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United States Environmental Protection Agency
Office of Water (4305)
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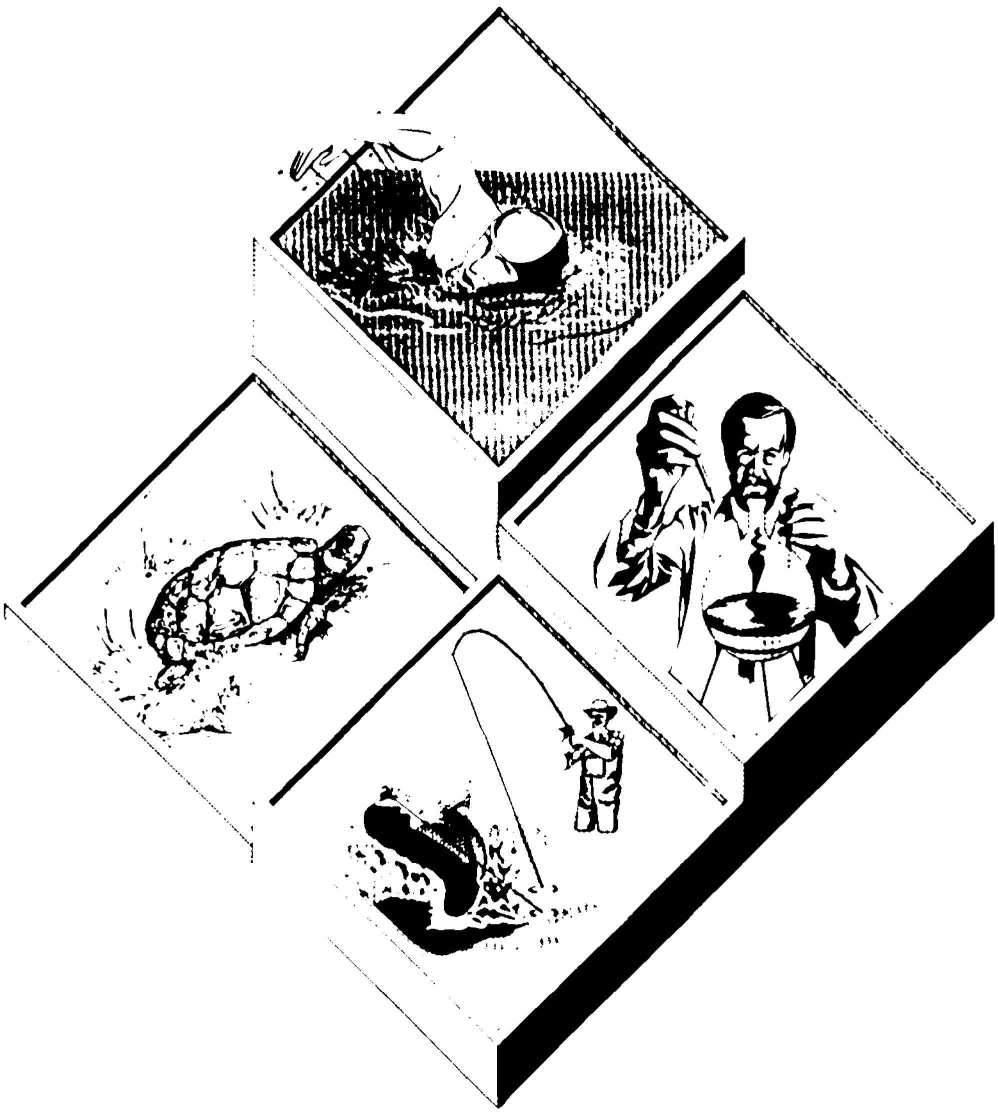
August 1994

EPA

Water Quality Standards Handbook: Second Edition

Appendixes



"... to restore and maintain the chemical, physical, and biological integrity of the Nation's waters."

Section 101(a) of the Clean Water Act

Contains update #1
August 1994

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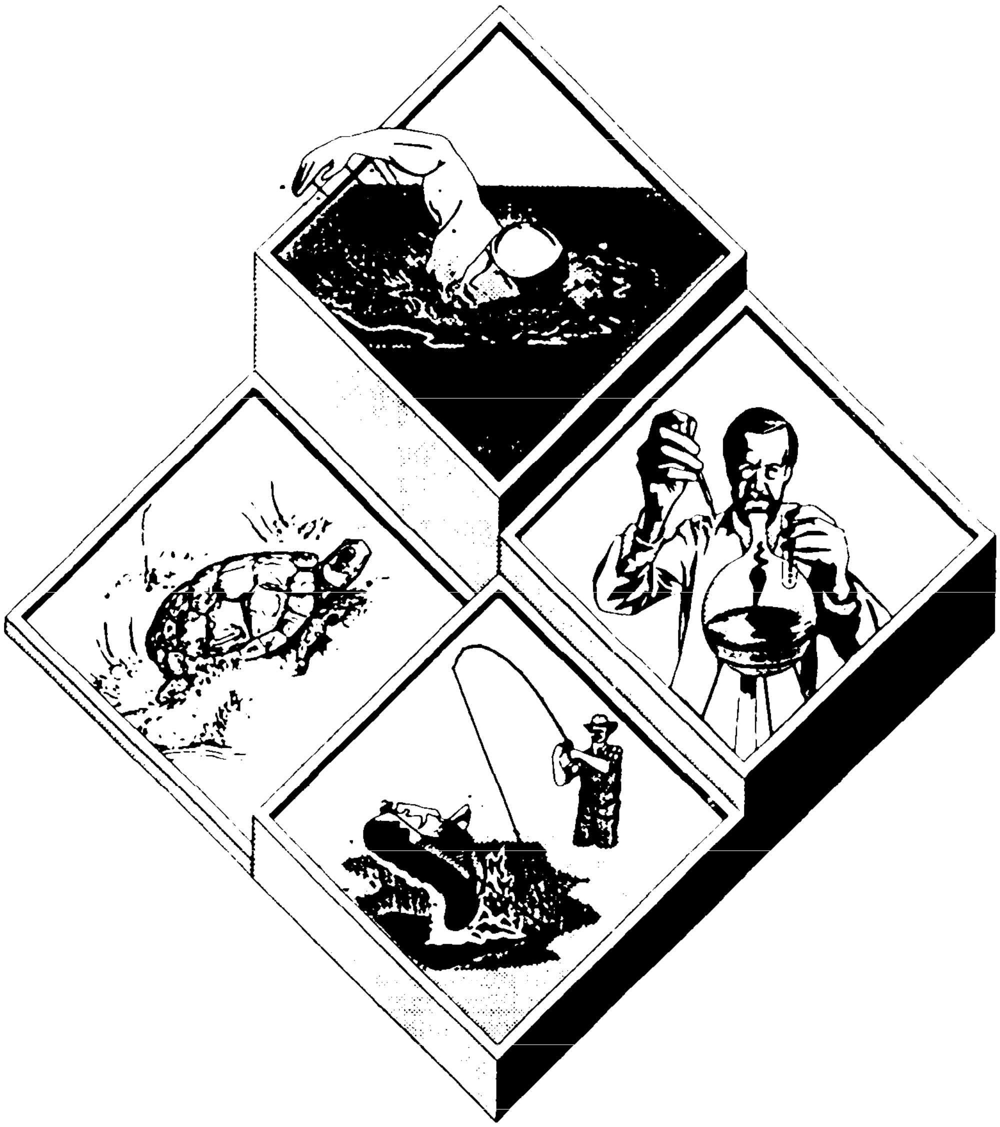

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APPENDIX A

Water Quality Standards Regulation

APPENDIX A

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

Water Quality Standards Regulation

(40 CFR 131; 48 FR 51405, Nov. 8, 1983; Revised through July 1, 1991; amended at 56 FR 64893, Dec. 12, 1991; 57 FR 60910, Dec. 22, 1992)

TITLE 40—PROTECTION OF ENVIRONMENT

CHAPTER I—ENVIRONMENTAL PROTECTION AGENCY

SUBCHAPTER D—WATER PROGRAMS

PART 131—WATER QUALITY STANDARDS

Authority: 33 U.S.C. 1251 *et seq.*

[Amended at 56 FR 64893, Dec. 12, 1991; 57 FR 60910, Dec. 22, 1992]

Subpart A—General Provisions

§131.1 Scope.

This part describes the requirements and procedures for developing, reviewing, revising and approving water quality standards by the States as authorized by section 303(c) of the Clean Water Act. The reporting or recordkeeping (information) provisions in this rule were approved by the Office of Management and Budget under 3504(b) of the Paperwork Reduction Act of 1980, U.S.C. 3501 *et seq.* (Approval number 2040-0049).

§131.2 Purpose.

A water quality standard defines the water quality goals of a water body, or portion thereof, by designating the use or uses to be made of the water and by setting criteria necessary to protect the uses. States adopt water quality standards to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act (the Act). "Serve the purposes of the Act" (as defined in sections 101(a)(2) and 303(c) of the Act) means that water quality standards should, wherever attainable, provide water quality for the protection and propagation of fish, shellfish and wildlife and for recreation in and on the water and take into consideration their use and value of public water supplies, propagation of fish, shellfish, and wildlife, recreation in and on the water, and agricultural, industrial, and other purposes including navigation.

Such standards serve the dual purposes of establishing the water quality goals for a specific water body and serve as the regulatory basis for the establishment of water-quality-based treatment controls and strategies beyond the technology-based levels of treatment required by sections 301(b) and 306 of the Act.

§131.3 Definitions.

(a) *The Act* means the Clean Water Act (Pub. L. 92-500, as amended, (33 U.S.C. 1251 *et seq.*))

(b) *Criteria* are elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use.

(c) *Section 304(a) criteria* are developed by EPA under authority of section 304(a) of the Act based on the latest scientific information on the relationship that the effect of a constituent concentration has on particular aquatic species and/or human health. This information is issued periodically to the States as guidance for use in developing criteria.

(d) *Toxic pollutants* are those pollutants listed by the Administrator under section 307(a) of the Act.

(e) *Existing uses* are those uses actually attained in the water body on or after November 28, 1975, whether or not they are included in the water quality standards.

(f) *Designated uses* are those uses specified in water quality standards for each water body or segment whether or not they are being attained

(g) *Use attainability analysis* is a structured scientific assessment of the factors affecting the attainment of the use which may include physical, chemical, biological, and economic factors as described in §131.10(g)

(h) *Water quality limited segment* means any segment where it is known that water quality does not meet applicable water quality standards, and/or is not expected to meet applicable water quality standards, even after the application of the technology bases effluent limitations required by sections 301(b) and 306 of the Act

(i) *Water quality standards* are provisions of State or Federal law which consist of a designated use or uses for the waters of the United States and water quality criteria for such waters based upon such uses. Water quality standards are to protect the public health or welfare, enhance the quality of water and serve the purposes of the Act

[§131.3(j)-(l) added at 56 FR 64893, Dec 12, 1991]

(j) *States* include: The 50 States, the District of Columbia, Guam, the Commonwealth of Puerto Rico, Virgin Islands, American Samoa, the Trust Territory of the Pacific Islands, the Commonwealth of the Northern Mariana Islands, and Indian Tribes that EPA determines qualify for treatment as States for purposes of water quality standards

(k) *Federal Indian Reservation, Indian Reservation*, or *Reservation* means all land within the limits of any Indian reservation under the jurisdiction of the United States Government, notwithstanding the issuance of any patent, and including rights-of-way running through the reservation."

(l) *Indian Tribe* or *Tribe* means any Indian Tribe, band, group, or community recognized by the Secretary of the Interior and exercising governmental authority over a Federal Indian reservation

§131.4 State authority.

(a) States (as defined in §131.3) are responsible for reviewing, establishing, and revising water quality standards. As recognized by section 510 of the Clean Water Act, States may develop water quality standards more stringent than required by this regulation. Consistent with section 101(g) and 518(a) of the Clean Water Act, water quality standards shall not be construed to superseder or abrogate rights to quantities of water

(b) States (as defined in §131.3) may issue certifications pursuant to the requirements of Clean Water Act section 401. Revisions adopted by States shall be applicable for use in issuing State certifications consistent with the provisions of §131.21(c).

(c) Where EPA determines that a Tribe qualifies for treatment as a State for purposes of water quality standards, the Tribe likewise qualifies for treatment as a State for purposes of certifications conducted under Clean Water Act section 401

[§131.4 revised at 56 FR 64893, Dec. 12, 1991]

§131.5 EPA authority.

[§131.5 former paragraphs (a)—(e) redesignated as new (a) and (a)(1)—(a)(5) at 56 FR 64893, Dec 12, 1991]

(a) Under section 303(c) of the Act, EPA is to review and to approve or disapprove State-adopted water quality standards. The review involves a determination of:

(1) Whether the State has adopted water uses which are consistent with the requirements of the Clean Water Act;

(2) Whether the state has adopted criteria that protect the designated water uses;

(3) Whether the State has followed its legal procedures for revising or adopting standards;

(4) Whether the State standards which do not include the uses specified in section 101(a)(2) of the Act are based upon appropriate technical and scientific data and analyses, and

(5) Whether the State submission meets the requirements included in §131.6 of this part. If EPA determines that State water quality standards are consistent with the factors listed in paragraphs (a) through (e) of this section, EPA approves the standards. EPA must disapprove the State water quality standards under section 303(c)(4) of the Act, if State adopted standards are not consistent with the factors listed in paragraphs (a) through (e) of this section. EPA may also promulgate a new or revised standard where necessary to meet the requirements of the Act

(b) Section 401 of the Clean Water Act authorizes EPA to issue certifications pursuant to the requirements of section 401 in any case where a State or interstate agency has no authority for issuing such certifications.

[§131.5(b) added at 56 FR 64893, Dec 12, 1991]

§131.6 Minimum requirements for water quality standards submission.

The following elements must be included in each State's water quality standards submitted to EPA for review

(a) Use designations consistent with the provisions of sections 101(a)(2) and 303(c)(2) of the Act

(b) Methods used and analyses conducted to support water quality standards revisions.

(c) Water quality criteria sufficient to protect the designated uses.

(d) An antidegradation policy consistent with §131.12.

(e) Certification by the State Attorney General or other appropriate legal authority within the State that the water quality standards were duly adopted pursuant to State law.

(f) General information which will aid the Agency in determining the adequacy of the scientific basis of the standards which do not include the uses specified in section 101(a)(2) of the Act as well as information on general policies applicable to State standards which may affect their application and implementation.

§131.7 Dispute resolution mechanism.

(a) Where disputes between States and Indian Tribes arise as a result of differing water quality standards on common bodies of water, the lead EPA Regional Administrator, as determined based upon OMB circular A-105, shall be responsible for acting in accordance with the provisions of this section

(b) The Regional Administrator shall attempt to resolve such disputes where

(1) The difference in water quality standards results in unreasonable consequences;

(2) The dispute is between a State (as defined in §131.3(j) but exclusive of all Indian Tribes) and a Tribe which EPA has determined qualifies to be treated as a State for purposes of water quality standards;

(3) A reasonable effort to resolve the dispute without EPA involvement has been made;

(4) The requested relief is consistent with the provisions of the Clean Water Act and other relevant law;

(5) The differing State and Tribal water quality standards have been adopted pursuant to State and Tribal law and approved by EPA; and

(6) A valid written request has been submitted by either the Tribe or the State.

(c) Either a State or a Tribe may request EPA to resolve any dispute which satisfies the criteria of paragraph (b) of this section. Written requests for EPA involvement should be submitted to the lead Regional Administrator and must include:

(1) A concise statement of the unreasonable consequences that are alleged to have arisen because of differing water quality standards;

(2) A concise description of the actions which have been taken to resolve the dispute without EPA involvement;

(3) A concise indication of the water quality standards provision which has resulted in the alleged unreasonable consequences;

(4) Factual data to support the alleged unreasonable consequences; and

(5) A statement of the relief sought from the alleged unreasonable consequences.

(d) Where, in the Regional Administrator's judgment, EPA involvement is appropriate based on the factors of paragraph (b) of this section, the Regional Administrator shall, within 30 days, notify the parties in writing that he/she is initiating an EPA dispute resolution action and solicit their written response. The Regional Administrator shall also make reasonable efforts to ensure that other interested individuals or groups have notice of this action. Such efforts shall include but not be limited to the following:

(1) Written notice to responsible Tribal and State Agencies, and other affected Federal Agencies,

(2) Notice to the specific individual or entity that is alleging that an unreasonable consequence is resulting from differing standards having been adopted on a common body of water,

(3) Public notice in local newspapers, radio, and television, as appropriate,

(4) Publication in trade journal newsletters, and

(5) Other means as appropriate.

(e) If in accordance with applicable State and Tribal law an Indian Tribe and State have entered into an agreement that resolves the dispute or establishes a mechanism for resolving a dispute, EPA shall defer to this agreement where it is consistent with the Clean Water Act and where it has been approved by EPA.

(f) EPA dispute resolution actions shall be consistent with one or a combination of the following options:

(1) *Mediation*. The Regional Administrator may appoint a mediator to mediate the dispute. Mediators shall be EPA employees, employees from other Federal agencies, or other individuals with appropriate qualifications.

(i) Where the State and Tribe agree to participate in the dispute resolution process, mediation with the intent to establish Tribal-State agreements, consistent with Clean Water Act section 518(d) shall normally be pursued as a first effort.

(ii) Mediators shall act as neutral facilitators whose function is to encourage communication and negotiation between all parties to the dispute.

(iii) Mediators may establish advisory panels, to consist in part of representatives from the affected parties, to study the problem and recommend an appropriate solution.

(iv) The procedure and schedule for mediation of individual disputes shall be determined by the mediator in consultation with the parties.

(v) If formal public hearings are held in connection with the actions taken under this paragraph, Agency requirements at 40 CFR 25.5 shall be followed.

(2) *Arbitration*. Where the parties to the dispute agree to participate in the dispute resolution process, the Regional Administrator may appoint an arbitrator or arbitration panel to arbitrate the dispute. Arbitrators and panel members shall be EPA employees, employees from other Federal agencies, or other individuals with appropriate qualifications. The Regional administrator shall select as arbitrators and arbitration panel members individuals who are agreeable to all parties, are knowledgeable concerning the requirements of the water quality standards program, have a basic understanding of the political and economic interests of Tribes and States involved, and are expected to fulfill the duties fairly and impartially.

(i) The arbitrator or arbitration panel shall conduct one or more private or public meetings with the parties and actively solicit information pertaining to the effects of differing water quality permit requirements on upstream and downstream dischargers, comparative risks to public health and the environment, economic impacts, present and historical water uses, the quality of the waters subject to such standards, and other factors relevant to the dispute such as whether proposed water quality criteria are more stringent than necessary to support designated uses, more stringent than natural background water quality or whether designated uses are reasonable given natural background water quality.

(ii) Following consideration of relevant factors as defined in paragraph (f)(2)(i) of this section, the arbitrator or arbitration panel shall have the authority and responsibility to provide all parties and the Regional Administrator with a written recommendation for resolution of the dispute. Arbitration panel recommendations shall, in general, be reached by majority vote. However, where the parties agree to binding arbitration, or where required by the Regional Administrator, recommendations of such arbitration panels may be unanimous decisions. Where binding or non-binding arbitration panels cannot reach a unanimous recommendation after a reasonable period of time, the Regional Administrator may direct the panel to issue a non-binding decision by majority vote.

(iii) The arbitrator or arbitration panel members may consult with EPA's Office of General Counsel on legal issues, but otherwise shall have no *ex parte* communications pertaining to the dispute. Federal employees who are arbitrators or arbitration panel members shall be neutral and shall not be predisposed for or against the position of any disputing party based on any Federal Trust responsibilities which their employers may have with respect to the Tribe. In addition, arbitrators or arbitration panel members who are Federal employees shall act independently from the normal hierarchy within their agency.

(iv) The parties are not obligated to abide by the arbitrator's or arbitration panel's recommendation unless they voluntarily entered into a binding agreement to do so.

(v) If a party to the dispute believes that the arbitrator or arbitration panel has recommended an action contrary to or inconsistent with the Clean Water Act, the party may appeal the arbitrator's recommendation to the Regional Administrator. The request for appeal must be in writing and must include a description of the statutory basis for altering the arbitrator's recommendation.

(vi) The procedure and schedule for arbitration of individual disputes shall be determined by the arbitrator or arbitration panel in consultation with parties.

(vii) If formal public hearings are held in connection with the actions taken under this paragraph. Agency requirements at 40 CFR 25.5 shall be followed.

(3) *Dispute Resolution Default Procedure.* Where one or more parties (as defined in paragraph (g) of this section) refuse to participate in either the mediation or arbitration dispute resolution processes, the Regional Administrator may appoint a single official or panel to review available information pertaining to the dispute and to issue a written recommendation for resolving the dispute. Review officials shall be EPA employees, employees from other Federal agencies, or other individuals with appropriate qualifications. Review panels shall include appropriate members to be selected by the Regional Administrator in consultation with the participating parties. Recommendations of such review officials or panels shall, to the extent possible given the lack of participation by one or more parties, be reached in a manner identical to that for arbitration of disputes specified in paragraphs (f)(2)(i) through (f)(2)(vii) of this section.

(g) *Definitions.* For the purposes of this section:

(1) *Dispute Resolution Mechanism* means the EPA mechanism established pursuant to the requirements of Clean Water Act section 518(e) for resolving unreasonable consequences that arise as a result of differing water quality standards that may be set by States and Indian Tribes located on common bodies of water.

(2) *Parties* to a State-Tribal dispute include the State and the Tribe and may, at the discretion of the Regional Administrator, include an NPDES permittee, citizen, citizen group, or other affected entity.

[§131.7 added at 56 FR 64893, Dec. 12, 1991]

§131.8 Requirements for Indian Tribes to be treated as States for purposes of water quality standards.

(a) The Regional Administrator, as determined based on OMB Circular A105, may treat an Indian Tribe as a State for purposes of the water quality standards program if the Tribe meets the following criteria:

(1) The Indian Tribe is recognized by the Secretary of the Interior and meets the definitions in §131.3(k) and (l),

(2) The Indian Tribe has a governing body carrying out substantial governmental duties and powers,

(3) The water quality standards program to be administered by the Indian Tribe pertains to the management and protection of water resources which are within the borders of the Indian reservation and held by the Indian Tribe, within the borders of the Indian reservation and held by the United States in trust for Indians, within the borders of the Indian reservation and held by a member of the Indian Tribe if such property interest is subject to a trust restriction on alienation, or otherwise within the borders of the Indian reservation, and

(4) The Indian Tribe is reasonably expected to be capable, in the Regional Administrator's judgment, of carrying out the functions of an effective water quality standards program in a manner consistent with the terms and purposes of the Act and applicable regulations.

(b) Requests by Indian Tribes for treatment as States for purposes of water quality standards should be submitted to the lead EPA Regional Administrator. The application shall include the following information:

(1) A statement that the Tribe is recognized by the Secretary of the Interior.

(2) A descriptive statement demonstrating that the Tribal governing body is currently carrying out substantial governmental duties and powers over a defined area. The statement shall:

(i) Describe the form of the Tribal government;

(ii) Describe the types of governmental functions currently performed by the Tribal governing body such as, but not limited to, the exercise of police powers affecting (or relating to) the health, safety, and welfare of the affected population, taxation, and the exercise of the power of eminent domain; and

(iii) Identify the source of the Tribal government's authority to carry out the governmental functions currently being performed.

(3) A descriptive statement of the Indian Tribe's authority to regulate water quality. The statement shall include:

(i) A map or legal description of the area over which the Indian Tribe asserts authority to regulate surface water quality;

(ii) A statement by the Tribe's legal counsel (or equivalent official) which describes the basis for the Tribes assertion of authority,

(iii) A copy of all documents such as Tribal constitutions, by-laws, charters, executive orders, codes, ordinances, and/or resolutions which support the Tribe's assertion of authority; and

(iv) an identification of the surface water for which the Tribe proposes to establish water quality standards.

(4) A narrative statement describing the capability of the Indian Tribe to administer an effective water quality standards program. The narrative statement shall include:

(i) A description of the Indian Tribe's previous management experience including, but not limited to, the administration of programs and services authorized by the Indian Self-Determination and Education Assistance Act (25 U.S.C. 450 *et seq.*), the Indian Mineral Development Act (25 U.S.C. 2101 *et seq.*), or the Indian Sanitation Facility Construction Activity Act (42 U.S.C. 2004a);

(ii) A list of existing environmental or public health programs administered by the Tribal governing body and copies of related Tribal laws, policies, and regulations;

(iii) A description of the entity (or entities) which exercise the executive, legislative, and judicial functions of the Tribal government;

(iv) A description of the existing or proposed, agency of the Indian Tribe which will assume primary responsibility for establishing, reviewing, implementing and revising water quality standards;

(v) A description of the technical and administrative capabilities of the staff to administer and manage an effective water quality standards program or a plan which proposes how the Tribe will acquire additional administrative and technical expertise. The plan must address how the Tribe will obtain the funds to acquire the administrative and technical expertise.

(5) Additional documentation required by the Regional Administrator which, in the judgment of the Regional Administrator, is necessary to support a Tribal request for treatment as a State.

(6) Where the Tribe has previously qualified for treatment as a State under a Clean Water Act or Safe Drinking Water Act program, the Tribe need only provide the required information which has not been submitted in a previous treatment as a State application.

(c) Procedure for processing an Indian Tribe's application for treatment as a State.

(1) The Regional Administrator shall process an application of an Indian Tribe for treatment as a State submitted pursuant to 131.8(b) in a timely manner. He shall promptly notify the Indian Tribe of receipt of the application.

(2) Within 30 days after receipt of the Indian Tribe's application for treatment as a State, the Regional Administrator shall provide appropriate notice. Notice shall:

(i) Include information on the substance and basis of the Tribe's assertion of authority to regulate the quality of reservation waters; and

(ii) Be provided to all appropriate governmental entities.

(3) The Regional Administrator shall provide 30 days for comments to be submitted on the Tribal application. Comments shall be limited to the Tribe's assertion of authority.

(4) If a Tribe's asserted authority is subject to a competing or conflicting claim, the Regional Administrator, after consultation with the Secretary of the Interior, or his designee, and in consideration of other comments received, shall determine whether the Tribe has adequately demonstrated that it meets the requirements of 131.8(a)(3).

(5) Where the Regional Administrator determines that a Tribe meets the requirements of this section, he shall promptly provide written notification to the Indian Tribe that the Tribe has qualified to be treated as a State for purposes of water quality standards and that the Tribe may initiate the formulation and adoption of water quality standards approvable under this part.

[§131.8 added at 56 FR 64893, Dec. 12, 1991]

Subpart B—Establishment of Water Quality Standards

§131.10 Designation of uses.

(a) Each State must specify appropriate water uses to be achieved and protected. The classification of the waters of the State must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish and wildlife, recreation in and on the water, agricultural, industrial, and other purposes including navigation. In no case shall a State adopt waste transport or waste assimilation as a designated use for any waters of the United States.

(b) In designating uses of a water body and the appropriate criteria for those uses, the State shall take into consideration the water quality standards of downstream waters and shall ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters.

(c) States may adopt sub-categories of a use and set the appropriate criteria to reflect varying needs of such sub-categories of uses, for instance, to differentiate between cold water and warm water fisheries.

(d) At a minimum, uses are deemed attainable if they can be achieved by the imposition of effluent limits required under sections 301(b) and 306 of the Act and cost-effective and reasonable best management practices for nonpoint source control.

(e) Prior to adding or removing any use, or establishing sub-categories of a use, the State shall provide notice and an opportunity for a public hearing under §131.20(b) of this regulation.

(f) States may adopt seasonal uses as an alternative to reclassifying a water body or segment thereof to uses requiring less stringent water quality criteria. If seasonal uses are adopted, water quality criteria should be adjusted to reflect the seasonal uses, however, such criteria shall not preclude the attainment and maintenance of a more protective use in another season.

(g) States may remove a designated use which is *not* an existing use, as defined in §131.3, or establish sub-categories of a use if the State can demonstrate that attaining the designated use is not feasible because:

(1) Naturally occurring pollutant concentrations prevent the attainment of the use; or

(2) Natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the use, unless these conditions may be compensated for by the discharge of sufficient volume of effluent discharges without violating State water conservation requirements to enable uses to be met; or

(3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place; or

(4) Dams, diversions or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use; or

(5) Physical conditions related to the natural features of the water body, such as the lack of a proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or

(6) Controls more stringent than those required by sections 301(b) and 306 of the Act would result in substantial and widespread economic and social impact.

(h) States may not remove designated uses if:

(1) They are existing uses, as defined in §131.3, unless a use requiring more stringent criteria is added; or

(2) Such uses will be attained by implementing effluent limits required under sections 301(b) and 306 of the Act and by implementing cost-effective and reasonable best management practices for nonpoint source control.

(i) Where existing water quality standards specify designated uses less than those which are presently being attained, the State shall revise its standards to reflect the uses actually being attained.

(j) A State must conduct a use attainability analysis as described in §131.3(g) whenever:

(i) The State designates or has designated uses that do not include the uses specified in section 101(a)(2) of the Act, or

(2) The State wishes to remove a designated use that is specified in section 101(a)(2) of the Act or to adopt subcategories of uses specified in section 101(a)(2) of the Act which require less *stringent criteria*.

(k) A State is not required to conduct a use attainability analysis under this regulation whenever designating uses which include those specified in section 101(a)(2) of the Act

§131.11 Criteria.

(a) *Inclusion of pollutants*

(1) States must adopt those water quality criteria that protect the designated use. Such criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect the designated use. For waters with multiple use designations, the criteria shall support the most sensitive use.

(2) *Toxic pollutants*. States must review water quality data and information on discharges to identify specific water bodies where toxic pollutants may be adversely affecting water quality or the attainment of the designated water use or where the levels of toxic pollutants are at a level to warrant concern and must adopt criteria for such toxic pollutants applicable to the water body sufficient to protect the designated use. Where a State adopts narrative criteria for toxic pollutants to protect designated uses, the State must provide information identifying the method by which the State intends to regulate point source discharges of toxic pollutants on water quality limited segments based on such narrative criteria. Such information may be included as part of the standards or may be included in documents generated by the State in response to the Water Quality Planning and Management Regulations (40 CFR part 35).

(b) Form of criteria. In establishing criteria, States should:

(1) Establish numerical values based on

(i) 304(a) Guidance; or

(ii) 304(a) Guidance modified to reflect site-specific conditions; or

(iii) Other scientifically defensible methods;

(2) Establish narrative criteria or criteria based upon biomonitoring methods where numerical criteria cannot be established or to supplement numerical criteria.

§131.12 Antidegradation policy.

(a) The State shall develop and adopt a statewide antidegradation policy and identify the methods for implementing such policy pursuant to this subpart. The antidegradation policy and implementation methods shall, at a minimum, be consistent with the following:

(1) Existing instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected.

(2) Where the quality of the waters exceed levels necessary to support propagation of fish, shellfish, and wildlife and recreation in and on the water, that quality shall be maintained and protected unless the State finds, after full satisfaction of the intergovernmental coordination and public participation provisions of the State's continuing planning process, that allowing lower water quality is necessary to accommodate important economic or social development in the area in which the waters are located. In allowing such degradation or lower water quality, the State shall assure water quality adequate to protect existing uses fully. Further, the State shall assure that there shall be achieved the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best management practices for nonpoint source control.

(3) Where high quality waters constitute an outstanding National resource, such as waters of National and State parks and wildlife refuges and waters of exceptional recreational or ecological significance, that water quality shall be maintained and protected.

(4) In those cases where potential water quality impairment associated with a thermal discharge is involved, the antidegradation policy and implementing method shall be consistent with section 316 of the Act.

§131.13 General policies.

States may, at their discretion, include in their State standards, policies generally affecting their application and implementation, such as mixing zones, low flows and variances. Such policies are subject to EPA review and approval.

Subpart C—Procedures for Review and Revision of Water Quality Standards

§131.20 State review and revision of water quality standards.

(a) *State review*. The State shall from time to time, but at least once every three years, hold public hearings for the purpose of reviewing applicable water quality standards and, as appropriate, modifying and adopting standards. Any water body segment with water quality standards that do not include the uses specified in section 101(a)(2) of the Act shall be re-examined every three years to determine if any new information has become available. If such new information indicates that the uses specified in section 101(a)(2) of the Act are attainable, the State shall revise its standards accordingly. Procedures States establish for identifying and reviewing water bodies for review should be incorporated into their Continuing Planning Process.

(b) *Public participation*. The State shall hold a public hearing for the purpose of reviewing water quality standards, in accordance with provisions of State law, EPA's water quality management regulation (40 CFR 130.3(b)(6)) and public participation regulation (40 CFR part 25). The proposed water quality standards revision and supporting analyses shall be made available to the public prior to the hearing.

(c) *Submittal to EPA*. The State shall submit the results of the review, any supporting analysis for the use attainability analysis, the methodologies used for site-specific criteria development, any general policies applicable to water quality standards and any revisions of the standards to the Regional Administrator for review and approval, within 30 days of the final State action to adopt and certify the revised standard, or if no revisions are made as a result of the review, within 30 days of the completion of the review.

§131.21 EPA review and approval of water quality standards.

(a) After the State submits its officially adopted revisions, the Regional Administrator shall either:

(1) Notify the State within 60 days that the revisions are approved, or

(2) Notify the State within 90 days that the revisions are disapproved. Such

notification of disapproval shall specify the changes needed to assure compliance with the requirements of the Act and this regulation, and shall explain why the State standard is not in compliance with such requirements. Any new or revised State standard must be accompanied by some type of supporting analysis.

(b) The Regional Administrator's approval or disapproval of a State water quality standard shall be based on the requirements of the Act as described in §§131.5, and 131.6.

(c) A State water quality standard remains in effect, even though disapproved by EPA, until the State revises it or EPA promulgates a rule that supersedes the State water quality standard.

(d) EPA shall, at least annually, publish in the FEDERAL REGISTER a notice of approvals under this section.

§131.22 EPA promulgation of water quality standards.

(a) If the State does not adopt the changes specified by the Regional Administrator within 90 days after notification of the Regional Administrator's disapproval, the Administrator shall promptly propose and promulgate such standard.

(b) The Administrator may also propose and promulgate a regulation, applicable to one or more States, setting forth a new or revised standard upon determining such a standard is necessary to meet the requirements of the Act.

(c) In promulgating water quality standards, the Administrator is subject to the same policies, procedures, analyses, and public participation requirements established for States in these regulations.

Subpart D—Federally Promulgated Water Quality Standards

§131.31 Arizona.

(a) Article 6, Part 2 is amended as follows:

(1) Reg. 6-2-6.11 shall read:

Reg. 6-2-6.11. Nutrient Standards. A. The mean annual total phosphate and mean annual total nitrate concentrations of the following waters shall not exceed the values given below nor shall the total phosphate or total nitrate concentrations of more than 10 percent of the samples in any year exceed the 90 percent values given below. Unless otherwise specified, indicated values also apply to tributaries to the named waters.

	Mean 90 pct annual value	
	Total phosphates as PO_4mg/l	Total nitrates as NO_3mg/l
1. Colorado River from Utah border to Willow Beach (main stem)	0.04-0.06	4-7
2. Colorado River from Willow Beach to Parker Dam (main stem)	0.06-0.10	5
3. Colorado River from Parker Dam to Imperial Dam (main stem)	0.08-0.12	5-7
4. Colorado River from Imperial Dam to Morelos Dam (main stem)	0.10-0.10	5-7
5. Gila River from New Mexico border to San Carlos Reservoir (excluding San Carlos Reservoir)	0.50-0.80	
6. Gila River from San Carlos Reservoir to Ashurst Hayden Dam (including San Carlos Reservoir)	0.30-0.50	
7. San Pedro River	0.30-0.50	
8. Verde River (except Granite Creek)	0.20-0.30	
9. Salt River above Roosevelt Lake	0.20-0.30	
10. Santa Cruz River from international boundary near Nogales to Sahuarita	0.50-0.80	
11. Little Colorado River above Lyman Reservoir	0.30-0.50	

B. The above standards are intended to protect the beneficial uses of the named waters. Because regulation of nitrates and phosphates alone may not be adequate to protect waters from eutrophication, no substance shall be added to any surface water which produces aquatic growth to the extent that such growths create a public nuisance or interference with beneficial uses of the water defined and designated in Reg. 6-2-6.5.

(2) Reg. 6-2-6.10 Subparts A and B are amended to include Reg. 6-2-6.11 in series with Regs. 6-2-6.6, 6-2-6.7 and 6-2-6.8.

§131.33 [Reserved]

§131.34 [Reserved]

§131.35 Colville Confederated Tribes Indian Reservation.

The water quality standards applicable to the waters within the Colville Indian Reservation, located in the State of Washington.

(a) *Background.*

(1) It is the purpose of these Federal water quality standards to prescribe minimum water quality requirements for the surface waters located within the exterior boundaries of the Colville Indian Reservation to ensure compliance with section 303(c) of the Clean Water Act.

(2) The Colville Confederated Tribes have a primary interest in the protection, control, conservation, and utilization of the water resources of the Colville Indian Reservation. Water quality standards have been enacted into tribal law by the Colville Business Council of the Confederated Tribes of the Colville Reservation, as the Colville Water Quality Standards Act, CTC Title 33 (Resolution No. 1984-526 (August 6, 1984) as amended by Resolution No. 1985-20 (January 18, 1985)).

(b) *Territory Covered.* The provisions of these water quality standards shall apply to all surface waters within the exterior boundaries of the Colville Indian Reservation.

(c) *Applicability, Administration and Amendment.*

(1) The water quality standards in this section shall be used by the Regional Administrator for establishing any water quality based National Pollutant Discharge Elimination System Permit (NPDES) for point sources on the Colville Confederated Tribes Reservation.

(2) In conjunction with the issuance of section 402 or section 404 permits, the Regional Administrator may designate mixing zones in the waters of the United States on the reservation on a case-by-case basis. The size of such mixing zones and the in-zone water quality in such mixing zones shall be consistent with the applicable procedures and guidelines in EPA's Water Quality Standards Handbook and the Technical Support Document for Water Quality Based Toxics Control.

(3) Amendments to the section at the request of the Tribe shall proceed in the following manner.

(i) The requested amendment shall first be duly approved by the Confederated Tribes of the Colville Reservation (and so certified by the Tribes Legal Counsel) and submitted to the Regional Administrator.

(ii) The requested amendment shall be reviewed by EPA (and by the State of Washington, if the action would affect a boundary water).

(iii) If deemed in compliance with the Clean Water Act, EPA will propose and promulgate an appropriate change to this section.

(4) Amendment of this section at EPA's initiative will follow consultation with the Tribe and other appropriate entities. Such amendments will then follow normal EPA rulemaking procedures.

(5) All other applicable provisions of this part 131 shall apply on the Colville Confederated Tribes Reservation. Special attention should be paid to §§131.6, 131.10, 131.11 and 131.20 for any amendment to these standards to be initiated by the Tribe.

(6) All numeric criteria contained in this section apply at all in-stream flow rates greater than or equal to the flow rate calculated as the minimum 7-consecutive day average flow with a recurrence frequency of once in ten years (7Q10); narrative criteria (§131.35(e)(3)) apply regardless of flow. The 7Q10 low flow shall be calculated using methods recommended by the U.S. Geological Survey.

(d) *Definitions*.

(1) "Acute toxicity" means a deleterious response (e.g., mortality, disorientation, immobilization) to a stimulus observed in 96 hours or less.

(2) "Background conditions" means the biological, chemical, and physical conditions of a water body, upstream from the point or non-point source discharge under consideration. Background sampling location in an enforcement action will be upstream from the point of discharge, but not upstream from other inflows. If several discharges to any water body exist, and an enforcement action is being taken for possible violations to the standards, background sampling will be undertaken immediately upstream from each discharge.

(3) "Ceremonial and Religious water use" means activities involving traditional Native American spiritual practices which involve, among other things, primary (direct) contact with water.

(4) "Chronic Toxicity" means the lowest concentration of a constituent causing observable effects (i.e., considering lethality, growth, reduced reproduction, etc.) over a relatively long period of time, usually a 28-day test period for small fish test species.

(5) "Council" or "Tribal Council" means the Colville Business Council of the Colville Confederated Tribes.

(6) "Geometric mean" means the "nth" root of a product of "n" factors.

(7) "Mean retention time" means the time obtained by dividing a reservoir's mean annual minimum total storage by the non-zero 30-day, ten-year low-flow from the reservoir.

(8) "Mixing Zone" or "dilution zone" means a limited area or volume of water where initial dilution of a discharge takes place; and where numeric water quality criteria can be exceeded but acutely toxic conditions are prevented from occurring.

(9) "pH" means the negative logarithm of the hydrogen ion concentration.

(10) "Primary contact recreation" means activities where a person would have direct contact with water to the point of complete submergence, including but not limited to skin diving, swimming, and water skiing.

(11) "Regional Administrator" means the Administrator of EPA's Region X.

(12) "Reservation" means all land within the limits of the Colville Indian Reservation, established on July 2, 1872 by Executive Order, presently containing 1,389,000 acres more or less, and under the jurisdiction of the United States government, notwithstanding the issuance of any patent, and including rights-of-way running through the reservation.

(13) "Secondary contact recreation" means activities where a person's water contact would be limited to the extent that bacterial infections of eyes, ears, respiratory, or digestive systems or urogenital areas would normally be avoided (such as wading or fishing).

(14) "Surface water" means all water above the surface of the ground within the exterior boundaries of the Colville Indian Reservation including but not limited to lakes, ponds, reservoirs, artificial impoundments, streams, rivers, springs, seeps and wetlands.

(15) "Temperature" means water temperature expressed in Centigrade degrees (C).

(16) "Total dissolved solids" (TDS) means the total filterable residue that passes through a standard glass fiber filter disk and remains after evaporation and drying to a constant weight at 180 degrees C. it is considered to be a measure of the dissolved salt content of the water.

(17) "Toxicity" means acute and/or chronic toxicity.

(18) "Tribe" or "Tribes" means the Colville Confederated Tribes.

(19) "Turbidity" means the clarity of water expressed as nephelometric turbidity units (NTU) and measured with a calibrated turbidimeter.

(20) "Wildlife habitat" means the waters and surrounding land areas of the Reservation used by fish, other aquatic life and wildlife at any stage of their life history or activity.

(e) *General considerations*. The following general guidelines shall apply to the water quality standards and classifications set forth in the use designation Sections.

(1) *Classification Boundaries*. At the boundary between waters of different classifications, the water quality standards for the higher classification shall prevail.

(2) *Antidegradation Policy*. This antidegradation policy shall be applicable to all surface waters of the Reservation.

(i) Existing in-stream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected.

(ii) Where the quality of the waters exceeds levels necessary to support propagation of fish, shellfish, and wildlife and recreation in and on the water, that quality shall be maintained and protected unless the Regional Administrator finds, after full satisfaction of the inter-governmental coordination and public participation provisions of the Tribes' continuing planning process, that allowing lower water quality is necessary to accommodate important economic or social development in the area in which the waters are located. In allowing such degradation or lower water quality, the Regional Administrator shall assure water quality adequate to protect existing uses fully. Further, the Regional Administrator shall assure that there shall be achieved the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best management practices for nonpoint source control.

(iii) Where high quality waters are identified as constituting an outstanding national or reservation resource, such as waters within areas designated as unique water quality management areas and waters otherwise of exceptional recreational or ecological significance, and are designated as special resource waters, that water quality shall be maintained and protected.

(iv) In those cases where potential water quality impairment associated with a thermal discharge is involved, this antidegradation policy's implementing method shall be consistent with section 316 of the Clean Water Act.

(3) *Aesthetic Qualities.* All waters within the Reservation, including those within mixing zones, shall be free from substances, attributable to wastewater discharges or other pollutant sources, that:

(i) Settle to form objectionable deposits;

(ii) Float as debris, scum, oil, or other matter forming nuisances;

(iii) Produce objectionable color, odor, taste, or turbidity;

(iv) Cause injury to, are toxic to, or produce adverse physiological responses in humans, animals, or plants; or

(v) Produce undesirable or nuisance aquatic life

(4) *Analytical Methods*

(i) The analytical testing methods used to measure or otherwise evaluate compliance with water quality standards shall to the extent practicable, be in accordance with the "Guidelines Establishing Test Procedures for the Analysis of Pollutants" (40 CFR part 136) When a testing method is not available for a particular substance, the most recent edition of "Standard Methods for the Examination of Water and Wastewater" (published by the American Public Health Association, American Water Works Association, and the Water Pollution Control Federation) and other or superseding methods published and/or approved by EPA shall be used.

(f) *General Water Use and Criteria Classes* The following criteria shall apply to the various classes of surface waters on the Colville Indian Reservation:

(1) *Class I (Extraordinary)*—

(i) *Designated uses.* The designated uses include, but are not limited to, the following:

(A) Water supply (domestic, industrial, agricultural).

(B) Stock watering.

(C) Fish and shellfish. Salmonid migration, rearing, spawning, and harvesting, other fish migration, rearing, spawning, and harvesting.

(D) Wildlife habitat.

(E) Ceremonial and religious water use.

(F) Recreation (primary contact recreation, sport fishing, boating and aesthetic enjoyment).

(G) Commerce and navigation.

(ii) *Water quality criteria.*

(A) Bacteriological Criteria—The geometric mean of the enterococci bacteria densities in samples taken over a 30 day period shall not exceed 8 per 100 milliliters, nor shall any single sample exceed an enterococci density of 35 per 100 milliliters. These limits are calculated as the geometric mean of the collected samples approximately equally spaced over a thirty day period.

(B) Dissolved oxygen—The dissolved oxygen shall exceed 9.5 mg/l.

(C) Total dissolved gas—concentrations shall not exceed 110 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures at any point of sample collection.

(D) Temperature—shall not exceed 16.0 degrees C due to human activities. Temperature increases shall not, at any time, exceed $t=23/(T+5)$.

(*1*) When natural conditions exceed 16.0 degrees C, no temperature increase will be allowed which will raise the receiving water by greater than 0.3 degrees C

(*2*) For purposes hereof, "t" represents the permissive temperature change across the dilution zone; and "T" represents the highest existing temperature in this water classification outside of any dilution zone.

(*3*) Provided that temperature increase resulting from nonpoint source activities shall not exceed 2.8 degrees C, and the maximum water temperature shall not exceed 10.3 degrees C.

(E) pH shall be within the range of 6.5 to 8.5 with a human-caused variation of less than 0.2 units.

(F) Turbidity shall not exceed 5 NTU over background turbidity when the background turbidity is 50 NTU or less, or have more than a 10 percent increase in turbidity when the background turbidity is more than 50 NTU.

(G) Toxic, radioactive, nonconventional, or deleterious material concentrations shall be less than those of public health significance, or which may cause acute or chronic toxic conditions to the aquatic biota, or which may adversely affect designated water uses.

(2) *Class II (Excellent).*—

(i) *Designated uses.* The designated uses include but are not limited to, the following.

(A) Water supply (domestic, industrial, agricultural).

(B) Stock watering

(C) Fish and shellfish Salmonid migration, rearing, spawning, and harvesting, other fish migration, rearing, spawning, and harvesting; crayfish rearing, spawning, and harvesting

(D) Wildlife habitat

(E) Ceremonial and religious water use.

(F) Recreation (primary contact recreation, sport fishing, boating and aesthetic enjoyment).

(G) Commerce and navigation

(ii) *Water quality criteria*

(A) Bacteriological Criteria—The geometric mean of the enterococci bacteria densities in samples taken over a 30 day period shall not exceed 16/100 ml, nor shall any single sample exceed an enterococci density of 75 per 100 milliliters These limits are calculated as the geometric mean of the collected samples approximately equally spaced over a thirty day period

(B) Dissolved oxygen—The dissolved oxygen shall exceed 8.0 mg/l.

(C) Total dissolved gas—concentrations shall not exceed 110 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures at any point of sample collection

(D) Temperature—shall not exceed 18.0 degrees C due to human activities. Temperature increases shall not, at any time, exceed $t=28/(T+7)$.

(*1*) When natural conditions exceed 18 degrees C no temperature increase will be allowed which will raise the receiving water temperature by greater than 0.3 degrees C.

(*2*) For purposes hereof, "t" represents the permissive temperature change across the dilution zone; and "T" represents the highest existing temperature in this water classification outside of any dilution zone.

(*3*) Provided that temperature increase resulting from non-point source activities shall not exceed 2.8 degrees C, and the maximum water temperature shall not exceed 18.3 degrees C

(E) pH shall be within the range of 6.5 to 8.5 with a human-caused variation of less than 0.5 units

(F) Turbidity shall not exceed 5 NTU over background turbidity when the background turbidity is 50 NTU or less, or have more than a 10 percent increase in turbidity when the background turbidity is more than 50 NTU.

(G) Toxic, radioactive, nonconventional, or deleterious material concentrations shall be less than those of public health significance, or which may cause acute or chronic toxic conditions to the aquatic biota, or which may adversely affect designated water uses.

(3) *Class III (Good)*

(i) *Designated uses.* The designated uses include but are not limited to, the following:

(A) Water supply (industrial, agricultural).

(B) Stock watering.

(C) Fish and shellfish: Salmonid migration, rearing, spawning, and harvesting; other fish migration, rearing, spawning, and harvesting; crayfish rearing, spawning, and harvesting.

(D) Wildlife habitat.

(E) Recreation (secondary contact recreation, sport fishing, boating and aesthetic enjoyment).

(F) Commerce and navigation.

(ii) *Water quality criteria.*

(A) Bacteriological Criteria—The geometric mean of the enterococci bacteria densities in samples taken over a 30 day period shall not exceed 33/100 ml, nor shall any single sample exceed an enterococci density of 150 per 100 milliliters. These limits are calculated as the geometric mean of the collected samples approximately equally spaced over a thirty day period.

(B) Dissolved oxygen.

	Early life stages[1,2]	Other life stages
7 day mean	9.5 (6.5)	[3]NA
1 day minimum[4]	8.0 (5.0)	6.5

[1] These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. The 3 mg/L differential is discussed in the dissolved oxygen criteria document (EPA 440/5-86-003, April 1986). For species that have early life stages exposed directly to the water column, the figures in parentheses apply.

[2] Includes all embryonic and larval stages and all juvenile forms to 30-days following hatching.

[3] NA (not applicable).

[4] All minima should be considered as instantaneous concentrations to be achieved at all times.

(C) Total dissolved gas concentrations shall not exceed 110 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures at any point of sample collection.

(D) Temperature shall not exceed 21.0 degrees C due to human activities. Temperature increases shall not, at any time, exceed t=34/(T+9).

(*1*) When natural conditions exceed 21.0 degrees C no temperature increase will be allowed which will raise the receiving water temperature by greater than 0.3 degrees C.

(*2*) For purposes hereof, "t" represents the permissive temperature change across the dilution zone; and "T" represents the highest existing temperature in this water classification outside of any dilution zone.

(*3*) Provided that temperature increase resulting from nonpoint source activities shall not exceed 2.8 degrees C, and the maximum water temperature shall not exceed 21.3 degrees C.

(E) pH shall be within the range of 6.5 to 8.5 with a human-caused variation of less than 0.5 units.

(F) Turbidity shall not exceed 10 NTU over background turbidity when the background turbidity is 50 NTU or less, or have more than a 20 percent increase in turbidity when the background turbidity is more than 50 NTU.

(G) Toxic, radioactive, nonconventional, or deleterious material concentrations shall be less than those of public health significance, or which may cause acute or chronic toxic conditions to the aquatic biota, or which may adversely affect designated water uses.

(4) *Class IV (Fair)*—

(i) *Designated uses.* The designated uses include but are not limited to, the following:

(A) Water supply (industrial).

(B) Stock watering.

(C) Fish (salmonid and other fish migration).

(D) Recreation (secondary contact recreation, sport fishing, boating and aesthetic enjoyment).

(E) Commerce and navigation.

(ii) *Water quality criteria.*

(A) Dissolved oxygen.

	During periods of salmonid and other fish migration	During all other time periods
30 day mean	6.5	5.5
7 day mean	[1]NA	[1]NA
7 day mean minimum	5.0	4.0
1 day minimum[2]	4.0	3.0

[1] NA (not applicable).

[2] All minima should be considered as instantaneous concentrations to be achieved at all times.

(B) Total dissolved gas—concentrations shall not exceed 110 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures at any point of sample collection.

(C) Temperature shall not exceed 22.0 degrees C due to human activities. Temperature increases shall not, at any time, exceed t=20/(T+2).

(*1*) When natural conditions exceed 22.0 degrees C, no temperature increase will be allowed which will raise the receiving water temperature by greater than 0.3 degrees C.

