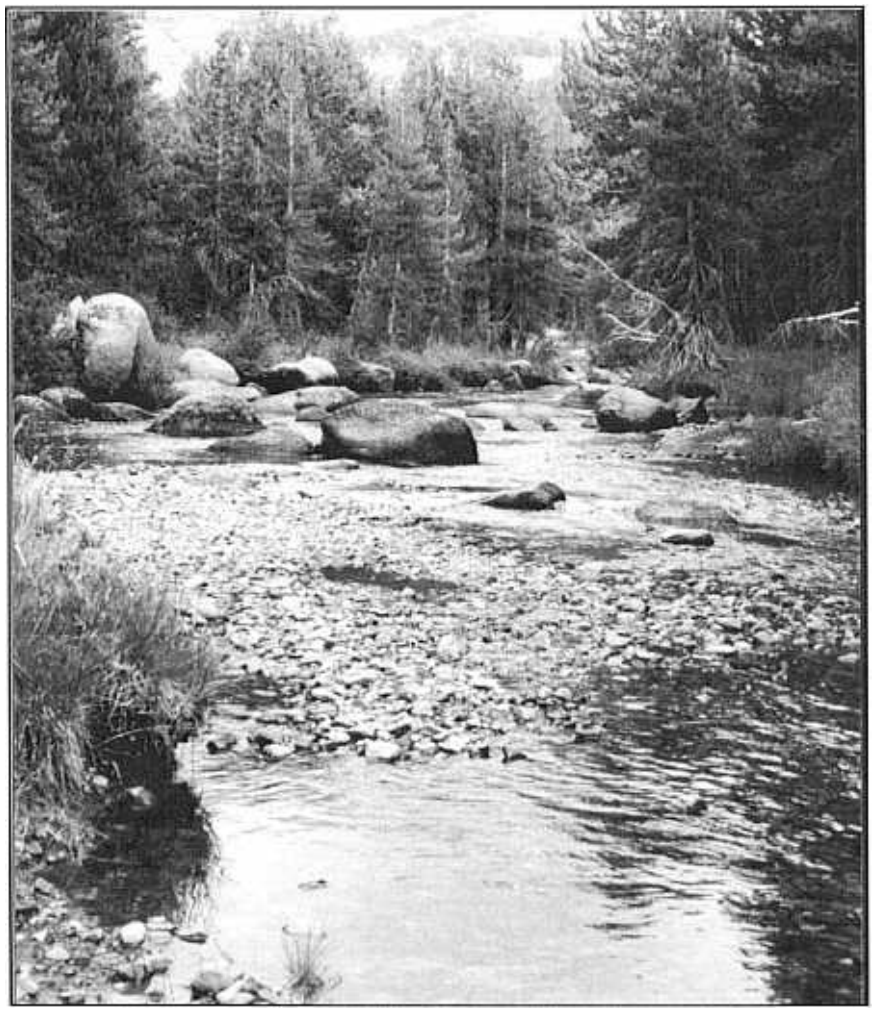
DRAFT

COMPILATION OF EXISTING GUIDANCE FOR THE DEVELOPMENT OF SITE-SPECIFIC WATER QUALITY OBJECTIVES IN THE STATE OF CALIFORNIA



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STATE WATER RESOURCES CONTROL BOARD
CALIFORNIA ENVIRONMENTAL PROTECTION AGENCY

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Prepared by:
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EXECUTIVE SUMMARY

This document provides the State of California, United States Environmental Protection Agency (USEPA) Region 9, other stakeholders and the general public with a compilation of existing guidance and information on the development of site-specific water quality objectives (SSOs). The document presents background information on default national aquatic life criteria, including their strengths and limitations. Published and unpublished methodologies for the derivation of site-specific objectives are described with links, references and case examples provided for greater detail. A section near the beginning of the document discusses site characteristics that are important to consider prior to granting approval for a study to modify an objective, as well as factors to consider if a site-specific objective is approved.

This document is for information purposes only, and is not intended to set policy. This document is a draft document because it will evolve as more information on preparing SSOs in California is developed. This document does not supersede and should be used in coordination with the State of California's Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California (SIP), and other State laws, polices and guidance documents.

Considerations in the Development of Site-specific Water Quality Objectives

Before such studies are initiated, a number of factors need to be considered in order to determine whether a site-specific objective is appropriate. Factors such as site definition and boundaries, the presence of endangered species, water quality characteristics of effluent dominated streams, and multiple site-specific objectives, require a careful evaluation of the anticipated levels of contaminants a site-specific objective may allow. Additional study to identify a cause or remedy a problem is needed before considering an SSO when a site is found to have ambient toxicity, an impaired in-stream biological community, or unacceptable whole effluent toxicity (WET). Follow-up monitoring should occur following the approval of a site-specific objective. This may include more rigorous monitoring when there is a greater uncertainty regarding the protection of the beneficial use of the water body, including downstream uses. Such monitoring could include more rigorous ambient and WET toxicity testing, in-stream biomonitoring, and monitoring of toxicant concentrations in biota and sediments.

Water quality standards programs incorporate antidegradation considerations in the development of site-specific water quality objectives. The concept of antidegradation can impact both the development of site-specific objectives as well as their application in NPDES permits. An antidegradation review considers whether lowering water quality is necessary to allow important economic or social development. The review evaluates whether all legal requirements for new and existing point sources and best management practices for nonpoint sources are achieved.

When undertaking a site-specific objective modification, it is critical that the State identify those interested parties (environmental, industrial, or governmental) that are likely to have an interest in the particular water body for which the site-specific objective is being considered. The primary Federal agencies that need to be contacted are the Region 9 Office of USEPA, and the appropriate field offices of the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS). All parties should be involved early in the process, remain informed of all anticipated economic and environmental impacts, and have access to the study data and discussion of data uncertainties.

Aquatic Life Based Criteria

Recommended national water quality criteria for the protection of aquatic life are based upon available toxicity data for any species that has reproducing wild populations in North America. Criteria are based upon a required minimum dataset of Genus Mean Acute Values that are calculated from all acceptable data for species within a genus. A rigorous set of guidelines is followed to derive criteria that reflect both short-term acute and long-term chronic protection. Because water quality criteria are primarily derived from studies in which organisms are exposed to a single contaminant in laboratory water, there are a number of site-specific conditions that can affect the toxicity of the contaminant which may cause a criterion to be over- or under-protective.

Site-Specific Objectives

Site-specific objectives adjust water quality objectives to account for their over- and under-protectiveness by using site-specific information and applicable Federal and State of California scientific guidance. Three USEPA published procedures and a number of other procedures allowed by USEPA based on proven scientific principles can be used to establish site-specific objectives. Of these procedures the most common is, the Water-effect Ratio (WER) Procedure which adjusts objectives to account for a site's water chemistry, followed by the Recalculation Procedure which adjusts objectives on the basis of the assemblage of species found in a particular site water, and the Resident Species Procedure which accounts for site water chemistry and the assemblage of resident species.

Water-effect Ratio Procedure

The waters effect ratio is the ratio of the toxicity of a chemical in site water to the chemical's toxicity in laboratory water. This procedure is commonly used when it is suspected that either chemical or physical aspects of receiving water or effluent (or both) will cause a pollutant to be less bioavailable and, therefore, less toxic. A minimum of three sets of side-by-side tests are conducted in which the toxicity of a contaminant is determined in site water and laboratory water. The effect level of the test determined in the site water divided by the effect level for the laboratory water is the water effect ratio. The Water-effect Ratio Procedure results in a multiplier which is applied to an existing water quality objective. If a chemical is less toxic in a site water, the multiplier is >1 and results in a higher objective; if a chemical is more toxic in a site water, the multiplier is <1 and results in a lower objective.

Recalculation Procedure

A water quality objective can be derived utilizing data for any North American aquatic species. It is possible that the level of protection provided by the suite of species in the national dataset is inappropriate for the species found at a particular site. The goal of the Recalculation Procedure is to eliminate from the database those taxa that are not resident (and not expected to be present) in the site waters, while keeping in the database resident species and those non-resident species that serve as toxicological surrogates for taxonomically related resident species for which no toxicological data are available. The end result of the Recalculation Procedure is that the remaining data are more representative of the sensitivities of species found at the site. The recalculation procedure consists of a systematic stepwise process which deletes (or adds) species

from the dataset following a set of stringent guidelines. A site-specific objective is "recalculated" from the adjusted dataset using the same procedure described in the national aquatic life guidelines.

Resident Species Procedure

The Resident Species Procedure accounts for site-specific conditions by testing the toxicity of the chemical of interest to resident species in site water. A site-specific objective is then calculated using the toxicity values and the procedure described in the national aquatic life guidelines. The over-riding problem with the Resident Species Procedure is that the number of genera in the generated dataset is usually so low (because of the cost of generating a robust dataset) that the statistical computation typically produces a criterion that is below that needed to protect the most sensitive species tested. For this reason, it is the least-used procedure for determining a site-specific objective.

Chemical Translator

Although not a type of site-specific objective, another approach which addresses the issue of the bioavailability of a chemical as do the WER and Resident Species Procedures is the chemical translator. In contrast to the WER and Resident Species Procedures, the chemical translator does not modify the water quality objective directly, but rather it translates the site-specific dissolved form of the objective to the total recoverable form. A chemical translator can be empirically determined by relating the dissolved metal concentration (operationally defined as the metal which passes through a 0.45 µm or a 0.40 µm filter) to the total recoverable concentration. Alternatively, the fraction of dissolved metal is derived using a partition coefficient. In this case, the coefficient is determined as a function of total suspended solids (although some other basis such as humic substances or particulate organic carbon may be used).

Other Site-specific Aquatic Life Objective Procedures

Two other alternative approaches are provided in this document for making site-specific modifications. The first approach is a two-step procedure to evaluate the possibility of changing the number of times a water quality objective can be exceeded over a three-year period by first assessing impairment, followed by an analysis of the historical concentrations of the chemical of interest. The second approach uses natural background concentrations. Where appropriate, USEPA recommends that site-specific objectives for aquatic life protection be derived using natural background concentrations of the toxic chemicals.

In contrast to the water-based value recommended in 1986, USEPA plans to recommend a revised chronic water quality objective for selenium which will be a tissue-based value designed to protect consumers of aquatic organisms including fish and wildlife. Fish are recognized in the literature as the group of organisms most sensitive to selenium. At the time of preparation of this document, USEPA had not yet published the Agency's recommendations for how the tissue-based objective will be implemented. Therefore, it is not feasible to recommend procedures for deriving a site-specific chronic tissue-based objective for selenium at this time. In addition to the USEPA's recommended national chronic criterion for selenium, an Inter-Agency task force is developing selenium water quality criteria for California watersheds. The goal of the criteria will be to protect aquatic ecosystems and their components, including wildlife and threatened and endangered species.

Endangered Species

To ensure compliance with the objectives of the Endangered Species Act and the California Endangered Species Act, special consideration must be given to listed species in the bodies where they are located. SSO development first requires identification of any listed species expected to reside at the site. References are provided in the document in which will aid in the research of the possible presence of listed species at and near the water body of interest. Once a listed species has been identified as a current or potential inhabitant in a specific water body or ecoregion, an analysis (biological evaluation) must be performed to determine if the SSO is likely to jeopardize the continued existence of that species. Several approaches are discussed including the use of surrogates to estimate the level of protection needed for a listed species.

Human Health-Based Objectives

Human health-based objectives are designed to protect humans from the adverse effects of pollutants. Site-specific criteria may be developed as long as the site-specific data, either toxicological or exposure-related, is justifiable. Examples of site-specific factors include site-specific fish consumption rate, bioaccumulation factors, and percent lipid in aquatic organisms. A methodology and case example are provided for the determination of a site-specific bioaccumulation factor.

In 2001, USEPA published a recommended fish tissue criterion of 0.3 mg methylmercury/kg to protect human health (USEPA 2001a). USEPA's preferred approach for relating a concentration of methymercury in fish tissue to a concentration of mercury in ambient water is to derive site-specific bioaccumulation factors based on water and fish collected in the water body of concern. Once a site-specific bioaccumulation factor has been determined for methylmercury, a dissolved site-specific objective can be calculated by dividing the tissue-based objective by the bioaccumulation factor. [*Note: At the time of this document publication, USEPA and California have not decided on a water quality objective for mercury. The information in this document presents USEPA's published recommendation for methylmercury. California may decide to use another approach*.]

Other Objectives and Guidelines

Wildlife objectives, sediment guidelines, and nutrient-based objectives are also effects-based concentrations (potentially influenced by reference conditions and antidegradation considerations) designed to restore and maintain the integrity of surface waters. A wildlife objective is a chemical concentration (in water, or in water or tissues), which, if not exceeded, will protect mammals and birds from the adverse impacts of chemical ingestion via food and/or water consumption. The methodology to derive wildlife objectives is similar to those for human health.

Sediment guidelines are designed to protect aquatic organisms from contaminants in sediments. It is anticipated that USEPA will publish sediment quality benchmarks which rely on equilibrium partitioning theory (percentage of a chemical which is actually biologically available in sediments or sediment interstitial water) to derive an effect level. The development of nutrient criteria incorporates the fundamental concepts of site-specific criteria. USEPA has relied on reference sites as the basis for establishing ecoregional nutrient criteria in an attempt to represent the physical, chemical, and biological

characteristics of the region.

LIST OF ACRONYMS

ACR = Acute-Chronic Ratios
AEPSC = American Electric Power Service Corporation
ASTM = American Society for Testing and Materials
AWQC = Ambient Water Quality Criteria
BAF = Bioaccumulation factors
BCC = Bioaccumulative Chemicals of Concern
BCF = Bioconcentration Factor
BSAF = Biota-Sediment Accumulation Factor
CABW= California Aquatic Bioassessment Workgroup
CART = Classification and Regression Trees
CCC = Criterion Continuous Concentration
CDFG = California Department of Fish and Game
CEQA = California Environmental Quality Act
CESA = California Endangered Species Act
CFR= Code of Federal Regulations
CMC = Criteria Maximum Concentration
CV = Coefficient of Variation
CWA = Clean Water Act
DFG = Department of Fish and Game
DOC = Dissolved Organic Carbon
EC_{50} = Concentration causing an effect to 50 percent of test population
EqP = Equilibrium Partitioning Theory
ESA = Endangered Species Act
ESB = Partitioning-Based Sediment Benchmarks
FACR = Final Acute-Chronic Ratio
FAV = Final Acute Value
FCM = Food-Chain Multiplier
FCV = Final Chronic Value
FWER= Final Water Effect Ratio
GM = Geometric Mean
GMAV= Genus Mean Acute Values
Guidelines = *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their Uses*
hWER= Highest WER
ICM = Interspecies Correlation Model
IRIS = Integrated Risk Information System
Kow = Chemical's Octanol-Water Partition Coefficient
KPCo = Kentucky Power Company
LC_{50} = Concentration causing lethal to 50 percent of test population
LDC = Least Disturbed Condition
LOAEL = Lowest Observed Adverse Effect Level
LOEC = Lowest Observed Effect Concentration
MDC = Minimally Disturbed Condition
MDR = Minimum Data Requirements

MHW = Moderately Hard Reconstituted Water
MOA = Memorandum of Agreement
MW = Megawatts
NMFS = National Marine Fisheries Service
NOAEL = No Observed Adverse Effect Level
NOEC = No Observed Effect Concentrations
NPDES = National Pollutant Discharge Elimination System
PAH = Polycyclic Aromatic Hydrocarbon
SMAV = Species Mean Acute Value
SMCV = Species Mean Chronic Value
SOP = Standard Operating Procedure
SQAG = Sediment Quality Assessment Guidelines
SRW= Soft Reconstituted Water
SSO = Site-Specific Objectives
TDR = Total-To-Dissolved Ratio
The Services = Field Offices of National Marine Fisheries Service and U.S. Fish and Wildlife Service
TIE = Toxicity Identification Evaluation
TL = Trophic Level
TMDL = Total Maximum Daily Load
TOC = Total Organic Carbon
TSD = Technical Support Document
TSS = Total Suspended Solids
TTD = Time-To-Death
USEPA = United States Environmental Protection Agency
USFWS = U.S. Fish and Wildlife Service
VDEQ = Virginia Department of Environmental Quality
WER= Water-effect Ratio
WET = Whole Effluent Toxicity
WLA = Waste Load Allocation
WQBEL = Water Quality-based Effluent Limit
WQC = National Water Quality Criteria
WQO = Water Quality Objective

1 INTRODUCTION

This guidance focuses on National Water Quality Criteria, site-specific water quality criteria, and site-specific water quality objectives. The terms "criteria" and "objective" can have slightly different regulatory connotations but, for purposes of this guidance, can be viewed as interchangeable, with "criteria" and "objective" applied to national and State of California waters respectively.

This document presents the standard approaches to develop site-specific objectives (SSO) using published USEPA guidance and generally discusses the conditions in which each is appropriate or not appropriate. In situations where site-specific criteria are appropriate, site-specific methods that evaluate the appropriateness of raising or lowering criteria are recommended and/or referred to. Case examples[1] are provided for select site-specific methods.

This document assembles guidance prepared by USEPA and other sources for use when developing site-specific objectives. It is a compendium of tools for use when considering and/or developing SSOs and brings together information that is available from many places into one place. While this document focuses primarily on the development of SSOs for toxic substances in surface waters, some information on the development of other criteria is included. **This document is for informational purposes only, and is not intended to set policy**. This document is a draft document because it will evolve as more information on preparing SSOs in California is developed. This document does not supersede and should be used in coordination with the State of California's *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (SIP), and other State laws, policies and guidance documents (e.g. the Basin Planning Amendment Process).