(*2*) For purposes hereof, "t" represents the permissive temperature change across the dilution zone; and "T" represents the highest existing temperature in this water classification outside of any dilution zone.

(D) pH shall be within the range of 6.5 to 9.0 with a human-caused variation of less than 0.5 units.

(E) Turbidity shall not exceed 10 NTU over background turbidity when the background turbidity is 50 NTU or less, or have more than a 20 percent increase in turbidity when the background turbidity is more than 50 NTU.

(F) Toxic, radioactive, nonconventional, or deleterious material concentrations shall be less than those of public health significance, or which may cause acute or chronic toxic conditions to the aquatic biota, or which may adversely affect designated water uses.

(5) *Lake Class*

(i) *Designated uses.* The designated uses include but are not limited to, the following:

(A) Water supply (domestic, industrial, agricultural).

(B) Stock watering.

(C) Fish and shellfish: Salmonid migration, rearing, spawning, and harvesting; other fish migration, rearing, spawning, and harvesting; crayfish rearing, spawning, and harvesting.

(D) Wildlife habitat.

(E) Ceremonial and religious water use.

(F) Recreation (primary contact recreation, sport fishing, boating and aesthetic enjoyment).

(G) Commerce and navigation.

(ii) *Water quality criteria*.

(A) Bacteriological Criteria. The geometric mean of the enterococci bacteria densities in samples taken over a 30 day period shall not exceed 33/100 ml, nor shall any single sample exceed an enterococci density of 150 per 100 milliliters. These limits are calculated as the geometric mean of the collected samples approximately equally spaced over a thirty day period.

(B) Dissolved oxygen—no measurable decrease from natural conditions.

(C) Total dissolved gas concentrations shall not exceed 110 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures at any point of sample collection.

(D) Temperature—no measurable change from natural conditions.

(E) pH—no measurable change from natural conditions.

(F) Turbidity shall not exceed 5 NTU over natural conditions.

(G) Toxic, radioactive, nonconventional, or deleterious material concentrations shall be less than those which may affect public health, the natural aquatic environment, or the desirability of the water for any use.

(6) *Special Resource Water Class (SRW)*—

(i) *General characteristics*. These are fresh or saline waters which comprise a special and unique resource to the Reservation. Water quality of this class will be varied and unique as determined by the Regional Administrator in cooperation with the Tribes.

(ii) *Designated uses*. The designated uses include, but are not limited to, the following:

(A) Wildlife habitat.

(B) Natural foodchain maintenance.

(iii) Water quality criteria.

(A) Enterococci bacteria densities shall not exceed natural conditions.

(B) Dissolved oxygen—shall not show any measurable decrease from natural conditions.

(C) Total dissolved gas shall not vary from natural conditions.

(D) Temperature—shall not show any measurable change from natural conditions.

(E) pH shall not show any measurable change from natural conditions.

(F) Settleable solids shall not show any change from natural conditions.

(G) Turbidity shall not exceed 5 NTU over natural conditions.

(H) Toxic, radioactive, or deleterious material concentrations shall not exceed those found under natural conditions.

(g) *General Classifications*. General classifications applying to various surface waterbodies not specifically classified under §131.35(h) are as follows:

(1) All surface waters that are tributaries to Class I waters are classified Class I, unless otherwise classified.

(2) Except for those specifically classified otherwise, all lakes with existing average concentrations less than 2000 mg/L TDS and their feeder streams on the Colville Indian Reservation are classified as Lake Class and Class I, respectively.

(3) All lakes on the Colville Indian Reservation with existing average concentrations of TDS equal to or exceeding 2000 mg/L and their feeder streams are classified as Lake Class and Class I respectively unless specifically classified otherwise.

(4) All reservoirs with a mean detention time of greater than 15 days are classified Lake Class.

(5) All reservoirs with a mean detention time of 15 days or less are classified the same as the river section in which they are located.

(6) All reservoirs established on pre-existing lakes are classified as Lake Class.

(7) All wetlands are assigned to the Special Resource Water Class.

(8) All other waters not specifically assigned to a classification of the reservation are classified as Class II.

(h) *Specific Classifications*. Specific classifications for surface waters of the Colville Indian Reservation are as follows:

(1) Streams

Alice Creek	Class III
Anderson Creek	Class III
Armstrong Creek	Class III
Barnaby Creek	Class II
Bear Creek	Class III
Beaver Dam Creek	Class II
Bridge Creek	Class II
Brush Creek	Class III
Buckhorn Creek	Class III
Cache Creek	Class III
Canteen Creek	Class I
Capoose Creek	Class III
Cobbs Creek	Class III
Columbia River from Chief Joseph Dam to Wells Dam	
Columbia River from northern Reservation boundary to Grand Coulee Dam (Roosevelt Lake)	
Columbia River from Grand Coulee Dam to Chief Joseph Dam	
Cook Creek	Class I
Cooper Creek	Class III
Cornstalk Creek	Class III
Cougar Creek	Class I
Coyote Creek	Class II
Deerhorn Creek	Class III
Dick Creek	Class III
Dry Creek	Class I
Empire Creek	Class III
Faye Creek	Class I
Forty Mile Creek	Class III
Gibson Creek	Class I
Gold Creek	Class II
Granite Creek	Class II
Grizzly Creek	Class III
Haley Creek	Class III
Hall Creek	Class II
Hall Creek West Fork	Class I
Iron Creek	Class III
Jack Creek	Class III
Jerred Creek	Class I
Joe Moses Creek	Class III
John Tom Creek	Class III
Jones Creek	Class I
Kartar Creek	Class III
Kincaid Creek	Class III
King Creek	Class III
Klondyke Creek	Class I
Lime Creek	Class III
Little Jim Creek	Class III
Little Nespelem	Class II
Louie Creek	Class III
Lynx Creek	Class II
Manila Creek	Class III
McAllister Creek	Class III
Meadow Creek	Class III
Mill Creek	Class II
Mission Creek	Class III
Nespelem River	Class II
Nez Perce Creek	Class III
Nine Mile Creek	Class II
Nineteen Mile Creek	Class III
No Name Creek	Class II
North Nanamkin Creek	Class III
North Star Creek	Class III
Okanogan River from Reservation north boundary to Columbia River	Class II
Olds Creek	Class I
Omak Creek	Class II
Onion Creek	Class II
Parmenter Creek	Class III
Peel Creek	Class III
Peter Dan Creek	Class III
Rock Creek	Class I
San Poil River	Class I
Sanpoil, River West Fork	Class II
Seventeen Mile Creek	Class III
Silver Creek	Class III
Sitdown Creek	Class III
Six Mile Creek	Class III
South Nanamkin Creek	Class III
Spring Creek	Class III
Stapaloop Creek	Class III
Stepstone Creek	Class III
Stranger Creek	Class II
Strawberry Creek	Class III
Swimptkin Creek	Class III
Three Forks Creek	Class I
Three Mile Creek	Class III
Thirteen Mile Creek	Class II
Thirty Mile Creek	Class II
Trail Creek	Class III
Twentyfive Mile Creek	Class III
Twentyone Mile Creek	Class III
Twentythree Mile Creek	Class III
Wannacot Creek	Class III

Wells Creek	Class I	LaFleur Lake	LC
Whitelaw Creek	Class III	Little Goose Lake	LC
Wilmont Creek	Class II	Little Owhi Lake	LC
(2) Lakes		McGinnis Lake	LC
Apex Lake	LC	Nicholas Lake	LC
Big Goose Lake	LC	Omak Lake	SRW
Bourgeau Lake	LC	Owhi Lake	SRW
Buffalo Lake	LC	Penley Lake	SRW
Cody Lake	LC	Rebecca Lake	LC
Crawfish Lakes	LC	Round Lake	LC
Camille Lake	LC	Simpson Lake	LC
Elbow Lake	LC	Soap Lake	LC
Fish Lake	LC	Sugar Lake	LC
Gold Lake	LC	Summit Lake	LC
Great Western Lake	LC	Twin Lakes	SRW
Johnson Lake	LC		

§131.36 Toxics criteria for those states not complying with Clean Water Act section 303(c)(2)(B).

(a) *Scope.* This section is not a general promulgation of the section 304(a) criteria for priority toxic pollutants but is restricted to specific pollutants in specific States.

(b) (1) *EPA's Section 304(a) Criteria for Priority Toxic Pollutants*

(#) COMPOUND (A)	CAS Number	B FRESHWATER Criterion Maximum Conc. d (ug/L) B1	B FRESHWATER Criterion Continuous Conc. d (ug/L) B2	C SALTWATER Criterion Maximum Conc. d (ug/L) C1	C SALTWATER Criterion Continuous Conc. d (ug/L) C2	D HUMAN HEALTH (10^{-6} risk for carcinogens) For Consumption of: Water & Organisms (ug/L) D1	D HUMAN HEALTH For Consumption of: Organisms Only (ug/L) D2
1 Antimony	7440360					14 a	4300 a
2 Arsenic	7440382	360 m	190 m	69 m	36 m	0.018 a,b,c	0.14 a,b,c
3 Beryllium	7440417					n	n
4 Cadmium	7440439	3.9 e,m	1.1 e,m	43 m	9.3 m	n	n
5a Chromium (III)	16065831	1700 e,m	210 e,m			n	n
b Chromium (VI)	18540299	16 m	11 m	1100 m	50 m	n	n
6 Copper	7440508	18 e,m	12 e,m	2.9 m	2.9 m		
7 Lead	7439921	82 e,m	3.2 e,m	220 m	8.5 m	n	n
8 Mercury	7439976	2.4 m	0.012 i	2.1 m	0.025 i	0.14	0.15
9 Nickel	7440020	1400 e,m	160 e,m	75 m	8.3 m	610 a	4600 a
10 Selenium	7782492	20	5	300 m	71 m	n	n
11 Silver	7440224	4.1 e,m		2.3 m			
12 Thallium	7440280					1.7 a	6.3 a
13 Zinc	7440666	120 e,m	110 e,m	95 m	86 m		
14 Cyanide	57125	22	5.2	1	1	700 a	220000 a,j
15 Asbestos	1332214					7,000,000 fibers/L k	
16 2,3,7,8-TCDD (Dioxin)	1746016					0.000000013 c	0.000000014 c
17 Acrolein	107028					320	780
18 Acrylonitrile	107131					0.059 a,c	0.66 a,c
19 Benzene	71432					1.2 a,c	71 a,c
20 Bromoform	75252					4.3 a,c	360 a,c
21 Carbon Tetrachloride	56235					0.25 a,c	4.4 a,c
22 Chlorobenzene	108907					680 a	21000 a,j
23 Chlorodibromomethane	124481					0.41 a,c	34 a,c
24 Chloroethane	75003						
25 2-Chloroethylvinyl Ether	110758						
26 Chloroform	67663					5.7 a,c	470 a,c
27 Dichlorobromomethane	75274					0.27 a,c	22 a,c

A			B		C		D	
			FRESHWATER		SALTWATER		HUMAN HEALTH (10^{-6} risk for carcinogens)	
							For Consumption of:	
(#)	COMPOUND	CAS Number	Criterion Maximum Conc. d (ug/L) B1	Criterion Continuous Conc. d (ug/L) B2	Criterion Maximum Conc. d (ug/L) C1	Criterion Continuous Conc. d (ug/L) C2	Water & Organisms (ug/L) D1	Organisms Only (ug/L) D2
28	1,1-Dichloroethane	75343						
29	1,2-Dichloroethane	107062					0.38 a,c	99 a,c
30	1,1-Dichloroethylene	75354					0.057 a,c	3.2 a,c
31	1,2-Dichloropropane	78875						
32	1,3-Dichloropropylene	542756					10 a	1700 a
33	Ethylbenzene	100414					3100 a	29000 a
34	Methyl Bromide	74839					48 a	4000 a
35	Methyl Chloride	74873					n	n
36	Methylene Chloride	75092					4.7 a,c	1600 a,c
37	1,1,2,2-Tetrachloroethane	79345					0.17 a,c	11 a,c
38	Tetrachloroethylene	127184					0.8 c	8.85 c
39	Toluene	108883					6800 a	200000 a
40	1,2-Trans-Dichloroethylene	156605						
41	1,1,1-Trichloroethane	71556					n	n
42	1,1,2-Trichloroethane	79005					0.60 a,c	42 a,c
43	Trichloroethylene	79016					2.7 c	81 c
44	Vinyl Chloride	75014					2 c	525 c
45	2-Chlorophenol	95578						
46	2,4-Dichlorophenol	120832					93 a	790 a,j
47	2,4-Dimethylphenol	105679						
48	2-Methyl-4,6-Dinitrophenol	534521					13.4	765
49	2,4-Dinitrophenol	51285					70 a	14000 a
50	2-Nitrophenol	88755						
51	4-Nitrophenol	100027						
52	3-Methyl-4-Chlorophenol	59507						
53	Pentachlorophenol	87865	20 f	13 f	13	7.9	0.28 a,c	8.2 a,c,j
54	Phenol	108952					21000 a	4600000 a,j
55	2,4,6-Trichlorophenol	88062					2.1 a,c	6.5 a,c
56	Acenaphthene	83329						

A			B		C		D	
			FRESHWATER		SALTWATER		HUMAN HEALTH (10^{-6} risk for carcinogens)	
			Criterion Maximum Conc. d (ug/L)	Criterion Continuous Conc. d (ug/L)	Criterion Maximum Conc. d (ug/L)	Criterion Continuous Conc. d (ug/L)	For Consumption of: Water & Organisms (ug/L)	Organisms Only (ug/L)
(#)	COMPOUND	CAS Number	B1	B2	C1	C2	D1	D2
57	Acenaphthylene	208968						
58	Anthracene	120127					9600 a	110000 a
59	Benzidine	92875					0.00012 a,c	0.00054 a,c
60	Benzo(a)Anthracene	56553					0.0028 c	0.031 c
61	Benzo(a)Pyrene	50328					0.0028 c	0.031 c
62	Benzo(b)Fluoranthene	205992					0.0028 c	0.031 c
63	Benzo(ghi)Perylene	191242						
64	Benzo(k)Fluoranthene	207089					0.0028 c	0.031 c
65	Bis(2-Chloroethoxy)Methane	111911						
66	Bis(2-Chloroethyl)Ether	111444					0.031 a,c	1.4 a,c
67	Bis(2-Chloroisopropyl)Ether	108601					1400 a	170000 a
68	Bis(2-Ethylhexyl)Phthalate	117817					1.8 a,c	5.9 a,c
69	4-Bromophenyl Phenyl Ether	101553						
70	Butylbenzyl Phthalate	85687						
71	2-Chloronaphthalene	91587						
72	4-Chlorophenyl Phenyl Ether	7005723						
73	Chrysene	218019					0.0028 c	0.031 c
74	Dibenzo(a,h)Anthracene	53703					0.0028 c	0.031 c
75	1,2-Dichlorobenzene	95501					2700 a	17000 a
76	1,3-Dichlorobenzene	541731					400	2600
77	1,4-Dichlorobenzene	106467					400	2600
78	3,3'-Dichlorobenzidine	91941					0.04 a,c	0.077 a,c
79	Diethyl Phthalate	84662					23000 a	120000 a
80	Dimethyl Phthalate	131113					313000	2900000
81	Di-n-Butyl Phthalate	84742					2700 a	12000 a
82	2,4-Dinitrotoluene	121142					0.11 c	9.1 c
83	2,6-Dinitrotoluene	606202						
84	Di-n-Octyl Phthalate	117840						
85	1,2-Diphenylhydrazine	122667					0.040 a,c	0.54 a,c

A			B		C		D	
			FRESHWATER		SALTWATER		HUMAN HEALTH (10^{-6} risk for carcinogens)	
							For Consumption of:	
(#)	COMPOUND	CAS Number	Criterion Maximum Conc. d (ug/L) B1	Criterion Continuous Conc. d (ug/L) B2	Criterion Maximum Conc. d (ug/L) C1	Criterion Continuous Conc. d (ug/L) C2	Water & Organisms (ug/L) D1	Organisms Only (ug/L) D2
86	Fluoranthene	206440					300 a	370 a
87	Fluorene	86737					1300 a	14000 a
88	Hexachlorobenzene	118741					0.00075 a,c	0.00077 a,c
89	Hexachlorobutadiene	87683					0.44 a,c	50 a,c
90	Hexachlorocyclopentadiene	77474					240 a	17000 a,j
91	Hexachloroethane	67721					1.9 a,c	8.9 a,c
92	Indeno(1,2,3-cd)Pyrene	193395					0.0028 c	0.031 c
93	Isophorone	78591					8.4 a,c	600 a,c
94	Naphthalene	91203						
95	Nitrobenzene	98953					17 a	1900 a,j
96	N-Nitrosodimethylamine	62759					0.00069 a,c	8.1 a,c
97	N-Nitrosodi-n-Propylamine	621647						
98	N-Nitrosodiphenylamine	86306					5.0 a,c	16 a,c
99	Phenanthrene	85018						
100	Pyrene	129000					960 a	11000 a
101	1,2,4-Trichlorobenzene	120821						
102	Aldrin	309002	3 g		1.3 g		0.00013 a,c	0.00014 a,c
103	alpha-BHC	319846					0.0039 a,c	0.013 a,c
104	beta-BHC	319857					0.014 a,c	0.046 a,c
105	gamma-BHC	58899	2 g	0.08 g	0.16 g		0.019 c	0.063 c
106	delta-BHC	319868						
107	Chlordane	57749	2.4 g	0.0043 g	0.09 g	0.004 g	0.00057 a,c	0.00059 a,c
108	4-4'-DDT	50293	1.1 g	0.001 g	0.13 g	0.001 g	0.00059 a,c	0.00059 a,c
109	4,4'-DDE	72559					0.00059 a,c	0.00059 a,c
110	4,4'-DDD	72548					0.00083 a,c	0.00084 a,c
111	Dieldrin	60571	2.5 g	0.0019 g	0.71 g	0.0019 g	0.00014 a,c	0.00014 a,c
112	alpha-Endosulfan	959988	0.22 g	0.056 g	0.034 g	0.0087 g	0.93 a	2.0 a
113	beta-Endosulfan	33213659	0.22 g	0.056 g	0.034 g	0.0087 g	0.93 a	2.0 a

A			B		C		D	
			FRESHWATER		SALTWATER		HUMAN HEALTH (10^{-6} risk for carcinogens)	
							For Consumption of:	
(#)	COMPOUND	CAS Number	Criterion Maximum Conc. d (ug/L) B1	Criterion Continuous Conc. d (ug/L) B2	Criterion Maximum Conc. d (ug/L) C1	Criterion Continuous Conc. d (ug/L) C2	Water & Organisms (ug/L) D1	Organisms Only (ug/L) D2
114	Endosulfan Sulfate	1031078					0.93 a	2.0 a
115	Endrin	72208	0.18 g	0.0023 g	0.037 g	0.0023 g	0.76 a	0.81 a,j
116	Endrin Aldehyde	7421934					0.76 a	0.81 a,j
117	Heptachlor	76448	0.52 g	0.0038 g	0.053 g	0.0036 g	0.00021 a,c	0.00021 a,c
118	Heptachlor Epoxide	1024573	0.52 g	0.0038 g	0.053 g	0.0036 g	0.00010 a,c	0.00011 a,c
119	PCB-1242	53469219		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
120	PCB-1254	11097691		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
121	PCB-1221	11104282		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
122	PCB-1232	11141165		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
123	PCB-1248	12672296		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
124	PCB-1260	11096825		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
125	PCB-1016	12674112		0.014 g		0.03 g	0.000044 a,c	0.000045 a,c
126	Toxaphene	8001352	0.73	0.0002	0.21	0.0002	0.00073 a,c	0.00075 a,c
Total No. of Criteria (h) =			24	29	23	27	91	90

Footnotes

a. Criteria revised to reflect current agency q_1* or RfD, as contained in the Integrated Risk Information System (IRIS). The fish tissue bioconcentration factor (BCF) from the 1980 criteria documents was retained in all cases.

b. The criteria refers to the inorganic form only.

c. Criteria in the matrix based on carcinogenicity (10^{6} risk). For a risk level of 10^{5}, move the decimal point in the matrix value one place to the right.

d. Criteria Maximum Concentration (CMC) = the highest concentration of a pollutant to which aquatic life can be exposed for a short period of time (1-hour average) without deleterious effects. Criteria Continuous Concentration (CCC) = the highest concentration of a pollutant to which aquatic life can be exposed for an extended period of time (4 days) without deleterious effects. ug/L = micrograms per liter.

e. Freshwater aquatic life criteria for these metals are expressed as a function of total hardness (mg/L), and as a function of the pollutant's water effect ratio, WER, as defined in §131.36(c). The equations are provided in matrix at §131.36(b)(2). Values displayed above in the matrix correspond to a total hardness of 100 mg/L and a water effect ratio of 1.0.

f. Freshwater aquatic life criteria for pentachlorophenol are expressed as a function of pH, and are calculated as follows. Values displayed above in the matrix correspond to a pH of 7.8.

$CMC = \exp(1.005(pH) - 4.830)$ $CCC = \exp(1.005(pH) - 5.290)$

g. Aquatic life criteria for these compounds were issued in 1980 utilizing the 1980 Guidelines for criteria development. The acute values shown are final acute values (FAV) which by the 1980 Guidelines are instantaneous values as contrasted with a CMC which is a one-hour average.

h. These totals simply sum the criteria in each column. For aquatic life, there are 30 priority toxic pollutants with some type of freshwater or saltwater, acute or chronic criteria. For human health, there are 91 priority toxic pollutants with either "water + fish" or "fish only" criteria. Note that these totals count chromium as one pollutant even though EPA has developed criteria based on two valence states. In the matrix, EPA has assigned numbers 5a and 5b to the criteria for chromium to reflect the fact that the list of 126 priority toxic pollutants includes only a single listing for chromium.

i. If the CCC for total mercury exceeds 0.012 ug/L more than once in a 3-year period in the ambient water, the edible portion of aquatic species of concern must be analyzed to determine whether the concentration of methyl mercury exceeds the FDA action level (1.0 mg/kg). If the FDA action level is exceeded, the State must notify the appropriate EPA Regional Administrator, initiate a revision of its mercury criterion in its water quality standards so as to protect designated uses, and take other appropriate action such as issuance of a fish consumption advisory for the affected area.

j. No criteria for protection of human health from consumption of aquatic organisms (excluding water) was presented in the 1980 criteria document or in the 1986 Quality Criteria for Water. Nevertheless, sufficient information was presented in the 1980 document to allow a calculation of a criterion, even though the results of such a calculation were not shown in the document.

k. The criterion for asbestos is the MCL (56 FR 3526, January 30, 1991).

l. This letter not used as a footnote.

m. Criteria for these metals are expressed as a function of the water effect ratio, WER, as defined in 40 CFR 131.36(c).

CMC = column B1 or C1 value × WER

CCC = column B2 or C2 value × WER

n. EPA is not promulgating human health criteria for this contaminant. However, permit authorities should address this contaminant in NPDES permit actions using the State's existing narrative criteria for toxics.

General Notes:

1. This chart lists all of EPA's priority toxic pollutants whether or not criteria recommendations are available. Blank spaces indicate the absence of criteria recommendations. Because of variations in chemical nomenclature systems, this listing of toxic pollutants does not duplicate the listing in Appendix A of 40 CFR Part 423. EPA has added the Chemical Abstracts Service (CAS) registry numbers, which provide a unique identification for each chemical.

2. The following chemicals have organoleptic based criteria recommendations that are not included on this chart (for reasons which are discussed in the preamble): copper, zinc, chlorobenzene, 2-chlorophenol, 2,4-dichlorophenol, acenaphthene, 2,4-dimethylphenol, 3-methyl-4-chlorophenol, hexachlorocyclopentadiene, pentachlorophenol, phenol.

3. For purposes of this rulemaking, freshwater criteria and saltwater criteria apply as specified in 40 CFR 131.36(c).

(2) *Factors for Calculating Metals Criteria*

$CMC = WER \exp\{m_A[\ln(\text{hardness})] + b_A\}$

$CCC = WER \exp\{m_C[\ln(\text{hardness})] + b_C\}$

$$CMC = WER \exp\{m_A[\ln(hardness)] + b_A\} \quad CCC = WER \exp\{m_C[\ln(hardness)] + b_C\}$$

	m_A	b_A	m_C	b_C
Cadmium	1 128	-3 828	0 7852	-3 490
Copper	0.9422	-1 464	0 8545	-1 465
Chromium (III)	0 8190	3 688	0 8190	1 561
Lead	1 273	-1 460	1 273	-4 705
Nickel	0 8460	3 3612	0 8460	1 1645
Silver	1 72	-6 52		
Zinc	0 8473	0 8604	0 8473	0 7614

Note The term "exp" represents the base e exponential function

(c) *Applicability*

(1) The criteria in paragraph (b) of this section apply to the States' designated uses cited in paragraph (d) of this section and supersede any criteria adopted by the State, except when State regulations contain criteria which are more stringent for a particular use in which case the State's criteria will continue to apply.

(2) The criteria established in this section are subject to the State's general rules of applicability in the same way and to the same extent as are the other numeric toxics criteria when applied to the same use classifications including mixing zones, and low flow values below which numeric standards can be exceeded in flowing fresh waters.

(i) For all waters with mixing zone regulations or implementation procedures, the criteria apply at the appropriate locations within or at the boundary of the mixing zones; otherwise the criteria apply throughout the waterbody including at the end of any discharge pipe, canal or other discharge point.

(ii) A State shall not use a low flow value below which numeric standards can be exceeded that is less stringent than the following for waters suitable for the establishment of low flow return frequencies (i.e., streams and rivers)

Aquatic Life

Acute criteria (CMC)	1 Q 10 or 1 B 3
Chronic criteria (CCC)	7 Q 10 or 4 B 3

Human Health

Non-carcinogens	30 Q 5
Carcinogens	Harmonic mean flow

Where:

CMC—criteria maximum concentration—the water quality criteria to protect against acute effects in aquatic life and is the highest instream concentration of a priority toxic pollutant consisting of a one-hour average not to be exceeded more than once every three years on the average;

CCC—criteria continuous concentration—the water quality criteria to protect against chronic effects in aquatic life is the highest instream concentration of a priority toxic pollutant consisting of a 4-day average not to be exceeded more than once every three years on the average;

1 Q 10 is the lowest one day flow with an average recurrence frequency of once in 10 years determined hydrologically;

1 B 3 is biologically based and indicates an allowable exceedence of once every 3 years. It is determined by EPA's computerized method (DFLOW model);

7 Q 10 is the lowest average 7 consecutive day low flow with an average recurrence frequency of once in 10 years determined hydrologically;

4 B 3 is biologically based and indicates an allowable exceedence for 4 consecutive days once every 3 years. It is determined by EPA's computerized method (DFLOW model);

30 Q 5 is the lowest average 30 consecutive day low flow with an average recurrence frequency of once in 5 years determined hydrologically; and the harmonic mean flow is a long term mean flow value calculated by dividing the number of daily flows analyzed by the sum of the reciprocals of those daily flows.

(iii) If a State does not have such a low flow value for numeric standards compliance, then none shall apply and the criteria included in paragraph (d) of this section herein apply at all flows.

(3) The aquatic life criteria in the matrix in paragraph (b) of this section apply as follows:

(i) For waters in which the salinity is equal to or less than 1 part per thousand 95% or more of the time, the applicable criteria are the freshwater criteria in Column B;

(ii) For waters in which the salinity is equal to or greater than 10 parts per thousand 95% or more of the time, the applicable criteria are the saltwater criteria in Column C; and

(iii) For waters in which the salinity is between 1 and 10 parts per thousand as defined in paragraphs (c)(3) (i) and (ii) of this section, the applicable criteria are the more stringent of the freshwater or saltwater criteria. However, the Regional Administrator may approve the use of the alternative freshwater or saltwater criteria if scientifically defensible information and data demonstrate that on a site-specific basis the biology of the waterbody is dominated by freshwater aquatic life and that freshwater criteria are more appropriate; or conversely, the biology of the waterbody is dominated by saltwater aquatic life and that saltwater criteria are more appropriate.

(4) *Application of metals criteria.*

(i) For purposes of calculating freshwater aquatic life criteria for metals from the equations in paragraph (b)(2) of this section, the minimum hardness allowed for use in those equations shall not be less than 25 mg/l, as calcium carbonate, even if the actual ambient hardness is less than 25 mg/l as calcium carbonate. The maximum hardness value for use in those equations shall not exceed 400 mg/l as calcium carbonate, even if the actual ambient hardness is greater than 400 mg/l as calcium carbonate. The same provisions apply for calculating the metals criteria for the comparisons provided for in paragraph (c)(3)(iii) of this section.

(ii) The hardness values used shall be consistent with the design discharge conditions established in paragraph (c)(2) of this section for flows and mixing zones.

(iii) The criteria for metals (compounds #1–#13 in paragraph (b) of this section) are expressed as total recoverable. For purposes of calculating aquatic life criteria for metals from the equations in footnote M. in the criteria matrix in paragraph (b)(1) of this section and the equations in paragraph (b)(2) of this section, the water-effect ratio is computed as a specific pollutant's acute or chronic toxicity values measured in water from the site covered by the standard, divided by the respective acute or chronic toxicity value in laboratory dilution water. The water-effect ratio shall be assigned a value of 1.0, except where the permitting authority assigns a different value that protects the designated uses of the water body from the toxic effects of the pollutant, and is derived from suitable tests on sampled water representative of conditions in the affected water body, consistent with the design discharge conditions established in paragraph (c)(2) of this section. For purposes of this paragraph, the term acute toxicity value is the toxicity test results, such as the concentration lethal to one-half of the test organisms (i.e., LC50) after 96 hours of exposure (e.g., fish toxicity tests) or the effect concentration to one-half of the test organisms, (i.e., EC50) after 48 hours of exposure (e.g., daphnia toxicity tests). For purposes of this paragraph, the term chronic value is the result from appropriate hypothesis testing or regression analysis of measurements of growth, reproduction, or survival from life cycle, partial life cycle, or early life stage tests. The determination of acute and chronic values shall be according to current standard protocols (e.g., those published by the American Society for Testing Materials (ASTM)) or other comparable methods. For calculation of criteria using site-specific values for both the hardness and the water effect ratio, the hardness used in the equations in paragraph (b)(2) of this section shall be as required in paragraph (c)(4)(ii) of this section. Water hardness shall be calculated from the measured calcium and magnesium ions present, and the ratio of calcium to magnesium shall be approximately the same in standard laboratory toxicity testing water as in the site water.

(d) *Criteria for Specific Jurisdictions*—

(1) *Rhode Island, EPA Region 1.*

(i) All waters assigned to the following use classifications in the Water Quality Regulations for Water Pollution Control adopted under Chapters 46-12, 42-17.1, and 42-35 of the General Laws of Rhode Island are subject to the criteria in paragraph (d)(1)(ii) of this section, without exception:

6.21 Freshwater	6.22 Saltwater
Class A	Class SA
Class B	Class SB
Class C	Class SC

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(1)(i) of this section:

Use classification	Applicable criteria
Class A Class B waters where water supply use is designated	These classifications are assigned the criteria in: Column D1—all
Class B waters where water supply use is not designated; Class C; Class SA; Class SB; Class SC	Each of these classifications is assigned the criteria in: Column D2—all

(iii) The human health criteria shall be applied at the 10^{-5} risk level, consistent with the State policy. To determine appropriate value for carcinogens, see footnote c in the criteria matrix in paragraph (b)(1) of this section.

(2) *Vermont, EPA Region 1.*

(i) All waters assigned to the following use classifications in the Vermont Water Quality Standards adopted under the authority of the Vermont Water Pollution Control Act (10 V.S.A., Chapter 47) are subject to the criteria in paragraph (d)(2)(ii) of this section, without exception:

Class A
Class B
Class C

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(2)(i) of this section:

Use classification	Applicable criteria
Class A Class B waters where water supply use is designated	This classification is assigned the criteria in Column B1—all Column B2—all Column D1—all
Class B waters where water supply use is not designated Class C	These classifications are assigned the criteria in: Column B1—all Column B2—all Column D2—all

(iii) The human health criteria shall be applied at the State-proposed 10^{-6} risk level.

(3) *New Jersey, EPA Region 2.*

(i) All waters assigned to the following use classifications in the New Jersey Administrative Code (N.J.A.C.) 7:9-4.1 et seq., Surface Water Quality Standards, are subject to the criteria in paragraph (d)(3)(ii) of this section, without exception.

N.J.A.C. 7:9-4.12(b): Class PL
N.J.A.C. 7:9-4.12(c): Class FW2
N.J.A.C. 7:9-4.12(d): Class SE1
N.J.A.C. 7:9-4.12(e): Class SE2
N.J.A.C. 7:9-4.12(f): Class SE3
N.J.A.C. 7:9-4.12(g): Class SC
N.J.A.C. 7:9-4.13(a): Delaware River Zones 1C, 1D, and 1E
N.J.A.C. 7:9-4.13(b): Delaware River Zone 2
N.J.A.C. 7:9-4.13(c): Delaware River Zone 3
N.J.A.C. 7:9-4.13(d): Delaware River Zone 4
N.J.A.C. 7:9-4.13(e): Delaware River Zone 5
N.J.A.C. 7:9-4.13(f): Delaware River Zone 6

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(3)(i) of this section:

Use classification	Applicable criteria
PL (Freshwater Pinelands), FW2	These classifications are assigned the criteria in: Column B1-all except #102, 105, 107, 108, 111, 112, 113, 115, 117, 118. Column B2—all excep #105, 107, 108, 111, 112, 113, 115, 117, 118, 119, 120, 121, 122, 123, 124, and 125 Column D1-all at a 10^{-6} risk level except #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, at a 10^{-5} risk level. Column D2—all at a 10^{-6} risk level except #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105; 23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, at a 10^{-5} risk level.
PL (Saline Water Pinelands), SE1, SE2, SE3, SC	These classifications are assigned the criteria in

Use classification	Applicable criteria
	Column C1—all except #102, 105, 107, 108, 111, 112, 113, 115, 117, and 118 Column C2—all except #105, 107, 108, 111, 112, 113, 115, 117, 118, 119, 120, 121, 122, 123, 124, and 125. Column D2—all at a 10^{-6} risk level except #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105; #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, at a 10^{-5} risk level
Delaware River zones 1C, 1D, 1E, 2, 3, 4, 5 and Delaware Bay zone 6	These classifications are assigned the criteria in Column B1—all Column B2—all Column D1—all at a 10^{-6} risk level except #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105; #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, at a 10^{-5} risk level. Column D2—all at a 10^{-6} risk level except #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, at a 10^{-5} risk level.
Delaware River zones 3, 4, and 5, and Delaware Bay zone 6	These classifications are assigned the criteria in Column C1—all Column C2—all Column D2—all at a 10^{-6} risk level except #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, #23, 30, 37, 38, 42, 68, 89, 91, 93, 104, 105, at a 10^{-5} risk level

(iii) The human health criteria shall be applied at the State-proposed 10^{-6} risk level for EPA rated Class A, B_1, and B_2 carcinogens; EPA rated Class C carcinogens shall be applied at 10^{-5} risk level. To determine appropriate value for carcinogens, see footnote c. in the matrix in paragraph (b)(1) of this section.

(4) *Puerto Rico, EPA Region 2.*

(i) All waters assigned to the following use classifications in the Puerto Rico Water Quality Standards (promulgated by Resolution Number R-83-5-2) are subject to the criteria in paragraph (d)(4)(ii) of this section, without exception.

Article 2.2.2—Class SB
Article 2.2.3—Class SC
Article 2.2.4—Class SD

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(4)(i) of this section:

Use classification	Applicable criteria
Class SD	This Classification is assigned the criteria in: Column B1—all, except: 10, 102, 105, 107, 108, 111, 112, 113, 115, 117, and 126 Column B2—all, except: 105, 107, 108, 112, 113, 115, and 117. Column D1—all, except 6, 14, 105, 112, 113, and 115. Column D2—all, except: 14, 105, 112, 113, and 115.
Class SB, Class SC	This Classification is assigned the criteria in: Column C1—all, except 4, 5b, 7, 8, 10, 11, 13, 102, 105, 107, 108, 111, 112, 113, 115, 117, and 126 Column C2—all, except 4, 5b, 10, 13, 108, 112, 113, 115, and 117. Column D2—all, except: 14, 105, 112, 113, and 115.

(iii) The human health criteria shall be applied at the State-proposed 10^{-5} risk level. To determine appropriate value for carcinogens, see footnote c. in the criteria matrix in paragraph (b)(1) of this section.

(5) *District of Columbia, EPA Region 3.*

(i) All waters assigned to the following use classifications in chapter 11 Title 21 DCMR, Water Quality Standards of the District of Columbia are subject to the criteria in paragraph (d)(5)(ii) of this section, without exception:

1101.2 Class C waters

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classification identified in paragraph (d)(5)(i) of this section:

Use classification	Applicable criteria
Class C	This classification is assigned the additional criteria in: Colum B2—#10, 118, 126 Colum D1—#15, 16, 44, 67, 68, 79, 80, 81, 88, 114, 116, 118 Colum D2—all

(iii) The human health criteria shall be applied at the State-adopted 10^{-6} risk level.

(6) *Florida, EPA Region 4.*

(i) All waters assigned to the following use classifications in Chapter 17-301 of the Florida Administrative Code (i.e., identified in Section 17-302.600) are subject to the criteria in paragraph (d)(6)(ii) of this section, without exception:

Class I
Class II
Class III

(ii) The following criteria from the matrix paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(6)(i) of this section:

Use classification	Applicable criteria
Class I	This classification is assigned the criteria in Column D1—#16
Class II and Class III (marine)	This classification is assigned the criteria in Column D2-#16
Class III (fresh water)	This classification is assigned the criteria in. Column D2—#16

(iii) The human health criteria shall be applied at the State-adopted 10^{-6} risk level.

(7) *Michigan, EPA Region 5.*

(i) All waters assigned to the following use classifications in the Michigan Department of Natural Resources Commission General Rules, R 323.1100 designated uses, as defined at R 323.1043. Definitions; A to N, (i.e., identified in Section (g) "Designated use") are subject to the criteria in paragraph (d)(7)(ii) of this section, without exception:

Agriculture
Navigation
Industrial Water Supply
Public Water Supply at the Point of Water Intake
Warmwater Fish

Other Indigenous Aquatic Life and Wildlife

Partial Body Contact Recreation

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(7)(i) of this section

Use classification	Applicable criteria
Public Water supply	This classification is assigned the criteria in Column B1—all, Column B2—all, Column D1—all
All other designations	These classifications are assigned the criteria in Column B1—all, Column B2—all, and Column D2—all

(iii) The human health criteria shall be applied at the State-adopted 10^{-5} risk level. To determine appropriate value for carcinogens, see footnote c in the criteria matrix in paragraph (b)(1) of this section.

(8) *Arkansas, EPA Region 6*

(i) All waters assigned to the following use classification in section 4C (Waterbody uses) identified in Arkansas Department of Pollution Control and Ecology's Regulation No 2 as amended and entitled, "Regulation Establishing Water Quality Standards for Surface Waters of the State of Arkansas" are subject to the criteria in paragraph (d)(8)(ii) of this section, without exception

Extraordinary Resource Waters

Ecologically Sensitive Waterbody

Natural and Scenic Waterways

Fisheries

(1) Trout

(2) Lakes and Reservoirs

(3) Streams

(a) Ozark Highlands Ecoregion

(b) Boston Mountains Ecoregion

(c) Arkansas River Valley Ecoregion

(d) Ouachita Mountains Ecoregion

(e) Typical Gulf Coastal Ecoregion

(f) Spring Water-influenced Gulf Coastal Ecoregion

(g) Least-altered Delta Ecoregion

(h) Channel-altered Delta Ecoregion

Domestic Water Supply

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classification identified in paragraph (d)(8)(i) of this section

Use classification	Applicable criteria
Extraordinary Resource Waters	
Ecologically Sensitive Waterbody	
Natural and Scenic Waterways	
Fisheries	
(1) Trout	
(2) Lakes and Reservoirs	
(3) Streams	
(a) Ozark Highlands Ecoregion	
(b) Boston Mountains Ecoregion	
(c) Arkansas River Valley Ecoregion	
(d) Ouachita Mountains Ecoregion	
(e) Typical Gulf Coastal Ecoregion	
(f) Spring Water-influenced Gulf Coastal Ecoregion	
(g) Least-altered Delta Ecoregion	
(h) Channel-altered Delta Ecoregion	These uses are each assigned the criteria in - Column B1 - #4 5a 5b 6 7 8 9 10 11 13 14; Column B2— #4, 5a, 5b, 6, 7, 8, 9, 10, 13, 14

(9) *Kansas, EPA Region 7*

(i) All waters assigned to the following use classification in the Kansas Department of Health and Environment regulations, K.A.R. 28-16-28b through K.A.R. 28-16-28f, are subject to the criteria in paragraph (d)(9)(ii) of this section, without exception

Section 28-16-28d

Section (2)(A)—Special Aquatic Life Use Waters

Section (2)(B)—Expected Aquatic Life Use Waters

Section (2)(C)—Restricted Aquatic Life Use Waters

Section (3)—Domestic Water Supply

Section (6)(c)—Consumptive Recreation Use

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(9)(i) of this section

Use classification	Applicable criteria
Sections (2)(A), (2)(B), (2)(C), (6)(C)	These classifactions are each assigned all criteria in Column B1, all except #9, 11, 13, 102, 105, 107, 108, 111-113, 115, 117, and 126; Column B2, all except #9, 13, 105, 107, 108, 111-113, 115, 117, 119-125, and 126, and Column D2, all except #9, 112, 113, and 115
Section (3)	This classification is assigned all criteria in Column D1, all except #9, 12, 112, 113, and 115

(iii) The human health criteria shall be applied at the State-proposed 10^{-6} risk level

(10) *California, EPA Region 9*

(i) All waters assigned any aquatic life or human health use classifications in the Water Quality Control Plans for the various Basins of the State ("Basin Plans"), as amended, adopted by the California State Water Resources Control Board ("SWRCB"), except for ocean waters covered by the Water Quality Control Plan for Ocean Waters of California ("Ocean Plan") adopted by the SWRCB with resolution Number 90-27 on March 22, 1990, are subject to the criteria in paragraph (d)(10)(ii) of this section, without exception. These criteria amend the portions of the existing State standards contained in the Basin Plans. More particularly these criteria amend water quality criteria contained in the Basin Plan Chapters specifying water quality objectives (the State equivalent of federal water quality criteria) for the toxic pollutants identified in paragraph (d)(10)(ii) of this section. Although the State has adopted several use designations for each of these waters, for purposes of this action, the specific standards to be applied in paragraph (d)(10)(ii) of this section are based on the presence in all waters of some aquatic life designation and the presence or absence of the MUN use designation (Municipal and domestic supply). (See Basin Plans for more detailed use definitions.)

Other Indigenous Aquatic Life and Wildlife

Partial Body Contact Recreation

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(7)(i) of this section

Use classification	Applicable criteria
Public Water supply	This classification is assigned the criteria in Column B1—all, Column B2—all, Column D1—all
All other designations	These classifications are assigned the criteria in: Column B1—all, Column B2—all, and Column D2—all

(iii) The human health criteria shall be applied at the State-adopted 10^{-5} risk level. To determine appropriate value for carcinogens, see footnote c in the criteria matrix in paragraph (b)(1) of this section.

(8) *Arkansas, EPA Region 6.*

(i) All waters assigned to the following use classification in section 4C (Waterbody uses) identified in Arkansas Department of Pollution Control and Ecology's Regulation No. 2 as amended and entitled, "Regulation Establishing Water Quality Standards for Surface Waters of the State of Arkansas" are subject to the criteria in paragraph (d)(8)(ii) of this section, without exception:

Extraordinary Resource Waters

Ecologically Sensitive Waterbody

Natural and Scenic Waterways

Fisheries

(1) Trout

(2) Lakes and Reservoirs

(3) Streams

(a) Ozark Highlands Ecoregion

(b) Boston Mountains Ecoregion

(c) Arkansas River Valley Ecoregion

(d) Ouachita Mountains Ecoregion

(e) Typical Gulf Coastal Ecoregion

(f) Spring Water-influenced Gulf Coastal Ecoregion

(g) Least-altered Delta Ecoregion

(h) Channel-altered Delta Ecoregion

Domestic Water Supply

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classification identified in paragraph (d)(8)(i) of this section:

Use classification	Applicable criteria
Extraordinary Resource Waters Ecologically Sensitive Waterbody Natural and Scenic Waterways Fisheries (1) Trout (2) Lakes and Reservoirs (3) Streams (a) Ozark Highlands Ecoregion (b) Boston Mountains Ecoregion (c) Arkansas River Valley Ecoregion (d) Ouachita Mountains Ecoregion (e) Typical Gulf Coastal Ecoregion (f) Spring Water-influenced Gulf Coastal Ecoregion (g) Least-altered Delta Ecoregion (h) Channel-altered Delta Ecoregion	These uses are each assigned the criteria in— Column B1— #4, 5a, 5b, 6, 7, 8, 9, 10, 11, 13, 14 Column B2— #4, 5a, 5b, 6, 7, 8, 9, 10, 13, 14

(9) *Kansas, EPA Region 7.*

(i) All waters assigned to the following use classification in the Kansas Department of Health and Environment regulations, K.A.R. 28-16-28b through K.A.R. 28-16-28f, are subject to the criteria in paragraph (d)(9)(ii) of this section, without exception.

Section 28-16-28d

Section (2)(A)—Special Aquatic Life Use Waters

Section (2)(B)—Expected Aquatic Life Use Waters

Section (2)(C)—Restricted Aquatic Life Use Waters

Section (3)—Domestic Water Supply

Section (6)(c)—Consumptive Recreation Use.

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(9)(i) of this section:

Use classification	Applicable criteria
Sections (2)(A), (2)(B), (2)(C) (6)(C)	These classifactions are each assigned all criteria in: Column B1, all except #9, 11, 13, 102, 105, 107, 108, 111-113, 115, 117, and 126; Column B2, all except #9, 13, 105, 107, 108, 111-113, 115, 117, 119-125, and 126, and Column D2, all except #9, 112, 113, and 115
Section (3)	This classification is assigned all criteria in: Column D1, all except #9, 12, 112, 113, and 115

(iii) The human health criteria shall be applied at the State-proposed 10^{-6} risk level.

(10) *California, EPA Region 9.*

(i) All waters assigned any aquatic life or human health use classifications in the Water Quality Control Plans for the various Basins of the State ("Basin Plans"), as amended, adopted by the California State Water Resources Control Board ("SWRCB"), except for ocean waters covered by the Water Quality Control Plan for Ocean Waters of California ("Ocean Plan") adopted by the SWRCB with resolution Number 90-27 on March 22, 1990, are subject to the criteria in paragraph (d)(10)(ii) of this section, without exception. These criteria amend the portions of the existing State standards contained in the Basin Plans. More particularly these criteria amend water quality criteria contained in the Basin Plan Chapters specifying water quality objectives (the State equivalent of federal water quality criteria) for the toxic pollutants identified in paragraph (d)(10)(ii) of this section. Although the State has adopted several use designations for each of these waters, for purposes of this action, the specific standards to be applied in paragraph (d)(10)(ii) of this section are based on the presence in all waters of some aquatic life designation and the presence or absence of the MUN use designation (Municipal and domestic supply). (See Basin Plans for more detailed use definitions.)

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the water and use classifications defined in paragraph (d)(10)(i) of this section and identified below

Water and use classification	Applicable criteria
Waters of the State defined as bays or estuaries except the Sacramento-San Joaquin Delta and San Francisco Bay	These waters are assigned the criteria in Column B1—pollutants 5a and 14 Column B2—pollutants 5a and 14 Column C1—pollutant 14 Column C2—pollutant 14 Column D2—pollutants 1, 12, 17, 18, 21, 22, 29, 30, 32, 33, 37, 38, 42-44, 46, 48, 49, 54, 59, 66, 67, 68, 78-82, 85, 89, 90, 91, 93, 95, 96, 98
Waters of the Sacramento—San Joaquin Delta and waters of the State defined as inland (i.e., all surface waters of the State not bays or estuaries or ocean) that include a MUN use designation	These waters are assigned the criteria in Column B1—pollutants 5a and 14 Column B2—pollutants 5a and 14 Column D1—pollutants 1, 12, 15, 17, 18, 21, 22, 29, 30, 32, 33, 37, 38, 42-48, 49, 59, 66, 68, 78-82, 85, 89, 90, 91, 93, 95, 96, 98
Waters of the State defined as inland without an MUN use designation	These waters are assigned the criteria in: Column B1—pollutants 5a and 14 Column B2—pollutants 5a and 14 Column D2—pollutants 1, 12, 17, 18, 21, 22, 29, 30, 32, 33, 37, 38, 42-44, 46, 48, 49, 54, 59, 66, 67, 68, 78-82, 85, 89, 90, 91, 93, 95, 96, 98
Waters of the San Joaquin River from the mouth of the Merced River to Vernalis	In addition to the criteria assigned to these waters elsewhere in this rule, these waters are assigned the criteria in Column B2—pollutant 10
Waters of Salt Slough, Mud Slough (north) and the San Joaquin River, Sack Dam to the mouth of the Merced River	In addition to the criteria assigned to these waters elsewhere in this rule, these waters are assigned the criteria in Column B1—pollutant 10 Column B2—pollutant 10
Waters of San Francisco Bay upstream to and including Suisun Bay and the Sacramento San Joaquin Delta	These waters are assigned the criteria in Column B1—pollutants 5a, 10* and 14 Column B2—pollutants 5a, 10* and 14 Column C1—pollutant 14 Column C2—pollutant 14 Column D2—pollutants 1, 12, 17, 18, 21, 22, 29, 30, 32, 33, 37, 38, 42-44, 46, 48, 49, 54, 59, 66, 67, 68, 78-82, 85, 89, 90, 91, 93, 95, 96, 98
All inland waters of the United States or enclosed bays and estuaries that are waters of the United States that include an MUN use designation and that the State has either excluded or partially excluded from coverage under its Water Quality Control Plan for Inland Surface Waters of California, Tables 1 and 2, or its Water Quality Control Plan for Enclosed Bays and Estuaries of California, Tables 1 and 2, or has deferred applicability of those tables. (Category (a), (b), and (c) waters described on page 6 of Water Quality Control Plan for Inland Surface Waters of California or page 6 of its Water Quality Control Plan for Enclosed Bays and Estuaries of California.)	These waters are assigned the criteria for pollutants for which the State does not apply Table 1 or 2 standards. These criteria are Column B1—all pollutants Column B2—all pollutants Column D1—all pollutants except #2
All inland waters of the United States that do not include an MUN use designation and that the State has either excluded or partially excluded from coverage under its Water Quality Control Plan for Inland Surface Waters of California, Tables 1 and 2, or has deferred applicability of these tables. (Category (a), (b), and (c) waters described on page 6 of Water Quality Control Plan Inland Surface Waters of California.)	

Water and use classification	Applicable criteria
	These waters are assigned the criteria for pollutants for which the State does not apply Table 1 or 2 standards. These criteria are: Column B1—all pollutants Column B2—all pollutants Column D2—all pollutants except #2
All enclosed bays and estuaries that are waters of the United States that do not include an MUN designation and that the State has either excluded or partially excluded from coverage under its Water Quality Control Plan for Inland Surface Waters of California, Tables 1 and 2, or its Water Quality Control Plan for Enclosed Bays and Estuaries of California, Tables 1 and 2, or has deferred applicability of those tables. (Category (a), (b), and (c) waters described on page 6 of Water Quality Control Plan for Inland Surface Waters of California or page 6 of its Water Quality Control Plan for Enclosed Bays and Estuaries of California.)	These waters are assigned the criteria for pollutants for which the State does not apply Table 1 or 2 standards. These criteria are: Column B1—all pollutants Column B2—all pollutants Column C1—all pollutants Column C2—all pollutants Column D2—all pollutants except #2

* The fresh water selenium criteria are included for the San Francisco Bay estuary because high levels of bioaccumulation of selenium in the estuary indicate that the salt water criteria are underprotective for San Francisco Bay.

(iii) The human health criteria shall be applied at the State-adopted 10^{-6} risk level.

(11) *Nevada, EPA Region 9.*

(i) All waters assigned the use classifications in Chapter 445 of the Nevada Administrative Code (NAC), Nevada Water Pollution Control Regulations, which are referred to in paragraph (d)(11)(ii) of this section, are subject to the criteria in paragraph (d)(11)(ii) of this section, without exception. These criteria amend the existing State standards contained in the Nevada Water Pollution Control Regulations. More particularly, these criteria amend or supplement the table of numeric standards in NAC 445.1339 for the toxic pollutants identified in paragraph (d)(11)(ii) of this section.