The State's Policy *for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California* (SIP) describes procedures to be used when developing SSOs. The SIP notes when it is appropriate to develop an SSO and what factors should be considered when initiating the development of an SSO. The SIP directs that Regional Water Quality Control Boards (RWQCBs) shall use scientifically defensible methods appropriate to the situation to derive the SSOs. Such methods may include USEPA-approved methods, such as the Water Effect Ratio Procedure, the Recalculation Procedure, and the Resident Species Procedure. Other methods that are scientifically defensible are allowed when appropriate. The complete text of Section 5.2 – Site-Specific Objectives of the SIP is included as Appendix A of this document.

The SIP also presents a flow chart (See Appendix B of this document), along with additional information, that may be used to guide the reader on when to embark on an SSO study.

[1] The case examples in the text and appendices of this document are summaries of site-specific studies that were conducted in several different states and are provided here as illustrations of approaches and methods. Because of regional environmental differences, their appearance in this document is not an endorsement of the methods for use in California.

2 CONSIDERATIONS IN THE DEVELOPMENT OF SITE-SPECIFIC WATER QUALITY OBJECTIVES

Prior to initiating or granting approval for an SSO study, there are several characteristics of the site that are important to consider. Among them are the toxicity history, recent biological and chemical assessments, and endangered species. Also, following the approval of an SSO, there may be a need for increased monitoring for effluent impacts to ensure full protection of beneficial uses. This section contains guidance on pre-study analysis of the site characteristics, and considerations for follow-up monitoring.

Endangered Species. The presence of endangered species does not preclude the opportunity to develop an SSO. The presence of endangered species does, however, add an extra burden of proof that any SSO will not cause either acute or chronic effects to the endangered species. Protection of in-stream species against direct toxicity will usually be determined by the magnitude of the difference between the SSO and the known toxicity of the chemical to either the endangered species, a specified surrogate species, or species in the several taxa most closely related to the endangered species (see Section 4, Endangered Species).

Because it is impossible to absolutely prove safety of any apparent no-effect concentration of a toxic chemical, evaluations of the SSO and the known toxic effect levels are necessary. The potential for subtle effects on behavior or biochemical processes that are not investigated in routine toxicity tests is real. Because of this possibility, a small difference between the SSO and contaminant concentrations likely to affect the endangered species should be considered reason to deny the SSO and require additional studies. However, the potential for subtle effects should not, a priori, be cause to deny the opportunity for an SSO when there is a large difference between the SSO and known concentrations likely to affect the endangered species. Although it isn't possible to quantitatively define "small" and "large," in most cases differences of 2x or less are considered small, and differences of 10x or more are considered large[2]. Such definitions should be based upon an analysis of available data relating the relative sensitivity of routine and non-routine endpoints, and, of course, such analysis should be as chemical-specific and species-specific as possible.

A Discharger's Whole Effluent Toxicity Limit is Exceeded. Based upon USEPA guidance, a site should not be considered for development of a site-specific objective using the Water-effect Ratio (WER) or Resident Species Procedure if the site is downstream of an effluent that routinely fails its whole-effluent toxicity (WET) testing requirements. The reason these approaches are not allowed in this case (even when the effluent toxicity is caused by a chemical other than the chemical for which the SSO is sought) is because the effluent matrix is likely to change after the toxicity is removed or reduced to an acceptable level. The change in the effluent matrix will likely affect the water-effect ratio; therefore, it is not appropriate to determine a WER while the effluent is unacceptably toxic. Note: An SSO may still be pursued using the Recalculation Procedure if there is unacceptable WET. However, it should be clearly demonstrated that the cause (in part or in whole) of the WET is not due to the chemical for which the SSO is sought.

[2] The definitions given for the terms "small" and "large" differences are provided for considerations but are not exact guidance.

Ambient Toxicity. California has an active program in which ambient water samples are collected and tested in the laboratory to evaluate acute and chronic toxicity to aquatic organisms. In those cases where an SSO is sought and the ambient water samples have been found to be toxic, additional work should be performed before beginning the SSO development process. It must be clearly demonstrated that the cause of toxicity in the ambient samples is not due (in part or in whole) to the chemical for which the SSO is desired.

Impaired Biological Community at the Site of Interest. A biological assessment of one or more communities (e.g., fish, macroinvertebrates, periphyton) at the site of interest may indicate that the site is impaired relative to a reference site or a reference condition. (See Section 3.10 for a discussion on bioassessment.) For a variety of reasons, the biological community at the site may be impaired, even if all of the water quality objectives have been attained. The impairment may be caused by toxics[3], elevated nutrients, increased temperature, invasive alien species, and/or habitat alteration; any of which may have point source, nonpoint source, and/or historical contamination origins. The relevance of an impaired biological community to the development of an SSO(s) is a complex issue that often does not have a simple means of resolution. The cause(s) and origin(s) of impairment can be very difficult to ascertain, and therefore it can be similarly difficult to assure that the chemical for which the SSO is being sought is not causing the impairment. If this is demonstrated, there may be management value in allowing an SSO in order to focus resources upon the source(s) of impairment rather than upon achieving a non-essential effluent goal.

However, before considering the development of an SSO, efforts should be made to associate the impairment with the source of stress. The Stressor Identification Guidance Document (EPA 822-B-00-025) describes the organization and analysis of available evidence to determine the cause of biological impairment. The guidance document suggests a sequence of activities from (1) reviewing available data, (2) forming possible scenarios that may explain the impairment, (3) evaluating the scenarios, to (4) deriving conclusions about the cause of impairment. The Stressor Identification Guidance Document can be downloaded at http://www.epa.gov/waterscience/biocriteria/stressors/stressorid.html.

As stated in Section 3.10, Biological Assessment, the State could approve initiating the site-specific development process if the site is found to be impaired, and the lines of evidence indicate the probable cause was not due to the chemical of interest. In this case, however, a requirement to conduct follow-up biological assessment should be added to ensure continued protection of the beneficial uses. If the site is found to be impaired, and lines of evidence indicate the probable cause was due in part or in whole to the chemical of interest, the State should disapprove the initiation of the site-specific development process.

Effluent Dominated Streams. There is no inherent reason that effluent dominated streams cannot be candidates for SSO proposals. Effluent dominance makes these streams likely candidates for discharger-initiated SSO consideration, due to both lack of dilution of potential toxic chemicals and the presence of concentrations of materials that could possible ameliorate toxicity e.g. dissolved organic carbon (DOC) and suspended solids. However, these same factors (lack of mixing/dilution) can expose aquatic life to high levels (approaching or equaling water quality objectives) of toxic substances without relief from eventual downstream dilution, such that one must carefully weigh these considerations from all

[3] Refers to toxics for which water quality objectives have not been developed, or the additive or synergistic effect of toxics at concentrations below water quality objective concentrations.

perspectives. More specifically, as discussed later in this document under Section 3.1.1.2 Important Considerations Related to Water Quality Criteria, effluent dominated waters, which lack significant effluent dilution, are also systems in which multiple contaminants can co-occur at elevated concentrations that approach the water quality objective levels. Because the toxicity of toxicants is often additive, this could result in the loss of sensitive species from effluent dominated waters. With regard to SSO development, special consideration should be given to effluent dominated streams to ensure protection from multiple contaminants.

The decision to proceed with an SSO for these waters should be guided by the same principles as for other waters, i.e.; WET test failure, ambient toxicity, endangered species, biological impairment, and persistent-chemical issues such as sediment quality and bioaccumulation (see sections related to these topics below).

Effluent dominated streams are frequently subjected to significant habitat modification that can influence which species reside at the site. Among the modifying factors not directly related to toxicity are greater hydraulic and thermal stability, scouring, sedimentation, and introduced species. The presence of these factors makes it difficult to select suitable reference sites for bioassessment and biocriteria. Care must be taken to sort out the various reasons for biological impairment which can be and often are the result of, multiple stressors including habitat modification and physical and chemical factors in the water column.

Application of the Resident Species Procedure at Sites with a Depauperate Fauna and Sites with Remnant Populations of Native Species

In the more recent description of the resident species procedure (*Water Quality Standards Handbook*, USEPA 1994a), the following statement is included: "If all the families at the site have been tested and the minimum dataset requirements have not been met, use the most sensitive resident family mean acute value as the site-specific Final Acute Value." This situation should be carefully implemented. The assumption that a mean acute value representing a family will fully protect an important species or "species of special concern" which belongs to this family should be evaluated as part of the SSO development.

Such situations are more apt to occur in intermittent or effluent dependent streams that contain a limited number of species. Some of these species, however, may be native species which are able to tolerate the altered conditions, and have lost habitat elsewhere due to dams or competition from non-natives in other reaches. These species may not be listed as endangered or threatened, but may be considered “species of special concern” and must be protected under the CWA. The reach of a stream targeted for an SSO may be important in a number of ways not immediately obvious without comprehensive study of the stream system, such as migration, juvenile habitat, or refugia during certain seasons or year types[4].

Several approaches can be used to derive an SSO for a site with a depauperate fauna in order to protect all resident species including any “species of special concern.” A protection factor can be applied to the

[4] Such a comprehensive study should also be conducted when the recalculation procedure is used to derive an SSO in a depauperate system believed to contain a “species of special concern.”

family mean value to protect the more sensitive species within the family. A second approach is to use a surrogate test(s) with a sensitive species in the family. (Note: the surrogate(s) would not need to occur at the site of interest.) The site-specific FAV in this case would be the effect value for the most sensitive species in the dataset. If the most sensitive family occurring at the site has more than one site species, data should be available for each species in that family, and the most sensitive species FAV should be used as the final FAV.

Multiple Site-specific Objectives. In situations where multiple SSOs are sought, additional efforts are recommended in order to assure protection of the beneficial uses. Since in all likelihood the chemicals of interest for SSOs have been either exceeding or close to exceeding the existing water quality objectives at the site, a bioassessment should be performed to determine if the site is biologically impaired. If the site is found to be biologically impaired, the above recommendations should be considered before any SSOs are permitted. If the bioassessment indicates no impairment and SSOs are developed, chronic toxicity tests should be conducted in the site water containing a mixture of all the chemicals for which SSOs were developed at the SSO concentration to demonstrate that there are no additive or synergistic effects.

Developing the Appropriate Type of Criteria. It is important that the appropriate type of criteria is developed for an SSO. For example, SSOs based on aquatic toxicity considerations would not be appropriate if the water body is designated for the MUN (Municipal and Domestic Supply) use and the proposed concentration to protect aquatic life is less stringent than the corresponding water quality criterion to protect drinking water.

Protection of Downstream Uses. The purpose of a water quality objective, including SSOs, is to protect the beneficial uses of the water body. In the case of SSOs, it is important to consider protecting the uses downstream of the site of interest for those chemicals that can have deleterious effects due to their persistent and/or bioaccumulative character. Selenium, for example, is a chemical for which the route of exposure for chronic toxicity is primarily via diet. Selenium bioconcentrates in trophic level 1 organisms to over 1000 times the concentration in the water, however selenium does not appear to biomagnify in the higher trophic levels. This characteristic of selenium accumulating in primary producers makes environments such as wetlands and backwaters particularly susceptible to selenium accumulation in fish, a group considered to be the most sensitive. Downstream uses also need to be considered for bioaccumulative inorganic (e.g., methylmercury) and organic chemicals. Depending on the food web structure and sediment characteristics of the site, elevated tissue residues can be found a considerable distance downstream of a point source discharge. These considerations can be addressed by increasing the range of the site to include sensitive habitats and the fate of persistent bioaccumulative chemicals.

Follow-up Monitoring. All sites for which SSOs have been developed should be part of a monitoring program to ensure the beneficial uses are being attained. Monitoring can consist of biomonitoring[5] to determine if the biological community is impaired relative to a reference condition, toxicological monitoring of water samples to see if the WER has changed, and/or chemical monitoring to document changes in water chemistry. The Interim WER Guidance (USEPA 1994b) offers the following considerations when using a WER.

[5] Depending on the chemical and/or site conditions, biomonitoring may include bioassessment of aquatic communities, wildlife and/or plant communities.

Periodic reevaluation of the WER.

a. WERs determined in upstream water take into account constituents contributed by point and nonpoint sources and natural runoff; thus a WER should be reevaluated whenever newly implemented controls or other changes substantially affect such factors as hardness, alkalinity, pH, suspended solids, organic carbon, or other toxic materials.

b. Most WERs determined by using downstream water are influenced more by the effluent than by the upstream water. Downstream WERs should be reevaluated whenever newly implemented controls or other changes might substantially impact the effluent, i.e., might impact the forms and concentrations of the chemical, hardness, alkalinity, pH, suspended solids, organic carbon, or other toxic materials. A special concern is the possibility of a shift from the discharge of a nontoxic form of the chemical to the discharge of a toxic form of the chemical, such that the concentration of the chemical does not increase; analytical chemistry might not detect the change that could be detected by toxicity tests.

Even if there have been no known changes, WERs should be reevaluated periodically. (The National Toxics Rule recommends that National Pollutant Discharge Elimination System (NPDES) permits include periodic determinations of WERs as a component of the monitoring requirements.) With advance planning, it should usually be possible to perform such reevaluations under conditions that are at least reasonably similar to those that control the permit limits (e.g., either design-flow or high-flow conditions), because those conditions should persist for a reasonable period of time. Periodic determination of WERs should be designed to answer questions, not just generate data.

Chemical Monitoring. Increased chemical monitoring of the upstream water, effluent, and/or downstream water, as appropriate, should be performed for water quality characteristics that may affect the toxicity of the chemical (e.g., hardness, alkalinity, pH, total organic carbon (TOC), and total suspended solids (TSS) to determine whether conditions change. The conditions at the times the samples were obtained should be retained on record for reference. The WER should be reevaluated whenever hardness, alkalinity, pH, TOC, and/or TSS decrease below the values that existed when the WERs were determined.

Biological Monitoring. If the SSO is a modification of an aquatic life objective, follow up bioassessment is encouraged to ensure that beneficial use of the water body is maintained. Annual bioassessments can be compared to reference and biological conditions of the site prior to SSO approval to determine if the use is being attained. Depending on the nature of the chemical, bioassessments may include bioaccumulation studies.

The Ratio of the New and Old Objective. The greater the increase in the objective, the more concern there should be about (1) the fate of any nontoxic chemical that contributes to the WER, and (2) changes in water quality at the site. The imposition of one or more conditions should be considered if the WER is used to raise the criterion by, for example, a factor of two, and especially if it is raised by a factor of five or more.

2.1 Antidegradation Considerations

Under the Federal water quality standards regulations, water quality standards include designated uses, criteria to protect those uses, and an antidegradation policy. The purpose of an antidegradation policy is to protect water quality that is higher than necessary to protect the uses. Any action taken that might result in lowering water quality is subject to the policy and triggers of an antidegradation review.

In California, lowering of water quality may be allowed when "...it has been demonstrated to the State that any change will be consistent with maximum benefit to the people of the State, [and] will not unreasonably affect present and anticipated beneficial use of such water....(State Board Resolution 68-16)" This language is consistent with the antidegradation requirements in the Federal water quality standards regulation (40 Code of Federal Regulations (CFR) part 131.12(a) (1) and (2)). The regulation requires that an antidegradation policy be established to maintain and protect existing beneficial uses and the water quality necessary to protect these uses. The regulation, however, allows a lowering of water quality if water quality exceeds the level necessary to support propagation of fish, shellfish, and wildlife, and recreation in and on the water; and the lowering is necessary to accommodate important economic or social development in the area of the waters. Lowering of water quality can take place only when existing uses will not be impaired. Intergovernmental coordination and public participation procedures must be followed. In addition, the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best management practices for nonpoint sources must be achieved. Finally, there must be a reasonable showing of the costs involved and the anticipated economic or social development that will be foregone. Making a determination to allow a lowering of water quality based on these factors is called an antidegradation review.

The adoption of site-specific objectives that are less stringent than existing criteria or objectives may ultimately result in a lowering of water quality. This action, hence, triggers the antidegradation policy. Under these circumstances, the regulator must consider whether an antidegradation review is more appropriately done at the water quality standards-setting phase or the permitting phase. In some cases, the impetus to develop site-specific objectives is driven by the desire to accommodate economic or social development in the affected area. In these circumstances, the regulator may have sufficient information such as economic data to perform an antidegradation review when the site-specific objectives are adopted. The advantage of doing an antidegradation review at this stage is that it obviates the need to do an additional review when NPDES permits are issued implementing the new objectives. Typically, however, the regulator will not have the necessary information to perform an antidegradation review when site-specific objectives are adopted. Under these circumstances, the determination whether a water quality lowering is justified will have to be done at the NPDES permitting stage.

Antidegradation considerations may or may not change applicable NPDES permit limits. There are several concerns regarding antidegradation implementation. One concern is that "creeping degradation" might result over time from a series of small changes, each inconsequential, whose overall effects may unintentionally degrade a water body.

Determinations made on the other elements of an antidegradation review could result in a decision to not apply the SSO, even though it may be acceptable scientifically. For example, after an SSO is derived, the State could decide that the economic savings that might result are not sufficient to offset the desire to simply maintain a higher level of water quality.

Because it is possible that the determination of an SSO will result in a more stringent limit, the more stringent limit should be reflected in water quality standards, and the designated use upgraded if necessary. In this situation, an antidegradation review is not required, because there is no lowering of water quality.

2.2 Consultation and Public Participation

Developing an SSO following acceptable scientific procedures is an optional component of the water quality standards program (40 CFR Part 131.11(b) (ii)). It is a matter of State discretion whether to adopt such objectives, although any affected party in the State can request the opportunity to develop the science for such a change in an objective. Since SSOs are part of water quality standards, their adoption must follow the administrative procedures of the State, including participation by the appropriate regional boards and other affected State agencies, and be approved by the Regional Board, State Board and USEPA.