(ii) The following criteria from matrix in paragraph (b)(1) of this section apply to the waters defined in paragraph (d)(11)(i) of this section and identified below:

Water and use classification	Applicable criteria
Waters that the State has included in NAC 445.1339 where Municipal or domestic supply is a designated use	These waters are assigned the criteria in: Column B1—pollutant #118 Column B2—pollutant #118 Column D1—pollutants #15, 16, 18, 19, 20, 21, 23, 26, 27, 29, 30, 34, 37, 38, 42, 43, 55, 58–62, 64, 66, 73, 74, 78, 82, 85, 87–89, 91, 92, 96, 98, 100, 103, 104, 105, 114, 116, 117, 118
Waters that the State has included in NAC 445.1339 where Municipal or domestic supply is not a designated use	These waters are assigned the criteria in: Column B1—pollutant #118 Column B2—pollutant #118 Column D2—all pollutants except #2

(iii) The human health criteria shall be applied at the 10^{-5} risk level, consistent with State policy. To determine appropriate value for carcinogens, see footnote c in the criteria matrix in paragraph (b)(1) of this section.

(12) *Alaska, EPA Region 10.*

(i) All waters assigned to the following use classifications in the Alaska Administrative Code (AAC), Chapter 18 (i.e., identified in 18 AAC 70.020) are subject to the criteria in paragraph (d)(12)(ii) of this section, without exception.

70.020.(1) (A) Fresh Water
70.020.(1) (A) Water Supply
(i) Drinking, culinary, and food processing,
(iii) Aquaculture;
70.020.(1) (B) Water Recreation
(i) Contact recreation,
(ii) Secondary recreation;
70.020.(1) (C) Growth and propagation of fish, shellfish, other aquatic life, and wildlife
70.020.(2) (A) Marine Water
70.020.(2) (A) Water Supply
(i) Aquaculture,
70.020.(2) (B) Water Recreation
(i) contact recreation,
(ii) secondary recreation,
70.020.(2) (C) Growth and propagation of fish, shellfish, other aquatic life, and wildlife;
70.020.(2) (D) Harvesting for consumption of raw mollusks or other raw aquatic life.

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(12)(i) of this section:

Use classification	Applicable criteria
(1)(A) i	Column B1—all Column B2—#10 Column D1

Use classification	Applicable criteria
	#s 2 16, 18-21 23, 26 27, 29 30, 32, 37, 38, 42-44 53 55, 59-62 64 66 68, 73, 74, 78 82 85 88 89 91-93, 96 98 102-105, 107-111 117-126
(1)(A) iii	Column B1—all Column B2—#10 Column D2 #s 2 14, 16, 18-21 22, 23, 26, 27, 29 30, 32, 37 38, 42-44 46 53, 54, 55, 59-62, 64, 66, 68, 73 74, 78 82, 85 88-93, 95, 96, 98, 102-105, 107-111, 115-126
(1)(B) i, (1)(B) ii, (1)(C)	Column B1—all Column B2—#10 Column D2 #s 2 14 16, 18-21 22, 23, 26 27 29 30 32 37 38, 42-44 46 53, 54, 55, 59-62 64 66, 68 73 74, 78, 82, 85 88-93, 95, 96, 98, 102-105, 107-111 115-126
(2)(A)ii, (2)(B)i, and (2)(B)ii, (2)(C), (2)(D)	Column C1—all Column C2—#10 Column D2 #s 2, 14, 16, 18-21, 22 23, 26, 27 29 30, 32, 37, 38, 42-44, 46, 53, 54 55, 59-62, 64 66 68, 73, 74, 78 82, 85, 88-93, 95, 96, 98, 102-105, 107-111, 115-126

(iii) The human health criteria shall be applied at the State-proposed risk level of 10^{-5}. To determine appropriate value for carcinogens, see footnote c in the criteria matrix in paragraph (b)(1) of this section.

(13) *Idaho, EPA Region 10*

(i) All waters assigned to the following use classifications in the Idaho Administrative Procedures Act (IDAPA), Chapter 16 (i.e., identified in IDAPA 16.01.2100,02-16.01 2100,07) are subject to the criteria in paragraph (d)(13)(ii) of this section, without exception.

16.01.2100.01.b Domestic Water Supplies

16.01.2100.02.a Cold Water Biota

16.01.2100.02.b Warm Water Biota

16.01.2100.02cc. Salmonid Spawning

16.01.2100.03.a Primary Contact Recreation

16.01.2100.03.b Secondary Contact Recreation

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(13)(i) of this section.

Use classification	Applicable criteria
01 b	This classification is assigned the criteria in Column D1—all except #14 and 115
02 a 02 b 02 cc	These classifications are assigned the criteria in Column B1—all Column B2—all Column D2—all
03 a	This classification is assigned the criteria in Column D2—all
03 b	This classification is assigned the criteria in Column D2—all

(iii) The human health criteria shall be applied at the 10^{-6} risk level, consistent with State policy.

(14) *Washington, EPA Region 10*

(i) All waters assigned to the following use classifications in the Washington Administrative Code (WAC), Chapter 173-201 (i.e., identified in WAC 173-201-045) are subject to the criteria in paragraph (d)(14)(ii) of this section, without exception.

173-201-045

Fish and Shellfish

Fish

Water Supply (domestic)

Recreation

(ii) The following criteria from the matrix in paragraph (b)(1) of this section apply to the use classifications identified in paragraph (d)(14)(i) of this section:

Use classification	Applicable criteria
Fish and Shellfish, Fish (Aquatic life criteria for arsenic and selenium withdrawn July 6, 1993 58 FR 36141)	These classifications are assigned the criteria in ~~Column B1 and B(2)—#2, 10~~ ~~Column C1—#2, 10~~ Column C2—~~#2~~, 6, ~~10~~, 14 Column D2—all
Water Supply (domestic)	These classifictions are assigned the criteria in Column D1—all
Recreation	This classification is assigned the criteria in Column D2—Marine waters and freshwaters not protected for domestic water supply

(iii) The human health criteria shall be applied at the State proposed risk level of 10^{-6}.

[§131.36 added at 57 FR 60910, Dec. 22, 1992]

APPENDIX B

Chronological Summary of Federal Water Quality Standards Promulgation Actions

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
OFFICE OF SCIENCE AND TECHNOLOGY
STANDARDS AND APPLIED SCIENCE DIVISION

JANUARY 1993

CHRONOLOGICAL SUMMARY OF FEDERAL WATER QUALITY STANDARDS PROMULGATION ACTIONS

	STATE	DATE	STATUS	REFERENCE	ACTION
1.	Kentucky	12/2/74	Final	39 FR 41709	Established statement in WQS giving EPA Administrator authority to grant a temporary exception to stream classification and/or criteria after case-by-case studies. Also, established statement that streams not listed in the WQS are understood to be classified as Aquatic Life and criteria for this use to be met.
2*.	Arizona	6/22/76	Final	41 FR 25000	Established nutrient standards for 11 streams.
3.	Nebraska	6/6/78	Final	43 FR 24529	Redesignated eight stream segments for full body contact recreation and three for partial body contact recreation and the protection of fish and wildlife.
4.	Mississippi	4/30/79	Final	44 FR 25223	Established dissolved oxygen criterion for all water uses recognized by the State. Established criterion for a daily average of not less than 5.0 mg/l with a daily instantaneous minimum of not less than 4.0 mg/l.

5.	Alabama	11/26/79	Proposed	44 FR 67442	Proposal to reestablish previously approved use classifications for segments of four navigable waterways, Five Mile Creek, Opossum Creek, Valley Creek, Village Creek, and upgrade the use designation of a segment of Village Creek from river mile 30 to its source.
6.	Alabama	2/14/80	Final	45 FR 9910	Established beneficial stream use classification for 16 streams: 8 were designated for fish and wildlife, 7 were upgraded to a fish and wildlife classification, 1 was designated as agricultural and industrial water supply. Proposed streams classification rulemaking for 7 streams withdrawn.
7.	North Carolina	4/1/80	Final	45 FR 21246	Nullified a zero dissolved oxygen standard variance in a segment of Welch Creek and reestablished the State's previous standard of 5 mg/l average, 4 mg/l minimum, except for lower concentrations caused by natural swamp conditions.
8.	Ohio	11/28/80	Final	45 FR 79053	(1) Established water use designation, (2) establish a DO criterion of 5 mg/l for warmwater use, (3) designated 17 streams as warmwater habitat, (4) placed 111 streams downgraded by Ohio into modified warmwater habitat, (5) revised certain provisions relating to mixing zones (principally on Lake Erie), (6) revised low flow and other exceptions to standards, (7) amended sampling and analytical protocols, and (8) withdrew EPA proposal to establish a new cyanide criterion.
9.	Kentucky	12/9/80	Final (withdrawal)	45 FR 81042	Withdrew the Federal promulgation action of 12/2/74 after adoption of ppropriate water quality standards by the State.

10. North Carolina	11/10/81	Final (withdrawal)	46 FR 55520	Withdrew the Federal promulgation action of 4/1/80 following State adoption of a dissolved oxygen criterion consistent with the Federally promulgated standard.
11. Ohio	2/16/82	Final (withdrawal)	47 FR 29541	Withdrew Federal promulgation of 11/28/80 because it was based on a portion of the water quality standards regulation that has been determined to be invalid.
12. Nebraska	7/26/82	Final (withdrawal)	47 FR 32128	Withdrew Federal promulgation action of 6/6/78 after adoption of appropriate water quality standards by the State.
13. Alabama	11/26/82	Final (withdrawal)	47 FR 53372	Withdrew the Federal promulgation action of 2/14/80 following State adoption of requirements consistent with the Federally promulgated standard.
14. Idaho	8/20/85	Proposed	50 FR 33672	Proposal to replace DO criterion downstream from dams, partially replace Statewide ammonia criterion, replace ammonia criterion for Indian Creek, and delete categorical exemption of dams from Antidegradation Policy.
15. Mississippi	4/4/86	Final (withdrawal)	51 FR 11581	Withdrew the Federal promulgation of 4/30/79 following State adoption of requirements consistent with the Federally promulgated standard.
16. Idaho	7/14/86	Final (withdrawal)	51 FR 25372	Withdrew portions of proposed rule to replace DO criterion downstream from dams and delete categorical exemptions of dams from antidegradation rule since State adopted acceptable standards in both instances.
17. Kentucky	3/20/87	Final	50 FR 9102	Established a chloride criterion of 600 mg/l as a 30-day average, not to exceed a maximum of 1,200 mg/l at any time.

18.	Idaho	7/25/88	Final (withdrawal)	53 FR 27882	Withdrew portion of proposed rule which would have established a Statewide ammonia criterion and a site-specific ammonia criterion applicable to lower Indian Creek since State adopted acceptable standards.
19*.	Coleville Indian Reservation	7/6/89	Final	54 FR 28622	Established designated uses and criteria for all surface waters on the Reservation.
20.	Kentucky	4/3/91	Final (withdrawal)	56 FR 13592	Withdrew the Federal promulgation of 3/20/87 after adoption of appropriate WQS by the State.
21*.	12 States 2 Territories	12/22/92	Final	57 FR 60848	Established numeric water quality for toxic pollutants (aquatic life and human health).
22.	Washington	7/6/93	Final (withdrawal)	58 FR 36141	Withdrew, in part, the Federal Promulgation of 12/22/92 after adoption of appropriate criteria by the State.

* Final federal rule remains in force

SUMMARY OF FEDERAL PROMULGATION ACTIONS

Total Number of Proposed or Final Rules	22
Final Standards Promulgated	10
Withdrawal of Final Standards	8
Federal Rules Remaining In Force	3
No Action Taken on Proposals or Proposal Withdrawn	3

APPENDIX C

Biological Criteria:
National Program Guidance
for Surface Waters

APPENDIX C

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

United States
Environmental Protection
Agency

Office of Water
Regulations and Standards (WH-585)
Washington, DC 20460

EPA-440/5-90-004
April 1990

EPA

Biological Criteria

National Program Guidance For Surface Waters

Printed on Recycled Paper

Biological Criteria

National Program Guidance for Surface Waters

Criteria and Standards Division
Office of Water Regulations and Standards
U. S. Environmental Protection Agency
401 M Street S.W.
Washington D.C. 20460

Contents

Part II: The Implementation Process

Acknowledgments

Development of this document required the combined effort of ecologists, biologists, and policy makers from States, EPA Regions, and EPA Headquarters. Initial efforts relied on the 1988 document Report of the National Workshop on Instream Biological Monitoring and Criteria that summarizes a 1987 workshop sponsored by the EPA Office of Water Regulations and Standards, EPA Region V, and EPA Environmental Research Laboratory–Corvallis. In December 1988, contributing and reviewing committees were established (see Appendix D). Members provided reference materials and commented on drafts. Their assistance was most valuable.

Special recognition goes to the Steering Committee who helped develop document goals and made a significant contribution toward the final guidance. Members of the Steering Committee include:

Robert Hughes, Ph.D.	*Chris Yoder*
Susan Davies	*Wayne Davis*
John Maxted	*Jimmie Overton*
James Plafkin, Ph.D.	*Dave Courtemanch*
Phil Larsen, Ph.D.	

Finally, our thanks go to States that recognized the importance of a biological approach in standards and pushed forward independently to incorporate biological criteria into their programs. Their guidance made this effort possible. Development of the program guidance document was sponsored by the U.S. EPA Office of Water Regulations and Standards and developed, in part, through U.S. EPA Contract No. 68-03-3533 to Dynamac Corporation. Thanks to Dr. Mark Southerland for his technical assistance.

Suzanne K. Macy Marcy, Ph.D.
Editor

In Memory of

James L. Plafkin, Ph.D.

Definitions

To effectively use biological criteria, a clear understanding of how these criteria are developed and applied in a water quality standards framework is necessary. This requires, in part, that users of biological criteria start from the same frame of reference. To help form this frame of reference, the following definitions are provided. Please consider them carefully to ensure a consistent interpretation of this document.

Definitions

- An **AQUATIC COMMUNITY** is an association of interacting populations of aquatic organisms in a given waterbody or habitat

- A **BIOLOGICAL ASSESSMENT** is an evaluation of the biological condition of a waterbody using biological surveys and other direct measurements of resident biota in surface waters.

- **BIOLOGICAL CRITERIA,** or biocriteria, are numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use.

- **BIOLOGICAL INTEGRITY** is functionally defined as the condition of the aquatic community inhabiting unimpaired waterbodies of a specified habitat as measured by community structure and function.

- **BIOLOGICAL MONITORING** is the use of a biological entity as a detector and its response as a measure to determine environmental conditions. Toxicity tests and biological surveys are common biomonitoring methods.

- A **BIOLOGICAL SURVEY,** or biosurvey, consists of collecting, processing and analyzing representative portions of a resident aquatic community to determine the community structure and function.

- A **COMMUNITY COMPONENT** is any portion of a biological community. The community component may pertain to the taxomonic group (fish, invertebrates, algae), the taxonomic category (phylum, order, family, genus, species), the feeding strategy (herbivore, omnivore, carnivore) or organizational level (individual, population, community association) of a biological entity within the aquatic community.

- **REGIONS OF ECOLOGICAL SIMILARITY** describe a relatively homogeneous area defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variable. Regions of ecological similarity help define the potential for designated use classifications of specific waterbodies.

- **DESIGNATED USES** are those uses specified in water quality standards for each waterbody or segment whether or not they are being attained.

- An **IMPACT** is a change in the chemical, physical or biological quality or condition of a waterbody caused by external sources.

- An **IMPAIRMENT** is a detrimental effect on the biological integrity of a waterbody caused by an impact that prevents attainment of the designated use.

- A **POPULATION** is an aggregate of interbreeding individuals of a biological species within a specified location.

- A **WATER QUALITY ASSESSMENT** is an evaluation of the condition of a waterbody using biological surveys, chemical-specific analyses of pollutants in waterbodies, and toxicity tests.

- An **ECOLOGICAL ASSESSMENT** is an evaluation of the condition of a waterbody using water quality and physical habitat assessment methods.

Executive Summary

The Clean Water Act (Act) directs the U.S. Environmental Protection Agency (EPA) to develop programs that will evaluate, restore and maintain the chemical, physical, and biological integrity of the Nation's waters. In response to this directive, States and EPA implemented chemically based water quality programs that successfully addressed significant water pollution problems. However, these programs alone cannot identify or address all surface water pollution problems. To create a more comprehensive program, EPA is setting a new priority for the development of biological water quality criteria. The initial phase of this program directs State adoption of narrative biological criteria as part of State water quality standards. This effort will help States and EPA achieve the objectives of the Clean Water Act set forth in Section 101 and comply with statutory requirements under Sections 303 and 304. The *Water Quality Standards Regulation* provides additional authority for biological criteria development.

In accordance with priorities established in the *FY 1991 Agency Operating Guidance*, States are to adopt narrative biological criteria into State water quality standards during the FY 1991-1993 triennium. To support this priority, EPA is developing a *Policy on the Use of Biological Assessments and Criteria in the Water Quality Program* and is providing this program guidance document on biological criteria.

This document provides guidance for development and implementation of narrative biological criteria. Future guidance documents will provide additional technical information to facilitate development and implementation of narrative and numeric criteria for each of the surface water types.

When implemented, biological criteria will expand and improve water quality standards programs, help identify impairment of beneficial uses, and help set program priorities. Biological criteria are valuable because they directly measure the condition of the resource at risk, detect problems that other methods may miss or underestimate, and provide a systematic process for measuring progress resulting from the implementation of water quality programs.

Biological criteria require direct measurements of the structure and function of resident aquatic communities to determine biological integrity and ecological function. They supplement, rather than replace chemical and toxicological methods. It is EPA's policy that biological survey methods be fully integrated with toxicity and chemical-specific assessment methods and that chemical-specific criteria, whole-effluent toxicity evaluations and biological criteria be used as independent evaluations of non-attainment of designated uses.

Biological criteria are narrative expressions or numerical values that describe the biological integrity of aquatic communities inhabiting waters of a given aquatic life use. They are developed under the assumptions that surface waters impacted by antnropogenic activities may contain impaired aquatic communities (the greater the impact the greater the expected impairment) and that surface waters not impacted by anthropogenic activities are generally not impaired. Measures of aquatic community structure and function in unimpaired surface waters functionally define biological integrity and form the basis for establishing the biological criteria.

Narrative biological criteria are definable statements of condition or attainable goals for a given use designation. They establish a positive statement about aquatic community characteristics expected to occur within a waterbody (e.g., "Aquatic life shall be as it naturally occurs" or "A natural variety of aquatic life shall be present and all functional groups well represented"). These criteria can be developed using existing information. Numeric criteria describe the expected attainable community attributes and establish values based on measures such as species richness, presence or absence of indicator taxa, and distribution of classes of organisms. To implement narrative criteria and develop numeric criteria, biota in reference waters must be carefully assessed. These are used as the reference values to determine if, and to what extent, an impacted surface waterbody is impaired.

Biological criteria support designated aquatic life use classifications for application in standards. The designated use determines the benefit or purpose to be derived from the waterbody; the criteria provide a measure to determine if the use is impaired. Refinement of State water quality standards to include more detailed language about aquatic life is essential to fully implement a biological criteria program. Data collected from biosurveys can identify consistently distinct characteristics among aquatic communities inhabiting different waters with the same designated use. These biological and ecological characteristics may be used to define separate categories within a designated use, or separate one designated use into two or more use classifications.

To develop values for biological criteria, States should (1) identify unimpaired reference waterbodies to establish the reference condition and (2) characterize the aquatic communities inhabiting reference surface waters. Currently, two principal approaches are used to establish reference sites: (1) the site-specific approach, which may require upstream-downstream or near field-far field evaluations, and (2) the regional approach, which identifies similarities in the physico-chemical characteristics of watersheds that influence aquatic ecology. The basis for choosing reference sites depends on classifying the habitat type and locating unimpaired (minimally impacted) waters.

Once reference sites are selected, their biological integrity must be evaluated using quantifiable biological surveys. The success of the survey will depend in part on the careful selection of aquatic community components (e.g., fish, macroinvertebrates, algae). These components should serve as effective indicators of high biological integrity, represent a range of pollution tolerances, provide predictable, repeatable results, and be readily identified by trained State personnel. Well-planned quality assurance protocols are required to reduce variability in data collection and to assess the natural variability inherent in aquatic communities. A quality survey will include multiple community components and may be measured using a variety of metrics. Since multiple approaches are available, factors to consider when choosing possible approaches for assessing biological integrity are presented in this document and will be further developed in future technical guidance documents.

To apply biological criteria in a water quality standards program, standardized sampling methods and statistical protocols must be used. These procedures must be sensitive enough to identify significant differences between established criteria and tested communities. There are three possible outcomes from hypothesis testing using these analyses: (1) the use is impaired, (2) the biological criteria are met, or (3) the outcome is indeterminate. If the use is impaired, efforts to diagnose the cause(s) will help determine appropriate action. If the use is not impaired, no action is required based on these analyses. The outcome will be indeterminate if the study design or evaluation was incomplete. In this case, States would need to re-evaluate their protocols.

If the designated use is impaired, diagnosis is the next step. During diagnostic evaluations three main impact categories must be considered: chemical, physical, and biological stress. Two questions are posed during initial diagnosis: (1) what are obvious potential causes of impairment, and (2) what possible causes do the biological data suggest? Obvious potential causes of impairment are often identified during normal field biological assessments. When an impaired use cannot be easily related to an obvious cause, the diagnostic process becomes investigative and iterative. Normally the diagnoses of biological impairments are relatively straightforward; States can use biological criteria to confirm impairment from a known source of impact.

There is considerable State interest in integrating biological assessments and criteria in water quality management programs. A minimum of 20 States now use some form of standardized biological assessments to determine the status of biota in State waters. Of these, 15 States are developing biological assessments for future criteria development. Five States use biological criteria to define aquatic life use classifications and to enforce water quality standards. Several States have established narrative biological criteria in their standards. One State has instituted numeric biological criteria.

Whether a State is just beginning to establish narrative biological criteria or is developing a fully integrated biological approach, the programmatic expansion from source control to resource management represents a natural progression in water quality programs. Implementation of biological criteria will provide new options for expanding the scope and application of ecological perspectives.

Part I

Program Elements

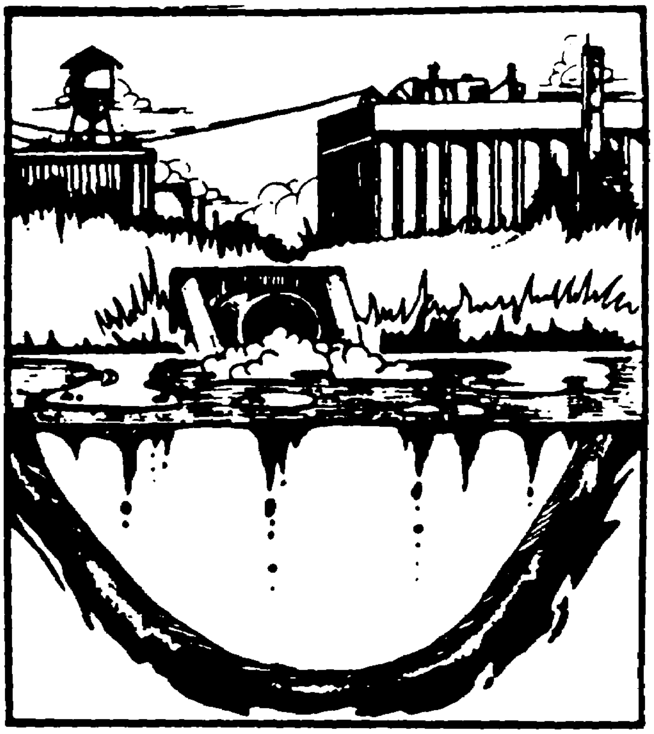
Chapter 1

Introduction

The principal objectives of the Clean Water Act are "to restore and maintain the chemical, physical and biological integrity of the Nation's waters" (Section 101). To achieve these objectives, EPA, States, the regulated community, and the public need comprehensive information about the ecological integrity of aquatic environments. Such information will help us identify waters requiring special protection and those that will benefit most from regulatory efforts.

To meet the objectives of the Act and to comply with statutory requirements under Sections 303 and 304, States are to adopt biological criteria in State standards. The *Water Quality Standards Regulation* provides additional authority for this effort. In accordance with the *FY 1991 Agency Operating Guidance*, States and qualified Indian tribes are to adopt narrative biological criteria into State water quality standards during the FY 1991-1993 triennium. To support this effort, EPA is developing a *Policy on the Use of Biological Assessments and Criteria in the Water Quality Program* and providing this program guidance document on biological criteria.

Like other water quality criteria, biological criteria identify water quality impairments, support regulatory controls that address water quality problems, and assess improvements in water quality from regulatory efforts. Biological criteria are numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use. They are developed through



Anthropogenic impacts, including point source discharges, nonpoint runoff, and habitat degradation continue to impair the nation's surface waters.

the direct measurement of aquatic community components inhabiting unimpaired surface waters.

Biological criteria complement current programs. Of the three objectives identified in the Act (chemical, physical, and biological integrity), current water quality programs focus on direct measures of chemical integrity (chemical-specific and whole-effluent toxicity) and, to some degree, physical integrity through several conventional criteria (e.g., pH, turbidity, dissolved oxygen). Implementation of these programs has significantly improved water quality. However, as we learn more about aquatic ecosystems it is apparent that other sources of waterbody impairment exist. Biological impairments from diffuse sources and habitat degradation can be greater than those caused by point source discharges (Judy et al. 1987; Miller et al. 1989). In Ohio, evaluation of instream biota indicated that 36 percent of impaired stream segments could not be detected using chemical criteria alone (see Fig. 1). Although effective for their purpose, chemical-specific criteria and whole-effluent toxicity provide only indirect evaluations and protection of biological integrity (see Table 1).

To effectively address our remaining water quality problems we need to develop more integrated and comprehensive evaluations. Chemical and physical integrity are necessary, but not sufficient conditions to attain biological integrity, and only when chemical, physical, and biological integrity are achieved, is ecological integrity possible (see Fig. 2). Biological criteria provide an essential third element for water quality management and serve as a natural progression in regulatory programs. Incorporating biological criteria into a fully integrated program directly protects the biological integrity of surface waters and provides indirect protection for chemical and physical integrity (see Table 2). Chemical-specific criteria, whole-effluent toxicity evaluations, and biological criteria, when used together, complement the relative strengths and weaknesses of each approach.

Figure 1.—Ohio Biosurvey Results Agree with Instream Chemistry or Reveal Unknown Problems

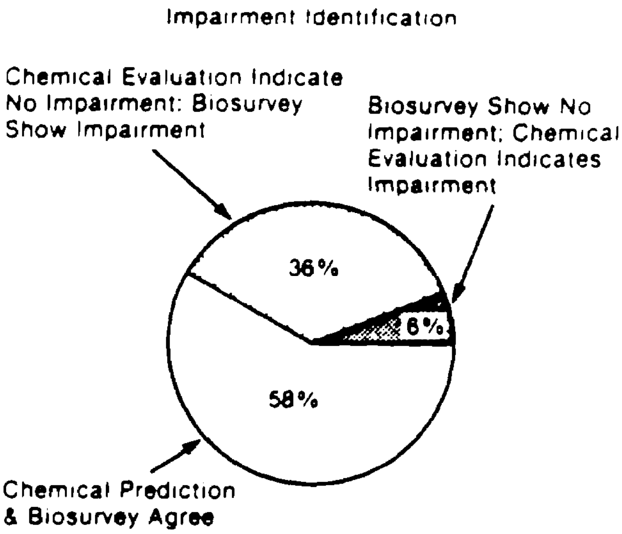


Fig. 1. In an intensive survey, 431 sites in Ohio were assessed using instream chemistry and biological surveys. In 36% of the cases, chemical evaluations implied no impairment but biological survey evaluations showed impairment. In 58% of the cases the chemical and biological assessments agreed. Of these, 17% identified waters with no impairment, 41% identified waters which were considered impaired. (Modified from Ohio EPA Water Quality Inventory, 1988.)

Biological assessments have been used in biomonitoring programs by States for many years. In this respect, biological criteria support earlier work. However, implementing biological criteria in water quality standards provides a systematic, structured, and objective process for making decisions about compliance with water quality standards. This distinguishes biological criteria from earlier use of biological information and increases the value of biological data in regulatory programs.

Table 1.—Current Water Quality Program Protection of the Three Elements of Ecological Integrity.

ELEMENTS OF ECOLOGICAL INTEGRITY	PROGRAM THAT DIRECTLY PROTECTS	PROGRAM THAT INDIRECTLY PROTECTS
Chemical Integrity	Chemical Specific Criteria (toxics) Whole Effluent Toxicity (toxics)	
Physical Integrity	Criteria for Conventionals (pH, DO, turbidity)	
Biological Integrity		Chemical Whole Effluent Toxicity (biotic response in lab)

Table 1. Current programs focus on chemical specific and whole-effluent toxicity evaluations. Both are valuable approaches for the direct evaluation and protection of chemical integrity. Physical integrity is also directly protected to a limited degree through criteria for conventional pollutants. Biological integrity is only indirectly protected under the assumption that by evaluating toxicity to organisms in laboratory studies estimates can be made about the toxicity to other organisms inhabiting ambient waters.

Table 2.—Water Quality Programs that Incorporate Biological Criteria to Protect Elements of Ecological Integrity.

ELEMENTS OF ECOLOGICAL INTEGRITY	DIRECTLY PROTECTS	INDIRECTLY PROTECTS
Chemical Integrity	Chemical Specific Criteria (toxics) Whole Effluent Toxicity (toxics)	Biocriteria (identification of impairment)
Physical Integrity	Criteria for conventionals (pH, temp., DO)	Biocriteria (habitat evaluation)
Biological Integrity	Biocriteria (biotic response in surface water)	Chemical/Whole Effluent Testing (biotic response in lab)

Table 2. When biological criteria are incorporated into water quality programs the biological integrity of surface waters may be directly evaluated and protected. Biological criteria also provide additional benefits by requiring an evaluation of physical integrity and providing a monitoring tool to assess the effectiveness of current chemically based criteria.

Figure 2.—The Elements of Ecological Integrity

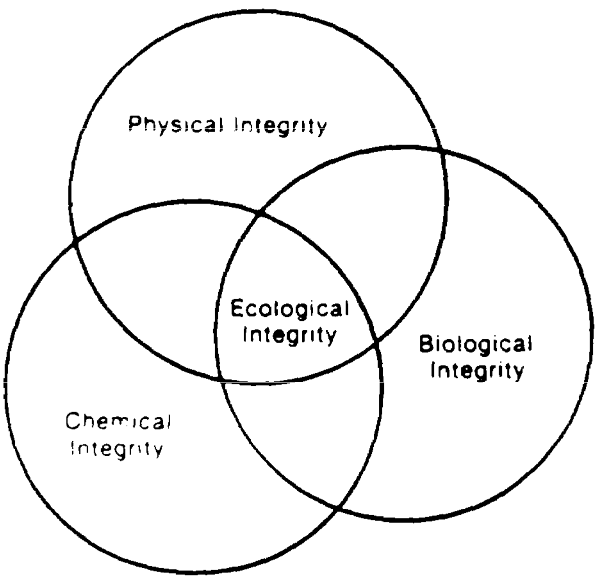


Fig. 2: Ecological Integrity is attainable when chemical, physical, and biological integrity occur simultaneously.

Value of Biological Criteria

Biological criteria provide an effective tool for addressing remaining water quality problems by directing regulatory efforts toward assessing the biological resources at risk from chemical, physical or biological impacts. A primary strength of biological criteria is the detection of water quality problems that other methods may miss or underestimate. Biological criteria can be used to determine to what extent current regulations are protecting the use.

Biological assessments provide integrated evaluations of water quality. They can identify impairments from contamination of the water column and sediments from unknown or unregulated chemicals, non-chemical impacts, and altered physical habitat. Resident biota function as continual monitors of environmental quality, increasing the likelihood of detecting the effects of episodic events (e.g., spills, dumping, treatment plant malfunctions, nutrient enrichment), toxic nonpoint source pollution (e.g., agricultural pesticides), cumulative pollution (i.e., multiple impacts over time or continuous low-level stress), or other impacts that periodic chemical sampling is unlikely to detect. Impacts on the physical habitat such as sedimentation from stormwater runoff and the effects of physical or structural habitat alterations (e.g., dredging, filling, channelization) can also be detected.

Biological criteria require the direct measure of resident aquatic community structure and function to determine biological integrity and ecological function. Using these measures, impairment can be detected and evaluated without knowing the impact(s) that may cause the impairment.

Biological criteria provide a regulatory framework for addressing water quality problems and offer additional benefits, including providing:

- the basis for characterizing high quality waters and identifying habitats and community components requiring special protection under State anti-degradation policies;
- a framework for deciding 319 actions for best control of nonpoint source pollution;
- an evaluation of surface water impairments predicted by chemical analyses, toxicity

testing, and fate and transport modeling (e.g., wasteload allocation);

- improvements in water quality standards (including refinement of use classifications);
- a process for demonstrating improvements in water quality after implementation of pollution controls;
- additional diagnostic tools.

The role of biological criteria as a regulatory tool is being realized in some States (e.g., Arkansas, Maine, Ohio, North Carolina, Vermont). Biological assessments and criteria have been useful for regulatory, resource protection, and monitoring and reporting programs. By incorporating biological criteria in programs, States can improve standards setting and enforcement, measure impairments from permit violations, and refine wasteload allocation models. In addition, the location, extent, and type of biological impairments measured in a waterbody provide valuable information needed for identifying the cause of impairment and determining actions required to improve water quality. Biological assessment and criteria programs provide a cost-effective method for evaluating water quality when a standardized, systematic approach to study design, field methods, and data analysis is established (Ohio EPA 1988a).

Process for Implementation

The implementation of biological criteria will follow the same process used for current chemical-specific and whole-effluent toxicity applications: national guidance produced by U.S. EPA will support States working to establish State standards for the implementation of regulatory programs (see Table 3). Biological criteria differ, however, in the degree of State involvement required. Because surface waters vary significantly from region to region, EPA will provide guidance on acceptable approaches for biological criteria development rather than specific criteria with numerical limitations. States are to establish assessment procedures, conduct field evaluations, and determine criteria values to implement biological criteria in State standards and apply them in regulatory programs.

The degree of State involvement required influences how biological criteria will be implemented. It is expected that States will implement these criteria in phases.

- **Phase I** includes the development and adoption of narrative biological criteria into State standards for all surface waters (streams, rivers, lakes, wetlands, estuaries). Definitions of terms and expressions in the narratives must be included in these standards (see the Narrative Criteria Section, Chapter 3). Adoption of narrative biological criteria in State standards provides the legal and programmatic basis for using ambient biological surveys and assessments in regulatory actions
- **Phase II** includes the development of an implementation plan. The plan should include program objectives, study design, research protocols, criteria for selecting reference conditions and community components, quality assurance and quality control procedures,

Table 3.—Process for Implementation of Water Quality Standards.

CRITERIA	EPA GUIDANCE	STATE IMPLEMENTATION	STATE APPLICATION
Chemical Specific	Pollutant specific numeric criteria	State Standards · use designation · numeric criteria · antidegradation	Permit limits Monitoring Best Management Practices Wasteload allocation
Narrative Free Forms	Whole effluent toxicity guidance	Water Quality Narrative · no toxic amounts translator	Permit limits Monitoring Wasteload allocation Best Management Practices
Biological	Biosurvey minimum requirement guidance	State Standards · refined use · narrative/numeric criteria · antidegradation	Permit conditions Monitoring Best Management Practices Wasteload allocation

Table 3 Similar to chemical specific criteria and whole effluent toxicity evaluations. EPA is providing guidance to States for the adoption of biological criteria into State standards to regulate sources of water quality impairment

and training for State personnel. In Phase II, States are to develop plans necessary to implement biological criteria for each surface water type.

- **Phase III** requires full implementation and integration of biological criteria in water quality standards. This requires using biological surveys to derive biological criteria for classes of surface waters and designated uses. These criteria are then used to identify nonattainment of designated uses and make regulatory decisions.

Narrative biological criteria can be developed for all five surface water classifications with little or no data collection. Application of narrative criteria in seriously degraded waters is possible in the short term. However, because of the diversity of surface waters and the biota that inhabit these waters, significant planning, data collection, and evaluation will be needed to fully implement the program. Criteria for each type of surface water are likely to be developed at different rates. The order and rate of development will depend, in part, on the development of EPA guidance for specific types of surface water. Biological criteria technical guidance for streams will be produced during FY 1991. The tentative order for future technical guidance documents includes guidance for rivers (FY 1992), lakes (FY 1993), wetlands (FY 1994) and estuaries (FY 1995). This order and timeline for guidance does not reflect the relative importance of these surface waters, but rather indicates the relative availability of research and the anticipated difficulty of developing guidance.

Independent Application of Biological Criteria

Biological criteria supplement, but do not replace, chemical and toxicological methods. Water chemistry methods are necessary to predict risks (particularly to human health and wildlife), and to diagnose, model, and regulate important water quality problems. Because biological criteria are able to detect different types of water quality impairments and, in particular, have different levels of sensitivity for detecting certain types of impairment compared to toxicological methods, they are not used in lieu of, or in conflict with, current regulatory efforts.

As with all criteria, certain limitations to biological criteria make independent application essential. Study design and use influences how sensitive biological criteria are for detecting community impairment. Several factors influence sensitivity: (1) State decisions about what is significantly different between reference and test communities, (2) study design, which may include community components that are not sensitive to the impact causing impairment, (3) high natural variability that makes it difficult to detect real differences, and (4) types of impacts that may be detectable sooner by other methods (e.g., chemical criteria may provide earlier indications of impairment from a bioaccumulative chemical because aquatic communities require exposure over time to incur the full effect).

Since each type of criteria (biological criteria, chemical-specific criteria, or whole-effluent toxicity evaluations) has different sensitivities and purposes, a criterion may fail to detect real impairments when used alone. As a result, these methods should be used together in an integrated water quality assessment, each providing an independent evaluation of nonattainment of a designated use. If any one type of criteria indicates impairment of the surface water, regulatory action can be taken to improve water quality. However, no one type of criteria can be used to confirm attainment of a use if another form of criteria indicates nonattainment (see Hypothesis Testing: Biological Criteria and the Scientific Method, Chapter 7). When these three methods are used together, they provide a powerful, integrated, and effective foundation for waterbody management and regulations.

How to Use this Document

The purpose of this document is to provide EPA Regions, States and others with the conceptual framework and assistance necessary to develop and implement narrative and numeric biological criteria and to promote national consistency in application. There are two main parts of the document. Part One (Chapters 1, 2, 3, and 4) includes the essential concepts about what biological criteria are and how they are used in regulatory programs. Part Two (Chapters 5, 6, and 7) provides an overview of the process that is essential for implementing a State biological criteria program. Specific chapters include the following:

Part I: PROGRAM ELEMENTS

- **Chapter 2, Legal Authority**, reviews the legal basis for biological criteria under the Clean Water Act and includes possible applications under the Act and other legislation.
- **Chapter 3, Conceptual Framework**, discusses the essential program elements for biological criteria, including what they are and how they are developed and used within a regulatory program. The development of narrative biological criteria is discussed in this chapter.
- **Chapter 4, Integration**, discusses the use of biological criteria in regulatory programs.

Part II: THE IMPLEMENTATION PROCESS

- **Chapter 5, The Reference Condition**, provides a discussion on alternative forms of reference conditions that may be developed by a State based on circumstances and needs.
- **Chapter 6, The Biological Survey**, provides some detail on the elements of a quality biological survey.
- **Chapter 7, Hypothesis Testing: Biological Criteria and the Scientific Method**, discusses how biological surveys are used to make regulatory and diagnostic decisions.
- **Appendix A** includes commonly asked questions and their answers about biological criteria.

Two additional documents are planned in the near term to supplement this program guidance document.

1. "*Biological Criteria Technical Reference Guide*" will contain a cross reference of technical papers on available approaches and methods for developing biological criteria (see tentative table of contents in Appendix B),
2. "*Biological Criteria Development by States*" will provide a summary of different mechanisms several States have used to implement and apply biological criteria in water quality programs (see tentative outline in Appendix C).

Both documents are planned for FY 1991. As previously discussed, over the next triennium technical guidance for specific systems (e.g., streams, wetlands) will be developed to provide guidance on acceptable biological assessment procedures to further support State implementation of comprehensive programs.

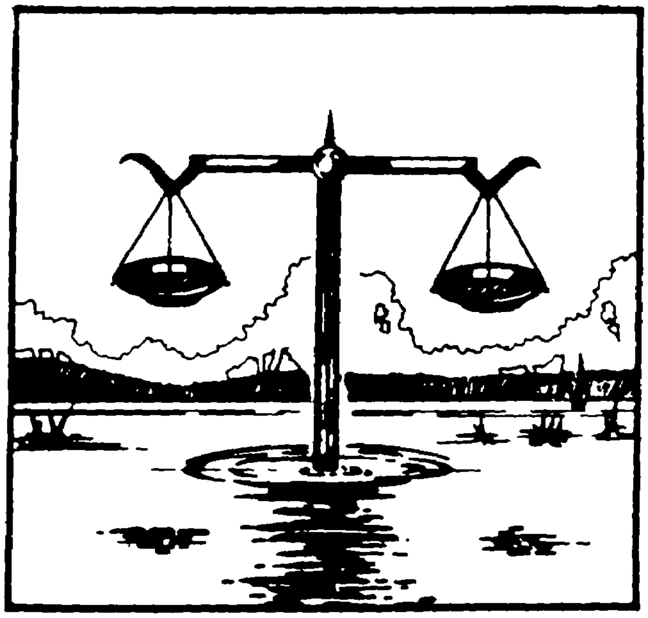
This biological criteria program guidance document supports development and implementation of biological criteria by providing guidance to States working to comply with requirements under the Clean Water Act and the Water Quality Standards Regulation. This guidance is not regulatory.

Chapter 2

Legal Authority

The Clean Water Act (Federal Water Pollution Control Act of 1972, Clean Water Act of 1977, and the Water Quality Act of 1987) mandates State development of criteria based on biological assessments of natural ecosystems.

The general authority for biological criteria comes from Section 101(a) of the Act which establishes as the objective of the Act the restoration and maintenance of the chemical, physical, and biological integrity of the Nation's waters. To meet this objective, water quality criteria must include criteria to protect biological integrity. Section 101(a)(2) includes the interim water quality goal for the protection and propagation of fish, shellfish, and wildlife. Propagation includes the full range of biological conditions necessary to support reproducing populations of all forms of aquatic life and other life that depend on aquatic systems. Sections 303 and 304 provide specific directives for the development of biological criteria.



Balancing the legal authority for biological criteria.

Section 303

Under Section 303(c) of the Act, States are required to adopt protective water quality standards that consist of uses, criteria, and antidegradation. States are to review these standards every three years and to revise them as needed.

Section 303(c)(2)(A) requires the adoption of water quality standards that "... serve the purposes of the Act," as given in Section 101. Section 303(c)(2)(B), enacted in 1987, requires States to adopt numeric criteria for toxic pollutants for which EPA has published 304(a)(1) criteria. The section further requires that, where numeric 304(a) criteria are not available, States should adopt criteria based on biological assessment and monitoring methods, consistent with information published by EPA under 304(a)(8).

These specific directives do not serve to restrict the use of biological criteria in other settings where they may be helpful. Accordingly, this guidance document provides assistance in implementing various sections of the Act, not just 303(c)(2)(B).

Section 304

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

- the effects of pollutants on aquatic community components (". . . plankton, fish, shellfish, wildlife, plant life . . .") and community attributes (". . . biological community diversity, productivity, and stability . . ."); in any body of water and;
- factors necessary ". . . to restore and maintain the chemical, physical, and biological integrity of all navigable waters . . ." for ". . . the protection of shellfish, fish, and wildlife for classes and categories of receiving waters . . ."

Potential Applications Under the Act

Development and use of biological criteria will help States to meet other requirements of the Act, including:

- setting planning and management priorities for waterbodies most in need of controls [Sec. 303(d)];
- determining impacts from nonpoint sources [i.e., Section 304(f) "(1) guidelines for identifying and evaluating the nature and extent of nonpoint sources of pollutants, and (2) processes, procedures, and methods to control pollution . . ."].
- biennial reports on the extent to which waters support balanced biological communities [Sec. 305(b)];
- assessment of lake trophic status and trends [Sec. 314];
- lists of waters that cannot attain designated uses without nonpoint source controls [Sec. 319];
- development of management plans and conducting monitoring in estuaries of national significance [Sec. 320];
- issuing permits for ocean discharges and monitoring ecological effects [Sec. 403(c) and 301(h)(3)];
- determination of acceptable sites for disposal of dredge and fill material [Sec. 404];

Potential Applications Under Other Legislation

Several legislative acts require an assessment of risk to the environment (including resident aquatic communities) to determine the need for regulatory action. Biological criteria can be used in this context to support EPA assessments under:

- *Toxic Substances Control Act (TSCA) of 1976*
- *Resource Conservation and Recovery Act (RCRA),*
- *Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA),*
- *Superfund Amendments and Reauthorization Act of 1986 (SARA),*
- *Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA);*
- *National Environmental Policy Act (NEPA);*
- *Federal Lands Policy and Management Act (FLPMA).*
- *The Fish and Wildlife Conservation Act of 1980*
- *Marine Protection, Research, and Sanctuaries Act*
- *Coastal Zone Management Act*

- ❑ *Wild and Scenic Rivers Act*
- ❑ *Fish and Wildlife Coordination Act, as Amended in 1965*

A summary of the applicability of these Acts for assessing ecological impairments may be found in *Risk Assessment Guidance for Superfund-Environmental Evaluation Manual (Interim Final) 1989.*

Other federal and State agencies can also benefit from using biological criteria to evaluate the biological integrity of surface waters within their jurisdiction and to the effects of specific practices on surface water quality. Agencies that could benefit include:

- ❑ **Department of the Interior** (*U.S. Fish and Wildlife Service, U.S. Geological Survey, Bureau of Mines, and Bureau of Reclamation, Bureau of Indian Affairs, Bureau of Land Management, and National Park Service*),
- ❑ **Department of Commerce** (*National Oceanic and Atmospheric Administration, National Marine Fisheries Service*),
- ❑ **Department of Transportation** (*Federal Highway Administration*)
- ❑ Department of Agriculture (*U.S. Forest Service, Soil Conservation Service*)
- ❑ **Department of Defense,**
- ❑ **Department of Energy,**
- ❑ **Army Corps of Engineers,**
- ❑ **Tennessee Valley Authority.**

Chapter 3

The Conceptual Framework

Biological integrity and the determination of use impairment through assessment of ambient biological communities form the foundation for biological criteria development. The effectiveness of a biological criteria program will depend on the development of quality criteria, the refinement of use classes to support narrative criteria, and careful application of scientific principles.

Aquatic communities assessed in unimpaired waterbodies (top) provide a reference for evaluating impairments in the same or similar waterbodies suffering from increasing anthropogenic impacts (bottom)

Premise for Biological Criteria

Biological criteria are based on the premise that the structure and function of an aquatic biological community within a specific habitat provide critical information about the quality of surface waters. Existing aquatic communities in pristine environments not subject to anthropogenic impact exemplify biological integrity and serve as the best possible goal for water quality. Although pristine environments are virtually non-existent (even remote waters are impacted by air pollution), minimally impacted waters exist. Measures of the structure and function of aquatic communities inhabiting unimpaired (minimally impacted) waters provide the basis for establishing a reference condition that may be compared to the condition of impacted surface waters to determine impairment.

Based on this premise, biological criteria are developed under the assumptions that: (1) surface waters subject to anthropogenic disturbance may contain impaired populations or communities of aquatic organisms—the greater the anthropogenic disturbance, the greater the likelihood and magnitude of impairment; and (2) surface waters not subject to anthropogenic disturbance generally contain unimpaired (natural) populations and communities of aquatic organisms exhibiting biological integrity.

Biological Integrity

The expression "biological integrity" is used in the Clean Water Act to define the Nation's objectives for water quality. According to Webster's New World Dictionary (1966), integrity is, "the quality or state of being complete; unimpaired." Biological integrity has been defined as "the ability of an aquatic ecosystem to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region" (Karr and Dudley 1981). For the purposes of biological criteria, these concepts are combined to develop a functional definition for evaluating biological integrity in water quality programs. Thus, biological integrity is functionally defined as:

> *the condition of the aquatic community inhabiting the unimpaired waterbodies of a specified habitat as measured by community structure and function.*

It will often be difficult to find unimpaired waters to define biological integrity and establish the reference condition. However, the structure and function of aquatic communities of high quality waters can be approximated in several ways. One is to characterize aquatic communities in the most protected waters representative of the regions where such sites exist. In areas where few or no unimpaired sites are available, characterization of least impaired systems approximates unimpaired systems. Concurrent analysis of historical records should supplement descriptions of the condition of least impaired systems. For some systems, such as lakes, evaluating paleoecological information (the record stored in sediment profiles) can provide a measure of less disturbed conditions.

Surface waters, when inhabited by aquatic communities, are exhibiting a degree of biological integrity. However, the best representation of biological integrity for a surface water should form the basis for establishing water quality goals for those waters. When tied to the development of biological criteria, the realities of limitations on biological integrity can be considered and incorporated into a progressive program to improve water quality.

Biological Criteria

Biological criteria are narrative expressions or numerical values that describe the biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use. While biological integrity describes the ultimate goal for water quality, biological criteria are based on aquatic community structure and function for waters within a variety of designated uses. Designated aquatic life uses serve as general statements of attained or attainable uses of State waters. Once established for a designated use, biological criteria are quantifiable values used to determine whether a use is impaired, and if so, the level of impairment. This is done by specifying what aquatic community structure and function should exist in waters of a given designated use, and then comparing this condition with the condition of a site under evaluation. If the existing aquatic community measures fail to meet the criteria, the use is considered impaired.

Since biological surveys used for biological criteria are capable of detecting water quality problems (use impairments) that may not be detected by chemical or toxicity testing, violation of biological criteria is sufficient cause for States to initiate regulatory action. Corroborating chemical and toxicity testing data are not required (though they may be desirable) as supporting evidence to sustain a determination of use impairment. However, a finding that biological criteria fail to indicate use impairment does not mean the use is automatically attained. Other evidence, such as violation of physical or chemical criteria, or results from toxicity tests, can also be used to identify impairment. Alternative forms of criteria provide independent assessments of nonattainment.

As stated above, biological criteria may be narrative statements or numerical values. States can establish general narrative biological criteria early in program development without conducting biological assessments. Once established in State standards, narrative biological criteria form the legal and programmatic basis for expanding biological assessment and biosurvey programs needed to implement narrative criteria and develop numeric biological criteria. Narrative biological criteria should become part of State regulations and standards.

Narrative Criteria

Narrative biological criteria are general statements of attainable or attained conditions of biological integrity and water quality for a given use designation. Although similar to the "free from" chemical water quality criteria, narrative biological criteria establish a positive statement about what should occur within a water body. Narrative criteria can take a number of forms but they must contain several attributes to support the goals of the Clean Water Act to provide for the protection and propagation of fish, shellfish, and wildlife. Thus, narrative criteria should include specific language about aquatic community characteristics that (1) must exist in a waterbody to meet a particular designated aquatic life use, and (2) are quantifiable. They must be written to protect the use. Supporting statements for the criteria should promote water quality to protect the most natural community possible for the designated use. Mechanisms should be established in the standard to address potentially conflicting multiple uses. Narratives should be written to protect the most sensitive use and support antidegradation.

Several States currently use narrative criteria. In Maine, for example, narrative criteria were established for four classes of water quality for streams and rivers (see Table 4). The classifications were based on the range of goals in the Act from "no discharge" to "protection and propagation of fish, shellfish, and wildlife" (Courtemanch and Davies 1987). Maine separated its "high quality water" into two categories, one that reflects the highest goal of the Act (no discharge, Class AA) and one that reflects high integrity but is minimally impacted by human activity (Class A). The statement "The aquatic life . . . shall be as naturally occurs" is a narrative biological criterion for both Class AA and A waters. Waters in Class B meet the use when the life stages of all indigenous aquatic species are supported and no detrimental changes occur in community composition (Maine DEP 1986). These criteria directly support refined designated aquatic life uses (see Section D, Refining Aquatic Life Use Classifications).

These narrative criteria are effective only if, as Maine has done, simple phrases such as "as naturally occurs" and "nondetrimental" are clearly operationally defined. Rules for sampling procedures and data analysis and interpretation should become part of the regulation or supporting documentation. Maine was able to develop these criteria and their supporting statements using avail-

Table 4.—Aquatic Life Classification Scheme for Maine's Rivers and Streams.

RIVERS AND STREAMS	MANAGEMENT PERSPECTIVE	LEVEL OF BIOLOGICAL INTEGRITY
Class AA	High quality water for preservation of recreational and ecological interests. No discharges of any kind permitted. No impoundment permitted.	Aquatic life shall be as naturally occurs.
Class A	High quality water with limited human interference. Discharges restricted to noncontact process water or highly treated wastewater of quality equal to or better than the receiving water. Impoundment permitted.	Aquatic life shall be as naturally occurs
Class B	Good quality water. Discharges of well treated effluents with ample dilution permitted.	Ambient water quality sufficient to support life stages of all indigenous aquatic species. Only nondetrimental changes in community composition may occur.
Class C	Lowest quality water. Requirements consistent with interim goals of the federal Water Quality Law (fishable and swimmable)	Ambient water quality sufficient to support the life stages of all indigenous fish species. Changes in species composition may occur but structure and function of the aquatic community must be maintained.

able data from water quality programs. To implement the criteria, aquatic life inhabiting unimpaired waters must be measured to quantify the criteria statement.

Narrative criteria can take more specific forms than illustrated in the Maine example. Narrative criteria may include specific classes and species of organisms that will occur in waters for a given designated use. To develop these narratives, field evaluations of reference conditions are necessary to identify biological community attributes that differ significantly between designated uses. For example in the Arkansas use class Typical Gulf Coastal Ecoregion (i.e., South Central Plains) the narrative criterion reads:

> *"Streams supporting diverse communities of indigenous or adapted species of fish and other forms of aquatic life. Fish communities are characterized by a limited proportion of sensitive species; sunfishes are distinctly dominant, followed by darters and minnows. The community may be generally characterized by the following fishes: Key Species—Redfin shiner, Spotted sucker, Yellow bullhead, Flier, Slough darter, Grass pickerel; Indicator Species—Pirate perch, Warmouth, Spotted sunfish, Dusky darter, Creek chubsucker, Banded pygmy sunfish (Arkansas DPCE 1988).*

In Connecticut, current designated uses are supported by narratives in the standard. For example, under Surface Water Classifications, Inland Surface Waters Class AA, the Designated Use is: "Existing or proposed drinking water supply; fish and wildlife habitat; recreational use; agricultural, industrial supply, and other purposes (recreation uses may be restricted)."

The supporting narratives include:

> *Benthic invertebrates which inhabit lotic waters: A wide variety of macroinvertebrate taxa should normally be present and all functional groups should normally be well represented . . . Water quality shall be sufficient to sustain a diverse macroinvertebrate community of indigenous species. Taxa within the Orders Plecoptera (stoneflies), Ephemeroptera (mayflies), Coleoptera (beetles), Tricoptera (caddisflies) should be well represented (Connecticut DEP 1987).*

For these narratives to be effective in a biological criteria program expressions such as "a wide variety" and "functional groups should normally be well represented" require quantifiable definitions that become part of the standard or supporting documentation. Many States may find such narratives in their standards already. If so, States should evaluate current language to determine if it meets the requirements of quantifiable narrative criteria that support refined aquatic life uses.

Narrative biological criteria are similar to the traditional narrative "free froms" by providing the legal basis for standards applications. A sixth "free from" could be incorporated into standards to help support narrative biological criteria such as "free from activities that would impair the aquatic community as it naturally occurs." Narrative biological criteria can be used immediately to address obvious existing problems.

Numeric Criteria

Numerical indices that serve as biological criteria should describe expected attainable community attributes for different designated uses. It is important to note that full implementation of narrative criteria will require similar data as that needed for developing numeric criteria. At this time, States may or may not choose to establish numeric criteria but may find it an effective tool for regulatory use.

To derive a numeric criterion, an aquatic community's structure and function is measured at reference sites and set as a reference condition. Examples of relative measures include similarity indices, coefficients of community loss, and comparisons of lists of dominant taxa. Measures of existing community structure such as species richness, presence or absence of indicator taxa, and distribution of trophic feeding groups are useful for establishing the normal range of community components to be expected in unimpaired systems. For example, Ohio uses criteria for the warmwater habitat use class based on multiple measures in different reference sites within the same ecoregion. Criteria are set as the 25th percentile of all biological index scores recorded at established reference sites within the ecoregion. Exceptional warmwater habitat index criteria are set at the 75th percentile (Ohio EPA 1988a). Applications such as this require an extensive data base and multiple reference sites for each criteria value.

To develop numeric biological criteria, careful assessments of biota in reference sites must be conducted (Hughes et al. 1986). There are numerous ways to assess community structure and function in surface waters. No single index or measure is universally recognized as free from bias. It is important to evaluate the strengths and weaknesses of different assessment approaches. A multimetric approach that incorporates information on species richness, trophic composition, abundance or biomass, and organism condition is recommended. Evaluations that measure multiple components of communities are also recommended because they tend to be more reliable (e.g., measures of fish and macroinvertebrates combined will provide more information than measures of fish communities alone). The weaknesses of one measure or index can often be compensated by combining it with the strengths of other community measurements.

The particular indices used to develop numeric criteria depend on the type of surface waters (streams, rivers, lakes, Great Lakes, estuaries, wetlands, and nearshore marine) to which they must be applied. In general, community-level indices such as the Index of Biotic Integrity developed for midwestern streams (Karr et al. 1986) are more easily interpreted and less variable than fluctuating numbers such as population size. Future EPA technical guidance documents will include evaluations of the effectiveness of different biological survey and assessment approaches for measuring the biological integrity of surface water types and provide guidance on acceptable approaches for biological criteria development.

Refining Aquatic Life Use Classifications

State standards consist of (1) designated aquatic life uses, (2) criteria sufficient to protect the designated and existing use, and (3) an antidegradation clause. Biological criteria support designated aquatic life use classifications for application in State standards. Each State develops its own designated use classification system based on the generic uses cited in the Act (e.g., protection and propagation of fish, shellfish, and wildlife). Designated uses are intentionally general. However, States may develop subcategories within use designations to refine and clarify the use class. Clarification of the use class is particularly helpful when a variety of surface waters with distinct characteristics fit within the same use class, or do not fit well into any category. Determination of nonattainment in these waters may be difficult and open to alternative interpretations. If a determination is in dispute, regulatory actions will be difficult to accomplish. Emphasizing aquatic community structure within the designated use focuses the evaluation of attainment/nonattainment on the resource of concern under the Act.

Flexibility inherent in the State process for designating uses allows the development of subcategories of uses within the Act's general categories. For example, subcategories of aquatic life uses may be on the basis of attainable habitat (e.g., cold versus warmwater habitat); innate differences in community structure and function, (e.g., high versus low species richness or productivity); or fundamental differences in important community components (e.g., warmwater fish communities dominated by bass versus catfish). Special uses may also be designated to protect particularly unique, sensitive, or valuable aquatic species, communities, or habitats.

Refinement of use classes can be accomplished within current State use classification structures. Data collected from biosurveys as part of a developing biocriteria program may reveal unique and consistent differences among aquatic communities inhabiting different waters with the same designated use. Measurable biological attributes could then be used to separate one class into two or more classes. The result is a refined aquatic life use. For example, in Arkansas the beneficial use Fisheries "provides for the protection and propagation of fish, shellfish, and other forms of aquatic life" (Arkansas DPCE 1988). This use is subdivided into Trout, Lakes and Reservoirs, and Streams. Recognizing that stream characteristics across regions of the State differed ecologically, the State further subdivided the stream designated uses into eight additional uses based on regional characteristics (e.g. Springwater-influenced Gulf Coastal Ecoregion Ouachita Mountains Ecoregion). Within this classification system, it was relatively straightforward for

Arkansas to establish detailed narrative biological criteria that list aquatic community components expected in each ecoregion (see Narrative Criteria section). These narrative criteria can then be used to establish whether the use is impaired.

States can refine very general designated uses such as high, medium, and low quality to specific categories that include measurable ecological characteristics. In Maine, for example, Class AA waters are defined as "the highest classification and shall be applied to waters which are outstanding natural resources and which should be preserved because of their ecological, social, scenic, or recreational importance." The designated use includes 'Class AA waters shall be of such quality that they are suitable . . . as habitat for fish and other aquatic life. The habitat shall be characterized as free flowing and natural." This use supports development of narrative criteria based on biological characteristics of aquatic communities (Maine DEP 1986; see the Narrative Criteria section).

Biological criteria that include lists of dominant or typical species expected to live in the surface water are particularly effective. Descriptions of impaired conditions are more difficult to interpret. However, biological criteria may contain statements concerning which species dominate disturbed sites, as well as those species expected at minimally impacted sites.

Most States collect biological data in current programs. Refining aquatic life use classifications and incorporating biological criteria into standards will enable States to evaluate these data more effectively.

Developing and Implementing Biological Criteria

Biological criteria development and implementation in standards require an understanding of the selection and evaluation of reference sites, measurement of aquatic community structure and function, and hypothesis testing under the scientific method. The developmental process is important for State water quality managers and their staff to understand to promote effective planning for resource and staff needs. This major program element deserves careful consideration and has been separated out in Part II by chapter for each developmental step as noted below. Additional guidance will be provided in future technical guidance documents.

The developmental process is illustrated in Figure 3. The first step is establishing narrative criteria in standards. However, to support these narratives, standardized protocols need to be developed to quanitify the narratives for criteria implementation. They should include data collection procedures, selection of reference sites, quality assurance and quality control procedures, hypothesis testing, and statistical protocols. Pilot studies should be conducted using these standard protocols to ensure they meet the needs of the program, test the hypotheses, and provide effective measures of the biological integrity of surface waters in the State.

Figure 3.—Process for the Development and Implementation of Biological Criteria

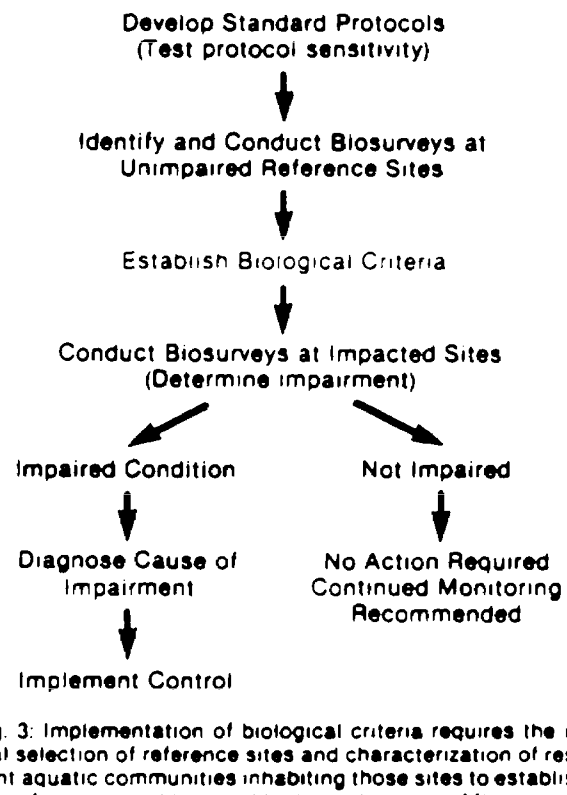


Fig. 3: Implementation of biological criteria requires the initial selection of reference sites and characterization of resident aquatic communities inhabiting those sites to establish the reference condition and biological criteria. After criteria development, impacted sites are evaluated using the same biosurvey procedures to assess resident biota. If impairment is found, diagnosis of cause will lead to the implementation of a control. Continued monitoring should accompany control implementation to determine the effectiveness of intervention. Monitoring is also recommended where no impairment is found to ensure that the surface water maintains or improves in quality.

The next step is establishing the reference condition for the surface water being tested. This reference may be site specific or regional but must establish the unimpaired baseline for comparison (see Chapter 5, The Reference Condition). Once reference sites are selected, the biological integrity of the site must be evaluated using carefully chosen biological surveys. A quality biological survey will include multiple community components and may be measured using a variety of metrics (see Chapter 6, The Biological Survey). Establishing the reference condition and conducting biological surveys at the reference locations provide the necessary information for establishing the biological criteria.

To apply biological criteria, impacted surface waters with comparable habitat characteristics are evaluated using the same procedures as those used to establish the criteria. The biological survey must support standardized sampling methods and statistical protocols that are sensitive enough to identify biologically relevant differences between established criteria and the community under evaluation. Resulting data are compared through hypothesis testing to determine impairment (see Chapter 7, Hypothesis Testing).

When water quality impairments are detected using biological criteria, they can only be applied in a regulatory setting if the cause for impairment can be identified. Diagnosis is iterative and investigative (see Chapter 7, Diagnosis). States must then determine appropriate actions to implement controls. Monitoring should remain a part of the biological criteria program whether impairments are found or not. If an impairment exists, monitoring provides a mechanism to determine if the control effort (intervention) is resulting in improved water quality. If there is no impairment, monitoring ensures the water quality is maintained and documents any improvements. When improvements in water quality are detected through monitoring programs two actions are recommended. When reference condition waters improve, biological criteria values should be recalculated to reflect this higher level of integrity. When impaired surface waters improve, states should reclassify those waters to reflect a refined designated use with a higher level of biological integrity. This provides a mechanism for progressive water quality improvement.

Chapter 4

Integrating Biological Criteria Into Surface Water Management

Integrating biological criteria into existing water quality programs will help to assess use attainment/nonattainment, improve problem discovery in specific waterbodies, and characterize overall water resource condition within a region. Ideally, biological criteria function in an iterative manner. New biosurvey information can be used to refine use classes. Refined use classes will help support criteria development and improve the value of data collected in biosurveys.

To integrate biological criteria into water quality programs, states must carefully determine where and how data are collected to assess the biological integrity of surface waters.

Implementing Biological Criteria

As biological survey data are collected, these data will increasingly support current use of biomonitoring data to identify water quality problems, assess their severity, and set planning and management priorities for remediation. Monitoring data and biological criteria should be used at the outset to help make regulatory decisions, develop appropriate controls, and evaluate the effectiveness of controls once they are implemented.

The value of incorporating biological survey information in regulatory programs is illustrated by evaluations conducted by North Carolina. In response to amendments of the Federal Water Pollution Control Act requiring secondary effluent limits for all wastewater treatment plants, North Carolina became embroiled in a debate over whether meeting secondary effluent limits (at considerable cost) would result in better water quality. North Carolina chose to test the effectiveness of additional treatment by conducting seven chemical and biological surveys before and after facility upgrades (North

Carolina DNRCD 1984). Study results indicated that moderate to substantial in-stream improvements were observed at six of seven facilities. Biological surveys were used as an efficient, cost-effective monitoring tool for assessing in-stream improvements after facility modification. North Carolina has also conducted comparative studies of benthic macroinvertebrate surveys and chemical-specific and whole-effluent evaluations to assess sensitivities of these measures for detecting impairments (Eagleson et al. 1990).

Narrative biological criteria provide a scientific framework for evaluating biosurvey, bioassessment, and biomonitoring data collected in most States. Initial application of narrative biological criteria may require only an evaluation of current work. States can use available data to define variables for choosing reference sites, selecting appropriate biological surveys, and assessing the response of local biota to a variety of impacts. States should also consider the decision criteria that will be used for determining appropriate State action when impairment is found.

Recent efforts by several States to develop biological criteria for freshwater streams provide excellent examples for how biological criteria can be integrated into water quality programs. Some of this work is described in the *National Workshop on Instream Biological Monitoring and Criteria* proceedings which recommended that "the concept of biological sampling should be integrated into the full spectrum of State and Federal surface water programs" (U.S. EPA 1987b). States are actively developing biological assessment and criteria programs; several have programs in place.

Biological Criteria in State Programs

Biological criteria are used within water programs to refine use designations, establish criteria for determining use attainment/nonattainment, evaluate effectiveness of current water programs, and detect and characterize previously unknown impairments. Twenty States are currently using some form of standardized ambient biological assessments to determine the status of biota within State waters. Levels of effort vary from bioassessment studies to fully developed biological criteria programs.

Fifteen States are developing aspects of biological assessments that will support future development of biological criteria. Colorado, Illinois, Iowa, Kentucky, Massachusetts, Tennessee, and Virginia conduct biological monitoring to evaluate biological conditions, but are not developing biological criteria. Kansas is considering using a community metric for water resource assessment. Arizona is planning to refine ecoregions for the State. Delaware, Minnesota, Texas, and Wisconsin are developing sampling and evaluation methods to apply to future biological criteria programs. New York is proposing to use biological criteria for site-specific evaluations of water quality impairment. Nebraska and Vermont use informal biological criteria to support existing aquatic life narratives in their water quality standards and other regulations. Vermont recently passed a law requiring that biological criteria be used to regulate through permitting the indirect discharge of sanitary effluents.

Florida incorporated a specific biological criterion into State standards for invertebrate species diversity. Species diversity within a waterbody, as measured by a Shannon diversity index, may not fall below 75 percent of reference values. This criterion has been used in enforcement cases to obtain injunctions and monetary settlements. Florida's approach is very specific and limits alternative applications.

Four States—Arkansas, North Carolina, Maine, and Ohio—are currently using biological criteria to define aquatic life use classifications and enforce water quality standards. These states have made biological criteria an integral part of comprehensive water quality programs.

- **Arkansas** rewrote its aquatic life use classifications for each of the State's ecoregions. This has allowed many cities to design wastewater treatment plants to meet realistic attainable dissolved oxygen conditions as determined by the new criteria.

- **North Carolina** developed biological criteria to assess impairment to aquatic life uses written as narratives in the State water quality standards. Biological data and criteria are used extensively to identify waters of special concern or those with exceptional water quality. In addition to the High Quality Waters (HQW) and Outstanding Resource Waters (ORW) designations, Nutrient Sensitive Waters (NSW) at risk for eutrophication are assessed using biological criteria. Although specific biological measures are not in the regulations, strengthened use of biological monitoring data to assess water quality is being proposed for incorporation in North Carolina's water quality standards.

■ **Maine** has enacted a revised Water Quality Classification Law specifically designed to facilitate the use of biological assessments. Each of four water classes contains descriptive aquatic life conditions necessary to attain that class. Based on a statewide database of macroinvertebrate samples collected above and below outfalls, Maine is now developing a set of dichotomous keys that serve as the biological criteria. Maine's program is not expected to have a significant role in permitting, but will be used to assess the degree of protection afforded by effluent limitations.

■ **Ohio** has instituted the most extensive use of biological criteria for defining use classifications and assessing water quality. Biological criteria were developed for Ohio rivers and streams using an ecoregional reference site approach. Within each of the State's five ecoregions, criteria for three biological indices (two for fish communities and one for macroinvertebrates) were derived. Ohio successfully uses biological criteria to demonstrate attainment of aquatic life uses and discover previously unknown or unidentified environmental degradation (e.g., twice as many impaired waters were discovered using biological criteria and water chemistry together than were found using chemistry alone). The upgraded use designations based on biological criteria were upheld in Ohio courts and the Ohio EPA successfully proposed their biological criteria for inclusion in the State water quality standards regulations.

States and EPA have learned a great deal about the effectiveness of integrated biological assessments through the development of biological criteria for freshwater streams. This information is particularly valuable in providing guidance on developing biological criteria for other surface water types. As previously discussed, EPA plans to produce supporting technical guidance for biological criteria development in streams and other surface waters. Production of these guidance documents will be contingent on technical progress made on each surface water type by researchers in EPA, States and the academic community.

EPA will also be developing outreach workshops to provide technical assistance to Regions and States working toward the implementation of biological criteria programs in State water quality management programs. In the interim, States should use the technical guidance currently available in the *Technical Support Manual(s): Waterbody Surveys and Assessments for Conducting Use Attainability Analysis* (U.S. EPA 1983b, 1984a,b).

During the next triennium, State effort will be focused on developing narrative biological criteria. Full implementation and integration of biological criteria will require several years. Using available guidance, States can complement the adoption of narrative criteria by developing implementation plans that include:

1. Defining program objectives, developing research protocols, and setting priorities;
2. Determining the process for establishing reference conditions, which includes developing a process to evaluate habitat characteristics;
3. Establishing biological survey protocols that include justifications for surface water classifications and selected aquatic community components to be evaluated; and
4. Developing a formal document describing the research design, quality assurance and quality control protocols, and required training for staff.

Whether a State begins with narrative biological criteria or moves to fully implement numeric criteria, the shift of the water quality program focus from source control to resource management represents a natural progression in the evolution from the technology-based to water quality-based approaches in water quality management. The addition of a biological perspective allows water quality programs to more directly address the objectives of the Clean Water Act and to place their efforts in a context that is more meaningful to the public.

Future Directions

Biological criteria now focus on resident aquatic communities in surface waters. They have the potential to expand in scope toward greater ecological integration. Ecological criteria may encompass the ambient aquatic communities in surface waters, wildlife species that use the same aquatic resources, and the aquatic community inhabiting the gravel and sediments underlying the surface waters and adjacent land (hyporheic zone); specific criteria may apply to physical habitat. These areas may represent only a few possible options for biological criteria in the future.

Many wildlife species depend on aquatic resources. If aquatic population levels decrease or if the distribution of species changes, food sources may be sufficiently altered to cause problems for wildlife species using aquatic resources. Habitat degradation that impairs aquatic species will often impact important wildlife habitat as well. These kinds of impairments are likely to be detected using biological criteria as currently formulated. In some cases, however, uptake of contaminants by resident aquatic organisms may not result in altered structure and function of the aquatic community. These impacts may go undetected by biological criteria, but could result in wildlife impairments because of bioaccumulation. Future expansion of biological criteria to include wildlife species that depend on aquatic resources could provide a more integrative ecosystem approach.

Rivers may have a subsurface flood plain extending as far as two kilometers from the river channel. Preliminary mass transport calculations made in the Flathead River basin in Montana indicate that nutrients discharged from this subsurface flood plain may be crucial to biotic productivity in the river channel (Stanford and Ward 1988). This is an unexplored dimension in the ecology of gravel river beds and potentially in other surface waters.

As discussed in Chapter 1, physical integrity is a necessary condition for biological integrity. Establishing the reference condition for biological criteria requires evaluation of habitat. The rapid bioassessment protocol provides a good example of the importance of habitat for interpreting biological assessments (Plafkin et al. 1989). However, it may be useful to more fully integrate habitat characteristics into the regulatory process by establishing criteria based on the necessary physical structure of habitats to support ecological integrity.

Part II

The Implementation Process

The implementation of biological criteria requires: (1) selection of unimpaired (minimal impact) surface waters to use as the reference condition for each designated use, (2) measurement of the structure and function of aquatic communities in reference surface waters to establish biological criteria, and (3) establishment of a protocol to compare the biological criteria to biota in impacted waters to determine whether impairment has occurred. These elements serve as an interactive network that is particularly important during early development of biological criteria where rapid accumulation of information is effective for refining both designated uses and developing biological criteria values. The following chapters describe these three essential elements.

Chapter 5

The Reference Condition

A key step in developing values for supporting narrative and creating numeric biological criteria is to establish reference conditions; it is an essential feature of environmental impact evaluations (Green 1979). Reference conditions are critical for environmental assessments because standard experimental controls are rarely available. For most surface waters, baseline data were not collected prior to an impact, thus impairment must be inferred from differences between the impact site and established references. Reference conditions describe the characteristics of waterbody segments least impaired by human activities and are used to define attainable biological or habitat conditions.

Reference conditions should be established by measuring resident biota in unimpaired surface waters.

Wide variability among natural surface waters across the country resulting from climatic, landform, and other geographic differences prevents the development of nationwide reference conditions. Most States are also too heterogeneous for single reference conditions. Thus, each State, and when appropriate, groups of States, will be responsible for selecting and evaluating reference waters within the State to establish biological criteria for a given surface water type or category of designated use. At least seven methods for estimating attainable conditions for streams have been identified (Hughes et al. 1986). Many of these can apply to other surface waters. References may be established by defining models of attainable conditions based on historical data or unimpaired habitat (e.g., streams in old growth forest). The reference condition established as before-after comparisons or concurrent measures of the reference water and impact sites can be based on empirical data (Hall et al. 1989).

Currently, two principal approaches are used for establishing the reference condition. A State may opt to (1) identify site-specific reference sites for each evaluation of impact or (2) select ecologically similar regional reference sites for comparison with impacted sites within the same region. Both approaches depend on evaluations of habitats to ensure that waters with similar habitats are compared. The designation of discrete habitat types is more fully developed for streams and rivers. Development of habitat types for lakes, wetlands, and estuaries is ongoing.

Site-Specific Reference Condition

A site-specific reference condition, frequently used to evaluate the impacts from a point discharge, is best for surface waters with a strong directional flow such as in streams and rivers (the upstream-downstream approach). However, it can also be used for other surface waters where gradients in contaminant concentration occur based on proximity to a source (the near field-far field approach). Establishment of a site-specific reference condition requires the availability of comparable habitat within the same waterbody in both the reference location and the impacted area.

A site-specific reference condition is difficult to establish if (1) diffuse nonpoint source pollution contaminates most of the water body; (2) modifications to the channel, shoreline, or bottom substrate are extensive; (3) point sources occur at multiple locations on the waterbody; or (4) habitat characteristics differ significantly between possible reference locations and the impact site (Hughes et al. 1986; Plafkin et al. 1989). In these cases, site-specific reference conditions could result in underestimates of impairment. Despite limitations, the use of site-specific reference conditions is often the method of choice for point source discharges and certain waterbodies, particularly when the relative impairments from different local impacts need to be determined.

The Upstream-Downstream Reference Condition

The upstream-downstream reference condition is best applied to streams and rivers where the habitat characteristics of the waterbody above the point of discharge are similar to the habitat characteristics of the stream below the point of discharge. One standard procedure is to characterize the biotic condition just above the discharge point (accounting for possible upstream circulation) to establish the reference condition. The condition below the discharge is also measured at several sites. If significant differences are found between these measures, impairment of the biota from the discharge is indicated. Since measurements of resident biota taken in any two sites are expected to differ because of natural variation, more than one biological assessment for both upstream and downstream sites is often needed to be confident in conclusions drawn from these data (Green, 1979). However, as more data are collected by a State, and particularly if regional characteristics of the waterbodies are incorporated, the basis for determining impairment from site-specific upstream-downstream assessments may require fewer individual samples. The same measures made below the "recovery zone" downstream from the discharge will help define where recovery occurs.

The upstream-downstream reference condition should be used with discretion since the reference condition may be impaired from impacts upstream from the point source of interest. In these cases it is important to discriminate between individual point source impact versus overall impairment of the system. When overall impairment occurs, the resident biota may be sufficiently impaired to make it impossible to detect the effect of the target point source discharger.

The approach can be cost effective when one biological assessment of the upstream reference condition adequately reflects the attainable condition of the impacted site. However, routine comparisons may require assessments of several upstream sites to adequately describe the natural variability of reference biota. Even so, measuring a series of site-specific references will likely continue to be the method of choice for certain point source discharges, especially where the relative impairments from different local impacts need to be determined.

The Near Field-Far Field Reference Condition

The near field-far field reference condition is effective for establishing a reference condition in surface waters other than rivers and streams and is particularly applicable for unique waterbodies (e.g., estuaries such as Puget Sound may not have comparable estuaries for comparison). To apply this method, two variables are measured (1) habitat characteristics, and (2) gradient of impairment. For reference waters to be identified within the same waterbody, sufficient size is necessary to separate the reference from the impact area so that a gradient of impact exists. At the same time, habitat characteristics must be comparable.

Although not fully developed, this approach may provide an effective way to establish biological criteria for estuaries, large lakes, or wetlands. For example, estuarine habitats could be defined and possible reference waters identified using physical and chemical variables like those selected by the Chesapeake Bay Program (U.S. EPA 1987a, e.g., substrate type, salinity, pH) to establish comparable subhabitats in an estuary. To determine those areas least impaired, a "mussel watch" program like that used in Narragansett Bay (i.e., captive mussels are used as indicators of contamination, (Phelps 1988)) could establish impairment gradients. These two measures, when combined, could form the basis for selecting specific habitat types in areas of least impairment to establish the reference condition.

Regional Reference Conditions

Some of the limitations of site-specific reference conditions can be overcome by using regional reference conditions that are based on the assumption that surface waters integrate the character of the land they drain. Waterbodies within the same watershed in the same region should be more similar to each other than to those within watersheds in different regions. Based on these assumptions, a distribution of aquatic regions can be developed based on ecological features that directly or indirectly relate to water quality and quantity, such as soil type, vegetation (land cover), land-surface form, climate, and land use. Maps that incorporate several of these features will provide a general purpose broad scale ecoregional framework (Gallant et al. 1989).

Regions of ecological similarity are based on hydrologic, climatic, geologic, or other relevant geographic variables that influence the nature of biota in surface waters. To establish a regional reference condition, surface waters of similar habitat type are identified in definable ecological regions. The biological integrity of these reference waters is determined to establish the reference condition and develop biological criteria. These criteria are then used to assess impacted surface waters in the same watershed or region. There are two forms of regional reference conditions: (1) paired watersheds and (2) ecoregions.

Paired Watershed Reference Conditions

Paired watershed reference conditions are established to evaluate impaired waterbodies, often impacted by multiple sources. When the majority of a waterbody is impaired, the upstream-downstream or near field-far field reference condition does not provide an adequate representation of the unimpaired condition of aquatic communities for the waterbody. Paired watershed reference conditions are established by identifying unimpaired surface waters within the same or very similar local watershed that is of comparable type and habitat. Variables to consider when selecting the watershed reference condition include absence of human disturbance, waterbody size and other physical characteristics, surrounding vegetation, and others as described in the "Regional Reference Site Selection" feature.

This method has been successfully applied (e.g., Hughes 1985) and is an approach used in Rapid Bioassessment Protocols (Plafkin et al. 1989). State use of this approach results in good reference conditions that can be used immediately in current programs. This approach has the added benefit of promoting the development of a database on high quality waters in the State that could form the foundation for establishing larger regional references (e.g., ecoregions.)

Ecoregional Reference Conditions

Reference conditions can also be developed on a larger scale. For these references, waterbodies of similar type are identified in regions of ecological similarity. To establish a regional reference condition, a set of surface waters of similar habitat type are identified in each ecological region. These sites must represent similar habitat type and be representative of the region. As with other reference conditions, the biological integrity of selected reference waters is determined to establish the reference. Biological criteria can then be developed and used to assess impacted surface waters in the same region. Before reference conditions may be established, regions of ecological similarity must be defined.

Regional Reference Site Selection

To determine specific regional reference sites for streams, candidate watersheds are selected from the appropriate maps and evaluated to determine if they are typical for the region. An evaluation of level of human disturbance is made and a number of relatively undisturbed reference sites are selected from the candidate sites. Generally, watersheds are chosen as regional reference sites when they fall entirely within typical areas of the region. Candidate sites are then selected by aerial and ground surveys. Identification of candidate sites is based on: (1) absence of human disturbance, (2) stream size, (3) type of stream channel, (4) location within a natural or political refuge, and (5) historical records of resident biota and possible migration barriers.

Final selection of reference sites depends on a determination of minimal disturbance derived from habitat evaluation made during site visits. For example, indicators of good quality streams in forested ecoregions include: (1) extensive, old, natural riparian vegetation; (2) relatively high heterogeneity in channel width and depth; (3) abundant large woody debris, coarse bottom substrate, or extensive aquatic or overhanging vegetation; (4) relatively high or constant discharge; (5) relatively clear waters with natural color and odor; (6) abundant diatom, insect, and fish assemblages; and (7) the presence of piscivorous birds and mammals.

One frequently used method is described by Omernik (1987) who combined maps of land-surface form, soil, potential natural vegetation, and land use within the conterminous United States to generate a map of aquatic ecoregions for the country. He also developed more detailed regional maps. The ecoregions defined by Omernik have been evaluated for streams and small rivers in Arkansas (Rohm et al. 1987), Ohio (Larsen et al. 1986; Whittier et al. 1987), Oregon (Whittier et al. 1988), Colorado (Gallant et al. 1989), and Wisconsin (Lyons 1989) and for lakes in Minnesota (Heiskary et al. 1987). State ecoregion maps were developed for Colorado (Gallant et al. 1989) and Oregon (Clarke et al. mss). Maps for the national ecoregions and six multi-state maps of more detailed ecoregions are available from the U.S. EPA Environmental Research Laboratory, Corvallis, Oregon.

Ecoregions such as those defined by Omernik (1987) provide only a first step in establishing regional reference sites for development of the reference condition. Field site evaluation is required to account for the inherent variability within each ecoregion. A general method for selecting reference sites for streams has been described (Hughes et al. 1986). These are the same variables used for comparable watershed reference site selection. Regional and on-site evaluations of biological factors help determine specific sites that best represent typical but unimpaired surface water habitats within the region. Details on this approach for streams is described in the "Regional Reference Site Selection" feature. To date, the regional approach has been tested on streams, rivers, and lakes. The method appears applicable for assessing other inland ecosystems. To apply this approach to wetlands and estuaries will require additional evaluation based on the relevant ecological features of these ecosystems (e.g. Brooks and Hughes, 1988).

Ideally, ecoregional reference sites should be as little disturbed as possible, yet represent waterbodies for which they are to serve as reference waters. These sites may serve as references for a large number of similar waterbodies (e.g., several reference streams may be used to define the reference condition for numerous physically separate streams if the reference streams contain the same range of stream morphology, substrate, and flow of the other streams within the same ecological region).

An important benefit of a regional reference system is the establishment of a baseline condition for the least impacted surface waters within the dominant land use pattern of the region. In many areas a return to pristine, or presettlement, conditions is impossible, and goals for waterbodies in extensively developed regions could reflect this. Regional reference sites based on the least impacted sites within a region will help water quality programs restore and protect the environment in a way that is ecologically feasible.

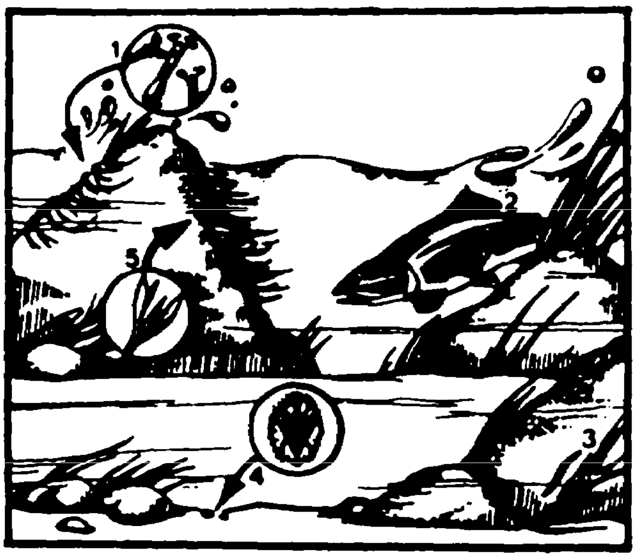
This approach must be used with caution for two reasons. First, in many urban, industrial, or heavily developed agricultural regions, even the least impacted sites are seriously degraded. Basing standards or criteria on such sites will set standards too low if these high levels of environmental degradation are considered acceptable or adequate. In such degraded regions, alternative sources for the regional reference may be needed (e.g., measures taken from the same region in a less developed neighboring State or historical records from the region before serious impact occurred). Second, in some regions the minimally-impacted sites are not typical of most sites in the region and may have remained unimpaired precisely because they are unique. These two considerations emphasize the need to select reference sites very carefully, based on solid quantitative data interpreted by professionals familiar with the biota of the region.

Each State, or groups of States, can select a series of regional reference sites that represent the attainable conditions for each region. Once biological criteria are established using this approach, the cost for evaluating local impairments is often lower than a series of measures of site-specific reference sites. Using paired watershed reference conditions immediately in regulatory programs will provide the added benefit of building a database for the development of regions of ecological similarity.

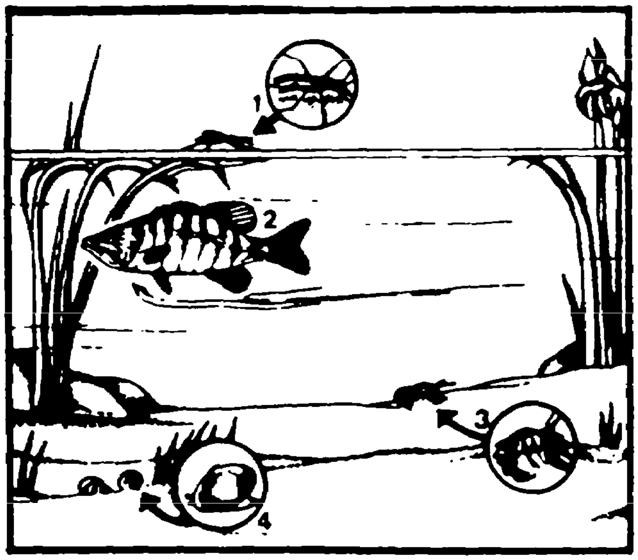
Chapter 6

The Biological Survey

A critical element of biological criteria is the characterization of biological communities inhabiting surface waters. Use of biological data is not new; biological information has been used to assess impacts from pollution since the 1890s (Forbes 1928), and most States currently incorporate biological information in their decisions about the quality of surface waters. However, biological information can be obtained through a variety of methods, some of which are more effective than others for characterizing resident aquatic biota. Biological criteria are developed using biological surveys; these provide the only direct method for measuring the structure and function of an aquatic community.



Different subhabitat within the same surface water will contain unique aquatic community components. In fast-flowing stream segments species such as (1) black fly larva; (2) brook trout; (3) water penny; (4) crane fly larva; and (5) water moss occur.



However, in slow-flowing stream segments, species like (1) water strider; (2) smallmouth bass; (3) crayfish; and (4) fingernail clams are abundant.

Biological survey study design is of critical importance to criteria development. The design must be scientifically rigorous to provide the basis for legal action, and be biologically relevant to detect problems of regulatory concern. Since it is not financially or technically feasible to evaluate all organisms in an entire ecosystem at all times, careful selection of community components, the time and place chosen for assessments, data gathering methods used, and the consistency with which these variables are applied will determine the success of the biological criteria program. Biological surveys must therefore be carefully planned to meet scientific and legal requirements, maximize information, and minimize cost.

Biological surveys can range from collecting samples of a single species to comprehensive evaluations of an entire ecosystem. The first approach is difficult to interpret for community assessment; the second approach is expensive and *impractical*. A balance between these extremes can meet program needs. Current approaches range between detailed ecological surveys, biosurveys of targeted community components, and biological indicators (e.g., keystone species). Each of these biosurveys has advantages and limitations. Additional discussion will be provided in technical guidance under development.

No single type of approach to biological surveys is always best. Many factors affect the value of the approach, including seasonal variation, waterbody size, physical boundaries, and other natural characteristics. Pilot testing alternative approaches in State waters may be the best way to determine the sensitivity of specific methods for evaluating biological integrity of local waters. Due to the number of alternatives available and the diversity of ecological systems, *individuals responsible for research design should be experienced biologists with expertise in the local and regional ecology of target surface waters.* States should develop a data management program that includes data analysis and evaluation and standard operating procedures as part of a Quality Assurance Program Plan.

When developing study designs for biological criteria, two key elements to consider include (1) selecting aquatic community components that will best represent the biological integrity of State surface waters and (2) designing data collection protocols to ensure the best representation of the aquatic community. Technical guidance currently available to aid the development of study design include: *Water Quality Standards Handbook* (U.S. EPA 1983a), *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses* (U.S. EPA 1983b); *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Volume II: Estuarine Systems* (U.S. EPA 1984a); and *Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Volume III: Lake Systems* (U.S. EPA 1984b). Future technical guidance will build on these documents and provide specific guidance for biological criteria development.

Selecting Aquatic Community Components

Aquatic communities contain a variety of species that represent different trophic levels, taxonomic groups, functional characteristics, and tolerance ranges. Careful selection of target taxonomic groups can provide a balanced assessment that is sufficiently broad to describe the structural and functional condition of an aquatic ecosystem, yet be sufficiently practical to use on a daily basis (Plafkin et al. 1989; Lenat 1988). When selecting community components to include in a *biological assessment, primary emphasis should go* toward including species or taxa that (1) serve as effective indicators of high biological integrity (i.e., those likely to live in unimpaired waters), (2) represent a range of pollution tolerances, (3) provide predictable, repeatable results, and (4) can be readily identified by trained State personnel.

Fish, macroinvertebrates, algae, and zooplankton are most commonly used in current bioassessment programs. The taxonomic groups chosen will vary depending on the type of aquatic ecosystem being assessed and the type of expected impairment. For example, benthic macroinvertebrate and fish communities are taxonomic groups often chosen for flowing fresh water. Macroinvertebrates and fish both provide valuable ecological information while fish correspond to the regulatory and *public perceptions of water quality* and reflect cumulative environmental stress over longer time frames. Plants are often used in wetlands, and algae are useful in lakes and estuaries to assess eutrophication. In marine systems, benthic macroinvertebrates and submerged aquatic vegetation may provide key community components. Amphipods, for example, dominate many aquatic communities and are more sensitive than other invertebrates such as polychaetes and molluscs to a wide variety of pollutants including hydrocarbons and heavy metals (Reich and Hart 1979; J.D. Thomas, pers. comm.).

It is beneficial to supplement standard groups with additional community components to meet specific goals, objectives, and resources of the assessment program. Biological surveys that use two or three taxonomic groups (e.g., fish, macroinvertebrates, algae) and, where appropriate, include different trophic levels within each group (e.g., primary, secondary, and tertiary consumers) will provide a more realistic evaluation of system biological integrity. This is analogous to using species from two or more taxonomic groups in bioassays. Impairments that are difficult to detect because of the temporal or spatial habits or the pollution tolerances of one group may be revealed through impairments in different species or assemblages (Ohio EPA 1988a).

Selection of aquatic community components that show different sensitivities and responses to the same perturbation will aid in identifying the nature of a problem. Available data on the ecological function, distribution, and abundance of species in a given habitat will help determine the most appropriate target species or taxa for biological surveys in the habitat. The selection of community components should also depend on the ability of the organisms to be accurately identified by trained State personnel. Attendent with the biological criteria program should be the development of identification keys for the organisms selected for study in the biological survey.

Biological Survey Design

Biological surveys that measure the structure and function of aquatic communities will provide the information needed for biological criteria development. Elements of community structure and function may be evaluated using a series of metrics. Structural metrics describe the composition of a community, such as the number of different species, relative abundance of specific species, and number and relative abundance of tolerant and intolerant species. Functional metrics describe the ecological processes of the community. These may include measures such as community photosynthesis or respiration. Function may also be estimated from the proportions of various feeding groups (e.g., omnivores, herbivores, and insectivores, or shredders, collectors, and grazers). Biological surveys can offer variety and flexibility in application. Indices currently available are primarily for freshwater streams. However, the approach has been used for lakes and can be developed for estuaries and wetlands.

Selecting the metric

Several methods are currently available for measuring the relative structural and functional well-being of fish assemblages in freshwater streams, such as the Index of Biotic Integrity (IBI); Karr 1981; Karr et al. 1986; Miller et al. 1988) and the Index of Well-being (IWB; Gammon 1976, Gammon et al. 1981). The IBI is one of the more widely used assessment methods. For additional detail, see the "Index of Biotic Integrity" feature.

Index of Biotic Integrity

The Index of Biotic Integrity (IBI) is commonly used for fish community analysis (Karr 1981). The original IBI was comprised of 12 metrics

- *six metrics evaluate species richness and composition*
  - *number of species*
  - *number of darter species*
  - *number of sucker species*
  - *number of sunfish species*
  - *number of intolerant species*
  - *proportion of green sunfish*
- *three metrics quantify trophic composition*
  - *proportion of omnivores*
  - *proportion of insectivorous cyprinids*
  - *proportion of piscivores*
- *three metrics summarize fish abundance and condition information*
  - *number of individuals in sample*
  - *proportion of hybrids*
  - *proportion of individuals with disease*

Each metric is scored 1 (worst), 3, or 5 (best) depending on how the field data compare with an expected value obtained from reference sites. All 12 metric values are then summed to provide an overall index value that represents relative integrity. The IBI was designed for midwestern streams; substitute metrics reflecting the same structural and functional characteristics have been created to accommodate regional variations in fish assemblages (Miller et al. 1988)

Several indices that evaluate more than one community characteristic are also available for assessing stream macroinvertebrate populations. Taxa richness, EPT taxa (number of taxa of the insect orders Ephemeroptera, Plecoptera, and Tricoptera), and species pollution tolerance values are a few of several components of these macroinvertebrate assessments. Example indices include the Invertebrate Community Index (ICI; Ohio EPA, 1988) and Hilsenhoff Biotic Index (HBI; Hilsenhoff, 1987).

Within these metrics specific information on the pollution tolerances of different species within a system will help define the type of impacts occurring in a waterbody. Biological indicator groups (intolerant species, tolerant species, percent of diseased organisms) can be used for evaluating community biological integrity if sufficient data have been collected to support conclusions drawn from the indicator data. In marine systems, for example, amphipods have been used by a number of researchers as environmental indicators (McCall 1977; Botton 1979; Mearns and Word 1982).

Sampling design

Sampling design and statistical protocols are required to reduce sampling error and evaluate the natural variability of biological responses that are found in both laboratory and field data. High variability reduces the power of a statistical test to detect real impairments (Sokal and Rohlf, 1981). States may reduce variability by refining sampling techniques and protocol to decrease variability introduced during data collection, and increase the power of the evaluation by increasing the number of replications. Sampling techniques are refined, in part, by collecting a representative sample of resident biota from the same component of the aquatic community from the same habitat type in the same way at sites being compared. Data collection protocols should incorporate (1) spatial scales (where and how samples are collected) and (2) temporal scales (when data are collected) (Green, 1979):

- **Spatial Scales** refer to the wide variety of subhabitats that exist within any surface water habitat. To account for subhabitats, adequate sampling protocols require selecting (1) the location within a habitat where target groups reside and (2) the method for collecting data on target groups. For example, if fish are sampled only from fast flowing riffles within stream A, but are sampled from slow flowing pools in stream B, the data will not be comparable.

- **Temporal Scales** refer to aquatic community changes that occur over time because of diurnal and life-cycle changes in organism behavior or development, and seasonal or annual changes in the environment. Many organisms go through seasonal life-cycle changes that dramatically affect their presence and abundance in the aquatic community. For example, macroinvertebrate data collected from stream A in March and stream B in May, would not be comparable because the emergence of insect adults after March would significantly alter the abundance of subadults found in stream B in May. Similar problems would occur if algae were collected in lake A during the dry season and lake B during the wet season.

Field sampling protocols that produce quality assessments from a limited number of site visits greatly enhance the utility of the sampling technique. Rapid bioassessment protocols, recently developed for assessing streams, use standardized techniques to quickly gather physical, chemical, and biological quantitative data that can assess changes in biological integrity (Plafkin et al. 1989). Rapid bioassessment methods can be cost-effective biological assessment approaches when they have been verified with more comprehensive evaluations for the habitats and region where they are to be applied.

Biological survey methods such as the IBI for fish and ICI for macroinvertebrates were developed in streams and rivers and have yet to be applied to many ecological regions. In addition, further research is needed to adapt the approach to lakes, wetlands, and estuaries, including the development of alternative structural or functional endpoints. For example, assessment methods for algae (e.g. measures of biomass, nuisance bloom frequency, community structure) have been used for lakes. Assessment metrics appropriate for developing biological criteria for lakes, large rivers, wetlands, and estuaries are being developed and tested so that a multi-metric approach can be effectively used for all surface waters.

Chapter 7

Hypothesis Testing: Biological Criteria and the Scientific Method

Biological criteria are applied in the standards program by testing hypotheses about the biological integrity of impacted surface waters. These hypotheses include the null hypothesis—the designated use of the waterbody is not impaired—and alternative hypotheses such as the designated use of the waterbody is impaired (more specific hypotheses can also be generated that predict the type(s) of impairment). Under these hypotheses specific predictions are generated concerning the kinds and numbers of organisms representing community structure and function expected or found in unimpaired habitats. The kinds and numbers of organisms surveyed in unimpaired waters are used to establish the biological criteria. To test the alternative hypotheses, data collection and analysis procedures are used to compare the criteria to comparable measures of community structure and function in impacted waters.

Multiple impacts in the same surface water such as discharges of effluent from point sources, leachate from landfills or dumps, and erosion from habitat degradation each contribute to impairment of the surface water. All impacts should be considered during the diagnosis process.

Hypothesis Testing

To detect differences of biological and regulatory concern between biological criteria and ambient biological integrity at a test site, it is important to establish the sensitivity of the evaluation. A 10 percent difference in condition is more difficult to detect than a 50 percent difference. For the experimental/survey design to be effective, the level of detection should be predetermined to establish sample size for data collection (Sokal and Rohlf 1981). Knowledge of expected natural variation, experimental error, and the kinds of detectable differences that can be expected will help determine sample size and location. This forms the basis for defining data quality objectives, standardizing data collection procedures, and developing quality assurance/ quality control standards.

Once data are collected and analyzed, they are used to test the hypotheses to determine if characteristics of the resident biota at a test site are significantly different from established criteria values for a comparable habitat. There are three possible outcomes:

1. The use is impaired when survey design and data analyses are sensitive enough to detect differences of regulatory importance, and significant differences were detected. The next step is to diagnose the cause(s) and source(s) of impairment.

2. The biological criteria are met when survey design and data analyses are sensitive enough to detect differences of regulatory significance, but no differences were found. In this case, no action is required by States based on these measures. However, other evidence may indicate impairment (e.g., chemical criteria are violated; see below).

3. The outcome is indeterminate when survey design and data analyses are not sensitive enough to detect differences of regulatory significance, and no differences were detected. If a State or Region determines that this is occurring, the development of study design and evaluation for biological criteria was incomplete. States must then determine whether they will accept the sensitivity of the survey or conduct additional surveys to increase the power of their analyses. If the sensitivity of the original survey is accepted, the State should determine what magnitude of difference the survey is capable of detecting. This will aid in re-evaluating research design and desired detection limits. An indeterminate outcome may also occur if the test site and the reference conditions were not comparable. This variable may also require re-evaluation.

As with all scientific studies, when implementing biological criteria, the purpose of hypothesis testing is to determine if the data support the conclusion that the null hypothesis is false (i.e., the designated use is not impaired in a particular waterbody). Biological criteria cannot prove attainment. This reasoning provides the basis for emphasizing independent application of different assessment methods (e.g., chemical verses biological criteria). No type of criteria can "prove" attainment; each type of criteria can disprove attainment.

Although this discussion is limited to the null and one alternative hypothesis, it is possible to generate multiple working hypotheses (Popper, 1968) that promote the diagnosis of water quality problems when they exist. For example, if physical habitat limitations are believed to be causing impairment (e.g., sedimentation) one alternative hypothesis could specify the loss of community components sensitive to this impact. Using multiple hypotheses can maximize the information gained from each study. See the Diagnosis section for additional discussion.

Diagnosis

When impairment of the designated use is found using biological criteria, a diagnosis of probable cause of impairment is the next step for implementation. Since biological criteria are primarily designed to detect water quality impairment, problems are likely to be identified without a known cause. Fortunately the process of evaluating test sites for biological impairment provides significant information to aid in determining cause.

During diagnostic evaluations, three main impact categories should be considered: chemical, physical, and biological. To begin the diagnostic process two questions are posed:

- What are the obvious causes of impairment?
- If no obvious causes are apparent, what possible causes do the biological data suggest?

Obvious causes such as habitat degradation, point source discharges, or introduced species are often identified during the course of a normal field biological assessment. Biomonitoring programs normally provide knowledge of potential sources of impact and characteristics of the habitat. As such, diagnosis is partly incorporated into many existing State field-oriented bioassessment programs. If more than one impact source is obvious, diagnosis will require determining which impact(s) is the cause of impairment or the extent to which each impact contributes to impairment. The nature of the biological impairment can guide evaluation (e.g., chemical contamination may lead to the loss of sensitive species, habitat degradation may result in loss of breeding habitat for certain species).

Case studies illustrate the effectiveness of biological criteria in identifying impairments and possible sources. For example, in Kansas three sites on Little Mill Creek were assessed using Rapid Bioassessment Protocols (Plafkin et al. 1989; see Fig. 4). Based on the results of a comparative analysis, habitats at the three sites were comparable and of high quality. Biological impairment, however, was identified at two of the three sites and directly related to proximity to a point source discharge from a sewage treatment plant. The severely impaired Site (STA 2) was located approximately 100 meters downstream from the plant. The slightly impaired Site (STA 3) was located between one and two miles downstream from the plant. However, the unimpaired Site (STA 1(R)) was approximately 150 meters upstream from the plant (Plafkin et al. 1989). This simple example illustrates the basic principles of diagnosis. In this case the treatment plant appears responsible for impairment of the resident biota and the discharge needs to be evaluated. Based on the biological survey the results are clear. However, impairment in resident populations of macroinvertebrates probably would not have been recognized using more traditional methods.