It is critical that the State identify all interested parties (environmental, industrial, or governmental) that are likely to have an interest in the particular water body for which an SSO is being considered, as the successful application of antidegradation concepts requires a balanced review of the data and potential impacts of the proposed action. Potential environmental advocacy groups include any of the local chapters of the national environmental organizations, plus any local groups that might exist. Also, industrial organizations and associations are candidates for participation. Numerous government organizations, including local municipal governments, regional or district governments, and various Federal agencies, are also likely to be interested in becoming involved. Obviously, if an NPDES permittee is responsible for initiating the action, the permittee will be involved. All interests should be invited and encouraged to participate. The assemblage of interested parties should be representative and not be dominated by a specific interest.

The primary Federal agencies that need to be contacted when developing SSOs are the Region 9 Office of USEPA, and the appropriate field offices of the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS). USEPA should be informed so that the State of California and USEPA can agree on the appropriateness of the procedure to be used to develop an SSO and to ensure that the State personnel understand the correct application of the available USEPA guidance. USEPA supports the concept of SSOs, and has indicated that it will review any SSO that was developed following the acceptable procedures. Any properly developed and adopted SSO must be approved by USEPA as a change to the State's water quality standards. Early discussions with USEPA will help to assure that the correct procedures are followed and will allow USEPA to become aware that such changes to water quality standards are being considered.

It is also extremely important that the State contact the field offices of NMFS and USFWS (the Services) at the outset to ascertain whether threatened or endangered species or their habitat exist where the SSOs are being considered. If these issues are not a concern, then no further action is required to protect federally listed species. In addition, there are a number of threatened/endangered species listed under the California Endangered Species Act that are not federally listed. These species must be considered in terms of protecting designated beneficial uses for RARE (Rare, Threatened or Endangered Species) and/or BIOL (Preservation of Biological Habitats) beneficial uses, and complying with California

Environmental Quality Act (CEQA) and California Endangered Species Act (CESA). On line reference for California threatened/endangered species and sensitive habitats include: Department of Fish and Game Natural Diversity Database www.dfg.ca.gov/whdab/html/cnddb.html, Audubon California Important Bird Areas report www.audubon-ca.org/IBA.htm, and California Native Plant Society Inventory of Rare and Endangered Plants www.northcoast.com/~cnps/cgi-bin/cnps/sensinv.cgi. If species of concern are present, there will likely be considerable discussion relating to the relevance of SSOs at that site and, depending on the availability of data, the process to derive the objectives could be more complex. If threatened or endangered species are present, SSO are likely to become more stringent. The Services and agencies outlined above may also have data useful in developing the objectives.

State agencies that need to be contacted when developing SSOs include the California Department of Fish and Game (CDFG) and other California Environmental Quality Act "responsible agencies" and "trustee agencies." Information regarding consultation with these agencies can be found at the Governor's Office of Planning and Research website at http://www.opr.ca.gov and the Resources Agency's CEQA website at http://www.ceres.ca.gov/ceqa. Other State agencies that may be sources of information for use in standards development include OEHHA and the departments of Pesticide Regulation, Toxic Substances Control, and Water Resources. The Central Valley Regional Water Quality Control Board's *A Compilation of Water Quality Goals*, available at http://www.swrcb.ca.gov/rwqcb5/available_documents/wq_goals/index.html, may provide additional information.

Depending on the situation, environmental groups may or may not be active in opposing or supporting a particular SSO. The State, using all the means at its disposal to make constituents aware of the planned action, needs to alert the environmental groups of what is planned, why it is being proposed, how it is to be accomplished, and what results may be anticipated. While industrial organizations and municipalities are likely to have the most data available in support of SSOs, environmental groups may also have critical information, and must be recognized as a legitimate partner in the process. Universities may also be a good source of data and assistance in deriving SSOs. They could also function in the role of a more neutral participant considering the science, environmental impacts, and anticipated benefits. In some cases, the proponents of developing the SSOs may be willing to support the services of consultants to fill the role of a neutral evaluator of the need for, and benefits of, SSOs. It is virtually certain, however, that if the State fails to seek the participation of all potentially affected parties, the effort to develop, adopt, and implement an SSO will be more complicated, more controversial, and less likely to succeed than if the time and effort is taken to seek active participation by the interested parties.

There are several general guidelines to follow to identify and work with stakeholders. First, it is important to recognize that all affected parties are legitimate partners in the SSO development process--it is not just the NPDES permit holder or those whose permits might be impacted. Second, it is important to ensure that all identifiable parties are involved early in the process, before key decisions have been made. Third, it is valuable to be clear as to the purpose of the task, how it will be undertaken, and what the anticipated environmental and economic impacts will be if a change is made, or alternatively if no change is made. Fourth, it is helpful to explain any uncertainties in the data and the potential impact if SSOs are implemented. Fifth, it is critical that all interested parties have access to all the data being considered. Finally, whoever is conducting the SSO studies (if not the State) must coordinate closely with the State and make sure that all the procedures required by the State to amend and/or adopt new water quality standards are followed in the process of developing the science for the State to consider an SSO.

3 AQUATIC LIFE BASED CRITERIA

National Water Quality Criteria (WQC) are established according to guidelines (Stephan et al 1985) developed by the USEPA. The guidelines provide procedures for collecting, evaluating, and using data on the toxicity of individual pollutants to aquatic organisms or consumers of those organisms. Most of the data describe acute toxicity (short-term exposure, 48-96 hour) or chronic toxicity (long-term exposure, usually exceeding 96 hours and encompassing all, or a significant portion, of an organism's life span). In addition, data regarding the effects of a pollutant on plants are considered, as well as data on the bioconcentration (current revisions also include bioaccumulation) of a pollutant into the tissues of an aquatic organism, and the effect of that tissue burden upon the consumers of aquatic organisms.

National criteria are established separately for freshwater and saltwater environments. Although updates and new criteria appear periodically, there are currently guideline-based freshwater and saltwater criteria for approximately three dozen chemicals, including the most toxic metals and pesticides, and several other important pollutants such as ammonia and cyanide (http://www.epa.gov/ost/pc/revcom.pdf or http://www.epa.gov/waterscience/pc/revcom.pdf). Most criteria are based upon protection of aquatic organisms as a result of the direct toxic effects of a pollutant, primarily through water exposure. The criteria for a few bioaccumulative chemicals are based on the protection of people (chronic criteria for mercury and some organic compounds), or fish (selenium). The derivation of these criteria is not based on standard aquatic toxicity test dataset, and therefore not amenable to standard site-specific approaches.

Aquatic life criteria for salt water organisms were developed for marine/estuarine organisms and may not necessarily be appropriate for the inland saline waters of California (e.g., Mono Lake, the Salton Sea, ephemeral desert playa lakes and wetlands). These waters support organisms adapted to salinity much higher than that of seawater and to high toxic trace element concentrations (recent studies at Mono Lake have shown that arsenic can be important in microbial energy cycling there). While SSOs would appear to be desirable for these waters, they probably share few species with those used to develop the Federal criteria, and the Resident Species Procedure might be the only appropriate method.

Most national criteria are based upon direct toxicity to aquatic organisms, and consist of an acute objective (the Criteria Maximum Concentration or CMC), and a chronic objective (the Criterion Continuous Concentration or CCC). Both the CMC and the CCC have three parts, an average concentration magnitude, an averaging duration, and a return period (Section 2.3 of the Technical Support Document for Water Quality-based Toxics Control, EPA/505/2-90-001). The magnitude (expressed as a concentration, e.g., 10 μg/L) is calculated from the toxicity dataset for each chemical. The duration and return periods are default periods stipulated by USEPA. Duration is the period over which a concentration must be averaged, and is usually 1 hr for the CMC and 4 days for the CCC (one exception is the 30-day averaging period for the ammonia CCC). The return period, without current exception, is one exceedance every three years, on the average.

A typical criterion statement reads:

> "The procedures described in the 'Guidelines for Deriving Numerical 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, freshwater [or saltwater] aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentration of [chemical] does not exceed [CCC] μg/L more than once every three years and if the one-hour average concentration does not exceed [CMC] μg/L more than once every three years on the average."

The CCC and the CMC are calculated to protect all aquatic organisms for which acceptable toxicant effect data are available, as represented by the known dataset describing the toxicity to North American aquatic species for each chemical. Both the CCC and the CMC can be reduced to protect any commercially, recreationally or ecologically important species that appears to be adversely affected by concentrations below the calculated criterion. Ecological importance is not precisely defined, but would include organisms particularly important as food due to their known or suspected critical presence in certain niches, at critical times, of particular sizes or as vital components of aquatic habitat; e.g. marine kelp.

National criteria are based upon available toxicity data for any species that has reproducing wild populations in North America. Criteria are based upon Genus Mean Acute Values (GMAVs) that are calculated from acceptable data for all species within a genus. Species values are themselves the geometric mean (Species Mean Acute Value, or SMAV) of all acceptable data for each species. Preference is given to acute toxicity tests that are flow-through (typically characterized by nearly continuous replacement of test solutions) in which the concentrations are confirmed by measurement of the test chemical in the test solutions. All appropriate data on toxicity to aquatic animals, aquatic plants, and consumers of aquatic animals are evaluated in developing a criterion.

3.1 Evaluation of Existing USEPA Criteria and SSO Guidance

3.1.1 National Guidelines and Criteria

National water quality criteria established using the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their Uses* (Stephan et al. 1985 and hereinafter referred to as "the Guidelines") are a significant improvement over previous criteria. Prior to the publication of the Guidelines, water quality criteria were usually developed *ad hoc*, and followed no set procedure. These earlier criteria were based upon best estimates provided by individuals or committees, and the methodologies used were sometimes inconsistent.

3.1.1.1 Strengths of Water Quality Criteria

The Guidelines provide a defined process for addressing a variety of toxicological data in a consistent manner. Among the factors considered in establishing current national water quality criteria, some of the more important are: provisions for minimum standards for data quality, a minimum dataset content requirement, and specific procedures for calculating criteria from the body of acceptable data. These procedures provided for criteria reflecting both short-term acute and long-term chronic protection. Where

appropriate data are available, current criteria reflect the influence of other water quality parameters on the toxicity of the criteria chemicals, particularly for certain metals and ammonia.

Current criteria have been developed considering toxicological data for both plants and animals, as well as data on the effects on predators feeding on contaminated prey. A similar provision allows for the protection of human health as a consequence of consuming contaminated fish and shellfish. Finally, for the purposes of developing permits, the current criteria provide both averaging and return periods.

A major advantage of the current criteria is that they do not rely upon a single most-sensitive test as the keystone of the criteria. Rather, the criteria are based upon an analysis of the distribution of acute effect concentrations using a statistical computation, usually based upon the four most sensitive genera and the total number of genera in the dataset.

3.1.1.2 Important Considerations Related to Water Quality Criteria

Despite the improvements the Guidelines provide for establishing water quality criteria, a number of important consideration still exist, and these lead to uncertainty regarding the level of protection actually provided by criteria. In regard to the requirement of a minimum database, the requisite minimum does not represent a very large number of taxa. As a result, most taxa are not represented, and an unknown proportion of those not represented may not be protected.

Acute toxicity data are limited to species and life stages that can be readily reared or collected in sufficient numbers to provide test organisms that meet acceptable criteria for taxonomic identification, health, uniformity of size and age, and control performance. Chronic data are limited to those species that will reproduce and/or readily grow in the laboratory. Whether these limitations bias the data toward more or less sensitive species is an open question. The only certainty is that most datasets include several taxa frequently found to be among the most sensitive taxa in very large datasets. Thus, one might conclude that the current criteria provide at least the level of protection intended, unless the tested species tend to be less sensitive on the average than the untested species.

However, national criteria for each chemical are established in isolation of potential effects of other chemical pollutants that might interact either additively, synergistically, or antagonistically. This is a necessity due to the seemingly infinite number of possible combinations of chemicals and their relative concentrations, as well as their temporal and spatial variations. These contaminant interactions may be detected by either whole effluent toxicity testing or by biological assessments of the in-stream community, which are absolutely essential supplements to numeric and narrative criteria to ensure adequate protection of aquatic life.

The statistical procedure used to compute a criterion calculates a 5^{th} percentile most-sensitive genus. Because the minimum acute toxicity database requires GMAVs for only eight genera, the 5^{th} percentile value can fall below the most-sensitive genus in the database, which is the case whenever the number of tested genera is fewer than about 20. Thus, for those chemicals with fewer than 20 tested genera, the criteria might provide a higher level of protection than intended, if the genera that are represented include at least one or two which are among the most sensitive. The only way to remove this potential bias towards overprotective criteria is to add genera to the acute toxicity database.

Water quality criteria are commonly considered to be safe concentrations. In practice, when criteria are applied to effluents at the end-of-pipe or the edge of a mixing zone, contaminant concentrations are subject to significant further dilution, so that the contaminant concentration for most of a body of water is considerably lower. Therefore, if criteria actually express the level of aquatic life protection intended, then most of the body of water is protected more than is the case at the point of compliance. However, situations (e.g., effluent dominated waters) where the contaminant concentration was routinely maintained at the criteria level in an entire body of water, then an accurate criterion would theoretically allow considerable harm to 5 percent of the aquatic species. This set of conditions could actually cause the disappearance of some species, which would be in contravention of many other goals of water quality protection, including endangered species protection and anti-degradation. Ultimately, the protection offered in such waters depends upon the relative sensitivity of site-species and the species represented in the criteria database.

As stated above, water quality criteria are tested independently and, therefore, implementation of their safe levels do not protect against the effect of multiple contaminants occurring at, or even below, their safe levels. Effluent dominated waters, which lack significant effluent dilution, are systems in which multiple contaminants can co-occur at elevated concentrations that approach objective levels. Because the toxicity of toxicants is often additive, this could result in the loss of sensitive species from effluent dominated waters. Currently, WET test results are the most direct way of detecting and evaluating the probable in-stream toxicity in such situations.

Experience in evaluating toxicity test data and performing toxicity tests has demonstrated that different life-stages of many species have different sensitivities to chemicals. According to the Guidelines, where multiple life stages have been tested, the most sensitive life stage can be used to establish species mean acute values. But for many species, the most-sensitive life stage may not have been tested, and therefore the overall sensitivity of the species may not be known. This uncertainty may result in a bias towards under-protective criteria.

Many chemical and physical characteristics of surface waters are considerably different from those waters typically used in laboratory toxicity testing. Among the more important characteristics are dissolved organic material and suspended solids, both known to reduce the toxicity of many chemicals through reduced bioavailability. For this reason, criteria based upon laboratory toxicity tests with laboratory-grade water may over-estimate toxicity in many natural waters, as well as in those waters containing a significant proportion of effluents high in dissolved organic matter and suspended solids. These circumstances can lead to over-protective criteria, which is particularly problematic given that criteria were formerly expressed in terms of total recoverable chemical concentration. The potential for overly restrictive criteria has been reduced for metals through the use of criteria based on dissolved concentrations, as opposed to total recoverable metal.

Although the current Guidelines for establishing criteria include procedures for adjusting for these and other water chemistry characteristics, these characteristics may not be reported in some studies and, even if reported, may not include a sufficient range of measurements to allow for calculation of the influence of the characteristics as they vary in concentration. Similarly, although plant data and animal and human consumer effect data can be considered in establishing criteria, these data are not commonplace, and much of the plant data is based upon studies in highly atypical media. Absence of this type of data may result in under-protective criteria, although consumption effects are common only with highly

bioaccumulatable materials for which data are often available. Similarly, plants are not often sensitive relative to animals, although marine kelp reproduction appears to be relatively sensitive to some materials.

Chronic criteria are established using the distribution of acute toxicity data, and are subject to any bias reflected in the acute toxicity dataset. In addition, chronic toxicity data are not as abundant as acute toxicity data, so that even the minimum chronic toxicity database of three test species is far less robust than the acute toxicity dataset. This is particularly important because chronic criteria are calculated based upon a weak (i.e., often variable) linkage to acute toxicity (the acute:chronic ratio) for the species for which both acute and chronic data are available. Any bias produced by this variability is probably random, and may therefore result in either over- or under-protection.

The averaging periods for aquatic life criteria (one hour for acute criteria, four days for chronic criteria) are "one size fits all." Depending upon how the averages are calculated and how dynamic the day-to-day variability is in the actual receiving water, these periods may be overprotective for most chemicals and most biota. The less dynamic the actual day-to-day variability of contaminant concentration, the more appropriate these averaging periods are for the biota in those situations.

In summary, there are many factors that could result in bias in either direction of national criteria developed using the Guidelines when applied at a particular site. If the magnitudes of the biases are small, or if they tend to cancel one another, the implications are not great. To limit bias with some of the more obvious uncertainties in the criteria, certain site-specific procedures have been developed for modifying national water quality criteria.

3.2 Site-Specific Objectives

National criteria are established on the basis of laboratory toxicity test data using either surface waters, well-waters, or reconstituted waters prepared using distilled or de-ionized water and reagent grade chemical salts. These test waters must be essentially free from any chemical contamination that might compromise the results of the toxicity tests. As a consequence, test waters are usually obtained from relatively pristine natural sources, or prepared using certified clean ingredients.

Criteria developed from these data are intended to be protective of aquatic organisms in all waters because they are based upon data for as many North American species as have been tested, and because the tests were generally conducted in high quality waters. The resultant dataset for any chemical may contain a large number of species, none of which are found in all water bodies. (There are some saltwater species in the national dataset that are found in most marine environments.) However, it is important to recognize that many bodies of water may contain dissolved or suspended materials that influence the toxicity of chemicals. These waters would not normally be considered as acceptable dilution waters for toxicity tests for developing national criteria. These two circumstances: 1) species representation, and 2) confounding water chemistry, can make national objectives that are either over-protective or under-protective for a particular body of water. USEPA has recognized this possibility, and has provided several procedures for modifying national criteria to more appropriately protect the species found in a specific water body. Although accounting for site-specific species representation and water body chemistry usually results in SSOs that are higher (less restrictive) than national criteria, it is possible that the presence of previously untested species and/or unusual water chemistry would require objectives that

are more protective than national criteria. SSO methods work equally well when more-protective objectives are needed.