In Maine, a more complex problem arose when effluents from a textile plant met chemical-specific and effluent toxicity criteria, yet a biological survey of downstream biota revealed up to 80 percent reduction in invertebrate richness below plant outfalls. Although the source of impairment seemed clear, the cause of impairment was more difficult to determine. By engaging in a diagnostic evaluation, Maine was able to determine that the discharge contained chemicals not regulated under current programs and that part of the toxicity effect was due to the sequential discharge of unique effluents (tested individually these effluents were not toxic; when exposure was in a particular sequence, toxicity occurred). Use of biological criteria resulted in the detection and diagnosis of this toxicity problem, which allowed Maine to develop workable alternative operating procedures for the textile industry to correct the problem (Courtemanch 1989, and pers. comm.).

During diagnosis it is important to consider and discriminate among multiple sources of impairment. In a North Carolina stream (see Figure 5) four sites were evaluated using rapid bioassessment techni-

Figure 4.—Kansas: Benthic Bioassessment of Little Mill Creek (Little Mill Creek = Site-Specific Reference) Relationship of Habitat and Bioassessment

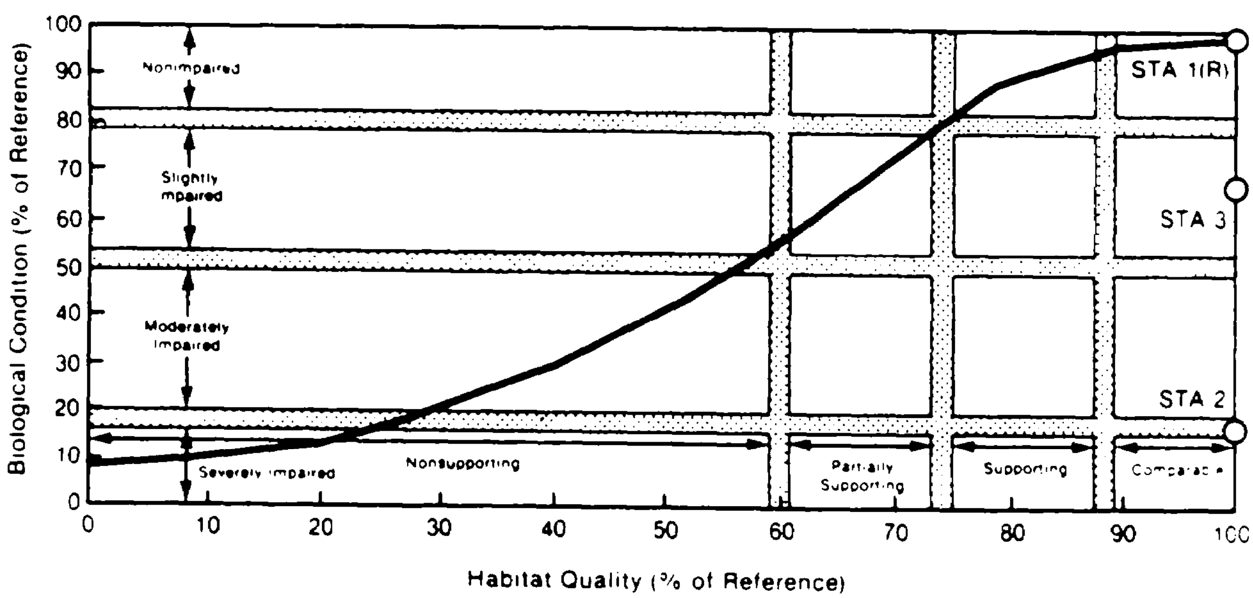


Fig. 4. Three stream segments sampled in a stream in Kansas using Rapid Bioassessment Protocols (Plafkin et al. 1989) reveals significant impairments at sites below a sewage treatment plant

Figure 5.—The Relationship Between Habitat Quality and Benthic Community Condition at the North Carolina Pilot Study Site.

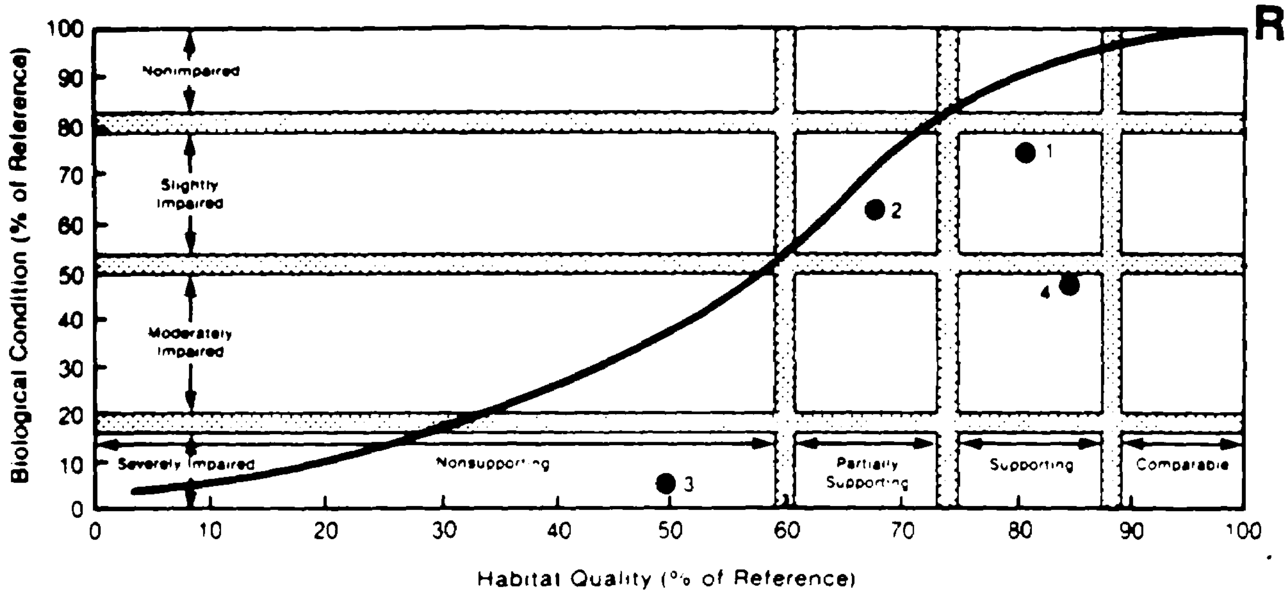


Fig. 5: Distinguishing between point and nonpoint sources of impairment requires an evaluation of the nature and magnitude of different sites in a surface water (Plafkin, et al. 1989)

ques. An ecoregional reference site (R) established the highest level of biological integrity for that stream type. Site (1), well upstream from a local town, was used as the upstream reference condition. Degraded conditions at Site (2) suggested nonpoint source problems and habitat degradation because of proximity to residential areas on the upstream edge of town. At Site (3) habitat alterations, nonpoint runoff, and point source discharges combined to severely degrade resident biota. At this site, sedimentation and toxicity from municipal sewage treatment effluent appeared responsible for a major portion of this degradation. Site (4), although several miles downstream from town, was still impaired despite significant improvement in habitat quality. This suggests that toxicity from upstream discharges may still be occurring (Barbour, 1990 pers. comm.). Using these kinds of comparisons, through a diagnostic procedure and by using available chemical and biological assessment tools, the relative effects of impacts can be determined so that solutions can be formulated to improve water quality.

When point and nonpoint impact and physical habitat degradation occur simultaneously, diagnosis may require the combined use of biological, physical, and chemical evaluations to discriminate between these impacts. For example, sedimentation of a stream caused by logging practices is likely to result in a decrease in species that require loose gravel for spawning but increase species naturally adapted to fine sediments. This shift in community components correlates well with the observed impact. However, if the impact is a point source discharge or nonpoint runoff of toxicants, both species types are likely to be impaired whether sedimentation occurs or not (although gravel breeding species can be expected to show greater impairment if sedimentation occurs). Part of the diagnostic process is derived from an understanding of organism sensitivities to different kinds of impacts and their habitat requirements. When habitat is good but water quality is poor, aquatic community components sensitive to toxicity will be impaired. However, if both habitat and water quality degrade, the resident community is likely to be composed of tolerant and opportunistic species.

When an impaired use cannot be easily related to an obvious cause, the diagnostic process becomes investigative and iterative. The iterative diagnostic process as shown in Figure 6 may require additional time and resources to verify cause and source. Initially, potential sources of impact are identified and mapped to determine location relative

Figure 6.—Diagnostic Process

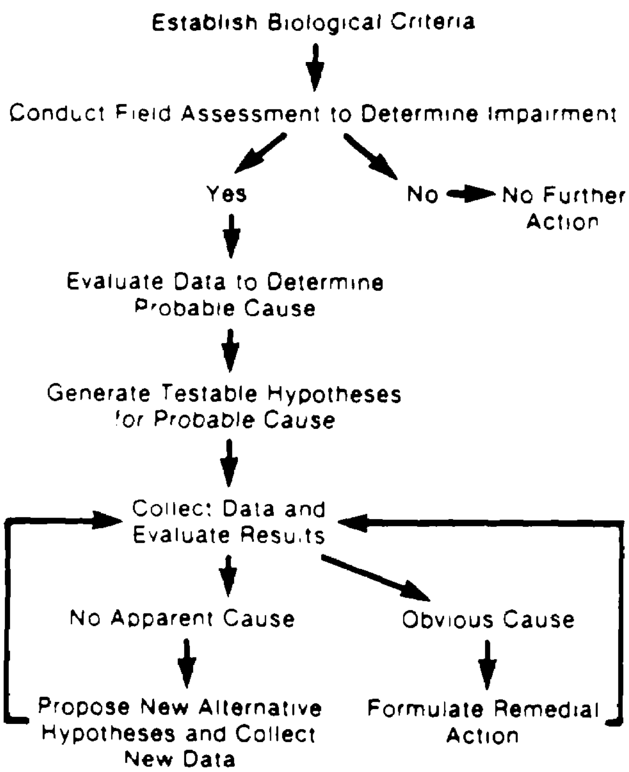


Fig. 6 The diagnostic process is a stepwise process for determining the cause of impaired biological integrity in surface waters. It may require multiple hypotheses testing and more than one remedial plan.

to the area suffering from biological impairment. An analysis of the physical, chemical, and biological characteristics of the study area will help identify the most likely sources and determine which data will be most valuable. Hypotheses that distinguish between possible causes of impairment should be generated. Study design and appropriate data collection procedures need to be developed to test the hypotheses. The severity of the impairment, the difficulty of diagnosis, and the costs involved will determine how many iterative loops will be completed in the diagnostic process.

Normally, diagnoses of biological impairment are relatively straightforward. States may use biological criteria as a method to confirm impairment from a known source of impact. However, the diagnostic process provides an effective way to identify unknown impacts and diagnose their cause so that corrective action can be devised and implemented

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Appendix A

Common Questions and Their Answers

Q. *How will implementing biological criteria benefit State water quality programs?*

A. State water quality programs will benefit from biological criteria because they:

a) directly assess impairments in ambient biota from adverse impacts on the environment;

b) are defensible and quantifiable;

c) document improvements in water quality resulting from agency action;

d) reduce the likelihood of false positives (i.e., a conclusion that attainment is achieved when it is not);

e) provide information on the integrity of biological systems that is compelling to the public.

Q. *How will biological criteria be used in a permit program?*

A. When permits are renewed, records from chemical analyses and biological assessments are used to determine if the permit has effectively prevented degradation and led to improvement. The purpose for this evaluation is to determine whether applicable water quality standards were achieved under the expiring permit and to decide if changes are needed. Biological surveys and criteria are particularly effective for determining the quality of waters subject to permitted discharges. Since biosurveys provide ongoing integrative evaluations of the biological integrity of resident biota, permit writers can make informed decisions on whether to maintain or restrict permit limits.

Q. *What expertise and staff will be needed to implement a biological criteria program?*

A. Staff with sound knowledge of State aquatic biology and scientific protocol are needed to coordinate a biological criteria program. Actual field monitoring could be accomplished by summer-hire biologists led by permanent staff aquatic biologists Most States employ aquatic biologists for monitoring trends or issuing site-specific permits.

Q. *Which management personnel should be involved in a biologically-based approach?*

A. Management personnel from each area within the standards and monitoring programs should be involved in this approach, including permit engineers, resource managers, and field personnel.

Q. *How much will this approach cost?*

A. The cost of developing biological criteria is a State-specific question depending upon many variables. However, States that have implemented a biological criteria program have found it to be cost effective (e.g., Ohio). Biological criteria provide an integrative assessment over time. Biota reflect multiple impacts. Testing for impairment of resident aquatic communities can actually require less monitoring than would be required to detect many impacts using more traditional methods (e.g., chemical testing for episodic events).

Q. What are some concerns of dischargers?

A. Dischargers are concerned that biological criteria will identify impairments that may be erroneously attributed to a discharger who is not responsible. This is a legitimate concern that the discharger and State must address with careful evaluations and diagnosis of cause of impairment. However, it is particularly important to ensure that waters used for the reference condition are not already impaired as may occur when conducting site-specific upstream-downstream evaluations. Although a discharger may be contributing to surface water degradation, it may be hard to detect using biosurvey methods if the waterbody is also impaired from other sources. This can be evaluated by testing the possible toxicity of effluent-free reference waters on sensitive organisms.

Dischargers are also concerned that current permit limits may become more stringent if it is determined that meeting chemical and whole-effluent permit limits are not sufficient to protect aquatic life from discharger activities. Alternative forms of regulation may be needed; these are not necessarily financially burdensome but could involve additional expense.

Burdensome monitoring requirements are additional concerns. With new rapid bioassessment protocols available for streams, and under development for other surface waters, monitoring resident biota is becoming more straightforward. Since resident biota provide an integrative measure of environmental impacts over time, the need for continual biomonitoring is actually lower than chemical analyses and generally less expensive. Guidance is being developed to establish acceptable research protocols, quality assurance/quality control programs and training opportunities to ensure that adequate guidance is available.

Q. What are the concerns of environmentalists?

A. Environmentalists are concerned that biological criteria could be used to alter restrictions on dischargers if biosurvey data indicate attainment of a designated use even though chemical criteria and/or whole-effluent toxicity evaluations predict impairment. Evidence suggests that this occurs infrequently (e.g., in Ohio, 6 percent of 431 sites evaluated using chemical-specific criteria and biosurveys resulted in this disagreement). In those cases where evidence suggests more than one conclusion, independent application applies. If biological criteria suggest impairment but chemical-specific and/or whole-effluent toxicity implies attainment of the use, the cause for impairment of the biota is to be evaluated and, where appropriate, regulated. If whole effluent and/or chemical-specific criteria imply impairment but no impairment is found in resident biota, the whole-effluent and/or chemical-specific criteria provide the basis for regulation.

Q. Do biological criteria have to be codified in State regulations?

A. State water quality standards require three components: (1) designated uses, (2) protective criteria, and (3) an antidegradation clause. For criteria to be enforceable they must be codified in regulations. Codification could involve general narrative statements of biological criteria, numeric criteria, and/or criteria accompanied by specific testing procedures. Codifying general narratives provides the most flexibility—specific methods for data collection the least flexibility—for incorporating new data and improving data gathering methods as the biological criteria program develops. States should carefully consider how to codify these criteria.

Q. How will biocriteria fit into the agency's method of implementing standards?

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

Q. Who determines the values for biological criteria and decides whether a waterbody meets the criteria?

The process of developing biological criteria, including refined use classes, narrative criteria, and numeric criteria, must include agency managers, staff biologists, and the public through public hearings and comment. Once criteria are established, determining attainment\nonattainment of a use requires biological and statistical evaluation based on established protocols. Changes in the criteria would require the same steps as the initial criteria: technical modifications by biologists, goal clarification by agency managers, and public hearings. The key to criteria development and revision is a clear statement of measurable objectives.

Q. *What additional information is available on developing and using biological criteria?*

A. This program guidance document will be supplemented by the document *Biological Criteria Development by States* that includes case histories of State implementation of biological criteria as narratives, numerics, and some data procedures. The purpose for the document is to expand on material presented in Part I. The document will be available in October 1990.

A general *Biological Criteria Technical Reference Guide* will also be available for distribution during FY 1991. This document outlines basic approaches for developing biological criteria in all surface waters (streams, rivers, lakes, wetlands, estuaries). The primary focus of the document is to provide a reference guide to scientific literature that describes approaches and methods used to determine biological integrity of specific surface water types.

Over the next triennium more detailed guidance will be produced that focuses on each surface water type (e.g., technical guidance for streams will be produced during FY 91). Comparisons of different biosurvey approaches will be included for accuracy, efficacy, and cost effectiveness.

Appendix B

Biological Criteria Technical Reference Guide

Table of Contents (*tentative*)

Appendix C

Biological Criteria Development by States

Table of Contents (*tentative*)

Appendix D

Contributors and Reviewers

Contributors

Gerald Ankley
USEPA Environmental Research Lab
6201 Congdon Blvd.
Duluth, MN 55804

John Arthur
USEPA
ERL-Duluth
6201 Congdon Blvd.
Duluth, MN 55804

Patricia Bailey
Division of Water Quality
Minnesota Pollution Control Agency
520 Lafayette Road
St. Paul, MN 55155

Joe Ball
Wisconsin DNR
Water Resource Management (WR/2)
P.O. Box 7291
Madison, WI 53707

Michael Barbour
EA Engineering, Science, and Technology Inc.
Hunt Valley/Loveton Center
15 Loveton Circle
Sparks, MD 21152

Raymond Beaumier
Ohio EPA
Water Quality Laboratory
1030 King Avenue
Columbus, OH 43212

John Bender
Nebraska Department of Environmental Control
P.O. Box 94877
State House Station
Lincoln, NE 69509

Mark Blosser
Delaware Department of Natural Resources - Water Quality Mgmt. Branch
P.O. Box 1401, 89 Kings Way
Dover, DE 19903

Robert Bode
New York State Department of
Environmental Conservation
Box 1397
Albany, NY 12201

Lee Bridges
Indiana Department of Environment Management
5500 W. Bradbury
Indianapolis, IN 46241

Claire Buchanan
Interstate Commission on Potomac River Basin
6110 Executive Boulevard Suite 300
Rockville, MD 20852-3903

David Courtemanch
Maine Department of Environmental Protection
Director, Division of Environmental
Evaluation and Lake Studies
State House No. 17
Augusta, ME 04333

Norm Crisp
Environmental Services Division
USEPA Region 7
25 Funston Road
Kansas City, KS 66115

Susan Davies
Maine Department of Environmental Protection
State House No. 17
Augusta, ME 04333

Wayne Davis
Environmental Scientist
Ambient Monitoring Section
USEPA Region 5
536 S. Clark St. (5-SMQA)
Chicago, IL 60605

Kenneth Duke
Battelle
505 King Avenue
Columbus, OH 43201-2693

Gary Fandrei
Minnesota Pollution Control Agency
Division of Water Quality
520 La Fayette Road North
St. Paul, MN 55155

Steve Fiske
Vermont Department of Environmental Conservation
6 Baldwin St.
Montpelier, VT 05602

John Giese
Arkansas Department Of Pollution Control and Ecology
P.O. Box 9583
8001 National Drive
Little Rock, AR 72209

Steven Glomb
Office of Marine and Estuarine Protection
USEPA (WH-556F)
401 M Street SW
Washington, DC 20460

Steve Goodbred
Division of Ecological Services
U. S. Fish and Wildlife Service
1825 B. Virginia Street
Annapolis, MD 21401

Jim Harrison
USEPA Region 4
345 Courtland St. (4WM-MEB)
Atlanta, GA 30365

Margarete Heber
Office of Water Enforcements and Permits
USEPA (EN-336)
401 M Street SW
Washington, DC 20460

Steve Hedtke
US EPA Environmental Research Lab
6201 Congdon Blvd.
Duluth, MN 55804

Robert Hite
Illinois EPA
2209 West Main
Marion, IL 62959

Linda Holst
USEPA Region 3
841 Chestnut Street
Philadelphia, PA 19107

Evan Hornig
USEPA Region 6
First Interstate Bank at Fountain Place
1445 Ross Avenue, Suite 1200
Dallas, TX 75202

William B. Horning II
Aquatic Biologist, Project Management Branch
USEPA/ORD Env. Monitoring Systems
3411 Church St.
Cincinnati, OH 45244

Robert Hughes
NSI Technology Services
200 SW 35th Street
Corvallis, OR 97333

Jim Hulbert
Florida Department of Environmental Regulation
Suite 232
3319 Maguire Blvd.
Orlando, FL 32803

James Kennedy
Institute of Applied Sciences
North Texas State University
Denton, TX 76203

Richard Langdon
Vermont Department of Environmental Conservation—10 North
103 S. Main Street
Waterbury, VT 05676

John Lyons
Special Projects Leader
Wisconsin Fish Research Section
Wisconsin Department of Natural Resources
3911 Fish Hatchery Rd.
Fitchburg, WI 53711

Anthony Maciorowski
Battelle
505 King Avenue
Columbus, OH 43201-2693

Suzanne Marcy
Office of Water Regulations and Standards
USEPA (WH 585)
401 M St. SW
Washington, DC 20460

Scott Mattee
Geological Survey of Alabama
PO Drawer O
Tuscaloosa, AL 35486

John Maxted
Delaware Department of Natural Resources and Environmental Control
39 Kings Highway, P.O. Box 1401
Dover, DE 19903

Jimmie Overton
NC Dept of Natural Resources and Community Development
P.O. Box 27687
512 N. Salisbury
Raleigh, NC 27611-7687

Steve Paulsen
Enviromental Research Center
University of Nevada - Las Vegas
4505 Maryland Parkey
Las Vegas, NV 89154

Loys Parrish
USEPA Region 8
P.O. Box 25366
Denver Federal Center
Denver, CO 80225

David Penrose
Environmental Biologist
North Carolina Department of Natural Resources and Community Development
512 N. Salisbury Street
Raleigh, NC 27611

Don Phelps
USEPA
Environmental Research Lab
South Ferry Road
Narragansett, RI 02882

Ernest Pizzuto
Connecticut Department Environmental Protection
122 Washington Street
Hartford, CT 06115

James Plafkin
Office of Water Regulations and Standards
USEPA (WH 553)
401 M Street, SW
Washington, DC 20460

Ronald Preston
Biological Science Coordinator
USEPA Region 3
Wheeling Office (3ES12)
303 Methodist Building
Wheeling, WV 26003

Ronald Raschke
Ecological Support Branch
Environmental Services Division
USEPA Region 4
Athens, GA 30613

Mark Southerland
Dynamac Corporation
The Dynamac Building
11140 Rickville Pike
Rockville, MD 20852

James Thomas
Newfound Harbor Marine Institute
Rt. 3, Box 170
Big Pine Key, FL 33043

Nelson Thomas
USEPA, ERL-Duluth
Senior Advisor for National Program
6201 Congdon Blvd.
Duluth, MN 55804

Randall Waite
USEPA Region 3
Program Support Branch (3WMIO)
841 Chesnut Bldg.
Philadelphia, PA 19107

John Wegrzyn
Manager, Water Quality Standards Unit
Arizona Department of Environmental Quality
2005 North Central Avenue
Phoenix, AZ 95004

Thom Whittier
NSI Technology Services
200 SW 35th Street
Corvallis, OR 97333

Bill Wuerthele
Water Management Division
USEPA Region 8 (WM-SP)
999 18th Street Suite 500
Denver, CO 80202

Chris Yoder
Asst. Manager, Surface Water Section
Water Quality Monitoring and Assessment
Ohio EPA-Water Quality Lab
1030 King Ave.
Columbus, OH 43212

David Yount
US EPA Environmental Research Lab
6201 Congdon Blvd.
Duluth, MN 55804

Lee Zeni
Interstate Commission on Potomac River Basin
6110 Executive Boulevard Suite 300
Rockville, MD 20852-3903

Reviewers

Paul Adamus
Wetlands Program
NSI Technology Services
200 S.W. 35th Street
Corvallis, OR 97333

Rick Albright
USEPA Region 10 (WD-139)
1200 6th Avenue NW
Seattle, WA 98101

Max Anderson
USEPA Region 5
536 S. Clark St. (5SCRL)
Chicago, IL 60605

Michael D. Bilger
USEPA Region 1
John F. Kennedy Building
Boston, MA 02203

Susan Boldt
University of Wisconsin Extension
Madison, WI

Paul Campanella
Office of Policy, Planning and Evaluation
USEPA (PM 222-A)
401 M St. S.W.
Washington, DC 20460

Cindy Carusone
New York Department of Environmental Conservation
Box 1397
Albany, NY 12201

Brian Choy
Hawaii Department of Health
645 Halekauwila St.
Honolulu, HI 96813

Bill Creal
Michigan DNR
Surface Water Quality Division
P.O. Box 30028
Lansing, MI 48909

Phil Crocker
Water Quality Management Branch
USEPA Region 6 / 1445 Ross Ave.
Dallas, TX 75202-2733

Kenneth Cummins
Appalachian Environmental Lab
University of Maryland
Frostburg, MD 21532

Jeff DeShon
Ohio EPA, Surface Water Section
1030 King Ave.
Columbus, OH 43212

Peter Farrington
Biomonitoring Assessments Officer
Water Quality Branch
Inland Waters Directorate
Environment Canada
Ottawa, Ontario K1A OH3

Kenneth Fenner
USEPA Region 5
Water Quality Branch
230 S. Dearborn
Chicago, IL 60604

Jack Freda
Ohio EPA
Surface Water Section
1030 King Avenue
Columbus, OH 43212

Toby Frevert
Illinois EPA
Division of Water Pollution Control
2200 Churchill Road
Springfield, IL 62706

Cynthia Fuller
USEPA GLNPO
230 S. Dearborn
Chicago, IL 60604

Jeff Gagler
USEPA Region 5
230 S. Dearborn (5WQS)
Chicago, IL 60604

Mary Jo Garrels
Maryland Department of the Environment
2500 Broening Highway
Building 30
Baltimore, MD 21224

Jim Giattina
USEPA Region 5
230 S. Dearborn (5WQP)
Chicago, IL 60604

Jim Green
Environmental Services Division
USEPA Region 3
303 Methodist Bldg.
11th and Chapline
Wheeling, WV 26003

Larindo Gronner
USEPA Region 4
345 Courtland St.
Atlanta, GA 30365

Martin Gurtz
U.S. Geological Survey, WRD
P.O. Box 2857
Raleigh, NC 27602-2857

Rick Hafele
Oregon Department Environmental Quality
1712 S.W. 11th Street
Portland, OR 97201

Steve Heiskary
MN Pollution Control Agency
520 Lafayette Road
St. Paul, MN 55155

Rollie Hemmett
USEPA Region 2
Environmental Services
Woodridge Avenue
Edison, NJ 08837

Charles Hocutt
Horn Point Environmental Laboratory
Box 775 University of Maryland
Cambridge, MD 21613

Hoke Howard
USEPA Region 4
College Station Road
Athens, GA 30605

Peter Husby
USEPA Region 9
215 Freemont St
San Francisco, CA 94105

Gerald Jacobi
Environmental Sciences
School of Science and Technology
New Mexico Highlands University
Las Vegas, NM 87701

James Karr
Department of Biology
Virginia Polytechnic Institute and State University
Blacksburg, VA 24061-0406

Roy Kleinsasser
Texas Parks and Wildlife
P.O. Box 947
San Marcos, TX 78667

Don Klemm
USEPA Environmental Monitoring and Systems Laboratory
Cincinnati, OH 45268

Robin Knox
Louisiana Department of Environment Quality
P.O. Box 44091
Baton Rouge, LA 70726

Robert Koroncai
Water Management Division
USEPA Region 3
847 Chestnut Bldg.
Philadelphia, PA 19107

Jim Kurztenbach
USEPA Region 2
Woodbridge Ave.
Rariton Depot Bldg. 10
Edison, NJ 08837

Roy Kwiatkowski
Water Quality Objectives Division
Water Quality Branch
Environment Canada
Ottawa, Ontario Canada
K1A OH3

Jim Lajorchak
EMSL-Cincinnati
U.S. Environmental Protection Agency
Cincinnati, OH

David Lenat
NC Dept of Natural Resources and Community Development
512 N. Salisbury St.
Raleigh, NC 27611

James Luey
USEPA Region 5
230 S. Dearborn (5WQS)
Chicago, IL 60604

Terry Maret
Nebraska Department of Environmental Control
Box 94877
State House Station
Lincoln, NE 69509

Wally Matsunaga
Illinois EPA
1701 First Ave., #600
Maywood, IL 60153

Robert Mosher
Illinois EPA
2200 Churchill Rd. #15
P.O. Box 19276
Springfield, IL 62794

Phillip Oshida
USEPA Region 9
215 Fremont Street
San Francisco, CA 94105

Bill Painter
USEPA, OPPE
401 M Street, SW (W435B)
Washington, DC 20460

Rob Pepin
USEPA Region 5
230 S. Dearborn
Chicago, IL 60604

Wayne Poppe
Tennessee Valley Authority
270 Haney Bldg.
Chattanooga, TN 37401

Walter Redmon
USEPA Region 5
230 S. Dearborn
Chicago, IL 60604

Landon Ross
Florida Department of
Environmental Regulation
2600 Blair Stone Road
Tallahassee, FL 32399

Jean Roberts
Arizona Department of
Environmental Quality
2655 East Magnolia
Phoenix, AZ 85034

Charles Saylor
Tennessee Valley Authority
Field Operations Eastern Area
Division of Services and Field
Operations
Norris, TN 37828

Robert Schacht
Illinois EPA
1701 First Avenue
Maywood, IL 60153

Duane Schuettpelz
Chief, Surface Water Standards and
Monitoring Section-Wisconsin
Department of Natural
Resources
Box 7921
Madison, WI 53707

Bruce Shackleford
Arkansas Department of Pollution
Control and Ecology
8001 National Drive
Little Rock, AR 72209

Larry Shepard
USEPA Region 5
230 S. Dearborn (5WQP)
Chicago, IL 60604

Jerry Shulte
Ohio River Sanitation Commission
49 E. 4th St., Suite 851
Cincinnati, OH 45202

Thomas Simon
USEPA Region 5
536 S. Clark St. (5SCRL)
Chicago, IL 60605

J. Singh
USEPA Region 5
536 Clark St. (5SCDO)
Chicago, IL 60605

Marc Smith
Biomonitoring Section
Ohio EPA
1030 King Avenue
Columbus, OH 43212

Denise Steurer
USEPA Region 5
230 S. Dearborn
Chicago, IL 60604

William Tucker
Supervisor, Water Quality
Monitoring
Illinois EPA
Division of Water Pollution Control
4500 S. Sixth Street
Springfield, IL 62706

Stephen Twidwell
Texas Water Commission
P.O. Box 13087
Capital Station
Austin, TX 78711-3087

Barbara Williams
USEPA Region 5
230 S. Dearborn
Chicago, IL 60604

APPENDIX D

National Guidance: Water Quality Standards for Wetlands

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

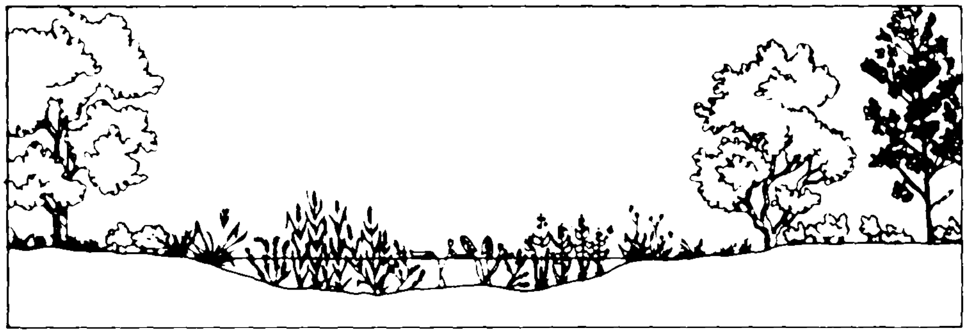
United States
Environmental Protection
Agency

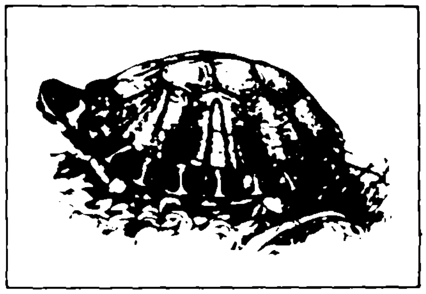
Office of Water
Regulations and Standards (WH-585)
Washinton, DC 20460

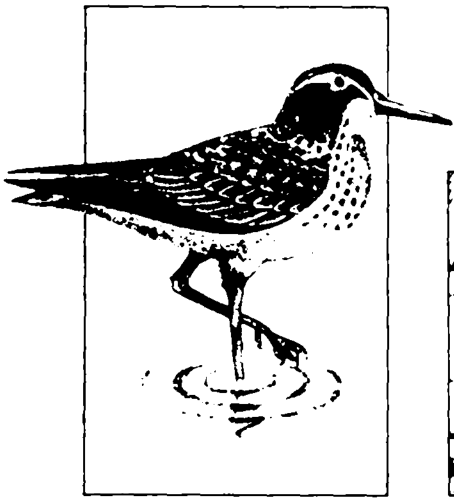
EPA 440/S-90-011
July 1990

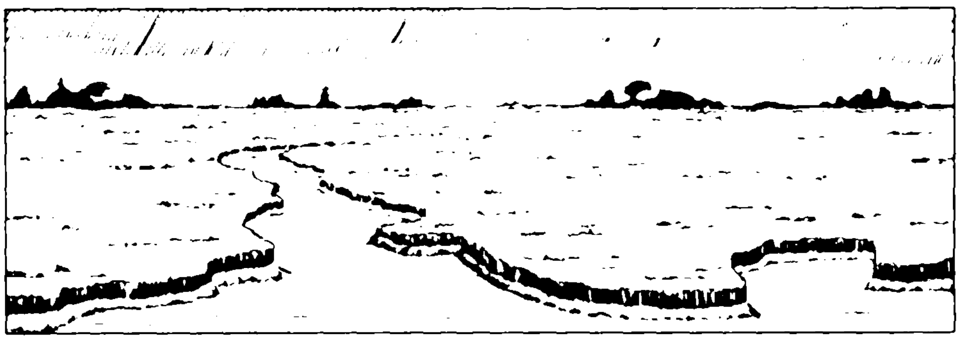
Water Quality Standards for Wetlands

National Guidance









WATER QUALITY STANDARDS FOR WETLANDS

National Guidance

July 1990

Prepared by:

U.S. Environmental Protection Agency
Office of Water Regulations and Standards
Office of Wetlands Protection

This document is designated as Appendix B to Chapter 2 - General Program Guidance of the Water Quality Standards Handbook, December 1983.

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UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF
WATER

MEMORANDUM

SUBJECT: Final Document: National Guidance on Water Quality Standards for Wetlands

FROM: Martha G. Prothro, Director
Office of Water Regulations and Standards

David G. Davis, Director
Office of Wetlands Protection

TO: Regional Water Division Directors
Regional Environmental Services Division Directors
Assistant Regional Administrator for Policy and Management, Region VII
OW Office Directors
State Water Quality Program Managers
State Wetland Program Managers

The following document entitled "National Guidance: Water Quality Standards for Wetlands" provides guidance for meeting the priority established in the FY 1991 Agency Operating Guidance to develop water quality standards for wetlands during the FY 1991-1993 triennium. This document was developed jointly by the Office of Water Regulations and Standards (OWRS) and the Office of Wetlands Protection (OWP), and reflects the comments we received on the February 1990 draft from EPA Headquarters and Regional offices, EPA laboratories, and the States.

By the end of FY 1993, the minimum requirements for States are to include wetlands in the definition of "State waters", establish beneficial uses for wetlands, adopt existing narrative and numeric criteria for wetlands, adopt narrative biological criteria for wetlands, and apply antidegradation policies to wetlands. Information in this document related to the development of biological criteria has been coordinated with recent guidance issued by OWRS; "Biological Criteria: National Program Guidance for Surface Waters", dated April 1990.

We are focusing on water quality standards for wetlands to ensure that provisions of the Clean Water Act currently applied to other surface waters are also being applied to wetlands. The document focuses on those elements of water quality standards that can be developed now using the overall structure of the water quality standards program and existing information and data sources related to wetlands. Periodically, our offices will provide additional information and support to the Regions and States through workshops and additional documents. We encourage you to let us know your needs as you begin developing wetlands standards. If you have any questions concerning this document, please contact us or have your staff contact Bob Shippen in OWRS (FTS-475-7329) or Doreen Robb in OWP (FTS-245-3906).

Attachment

cc: LaJuana Wilcher
Robert Wayland

EXECUTIVE SUMMARY

Background

This document provides program guidance to States on how to ensure effective application of water quality standards (WQS) to wetlands. This guidance reflects the level of achievement EPA expects the States to accomplish by the end of FY 1993, as defined in the Agency Operating Guidance, FY 1991, Office of Water. The basic requirements for applying State water quality standards to wetlands include the following:

- Include wetlands in the definition of "State waters."
- Designate uses for all wetlands.
- Adopt aesthetic narrative criteria (the "free froms") and appropriate numeric criteria for wetlands.
- Adopt narrative biological criteria for wetlands.
- Apply the State's antidegradation policy and implementation methods to wetlands.

Water quality standards for wetlands are necessary to ensure that the provisions of the Clean Water Act (CWA) applied to other surface waters are also applied to wetlands. Although Federal regulations implementing the CWA include wetlands in the definition of "waters of the U.S." and therefore require water quality standards, a number of States have not developed WQS for wetlands and have not included wetlands in their definitions of "State waters." Applying water quality standards to wetlands is part of an overall effort to protect and enhance the Nation's wetland resources and provides a regulatory basis for a variety of programs to meet this goal. Standards provide the foundation for a broad range of water quality management activities including, but not limited to, monitoring under Section 305(b), permitting under Sections 402 and 404, water quality certification under Section 401, and the control of NPS pollution under Section 319.

With the issuance of this guidance, EPA proposes a two- phased approach for the development of WQS for wetlands. Phase 1 activities presented in this guidance include the development of WQS elements for wetlands based upon existing information and science to be implemented within the next triennium. Phase 2 involves the further refinement of these basic elements using new science and program developments. The development of WQS for all surface waters is an iterative process.

Definition

The first and most important step in applying water quality standards to wetlands is ensuring that wetlands are legally included in the scope of States' water quality standards programs. States may accomplish this by adopting a regulatory definition of "State waters" at least as inclusive as the Federal definition of waters of the U.S." and by adopting an appropriate definition for "wetlands." States may also need to remove or modify regulatory language that explicitly or implicitly limits the authority of water quality standards over wetlands.

Use Designation

At a minimum, all wetlands must have uses designated that meet the goals of Section 101(a)(2) of the CWA by providing for the protection and propagation of fish, shellfish, and wildlife and for recreation in and on the water, unless the results of a use attainability analysis (UAA) show that the CWA Section 101(a)(2) goals cannot be achieved. When designating uses for wetlands, States may choose to use their existing general and water-specific classification systems, or they may set up an entirely different system for wetlands reflecting their unique functions. Two basic pieces of information are useful in classifying wetland uses: (1) the structural types of wetlands and (2) the functions and values associated with such types of wetlands. Generally, wetland functions directly relate to the physical, chemical, and biological integrity of wetlands. The protection of these functions through water quality standards also may be needed to attain the uses of waters adjacent to, or downstream of, wetlands.

Criteria

The Water Quality Standards Regulation (40 CFR 131.11(a)(1)) requires States to adopt criteria sufficient to protect designated uses that may include general statements (narrative) and specific numerical values (i.e., concentrations of contaminants and water quality characteristics). Most State water quality standards already contain many criteria for various water types and designated use classes that may be applicable to wetlands.

Narrative criteria are particularly important in wetlands, since many wetland impacts cannot be fully addressed by numeric criteria. Such impacts may result from the discharge of chemicals for which there are no numeric criteria in State standards, nonpoint sources, and activities that may affect the physical and/or biological, rather than the chemical, aspects of water quality (e.g., discharge of dredged and fill material). Narratives should be written to protect the most sensitive designated use and to support existing uses under State antidegradation policies. In addition to other narrative criteria, narrative biological criteria provide a further basis for managing a broad range of activities that impact the biological integrity of wetlands and other surface waters, particularly physical and hydrologic modifications. Narrative biological criteria are general statements of attainable or attained conditions of biological integrity and water quality for a given use designation. EPA has published national guidance on developing biological criteria for all surface waters.

Numeric criteria are specific numeric values for chemical constituents, physical parameters, or biological conditions that are adopted in State standards. Human health water quality criteria are based on the toxicity of a contaminant and the amount of the contaminant consumed through ingestion of water and fish regardless of the type of water. Therefore, EPA's chemical-specific human health criteria are directly applicable to wetlands. EPA also develops chemical-specific numeric criteria recommendations for the protection of freshwater and saltwater aquatic life. The numeric aquatic life criteria, although not designed specifically for wetlands, were designed to be protective of aquatic life and are generally applicable to most wetland types. An exception to this are pH-dependent criteria, such as ammonia and pentachlorophenol, since wetland pH may be outside the normal range of 6.5-9.0. As in other waters, natural water quality characteristics in some wetlands may be outside the range established for uses designated in State standards. These water quality characteristics may require the development of criteria that reflect the natural background conditions in a specific wetland or wetland type. Examples of some of the wetland characteristics that may fall into this category are dissolved oxygen, pH, turbidity, color, and hydrogen sulfide.

Antidegradation

The antidegradation policies contained in all State standards provide a powerful tool for the protection of wetlands and can be used by States to regulate point and nonpoint source discharges to wetlands in the same way as other surface waters. In conjunction with beneficial uses and narrative criteria, antidegradation can be used to address impacts to wetlands that cannot be fully addressed by chemical criteria, such as physical and hydrologic modifications. With the inclusion of wetlands as "waters of the State," State antidegradation policies and their implementation methods will apply to wetlands in the same way as other surface waters. State antidegradation policies should provide for the protection of existing uses in wetlands and the level of water quality necessary to protect those uses in the same manner as provided for other surface waters; see Section 131.12(a)(1) of the WQS regulation. In the case of fills, EPA interprets protection of existing uses to be met if there is no significant degradation as defined according to the Section 404(b)(1) guidelines. State antidegradation policies also provide special protection for outstanding natural resource waters.

Implementation

Implementing water quality standards for wetlands will require a coordinated effort between related Federal and State agencies and programs. Many States have begun to make more use of CWA Section 401 certification to manage certain activities that impact their wetland resources on a physical and/or biological basis rather than just chemical impacts. Section 401 gives the States the authority to grant, deny, or condition certification of Federal permits or licenses that may result in a discharge to "waters of the U.S." Such action is taken by the State to ensure compliance with various provisions of the CWA, including the State's water quality standards. Violation of water quality standards is often the basis for denials or conditioning through Section 401 certification.

Natural wetlands are nearly always "waters of the U.S." and are afforded the same level of protection as other surface waters with regard to standards and minimum wastewater treatment requirements. Water quality standards for wetlands can prevent the misuse and overuse of natural wetlands for treatment through adoption of proper uses and criteria and application of State antidegradation policies. The Water Quality Standards Regulation (40 CFR 131.10(a)) states that, "in no case shall a State adopt waste transport or waste assimilation as a designated use for any 'waters of the U.S.'." Certain activities involving the discharge of pollutants to wetlands may be permitted; however, as with other surface waters, the State must ensure, through ambient monitoring, that permitted discharges to wetlands preserve and protect wetland functions and values as defined in State water quality standards. For municipal discharges to natural wetlands, a minimum of secondary treatment is required, and applicable water quality standards for the wetland and adjacent waters must be met. EPA anticipates that the policy for stormwater discharges to wetlands will have some similarities to the policies for municipal wastewater discharges to wetlands.

Many wetlands, through their assimilative capacity for nutrients and sediment, also serve an important water quality control function for nonpoint source pollution effects on waters adjacent to, or downstream of, the wetlands. Section 319 of the CWA requires the States to complete assessments of nonpoint source (NPS) impacts to State waters, including wetlands, and to prepare management programs to control NPS impacts. Water quality standards for wetlands can form the basis for these assessments and management programs for wetlands.

In addition, States can address physical and hydrological impacts on wetland quality through the application of narrative criteria to protect existing uses and through application of their antidegradation policies. The States should provide a linkage in their water quality standards to the determination of "significant degradation" as required under EPA guidelines (40 CFR 230.10(c)) and other applicable State laws affecting the disposal of dredged or fill materials in wetlands.

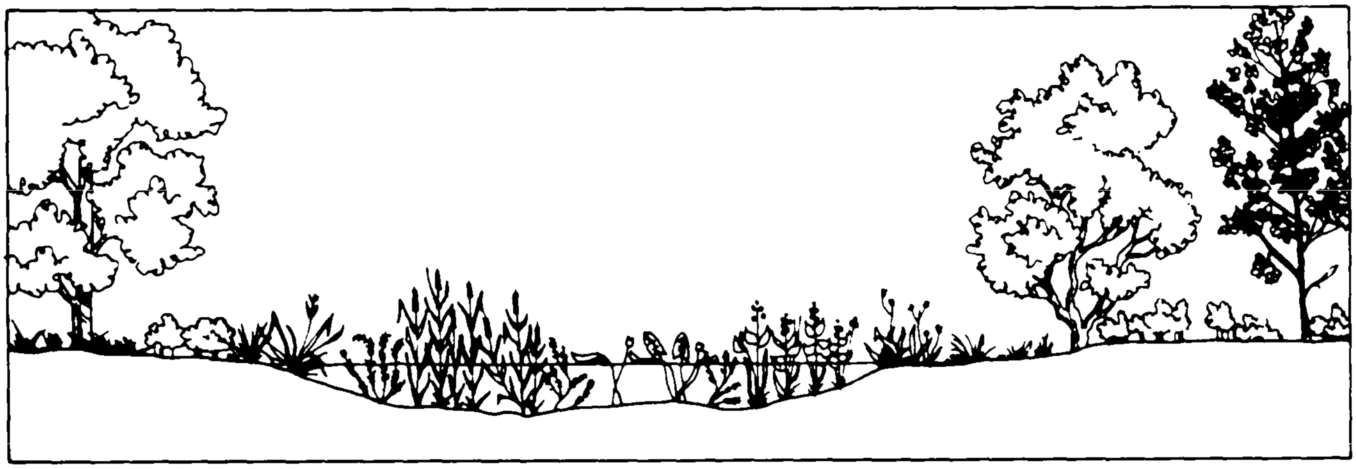
Finally, water quality management activities, including the permitting of wastewater and stormwater discharges, the assessment and control of NPS pollution, and waste disposal activities (sewage sludge, CERCLA, RCRA) require sufficient monitoring to ensure that the designated and existing uses of "waters of the U.S." are maintained and protected. The inclusion of wetlands in water quality standards provides the basis for conducting both wetland-specific and status and trend monitoring of State wetland resources. Monitoring of activities impacting specific wetlands may include several approaches, including biological measurements (i.e., plant, macroinvertebrate, and fish), that have shown promise for monitoring stream quality. The States are encouraged to develop and test the use of biological indicators.

Future Directions

Development of narrative biological criteria are included in the first phase of the development of water quality standards for wetlands. The second phase involves the implementation of numeric biological criteria. This effort requires the detailed evaluation of the components of wetland communities to determine the structure and function of unimpaired wetlands. Wetlands are important habitats for wildlife species. It is therefore also important to consider wildlife in developing criteria that protect the functions and values of wetlands. During the next 3 years, the Office of Water Regulations and Standards is reviewing aquatic life water quality criteria to determine whether adjustments in the criteria and/or alternative forms of criteria (e.g., tissue concentration criteria) are needed to adequately protect wildlife species using wetland resources. EPA's Office of Water Regulations and Standards is also developing guidance for EPA and State surface water monitoring programs that will be issued by the end of FY 1990. Other technical guidance and support for the development of State water quality standards will be forthcoming from EPA in the next triennium.

Chapter 1.0

Introduction



Our understanding of the many benefits that wetlands provide has evolved rapidly over the last 20 to 30 years. Recently, programs have been developed to restore and protect wetland resources at the local, State, and Federal levels of government. At the Federal level, the President of the United States established the goal of "no net loss" of wetlands, adapted from the National Wetlands Policy Forum recommendations (The Conservation Foundation 1988). Applying water quality standards to wetlands is part of an overall effort to protect the Nation's wetland resources and provides a regulatory basis for a variety of programs for managing wetlands to meet this goal.

As the link between land and water, wetlands play a vital role in water quality management programs. Wetlands provide a wide array of functions including shoreline stabilization, nonpoint source runoff filtration, and erosion control, which directly benefit adjacent and downstream waters. In addition, wetlands provide important biological habitat, including nursery areas for aquatic life and wildlife, and other benefits such as groundwater recharge and recreation. Wetlands comprise a wide variety of aquatic vegetated systems including, but not limited to, sloughs, prairie potholes, wet meadows, bogs, fens, vernal pools, and marshes. The basic elements of water quality standards (WQS), including designated uses, criteria, and an antidegradation policy, provide a sound legal basis for protecting wetland resources through State water quality management programs.

Water quality standards traditionally have been applied to waters such as rivers, lakes, estuaries, and oceans, and have been applied tangentially, if at all, to wetlands by applying the same uses and criteria to wetlands as to adjacent perennial waters. Isolated wetlands not directly associated with perennial waters generally have not been addressed in State water quality standards. A recent review of State water quality standards (USEPA 1989d) shows that only half of the States specifically refer to wetlands, or use similar terminology, in their water quality standards. Even where wetlands are referenced, standards may not be tailored to reflect the unique characteristics of wetlands.

Water quality standards specifically tailored to wetlands provide a consistent basis for the development of policies and technical procedures for managing activities that impact wetlands. Such water quality standards provide the goals for Federal and State programs that regulate discharges to wetlands, particularly those under CWA Sections 402 and 404 as well as other regulatory programs (e.g., Sections 307, 318, and 405) and nonregulatory programs (e.g., Sections 314, 319, and 320). In addition, standards play a critical role in the State 401 certification process by providing the basis for approving, conditioning, or denying Federal permits and licenses, as appropriate. Finally, standards provide a benchmark against which to assess the many activities that impact wetlands.

1.1 Objectives

The objective of this document is to assist States in applying their water quality standards regulations to wetlands in accordance with the *Agency Operating Guidance* (USEPA 1990a), which states:

> *By September 30, 1993, States and qualified Indian Tribes must adopt narrative water quality standards that apply directly to wetlands. Those Standards shall be established in accordance with either the National Guidance, Water Quality Standards for Wetlands...or some other scientifically valid method. In adopting water quality standards for wetlands, States and qualified Indian Tribes, at a minimum, shall: (1) define wetlands as "State waters"; (2) designate uses that protect the structure and function of wetlands; (3) adopt aesthetic narrative criteria (the "free froms") and appropriate numeric criteria in the standards to protect the designated uses; (4) adopt narrative biological criteria in the standards; and (5) extend the antidegradation policy and implementation methods to wetlands. Unless results of a use attainability analysis show that the section 101(a) goals cannot be achieved, States and qualified Indian Tribes shall designate uses for wetlands that provide for the protection of fish, shellfish, wildlife, and recreation. When extending the antidegradation policy and implementation methods to wetlands, consideration should be given to designating critical wetlands as Outstanding National Resource Waters. As necessary, the antidegradation policy should be revised to reflect the unique characteristics of wetlands.*

This level of achievement is based upon existing science and information, and therefore can be completed within the FY 91-93 triennial review cycle.

Initial development of water quality standards for wetlands over the next 3 years will provide the foundation for the development of more detailed water quality standards for wetlands in the future based on further research and policy development (see Chapter 7.0.). Activities defined in this guidance are referred to as "Phase 1 activities," while those to be developed over the longer term are referred to as "Phase 2 activities." Developing water quality standards is an iterative process.

This guidance is not regulatory, nor is it designed to dictate specific approaches needed in State water quality standards. The document addresses the minimum requirements set out in the Operating Guidance, and should be used as a guide to the modifications that may be needed in State standards. EPA recognizes that State water quality standards regulations vary greatly from State to State, as do wetland resources. This guidance suggests approaches that States may wish to use and allows for State flexibility and innovation.

1.2 Organization

Each chapter of this document provides guidance on a particular element of Phase 1 wetland water quality standards that EPA expects States to undertake during the next triennial review period (i.e., by September 30, 1993). For each chapter, a discussion of what EPA considers to be minimally acceptable is followed by subsections providing information that may be used to meet, and go beyond, the minimum requirements during Phase 1. Documents referenced in this guidance provide further information on specific topics and may be obtained from the sources listed in the "References" section. The following paragraphs introduce each of the chapters of this guidance.

Most wetlands fall within the definition of "waters of the U.S." and thus require water quality standards. EPA expects States by the end of FY 1993 to include wetlands in their definition of "State waters" consistent with the Federal definition of "waters of the U.S." Guidance on the inclusion of wetlands in the definition of "State waters" is contained in Chapter 2.0.

The application of water quality standards to wetlands requires that States designate appropriate uses consistent with Sections 101(a)(2) and 303(c)(2) of the Clean Water Act (CWA). EPA expects States by the end of FY 1993 to establish designated uses for all wetlands. Discussion of designated uses is contained in Chapter 3.0.

The WQS regulation (40 CFR 131) requires States to adopt water quality criteria sufficient to protect designated uses. EPA expects the States, by the end of FY 1993, to adopt aesthetic narrative criteria (the "free froms"), appropriate numeric criteria, and narrative biological criteria for wetlands. Narrative criteria are particularly important for wetlands, since many activities may impact upon the physical and biological, as well as chemical, components of water quality. Chapter 4.0 discusses the application of narrative and numeric criteria to wetlands.

EPA also expects States to fully apply antidegradation policies and implementation methods to wetlands by the end of FY 1993. Antidegradation can provide a powerful tool for the protection of wetlands, especially through the requirement for full protection of existing uses as well as the States' option of designating wetlands as outstanding national resource waters. Guidance on the application of State antidegradation policies to wetlands is contained in Chapter 5.0.

Many State water quality standards contain policies affecting the application and implementation of water quality standards (e.g., variances, mixing zones). Unless otherwise specified, such policies are presumed to apply to wetlands in the same manner as to other waters of the State. States should consider whether such policies should be modified to reflect the characteristics of wetlands. Guidance on the implementation of water quality standards for wetlands is contained in Chapter 6.0.

Application of standards to wetlands will be an iterative process; both EPA and the States will refine their approach based on new scientific information as well as experience developed through State programs. Chapter 7.0 outlines Phase 2 wetland standards activities for which EPA is planning additional research and program development.

1.3 Legal Authority

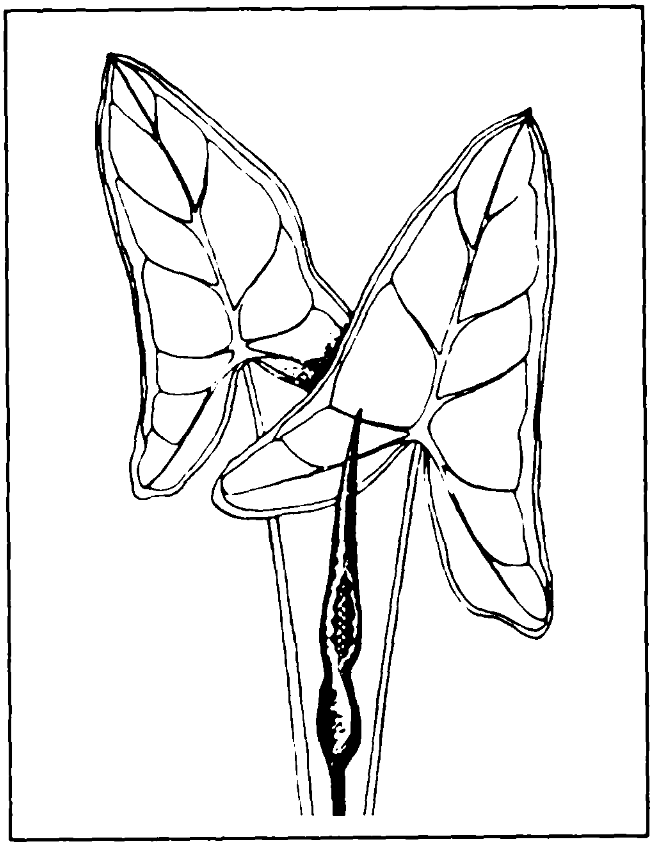
The Clean Water Act requires States to develop water quality standards, which include designated uses and criteria to support those uses, for "navigable waters." CWA Section 502(7) defines "navigable waters" as "waters of the U.S." "Waters of the U.S." are, in turn, defined in Federal regulations developed for the National Pollution Discharge Elimination System (40 CFR 122.2) and permits for the discharge of dredged or fill material (40 CFR 230.3 and 232.2). "Waters of the U.S." include waters subject to the ebb and flow of the tide; interstate waters (including interstate wetlands) and intrastate waters (including wetlands), the use, destruction, or degradation of which could affect interstate commerce; tributaries of the above; and wetlands adjacent to the above waters (other than waters which are themselves waters). See Appendix B for a complete definition.

> The term "wetlands" is defined in 40 CFR 232.2(r) as:
>
> *Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.*

This definition of "waters of the U.S.," which includes most wetlands, has been debated in Congress and upheld by the courts. In 1977, a proposal to delete CWA jurisdiction over most wetlands for the purpose of the Section 404 permit program was defeated in the Senate. The debate on the amendment shows a strong congressional awareness of the value of wetlands and the importance of retaining them under the statutory scheme. Various courts have also upheld the application of the CWA to wetlands. See, e.g., *United States* v. *Riverside Bayview Homes*, 474 U.S. 121 (1985); *United States* v. *Byrd*, 609 F.2d 1204 (7th Cir. 1979); *Avoyelles Sportsmen's League* v. *Marsh*, 715 F.2d 897 (5th

Cir. 1983); *United States* v. *Leslie Salt* [1990 decision]. The practical effect is to make nearly all wetlands "waters of the U.S."

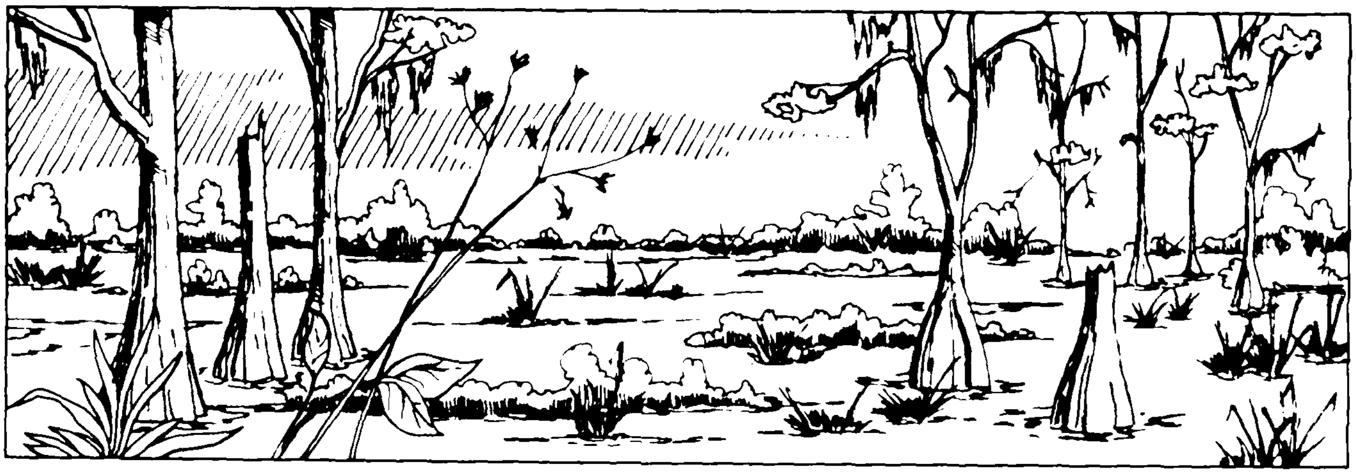
Created wastewater treatment wetlands[1] designed, built, and operated solely as wastewater treatment systems are generally not considered to be waters of the U.S. Water quality standards that apply to natural wetlands generally do not apply to such created wastewater treatment wetlands. Many created wetlands, however, are designed, built, and operated to provide, in addition to wastewater treatment, functions and values similar to those provided by natural wetlands. Under certain circumstances, such created multiple use wetlands may be considered waters of the U.S. and as such would require water quality standards. This determination must be made on a case-by-case basis, and may consider factors such as the size and degree of isolation of the created wetlands and other appropriate factors.



1 Different offices within EPA use different terminology (e.g., "create" or "constructed") to describe wastewater treatment wetlands. This terminology is evolving; for purposes of this guidance document, the terms are interchangeable in meaning.

Chapter 2.0

Inclusion of Wetlands in the Definition of State Waters



The first, and most important, step in applying water quality standards to wetlands is ensuring that wetlands are legally included in the scope of States' water quality standards programs. EPA expects States' water quality standards to include wetlands in the definition of "State waters" by the end of FY 1993. States may accomplish this by adopting a regulatory definition of "State waters" at least as inclusive as the Federal definition of "waters of the U.S." and by adopting an appropriate definition for "wetlands." For example, one State includes the following definitions in their water quality standards:

> *"Surface waters of the State"... means all streams,... lakes..., ponds, marshes, wetlands* or other waterways...
>
> *"Wetlands" means areas of land where the water table is at, near or above the land surface long enough each year to result in the formation of characteristically wet (hydric) soil types, and support the growth of water dependent (hydrophytic) vegetation. Wetlands include, but are not limited to, marshes, swamps, bogs, and other such low-lying areas.*

States may also need to remove or modify regulatory language that explicitly or implicitly limits the authority of water quality standards over wetlands. In certain instances, such as when water quality standards are statutory or where a statute defines or limits regulatory authority over wetlands, statutory changes may be needed.

The CWA does not preclude States from adopting, under State law, a more expansive definition of "waters of the State" to meet the goals of the act. Additional areas that could be covered include riparian areas, floodplains, vegetated buffer areas, or any other critical areas identified by the State. Riparian areas and floodplains are important and severely threatened ecosystems, particularly in the arid and semiarid West. Often it is technically difficult to separate, jurisdictionally, wetlands subject to the provisions of the CWA from other areas within the riparian or floodplain complex.

States may choose to include riparian or floodplain ecosystems as a whole in the definition of "waters of the State" or designate these areas for special protection in their water quality standards through several mechanisms, including definitions, use classifications, and antidegradation. For example, the regulatory definition of "waters of the State" in one State includes:

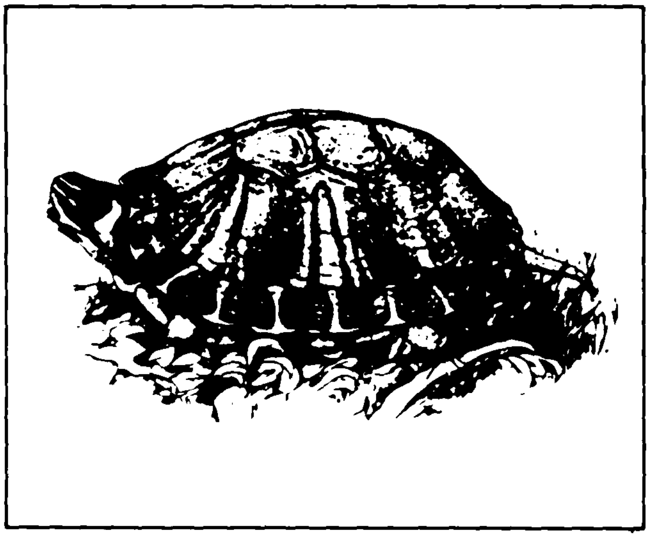
> *...The flood plain of free flowing waters determined by the Department...on the basis of the 100-year flood frequency.*

In another State, the definition of a use classification states:

> *This beneficial use is a combination of the characteristics of the watershed expressed in the water quality and the riparian area.*

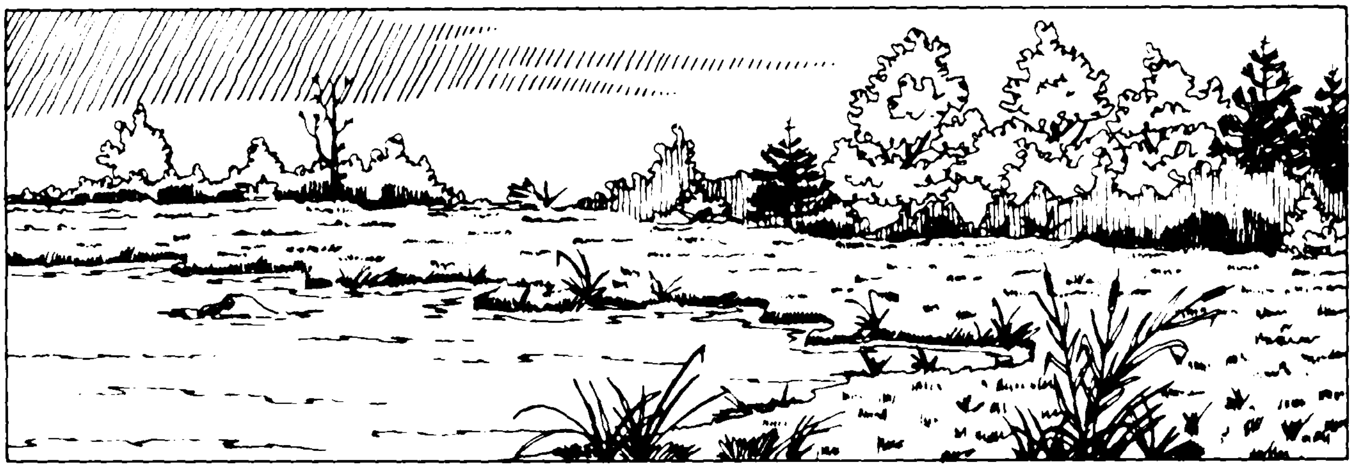
And in a third State, the antidegradation protection for high-quality waters provides that:

> These waters shall not be lowered in quality...unless it is determined by the commission that such lowering will not do any of the following:
>
> > *...[b]ecome injurious to the value or utility of riparian lands...*



Chapter 3.0

Use Classification



At a minimum, EPA expects States by the end of FY 1993 to designate uses for all wetlands, and to meet the same minimum requirements of the WQS regulation (40 CFR 131.10) that are applied to other waters. Uses for wetlands must meet the goals of Section 101(a)(2) of the CWA by providing for the protection and propagation of fish, shellfish, and wildlife and for recreation in and on the water, unless the results of a use attainability analysis (UAA) show that the CWA Section 101(a)(2) goals cannot be achieved. The Water Quality Standards Regulation (40 CFR 131.10(c)) allows for the designation of subcategories of a use, an activity that may be appropriate for wetlands. Pursuant to the WQS Regulation (40 CFR 131.10(i)), States must designate any uses that are presently being attained in the wetland. A technical support document is currently being developed by the Office of Water Regulations and Standards for conducting use attainability analyses for wetlands.

The propagation of aquatic life and wildlife is an attainable use in virtually all wetlands. Aquatic life protection need not refer only to year-round fish and aquatic life. Wetlands often provide valuable seasonal habitat for fish and other aquatic life, amphibians, and migratory bird reproduction and migration. States should ensure that aquatic life and wildlife uses are designated for wetlands even if a limited habitat is available or the use is attained only seasonally.

Recreation in and on the water, on the other hand, may not be attainable in certain wetlands that do not have sufficient water, at least seasonally. However, States are also encouraged to recognize and protect recreational uses that do not directly involve contact with water, e.g., hiking, camping, bird watching.

The WQS regulation requires a UAA wherever a State designates a use that does not include the uses specified in Section 101(a)(2) of the CWA; see 40 CFR Part 131.10(j). This need not be an onerous task for States when deciding whether certain recreational uses are attainable. States may conduct generic UAAs for entire classes or types of wetlands based on the demonstrations in 40 CFR Part 131.10(g)(2). States must, however, designate CWA goal uses wherever these are attainable, even where attainment may be seasonal.

When designating uses for wetlands, States may choose to use their existing general and water-specific classification systems, or they may set up an entirely different system for wetlands. Each of these approaches has advantages and disadvantages, as discussed below.

Some States stipulate that wetlands are designated for the same uses as the adjacent waters. States may also apply their existing general classification system to designate uses for specific wetlands or groups of wetlands. The advantage of these approaches is that they do not require States to expend additional effort to develop specific wetland uses, or determine specific functions and values, and can be generally used to designate the CWA goal uses for wetlands. However, since wetland attributes may be significantly different than those of other waters, States with general wetland use designations will need to review the uses for individual wetlands in more detail when assessing activities that may impair the specific "existing uses" (e.g., functions and values). In addition, the "adjacent" approach does not produce uses for "isolated" wetlands.

Owing to these differences in attributes, States should strongly consider adopting a separate use classification system for wetlands based on wetland type and/or beneficial use (function and value). This approach initially requires more effort in developing use categories (and specific criteria that may be needed for them), as well as in determining what uses to assign to specific wetlands or groups of wetlands. The greater the specificity in designating uses, however, the easier it is for States to justify regulatory controls to protect those uses. States may wish to designate beneficial uses for individually named wetlands, including outstanding wetlands (see Section 6.3), although this approach may be practical only for a limited number of wetlands. For the majority of their wetlands, States may wish to designate generalized uses for groups of wetlands based on region or wetland type.

Two basic pieces of information are useful in classifying wetland uses: (1) the structural types of wetlands; and (2) the functions and values associated with such types of wetlands. The functions and values of wetlands are often defined based upon structural type and location within the landscape or watershed. The understanding of the various wetland types within the State and their functions and values provides the basis for a comprehensive classification system applicable to all wetlands and all wetland uses. As with other waters, both general and waterbody-specific classifications may be needed to ensure that uses are appropriately assigned to all wetlands in the State. Appropriate and definitive use designations allow water quality standards to more accurately reflect both the "existing" uses and the States' goals for their wetland resources, and to allow standards to be a more powerful tool in protecting State wetlands. Sections 3.1 through 3.3 provide further information on wetland types, functions, and values, and how these can be used to designate uses for wetlands.

3.1 Wetland Types

A detailed understanding of the various wetland types within the State provides the basis for a comprehensive classification system. The classification system most often cited and used by Federal and State wetland permit programs was developed by Cowardin et al. (1979) for the U.S. Fish and Wildlife Service (FWS); see Figure 1. This system provides the basis for wetland-related activities within the FWS. The Cowardin system is hierarchical and thus can provide several levels of detail in classifying wetlands. The "System" and "Subsystem" levels of detail appear to be the most promising for water quality standards. The "Class" level may be useful for designating uses for specific wetlands or wetland types. Section 3.3 gives an example of how one State uses the Cowardin system to generate designated uses for wetlands.

Under the Emergency Wetlands Resources Act of 1986, the FWS is required to complete the mapping of wetlands within the lower 48 States by 1998 through the National Wetlands Inventory (NWI) and to assess the status of the nation's wetland resources every 10 years. The maps and status and trend reports may help States understand the extent of their wetlands and wetland types and ensure that all wetlands are assigned appropriate uses. To date, over 30,000 detailed 1:24,000 scale maps have been completed, covering approximately 60 percent of

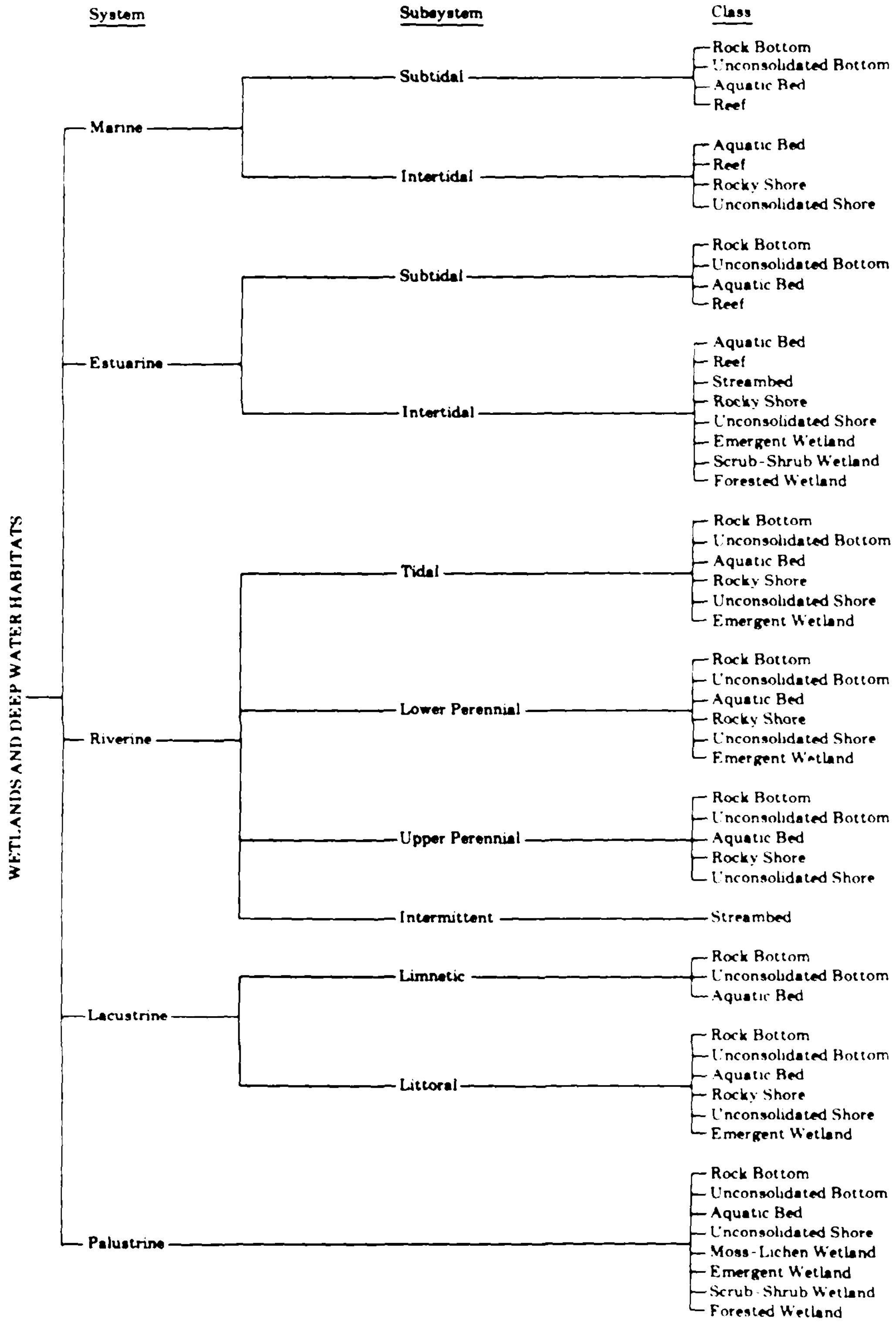


Figure 1. Classification hierarchy of wetlands and deepwater habitats, showing Systems, Subsystems, and Classes. The Palustrine System does not include deepwater habitats (from Cowardin et al., 1979).

the coterminous United States and 16 percent of Alaska[2]

In some States, wetland maps developed under the NWI program have been digitized and are available for use with geographic information systems (GIS). To date, more than 5,700 wetland maps representing 10.5 percent of the coterminous United States have been digitized. Statewide digital databases have been developed for New Jersey, Delaware, Illinois, Maryland, and Washington, and are in progress in Indiana and Virginia. NWI digital data files also are available for portions of 20 other States. NWI data files are sold at cost in 7.5-minute quadrangle units. The data are provided on magnetic tape in MOSS export, DLG3 optional, ELAS, and IGES formats[3]. Digital wetlands data may expedite assigning uses to wetlands for both general and wetland-specific FIC classifications.

The classification of wetlands may benefit from the use of salinity concentrations. The Cowardin classification system uses a salinity criterion of 0.5 ppt ocean-derived salinity to differentiate between estuarine and freshwater wetlands. Differences in salinity are reflected in the species composition of plants and animals. The use of salinity in the classification of wetlands may be useful in restricting activities that would alter the salinity of a wetland to such a degree that the wetland type would change. These activities include, for example, the construction of dikes to convert a saltwater marsh to a freshwater marsh or the dredging of channels that would deliver saltwater to freshwater wetlands.

3.2 Wetland Functions and Values

Many approaches have been developed for identifying wetland functions and values. Wetland evaluation techniques developed prior to 1983 have been summarized by Lonard and Clairain (1985), and EPA has summarized assessment methodologies developed since 1983 (see Appendix C). EPA has also developed guidance on the selection of a methodology for activities under the Section 404 program entitled *Draft Guidance to EPA Regional Offices on the Use of Advance Identification Authorities Under Section 404 of the Clean Water Act* (USEPA 1989a). States may develop their own techniques for assessing the functions and values of their wetlands.

General wetland functions that directly relate to the physical, chemical, and biological integrity of wetlands are listed below. The protection of these functions through water quality standards also may be needed to attain the uses of waters adjacent to, or downstream of, wetlands.

- Groundwater Recharge/Discharge
- Flood Flow Alteration
- Sediment Stabilization
- Sediment/Toxic Retention
- Nutrient Removal/Transformation
- Wildlife Diversity/Abundance
- Aquatic Diversity/Abundance
- Recreation

Methodologies that are flexible with regard to data requirements and include several levels of detail have the greatest potential for application to standards. One such methodology is the Wetland Evaluation Technique developed by Adamus, et al. (1987) for the U.S. Army Corps of Engineers and the

2 Information on the availability of draft and final maps may be obtained for the coterminous United States by calling 1-800-USA-MAPS or 703-860-6045 in Virginia. In Alaska, the number is 907-271-4159, and in Hawaii the number is 808-548-2861. Further information on the FWS National Wetlands Inventory (NWI) may be obtained from the FWS Regional Coordinators listed in Appendix D.

3 For additional information on digital wetland data contact: USFWS, National Wetlands Inventory Program, 9720 Executive Center Drive, Monroe Building, Suite 101, St. Petersburg, FL 33702; 813-893-3624, FTS 826-3624.

Department of Transportation. The Wetland Evaluation Technique was designed for conducting an initial rapid assessment of wetland functions and values in terms of social significance, effectiveness, and opportunity. Social significance assesses the value of a wetland to society in terms of its special designation, potential economic value, and strategic location. Effectiveness assesses the capability of a wetland to perform a function because of its physical, chemical, or biological characteristics. Opportunity assesses the [opportunity] of a wetland to perform a function to its level of capability. This assessment results in "high," "moderate," or "low" ratings for 11 wetland functions in the context of social significance, effectiveness, and opportunity. This technique also may be useful in identifying outstanding wetlands for protection under State antidegradation policies; see Section 5.3.

The FWS maintains a Wetlands Values Database that also may be useful in identifying wetland functions and in designating wetland uses. The data are keyed to the Cowardin-based wetland codes identified on the National Wetland Inventory maps. The database contains scientific literature on wetland functions and values. It is computerized and contains over 18,000 citations, of which 8,000 are annotated. For further information, contact the NWI Program (see Section 3.1) or the FWS National Ecology Research Center[4]. In addition, State wetland programs, EPA Regional wetland coordinators, and FWS Regional wetland coordinators can provide information on wetland functions and values on a State or regional basis; see Appendix D.

3.3 Designating Wetland Uses

The functions and values of specifically identified and named wetlands, including those identified within the State's water-specific classification system and outstanding national resource water (ONRW) category, may be defined using the Wetland Evaluation Technique or similar methodology. For the general classification of wetlands, however, States may choose to evaluate wetland function and values for all the wetlands within the State based on wetland type (using Cowardin (1979); see Figure 1). One State applies its general use classifications to different wetland types based on Cowardin's system level of detail as illustrated in Figure 2. Note that the State's uses are based on function, and the designation approach links specific wetland functions to a given wetland type. The State evaluates wetlands on a case-by-case basis as individual permit decisions arise to ensure that designated uses are being protected and have reflected existing uses.

4 USFWS; Wetlands Values Database, National Ecology Research Center, 4512 McMurray, Ft. Collins, CO 80522; 303-226-9407.

BENEFICIAL USE	WETLAND TYPE (Cowardin) MARINE	ESTUARINE	RIVERINE	LACUSTRINE	PALUSTRINE
Municipal and Domestic Supply	-	-	x	x	x
Agricultural Supply	-	x	x	x	x
Industrial Process Supply	-	x	o	o	-
Groundwater Recharge	x	x	x	x	x
Freshwater Replenishment	-	-	x	x	x
Navigation	x	x	x	x	x
Water Contact Recreation	x	x	x	x	x
Non-Contact Water Recreation	x	x	x	x	x
Ocean Commercial and Sport Fishing	x	x	-	-	-
Warm Fresh Water Habitat	-	-	x	x	x
Cold Fresh Water Habitat	-	-	x	x	x
Preservation of Areas of Special Biological Significance	-	-	-	-	-
Wildlife Habitat	x	x	x	x	x
Preservation of Rare and Endangered Species	x	x	x	x	x
Marine Habitat	x	x	-	-	-
Fish Migration	x	x	x	x	-
Shellfish Harvesting	x	x	x	-	-
Estuarine Habitat	-	x	-	-	-

x = existing beneficial use
o = potential beneficial use

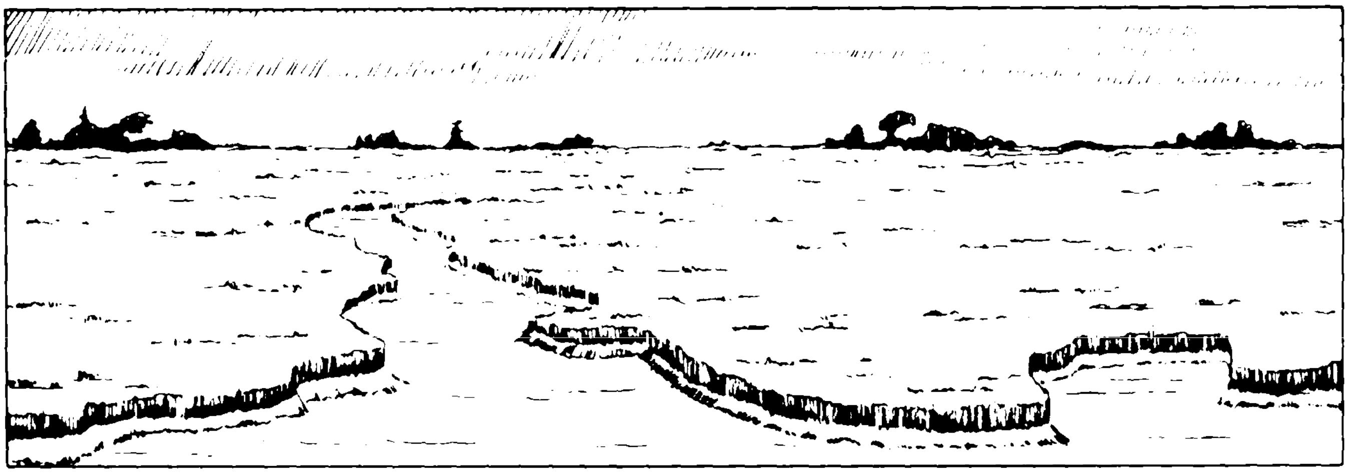
Figure 2. Example Existing and Potential Uses of Wetlands

Alternatively, a third method may use the location of wetlands within the landscape as the basis for establishing general functions and values applicable to all the wetlands within a defined region. EPA has developed a guidance entitled *Regionalization as a Tool for Managing Environmental Resources* (USEPA 1989c). The guidance illustrates how various regionalization techniques have been used in water quality management, including the use of the ecoregions developed by EPA's Office of Research and Development, to direct State water quality standards and monitoring programs. These approaches also may be useful in the classification of wetlands.

EPA's Office of Research and Development is currently refining a draft document that will provide useful information to States related to use classification methodologies (Adamus and Brandt - Draft). There are likely many other approaches for designating uses for wetlands, and the States are encouraged to develop comprehensive classification systems tailored to their wetland resources. As with other surface waters, many wetlands are currently degraded by natural and anthropogenic activities. The classification of wetlands should reflect the potential uses attainable for a particular wetland, wetland type, or class of wetland.

Chapter 4.0

Criteria



The Water Quality Standards Regulation (40 CFR 131.11(a)(1)) requires States to adopt criteria sufficient to protect designated uses. These criteria may include general statements (narrative) and specific numerical values (i.e., concentrations of contaminants and water quality characteristics). At a minimum, EPA expects States to apply aesthetic narrative criteria (the "free froms") and appropriate numeric criteria to wetlands and to adopt narrative biological criteria for wetlands by the end of FY 1993. Most State water quality standards already contain many criteria for various water types and designated use classes, including narrative criteria and numeric criteria to protect human health and freshwater and saltwater aquatic life, that may be applicable to wetlands.

In many cases, it may be necessary to use a combination of numeric and narrative criteria to ensure that wetland functions and values are adequately protected. Section 4.1 describes the application of narrative criteria to wetlands and Section 4.2 discusses application of numeric criteria for protection of human health and aquatic life.

4.1 Narrative Criteria

Narrative criteria are general statements designed to protect a specific designated use or set of uses. They can be statements prohibiting certain actions or conditions (e.g., "free from substances that produce undesirable or nuisance aquatic life") or positive statements about what is expected to occur in the water (e.g., "water quality and aquatic life shall be as it naturally occurs"). Narrative criteria are used to identify impacts on designated uses and as a regulatory basis for controlling a variety of impacts to State waters. Narrative criteria are particularly important in wetlands, since many wetland impacts cannot be fully addressed by numeric criteria. Such impacts may result from the discharge of chemicals for which there are no numeric criteria in State standards, from nonpoint sources, and from activities that may affect the physical and/or biological, rather than the chemical, aspects of water quality (e.g., discharge of dredged and fill material). The Water Quality Standards Regulation (40 CFR 131.11(b)) states that "States should...include narrative criteria in their standards where numeric criteria cannot be established or to supplement numeric criteria."

4.1.1 General Narrative Criteria

Narrative criteria within the water quality standards program date back to at least 1968 when five "free froms" were included in *Water Quality Criteria* (the Green Book), (FWPCA 1968). These "free froms" have been included as "aesthetic criteria" in EPA's most recent Section 304(a) criteria summary document, *Quality Criteria for Water - 1986* (USEPA 1987a). The narrative criteria from these documents state:

All waters [shall be] free from substances attributable to wastewater or other discharge that:

(1) *settle to form objectionable deposits;*

(2) *float as debris, scum, oil, or other matter to form nuisances;*

(3) *produce objectionable color, odor, taste, or turbidity;*

(4) *injure or are toxic or produce adverse physiological responses in humans, animals or plants; and*

(5) *produce undesirable or nuisance aquatic life.*

The *Water Quality Standards Handbook* (USEPA 1983b) recommends that States apply narrative criteria to all waters of the United States. If these or similar criteria are already applied to all State waters in a State's standards, the inclusion of wetlands in the definition of "waters of the State" will apply these criteria to wetlands.

4.1.2 Narrative Biological Criteria

Narrative biological criteria are general statements of attainable or attained conditions of biological integrity and water quality for a given use designation. Narrative biological criteria can take a number of forms. As a sixth "free from," the criteria could read "free from activities that would substantially impair the biological community as it naturally occurs due to physical, chemical, and hydrologic changes," or the criteria may contain positive statements about the biological community existing or attainable in wetlands.

Narrative biological criteria should contain attributes that support the goals of the Clean Water Act, which provide for the protection and propagation of fish, shellfish, and wildlife. Therefore, narrative criteria should include specific language about community characteristics that (1) must exist in a wetland to meet a particular designated aquatic life/wildlife use, and (2) are quantifiable. Supporting statements for the criteria should promote water quality to protect the most natural community associated with the designated use. Mechanisms should be established in the standard to address potentially conflicting multiple uses. Narratives should be written to protect the most sensitive designated use and to support existing uses under State antidegradation policies.

In addition to other narrative criteria, narrative biological criteria provide a further basis for managing a broad range of activities that impact the biological integrity of wetlands and other surface waters, particularly physical and hydrologic modifications. For instance, hydrologic criteria are one particularly important but often overlooked component to include in water quality standards to help maintain wetlands quality. Hydrology is the primary factor influencing the type and location of wetlands. Maintaining appropriate hydrologic conditions in wetlands is critical to the maintenance of wetland functions and values. Hydrologic manipulations to wetlands have occurred nationwide in the form of flow alterations and diversions, disposal of dredged or fill material, dredging of canals through wetlands, and construction of levees or dikes. Changes in base flow or flow regime can severely alter the plant and animal species composition of a wetland, and destroy the entire wetland system if the change is great enough. States should consider the establishment of criteria to regulate hydrologic alterations to wetlands. One State has adopted the following language and criteria to maintain and protect the natural hydrologic conditions and values of wetlands:

Natural hydrological conditions necessary to support the biological and physical characteristics naturally present in wetlands shall be protected to prevent significant adverse impacts on:

(1) *Water currents, erosion or sedimentation patterns;*

(2) *Natural water temperature variations;*

(3) *The chemical, nutrient and dissolved oxygen regime of the wetland;*

(4) *The normal movement of aquatic fauna;*

(5) *The pH of the wetland; and*

(6) *Normal water levels or elevations.*

One source of information for developing more quantifiable hydrologic criteria is the Instream Flow Program of the U.S. Fish and Wildlife Service, which can provide technical guidance on the minimum flows necessary to attain various water uses.

Narrative criteria, in conjunction with antidegradation policies, can provide the basis for determining the impacts of activities (such as hydrologic modifications) on designated and existing uses. EPA has published national guidance on developing biological criteria for all surface waters (USEPA 1990b). EPA's Office of Research and Development also has produced a literature synthesis of wetland biomonitoring data on a State-by-State basis, which is intended to support the development of narrative biological criteria (Adamus and Brandt - Draft).

4.2 Numeric Criteria

Numeric criteria are specific numeric values for chemical constituents, physical parameters, or biological conditions that are adopted in State standards. These may be values not to be exceeded (e.g., toxics), values that must be exceeded (e.g., dissolved oxygen), or a combination of the two (e.g., pH). As with all criteria, numeric criteria are adopted to protect one or more designated uses. Under Section 304(a) of the Clean Water Act, EPA publishes numeric national criteria recommendations designed to protect aquatic organisms and human health. These criteria are summarized in *Quality Criteria for Water* - 1986 (USEPA 1987a). These criteria serve as guidelines from which States can develop their own numeric criteria, taking into account the particular uses designated by the State.

4.2.1 Numeric Criteria - Human Health

Human health water quality criteria are based on the toxicity of a contaminant and the amount of the contaminant consumed through ingestion of water and fish regardless of the type of water. Therefore, EPA's chemical-specific human health criteria are directly applicable to wetlands. A summary of EPA human health criteria recommendations is contained in *Quality Criteria for Water - 1986.*

Few wetlands are used directly for drinking water supplies. Where drinking water is a designated or existing use for a wetland or for adjacent waters affected by the wetland, however, States must provide criteria sufficient to protect human health based on water consumption (as well as aquatic life consumption if appropriate). When assessing the potential for water consumption, States should also evaluate the wetland's groundwater recharge function to assure protection of drinking water supplies from that source as well.

The application of human health criteria, based on consumption of aquatic life, to wetlands is a function of the level of detail in the States' designated uses. If all wetlands are designated under the State's general "aquatic life/wildlife" designation, consumption of that aquatic life is assumed to be an included use and the State's human health criteria based on consumption of aquatic life will apply throughout. However, States that adopt a more detailed use classification system for wetlands (or wish to derive site-specific human health criteria for wetlands) may wish to selectively apply human health criteria to those wetlands where consumption of aquatic life is designated or likely to occur (note that a UAA will be required where CWA goal uses are not designated). States may also wish to adjust the exposure assumptions used in deriving human health criteria. Where it is known that exposure to individuals at a certain site, or within a certain category of wetland, is likely to be different from the assumed exposure underlying the States' criteria, States may wish to consider a reasonable estimate of the actual exposure and take this estimate into account when calculating the criteria for the site.

4.2.2 Numeric Criteria - Aquatic Life

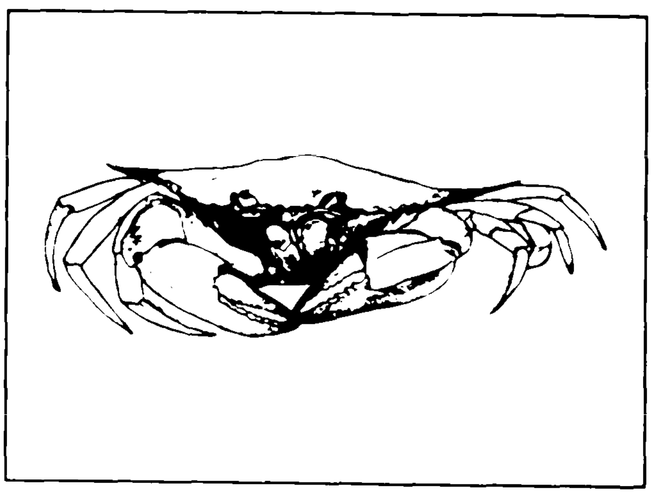
EPA develops chemical-specific numeric criteria recommendations for the protection of freshwater and saltwater aquatic life. These criteria may be divided into two basic categories: (1) chemicals that cause toxicity to aquatic life such as metals, ammonia, chlorine, and organics; and (2) other water quality characteristics such as dissolved oxygen, alkalinity, salinity, pH, and temperature. These criteria are currently applied directly to a broad range of surface waters in State standards, including lakes, impoundments, ephemeral and perennial rivers and streams, estuaries, the oceans, and in some instances, wetlands. A summary of EPA's aquatic life criteria recommendations is published in *Quality Criteria for Water - 1986*. The numeric aquatic life criteria, although not designed specifically for wetlands, were designed to be protective of aquatic life and are generally applicable to most wetland types.

EPA's aquatic life criteria are most often based upon toxicological testing under controlled conditions in the laboratory. The EPA guidelines for the development of such criteria (Stephan et al., 1985) require the testing of plant, invertebrate, and fish species. Generally, these criteria are supported by toxicity tests on invertebrate and early life stage fish commonly found in many wetlands. Adjustments based on natural conditions, water chemistry, and biological community conditions may be appropriate for certain criteria as discussed below. EPA's Office of Research and Development is currently finalizing a draft document that provides additional technical guidance on this topic, including site-specific adjustments of criteria (Hagley and Taylor - Draft).

As in other waters, natural water quality characteristics in some wetlands may be outside the range established for uses designated in State standards. These water quality characteristics may require the development of criteria that reflect the natural background conditions in a specific wetland or wetland type. States routinely set criteria for specific waters based on natural conditions. Examples of some of the wetland characteristics that may fall into this category are dissolved oxygen, pH, turbidity, color, and hydrogen sulfide.

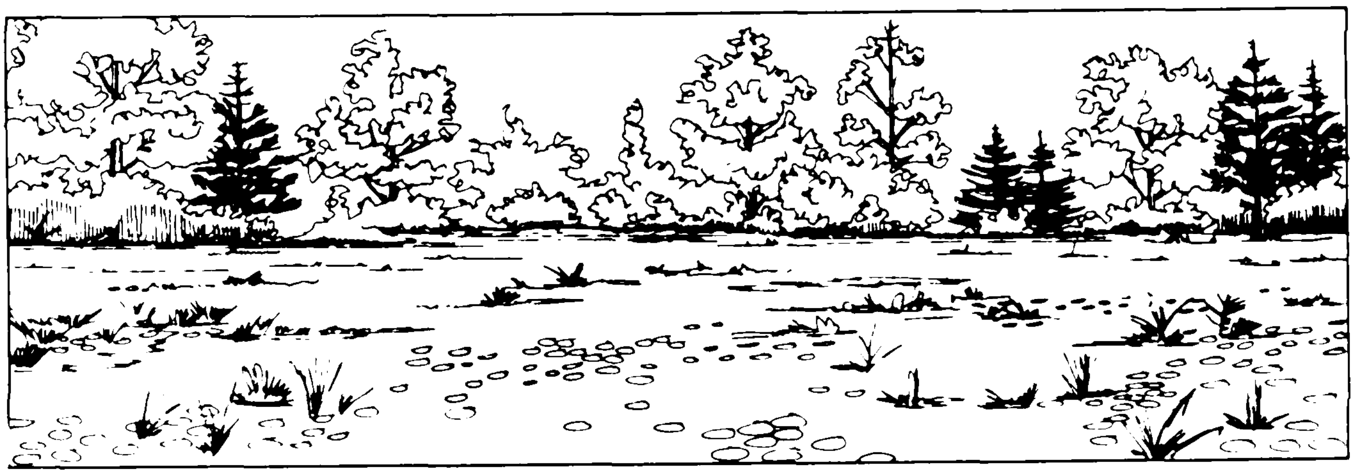
Many of EPA's aquatic life criteria are based on equations that take into account salinity, pH, temperature and/or hardness. These may be directly applied to wetlands in the same way as other water types with adjustments in the criteria to reflect these water quality characteristics. However, two national criteria that are pH dependent, ammonia and pentachlorophenol, present a different situation. The pH in some wetlands may be outside the pH range of 6.5-9.0 units for which these criteria were derived. It is recommended that States conduct additional toxicity testing if they wish to derive criteria for ammonia and pentachlorophenol outside the 6.5-9.0 pH range, unless data are already available.

States may also develop scientifically defensible site-specific criteria for parameters whose Statewide values may be inappropriate. Site-specific adjustments may be made based on the water quality and biological conditions in a specific water, or in waters within a particular region or ecoregion. EPA has developed guidance on the site-specific adjustment of the national criteria (USEPA 1983b). These methods are applicable to wetlands and should be used in the same manner as States use them for other waters. As defined in the Handbook, three procedures may be used to develop site-specific criteria: (1) the recalculation procedures, (2) the indicator species procedures, and (3) the resident species procedures. These procedures may be used to develop site-specific numeric criteria for specific wetlands or wetland types. The recalculation procedure is used to make adjustments based upon differences between the toxicity to resident organisms and those used to derive national criteria. The indicator species procedure is used to account for differences in the bioavailability and/or toxicity of a contaminant based upon the physical and chemical characteristics of site water. The resident species procedure accounts for differences in both species sensitivity and water quality characteristics



Chapter 5.0

Antidegradation



The antidegradation policies contained in all State standards provide a powerful tool for the protection of wetlands and can be used by States to regulate point and nonpoint source discharges to wetlands in the same way as other surface waters. In conjunction with beneficial uses and narrative criteria, antidegradation can be used to address impacts to wetlands that cannot be fully addressed by chemical criteria, such as physical and hydrologic modifications. The implications of antidegradation to the disposal of dredged and fill material are discussed in Section 5.1 below. At a minimum, EPA expects States to fully apply their antidegradation policies and implementation methods to wetlands by the end of FY 1993. No changes to State policies are required if they are fully consistent with the Federal policy. With the inclusion of wetlands as "waters of the State," State antidegradation policies and their implementation methods will apply to wetlands in the same way as other surface waters. The WQS regulation describes the requirements for State antidegradation policies, which include full protection of existing uses (functions and values), maintenance of water quality in high-quality waters, and a prohibition against lowering water quality in outstanding national resource waters. EPA guidance on the implementation of antidegradation policies is contained in the *Water Quality Standards Handbook* (USEPA 1983b) and *Questions and Answers on: Antidegradation* (USEPA 1985a).

5.1 Protection of Existing Uses

State antidegradation policies should provide for the protection of existing uses in wetlands and the level of water quality necessary to protect those uses in the same manner as for other surface waters; see Section 131.12(a)(1) of the WQS regulation. The existing use can be determined by demonstrating that the use or uses have actually occurred since November 28, 1975, or that the water quality is suitable to allow the use to be attained. This is the basis of EPA's antidegradation policy and is important in the wetland protection effort. States, especially those that adopt less detailed use classifications for wetlands, will need to use the existing use protection in their antidegradation policies to ensure protection of wetland values and functions.

Determination of an existing aquatic life and wildlife use may require physical, chemical, and biological evaluations through a waterbody survey and assessment. Waterbody survey and assessment guidance may be found in three volumes entitled *Technical Support Manual for Conducting Use Attainability Analyses* (USEPA 1983b, 1984a, 1984b). A technical support manual for conducting use attainability analyses for wetlands is currently under development by the Office of Water Regulations and Standards.

In the case of wetland fills, EPA allows a slightly different interpretation of existing uses under the antidegradation policy. This interpretation has been addressed in the answer to question no. 13 in *Questions and Answers on: Antidegradation* (USEPA 1985a), and is presented below:

> *Since a literal interpretation of the antidegradation policy could result in preventing the issuance of any wetland fill permit under Section 404 of the Clean Water Act, and it is logical to assume that Congress intended some such permits to be granted within the framework of the Act, EPA interprets 40 CFR 131.12(a)(l) of the antidegradation policy to be satisfied with regard to fills in wetlands if the discharge did not result in "significant degradation" to the aquatic ecosystem as defined under Section 230.10(c) of the Section 404(b)(l) guidelines. If any wetlands were found to have better water quality than "fishable/swimmable," the State would be allowed to lower water quality to the no significant degradation level as long as the requirements of Section 131.12(a)(2) were followed. As for the ONRW provision of antidegradation (131.12(a)(3)), there is no difference in the way it applies to wetlands and other waterbodies.*

The Section 404(b)(1) Guidelines state that the following effects contribute to significant degradation, either individually or collectively:

> *...significant adverse effects on (1) human health or welfare, including effects on municipal water supplies, plankton, fish, shellfish, wildlife, and special aquatic sites (e.g., wetlands); (2) on the life stages of aquatic life and other wildlife dependent on aquatic ecosystems, including the transfer, concentration or spread of pollutants or their byproducts beyond the site through biological, physical, or chemical process; (3) on ecosystem diversity, productivity and stability, including loss of fish and wildlife habitat or loss of the capacity of a wetland to assimilate nutrients, purify water or reduce wave energy; or (4) on recreational, aesthetic, and economic values.*

These Guidelines may be used by States to determine "significant degradation" for wetland fills. Of course, the States are free to adopt stricter requirements for wetland fills in their own antidegradation policies, just as they may adopt any other requirements more stringent than Federal law requires. For additional information on the linkage between water quality standards and the Section 404 program, see Section 6.2 of this guidance.

5.2 Protection of High-Quality Wetlands

State antidegradation policies should provide for water quality in "high quality wetlands" to be maintained and protected, as prescribed in Section 131.12(a)(2) of the WQS regulation. State implementation methods requiring alternatives analyses, social and economic justifications, point and nonpoint source control, and public participation are to be applied to wetlands in the same manner they are applied to other surface waters.

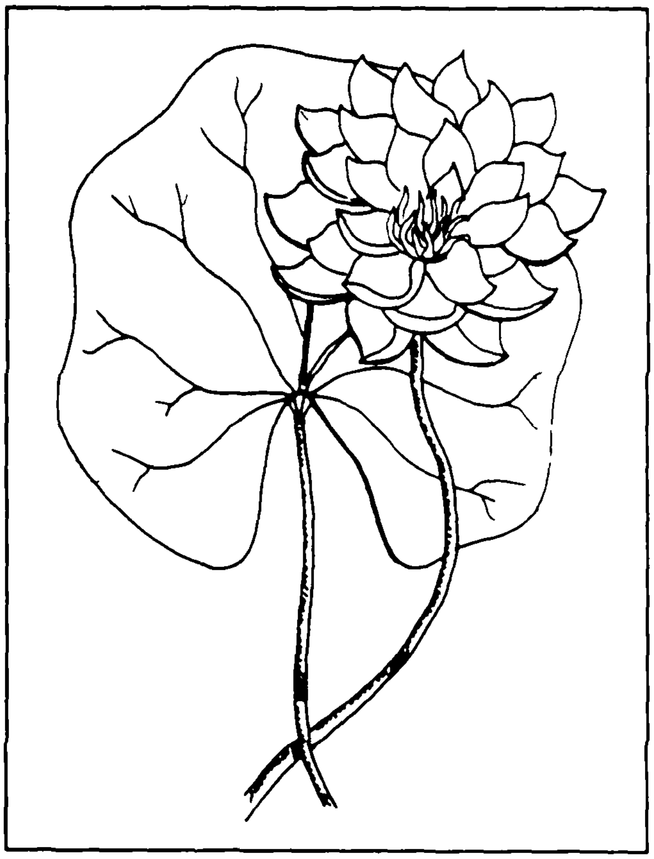
5.3 Protection of Outstanding Wetlands

Outstanding national resource waters (ONRW) designations offer special protection (i.e., no degradation) for designated waters, including wetlands. These are areas of exceptional water quality or recreational/ecological significance. State antidegradation policies should provide special protection to wetlands designated as outstanding national resource waters in the same manner as other surface waters; see Section 131.12(a)(3) of the WQS regulation and EPA guidance *Water Quality Standards Handbook* (USEPA 1983b), and *Questions and Answers on: Antidegradation* (USEPA 1985a). Activities that might trigger a State analysis of a wetland for possible designation as an ONRW are no different for wetlands than for other waters.