A decision to pursue an SSO is usually prompted by one or more of the following situations:

- the inability to meet current permit limits;
- indications (e.g., from a bioassessment, or WET testing) that current discharges are not adversely impacting the aquatic community;
- chemical characteristics of the effluent or receiving water (e.g., high concentrations of suspended solids or organic carbon) that suggest reduced contaminant toxicity in that matrix;
- indications (e.g., from a bioassessment, WET testing, or a toxicity identification evaluation [TIE]) that the allowed discharge limits are adversely impacting the aquatic community; or
- the presence of an untested species of particular importance or suspected high sensitivity, including rare, threatened, or endangered species.

Although any of the above conditions could trigger an SSO study by various parties, SSO studies are predominantly discharger initiated, and typically result from one or more of the first three situations described above. Although the level of protection and attainability are important motivations for those seeking SSOs, the various procedures are structured with the intent of developing more-appropriate SSOs to achieve the level of protection intended by the national criteria, and should be applied with that goal in mind.

There are three current USEPA published procedures for establishing SSOs, and a number of other procedures allowed by USEPA based on proven scientific principles. This document presents all of the available procedures for developing SSOs. The most commonly used procedure for adjusting criteria to account for a site's water chemistry is the WER procedure. The second most commonly used procedure provides for adjustments to national criteria on the basis of the assemblage of species found in a particular site water (the Recalculation Procedure). A third procedure, the Resident Species Procedure, accounts for site water chemistry and the assemblage of resident species through tests of resident species in site water. The strengths and weaknesses of these three approaches are discussed in Sections 3.3-3.5.

Before selecting an SSO procedure, or combination of procedures, it is important to understand the probable direction and magnitude of difference between a criterion and an SSO. This understanding requires both a familiarity with the particular criteria dataset, and a familiarity with the effect of the calculation on the magnitude of the Final Acute Value (FAV) as the number and range of GMAVs changes. A reasonable understanding of these relationships requires a familiarity with the procedure for calculating FAVs and the development of the calculation procedure (Stephan et al. 1985). This understanding is an important consideration for determining if the Recalculation Procedure and the Resident Species Procedure should be used at a site. A discussion containing hypothetical examples of the effect of the number and range of GMAVs on the FAV is provided in Appendix C.

Several methodologies for the development of SSOs are given in the following sections. It should be noted that laboratories providing analytical support of permit monitoring are to possess proper certification issued by the Department of Health Services as required by Water Code Section 13176. It should also be noted that Quality Assurance Project Plans (QAPPs) are required whenever there is collection of environmental data, especially for those projects funded by federal dollars. For additional information, contact the State Board's Quality Assurance Program Manager.

3.3 Water-effect Ratio Procedure

3.3.1 Water-effect Ratio Procedure Overview

The WER procedure is, by far, the most commonly used method to derive an SSO. The WER is the ratio of the toxicity of a chemical in site water to the chemical's toxicity in laboratory water. This approach has the major advantage of not reducing the total number of species in the database, thus maintaining the degree of statistical robustness in the original calculation of the 5th percentile most-sensitive genus. The primary intent of the WER procedure is to quantify the combined effects of all potential interactions, among the various chemical components of the receiving water and the effluent, on the toxicity of the chemical of interest. In many cases, these interactions tend to reduce the toxicity of a substance, resulting in a higher water quality objective. A properly designed study will be able to utilize hydrological and chemical data in order to provide an objective that is applicable to specific site water quality conditions. This knowledge is important for adequately defining the site or sites to which the new objective is to be applied, as well as for selecting an appropriate sampling schedule that includes likely periods of greatest biological availability/toxicity based upon receiving water hydrology and chemistry. In addition, sometimes it is desirable to develop an SSO that is a function of receiving water chemistry, e.g., TSS, in which case the sampling schedule must encompass a representative range of site TSS concentrations.

In order to use the WER procedure, an adequate knowledge of site hydrology, ambient water chemistry, and waste water characteristics must be acquired in order to develop a defensible study plan. Obtaining appropriate data is likely to require both an investigation of historic data as well as the development of current flow and chemical data that are most germane to the WER that is being developed. This data-gathering process is also necessary to make an informed decision regarding the extent of the site or sites that are being considered in the development and application of the WER. The cost of obtaining these data, as well as the cost of conducting all the potential biological and chemical tests required, is often considerable.

The WER Procedure is commonly used when it is suspected that either chemical or physical aspects of receiving water or effluent (or both) will cause a pollutant to be less bioavailable and, therefore, less toxic. Laboratory waters used for rearing, holding, and conducting toxicity tests to establish national WQOs are routinely good quality, pollutant-free surface, groundwater well, or reconstituted waters. Results from such tests help assure that WQOs will be protective in most surface waters, and reduce the likelihood of criticism that toxicity is due to constituents of the laboratory dilution water. However, effluents and natural surface waters may contain meaningful concentrations of DOC, suspended particulates, or other materials, which can reduce the bioavailability (and therefore toxicity) of many pollutants. In a few cases, there may also exist natural surface waters or effluents whose composition enhances the bioavailability of pollutants. The WER procedure provides a means for developing SSOs that take into account changes in bioavailability and toxicity, in either direction.

Guidance to determine a WER is provided in USEPA's Interim Guidance on Determination and Use of WERs for Metals (USEPA 1994b; also in Appendix L of the Water Quality Standards Handbook: 2nd Edition, [USEPA 1994a]). A self-extracting zip file of the WER guidance can be downloaded from the website, http://www.epa.gov/seahome/wer.html. The WER procedure results in a multiplier which is applied to an existing water quality objective. If a chemical is less toxic in a site water, the multiplier is >1, and results in a higher SSO; if a chemical is more toxic in a site water, the multiplier is <1, resulting in a lower SSO. The multiplier (the WER) is simply the ratio of the toxicity in site water compared to the toxicity in typical laboratory dilution water. The ratio is determined from simultaneous side-by-side toxicity tests in the two types of waters. The WER equals:

$$\frac{\text{Site Water } LC_{50}}{\text{Lab Water } LC_{50}}$$

Because toxicity tests are not exact measures of toxicity but rather are experimental estimates influenced by many biological, chemical, and physical variables the WER procedure requires that at least three sets of tests be conducted with at least three weeks between sets and, if possible, covering a variety of potentially important conditions, e.g., streamflow, tidal phase, suspended solids, and season. In addition because it is assumed, but is not certain, that different taxonomic groups will respond similarly in WER tests at least one set of tests must use both a fish species and an invertebrate species in order to test the similar-response assumption. Before entering into a full-scale WER series, it is practical to conduct a trial WER test pair with a sensitive species in order to determine the likelihood of obtaining results that are satisfactory to those interested in deriving an SSO. If properly conducted, the trial tests can be used as the first in the required series of WER tests.

3.3.2 WER Procedure Stipulations

- Toxicity tests must be conducted following USEPA or American Society for Testing and Materials (ASTM) methods.

- Two laboratory dilution water toxicity tests must be confirmed by another laboratory (results listed in Table 1 of criteria documents are acceptable for confirmation as a second laboratory).

- WERs must be determined only for one chemical at a time, not several simultaneously in the same solutions.

- WERs cannot be extrapolated from one chemical to another, one effluent to another, or one site water to another.

3.3.3 Two WER Methods

Current guidance provides two different methods for obtaining site-water test solutions for conducting WER tests. Method 1 utilizes laboratory-prepared mixtures of effluent and upstream water, while Method 2 utilizes water collected from one or more sampling points in a (usually large) water body such as a lake, estuary, or long reach of a river. USEPA recommends that the methods be used as follows:

3.3.3.1 Method 1:

- is applied in the vicinities of plumes;
- is used to determine CMCWERs and most CCCWERs;
- is used primarily in flowing fresh waters, such as rivers and streams;
- applies to effluent dominated streams or where design flows are zero;
- can also be used to determine CMCWERs for near-field effects in such large water bodies as oceans, large lakes, reservoirs, and estuaries.

3.3.3.2 Method 2:

- is used to determine WERs that apply outside the area of plumes in large bodies of water (i.e., CCCWERs);
- may be used to determine WERs in advance of a proposed discharge into a water body;
- involves sampling site water at various times, locations, and depths as appropriate to identify the range of WERs that apply to the body of water defined as the site(s);
- results in experimentally determined WERs that are used to decide how many SSOs should be derived for the body of water (how many sites), especially to address spatial and temporal variability in receiving water characteristics;
- requires substantially more resources than Method 1.

3.3.4 Streamlined WER for Copper

In a procedural modification issued by USEPA (2001b), a streamlined WER approach for copper only was introduced for situations where ambient copper concentrations are elevated by the continuous discharge of point source effluents. The basis for this streamlined procedure is the substantial amount of data generated while developing WERs for copper in freshwater. USEPA states that with adequate basis in data, the approach might be possible for saltwater also, and might be acceptable for other pollutants. This procedure is currently approved only for copper. A comparison of the streamlined procedure and the standard WER procedure is presented in Table 1.

Table 1. Comparison of Streamlined Procedure and the 1994 Standard Interim Procedure.

Characteristic	Standard	Streamlined
Applicability	Universal	Only for copper from continuous discharges
Minimum number of sampling events	3	2 (with several restrictions)
Minimum number of WER measurements	4	2
Minimum number of WERs for calculating site WER	3	2
Preparation of simulated downstream water	Mix effluent with upstream water in proportion to flows at sampling time	Mix effluent with upstream water in proportion to flows at design conditions
Calculation of sample WER	Site water LC_{50} ÷ Lab water LC_{50}	Site water LC_{50} ÷ the greater of lab water LC_{50} or criteria document SMAV
Calculation of final site WER	Complicated scheme with six "if...then...else" clauses and 12 possible paths	Geometric mean of the two measured WERS

3.3.5 Reporting Requirements for WERs {http://www.epa.gov/seahome/wer.html}

A report of the experimental determination of a WER to the appropriate regulatory authority should include the following:

1. The name(s) of the investigator(s), laboratory name and location, dates of initiation and termination of the study.

2. A description of the laboratory dilution water, including source, preparation, and any demonstrations that the dilution water is acceptable for the survival, growth, and reproduction of aquatic species.

3. The name, location, and description of the discharger, a description of the effluent, and the design flows of the effluent and upstream water.

4. A description of each sampling event, including station location, sampling date, and time. The description must include reasons for the selection of sampling locations, dates, and times. The flows of the upstream water and effluent each time samples are collected must be reported.

5. The procedures used to obtain, transport, and store the effluent and upstream water samples.

6. Any pretreatment, such as filtration, of the effluent, site water, and/or laboratory dilution water.

7. The results of all chemical and physical measurements of the site water and laboratory dilution water, including hardness (or salinity), alkalinity, pH, and concentrations of total recoverable metal, dissolved metal, total suspended solids, and total organic carbon.

8. A description of the experimental design, test chambers, depth and volume of solution in the chambers, loading and lighting, and numbers of organisms and chambers per treatment.

9. The source and grade of the test chemical, how the stock solution was prepared, including any acids, bases, or solvents used.

10. The source of the test organisms, scientific name and how the species identification was verified. The age, life stage, means and ranges of weights and/or lengths, observed diseases, treatments, holding and acclimation procedures, and food must also be reported.

11. The average and range of the temperature, pH, hardness (or salinity), and the concentration of dissolved oxygen (as percent saturation and as mg/L) during acclimation, and the method used to measure these parameters.

12. A detailed report of each toxicity test, including: the average and range of the measured concentration of dissolved oxygen (as percent saturation and as mg/L); the average and range of the test temperature and the measurement method; the schedule for taking samples of test solutions and the methods used to obtain, prepare and store samples; the total recoverable and dissolved concentrations of the test material in each treatment.

13. All differences, other than the dilution water and the concentrations of metal in the test solutions, between the side-by-side tests using laboratory dilution water and site water.

14. A comparison of results obtained with the primary and secondary tests.

15. The WERs and an explanation of their calculation from the data obtained.

A report of the derivation of a Final Water Effect Ratio (FWER) must include the following:

1) A report describing the determination of each WER used in the derivation of the FWER; all WERs determined with secondary tests must be reported along with all WERs that were determined with the primary test.

2) The design flow of the upstream water and the effluent and the hardness used in the derivation of the permit limits, if the criterion for the metal is hardness-dependent.

3) A summary table must be presented that contains the following for each WER that was derived:

a) The value of the WER and the two endpoints from which it was calculated;

b) The highest WER (hWER) calculated from the WER;

c) The test type and species that was used;

d) The date the samples of effluent and site water were collected;

e) The flows of the effluent and upstream water when the samples were taken;

f) The following information concerning the laboratory dilution water, effluent, upstream water, and actual and/or simulated downstream water: hardness (salinity), alkalinity, pH, and concentrations of total recoverable metal, dissolved metal, TSS, and TOC.

g) A detailed explanation of how the FWER was derived from the WERs that are in the summary table.

A case example of how an SSO for hexavalent chromium was derived using the WER procedure is given in Appendix D.

3.4 Recalculation Procedure

3.4.1 Recalculation Procedure Overview

The goal of the Recalculation Procedure is to eliminate from the database those taxa that are not resident (and not expected to be present) in the site waters, while keeping in the database non-resident species that serve as toxicological surrogates for taxonomically related resident species. The process is based upon the assumption of taxonomic surrogacy, which is the assumption that species within genera are more similar in sensitivity than all species, genera within a family are more similar than all genera, and so on through taxonomic family, order, and class. If any taxon is not present at the site, then the dataset can be purged of all data for that taxon. For example, if no mayflies are present at the site, then, in most cases, all mayfly data can be purged from the dataset. However, if one mayfly species is present at the site, and no data were available for that species, data for at least the most closely related mayfly would be retained in the dataset.

The end result of the Recalculation Procedure is that the remaining data are more representative of the sensitivities of species found at the site. When this process raises or eliminates the mean acute values for the four most sensitive genera, it can provide for a higher site-specific criterion. Although the Recalculation Procedure provides a more representative array of test data for a particular site, the total number of genera represented in the database may decline appreciably through the deletion process. In this case, the statistical calculation of the 5^{th} percentile most-sensitive genus may result in a lower site-specific criterion. This loss of statistical robustness is the primary problem with the use of the

Recalculation Procedure. In addition, although taxonomic surrogacy is probably a reasonable assumption, there could be instances where the process fails to provide a good estimate of sensitivity for a resident species. In contrast, sensitive non-resident species that are deleted may in fact be representative of sensitivities of taxonomic categories that are resident but untested.

National criteria can utilize data for any North American aquatic species, and USEPA guidelines require acute toxicity data for a minimum of eight families, and chronic toxicity data for a minimum of three species (Minimum Dataset Requirements can be found in Appendix E). The minimum dataset must also include species representing several specified taxonomic groups or ecological niches. While a minimum dataset requires only eight families, many national criteria are based upon data for dozens of species with a wide range of sensitivities to the chemical. It is possible that the level of protection provided by the suite of species in the national dataset is inappropriate for the species found at a particular site. USEPA noted that there are at least three reasons why the level of protection might be inappropriate:

1) The national dataset for the chemical of interest contains aquatic species that are sensitive to many pollutants, but these and comparably sensitive species might not be resident at the site.

2) One or more species that are critical at the site might be sensitive to the pollutant and not represented in the national dataset and therefore require a lower criterion.

3) The species resident at the site might represent a narrower mix of species than those in the national dataset due to a limited range of natural environmental conditions at a site.

The Recalculation Procedure is structured so that corrections and additions can be made to the national dataset without the deletion process being used, to take into account taxa that are resident at the site. In effect, this procedure makes it possible to update the national aquatic life criterion. Recalculation by adding new taxa without using the deletion process has two potential outcomes, and either or both are possible: 1) a more-sensitive species might be added to the database, and this could result in a lower objective, and 2) an increase in the total number of genera (i.e., "n") in the database could result in a slightly higher objective due to the effect of a larger "n" on the calculation of the 5th percentile GMAV. Note: When the WER (Section 3.3) procedure is used, the laboratory water acute values are candidates for addition to the national database.

The Recalculation Procedure is intended to account for toxicological differences between the aquatic species that are resident at the site and those that were used to derive the national criterion. The term "resident" includes the species, genera, families, orders, classes, and phyla that:

- are usually present at the site;
- are present at the site only seasonally due to migration;
- are present intermittently because they periodically return to or extend their ranges into the site;
- were present at the site in the past, are not currently present at the site due to degraded conditions, and are expected to return to the site when conditions improve; and

- are present in nearby bodies of water, are not currently present at the site due to degraded conditions, and are expected to be present at the site when conditions improve.

The resident taxa cannot be determined merely by a one-time sampling downstream and/or upstream of the site. Resident species should not include species that were once present at the site but cannot exist at the site now due to permanent physical alteration of the habitat, e.g., the flooding of stream habitat by dams.

The definition of the "site" can be extremely important when using the Recalculation Procedure. For example, the number of resident taxa will generally decrease as the size of the site decreases. Also, if the site is defined to be very small, the permit limit might be controlled by a criterion or objective that applies immediately downstream of the site boundary.

The concept of the Recalculation Procedure is to create a dataset that is appropriate for deriving a site-specific criterion by modifying the national dataset by correction, addition, or deletion of data. Corrections and additions to the national dataset must include all USEPA-approved modifications, including any proposed during the current recalculation process (contact USEPA for this information). All corrections and additions that have been approved by USEPA are required, whereas use of the deletion process is optional.