The following list provides general information on wetlands that are likely candidates for protection as ONRWs. *It also may be used to identify specific* wetlands for use designation under the State's wetland classification system; see Chapter 4.0. Some of these information sources are discussed in greater detail in EPA's guidance entitled *Wetlands and Section 401 Certification: Opportunities and Guidelines for States and Eligible Indian Tribes* (USEPA 1989f); see Section 6.1.

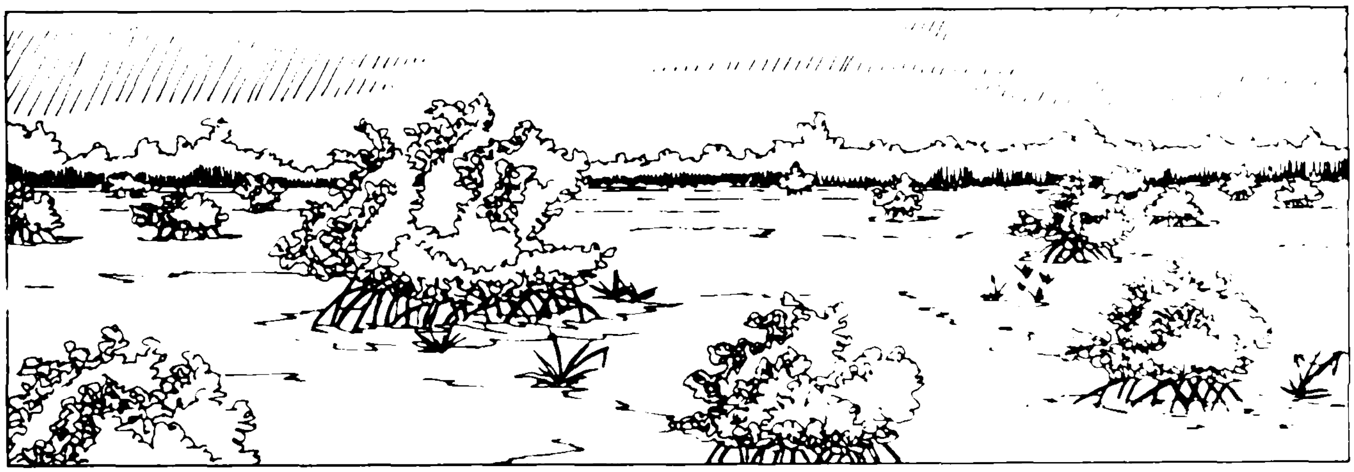
- Parks, wildlife management areas, refuges, wild and scenic rivers, and estuarine sanctuaries;
- Wetlands adjacent to ONRWs or other high-quality waters (e.g., lakes, estuaries shellfish beds);
- Priority wetlands identified under the Emergency Wetlands Resources Act of 1986 through Statewide Outdoor Recreation Plans (SORP) and Wetland Priority Conservation Plans;
- Sites within joint venture project areas under the North American Waterfowl Management Plan;
- Sites under the Ramsar (Iran) Treaty on Wetlands of International Importance;
- Biosphere reserve sites identified as part of the "Man and the Biosphere" Program sponsored by the United Nations;
- Natural heritage areas and other similar designations established by the State or private organizations (e.g., Nature Conservancy); and
- Priority wetlands identified as part of comprehensive planning efforts conducted at the local, State, Regional, or Federal levels of government; e.g., Advance Identification (ADID) program under Section 404 and Special Area Management Plans (SAMPs) under the 1980 Coastal Zone Management Act.

The Wetland Evaluation Technique; Volume II: Methodology (Adamus et al., 1987) provides additional guidance on the identification of wetlands with high ecological and social value; see Section 3.2.



Chapter 6.0

Implementation



Implementing water quality standards for wetlands will require a coordinated effort between related Federal and State agencies and programs. In addition to the Section 401 certification for Federal permits and licenses, standards have other potential applications for State programs, including landfill siting, fish and wildlife management and aquisition decisions, and best management practices to control nonpoint source pollution. Many coastal States have wetland permit programs, coastal zone management programs, and National Estuary Programs; and the development of water quality standards should utilize data, information and expertise from these programs. For all States, information and expertise is available nationwide from EPA and the Corps of Engineers as part of the Federal 404 permit program. State wildlife and fisheries departments can also provide data, advice, and expertise related to wetlands. Finally, the FWS can provide information on wetlands as part of the National Wetlands Inventory program, the Fish and Wildlife Enhancement Program, the Endangered Species and Habitat Conservation Program, the North American Waterfowl Management Program and the National Wildlife Refuge program. EPA and FWS wetland program contacts are included in Appendix D.

This section provides information on certain elements of standards (e.g., mixing zones) and the relationship between wetland standards and other water-related activities and programs (e.g., monitoring and CWA Sections 401, 402, 404, and 319). As information is developed by EPA and the States, EPA will periodically transfer it nationwide through workshops and program summaries. EPA's Office of Water Regulations and Standards has developed an outreach program for providing this information.

6.1 Section 401 Certification

Many States have begun to make more use of CWA Section 401 certification to manage certain activities that impact their wetland resources. Section 401 gives the States the authority to grant, deny, or condition certification of Federal permits or licenses (e.g., CWA Section 404 permits issued by the U.S. Army Corps of Engineers, Federal Energy

Regulatory Commission licenses, some Rivers and Harbors Act Sections 9 and 10 permits, and CWA Section 402 permits where issued by EPA) that may result in a discharge to "waters of the U.S." Such action is taken by the State to ensure compliance with various provisions of the CWA. Violation of water quality standards is often the basis for denials or conditioning through Section 401 certification. In the absence of wetland-specific standards, States have based decisions on their general narrative criteria and antidegradation policies. The Office of Wetlands Protection has developed a handbook for States entitled *Wetlands and 401 Certification: Opportunities and Guidelines for States and Eligible Indian Tribes* (USEPA 1989g) on the use of Section 401 certification to protect wetlands. This document provides several examples wherein States have applied their water quality standards to wetlands; one example is included in Appendix E.

The development of explicit water quality standards for wetlands, including wetlands in the definition of "State waters," uses, criteria, and antidegradation policies, can provide a strong and consistent basis for State 401 certifications.

6.2 Discharges to Wetlands

The Water Quality Standards Regulation (40 CFR 131.10(a)) states that, "in no case shall a State adopt waste transport or waste assimilation as a designated use for any 'waters of the U.S.'." This prohibition extends to wetlands, since they are included in the definition of "waters of the U.S." Certain activities involving the discharge of pollutants to wetlands may be permitted, as with other water types, providing a determination is made that the designated and existing uses of the wetlands and downstream waters will be maintained and protected. As with other surface waters, the State must ensure, through ambient monitoring, that permitted discharges to wetlands preserve and protect wetland functions and values as defined in State water quality standards, see Section 6.4.

Created wastewater treatment wetlands that are not impounded from waters of the United States and are designed, built, and operated solely as wastewater treatment systems, are a special case, and are not generally considered "waters of the U.S." Some such created wetlands, however, also provide other functions and values similar to those provided by natural wetlands. Under certain circumstances, such created, multiple use wetlands may be considered "waters of the U.S.," and as such, would be subject to the same protection and restrictions on use as natural wetlands (see *Report on the Use of Wetlands for Municipal Wastewater Treatment and Disposal* (USEPA 1987b)). This determination must be made on a case-by-case basis, and may consider factors such as the size and degree of isolation of the created wetland.

6.2.1 Municipal Wastewater Treatment

State standards should be consistent with the document developed by the Office of Municipal Pollution Control entitled *Report on the Use of Wetlands for Municipal Wastewater Treatment and Disposal* (USEPA 1987b), on the use of wetlands for municipal wastewater treatment. This document outlines minimum treatment and other requirements under the CWA for discharges to natural wetlands and those specifically created and used for the purpose of wastewater treatment.

The following is a brief summary of the above-referenced document. For municipal discharges to natural wetlands, a minimum of secondary treatment is required, and applicable water quality standards for the wetland and adjacent waters must be met. Natural wetlands are nearly always "waters of the U.S." and are afforded the same level of protection as other surface waters with regard to standards and minimum treatment requirements. There are no minimum treatment requirements for wetlands created solely for the purpose of wastewater treatment that do not qualify as "waters of the U.S." The discharge from the created wetlands that do not qualify as "waters of the U.S." must meet applicable standards for the receiving water. EPA encourages the expansion of wetland resources through the creation of engineered wetlands while allowing the use of natural wetlands for wastewater treatment only under limited conditions. Water quality standards for wetlands can prevent the misuse and overuse of natural wetlands for treatment through adoption of proper uses and criteria and application of State antidegradation policies.

6.2.2 Stormwater Treatment

Stormwater discharges to wetlands can provide an important component of the freshwater supply to wetlands. However, stormwater discharges from various land use activities can also contain a significant amount of pollutants. Section 402(p)(2) of the Clean Water Act requires that EPA, or States with authorized National Pollutant Discharge Elimination System (NPDES) programs, issue NPDES permits for certain types of stormwater discharges. EPA is in the process of developing regulations defining the scope of this program as well as developing permits for these discharges. Stormwater permits can be used to require controls that reduce the pollutants discharged to wetlands as well as other waters of the United States. In addition, some of the stormwater management controls anticipated in permits will require creation of wetlands or structures with some of the attributes of wetlands for the single purpose of water treatment.

EPA anticipates that the policy for stormwater discharges to wetlands will have some similarities to the policies for municipal wastewater discharges to wetlands. Natural wetlands are "waters of the United States" and are afforded a level of protection with regard to water quality standards and technology-based treatment requirements. The discharge from created wetlands must meet applicable water quality standards for the receiving waters. EPA will issue technical guidance on permitting stormwater discharges, including permitting stormwater discharges to wetlands, over the next few years.

6.2.3 Fills

Section 404 of the CWA regulates the discharge of dredged and fill material into "waters of the U.S." The Corps of Engineers' regulations for the 404 program are contained in 33 CFR Parts 320-330, while EPA's regulations for the 404 program are contained in 40 CFR Part 230-33.

One State uses the following guidelines for fills in their internal Section 401 review guidelines:

(a) *if the project is not water dependent, certification is denied;*

(b) *if the project is water dependent, certification is denied if there is a viable alternative (e.g., available upland nearby is a viable alternative);*

(c) *if no viable alternatives exist and impacts to wetland cannot be made acceptable through conditions on certification (e.g., fish movement criteria, creation of floodways to bypass oxbows, flow through criteria), certification is denied.*

Some modification of this may be incorporated into States' water quality standards. The States are encouraged to provide a linkage in their water quality standards to the determination of "significant degradation" as required under EPA guidelines (40 CFR 230.10(c)) and other applicable State laws affecting the disposal of dredged or fill materials in wetlands; see Section 5.1.

6.2.4 Nonpoint Source Assessment and Control

Wetlands, as with other waters, are impacted by nonpoint sources of pollution. Many wetlands, through their assimilative capacity for nutrients and sediment, also can serve an important water quality control function for nonpoint source pollution effects on waters adjacent to, or downstream of, the wetlands. Water quality standards play a pivotal role in both of the above. First, Section 319 of the CWA requires the States to complete assessments of nonpoint source (NPS) impacts to State waters, including wetlands, and to prepare management programs to control NPS impacts. Water quality standards for wetlands can form the basis for these assessments and management programs for wetlands. Second, water quality standards requirements for other surface waters such as rivers, lakes, and estuaries can provide an impetus for States to protect, enhance, and restore wetlands to help achieve nonpoint source control and water quality standards objectives for adjacent and downstream waters. The Office of Water Regulations and Standards and the Office of Wetlands Protection have developed guidance on the coordination of wetland and NPS control programs entitled *National Guidance - Wetlands and Nonpoint Source Control Programs* (USEPA 1990c).

6.3 Monitoring

Water quality management activities, including the permitting of wastewater and stormwater discharges, the assessment and control of NPS pollution, and waste disposal activities (sewage sludge, CERCLA, RCRA) require sufficient monitoring to ensure that the designated and existing uses of "waters of the U.S." are maintained and protected. In addition, Section 305(b) of the CWA requires

States to report on the overall status of their waters in attaining water quality standards. The inclusion of wetlands in water quality standards provides the basis for conducting both wetland-specific and status and trend monitoring of State wetland resources. Information gathered from the 305(b) reports may also be used to update and refine the designated wetland uses. The monitoring of wetlands is made difficult by limitations in State resources. Where regulated activities impact wetlands or other surface waters, States should provide regulatory incentives and negotiate monitoring responsibilities of the party conducting the regulated activity.

Monitoring of activities impacting specific wetlands may include several approaches. Monitoring methods involving biological measurements, such as plant, macroinvertebrate, and fish (e.g., biomass and diversity indices), have shown promise for monitoring stream quality (Plafkin et al., 1989). These types of indicators have not been widely tested for wetlands; see Section 7.1. However, the State of Florida has developed biological criteria as part of their regulations governing the discharge of municipal wastewater to wetlands[5]. The States are encouraged to develop and test the use of biological indicators. Other more traditional methods currently applied to other surface waters, including but not limited to the use of water quality criteria, sediment quality criteria, and whole effluent toxicity, are also available for conducting monitoring of specific wetlands.

Discharges involving persistent or bioaccumulative contaminants may necessitate the monitoring of the fate of such contaminants through wetlands and their impacts on aquatic life and wildlife. The exposure of birds and mammals to these contaminants is accentuated by the frequent use of wetlands by wildlife and the concentration of contaminants in wetlands through sedimentation and other processes. States should conduct monitoring of these contaminants in wetlands, and may require such monitoring as part of regulatory activities involving these contaminants.

Status and trend monitoring of the wetland resources overall may require additional approaches; see Section 3.1. Given current gaps in scientific knowledge concerning indicators of wetland quality, monitoring of wetlands over the next few years may focus on the spatial extent (i.e., quantity) and physical structure (e.g., plant types, diversity, and distribution) of wetland resources. The tracking of wetland acreage and plant communities using aerial photography can provide information that can augment the data collected on specific activities impacting wetlands, as discussed above.

EPA has developed guidance on the reporting of wetland conditions for the Section 305(b) program entitled *Guidelines for the Preparation of the 1990 State Water Quality Assessment 305(b) Report* (USEPA 1989b). When assessing individual specific wetlands, assessment information should be managed in an automated data system compatible with the Section 305(b) Waterbody System. In addition, the NWI program provides technical procedures and protocols for tracking the spatial extent of wetlands for the United States and subregions of the United States. These sources provide the framework for reporting on the status and trends of State wetland resources.

6.4 Mixing Zones and Variances

The guidance on mixing zones in the *Water Quality Standards Handbook* (USEPA 1983b) and the *Technical Support Document for Water Quality-Based Toxics Control* (TSD) (USEPA 1985b) apply to all surface waters, including wetlands. This includes the point of application of acute and chronic criteria. As with other surface waters, mixing zones may be granted only when water is present, and may be developed specifically for different water types. Just as mixing zone procedures are often different for different water types and flow regimes (e.g., free flowing streams versus lakes and estuaries), separate procedures also may be developed specifically for wetlands. Such procedures should meet the requirements contained in the TSD.

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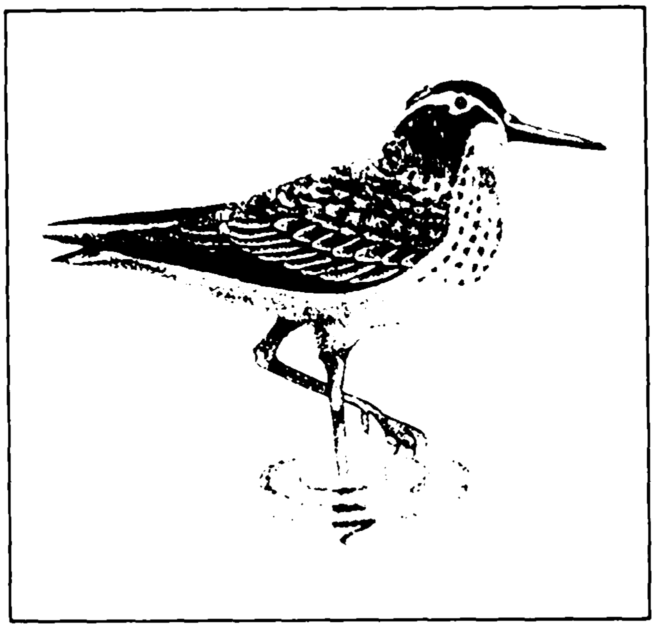
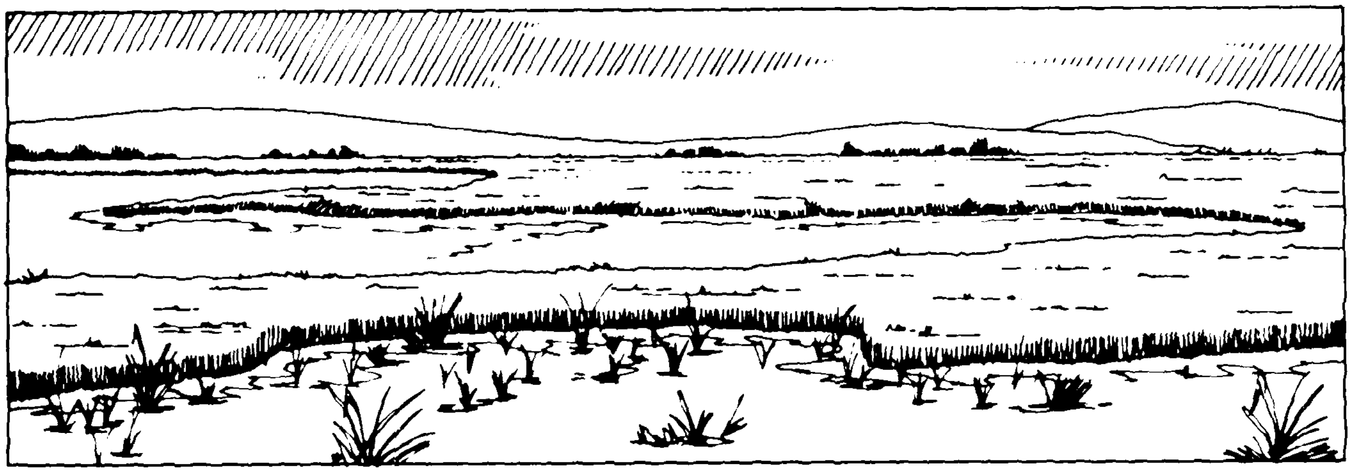
5 Florida Department of Environmental Regulations; State Regulations Part I, "Domestic Wastewater Facilities," Subpart C, "Design/Performance Considerations," 17-6.055, "Wetlands Applications."

As in other State waters, variances may be granted to discharges to wetlands. Variances must meet one or more of the six requirements for the removal of a designated use (40 CFR Part 131.10(g)) and must fully protect any existing uses of the wetland.



Chapter 7.0

Future Directions



EPA's Office of Water Regulations and Standards' planning document *Water Quality Standards Framework* (USEPA - Draft 1989e), identifies the major objectives for the program and the activities necessary to meet these objectives. Activities related to the development of water quality standards for wetlands are separated into two phases: (1) Phase 1 activities to be developed by the States by the end of FY 1993, discussed above; and (2) Phase 2 activities that will require additional research and program development, which are discussed below.

7.1 Numeric Biological Criteria for Wetlands

Development of narrative biological criteria is included in the first phase of the development of water quality standards for wetlands; see Section 5.1.2. The second phase involves the implementation of numeric biological criteria. This effort requires the detailed evaluation of the components of wetland communities to determine the structure and function of unimpaired wetlands. These measures serve as reference conditions for evaluating the integrity of other wetlands. Regulatory activities involving discharges to wetlands (e.g., CWA Sections 402 and 404) can provide monitoring data to augment data collected by the States for the development of numeric biological criteria; see Section 7.4. The development of numeric biological criteria for wetlands will require additional research and field testing over the next several years.

Biological criteria are based on local and regional biotic characteristics. This is in contrast to the nationally based chemical-specific aquatic life criteria developed by EPA under controlled laboratory conditions. The States will have primary responsibility for developing and implementing biological criteria for their surface waters, including wetlands, to reflect local and regional differences in resident biological communities. EPA will work closely with the States and the EPA Office of Research and Development to develop and test numeric biological criteria for wetlands. Updates on this work will be provided through the Office of Water Regulations and Standards, Criteria and Standards Division's regular newsletter.

7.2 Wildlife Criteria

Wetlands are important habitats for wildlife species. It is therefore important to consider wildlife in developing criteria that protect the functions and values of wetlands. Existing chemical-specific aquatic life criteria are derived by testing selected aquatic organisms by exposing them to contaminants in water. Although considered to be protective of aquatic life, these criteria often do not account for the bioaccumulation of these contaminants, which may cause a major impact on wildlife using wetland resources. Except for criteria for PCB, DDT, selenium, and mercury, wildlife have not been included during the development of the national aquatic life criteria.

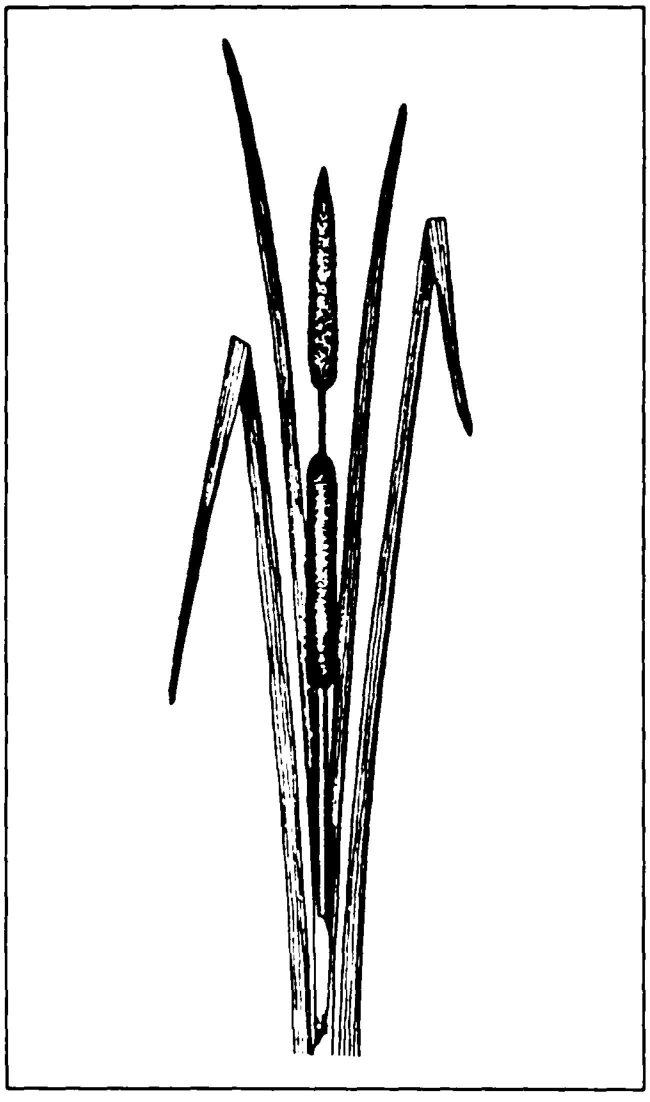
During the next 3 years, the Office of Water Regulations and Standards is reviewing aquatic life water quality criteria to determine whether adjustments in the criteria and/or alternative forms of criteria (e.g., tissue concentration criteria) are needed to adequately protect wildlife species using wetland resources. Since wetlands may not have open surface waters during all or parts of the year, alternative tissue based criteria based on contaminant concentrations in wildlife species and their food sources may become important criteria for evaluating contaminant impacts in wetlands, particularly those that bioaccumulate. Based on evaluations of current criteria and wildlife at risk in wetlands, national criteria may be developed.

7.3 Wetlands Monitoring

EPA's Office of Water Regulations and Standards is developing guidance for EPA and State surface water monitoring programs that will be issued by the end of FY 1990. This guidance will (1) encourage States to use monitoring data in a variety of program areas to support water quality management decisions; and (2) provide examples of innovative monitoring techniques through the use of case studies. The uses of data pertinent to wetlands that will be discussed include the following:

- refining use classification systems by developing physical, chemical, and biological water quality criteria, goals, and standards that account for regional variation in attainable conditions;
- identifying high-quality waters deserving special protection;
- using remote-sensing data;
- using integrated assessments to detect subtle ecological impacts; and
- identifying significant nonpoint sources of pollution that will prevent attainment of uses.

One or more case studies will address efforts to quantify the extent of a State's wetlands and to identify sensitive wetlands through their advance identification (USEPA 1989a).



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1 USEPA, Office of Wetlands Protection
Wetlands Strategies and State Programs Division
401 M St., S.W. (A-104F)
Washington, DC 20460
(202) 382-5048

2 USEPA, Office of Water Regulations and Standards
Assessment and Watershed Protection Division
401 M St., S.W. (WH-553)
Washington, DC 20460
(202) 382-7040

3 National Technical Information Service (NTIS)
5285 Front Royal Road
Springfield, VA 22116
(703) 487-4650

4 USEPA, Office of Water Regulations and Standards
Criteria and Standards Division
401 M St., S.W. (WH-585)
Washington, DC 20460
(202) 475-7315

5 Out of print. A revised Technical Support Document for Water Quality-based Toxics Control will be available October 1990 from:
Office of Water Enforcement and Permits
Permits Division
401 M St., S.W. (EN-336)
Washington, DC 20460

6 U.S. Government Printing Office
North Capitol St., N.W.
Washington, DC 20401
(202) 783-3238
a Order No. 024-010-00524-6
b Order No. 955-002-0000-8

7 USEPA, Water Policy Office
401 M St., S.W. (WH-556)
Washington, DC 20460
(202) 382-5818

8 USEPA, Office of Research and Development
Environmental Research Laboratory
200 SW 35th St.
Corvallis, OR 97333
(503) 420-4666

9 USEPA, Office of Municipal Pollution Control
401 M St., S.W. (WH-546)
Washington, DC 20460
(202) 382-5850

10 The Conservation Foundation
1250 Twenty-Fourth St., N.W.
Washington, DC 20037
(202) 293-4800

11 U.S. Army, Corps of Engineers
Wetlands Research Program
(601) 634-3774

12 USEPA, Office of Research and Development
Environmental Research Laboratory
Duluth, MN 55804
(218) 780-5549

Appendix A

Glossary

Ambient Monitoring - Monitoring within natural systems (e.g., lakes, rivers, estuaries, wetlands) to determine existing conditions.

Created Wetland - A wetland at a site where it did not formerly occur. Created wetlands are designed to meet a variety of human benefits including, but not limited to, the treatment of water pollution discharges (e.g., municipal wastewater, stormwater) and the mitigation of wetland losses permitted under Section 404 of the Clean Water Act. This term encompasses the term "constructed wetland" as used in other EPA guidance and documents.

Enhancement - An activity increasing one or more natural or artificial wetland functions. For example, the removal of a point source discharge impacting a wetland.

Functions - The roles that wetlands serve, which are of value to society or the environment.

Habitat - The environment occupied by individuals of a particular species, population, or community.

Hydrology - The science dealing with the properties, distribution, and circulation of water both on the surface and under the earth.

Restoration - An activity returning a wetland from a disturbed or altered condition with lesser acreage or functions to a previous condition with greater wetland acreage or functions. For example, restoration might involve the plugging of a drainage ditch to restore the hydrology to an area that was a wetland before the installation of the drainage ditch.

Riparian - Areas next to or substantially influenced by water. These may include areas adjacent to rivers, lakes, or estuaries. These areas often include wetlands.

Upland - Any area that does not qualify as wetland because the associated hydrologic regime is not sufficiently wet to elicit development of vegetation, soils and/or hydrologic characteristics associated with wetlands, or is defined as open waters.

Waters of the U.S. - See Appendix B for Federal definition; 40 CFR Parts 122.2, 230.3, and 232.2

Wetlands - Those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas. See Federal definition contained in Federal regulations: 40 CFR Parts 122.2, 230.3, and 232.2.

Appendix B

The Federal definition of "waters of the United States" (40 CFR Section 232.2(q)) is:

(1) All waters which are currently used, were used in the past, or may be susceptible to use in interstate or foreign commerce, including all waters which are subject to the ebb and flow of the tide;

(2) All interstate waters including interstate wetlands;

(3) All other waters such as intrastate lakes, rivers, streams (including intermittent streams), mudflats, sandflats, wetlands, sloughs, prairie potholes, wet meadows, playa lakes, or natural ponds, the use, degradation or destruction of which would or could affect interstate or foreign commerce including any such waters:

- (i) Which are or could be used by interstate or foreign travelers for recreational or other purposes; or
- (ii) From which fish or shellfish could be taken and sold in interstate or foreign commerce;
- (iii) Which are used or could be used for industrial purposes by industries in interstate commerce;*

(4) All impoundments of waters otherwise defined as waters of the United States under this definition;

(5) Tributaries of waters identified in paragraphs 1-4;

(6) The territorial sea; and

(7) Wetlands adjacent to waters (other than waters that are themselves wetlands) identified in 1-6; waste treatment systems, including treatment ponds or lagoons designed to meet the requirements of CWA (other than cooling ponds as defined in 40 CFR 423.11(m) which also meet criteria in this definition) are not waters of the United States.

(*Note: EPA has clarified that waters of the U.S. under the commerce connection in (3) above also include, for example, waters:

Which are or would be used as habitat by birds protected by Migratory Bird Treaties or migratory birds which cross State lines;
Which are or would be used as habitat for endangered species;
Used to irrigate crops sold in interstate commerce.)

Appendix C

Information on the Assessment of Wetland Functions and Values

Summary of Methodologies Prior to 1983 (Lonard and Clairain 1986)

Introduction

Since 1972, a wide variety of wetlands evaluation methodologies have been developed by Federal or State agencies, private consulting firms, and the academic community. These evaluation methods have been developed to ascertain all or selected wetland functions and values that include habitat; hydrology, including water quality recreation; agriculture/silviculture; and heritage functions.

Publications by the U.S. Water Resources Council (Lonard et al., 1981) and the U.S. Army Engineer Waterways Experiment Station (Lonard et al., 1984) documented and summarized pre-1981 wetland evaluation methods. The two documents include a critical review of the literature, identification of research needs, and recommendations for the improvement of wetlands evaluation methodologies. Methodology analyses include an examination of wetlands functions; geographic features; personnel requirements for implementation, data requirements, and products; field testing; flexibility; and administrative uses. Recently, the U.S. Environmental Protection Agency, with technical assistance from WAPORA, Inc. (1984) summarized freshwater wetland evaluation methodologies related to primary and cumulative impacts published prior to 1981. The specific objective of this paper is to present a summary of wetlands evaluation methodologies identified from the pre-1981 literature, and to present an update of methodologies published since 1981.

Methods

In 1981, a U.S. Army Engineer Waterways Experiment Station (WES) study team evaluated 40 wetlands evaluation methodologies according to several screening criteria, and examined 20 of the methodologies in detail using a series of descriptive parameters (Lonard et al., 1981). The criteria and parameters were developed to ensure consistency during review and analysis of methodologies. Five additional methodologies proposed since 1981 have been analyzed and summarized for this paper using the same criteria. This does not suggest, however, that only five methodologies have been developed since 1981.

Available Wetlands Evaluation Methodologies

Abstracts of 25 wetlands evaluation methodologies that met the WES study team's criteria include the following:

1. Adamus, P.R., and Stockwell, L.T. 1983. "A Method for Wetland Functional Assessment. Volume I. Critical Review and Evaluation Concepts," U.S. Department of Transporta-

tion. Federal Highway Administration. Office of Research, Environmental Division. Washington, D.C. 20590; and Adamus, P.R. 1983. "A Method for Wetland Functional Assessment. Volume II. The Method," U.S. Department of Transportation. Federal Highway Administration. Office of Research, Environmental Division. Washington, D.C. 20590.

Volume I of the method provides a detailed literature review and discussion of the rationale of the method. The wetland functional assessment or evaluation methodology presented in Volume II consists of three separate procedures. Procedure I, referred to as a "Threshold Analysis," provides a methodology for estimating the probability that a single wetland is of high, moderate, or low value for each of 11 wetland functions discussed in detail in Volume I. This procedure is based on assessment of 75 bio-physical wetland features obtained from office, field, and quantitative studies. It also incorporates consideration of the social significance of the wetland as indicated by public priorities. The priorities are determined based on results of a series of questions that the evaluator must consider. Procedure II, designed as a "Comparative Analysis," provides parameters for estimating whether one wetland is likely to be more important than another for each wetland function, and Procedure II, referred to as "Mitigation Analysis," provides an outline for comparing mitigation alternatives and their reasonableness." The evaluation methodology is qualitative in its approach.

2. Brown, A., Kittle, P., Dale, E.E., and Huffman, R.T. 1974. "Rare and Endangered Species, Unique Ecosystems, and Wetlands," Department of Zoology and Department of Botany and Bacteriology. The University of Arkansas, Fayetteville, Arkansas.

The Arkansas Wetlands Classification System contains a two-part, multivariate approach for evaluating freshwater wetlands for maximum wildlife production and diversity. Initially, Arkansas wetlands were qualitatively classified as prime or nonprime wetlands habitats according to use by man. A numerical value for a wetland was determined by calculating a subscore, which was based on the multiplication of a significance coefficient by a determined weighted value. The values for each variable were summed, and a total wetland qualitative value was obtained for use by decision makers.

3. Dee, N., Baker, J., Drobney, N., Duke, K., Whitman, I., and Fahringer, D. 1973. "Environmental Evaluation System for Water Resources Planning," *Water Resources Research*, Vol 9, No. 3, pp 523-534.

The Environmental Evaluation System (EES) is a methodology for conducting environmental impact analysis. It was developed by an interdisciplinary research team, and is based on a hierarchical arrangement of environmental quality indicators, an arrangement that classifies the major areas of environmental concern into major categories, components, and ultimately into parameters and measurements of environmental quality. The EES provides for environmental impact evaluation in four major categories: ecology, environmental pollution, aesthetics, and human interest. These four categories are further broken down into 18 components, and finally into 78 parameters. The EES provides a means for measuring or estimating selected environmental impacts of large-scale water resource development projects in commensurate units termed environmental impact units (EIU). Results of using the EES include a total score in EIU "with" and "without" the proposed project; the difference between the two scores in one measure of environmental impact. Environmental impact scores developed in the EES are based on the magnitude of specific environmental impacts and their relative importance. Another major output from the EES is an indication of major adverse impacts called "red flags," which are of concern of and by themselves. These red flags indicate "fragile" elements of the environment that must be studied in more detail. (Authors' abstract.)

4. Euler, D.L., Carreiro, F.T., McCullough, G.B., Snell, E.A., Glooschenko, V., and Spurr, R.H. 1983. "An Evaluation System for Wetlands of Ontario South of the Precambrian Shield," First Edition. Ontario Ministry of Natural Resources and Canadian Wildlife Service, Ontario Region. Variously paged.

The methodology was developed to evaluate a wide variety of wetland functions that include biological, social, hydrological, and special features. The procedures includes a rationale of scientific and technical literature for wetlands values, the evaluation methodology, a step-by-step procedure manual, a wetland data record, and a wetland evaluation record. The procedure was developed to evaluate and rank a wide variety of inland wetlands located in Ontario, Canada, south of the Precambrian Shield.

5. Fried, E. 1974. "Priority Rating of Wetlands for Acquisition," *Transaction of the Northeast Fish and Wildlife Conference,* Vol 31, pp 15-30.

New York State's Environmental Quality Bond Act of 1972 provides $5 million for inland wetland acquisition, $18 million for tidal wetlands acquisition, and $4 million for wetlands restoration. A priority rating system, with particular emphasis on inland wetlands, was developed to guide these programs. The governing equation was: priority rating = (P + V + A) x 5, where the priority rating is per acre desirability for acquisition, P is biological productivity, V is vulnerability, and A is additional factors. Both actual and potential conditions could be rated. The rating system was successfully applied to some 130 inland wetlands. Using a separate equation, wetland values were related to costs. (Authors's abstract.)

6. Galloway, G.E. 1978. "Assessing Man's Impact on Wetlands," Sea Grant Publications Nos. UNC-SG-78-17 or UNC-WRRI-78-136, University of North Carolina, Raleigh, North Carolina.

The Wetland Evaluation System (WES) proposed by Galloway emphasizes a system approach to evaluate man's impact on a wetland ecosystem. Impacts are determined and compared for "with" and "without" project conditions. The advice of an interdisciplinary team, as well as the input of local elected officials and laymen, are included as part of the WES model. Parameters that make up a wetland are assessed at the macro-level, and the results of the evaluation are displayed numerically and graphically with computer assisted techniques.

7. Golet, F.C. 1973. "Classification Evaluation of Freshwater Wetlands as Wildlife Habitat in the Glaciated Northeast," *Transactions of the Northeast Fish and Wildlife Conference,* Vol 30, pp 257-279.

A detailed classification system for freshwater wetlands is presented along with 10 criteria for the evaluation of wetlands as wildlife habitat. The results are based on a 2-year field study of over 150 wetlands located throughout the state of Massachusetts. The major components of the classification system include wetland classes and subclasses, based on the dominant life form of vegetation and surface water depth and permanence; size categories; topographic and hydrologic location; surrounding habitat types; proportions and interspersion of cover and water; and vegetative interspersion. These components are combined with wetland juxtaposition and water chemistry to produce criteria for a wetland evaluation. Using a system of specification and ranks, wetlands can be arranged according to their wildlife value for decision-making. (Author's abstract.) "At this point, the system has been used in numerous states on thousands of wetlands; recent revisions have resulted in such use." (F.C. Golet)

8. Gupta, T.R., and Foster, J.H. 1973. "Valuation of Visual-Cultural Benefits from Freshwater Wetlands in Massachusetts," *Journal of the Northeastern Agricultural Council*, Vol 2, No 1, pp 262-273.

The authors suggested an alternative to the "willingness to pay" approaches for measuring the social values of natural open space and recreational resources. The method combines an identification and measurement of the physical qualities of the resource by landscape architects. Measurement values were expressed in the context of the political system and current public views. The procedure is demonstrated by its application to freshwater wetlands in Massachusetts.

9. Kibby, H.V. 1978. "Effects of Wetlands on Water Quality," *Proceedings of the Symposium on Strategies for Protection and Management of Floodplain Wetlands and other Riparian Ecosystems*, General Technical Report No. GTR-WO-12, U.S. Department of Agriculture, Forest Service, Washington, D.C.

Wetlands potentially have significant effects on water quality. Significant amounts of nitrogen are assimilated during the growing season and then released in the fall and early spring. Phosphorus, while assimilated by wetlands, is also released throughout the year. Some potential management tools for evaluating the effect of wetlands on water quality are discussed. (Author's abstract.)

10. Larson, J.S. (ed.) 1976. "Models for Assessment of Freshwater Wetlands," Publication No. 32. Water Resources Research Center, University of Massachusetts, Amherst, Massachusetts.

Four submodels for relative and economic evaluation of freshwater wetlands are presented within a single, 3-phase elimination model. The submodels treat wildlife, visual-cultural, groundwater, and economic values.

The wildlife and visual-cultural models are based on physical characteristics that, for the most part, can be measured on existing maps and aerial photographs. Each characteristic is given values by rank and coefficient. A relative numerical score is calculated for the total wetland characteristics and used to compare it with a broad range of northeastern wetlands or with wetlands selected by the user. The groundwater model places wetlands in classes of probable groundwater yield, based on surficial geologic deposits under the wetland.

The economic submodel suggests values for wildlife, visual-cultural aspects, groundwater, and flood control. Wildlife values are derived from the records of state agency purchases of wetlands with sportsmen's dollars for wildlife management purposes. Visual-cultural economic values are based on the record of wetland purposes for open space values by municipal conservation commissions. Groundwater values stem from savings realized by selection of a drilled public water supply over a surface water source. Flood control values are based on U.S. Army Corps of Engineers data on flood control values of the Charles River, Massachusetts, mainstream wetlands.

The submodels are presented within the framework of an overall 3-phase eliminative model. Phase I identifies outstanding wetlands that should be protected at all costs. Phase II applies the wildlife, visual-cultural, and groundwater submodels to those wetlands that do not meet criteria for outstanding wetlands. Phase III develops the economic values of the wetlands evaluated in Phase II.

The models are intended to be used by local, regional, and state resource planners and wetlands regulation agencies. (Author's abstract.)

11. Marble, A.D., and Gross, M. 1984. "A Method for Assessing Wetland Characteristics and Values," *Landscape Planning*, Vol 11, pp 1-17.

The method presented for assessing wetland values identified the relative importance of wetlands in providing wildlife habitat, flood control, and improvement of surface water quality. All wetlands in the study area were categorized on the basis of their landscape position of hilltop, hillside, or valley. Each of the wetland values measured were then related to the corresponding landscape position categories. Valley wetlands were found to be most valuable in all instances. The method provides information on wetland values that can be simply gathered and easily assessed, requiring only available data and a minimum of resources. Implementation of this method on a regional or municipality-wide basis can provide decision makers with readily accessible and comparative information on wetland values. (Authors' abstract.)

12. Michigan Department of Natural Resources. 1980. "Manual for Wetland Evaluation Techniques: Operation Draft," Division of Land Resource Programs, Lansing, Michigan. 29 pp.

The Michigan Department of Natural Resources (MDNR) Wetland Evaluation Technique is designed to assist decision makers on permit applications involving projects where significant impacts are anticipated. The manual describes the criteria to be used in evaluating any particular wetland. The technique provides a means of evaluating the status of existing wetlands as well as potential project-related impacts on wetland structure and aerial extent. One part of the technique requires examination of six basic features of wetlands, including: (1) hydrologic functions; (2) soil characteristics; (3) wildlife habitat/use evaluation; (4) fisheries habitat/use; (5)

nutrient removal/recycling functions; (6) removal of suspended sediments. A second part of the analysis includes consideration of public interest concerns. This method also includes brief consideration of cumulative, cultural/historic, and economic impacts.

13. Reppert, R.T., Sigleo, W., Stakhiv, E., Messman, L., and Meyers, C. 1979. "Wetland Values: Concepts and Methods for Wetlands Evaluation," IWR Research Report 79-R-1, U.S. Army Engineer Institute for Water Resources, Fort Belvoir, Virginia.

The evaluation of wetlands is based on the analysis of their physical, biological, and human use characteristics. The report discusses these functional characteristics and identifies specific criteria for determining the efficiency with which the respective functions are performed.

Two potential wetlands evaluation methods are described. One is a non-quantitative method in which individual wetland areas are evaluated based on the deductive analysis of their individual functional characteristics. The other is a semi-quantitative method in which the relative values of two or more site alternatives are established through the mathematical rating and summation of their functional relationships.

The specific functions and values of wetlands that are covered in this report are (1) natural biological functions, including food chain productivity and habitat; (2) their use as sanctuaries, refuges, or scientific study areas; (3) shoreline protection; (4) groundwater recharge; (5) storage for flood and stormwater; (6) water quality improvement; (7) hydrologic support; and (8) various cultural values. (Authors' abstract.)

14. Shuldiner, P.W., Cope, D.F., and Newton, R.B. 1979. "Ecological Effects on Highway Fills of Wetlands," Research Report. National Cooperative Highway Research Program Report No. 218A, Transportation Research Board, National Research Council, Washington, D.C.; and Shuldiner, P.W., Cope, D.F., and Newton, R.B. 1979. "Ecological Effects of Highway Fills on Wetlands," User's Manual. National Cooperative Highway Research Program Report No. 218B, Transportation Research Board, National Research Council, Washington, D.C.

The two reports include a Research Report and a User's Manual to provide, in concise format, guidelines and information needed for the determination of the ecological effects that may result from the placement of highway fills on wetlands and associated floodplains, and to suggest procedures by which deleterious impacts can be minimized or avoided. The practices that can be used to enhance the positive benefits are also discussed. Both reports cover the most common physical, chemical, and biological effects that the highway engineer is likely to encounter when placing fills in wetlands, and displays the effects and their interactions in a series of flowcharts and matrices.

15. SCS Engineers. 1979. "Analysis of Selected Functional Characteristics of Wetlands," Contract No. DACW73-78-R-0017, Reston, Virginia.

The investigation focused on identifying factors and criteria for assessing the wetland functions of water quality improvement, groundwater recharge, storm and floodwater storage, and shoreline protection. Factors and criteria were identified that could be used to develop procedures to assist Corps personnel in wetlands assessing the values of general wetland types and of specific wetlands in performing the functions indicated. To the extent possible, procedures were then outlined that allow the application of these criteria in specific sites.

16. Smardon, R.D. 1972. "Assessing Visual-Cultural Values on Inland Wetlands in Massachusetts," Master of Science Thesis. University of Massachusetts. Amherst, Massachusetts.

This study deals with the incorporation of visual-cultural values of inland wetlands into the decision making process of land use allocation of inland wetlands in Massachusetts. Visual-cultural values of inland wetlands may be defined as visual, recreational, and educational values of inland wetlands to society. The multivariate model is an eliminative and comparative model that has three levels of evaluation. The first level identifies those wetlands that are outstanding natural areas, have regional landscape value, or are large wetland systems.

These wetlands have top priority for preservation. The second level is a rating and ranking system. At this stage, the combined natural resource values of the wetland are evaluated. Wetlands with high ratings or rank from this level are eliminated and have the next highest priority for preservation or some sort of protection. The third level evaluation considers the cultural values (e.g., accessibility, location near schools) of wetlands. The model is designed to be utilized at many different levels of decision making. For example, it can be used by state agencies, town conservation commissions, and conceivably could be used by other states in northeastern United States. (Author's abstract.)

17. Solomon, R.D., Colbert, B.K., Hansen, W.J., Richardson, S.E., Ganter, L.W., and Vlachos, E.C. 1977. "Water Resources Assessment Methodology (WRAM)--Impact Assessment and Alternative Evaluation," Technical Report Y-77-1, Environmental Effects Laboratory, U.S. Army Engineer Waterways Experiment Station, CE, Vicksburg, Mississippi.

This study presented a review of 54 impact assessment methodologies and found that none entirely satisfied the needs or requirements for the Corps' water resources project and programs. However, salient features contained in several of the methodologies were considered pertinent and were utilized to develop a water resources assessment methodology (WRAM). One of the features consisted of weighting impacted variables and scaling the impacts of alternatives. The weighted rankings technique is the basic weighting and scaling tool used in this methodology. Principal components of WRAM include assembling an interdisciplinary team; selecting and ensuring assessment variables; identifying, predicting, and evaluating impacts and alternatives; and documenting the analysis. Although developed primarily for use by the Corps in water resources management, WRAM is applicable to other resources agencies.

18. State of Maryland Department of Natural Resources. Undated. "Environmental Evaluation of Coastal Wetlands (Draft)," Tidal Wetlands Study, pp 181-208.

The Maryland scheme for the evaluation of coastal wetlands is based on the recognition of 32 distinct types of vegetation in the marshes and swamps of tidewater areas of the state. Rankings of vegetation types were developed and parameters for the evaluation of specific areas of wetlands were described. The application of the scheme is explained and demonstrated. Guidance is provided for the interpretation of results. The application of the Maryland scheme requires a detailed inventory of the types of vegetation in the area selected for evaluation.

19. U.S. Army Engineer District, Rock Island. 1983. "Wetland Evaluation Methodology," Wisconsin Department of Natural Resources, Bureau of Water Regulation and Zoning.

The Wetland Evaluation Methodology is a shortened and revised version of a technique developed for the Federal Highway Administration (FHWA) (see Adamus, 1983; Number 1). The FHWA technique was designed to assess all wetland types whereas the Wetland Evaluation Methodology assesses those wetlands in Wisconsin (e.g., assessment procedures in the FHWA technique for estuarine marshes have been omitted from the Wetland Evaluation Methodology). Other changes have also been incorporated into the Wetland Evaluation Methodology to more closely reflect other regional conditions.

20. U.S. Army Engineer Division, Lower Mississippi Valley. 1980. "A Habitat Evaluation System for Water Resources Planning," U.S. Army Corps of Engineers, Lower Mississippi Valley Division, Vicksburg, Mississippi.

A methodology is presented for determining the quality of major habitat types based on the description and quantification of habitat characteristics. Values are compared for existing baseline conditions, future conditions without the project, and with alternative project conditions. Curves, parameter characteristics, and descriptive information are included in the appendices. The Habitat Evaluation System (HES) procedure includes the following steps for evaluating impacts of a water resource development project. The steps include: (1) obtaining habitat type or land use acreage; (2) deriving Habitat Quality Index scores; (3) deriving Habitat Unit Values; (4) projecting Habitat Unit Values for the future "with" and "without" project conditions; (5)

using Habitat Unit Values to assess impacts of project conditions; and (6) determining mitigation requirements.

21. U.S. Army Engineer Division, New England. 1972. "Charles River: Main Report and Attachments," Waltham, Massachusetts.

The study was a long-term project directed by the U.S. Army Corps of Engineers to study the resources of the Charles River Watershed in eastern Massachusetts. It had an emphasis on how to control flood damage in the urbanized lower watershed, and how to prevent any significant flood damage in the middle and upper watershed. Seventeen crucial wetlands were identified for acquisition to maintain flood storage capacity in the watershed as a nonstructural alternative for flood protection in the lower Charles River basin. Various aspects of the watershed were studied in an interdisciplinary fashion.

22. U.S. Department of Agriculture. 1978. "Wetlands Evaluation Criteria--Water and Related Land Resources of the Coastal Region, Massachusetts," Soil Conservation Service, Amherst, Massachusetts.

A portion of the document contains criteria used to evaluate major wetlands in the coastal region of Massachusetts. Each of the 85 wetlands evaluated was subjected to map study and field examination. Ratings were assigned based on point values obtained for various attributes. A rationale for each evaluation item was developed to explain the development of the criteria.

23. U.S. Fish and Wildlife Service. 1980. "Habitat Evaluation Procedures (HEP) Manual (102ESM)," Washington, D.C.

HEP is a method that can be used to document the quality and quantity of available habitat for selected wildlife species. HEP provides information for two general types of wildlife habitat comparisons: (1) the relative value of different areas at the same point in time; and (2) the relative value of the same area at future points in time. By combining the two types of comparisons, the impact of proposed or anticipated land and water changes on wildlife habitat can be quantified. This document described HEP, discusses some probable applications, and provides guidance in applying HEP in the field.

24. Virginia Institute of Marine Science. Undated. "Evaluation of Virginia Wetlands," (mimeographed). Glouchester Point, Virginia.

The authors presented a procedure to evaluate the wetlands of Virginia. The objective of the wetland evaluation program was to recognize wetlands that possess great ecological significance as well as those of lesser significance. Two broad categories of criteria were utilized in evaluating the ecological significance of wetlands: (1) the interaction of wetlands with the marine environment; and (2) the interaction of the wetland with the terrestrial environment. A formula was developed to incorporate various factors into "relative ecological significance values."

25. Winchester, B.H., and Harris, L.D. 1979. "An Approach to Valuation of Florida Freshwater Wetlands," *Proceedings of the Sixth Annual Conference on the Restoration and Creation of Wetlands*, Tampa, Florida.

A procedure was presented for estimating the relative ecological and functional value of Florida freshwater wetlands. Wetland functions evaluated by this procedure include water quality enhancement, water detention, vegetation diversity and productivity, and wildlife habitat value. The field parameters used in the assessment were wetland size, contiguity, structural vegetative diversity, and an edge-to-area ration. The procedure was field tested and was time- and cost-effective. Allowing flexibility in both the evaluative criteria used and the relative weight assigned to each criterion, the methodology is applicable in any Florida region for which basic ecological data are available.

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Wetland Assessment Techniques Developed Since 1983 (USEPA 1989a)

- Wetlands Evaluation Technique (Adamus, et al. 1987). This nationally applicable procedure has been used in at least six ADIDs to date, mostly in its original form (known popularly as the "FHWA" or "Adamus" method). It has since been extensively revised and is available at no cost (with simple software) from the Corps of Engineers Wetlands Research Program (contact: Buddy Clairain, 601-634-3774). Future revisions are anticipated.