The Recalculation Procedure is more likely to result in lowering a criterion if the net result of addition and deletion is to decrease the number of genera in the dataset, whereas the procedure is more likely to result in raising a criterion if the net result of addition and deletion is to increase the number of genera in the dataset (see Appendix C).

The concept of "correction" includes removal of data that should not have been in the national dataset in the first place. The concept of "correction" does not include removal of a datum from the national dataset just because the quality of a datum is claimed to be suspect. If additional data are available for the same species, USEPA will decide which data should be used, based on the available guidance (Stephan et al. 1985); also, data based on measured concentrations are usually preferable to those based on nominal concentrations. Selective corrections and additions are not allowed, i.e., all corrections on USEPA's newest list must be made. If the new dataset does not satisfy the applicable Minimum Data Requirements (MDRs), additional pertinent data must be generated (Appendix E). Once the new data are approved by USEPA, the Recalculation Procedure must be re-initiated with the addition of the new data.

Additions and corrections must be made as described above before the deletion process can be started. Selective deletions are not allowed; rather, the deletion process described below must be applied to all species in the national dataset. The step-wise process of deletion will identify which species must be deleted, and which species must not be deleted. Comprehensive information must be obtained concerning resident species prior to any deletion.

Acceptable pertinent toxicological data must be available for at least one species in each class of aquatic invertebrates, amphibians, and fish that contains a species that is a critical species at the site. For each aquatic plant, invertebrate, amphibian, and fish species that is resident at the site and is listed as threatened or endangered under Section 4 of the Endangered Species Act

(http://endangered.fws.gov/wildlife.html#Species), data must be available or be generated for that species or for an acceptable surrogate species (see Section 4, Endangered Species). Data for a surrogate species must be used as if they are data for resident species. If additional data are needed, they must be generated using acceptable procedures (Stephan et al. 1985), then be approved by USEPA, and the Recalculation Procedure must be re-initiated with the addition of the new data.

Data generation may also be required after the deletion process is completed. Even if one or more species are deleted, there still are MDRs that must be satisfied (Appendix E). If the data remaining after deletion do not satisfy the applicable MDRs, additional toxicity tests must be conducted using acceptable procedures (Stephan et al. 1985), so that all MDRs are satisfied. If the new data are approved by USEPA, the Recalculation Procedure must be re-initiated with the addition of new data.

Additional chronic tests are not required because the national Final Acute-Chronic Ratio (FACR) may be used to calculate the site-specific Final Chronic Value (FCV). If acute-chronic ratios (ACRs) are available or are generated so that the chronic MDRs are satisfied using only resident species, a site-specific FACR may be derived and used in place of the national FACR. Because a FACR was not used to derive the freshwater CCC for cadmium, this CCC can only be modified the same way as a FAV; which approach is acceptable will depend on which species are deleted.

If species deletion is to be evaluated, the following deletion process must be applied:

1) Obtain a copy of the national dataset, i.e., Tables 1, 2, and 3 in the national criteria document.

2) Make corrections in and/or additions to the national dataset, as described above.

3) Group all the species in the dataset taxonomically by phylum, class, order, family, genus, and species (other taxonomic groups are sometimes encountered, e.g., tribe, and should be included) (see Table 2 in Appendix F).

4) Circle each species that satisfies the definition of "resident," including any species that are identified as acceptable surrogates for threatened or endangered resident species.

5) Use the step-wise process, shown below, to determine which of the uncircled species must be deleted and which must not be deleted.

Species Deletion Step-wise Process

Step 1. Is the genus resident at the site?

- No. Go to step 2.
- Yes Within the genus, are there resident species that are not in the dataset?
  - No. Delete the uncircled species.*
  - Yes. Retain the uncircled species.*

Species Deletion Step-wise Process

Step 2. Is the family resident at the site?

- No. Go to step 3.
- Yes. Within the family, are there resident genera that are not in the dataset?
  - No. Delete the uncircled species.*
  - Yes. Retain the uncircled species.*

Step 3. Is the order resident at the site?

- No. Go to step 4.
- Yes. Does the dataset contain a circled species that is in the same order?
  - No. Retain the uncircled species.*
  - Yes. Delete the uncircled species.*

Step 4. Is the class resident at the site?

- No. Go to step 5.
- Yes. Does the dataset contain a circled species that is in the same class?
  - No. Retain the uncircled species.*
  - Yes. Delete the uncircled species.*

Step 5. Is the phylum resident at the site?

- No. Delete the uncircled species.*
- Yes. Does the dataset contain a circled species that is in the same phylum?
  - No. Retain the uncircled species.*
  - Yes. Delete the uncircled species.*

* Continue the deletion process by starting at step 1 for each uncircled species until all uncircled species in the dataset have been considered.

The species that are circled and those that are retained constitute the site-specific dataset. (An example of the deletion process is given in the case study given in Appendix F.)

This deletion process is designed to ensure that:

- each resident species in the national dataset is also in the site-specific dataset;

- each resident species not in the national dataset is represented in the site-specific dataset by all species in the national dataset that are in the same genus;

- each resident genus not in the national dataset is represented in the site-specific dataset by all genera in the national dataset that are in the same family; and

- each resident order, class, and phylum in the national dataset is represented in the site-specific dataset by the one or more species in the national dataset most closely related to the resident species in that order, class, and phylum.

The initial MDRs for the Recalculation Procedure are the same as those for the derivation of a national criterion (Appendix E). If a specific MDR cannot be satisfied by any resident species, a taxonomically similar resident species must be substituted in order to meet the eight MDRs as follows:

- if no species of the type required is resident at the site, but a species in the same order is, the MDR can only be satisfied by data for a resident species in that order;

- if no species in the order is resident at the site, but a species in the class is, the MDR can only be satisfied by data for a resident species in that class;

- if no species in the same class is resident at the site, but a species in the phylum is, the MDR can only be satisfied by data for a resident species in that phylum;

- if no species in the same phylum is resident at the site, any resident species that is not used to satisfy a different MDR can be used to satisfy the MDR;

- if additional data are generated using acceptable procedures (Stephan et al. 1985) and they are approved by USEPA, the Recalculation Procedure must be re-initiated with the addition of the new data.

National Water Quality Objectives are based upon GMAVs. The reason that GMAVs are used is that public comments regarding the first round of national criteria and the guidelines for their derivation suggested that the use of SMAVs resulted in too many SMAVs for cladocerans and salmonids being included in the FAV calculation, driving the criteria lower than appropriate for many waters with few (or no) such species. The USEPA agreed that such an effect is possible, especially for chemicals with small datasets and, following further analysis of taxonomic and toxicological relationships, decided upon the use of GMAVs in the calculation of the FAV for national criteria.

However, when the Recalculation Procedure is used with species deletion, there should be no species left in the dataset that is not either a resident species or a species that is the most appropriate surrogate for a resident species. For this reason, it should be acceptable to utilize SMAVs for the calculation of FAVs when an SSO is developed using the Recalculation Procedure with species deletion. Where there is only one species in each genus remaining in the dataset, this is the same as using GMAVs. If there are two or more species in any genera, the result of using SMAVs rather than GMAVs would be a larger “n” in

calculating the 5th percentile SMAV. When a resident species is represented in the dataset by co-surrogate species, the geometric mean of the two or more co-surrogate species should still be used.

The complete SMAV dataset should be examined to determine which of the four lowest SMAVs are for resident species, and which are for surrogate species. The more that the FAV is controlled by surrogate species, the more appropriate it would be to attempt to conduct toxicity tests with the one or more resident species represented by the surrogate species. There are 16 possible patterns of surrogate and resident species for the four lowest SMAVs (Table 2).

The potential importance of surrogate species increases from case A to case P. As illustrated in Table 2, replacing any SMAV with another SMAV that decreases the variability among the four lowest SMAVs will usually result in a higher FAV, while replacing a SMAV with another that increases the variability among the four lowest SMAVs will usually result in a lower FAV. The decision to conduct resident species testing will be guided by several considerations, including:

- does the surrogate species SMAV introduce a great deal of variability? This could be important in cases E through P, especially in cases M through P, and basically any time the 1st or 4th lowest SMAV is for a surrogate species.

- is it practical to conduct tests with the one or more resident species for which the surrogate has been retained in the dataset?

- are there indications that the resident species might be considerably more, or less, sensitive than its surrogate?

When developing toxicological data (usually LC_{50}s or EC_{50}s) for resident species, it can be difficult to obtain sufficient numbers of test organisms. Two alternatives should be considered. First, there may be other nonresident species that are more readily available than the resident species, and more closely related taxonomically than the species identified as surrogate from the recalculation dataset. Determining LC/EC_{50}s for the closer surrogate species should provide for a better estimate of the sensitivity of the resident species.

Table 2. Sixteen Possible Patterns of Surrogate and Resident Species for the Four Lowest SMAVs.

	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P
1st lowest SMAV	R	R	R	R	R	R	R	R	S	S	S	S	S	S	S	S
2nd lowest SMAV	R	R	S	S	R	S	S	R	R	R	S	S	R	R	S	S
3rd lowest SMAV	R	S	R	S	R	R	S	S	R	S	R	S	R	S	R	S
4th lowest SMAV	R	R	R	R	S	S	S	S	R	R	R	R	S	S	S	S

A second potential method is a limited bioassay approach. Because the actual SMAV concentration is important only if it is for one of the four most sensitive species, it would be sufficient to adequately demonstrate that a particular resident species is less sensitive than the fourth most sensitive species. Thus, if the fourth most sensitive species has a SMAV of 100 μg/L, it is necessary only to demonstrate that a site species shows >50 percent survival at 100 μg/L in an acceptable toxicity test. This can be done without conducting a full-scale, multiple-concentration toxicity test. For example, if 10 organisms were exposed (in typical laboratory water) at 100 μg/L for 48 or 96 hours, as appropriate for the species, and **none** died, that should constitute reasonable evidence that the SMAV is >100 μg/L and that SMAV (>100 μg/L) should be added to the dataset. Two additional considerations to this approach are: 1) with small numbers of test organisms even 30 or 40 percent mortality should not be taken as proof of <50 percent mortality, and 2) if the tested life stages are known to be considerably less sensitive, this needs to be considered (e.g., perhaps the test should be conducted at a higher concentration (e.g., 200 µg/L). It should be noted, however, that if >50% of the test organisms die, then the SMAV would be important to the derivation of the FAV, but is unknown. Under the latter circumstance, either the SMAV must be determined by appropriate test procedures, or no SSO can be developed.

If fewer than eight families of aquatic invertebrates, amphibians, and fishes are resident at the site, an SMAV must be available for at least one species in each of the families and the following special version of the Recalculation Procedure must be used:

- data must be available for at least one resident species in each of the families resident at the site;
- the lowest SMAV that is available for a resident species must be used as the FAV; and

- the site-specific CMC and CCC must be calculated as described in the process below:

Determining the CMC and/or CCC

1. Determining the FAV:
    - If the eight-family MDRs are satisfied, the site-specific FAV must be calculated from SMAVs using the procedure described in the national aquatic life guidelines (Stephan et al. 1985).
    - If fewer than eight families of aquatic invertebrates, amphibians, and fishes are resident at the site, the lowest SMAV that is available for a resident species must be used as the FAV.
2. The site-specific CMC must be calculated by dividing the site-specific FAV by 2.
3. The site-specific FCV must be calculated by dividing the site-specific FAV by the national FACR (or by a site-specific FACR if one is derived). Note: Because a FACR was not used to derive the national CCC for cadmium in fresh water, the site-specific CCC equals the site-specific FCV.
4. The calculated FAV, CMC, and/or CCC must be lowered, if necessary, to
    a) protect an aquatic plant[6], invertebrate, amphibian, or fish species that is a critical species at the site, and
    b) ensure that the criterion is not likely to jeopardize the continued existence of any endangered or threatened species listed under Section 4 of the Endangered Species Act or result in the destruction or adverse modification of such species' critical habitat.

Results of use of the Recalculation Procedure must be submitted in a report that includes the following information:

- a list of all resident species of aquatic invertebrates, amphibians, and fishes, along with the source of the information;
- a list of all aquatic plant, invertebrate, amphibian, and fish species that are critical species at the site, including all resident species that are listed as threatened or endangered under section 4 of the Endangered Species Act;

[6] Toxicity data for algae, including marine macroalgae, are often available or can be obtained by established testing procedures. Toxicity data for freshwater macrophytes are rare, and they are not usually considered to be as sensitive as other freshwater taxa.

- a site-specific version of Table 1 from a criteria document produced by the USEPA after 1984;
- a site-specific version of Table 3 from a criteria document produced by the USEPA after 1984;
- a list of all species that were deleted;
- the new calculated FAV, CMC, and/or CCC;
- the lowered FAV, CMC, and/or CCC, if one or more were lowered to protect a specific species.

A case example of how an SSO for aluminum was derived using the Recalculation Procedure is given in Appendix F.

3.5 Resident Species Procedure

3.5.1 Resident Species Procedure Overview

Probably the least-used procedure for establishing an SSO is the Resident Species Procedure. This process provides the opportunity to generate a dataset using only species resident at the site. Thus, none of the species in the dataset are surrogates. In addition, if the tests are run in site water, the new objective will incorporate the advantage of the WER process by automatically taking into account the bioavailability influences of all the chemical and physical characteristics of the site water. Data for some resident species may already be in the national dataset and can usually be retained in the resident species dataset without further testing. However, resident species data already in the national dataset will not usually be reflective of the influences of site water.

The over-riding problem with the Resident Species Procedure is that the number of genera in the generated dataset is usually so low (because of the cost of generating a robust dataset) that the statistical computation of 5th percentile most-sensitive genus typically produces a criterion that is below that needed to protect the most sensitive species tested. In addition, it is necessary to obtain test organisms of the appropriate species. This can require the collection of fish and invertebrates from site waters in sufficient numbers, uniformity of size, and health to provide for the necessary toxicity tests.

USEPA guidance has provided the alternative to prepare an SSO by developing a dataset using only resident species. This approach can be performed using either site water or any typical laboratory water, although it is probably more appropriate and preferable to use site water. This procedure requires that minimum database requirements are met as appropriate for the resident species at the site. As demonstrated in Figures 1 through 3 of Appendix C, the FAV will usually fall below the lowest GMAV, unless at least 20 GMAVs are available in the dataset. As site size is increased, the number of resident species will also increase (it can never decrease with increasing size). This consideration is important, as collecting, holding and testing individuals of resident species will often prove to be either impossible or impractical. However, as site size increases, the range of water chemistry variables will also increase,

making the definition of site water more difficult, and possibly making it necessary to conduct tests in several site waters representative of the range of water chemistry conditions.

In general, the Resident Species Procedure is not a practical method for developing an SSO, due to the large costs of testing and obtaining an adequate number of GMAVs. Where one or more species of special concern are present at a site, the use of the Recalculation Procedure, following addition of data for the species of special concern, is much more practical. Regardless of the calculation of the FAV from the augmented dataset, the FAV can always be lowered to the SMAV of a species considered critical to the site.

A case example of using the Resident Species Procedure to derive a site-specific criterion for copper is given in Appendix G.

Note:
The discussion of the guidances discussed above pertain only to the methodologies USEPA has published. There are other proposed approaches for developing site-specific criteria for which USEPA has issued policy statements, but has not published methods, and those approaches are therefore not addressed in the above review. The proposed methods include using natural background concentrations of chemicals for site-specific criteria, establishing site-specific criteria to protect endangered species, development of site-specific bioaccumulation factors for use in human health water quality criteria, development of site-specific default values used in Ambient Water Quality Criteria (AWQC) calculations, and wildlife-based criteria.

3.6 The Chemical Translator

An approach which addresses the issue of the bioavailability of a chemical, as do the WER and Resident Species Procedures, is the chemical translator. In contrast to the WER and Resident Species Procedures, the chemical translator does not modify the WQO directly, but rather it translates the site-specific dissolved form of the WQO to the total recoverable form.

The State Board's *Policy for Implementation of Toxics Standards for Inland Surface Waters, Enclosed Bays, and Estuaries of California (SIP) Section 1.4.1* provides California requirements for development of metal translators as part of NPDES permitting. These requirements are included as Appendix H.

For a number of years USEPA has recommended using dissolved metal concentrations (as opposed to total recoverable metal concentrations) to set water quality criteria, because dissolved metal more closely approximates the bioavailable fraction in the water column than does total recoverable metal (USEPA 1993). USEPA believes that the primary toxic mechanism for organisms that live in the water column is adsorption to, or uptake across, the gills; this physiological process requires a metal to be in the dissolved form. USEPA does not make the claim that particulate metal is nontoxic, but rather that particulate metal appears to be substantially less toxic than dissolved metal (USEPA 1996).

Dissolved metal is operationally defined as the metal which passes through a 0.45 μm or a 0.40 μm filter, while particulate metal is operationally defined as total recoverable metal minus dissolved metal. The operationally defined dissolved fraction of the metal includes particulate metal that is sufficiently small to pass through the filter, or that is absorbed to, or complexed with, organic colloids and ligands which also

pass through the filter. Some or all of the operationally defined dissolved metal, may not be biologically available.

By regulation (40 CFR 122.45(c)), NPDES permit limits, in most instances, must be expressed as total recoverable metal, and compliance must be assessed by measuring total recoverable metal. Because of differences in the chemical characteristics of effluents and receiving waters, there can be changes in metal partitioning between dissolved and adsorbed forms when the effluent and receiving water mix. An additional calculation can be performed and applied to determine the dissolved fraction of a metal specific to that site (a translator). This translator calculation answers the question "What is the total metal concentration in the discharged effluent that will not exceed the dissolved metal criterion in the receiving water body?"

The translator is used to convert the dissolved metal criterion back to a total metal concentration for evaluating compliance with a total metal NPDES permit limit. USEPA defines the translator as the fraction of the total recoverable metal in the downstream water that is dissolved, f_d. The translator is used to estimate the concentration of total recoverable metal in the effluent discharge that equates to (or results in) the criterion concentration in the receiving water body. Some states (Michigan, for example) define the translator as the total-to-dissolved ratio (TDR), which is the reciprocal of the f_d. The difference in the two forms of metal is reflected in the way that compliance with the total metal permit limit is determined. When f_d is used, the dissolved criterion is divided by f_d to assess permit compliance; however, when TDR is used, the dissolved criterion is multiplied by the TDR to determine permit compliance.

USEPA provides dischargers with the option of developing site-specific translators (i.e., determine the f_d ratio in the site water) for individual water bodies. Two procedures can be used to develop site-specific translators; complete guidance is provided by USEPA (USEPA 1996). The most straightforward approach for translating a metal criterion from a dissolved water quality criterion to a total recoverable effluent concentration is the direct analysis approach. The translator is the fraction of total recoverable metal that is dissolved, and can be determined directly by measuring the dissolved and total recoverable metal concentrations in water samples collected from the well-mixed effluent/receiving water matrix (i.e., at or below the edge of the mixing zone). In this approach, a number of samples are collected over time (e.g., monthly for a year), and an f_d value is determined for each sample, where $f_d=C_d/C_T$ [Eqn.1], where C_d is the dissolved metal concentration and C_T is the total metal concentration. The translator is then calculated as the geometric mean (GM) of the dissolved fractions.

In the second approach, f_d is derived using a partition coefficient, K_D. In this case, the coefficient is determined as a function of total suspended solids (although some other basis such as humic substances or particulate organic carbon may be used). The partition coefficient for the equilibrium $C_d+TSS=C_p$, where C_p is the bulk particulate-adsorbed metal concentration is expressed as:

$$K_D=C_p/(C_d \cdot TSS) \qquad \text{[Eqn.2].}$$

C_p equals C_T-C_d. Substituting for C_p in Eqn. 2, and rearranging, yields:

$$C_T/C_d\ (TDR) = 1+K_D \cdot TSS \qquad \text{[Eqn. 3], or}$$

$$C_d/C_T\ (f_d) = (1+K_D \cdot TSS)^{-1} \qquad \text{[Eqn. 4].}$$

As in the first approach, numerous samples are collected over time, and the f_d (or TDR) and TSS concentrations measured at the site are fit to a least squares regression, the slope of which is K_D. The established K_D is then used to determine the translator using Eqn. 3 or 4 and inserting a TSS value representative of some critical condition, e.g., low flow conditions.

3.6.1 Case History at Naval Bases in Virginia (Strom, 1998)

The chemical translator approach was used to develop appropriate chemical translators (ratios of total recoverable metal fractions to dissolved metal fractions) at eight locations near Newport, Virginia.

Samples were collected 15 times from each sampling site, approximately every two weeks. Consistent with Virginia Department of Environmental Quality (VDEQ) guidance, clean collection and analysis techniques were used (which is critical when addressing water quality criteria in the parts per billion range). The Navy is working with VDEQ to finalize preliminary results.

The copper translator averaged 1.47 (ranging from 1.21 to 2.18); the cadmium translator averaged 1.32 (ranging from 1.06 to 1.71), and the zinc translator averaged 2.13 (ranging from 1.31 to 3.61).

3.6.2 Case History at Lower South San Francisco Bay (RWQCB, 2002)

Chemical translators (ratio of dissolved to total recoverable metal fractions) were developed for copper and nickel based on the analysis of biweekly samples from 12 sites in lower South San Francisco Bay (Dumbarton to sloughs), February 1997 to August 2000.

The dataset, including the variables site, season, TSS, and tide, was analyzed using the CART 3.6.3 (Classification and Regression Trees) program which uses decision trees to display how data may be classified or predicted. These decision trees built by CART were used to identify the important parametric conditions in classifying the translator, f_d, values.

Chemical translator data were analyzed via CART analysis using the translator as the response variable and site, season (wet or dry), TSS, and tide as input variables. Based on the analysis, translators in lower South San Francisco Bay did not show strong temporal variation due to season or tides. CART analysis also indicated the site was not a good predictor of the translator value. The analysis did, however, indicate that chemical translators from sites 11 and 12 were significantly lower than the other 10 sites. Because of this, these sites were dropped from all subsequent translator calculations because they were deemed not representative of conditions in lower South San Francisco Bay.

CART and other analyses indicated that the most important variable in predicting the chemical translators for copper and nickel was TSS. Because of the dependence of these translators on TSS, regression equations relating f_d to TSS were developed. Linear regressions with log TSS and log translator (including data from all sites except 11 and 12; n = 508 points) provided the best fit, with r^2 values of 0.715 achieved for both metals. The following relationship was derived for copper:

$$\log \text{Translator} = 0.514 - 0.482 \log \text{TSS}$$

95% confidence intervals of the linear regression (representative of TSS values for the sites under consideration) were:

upper bound-
log Translator = 0.570 – 0.454 log TSS

lower bound-
log Translator = 0.458 – 0.511 log TSS

The following relationship was derived for nickel:

log Translator = 0.439 – 0.500 log TSS

95% confidence intervals of the linear regression were:

upper bound-
log Translator = 0.508 – 0.465 log TSS

lower bound-
log Translator = 0.369 – 0.536 log TSS

Table 3 compares the chemical translators for copper and nickel based purely on the ratio of dissolved to total recoverable metal, f_d, and in relation to its relationship with TSS.

Table 3. Chemical Translators for Copper and Nickel (Ratio of Dissolved to Total Recoverable Metal Fractions) in Lower South San Francisco Bay, CA.

		Linear Regression with TSS[a]		
Metal	f_d	Median	Upper bound	Lower bound
Copper	0.45	0.42	0.53	0.32
Nickel	0.33	0.32	0.44	0.24

[a] Translator based on a median TSS of 72 mg/L for 10 stations in lower South San Francisco Bay.

The California Regional Water Quality Control Board recommended a translator of 0.53 and 0.44 for copper and nickel, respectively, based on the upper bound of the linear regression equation and the median TSS value for the 10 sites in lower South San Francisco Bay of 72 mg/L.

3.7 Selenium Water Quality Objective

USEPA's plans to recommend a national chronic water quality objective for selenium that is derived using a different approach from most other water quality objectives. Selenium is bioaccumulative, and diet has been proven to be the primary route of exposure causing chronic toxicity to aquatic life. These two characteristics of selenium have warranted the use of field data to derive the chronic objective, and the exclusion of laboratory data in which organisms are exposed to selenium solely through the water. As a result, the selenium chronic objective will not be developed in the typical manner of using a toxicity test dataset compiled from acceptable tests, as defined in the Guidelines. The standard procedures for

developing site-specific objectives, presented in Sections 3.3, 3.4 and 3.5 of this document, use methodologies designed to modify water quality objectives based on the typical approach of using toxicity datasets (as prescribed by the Guidelines); these methodologies do not apply for selenium.

In addition to the USEPA's recommended national chronic criterion for selenium, a task force consisting of USGS, the Services, USEPA Region 9 and USEPA headquarters are developing selenium water quality criteria for all of California's watersheds. The goal of the criteria will be to protect aquatic ecosystems and their components including wildlife and threatened and endangered species. The Great Lakes Initiative Wildlife Methodology will be used to calculate water column values to protect wildlife. A risk assessment will be performed to ensure the wildlife criteria will be protective of aquatic life including threatened and endangered species. The selenium criteria developed by the Inter-Agency task force is expected to be proposed in a water quality standards package in 2006.

3.8 Determination of a Site-specific Increase in the Average Frequency of Allowed Excursions of a Water Quality Objective

An alternative to the modification of a water quality objective is to increase the number of times the objective can be exceeded over a three-year period (return period). This approach can be used for any water quality objective for the protection of aquatic life. It is limited to those chemicals for which effect concentrations are known for tissues of aquatic organisms. USEPA recommends a once in three year average frequency for excursions of both acute and chronic criteria (USEPA 1991a). The purpose of the average frequency of allowed excursions is to provide an appropriate period of time (on the average) for the aquatic community to 1) recover from the effects of an excursion; and 2) to function normally for a period of time before the next excursion. As stated in USEPA's *Technical Support Document for Water Quality-based Toxics Control* (TSD), the allowable frequency of excursions depends on site-specific factors, and more frequent excursions may be acceptable in certain situations. The ability of the aquatic community to withstand or recover from an excursion of a water quality criterion is dependent on a variety of toxicological and site-specific conditions, such as the toxicity of the contaminant, the magnitude and duration of the excursion, water quality characteristics, the type of aquatic species present and the presence (or lack) of refugia from which the aquatic community can be repopulated.

A two-step approach to develop a site-specific increase in the average frequency of allowed excursions for a water quality objective is as follows:

1) Conduct a field assessment to determine whether the historical concentrations discharged from an outfall[7] have caused impairment to the downstream biological community; and

2) If step 1 demonstrates that there has been no impairment to the downstream biological community, determine a site-specific average frequency of allowed excursions for the chemical of interest based on the historical effluent discharge and ambient data.

This two-step approach can be used to allow an effluent discharge to comply with the State's water quality objective. General guidance with a hypothetical example is provided below.

[7] The scenario used for this procedure uses an outfall as the entity for which a site-specific modification is desired. The procedure can be applied to non point sources or for ambient waters.

Step 1. Assessment of Risk to the Downstream Aquatic Community as a Result of the Exposure to the Chemical of Interest Discharged from the Outfall

Conduct an assessment in the receiving stream to determine if the chemical of interest is accumulating in resident macroinvertebrates and/or fish to unacceptable concentrations. Compare the residue levels in the downstream organisms to "safe levels" reported in the literature to determine if the downstream organisms are at risk.

Step 2. Determine a Site-specific Average Frequency of Allowed Excursions for the Chemical of Interest

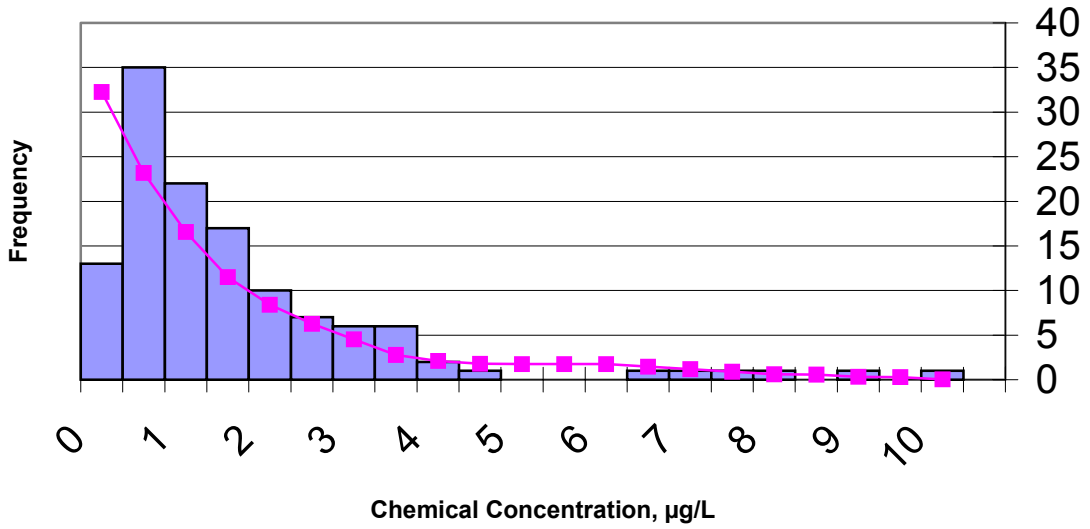
If the results of the risk assessment in Step 1 indicate that the discharge concentrations of the chemical of interest from the outfall are not causing impairment to the downstream aquatic community, Step 2 can be performed. Step 2 consists of an analysis of the frequency of excursions[8] of concentrations of the chemical of interest discharged from the outfall prior to the collection of macroinvertebrates and fish sampled for residue analysis. This analysis will determine the average number of allowable excursions over a three year period.

Figure 1 shows a distribution of chemical concentrations over a three year period. The bars in the figure represent the number of times the measured chemical concentration was observed in an 8-year period. The line represents the probability that each chemical concentration range (<1, 1-2, 2-3, etc.) will be exceeded for any given sample, expressed as the number of times the concentration was exceeded during three years of sampling and analysis. The probabilities of exceeding a given chemical concentration were calculated by determining the inverse of the cumulative distribution of the observed values. This distribution assigns zero probability to observing any value higher than has been observed in the past. To account for the possibility of higher values in the future, the inverse cumulative distribution was calculated with 10 percent exponential smoothing. The effect of this manipulation is to smooth the curve and to "shift" the distribution slightly to the right, while retaining the basic shape of the observed relationship.

For a hypothetical water quality objective of 5 µg/L, the line representing the probability that a given chemical concentration will be exceeded over a three year period indicates that the water quality objective will be exceeded two times during that time period (Figure 1). Based on this evaluation, it can be concluded that two excursions of the 5 µg/L criterion over a three year period will not impair the downstream uses. It is important to note that this evaluation does not necessarily reflect the capacity of the system with respect to the number of allowable excursions over a three year period. This evaluation was based on the historical chemical concentrations which have not caused impairment; it is possible that a greater number of excursions over a three year period will not impair the downstream uses. To ensure that water quality standards are being attained, that is, the downstream uses are being protected, macroinvertebrate and/or fish tissue should be analyzed for residues if there are two or more excursions over any three year period.

[8] An excursion in this case is defined as when the concentration of the chemical of interest from the outfall exceeds the water quality objective.

Figure 1. Distribution of Chemical Concentrations and Expected Excursions in Three Years



3.9 Establishing Site-Specific Water Quality Objectives using Natural Background Concentrations of Toxics

Toxics can be found naturally in surface waters in concentrations above the national criteria, as published under section 304(a) of CWA. Natural background concentrations of contaminants imply that there is no input of contaminants from anthropogenic sources. In those cases, the USEPA policy, issued in a 1997 memorandum from Tudor Davies (Office of Science and Technology), recommends that SSOs for aquatic life protection be derived using the natural concentrations of the toxic chemicals. This memorandum can be reviewed at USEPA's website at http://www.epa.gov/waterscience/library/wqcriteria/naturalback.pdf. However, methods to establish or test for compliance with SSOs based on natural background concentrations have not yet been published by USEPA.

The procedure provided in Appendix I describes the application of statistical techniques to test for violations of SSOs based on variation in natural background chemical concentrations. The premise for this procedure is that the assemblage of organisms that reside at the site are naturally exposed to a range of concentrations of the substance, and are not impaired by the chemical. It is assumed that a significant departure from the range of concentrations typically found at the site is a potential risk to the biological community, although there may or may not be any biological information to support this assumption.

The statistical procedure in Appendix I can be used to assess if the downstream site contains a significantly greater concentration than the upstream site. The recommended statistical procedure (Appendix I) for verifying that impairment is an option, but not the only one. The direction to compare data for upstream/downstream sites, or water bodies and reference sites, may not be appropriate for springs and "unique" internally drained saline lakes. In general, the obvious absence of significant

anthropogenic sources of toxic/radioactive elements, together with knowledge of local geology, should be adequate evidence that sources are natural.

3.10 Biological Assessment

Biological assessments are an evaluation of the biological condition of a water body using biosurvey data and other direct measurements of resident biota in surface waters. Bioassessment data can be used to set protection and restoration goals, to decide what in-stream parameters to monitor and how to interpret what is found, to identify stresses to the water body and decide how they should be controlled, and to assess and report on the effectiveness of management actions. Bioassessment is also the tool by which biological criteria are derived. Biological criteria are narrative or numeric expressions that describe the reference condition (structure and function) of aquatic communities inhabiting waters of a given designated aquatic life use. Biocriteria are based on the numbers and kinds of organisms present and are regulatory-based biological measurements.

Many states, including California, are in the process of developing biological criteria as part of their water quality standards program. The California Department of Fish and Game (DFG) has an active bioassessment program administered by the California Aquatic Bioassessment Workgroup (CABW). CABW has been active in defining and testing sets of procedures for sampling aquatic communities, establishing reference conditions, and developing quality assurance and quality control procedures. Several other acceptable protocols for bioassessment are either being developed or used throughout California (e.g., Region 6, U.S. Forest Service).

With respect to the development of SSOs, biological assessment data can be used to address a number of issues. Prior to beginning an SSO study, a biological assessment of the site of interest relative to a reference condition can be used to indicate whether or not the site is biologically impaired. Such information could be used to determine if the site is a candidate for SSO development. For example, the first two of the following three outcomes could result in a situation where an increase in a water quality objective is desired at a site where the water quality objective is intermittently or consistently exceeded; whereas the third situation suggests that an SSO may not be appropriate: 1) if the site is not impaired, the State would have the assurance that the aquatic life uses of the water body are being attained at concentrations greater than the water quality objective, and could grant approval to begin the study; 2) if the site is impaired, and lines of evidence (e.g., chemical data, sediment and/or ambient water toxicity identification evaluations, signature responses, etc.) indicate the probable cause of the impairment was not due to the chemical of interest, the State could approve beginning the site-specific development process with a requirement to conduct follow-up biological assessment (see Conditions, Section 2;) 3) if the site is impaired, and lines of evidence indicate the probable cause is due in part or in whole to the chemical of interest, the State could disapprove beginning the site-specific development process.

Biological assessment data could also be used in the development process in defining what species "reside at the site" in the Recalculation Procedure (see Section 3.4). The term "reside at the site" refers to species that could be residents at the site if there was no biological impairment. As part of its mission, CABW establishes reference sites, and has proposed a procedure for assessing reference sites. As stated on CABW's website, http://www.dfg.ca.gov/cabw/cabwhome.html, reference sites represent the minimally disturbed condition (MDC) or least disturbed condition (LDC) of a water body. MDC sites represent the absence of significant human disturbance. MDC sites change little over time, mostly due to

natural processes, and thus provide a stable benchmark. Historical information may be used to help describe MDC.

The LDC represents the best available physical, chemical, and biological conditions given today's state of the landscape. The LDC will change over time as land use and management practices change, and thus is not a "target" or upper bound of water quality potential. Therefore, the assemblage of organisms found at reference sites would represent the list of taxa that would "reside at the site" of interest in an SSO study. Of course, the reference site used in this context should be located in the same ecoregion of the site of interest and have similar size characteristics (e.g., headwater stream, wadeable stream, etc.). Information on the procedures used to assess a reference site and information on reference sites located within California can be found on the CABW website cited above.

The most decisive method for determining if an SSO is protecting a beneficial use (i.e., aquatic life use) of a water body is biological assessment. In cases where there is uncertainty about the level of protection afforded by an SSO, a follow-up monitoring program using the biological assessment procedures provided in CABW's website provides the opportunity to perform an accurate assessment of the SSO.

4 ENDANGERED SPECIES

4.1 Background

Surface waters protected under the CWA are critical to the survival of threatened and endangered species. It is currently estimated that 85 percent of all Federally-listed species (1,258 total in the United States) use wetlands and aquatic habitats, and pollution is a factor contributing to their listing as threatened or endangered in over half of the cases (USEPA 2002). Approximately 60 percent of the total number of threatened and endangered species listed in the United States are plants (for a general summary of information on Federally-listed species go to http://endangered.fws.gov/wildlife.html#Species). Currently, 19 amphibians, 115 fishes, 70 clams, and 21 crustaceans appear on the United States endangered and threatened species list. California alone accounts for 292 of the 1258 species that are listed. Federally listed species are given at http://ecos.fws.gov/webpage/webpage_usa_lists.html?state=CA. Additionally, there are a number of threatened/endangered species listed under the CESA that are not federally listed. The CDFG maintains a database (Rarefind) of known occurrences of both state and federally listed species.

4.2 Memorandum of Agreement between USEPA and the Services

On January 18, 2001, the USEPA and the Services signed a Memorandum of Agreement (MOA) designed to improve methods for coordinating compliance with sections 303(c) and 402 of the CWA, and Section 7 of the Endangered Species Act (ESA). The MOA provides clear mechanisms for protecting and promoting the recovery of Federally-listed species (threatened and endangered) and their supporting ecosystems (USEPA 2001c). The MOA can be downloaded from USEPA's website at http://www.epa.gov/waterscience/standards/esa.html. Under section 7 of the ESA, any action authorized, funded, or carried out by USEPA (or other Federal Agency) that may adversely affect a Federally-listed species is subject to the consultation requirements set forth in the ESA. The development and

implementation of site-specific water quality objectives by the State or authorized Tribes are affected by this MOA because USEPA approves State and Tribal water quality standards and total maximum daily loads[9], and either approves NPDES delegation to States and Tribes, or issues NPDES permits, all of which constitute Federal actions. Under the MOA, EPA will seek the assistance and comment of the Services during a State's or Tribe's development of water quality standards. The subject of this section, however, pertains only to the development of site-specific water quality objectives in water bodies where Federally-listed species reside (e.g., use and selection of surrogate species and/or tests to adjust national criteria), and not the coordinated efforts of EPA and the Services to comply with the ESA *per se*.

4.3 Identification of Threatened and Endangered Species

To ensure compliance with the objectives of the ESA and the CESA (i.e., protecting and promoting the recovery of threatened and endangered species and their supporting ecosystems), special consideration must be given to Federally-listed and State-listed species in the bodies where they are located. Water quality objectives may be under- protective if the listed species is more sensitive than the species included in the national data set. As a result, derivation of SSOs first requires identification of any Federally-listed and State-listed species expected to reside at the site. Because some of these species may reside either seasonally or intermittently at the site, and may be excluded from a site because of existing temporary conditions including pollution, it is important to research their possible presence in adjacent water bodies, or perhaps even the entire ecoregion. Federally-listed species within each state can be found at http://endangered.fws.gov/wildlife.html#Species.

The "Rarefind" database, maintained by the CDFG, is a good starting point for determining if any listed species occur in the area of the proposed SSO. The Division of Water Rights (DWR) of the SWRCB holds a license to use the database. Regional Boards can contact the DWR in order to obtain a copy of the database. Currently, MaryLisa Lynch is the contact person at DWR. She can be reached at (916) 341-5365 or Mlynch@waterrights.swrcb.ca.gov. Other agencies or individuals will need to contact the CDFG's Wildlife and Habitat Data Analysis Branch at: 1807 13th Street, Suite 202, Sacramento, CA 95841; phone (916) 322-2493; web address: http://www.dfg.ca.gov/whdab/index.html. Additionally, the regional offices of the NMFS, the USFWS, and the CDFG should be contacted for any current information on listed species.

4.4 Assessing Toxicity to Listed Species

Once a listed species has been identified as a current or potential inhabitant in a specific water body or ecoregion, an analysis (biological evaluation) must be performed to determine if the SSO is likely to jeopardize the continued existence of that species. If it is, the objective must be modified to protect that individual species. This may be difficult because of the paucity of appropriate toxicological data regarding many of the Federally-listed species. The primary reasons for the lack of toxicity test data using Federally-listed species are: 1) low population numbers; 2) lack of techniques to maintain and propagate the species; and 3) management and procedural issues associated with the use of Federally-

[9] Total maximum daily loads are essentially a mass balance of a pollutant flowing in and out of a watershed (Whittenmore and Ice, 2001). They are used to calculate the maximum amount of a pollutant that a water body can receive and still meet applicable water quality standards.

listed species in toxicity tests. Indeed, standardized toxicity test procedures based on the concentration-response approach generally require large numbers of test animals to attain sufficient statistical power. An alternative testing approach is the use of time-response models, which predict effects as a function of exposure duration and concentration (see Chew and Hamilton (1985), Newman and Dixon (1996), and Newman and McCloskey (1996) for a general discussion). Time-response models using time-to-death (TTD) as the measured response variable hold the advantage over standardized concentration-response methodologies because the endpoint is associated with an individual, thus the overall numbers of animals required for testing could be markedly reduced. If toxicological data are not available, or simply cannot be generated for the listed species, selection of appropriate ***surrogate*** toxicity data may be necessary to estimate the level of protection needed for the species of interest for a given chemical. Selection of an appropriate surrogate is important, since acute toxicity differences between the least and the most sensitive aquatic species (freshwater) for tested chemicals average two orders of magnitude (Mayer and Ellersieck 1986).

4.4.1 Historical Use of Surrogate Toxicity Data in the Water Quality Standards Program

Use of toxicological data from tests with surrogate species to protect resident species is integral to the water quality standards program, and standard for national criteria development. The premise for such a convention is based primarily on the argument that if acceptable data are available for a large number of taxa from a variety of taxonomic and functional groups, a reasonable level of protection will be provided (Stephan et al. 1985). This argument is relevant, unless an ecologically, commercially or recreationally important species, or threatened or endangered species, is very sensitive. The current eight family rule given in the guidelines (Stephan et al. 1985) is restrictive in that in many cases it does not represent all the taxa residing at a locale, but it is expected to represent the sensitivities of untested resident species, including those that are endangered or threatened.

4.4.2 Considerations Regarding Use of Surrogates and Species Sensitivity

Few studies compare the contaminant sensitivities of related species within general taxonomic groups (e.g., order, family), and even fewer studies have evaluated the sensitivity of Federally-listed species within a genus (Sappington et al. 2001). Most of the available data are for fish, and toxicological data on endangered species within other phyla, e.g., Arthropoda and Mollusca are lacking. Out of 4,901 acute toxicity tests with 410 chemicals and 66 freshwater species, the rank order of sensitivity (most to least) of the seven most commonly tested families was as follows: Pteronarcidae (stoneflies), Daphnidae, Gammaridae, Salmonidae, Centrarchidae, Ictaluridae, and Cyprinidae (Mayer and Ellersieck 1986). The class Insecta was the most sensitive 50 percent of the time, followed by Crustacea (31 percent), Osteichthyes (19 percent) and Amphibia (0 percent).

The most comprehensive study to date comparing the sensitivity to contaminants of endangered fish species with surrogate species is that of Sappington et al. (2001). The authors employed standard static acute toxicity test procedures to compare the sensitivity of rainbow trout (the most common cold water fish test species), fathead minnows (most common warm water fish test species), and sheepshead minnows (the most common saltwater fish test species) as surrogate test species for the following Federally-listed species: Apache trout, Lahontan cutthroat trout, greenback cutthroat trout, bonytail chub, Colorado pikeminnow, razorback sucker, Leon Springs pupfish, and desert pupfish. These species

represent the families Salmonidae, Cyprinidae, and Cyprinodontidae. Chemicals tested were carbaryl, copper,
4-nonylphenol, pentachlorophenol, and permethrin. The sensitivities (denoted by the 96-hr LC_{50} value) of the listed species were within a factor of two of the surrogate test species (i.e., rainbow trout, fathead minnow, and sheepshead minnow) in all but two of 36 possible outcomes. Furthermore, the 96-hr LC_{50} values for the listed species were lower than that of their surrogates in only about 10 of the surrogate/listed species comparisons. Based on these findings, the authors concluded that a safety factor of 0.5x the SMAV could be used as a conservative estimate for listed cold-water, warm-water, and euryhaline species.

4.4.3 Deriving Toxicity Values from Surrogate Data for Aquatic Species

Guidance on assessing whether the 304(a) aquatic life criteria are protective of Federally-listed species is currently in development for the national consultations between USEPA and the Services. Although the methodology used in the national consultations was not designed with the intent for criteria development, it does provide several methods for deriving toxicity values where there are no chemical specific data for a given species and where only surrogate data are available. It should be noted that where toxicity data are available for the species of interest, only these data should be used for designating an effects concentration for a given species, and it will not be necessary to rely on the following methods, which rely on surrogate data. The State of California has decided to incorporate these methods for deriving toxicity values from surrogate data into this Guidance.

4.4.3.1 Methods for Deriving Toxicity Values from Surrogate Data for Aquatic Species

The following methods from the national consultations are useful for deriving both acute and chronic toxicity values from surrogate data, although they are described below in the context of acute toxicity.

(1) Interspecies Correlation Estimate (ICE) Method: Interspecies Correlation Estimates (ICEs) can be used up through the taxonomic level of family. Species to species ICEs are based on regression analyses of LC_{50}s measured for a given species to LC_{50}s measured for the same chemicals for commonly used surrogate species, preferably based on a minimum of five tested chemicals. If the surrogate species has been tested for the chemical of interest, but the given species of interest has not, such relationships can be used to estimate the LC_{50} for the chemical and species of interest. When there is no ICE for the given species, an ICE for its genus or family may be used. These higher taxa ICEs are derived the same as for individual species, except that each genus or family must be represented by at least two species. Due to the uncertainty in such correlations, the LC_{50} estimates can be based on a lower confidence bound of the ICE.

(2) Direct Percentile Estimation (DPE) Method: If several surrogates within the same taxonomic unit as the species of interest have been the subject of acute toxicity tests, they can be considered to be a sample of the toxic sensitivities within that taxonomic unit and be used to estimate the distribution of those sensitivities, and thus estimate the LC_{50}s for the species of interest. A percentile in this distribution is then specified. Accordingly, the actual toxicity for the species of interest has a specified likelihood of being greater than this estimated percentile. There should be at least four surrogates within a taxonomic group in which the species of interest belongs in order to use DPE. Variability between species generally becomes too great above the class level to reliably use the DPE method. DPE estimates must be

represented by at least two taxa from the next lowest level. (For example, a DPE estimate at the family level must include data from at least two genera within that family).

Any surrogate method, works better for surrogates that are taxonomically close to the species of interest. For example, if four or more surrogates within the genus of the species of interest have been tested, the DPE method would be applied to these, ignoring more distantly related surrogates. If a sufficient number of surrogates are not available within the genus, then the taxonomic unit would expand progressively to family, order, and class until four or more surrogates are available. A similar progressive prioritization can be applied to the ICE method through the family level.

Because the DPE procedures require at least four tested species within the taxonomic level at which calculations are made, they may sometimes overestimate or underestimate risk to the species of interest. Risk may be overestimated when the data used in DPE calculations include a high proportion of sensitive members of a broad taxon when more closely related species (but less than the required four) are known to be tolerant. When such extreme deviations are noted (e.g., where there is more than a 10-fold difference between a given percentile estimate and the available data within a taxonomic group), the analysis can consider further information regarding whether the selected LC_{50} is over protective or under protective. This could involve examination of the relative sensitivities within and between the taxa in question for other similar chemicals.

4.4.3.2 Acute Toxicity Endpoints

Once an LC_{50} is selected or estimated for the species of interest, it should be reduced because 50 percent mortality is a much greater-than-desired level of effect for this risk assessment. LC_{50}s can be divided by two, which generally results in a concentration near or below the lethality threshold.

4.4.3.3 Chronic Toxicity Endpoints and Extrapolations

For most tests, the No Observed Effect Concentrations (NOECs) can be used. However, for tests with widely-spread concentrations, the NOEC can be lower than the true chronic effects concentration (EC) that represents no effect. Therefore, if the actual NOEC is more than two-fold below the Lowest Observed Effect Concentrations (LOEC), the concentration used for these tests will be the Chronic Value divided by 1.4. This results in a concentration that is above the NOEC, but which approaches the NOEC as the dilution factor approaches two-fold.

Where chronic toxicity data are not available for the species of interest it will be necessary to estimate chronic EC values. In estimating chronic EC values, acute-to-chronic ratios (ACRs) are calculated for each species that has both acute LC_{50} and chronic EC toxicity data (i.e., ACR=LC_{50}/chronic EC). The relationship of these ACRs to species sensitivity and taxonomy are then evaluated. If no significant relationships are found, the Final Acute-to-Chronic Ratio (FACR) from the criteria document will be applied to any species as needed. If significant relationships do exist as a function of species sensitivity and/or taxonomy, chronic ECs will be estimated from acute LC_{50}s based on a linear regression model.

4.4.4 Deriving Toxicity Values from Surrogate Data for Aquatic-Dependent Species

In assessing risk to aquatic-dependent species, it will be necessary to identify an exposure concentration, since exposure occurs as residue levels in the prey organisms, as well as determining the effects concentration. In both cases (i.e., exposure and effects assessment) there is a range of values from which to choose: for the exposure assessment, there will be a range of bioaccumulation factors (BAFs), and for the effects assessment, there will be a range of surrogate dietary effects concentrations. A point in these ranges is then selected that represents an appropriate level of risk in protecting a given species. In identifying the range of BAFs, appropriate BAFs are those based on water column concentrations at or near the chronic criterion (CCC).

4.4.5 Exposure Concentrations in the Water Column

Pollutant concentrations in the water column are assumed to be at the acute criteria for very short durations and at the chronic criteria for the remainder of the time.

5 HUMAN HEALTH-BASED OBJECTIVES

Human health-based objectives are designed to protect humans from the adverse effects of pollutants. With the current exception of methylmercury, all human health objectives are in the form of AWQC, and are derived using equations that include toxicological and exposure parameters. *The Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (USEPA 2000a) provides the following three equations for deriving AWQC based on noncancer effects and cancer effects (nonlinear and linear low-dose extrapolation):

Noncancer Effects

$$AWQC = RfD * RSC * \left(\frac{BW}{DI + \sum_{i=2}^{4} \left(FI_i * BAF_i \right)} \right)$$

Cancer Effects: Nonlinear Low-Dose Extrapolation

$$AWQC = \frac{POD}{UF} * RSC * \left(\frac{BW}{DI + \sum_{i=2}^{4} \left(FI_i * BAF_i \right)} \right)$$

Cancer Effects: Linear Low-Dose Extrapolation

$$AWQC = RSD * \left(\frac{BW}{DI + \sum_{i=2}^{4} (FI_i * BAF_i)} \right)$$

where:

AWQC = Ambient Water Quality Criterion (mg/L).

RfD = Reference dose for noncancer effects (mg/kg-day).

POD = Point of departure for carcinogens based on a nonlinear low-dose extrapolation (unitless).

UF = Uncertainty factor for carcinogens based on a nonlinear low-dose extrapolation (unitless).

RSD = Risk-specific dose for carcinogens based on a linear low-dose extrapolation (mg/kg-day) (dose associated with a target risk, such as 10^{-6}).

RSC = Relative source contribution factor to account for non-water sources of exposure. (Not used for linear carcinogens.) May be either a percentage (multiplied) or amount subtracted, depending on whether multiple criteria are relevant to the chemical.

BW = Human body weight (default = 70 kg for adults).

DI = Drinking water intake (default = 2 L/day for adults).

FI_i = Fish intake at trophic level (TL) I (I = 2, 3, and 4) (defaults for total intake = 0.0175 kg/day for general adult population and sport anglers, and 0.1424 kg/day for subsistence fishers). TL breakouts for the general adult population and sport anglers are: TL2 = 0.0038 kg/day; TL3 = 0.0080 kg/day; and TL4 = 0.0057 kg/day.

BAF_i = Bioaccumulation factor at TL I (I = 2, 3, and 4), lipid normalized (L/kg).

The National Recommended Water Quality Criteria: 2002 Human Health Criteria Matrix contains information regarding the calculation of human health criteria (http://epa.gov/waterscience/criteria/hh_calc_matrix.pdf). The information provided in this document includes RfDs, RSCs, fish intake values and BCF/BAF values used to calculate the criteria using the equations given above.

The 2000 Human Health Methodology allows for site-specific modifications by states and Tribes to reflect local environmental conditions and human exposure patterns. USEPA states that site-specific criteria may be developed as long the site-specific data, either toxicological or exposure-related, is

justifiable. For example, when using a site-specific fish consumption rate, a state should use a value that represents at least the central tendency of the population surveyed (either sport or subsistence fishers, or both). If a site-specific fish consumption rate for these categories is lower than a USEPA default value, it may be used in calculating the AWQC. However, to justify such a level (either higher or lower than USEPA default values), the State should assemble appropriate survey data to arrive at a defensible site-specific fish consumption rate. USEPA has published the "Guidance for Conducting Fish and Wildlife Consumption Surveys," which provides guidance on methods for obtaining consumption rates for use in developing or modifying water quality standards. This document can be accessed on USEPA's website at http://www.epa.gov/waterscience/fish/fishguid.pdf.

Appropriate justification is also needed for site-specific modification of the AWQC using environmental factors such as BAF and percent lipid. Considerations and guidance on developing a site-specific BAF is given in the following section. In the case of deviations from toxicological values (i.e., Integrated Risk Information System (IRIS) values: verified noncancer and cancer assessments), USEPA strongly recommends that the data upon which the deviation is based be presented to and approved by USEPA before a site-specific criterion is developed.

California's Office of Environmental Health Hazard Assessment has issued the publication *Chemicals in Fish: Consumption of Fish and Shellfish in California and the United States* dated October of 2001. The Final Report is based on an extensive review of fish consumption surveys conducted throughout the United States, and includes a critical evaluation of these studies. The information and recommendations contained in the report are planned to be used for supporting the functions and activities of various state programs including development of fish advisories, development of water quality objectives for environmental contaminants, and other exposure-related assessments. The document is available at: http://www.oehha.ca.gov/fish/special_reports/fishconsum.html#download.

5.1 Site-Specific Bioaccumulation Factors

[*Note: Development of site-specific bioaccumulation factors is complex, has not yet been implemented in California, and can be a fairly costly process. Before beginning an effort of this magnitude, it would be prudent to determine whether this process is acceptable to stakeholders.*]

BAFs are partition coefficients that describe the concentration of the chemical in the aquatic organism's tissues relative to the concentration of the chemical in the water. BAFs reflect the degree to which a chemical will accumulate in the tissues of an aquatic organism when that chemical is present in the aquatic environment. National BAFs are measured or predicted using some or all of the following four methods, depending on the type of chemical and its properties.

1. a measured BAF obtained from a field study;

2. a BAF predicted from a field-measured biota-sediment accumulation factor (BSAF);

3. a BAF predicted from a laboratory-measured bioconcentration factor (BCF), with or without any adjustment from a food-chain multiplier (FCM);

4. a BAF predicted from a chemical's octanol-water partition coefficient (Kow), with or without a FCM.

Although up to four methods can be used to derive a BAF, the type of chemical will influence the method selected. Chemicals that bioaccumulate can be classified into three categories: nonionic organic chemicals, ionic organic chemicals, and inorganic chemicals. USEPA's *Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health* (2000) provides guidance on deriving BAFs for each type of chemical. This document can be accessed on USEPA's website at www.epa.gov/waterscience/humanhealth/method/covertoc.pdf.

The procedures to derive BAFs in the 2000 Human Health Methodology allow states and Tribes the ability to make adjustments to national BAFs to reflect local conditions. Site-specific conditions that can affect the BAF, such as lipid content in locally consumed aquatic biota and the organic carbon content of the surface water and/or sediment, are accounted for in field-measured BAFs and BSAFs. Guidance and data for adjusting national BAFs to site-specific conditions will be provided in the Bioaccumulation Technical Support Document, a document USEPA expects to publish in 2003. USEPA also plans to publish detailed guidance on designing and conducting field bioaccumulation studies for measuring BAFs and BSAFs. The purpose of the field guidance document is to provide guidance for state and tribal resource management agencies to obtain the information required to generate water body-specific BAFs and BSAFs for organic hydrophobic chemicals, which can be used to derive baseline BAFs. The guidance will include approaches and methods which are applicable to virtually any water body in the United States. The document also will describe how an effective field sampling program should be structured, including recommendations and considerations regarding field sampling procedures, laboratory methods, statistical analyses and data quality requirements for the generation of scientifically supportable BAFs and BSAFs. By following the recommendations in the document, investigators will foster comparability and utility of data across agencies, especially in situations involving interstate waters. A case study for the development of a site-specific BAF for benzo[a]pyrene followed the approach and procedures that will be provided in USEPA's field guidance (Appendix J).

5.1.1 Methylmercury

[*Note: At the time of this document publication, USEPA and California have not decided on a water quality objective for mercury. This section presents USEPA's published recommendation for methylmercury. California may decide to use another approach.*]

In 2001, USEPA published a recommended fish tissue criterion of 0.3 mg methylmercury/kg to protect human health. USEPA recognizes that a state's water quality objective in the form of a fish tissue residue value may pose implementation challenges under traditional water quality-based control programs (USEPA 2001a). Under a water quality-based approach to controlling pollutants, NPDES permit compliance with water quality standards is usually determined by comparing the allowable concentration of a pollutant in the water column to the actual pollutant concentration measured in the water column over some specific period of time. Mechanisms to control pollutants in water bodies usually involve determining the allowable discharge load to a water body by conducting Total Maximum Daily Amount (TMDL) and wasteload allocation (WLA) calculations. Because the traditional approach for monitoring, measuring compliance, and controlling the discharge of a pollutant is based on the concentration of the pollutant in water, a mechanism is needed to relate concentrations of mercury in fish tissue to

concentrations in water. At present, methylmercury is the only contaminant for which there is a fish tissue value for the protection of human health.

USEPA's preferred approach for relating a concentration of methymercury in fish tissue to a concentration of mercury in ambient water is to derive site-specific BAFs based on water and fish collected in the water body of concern (USEPA 2001a). USEPA prefers the use of site-specific BAFs because they inherently incorporate the net effects of the biotic and abiotic factors at a particular location those factors can affect bioaccumulation in the aquatic food chain, and accounting for those factors provides an accurate perspective on the uptake of contaminants. When sampling fish and water to derive a site-specific BAF, it is important to consider how best to sample, so that issues such as seasonal variability in fish exposure, and fish size are taken into account, as well as temporal and spatial variability of methylmercury in the water.

Once a site-specific BAF has been determined for methylmercury, a dissolved SSO can be calculated using the following equation:

$$Methylmercury\ SSO, \mu g / L = \frac{0.3 \mu g\ methylmercury / g}{BAF\ value * 10^{-3}}$$

USEPA recognizes that states and authorized Tribes will need additional specific procedures and water quality program guidance in order to implement methylmercury water quality objectives based on a tissue residue level. These procedures include, but are not necessarily limited to: (1) an analytical method for detecting and measuring methylmercury concentrations in fish and water; (2) a field sampling plan for collecting fish and protocols for laboratory analysis and data interpretation; (3) a procedure for translating methylmercury concentrations in fish to total methylmercury concentrations in ambient surface water or effluent; (4) data quality objectives and associated procedures for determining attainment of the water quality objective and status of beneficial use impairment based on fish residue data; (5) harmonization with fish consumption advisory programs; (6) procedures for determining the need for a water quality-based effluent limit (WQBEL) in NPDES permits for point source discharges of mercury; (7) procedures for developing and implementing WQBELs for NPDES permits; and (8) procedures for developing targets for TMDL load and wasteload allocations.

6 WILDLIFE, SEDIMENT, AND NUTRIENT-BASED OBJECTIVES

Aside from aquatic life and human health criteria, there are other criteria and guidelines designed to restore and maintain the integrity of surface waters. These are treated separately here because the methodology and approaches for their use in deriving site-specific water quality objectives for a given water body or reach are not routinely employed, or are still evolving. Even though the available methods or approaches themselves are not used routinely for the intents and purposes of this document, they still require consideration as a potential tool for deriving site-specific water quality objectives in the future. The subsections below provide a brief introduction and summary of the information available for wildlife, sediment, and nutrient-based objectives.

6.1 Wildlife-based Objectives

Wildlife criteria are derived to establish ambient concentrations of chemicals which, if not exceeded, will protect mammals and birds from adverse impacts from that chemical due to consumption of food and/or water. USEPA's first water quality criteria specifically for the protection of wildlife have been developed for the Great Lakes System.

The Final Water Quality Guidance for the Great Lakes System (Final Rule, Federal Register: March 23, 1995, Volume 60, Number 56; http://www.epa.gov/owow/tmdl/glsprohibit.pdf) contains numeric criteria to protect wildlife from four pollutants, and a methodology to derive Tier I criteria for additional bioaccumulative chemicals of concern (BCCs).

The methodology from wildlife criteria is based largely on the noncancer human health model. It focuses, however, on endpoints related to reproduction and population survival rather than the survival of individual members of a species. The methodology incorporates pollutant-specific effects data for a variety of mammals and birds, and species-specific exposure parameters for two mammals and three birds representative of mammals and birds resident in the Great Lakes Basin which are likely to experience significant exposure to bioaccumulative contaminants through the aquatic food web.

Wildlife criteria for birds and mammals are developed using the following equation:

$$WV = \frac{\frac{TD}{UF_A \ x \ UF_S \ x \ UF_L} x \ Wt}{W + \sum (F_{TLi} \ x \ BAF_{TLi}^{WL})}$$

where:

WV = wildlife value in milligrams of substance per liter (mg/l);

TD = test dose in milligrams of substance per kilograms per day (mg/kg-d) for the test species; either a No Observed Adverse Effect Level (NOAEL) or a Lowest Observed Adverse Effect Level (LOAEL);

UF_A = uncertainty factor for extrapolating toxicity data across species (unitless). A species-specific UF is selected and applied to each representative species, consistent with the equation;

UF_S = UF for extrapolating from subchronic to chronic exposures (unitless);

UF_L = UF for LOAEL to NOAEL extrapolations (unitless);

Wt = average weight in kilograms (kg) for the representative species;

W = average daily volume of water consumed in liters per day (L/d) by the representative species;

F_{TLi} = average daily amount of food consumed from TL i in kilograms per day (kg/d) by the representative species; and

BAF^{WL}_{TLi} = bioaccumulation factor for wildlife food in TL i in liters per kilogram (L/kg), developed using the BAF methodology contained in rule 3745-1-37 of the Administrative Code. For consumption of piscivorous birds by other birds (e.g., herring gull by eagles), the BAF is derived by multiplying the TL three BAF for fish by a biomagnification factor to account for the biomagnification from fish to the consumed birds.

Similar to the human health-based objectives discussed in Section 5, site-specific considerations for wildlife criteria can be toxicological and exposure-related. Given the difficulty and expense in generating toxicological values for wildlife, the most likely approach for modifying a wildlife criterion is to determine a site-specific BAF. The guidance presented in Section 5.1 of this document would also apply to wildlife, with attention given to the target species for which the criterion is derived. For example, if the criterion is being derived to protect otters living in a coldwater stream, the BAF should be derived using the whole body tissue concentration for salmonids. If the criterion is intended to protect the belted kingfisher inhabiting the corridor of a warmwater stream, whole body concentrations of minnows and shiners should be used in the BAF determination.

6.2 Sediment Quality-based Objectives

Contaminated sediments are known to have profound impacts on the overall quality of our nation's waters (USEPA 2001d). According to the draft report titled: *Incidence and Severity of Sediment Contamination in Surface Waters of the United States, National Sediment Quality Survey* (http://www.epa.gov/waterscience/cs/surveyfs.html), 3.3 percent of the nation's waters have sediments where chemical concentrations in the sediments are probably causing adverse effects to aquatic life and pose a risk to human health through the consumption of contaminated fish and shellfish. Another 3 percent of the nation's waters are suspected of having sediments causing adverse effects, and 2.5 percent thought to be safe for aquatic life. There are virtually no data on sediments from the other 91.2 percent of the nation's waters. Furthermore, no national sediment quality criteria have been published to date, and only one state, Washington, has adopted Apparent Effects Thresholds (dry weight chemical concentrations above which effects have always been observed) for sediments into their state water quality standards program (www.ecy.wa.gov/programs/tcp/smu/sed_chem.htm).

The SWRCB is required under the California Water Code to develop sediment quality objectives for enclosed bays and estuaries. While the scope and objectives of this program have not been finalized, SWRCB staff anticipate that the tools and protocols developed and approved for use in the statewide program will assist RWQCBs in the development of site-specific sediment quality objectives. SWRCB staff also believe that the methods used to develop regional or statewide numeric objectives or threshold values will also be applicable to local or site specific situations. Currently, site-specific values are used for sediment cleanup operations by the coastal Regional Boards. These values are typically established using multiple indicator tools such as chemistry, benthic community, toxicity and bioaccumulation (weight of evidence approach) coupled with comparison to various sediment quality guidelines.

Several federal guidance documents for developing sediment objectives exist, and are provided for informational purposes in the following table.

Table 4. Guidance Documents for Establishing Sediment Quality Objectives.

Document	Publisher	Available at:
Prediction of sediment toxicity using consensus-based freshwater sediment quality guidelines. USEPA#: 905/R-00/007 YEAR: 2000	USGS/USEPA	http://www.epa.gov/waterscience/cs/guidelines.htm
Sediment Quality Guidelines Developed for the National Status and Trends Program. YEAR: 1999	NOAA	http://response.restoration.noaa.gov/cpr/sediment/SQGs.html
Technical Basis for Deriving Sediment Quality Criteria for Nonionic Organic Contaminants for the Protection of Benthic Organisms by Using EqP. USEPA#: 822R02041 Draft Update: 2002	USEPA	http://yosemite.epa.gov/water/owrccatalog.nsf/e673c95b11602f2385256ae1007279fe/a8c29814a7d047b485256c5900484d05?OpenDocument&CartID=13080-103246
Guidelines for Deriving Site-specific Sediment Quality Criteria for the Protection of Benthic Organisms. USEPA#: 822R93017 YEAR:1983	USEPA	http://yosemite.epa.gov/water/owrccatalog.nsf/065ca07e299b464685256ce50075c11a/1bc869bbc4728fb885256b0600723bd4?OpenDocument&CartID=13080-103246
1994 Florida Sediment Quality Assessment Guidelines (SQAGs) YEAR: 1994	FL Dept. Env. Protection	http://www.dep.state.fl.us/waste/quick_topics/publications/pages/default.htm
Technical Guidance for Screening Contaminated Sediments YEAR: 1993 Updated: 1994, 1998, 1999	NY Dept. Env. Conservation	http://www.dec.state.ny.us/website/dfwmr/habitat/seddoc.pdf
Outline of Tiered Decision Making Process in Establishing Sediment Quality Objectives and Making Sediment Management Decisions.	WI Dept. Nat. Resources	Available Upon Request

Note that the procedures mentioned above should only be used after appropriate sediment assessment procedures have been employed. For example, if bioassays demonstrate that a sediment is toxic, risk assessment procedures to identify the chemicals causing the observed effects, such as sediment toxicity identification evaluations, can be used (USEPA 1991b; Ho *et al*., 1997). Until such time that other appropriate tools can be developed, USEPA advocates the use of the sediment quality triad to address issues related to contaminated sediments (USEPA 2000b; information can also be found at http://www.epa.gov/waterscience/cs/). This weight-of-evidence approach relies on three types of

information (sediment toxicity, sediment chemistry, and benthic community abundance and composition) to help identify which chemical(s) could be causing toxicity. Again, this type of critical information should be obtained prior to any attempt to employ site-specific modifications to an ESB.

6.3 Nutrient-based Objectives

The National Nutrient Criteria Program was established to address the widespread problem of excessive concentrations of nutrients in surface waters. Approximately half of the nation's impaired streams, rivers, lakes, reservoirs, estuaries, and coastal marine waters are impaired as a result of nutrient enrichment (USEPA 2000c). High concentrations of nitrogen and phosphorus are the primary cause of eutrophication, which may lead to algal blooms, growth of macrophytes, shifts in species composition (flora and fauna), low dissolved oxygen, and fish kills. In addition to total nitrogen and total phosphorus concentrations (the causal variables), the National Nutrient Criteria Program calls for monitoring Secchi depth, chlorophyll *a* and algal turbidity. These response variables indicate the level of eutrophication. It is important to recognize, however, a water body with short retention time could be clear, and still transport an excessive load of nitrogen and/or phosphorus downstream.

The development of nutrient criteria incorporates the fundamental concepts of site-specific criteria. In recognition of natural variations in the concentration of soil (parent material) nutrients and precipitation regimes, and variations in water flow among distinct water body types (e.g., streams and rivers, lakes and reservoirs, estuaries, and coastal marine waters), the USEPA recommends the development of nutrient criteria for specific regions and water body types (USEPA 1998). USEPA relied on reference sites as the basis for establishing ecoregional nutrient criteria in an attempt to represent the physical, chemical, and biological characteristics of the region. Further information on the derivation of and basis for USEPA's recommended nutrient criteria can be found at USEPA's website, http://www.epa.gov/ost/standards/nutrient.html.

The development of more specific nutrient criteria is encouraged where possible, but USEPA acknowledges that this approach to developing nutrient criteria may not be practical in many cases given the costs and efforts required to develop nutrient criteria for individual water bodies within a region. Nutrient ecoregions delineate areas of broad scale similarities in geography, ecosystems, and related nutrient conditions which could be refined and further subdivided to develop site-specificity. The ecoregional reference conditions and criteria are believed to be a reasonable alternative approach to the single, nationwide criterion approach which fails to address regional variability and the too refined individual water body approach. California will rely on a different approach than those proposed by USEPA. A final workplan detailing nutrient criteria development procedures will be available by summer 2003.

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