- Bottomland Hardwoods WET (Adamus 1987). This is a simplified, regionalized version of WET, applicable to EPA Regions 4 and 6. It is available from OWP (contact: Joe DaVia at 202-475-8795). Supporting software is being developed, and future revisions are anticipated.

- Southeastern Alaska WET (Adamus Resource Assessment 1987). This is also a simplified, regionalized version of WET.

- Minnesota Method (U.S. Army Corps of Engineers-St.Paul, 1988). This was a joint State-Federal effort that involved considerable adaptation of WET. A similar effort is underway in Wisconsin.

- Onondaga County Method (SUNY-Syracuse 1987). This was adapted from WET by Smardon and others at the State University of New York.

- Hollands-Magee Method. This is a scoring technique developed by two consultants and has been applied to hundreds of wetlands in New England and part of Wisconsin (contact: Dennis Magee at 603-472-5191). Supporting software is available.

- Ontario Method (Euler et al. 1983). This is also a scoring technique, and was extensively peer-reviewed in Canada. (Contact: Valanne Glooschenko, 416-965-7641).

- Connecticut Method (Amman et al. 1986). This is a scoring technique developed for inland municipal wetland agencies.

- Marble-Gross Method (Marble and Gross 1984). This was developed for a local application in Connecticut.

- Habitat Evaluation System (HES) (Tennessee Dept. of Conservation 1987). This is a revised version of a Corps-sponsored method used to evaluate Lower Mississippi wildlife habitat.

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Appendix D

REGIONAL COORDINATORS

Regional Water Quality Standards Coordinators
U.S. Environmental Protection Agency (USEPA)

Eric Hall, WQS Coordinator
USEPA, **Region 1**
Water Management Division
JFK Federal Building
Boston, MA 02203
(FTS) 835-3533
(617) 565-3533

Rick Balla, WQS Coordinator
USEPA, **Region 2**
Water Management Division
26 Federal Plaza
New York, NY 10278
(FTS) 264-1559
(212) 264-1559

Linda Holst, WQS Coordinator
USEPA, **Region 3**
Water Management Division
841 Chestnut Street
Philadelphia, PA 19107
(FTS) 597-0133
(215) 597-3425

Fritz Wagener, WQS Coordinator
USEPA, **Region 4**
Water Management Division
345 Courtland Street, N.E.
Atlanta, GA 30365
(FTS) 257-2126
(404) 347-2126

Larry Shepard, WQS Coordinator
USEPA, **Region 5** (TUD-8)
Water Management Division
230 South Dearborn Street
Chicago, IL 60604
(FTS) 886-0135
(312) 886-0135

David Neleigh, WQS Coordinator
USEPA, **Region 6**
Water Management Division
1445 Ross Avenue
First Interstate Bank Tower
Dallas, TX 75202
(FTS) 255-7145
(214) 655-7145

John Houlihan, WQS Coordinator
USEPA, **Region 7**
Water Compliance Branch
726 Minnesota Avenue
Kansas City, KS 66101
(FTS) 276-7432
(913) 551-7432

Bill Wuerthele, WQS Coordinator
USEPA, **Region 8** (8WM-SP)
Water Management Division
999 18th Street
Denver, CO 80202-2405
(FTS) 330-1586
(303) 293-1586

Phil Woods, WQS Coordinator
USEPA, **Region 9**
Water Management Division (W-3-1)
75 Hawthorne Street
San Francisco, CA 94105
(FTS) 484-1994
(415) 744-1994

Sally Marquis, WQS Coordinator
USEPA, **Region 10**
Water Management Division (WD-139)
1200 Sixth Avenue
Seattle, WA 98101
(FTS) 399-2116
(206) 442-2116

Regional Wetland Program Coordinators
U.S. Environmental Protection Agency (USEPA)

Doug Thompson, Wetlands Coordinator
USEPA, **Region 1**
Water Management Division
Water Quality Branch
John F. Kennedy Federal Building
Boston, Massachusetts 02203-2211
(FTS) 835-4422
(617) 565-4422

Dan Montella, Wetlands Coordinator
USEPA, **Region 2**
Water Management Division
Marine & Wetlands Protection Branch
26 Federal Plaza
New York, New York 10278
(FTS) 264-5170
(212) 264-5170

Barbara D'Angelo, Wetlands Coordinator
USEPA, **Region 3**
Environmental Service Division
Wetlands and Marine Policy Section
841 Chestnut Street
Philadelphia, Pennsylvania 19107
(FTS) 597-9301
(215) 597-9301

Tom Welborn, Wetlands Coordinator
(Regulatory Unit)
Gail Vanderhoogt, Wetlands Coordinator
(Planning Unit)
USEPA, **Region 4**
Water Management Division
Water Quality Branch
345 Courtland Street, N.E.
Atlanta, Georgia 30365
(FTS) 257-2126
(404) 347-2126

Doug Ehorn, Wetland Coordinator
USEPA, **Region 5**
Water Management Division
Water Quality Branch
230 South Dearborn Street
Chicago, Illinois 60604
(FTS) 886-0243
(312) 886-0243

Jerry Saunders, Wetlands Coordinator
USEPA, **Region 6**
Environmental Services Division
Federal Activities Branch
12th Floor, Suite 1200
1445 Ross Avenue
Dallas, Texas 75202
(FTS) 255-2263
(214) 655-2263

Diane Hershberger, Wetlands Coordinator
Assistant Regional Administrator for
Policy and Management
USEPA, **Region 7**
Environmental Review Branch
726 Minnesota Avenue
Kansas City, Kansas 66101
(FTS) 276-7573
(913) 551-7573

Gene Reetz, Wetlands Coordinator
USEPA, **Region 8**
Water Management Division
State Program Management Branch
One Denver Place, Suite 500
999 18th Street
Denver, Colorado 80202-2405
(FTS) 330-1565
(303) 293-1565

Phil Oshida, Wetlands Coordinator
USEPA, **Region 9**
Water Management Division
Wetlands, Oceans and Estuarine Branch
1235 Mission Street
San Francisco, California 94103
(FTS) 464-2187
(415) 744-2180

Bill Riley, Wetlands Coordinator
USEPA, **Region 10**
Water Management Division
Environmental Evaluation Branch
1200 Sixth Avenue
Seattle, Washington 98101
(FTS) 399-1412
(206) 422-1412

Regional Wetland Program Coordinators
U.S. Fish and Wildlife Service (USFWS)

Region 1	California, Hawaii, Idaho, Nevada, Oregon, Washington RWC: Dennis Peters ASST: Howard Browers	Regional Wetland Coordinator USFWS, Region 1 Fish and Wildlife Enhancement 1002 N.E. Holladay Street Portland, Oregon 97232-4181 COM: 503/231-6154 FTS: 429-6154
Region 2	Arizona, New Mexico Oklahoma, Texas RWC: Warren Hagenbuck ASST: Curtis Carley	Regional Wetland Coordinator USFWS, Region 2 Room 4012 500 Gold Avenue, SW Albuquerque, New Mexico 87103 COM: 505/766-2914 FTS: 474-2914
Region 3	Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, Wisconsin RWC: Ron Erickson ASST: John Anderson	Regional Wetland Coordinator USFWS, Region 3 Fish and Wildlife Enhancement Federal Building, Ft Snelling Twin Cities, Minnesota 55111 COM: 612/725-3536 FTS: 725-3536
Region 4	Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Puerto Rico, South Carolina, Tennessee, Virgin Islands RWC: John Hefner ASST: Charlie Storrs	Regional Wetland Coordinator USFWS, Region 4 R.B. Russell Federal Building 75 Spring Street, S.W. Suite 1276 Atlanta, Georgia 30303 COM: 404/331-6343 FTS: 841-6343

Region 5	Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New York, New Jersey, Pennsylvania, Rhode Island, Vermont, Virginia, West Virginia RWC: Ralph Tiner ASST: Glenn Smith	Regional Wetland Coordinator USFWS, Region 5 One Gateway Center, Suite 700 Newton Corner, MA 02158 COM: 617/965-5100 FTS: 829-9379
Region 6	Colorado, Kansas, Montana, Nebraska, North Dakota, South Dakota, Utah, Wyoming RWC: Chuck Elliott ASST: Bill Pearson	Regional Wetland Coordinator USFWS, Region 6 Fish and Wildlife Enhancement P.O. Box 25486 Denver Federal Center Denver, Colorado 80225 COM: 303/236-8180 FTS: 776-8180
Region 7	Alaska RWC: Jon Hall ASST: David Dall	Regional Wetland Coordinator USFWS, Region 7 1011 East Tudor Road Anchorage, Alaska 99503 COM: 907/786-3403 or 3471 FTS: (8) 907/786-3403

Appendix E

EXAMPLE OF STATE CERTIFICATION ACTION INVOLVING WETLANDS UNDER CWA SECTION 401

The dam proposed by the City of Harrisburg was to be 3,000 feet long and 17 feet high. The dam was to consist of 32 bottom-hinged flap gates. The dam would have created an impoundment with a surface area of 3,800 acres, a total storage capacity of 35,000 acre-feet, and a pool elevation of 306.5 feet. The backwater would have extended approximately 8 miles upstream on the Susquehanna River and approximately 3 miles upstream on the Conodoguinet Creek.

The project was to be a run-of-the-river facility, using the head difference created by the dam to create electricity. Maximum turbine flow would have been 10,000 cfs (at a nethead of 12.5), and minimum flow would have been 2,000 cfs. Under normal conditions, all flows up to 40,000 cfs would have passed through the turbines.

The public notice denying 401 certification for this project stated as follows:

1. The construction and operation of the project will result in the significant loss of wetlands and related aquatic habitat and acreage. More specifically:

    a. The destruction of the wetlands will have an adverse impact on the local river ecosystem because of the integral role wetlands play in maintaining that ecosystem.

    b. The destruction of the wetlands will cause the loss of beds of emergent aquatic vegetation that serve as habitat for juvenile fish. Loss of this habitat will adversely affect the relative abundance of juvenile and adult fish (especially smallmouth bass).

    c. The wetlands which will be lost are critical habitat for, among other species, the yellow crowned night heron, black crowned night heron, marsh wren and great egret. In addition, the yellow crowned night heron is a proposed State threatened species, and the marsh wren and great egret are candidate species of special concern.

    d. All affected wetlands areas are important and, to the extent that the loss of these wetlands can be mitigated, the applicant has failed to demonstrate that the mitigation proposed is adequate. To the extent that adequate mitigation is possible, mitigation must include replacement in the river system.

    e. Proposed riprapping of the shoreline could further reduce wetland acreage. The applicant has failed to demonstrate that there will not be an adverse water quality and related habitat impact resulting from riprapping.

f. Based upon information received by the Department, the applicant has underestimated the total wetland acreage affected.

2. The applicant has failed to demonstrate that there will be no adverse water quality impacts from increased groundwater levels resulting from the project. The ground water model used by the applicant is not acceptable due to erroneous assumptions and the lack of a sensitivity analysis. The applicant has not provided sufficient information concerning the impact of increased groundwater levels on existing sites of subsurface contamination, adequacy of subsurface sewage system replacement areas and the impact of potential increased surface flooding. Additionally, information was not provided to adequately assess the effect of raised groundwater on sewer system laterals, effectiveness of sewer rehabilitation measures and potential for increased flows at the Harrisburg wastewater plant.

3. The applicant has failed to demonstrate that there will not be a dissolved oxygen problem as a result of the impoundment. Present information indicates the existing river system in the area is sensitive to diurnal, dissolved oxygen fluctuation. Sufficient information was not provided to allow the Department to conclude that dissolved oxygen standards will be met in the pool area. Additionally, the applicant failed to adequately address the issue of anticipated dissolved oxygen levels below the dam.

4. The proposed impoundment will create a backwater on the lower three miles of the Conodoguinet Creek. Water quality in the Creek is currently adversely affected by nutrient problems. The applicant has failed to demonstrate that there will not be water quality degradation as a result of the impoundment.

5. The applicant has failed to demonstrate that there will not be an adverse water quality impact resulting from combined sewer overflows.

6. The applicant has failed to demonstrate that there will not be an adverse water quality impact to the 150-acre area downstream of the proposed dam and upstream from the existing Dock Street dam.

7. The applicant has failed to demonstrate that the construction and operation of the proposed dam will not have an adverse impact on the aquatic resources upstream from the proposed impoundment. For example, the suitability of the impoundment for smallmouth bass spawning relative to the frequency of turbid conditions during spawning was not adequately addressed and construction of the dam and impoundment will result in a decrease in the diversity and density of the macroinvertebrate community in the impoundment area.

8. Construction of the dam will have an adverse impact on upstream and downstream migration of migratory fish (especially shad). Even with the construction of fish passageways for upstream and downstream migration, significant declines in the numbers of fish successfully negotiating the obstruction are anticipated.

9. The applicant has failed to demonstrate that there will not be an adverse water quality impact related to sedimentation within the pool area.

APPENDIX E

An Approach for Evaluating Numeric Water Quality Criteria for Wetlands Protection

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

AN APPROACH FOR EVALUATING NUMERIC WATER QUALITY CRITERIA FOR WETLANDS PROTECTION

by

Cynthia A. Hagley and Debra L. Taylor
Asci Corporation
Duluth, Minnesota 55804

Project Officer

William D. Sanville
Project Leader
Environmental Research Laboratory
Duluth, Minnesota 55804

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ABSTRACT

Extension of the national numeric aquatic life criteria to wetlands has been recommended as part of a program to develop standards and criteria for wetlands. This report provides an overview of the need for standards and criteria for wetlands and a description of the numeric aquatic life criteria. The numeric aquatic life criteria are designed to be protective of aquatic life and their uses for surface waters, and are probably applicable to most wetland types. This report provides a possible approach, based on the site-specific guidelines, for detecting wetland types that might not be protected by direct application of national numeric criteria. The evaluation can be simple and inexpensive for those wetland types for which sufficient water chemistry and species assemblage data are available, but will be less useful for wetland types for which these data are not readily available. The site-specific approach is described and recommended for wetlands for which modifications to the numeric criteria are considered necessary. The results of this type of evaluation, combined with information on local or regional environmental threats, can be used to prioritize wetland types (and individual criteria) for further site-specific evaluations and/or additional data collection. Close coordination among regulatory agencies, wetland scientists, and criteria experts will be required.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF
WATER

MEMORANDUM

SUBJECT: Numeric Water Quality Criteria for Wetlands

FROM: William R. Diamond, Director
Standards and Applied Science Division
Office of Science and Technology

TO: Water Management Division Directors (Regions I-X)
Environmental Services Division Directors (Regions I-X)

State Water Pollution Control Agency Directors

The purpose of this memorandum is to provide you with a copy of a report entitled "An Approach for Evaluation of Numeric Water Quality Criteria for Wetlands Protection", prepared by EPA's Environmental Research Laboratory in Duluth, Minnesota. This report was requested in the early stages of planning for wetland water quality standards to assess the applicability of EPA's existing numeric aquatic life criteria methodology for wetlands. This report was prepared by the Wetlands Research Program and is part of the Agency's activities to assist States with developing water quality standards for wetlands.

The report evaluates EPA's numeric aquatic life criteria to determine how they can be applied to wetlands. Numeric aquatic life criteria are designed to be protective of aquatic life for a wide range of surface water types. The report suggests that most numeric aquatic life criteria are applicable to most wetland types.

However, there are some wetland types where EPA's criteria are not appropriate. This report presents an approach that States may use as a screening tool to detect those wetland types that may be under- or overprotected by EPA's criteria. The proposed approach relies on data readily available from EPA's 304(a) criteria documents, as well as species assemblages and water quality data from individual wetland types. The results of this type of simple evaluation can be used to prioritize wetland types where further evaluation may be needed prior to setting criteria. Two example analyses of the approach are included in the report. EPA's site-specific criteria development guidelines can then be used to modify criteria if appropriate.

This report compiles existing information from EPA's 304(a) criteria guidance documents and site-specific criteria methodologies and does not contain new guidance or policy. The report has been peer reviewed by ERL/Duluth scientists who develop EPA's 304 criteria. The report also has been reviewed by the Standards and Applied Science Division and the Wetlands Division.

If you have additional questions on the information contained in this report or its applications, contact the following persons: David Sabock, Water Quality Standards Branch, at 202-475-7315 regarding designated uses and water quality standards policies; Bob April, Ecological Risk Assessment Branch, at 202-475-7315, regarding EPA's aquatic life criteria; or Bill Sanville, Environmental Research Laboratory/Duluth, at 218-720-5500, regarding the research for this report.

Attachment

cc: Water Quality Branch Chiefs (Regions I-X)
Water Quality Standards Coordinators (Regions I-X)
Wetlands Coordinators (Regions I-X)
David Sabock
Bob April
Bill Sanville

CONTENTS

TABLES

ACKNOWLEDGEMENTS

Preparation of this document has been funded by the U.S. Environmental Protection Agency. This document has been prepared at the EPA Environmental Research Laboratory in Duluth, Minnesota, through Contract # 68033544 to AScI Corporation. This document has been subjected to the Agency's peer and administrative review. Excellent reviews and assistance were received from C. Stephan, R. Spehar, C. Johnston, E. Hunt, D. Robb, and J. Minter.

SECTION 1

INTRODUCTION

NEED FOR STANDARDS FOR WETLANDS

Wetlands have been studied and appreciated for a relatively short time in relation to other types of aquatic systems. The extent of their value in the landscape has only recently been recognized; in fact, a few decades ago government policies encouraged wetland drainage and conversion. Wetlands traditionally have been recognized as important fish and wildlife habitats, and it is estimated that over one-third of U.S. endangered species require wetland habitat for their continued existence. Some of their many other values, however, have become apparent only recently. These include attenuation of flood flows, groundwater recharge, shoreline and stream bank stabilization, filtering of pollutants from point and nonpoint sources, unique habitats for both flora and fauna, and recreational and educational opportunities.[1]

Impacts to Wetlands

Despite new appreciation of the valuable functions that wetlands perform in the landscape, they continue to be destroyed and altered at a rapid pace. Since pre-settlement times over half of the wetlands in the continental U.S. have been destroyed, and losses over the last few decades have remained high.[2] These figures only represent actual loss of acreage and do not account for alterations to or contamination of still-extant wetlands. The causes of wetland destruction and degradation include:[3]

* Urbanization - Resulting in drainage and filling, contamination, and ecological isolation of wetlands.

* Agriculture Conversion - Drainage, cropping, and grazing which change or destroy wetland structure and ecological function.

* Water Resource Development - Water flow alterations to wetlands from diking, irrigation diversions, alterations to rivers for navigation, diversions for

water supply, and groundwater pumping. These result in changes in the hydrology that sustains the wetland system.

* Chemical Pollution - From point and nonpoint sources, hazardous waste sites, mining, and other activities. These can overwhelm the assimilative capacity of wetlands or be toxic to wetland organisms.

* Biological Disturbances - Introduction or elimination of plant and animal species that affect ecosystem processes.

Gaps in Federal Regulatory Programs

Existing Federal regulatory programs intended to reduce some of the impacts described above leave major gaps in the protection of wetlands. Section 404 of the Clean Water Act (CWA) requires a permit to be obtained from the Army Corps of Engineers, in cooperation with the U.S. Environmental Protection Agency (EPA), before dredged material or fill can be discharged into waters of the United States. Alterations such as drainage, water diversion, and chemical contamination are not covered by Section 404 unless material will be discharged into the wetland in association with such alterations. The Resource Conservation and Recovery Act, which regulates the disposal of hazardous wastes, and the CWA, which regulates contamination from waste-water discharges and nonpoint-source pollution, could provide protection from certain impacts, but they have not been used consistently to regulate impacts to wetlands. Programs designed to protect endangered species, migratory birds, and marine mammals have also been used to reduce impacts to wetlands, but "the application of these programs also has been uneven."[4]

Gaps in State Regulatory Programs

Wetland regulations vary greatly among States. Some States are now developing narrative standards for wetlands (e.g. Wisconsin, Rhode Island, and others). On the other hand, although wetlands are included in the Federal definition of "waters of the United States" and are protected by Section 101(a) of the CWA, not all States include them as "waters of the State" in their definitions. A review conducted in 1989 by the EPA Office of Wetlands Protection and the Office of Water Regulations and Standards found that only 27 of 50 States mentioned wetlands in definitions of State waters. The review verified that there generally is a lack of consideration given to water quality standards for wetlands.[5]

Effective Use of Existing Regulatory Options

Although some impacts (e.g. excavation, most drainage, and destruction of vegetation) are not addressed by the current implementation of existing regulations and programs, much of the chemical contamination of wetlands could be controlled through existing Federal and State water pollution control laws.[4] The National Wetlands Policy Forum recommended that EPA and State water pollution control agencies review the implementation of their water quality programs to ensure that the chemical integrity of wetlands is adequately protected. The Forum stressed the need to develop water quality standards designed to protect sensitive wetlands.[5]

Under Section 401 of the CWA, States have authority to authorize, condition, or deny all Federal permits or licenses in order to comply with State water quality standards, including, but not limited to, Sections 402 and 404 of the CWA, Sections 9 and 10 of the Rivers and Harbors Act, and Federal Energy Regulatory Commission licenses. States with water quality standards that apply to or are specifically designed for wetlands can use 401 certification much more effectively as a regulatory tool.

As wetlands receive more recognition as important components of State water resources, the need for testing the applicability of some existing guidelines and standards to wetlands regulation becomes more apparent.

PROPOSED APPROACH TO DEVELOPMENT OF WETLAND STANDARDS

The EPA Office of Water Regulations and Standards and Office of Wetlands Protection recently completed a document entitled, "National Guidance: Water Quality Standards for Wetlands."[6] It recommends a two-phased approach for the development of water quality standards for wetlands. In the first 3-year phase of this program, standards for wetlands would be developed using existing information in order to provide protection to wetlands consistent with the protection afforded other State waters. Technical support for this initial phase will be provided through documents such as this one, which focuses on the application of existing numeric criteria to wetlands. These criteria are widely used. Applying them to wetlands requires a small amount of effort and can be accomplished quickly.

The development of narrative biocriteria is also required in the initial phase of standards development. The long-term goal (3-10 years) of this program is to develop numeric biocriteria for wetlands. It is anticipated that both narrative and numeric biocriteria can provide a more integrative estimate of whole-wetland health and better identification of impacts and trends than can be attained by traditional numeric chemical criteria. Field-based, community-level biosurveys can be implemented to complement, and help validate, laboratory-based conclusions. Results of such surveys can be used to monitor wetlands for degradation and establish narrative or numeric biocriteria or guidance which take into account "real world" biological interactions and the interactions of multiple stressors.

More information on the development of numeric biocriteria will be available in a guidance document in coming years. Technical guidance to support the development of biological criteria for wetlands has also been prepared.[7] This guidance provides a synthesis of technical information on field studies of inland wetland biological communities.

PURPOSE OF THIS DOCUMENT

A number of steps are needed to develop wetland standards. The document, "National Guidance: Water Quality Standards for Wetlands," mentioned above, provides general guidelines to the States for each of the following steps: the inclusion of wetlands in definitions of State waters, the relationship between wetland standards and other water-related programs, use classification systems for wetlands, the definition of wetland functions and values, the applicability of existing narrative and numeric water quality criteria to wetlands, and the application of antidegradation policies to wetlands.

The technical document for biological criteria[7] and this report are companions to the guidance document described above. This report is directed primarily toward wetland scientists unfamiliar with water quality regulation and is intended to provide a basis for dialogue between wetland scientists and criteria experts regarding adapting numeric aquatic life criteria to wetlands. More specifically:

1) It provides background information and an overview of water quality standards and numeric chemical criteria, including application to wetlands.

2) The need for evaluating numeric water quality criteria is discussed. The site-specific guidelines are introduced and discussed in two contexts: a) as an initial screening tool to ensure that water quality in extreme wetland types is adequately protected by criteria, and b) in terms of using the site-specific guidelines to modify criteria for wetlands where criteria might be over or underprotective.

3) An approach is described that uses information available from criteria documents and is designed to: a) detect wetland types where water quality is not clearly protected by existing criteria, and b) help prioritize further evaluations and research efforts.

4) A simple test of the approach is presented with two examples. Results are not considered conclusive and are presented only as an example of the procedure.

Most of the data and examples are based on the freshwater acute criteria. A similar approach should be equally applicable to the saltwater acute criteria and to both saltwater and freshwater chronic criteria.

SECTION 2

CURRENT SURFACE WATER STANDARDS AND CRITERIA

This section describes how criteria are used in State standards, how national numeric criteria are derived, and what options are currently available for modifying national aquatic life criteria.

DESCRIPTION OF STANDARDS AND CRITERIA

Surface waters are protected by Section 101(a) of the CWA with the goal: "to restore and maintain the chemical, physical, and biological integrity of the nation's waters." State water quality standards are developed to meet this goal.

State Standards

There are two main components to establishing a standard: 1) The level of water quality attainable for a particular waterbody, or the designated use of that waterbody (e.g. recreational, fishery, etc.) is determined; 2) Water quality criteria (usually a combination of narrative and numeric) are established to protect that designated use. Water quality standards also contain an antidegradation policy "to maintain and protect existing uses and water quality, to provide protection for higher quality waters, and to provide protection for outstanding national resource waters."[8] State standards for a particular waterbody must be met when discharging wastewaters. The "National Guidance: Water Quality Standards for Wetlands"[6] outlines a basic program to achieve these goals for wetlands.

Aquatic Criteria

Narrative Criteria--

Narrative criteria are statements, usually expressed in a "free from ..." format. For example, all States have a narrative statement in their water quality standards which requires that their waters not contain "toxic substances in toxic amounts." Narrative criteria are typically applied at the State level when combinations of pollutants must be controlled or when pollutants are present which are not listed in State water quality standards.[8] States must document the process by which they propose to implement these narrative criteria in their standards.

Numeric Criteria--

Pollutant-specific numeric criteria are used by the States when it is necessary to control individual pollutants in order to protect the designated use of a waterbody. Fate and transpor models commonly are used to translate these criteria into permit limits for individual dischargers. Some criteria apply State-wide and others are specific to particular designated uses or waterbodies.

National numeric criteria are developed by EPA based on best available scientific information. They serve as recommendations to assist States in developing their own criteria and to assist in interpreting narrative criteria.[9] These include human health and aquatic life pollutant-specific criteria and whole effluent toxicity criteria. Sediment criteria are now being developed. States can adopt national numeric criteria directly. Alternatively, site-specific criteria may be developed using EPA-specified guidelines, and State-specific criteria can be derived using procedures developed by the State.[8]

DEVELOPMENT OF NATIONAL AQUATIC LIFE NUMERIC CRITERIA

National aquatic life criteria are usually derived using single-species laboratory toxicity tests. Tests are repeated with a wide variety of aquatic organisms for each chemical. The criteria are designed to protect against unacceptable effects to aquatic organisms or their uses caused by exposures to high concentrations for short periods of time (acute effects), to lower concentrations for longer periods of time (chronic effects), and to combinations of both.[9] EPA criteria are composed of 1) magnitude (what concentration of a pollutant is allowable); 2) duration of exposure (the period of time over which the in-stream concentration is averaged for comparison with criteria concentrations); and 3) frequency (how often the criterion can be exceeded without unacceptably affecting the community).[10] Separate criteria are determined for fresh water and salt water. Field data are used when appropriate.

All acceptable data regarding toxicity to fish and invertebrates are evaluated for inclusion in the criteria. Data on toxicity to aquatic plants are evaluated to determine whether concentrations of the chemical that do not cause unacceptable effects to aquatic animals will cause unacceptable effects to plants. Bioaccumulation data are examined to determine if residues in the organisms might exceed FDA action levels or cause known effects on the wildlife that consume them. For a complete description of the procedures for deriving ambient criteria, consult the "National Guidelines" (1985).[9]

Numeric water quality criteria are designed to protect most of the species inhabiting a site.[9] A wide variety of taxa with a range of sensitivities are required for deriving criteria. Guidelines are followed to determine the availability of sufficient experimental data from enough appropriate taxa to derive a criterion. For example, to derive a freshwater Final Acute Value for a chemical, results of acute tests with at least one species of freshwater animal in at least eight different families are required. Acute and chronic values can be made to be a function of a water quality characteristic such as Ph, salinity, or hardness, when it is determined that these characteristics impact toxicity, and enough data exist to establish the relationship. Table 1 lists the chemicals for which freshwater aquatic life criteria have been developed and indicates which of those criteria are pH, hardness, or temperature dependent.

SITE-SPECIFIC GUIDELINES

An option for modifying national aquatic life water quality criteria to reflect local conditions is presented in the site-specific guidelines. States may develop site-specific criteria by modifying the national criteria for sites where 1) water quality characteristics, such as pH, hardness, temperature, etc., that might impact toxicity of the pollutants of concern differ from the laboratory water used in developing the criterion; or 2) the types of organisms at the site differ from, and may be more or less sensitive than, those used to calculate the criterion; or 3) both may be true. Site-specific criteria take local conditions into account to provide an appropriate level of protection. They can also be used to set seasonal criteria when there is high temporal variability.[8]

A testing program can be used to determine whether site-specific modifications to criteria are necessary. This program may include water quality sampling and analysis, a biological survey, and acute and chronic toxicity tests.[11] If site-specific modifications are deemed necessary, 3 separate procedures are available for using site-specific guidelines to modify criteria values, including the recalculation procedure, the indicator species procedure, and the resident species procedure. These will be discussed more fully in the next section.

SECTION 3

THE NEED FOR EVALUATING NUMERIC WATER QUALITY CRITERIA: USE OF THE SITE-SPECIFIC GUIDELINES

OVERALL RELEVANCE OF CRITERIA TO WETLANDS

The national aquatic life criteria have been developed to provide guidance to the States for the protection of aquatic life and their uses in a variety of surface waters. They are designed to be conservative and "... have been developed on the theory that effects which occur on a species in appropriate laboratory tests will generally occur on the same species in comparable field situations. All North American bodies of water and resident aquatic species and their uses are meant to be taken into account, except for a few that may be too atypical ..."[9] A wide variety of taxonomic groups sensitive to many materials are used in testing, including many taxa common to both wetlands and other surface waters. In order to ensure that criteria are appropriately protective, water used for testing is low in particulate matter and organic matter, because these substances can reduce availability and toxicity of some chemicals. For these reasons, the "National Guidance: Water Quality Standards for Wetlands" states that, in most cases, criteria should be protective of wetland biota.[6]

Although the water quality criteria are probably generally protective of wetlands and provide the best currently available tool for regulating contamination from specific pollutants, there are many different types of wetlands with widely variable conditions. There might be some wetland types where the resident biota or chemical and physical conditions are substantially different from what the criteria were designed to protect. These differences could result in underprotection or overprotection of the wetland resource. This section discusses the use of site-specific guidelines for wetland types for which certain criteria might be over or underprotective, but its primary focus is to provide a mechanism to identify wetland types that might be underprotected by certain criteria and that might require further research.

WETLAND VARIABILITY

Wetlands are usually located at the interface between terrestrial systems and truly aquatic systems, and so combine attributes of both.[12] They are intermediate between terrestrial and aquatic systems in the amount of water they store and process and are very sensitive to changes in hydrology.[12] Their chemical and physical properties, such as nutrient availability, degree of substrate anoxia, soil salinity, sediment properties, and pH are influenced greatly by hydrologic conditions. Attendees at a Wetlands Water Quality Workshop (held in Easton, Maryland in August, 1988) listed the most common ways in which wetlands differ from "typical" surface waters: higher concentrations of organic carbon and particulate matter, more variable and generally lower pH, more variable and generally lower dissolved oxygen, more variable temperatures, and more transient availability of water.[13]

There is also high variability among wetland types. Wetlands, by definition, share hydrophytic vegetation, hydric soils, and a water table at or near the surface at some time during the growing season. Beyond these shared features, however, there is tremendous hydrological, physical, chemical, and biological variability. For example, an early classification system for wetlands, "Circular 39", listed 20 distinctly different wetland types[14], and the present "Cowardin" system lists 56 classes of wetlands.[15] This variability makes it important to evaluate different wetland types individually.

USE OF THE SITE-SPECIFIC GUIDELINES FOR WETLANDS

The site-specific guidelines outlined in Section 2 are designed to address the chemical and biological variability described above. Determining the need for site-specific modifications to criteria requires a comparison of the aquatic biota and chemical conditions at the site to those used for establishing the criterion. This comparison is useful for identifying wetland types that might require additional evaluation. The three site-specific options are discussed in the context of their general relevance to wetlands and are used in this discussion to provide a framework for evaluating the protectiveness of criteria for wetlands.

In most cases, because of the conservative approach used in the derivation of the criteria, use of the site-specific guidelines to modify criteria results in no change or in their relaxation, provided that an adequate number of species are used in the calculations. However, criteria can also become more restrictive. Newly tested species could be especially sensitive to certain pollutants, or extreme water conditions found in some surface waters or wetland types might not reduce the toxicity of a chemical. Disease, parasites, predators, other pollutants, contaminated or insufficient food, and fluctuating and extreme conditions might all affect the ability of organisms to withstand toxic pollutants.[9]

Appropriateness of Testing Organisms: Recalculation Procedure

The first option given in the site-specific guidelines is the recalculation procedure.[8,11] This approach is designed to take into account differences between the sensitivity of resident species and those used to calculate a criterion for the material of concern. It involves eliminating data from the criterion database for species that are not resident at that site. It could require additional resident species testing in laboratory water if the number of species remaining for recalculating the criterion drops below the minimum data requirements. "Resident" species include those that seasonally or intermittently exist at a site.[11,16]

Use of the recalculation procedure will not necessarily result in a higher acute criterion value (less restrictive), even if sensitive species are eliminated from the dataset and minimum family requirements are met. The number of families used to calculate Final Acute Values is important. If a number of non-wetland species are dropped out of the calculation without adding a sufficient number of new species, a lower (more restrictive) Final Acute Value can result, because data are available for fewer species.[11]

Similarity of Required Taxa and Typical Wetland Species--

The variety of test species required to establish the national numeric criteria was chosen to represent a wide range of taxa having a wide range of habitat requirements and sensitivity to toxicants. Establishment of a freshwater Final Acute Value for a chemical requires a minimum of 8 different types of families to be tested. These include: 1) the family Salmonidae; 2) a second family of fish, preferably a warmwater species; 3) a third family in the phylum Chordata (fish, amphibian, etc.); 4) a planktonic crustacean; 5) a benthic crustacean; 6) an insect; 7) a family in a phylum other than Arthropoda or Chordata; and 8) a family in any order of insect or phylum not already represented.[9]

When a required type of family does not exist at a site, the guidelines for the recalculation procedure specify that substitutes from a sensitive family, resident in the site, should be added to meet the minimum family data requirement. Should it happen that all resident families have been tested and the minimum data requirements still have not been met, the acute toxicity value from the most sensitive resident family that has been tested should be used as the site-specific value.

Most of the required families are probably well-represented in most wetland types. Some types of wetlands, however, seldom or never contain fish, and most wetland types do not support salmonids or aquatic insects requiring flowing water.

General Evaluation of Species Suitability--

Table 2 presents six criterion chemicals chosen as examples and the eight taxonomic groups required to establish criteria. The chemicals include two organochlorines: polychlorinated biphenyls (PCBs - used in industrial applications, environmentally-persistent, bioaccumulate) and pentachlorophenol (widely used fungicide and bactericide); one organophosphate: parathion (insecticide); two metals: zinc and chromium(VI); and cyanide.

The species used for acute toxicity testing for each of the six chemicals have been broken down by taxonomic group and evaluated based on the likelihood that those species can be found in wetlands. Except for the unsuitability of the Salmonidae to most wetland types, most of the taxonomic groups are well-represented for the six chemicals used as examples. Wetland species were not present in the list of species used to calculate the Final Acute Value for the "non-arthropod/non-chordate" and "another insect or new phylum" groups for a few of the criteria. This is not because these groups are not represented in wetlands. These are very general classifications. For example, the "non-arthropod/non-chordate" group can include rotifers, annelids, and mollusks among other phyla, all of which should have many representatives in most types of wetlands. There is a large degree of variation in the total number of species tested for the six chemicals used as examples, ranging from 10 fish and invertebrates for polychlorinated biphenyls (PCBs) to 45 for zinc (Table 7). Criteria based on smaller numbers of species are less likely to include a sufficient number of wetland species to fulfill the minimum family requirements. Additional toxicity testing, using laboratory water and wetland species from the missing families, can be done to fill these gaps.

While the general taxonomic groups required for toxicity testing are fairly well represented in wetlands, the similarity between the genera and species inhabiting individual wetland types and those used for criteria testing varies widely among criteria and wetland types. Species chosen for toxicity testing were seldom or never chosen with wetlands in mind. In addition, relatively little is known about species assemblages in some types of wetlands (particularly in those lacking surface waters, such as wet meadows or bogs). Defining typical wetland taxa is difficult. For example, while most types of wetlands do not support salmonids, Coho salmon are highly dependent on wetlands in Alaska, where there is a higher percentage and acreage of wetlands than in any other State. Part of the utility of the evaluation proposed here is in identifying where significant gaps in data exist.

Influence of Cofactors: Indicator Species Procedure

The second of the three site-specific procedures, the indicator species procedure, accounts for differences in biological availability and/or toxicity of a material caused by physical and/or chemical characteristics of the site water, or cofactors. For the acute test, the effect of site water is compared to the effect of laboratory water, using at least two resident species or acceptable non-resident species (one fish and one invertebrate) as indicators. A ratio is determined, which is used to modify the Final Acute Value. See Carlson et al. (1984) for information and guidelines for determination of site-specific chronic values.[11]

Suitability of Standard Testing Conditions--

Standard aquatic toxicity tests are performed using natural or reconstituted dilution water that should not of itself affect the results of toxicity tests. For example, organic carbon and particulate matter are required to be low to avoid sorption or complexation of toxicants, which might lower the toxicity or availability of some criterion chemicals. Recommended acute test conditions for certain water quality characteristics of fresh and salt water are listed in Table 3. Wetlands, as well as some types of surface waters, can have values far outside the ranges used for standard testing for some of these characteristics (most notably total organic carbon, particulate matter, pH, and dissolved oxygen). Wetland types can be evaluated to identify these extremes.

Wetland Cofactors--

Many water quality characteristics can 1) act as cofactors to affect the toxicity of pollutants (e.g. alkalinity/acidity, hardness, ionic strength, organic matter, temperature, dissolved oxygen, suspended solids); 2) can be directly toxic to organisms (e.g. un-ionized ammonia, high or low pH, hydrogen sulfide, low dissolved oxygen); or 3) can interfere mechanically with feeding and reproduction (e.g. suspended solids). The criteria for some of these water quality characteristics can be naturally exceeded in many wetland types, as well as in some lakes and streams.

Hardness, pH, and temperature adjustments built into a few of the criteria account for effects from these cofactors in a few cases, but no other cofactors are now included in the criteria, despite some known effects. For example, alkalinity, salinity, and suspended solids, in addition to pH and hardness, are known to affect the toxicities of heavy metals and ammonia. These cofactors are not included in the criteria primarily because there are insufficient data.[9] For example, most toxicity tests have been performed under conditions of low or high salinity, so that estuaries, where salinity values can vary greatly, may require salinity-dependent site-specific criteria for some metals.[11] An initial evaluation of the adequacy of protection provided to a wetland type by a criterion should take possible cofactor effects into account.

Combination: Resident Species Procedure

The resident species procedure accounts for differences in both species sensitivity and water quality characteristics.[11] This procedure is costly, because it requires that a complete minimum dataset be developed using site water and resident species. It is designed to compensate concurrently for differences in the sensitivity range of species represented in the dataset used to derive the criterion and for site water differences which may markedly affect the biological availability and/or toxicity of the chemical.[11]

AQUATIC PLANTS

One of the most notable differences between wetlands and other types of surface waters is the dominance (and importance) of aquatic macrophytes and other hydrophytic vegetation in wetlands. Aquatic plants probably constitute the majority of the biomass in most wetland types.

Few data concerning toxicity to aquatic plants are currently required for deriving aquatic life criteria. Traditionally, procedures for aquatic toxicity tests on plants have not been as well developed as for animals. Although national numeric criteria development guidelines state that results of a test with a freshwater alga or vascular plant "should be available" for establishing a criterion, they do not require that information.[9] The Final Plant Value is the lowest (most sensitive) result from tests with important aquatic plant species (vascular plant or alga), in which the concentrations of test material were measured and the endpoint was biologically important. Plant values are compared to animal values to determine the relative sensitivities of aquatic plants and animals. If plants are "among the aquatic organisms that are most sensitive to the material," results of a second test with a plant from another phylum are included.[9]

Results of tests with plants usually indicate that criteria which protect aquatic animals and their uses also protect aquatic plants and their uses.[9] As criteria are evaluated for their suitability for wetlands, however, plant values should be examined carefully. Additional plant testing may be advisable in some cases. If site-specific adjustments are made to some criteria, they could result in less restrictive acute and chronic values for animals. Some plant values could then be as sensitive or more sensitive than the animal values. Chemicals with fairly sensitive plant values include: aluminum, arsenic(III), cadmium, chloride, chromium(VI), cyanide, and selenium(VI). For example, fish are generally much more sensitive to cyanide than invertebrates. If the recalculation procedure was used to develop a site-specific cyanide criterion for a wetland type containing no fish, values for these sensitive species would be replaced in the calculation, possibly by less sensitive species. A less restrictive criterion could result, possibly making the plant value more sensitive than the animal value. Therefore, additional consideration should be given to plant toxicity data for wetland systems.

SECTION 4

EVALUATION PROGRAM

The direct application of existing aquatic life criteria to wetlands is assumed to be reasonable in most cases. It provides a practical approach towards protecting the biological integrity of wetlands. The following evaluation program offers a possible strategy to identify extreme wetland types that might be underprotected by some criteria, to prioritize wetland types and criterion chemicals for further testing or research, and to identify gaps in available data. The approach can be helpful for identifying those instances where modifications to existing criteria might be advisable. The proposed evaluation program offers a screening tool to begin to answer the following questions: 1) Are there some wetland types for which certain criteria are underprotective? 2) For criteria in wetland types that cannot be applied directly, can site-specific guidelines be used to modify the criteria to protect the wetland? 3) Will additional toxicity testing under wetland conditions and with wetland species be necessary in some cases in order to establish site-specific criteria?

The proposed approach relates species and water quality characteristics of individual wetland types to species and water quality characteristics important in deriving each criterion. It involves identifying wetland types of concern, identifying cofactors possibly affecting toxicity for the criteria of interest, gathering data on the biota and water quality characteristics of the wetland type, and comparing to data used to derive the criterion.

CLASSIFICATION

The proposed program for the evaluation of the suitability of aquatic life criteria discussed in this section can be done separately for individual wetland types. These can be defined in the classification process, which is the first step in developing standards for wetlands. The classification process requires the identification of the various structural types of wetlands and identification of their functions and values.[6] The classification should provide groups of wetlands that are similar enough structurally and functionally so that they can reasonably be expected to respond in kind to inputs of toxic chemicals.

EVALUATING THE APPROPRIATENESS OF DIRECT APPLICATION OF CRITERIA

Information Needed

1. Identification of cofactors. Cofactors potentially affecting mobility and biological availability for each criterion chemical should be identified. Cofactors known to affect each criterion chemical are listed in individual national criteria documents and are summarized in Table 4. The absence of a relationship between a cofactor and a chemical on Table 4 does not ensure that no relationship exists, merely that none was discussed in the criteria document. The chemistry of the effects of the cofactors on the chemicals is often very complicated, and limited data are available regarding some of the relationships. The approach presented here is simplistic and is geared toward directing further efforts. Other sources of information, in addition to the criteria documents, should be consulted when actually applying this approach. Criteria that include hardness- or pH-dependent correction factors (Table 1) should apply directly to wetlands unless the wetland type has extremes of pH or hardness well outside the ranges used in toxicity testing. For example, the pH of acid bogs can be as low as 3.5, well below the 6.5 lower limit for toxicity testing (Table 3).

2. Comparison to wetland water chemistry. Natural levels and variability of those cofactors should be identified as well as possible for each major wetland type of interest. Wetlands-related information can be accumulated through consultation with wetland researchers, through literature searches, and from monitoring agencies.

3. Comparison of species lists. Species lists of fish, invertebrates, and plants should be compiled for each wetland type and compared to lists of species used for testing each criterion. Lists should be evaluated on two levels: a) Species level - Are the species used for toxicity testing representative (the same species or genera, or "similar" in terms of sensitivity to toxicants) of the species found in the wetland type? b) Family level - Does the wetland contain suitable representatives for each of the families listed in the minimum family requirements?[8,11] Consultation with fish and invertebrate specialists, plant ecologists, and wetlands expe -s will be necessary to do this comparison.

Adoption of Existing Water Quality Criterion

The existing water quality criterion should be suitable for that wetland type if the following are true:

1. Important cofactor levels are not naturally exceeded in the wetland to a degree that might seriously affect toxicity or availability of the chemical. Would toxicity likely be higher, lower, or not influenced by typical levels or extremes of a particular cofactor in a particular wetland type?

2. Sufficient species or genera used for aquatic toxicity testing are found in the wetland type so that the minimum family requirements can be met by resident wetland species. Consultation between wetland scientists and criteria experts will be necessary in many cases to make judgements on how well-represented some wetland types are.

3. The criterion itself is not naturally exceeded in the wetland.

DEVELOPING SITE-SPECIFIC CRITERIA

When one or more of these stipulations is not true or when insufficient data are available, more evaluation is advisable. Again, consultation between wetland scientists and criteria experts might be helpful in prioritizing those wetland types for which additional protection, or additional research, might be needed for some chemicals. Once a priority list for further evaluation is established, an approach to obtaining the additional required data can be determined. It might be possible to group wetlands by type, and possibly by designated use, and then develop site-specific criteria for all wetlands of that type in the State.

SECTION 5

EXAMPLE ANALYSES

Evaluations of the applicability of the six criteria listed in Table 2 will be made for two sets of wetland data, including shallow marshes and prairie potholes. The analyses in these examples were made with limited data for each wetland type and are preliminary. They have been compiled to be used only as illustrations of the usefulness of this approach.

EXAMPLE 1

The first example is based on a wetland study taking place in southcentral Minnesota. The wetlands are being studied to evaluate the effects of disturbance on water quality, as well as the effects of pesticides on wetland communities. Therefore chemical and biological data have been collected.[18]

Classification

The wetland study sites are primarily shallow marshes (freshwater palustrine, persistent emergent, semi-permanently or seasonally-flooded, according to Cowardin[15]), dominated by *Phalaris* (reed canary grass) and *Typha* (cattails), but also include a small number of wet meadow/seasonally-flooded wetlands, deep marsh, shrub/scrub + woody wetlands, and ponds.

Steps 1 and 2: Identification of Cofactors and Comparison to Wetland Water Chemistry

Cofactors are identified for criteria chemicals in Table 4. Some water quality characteristics averaged for 5 seasons for the Minnesota wetlands are summarized in Table 5.

Although some water chemistry conditions in the shallow marshes were within the ranges of the aquatic toxicity testing conditions, others were exceeded (Table 3). Wetland values for pH were well within the 6.5-9.0 range allowed for testing, so criteria having pH as a possible cofactor affecting toxicity and/or biological availability should not be underprotective because of pH effects. As Table 4 shows, PCP, chromium(VI), zinc, and cyanide can be more toxic at low pH values, so a very acidic wetland might require additional evaluation in regard to pH. The PCP criterion has an adjustment factor for pH, which indicates that enough suitable data are available to allow this relationship to be incorporated into the criterion.

Hardness values were not available for these marshes, but were probably fairly low since alkalinity was low. Table 4 lists hardness as a cofactor for zinc and chromium(VI). Table 1 reveals that the zinc criterion has an adjustment factor for hardness, so any effect of hardness on zinc toxicity and/or biological availability is already included in the criterion and does not have to be considered further. Chromium(VI) is more toxic at low alkalinity and hardness, but the criterion was derived using soft water and should be protective for the wetlands.

Total organic carbon (TOC) was highly variable in the wetlands and generally well above the 5 mg/L limit for toxicity testing. However parathion and zinc, the two criteria with TOC cofactor effects, have reduced toxicity and/or biological availability at high levels of organic matter (Table 4), so criteria should be protective.

Dissolved oxygen (DO) was highly variable in the wetlands and reached very low levels in late summer. The shallow waters of the marshes were extremely warm on hot summer days. Toxicity and/or biological availability is increased by low DO and high temperatures for PCBs, PCP, and cyanide. These relationships will require further evaluation.

Step 3: Comparisons of Species Lists

In Step 3, fish, invertebrates, and plants inhabiting the wetlands are compared to species used in testing each criterion. For these examples, only the acute toxicity lists have been consulted. A list of genera common to both the marshes and to the toxicity tests was compiled for each criterion. When identical species were not found, species from the same genus were compared to determine whether habitat requirements are suitable enough to include them as representative species for these wetlands. The shortened list of marsh species the same as, or similar to, species used for toxicity testing was examined to determine whether the minimum family requirements for acute toxicity tests could be met for each criterion. Table 6 contains a list of marsh genera that could be used to fulfill minimum family requirements for each criterion. Appendix A contains a list of the sources that have been consulted in making this comparison.

The aquatic species found in the Minnesota wetlands were fairly well-represented by the acute toxicity test species for the six chemicals used in this example. The percentages of total species tested that have not been found in these wetlands were below 50% for all six criteria (Table 7). Except for PCBs, for which no plant value is available, plant species tested overlapped with species occurring in the wetlands. The absence of salmonids in wetlands was the only consistent omission.

Of all the species tested, the salmonids are the most sensitive to PCP and cyanide and are much more sensitive than most invertebrate species. The inclusion of highly sensitive salmonid data in the criteria calculations probably ensures that these two criteria are adequately protective when applied to wetlands not containing this sensitive family (not considering cofactor effects). It would perhaps be more important to consider the effects of the absence of salmonids in Minnesota marshes for criteria where salmonids are among the least sensitive species, including parathion and chromium(VI). In this case, the presence of salmonid toxicity data in the criterion calculation, despite their absence from the wetlands, could possibly cause the criterion to be less restrictive than is appropriate for the wetland.

Salmonids do not occur in the wetlands included in this example. Three criteria were missing an additional required taxonomic group (from Table 6: PCBs, chromium(VI), and cyanide). There are certainly representatives of this taxonomic group (nonarthropod/nonchordate) inhabiting the wetlands, but the genera used for toxicity tests did not correspond to the wetland genera. These three criteria have the least species on the acute toxicity list, so there are less species to compare to, in relation to the other criteria (Table 7). Toxicity experts and wetland biologists might be able to fill some of these data gaps by reaching conclusions on the suitability of wetland species to fulfill the minimum family requirements.

EXAMPLE 2

This example is based on data for a number of oligosaline prairie pothole wetlands in southcentral North Dakota.[19,20] Oligosaline is defined as ranging from 0.5-5 g/kg salinity, or specific conductance of 800-8,000 µS/cm at 25°C.[15] The chemical types of the majority of wetlands used in this example include magnesium bicarbonate, magnesium sulfate, and sodium sulfate.[20]

Classification

Wetlands included in this example are semipermanent (cover type 4 of the classification system developed by Stewart and Kantrud for the glaciated prairie region)[21], containing wet meadow, shallow marsh, and deep marsh. Classification of these wetlands based on the Cowardin system can be found in Kantrud et al.[20]

Steps 1 and 2: Identification of Cofactors and Comparison to Wetland Water Chemistry

Cofactors are identified for criteria chemicals in Table 4. Water quality data for the prairie pothole wetlands are summarized in Table 8. A comparison of water chemistry conditions for the prairie potholes with standard toxicological testing conditions (Table 3) reveals a number of differences.

These wetlands are extremely alkaline and saline compared to water used for freshwater toxicity testing. Salinity (reported as specific conductance) can vary greatly over the year and is concentrated by the high rates of evaporation and transpiration that take place in the summer. A number of the wetlands have pH values above the 6.5-9.0 range that the criteria are designed to protect. No data were available for total organic carbon (TOC), but dissolved organic carbon values from other prairie pothole systems were generally well above the TOC limit of 5 mg/L used for toxicity testing.[22] As in Example 1, hardness can be eliminated from consideration as a cofactor, because toxicity and/or biological availability is decreased as hardness increases. Similarly, the probable high TOC levels would decrease toxicity and/or biological availability for zinc and chromium(VI). The high pH values should cause decreased toxicity and/or biological availability. Bioavailability of zinc is reduced in high ionic strength waters such as these.

Dissolved oxygen (DO) levels drop in the winter and in middle to late summer, allowing anoxic conditions to develop. Although no aquatic temperature data were available, the Dakotas have moderately hot summers (mean July temperature of 22.3°C).[20] The shallow waters of the prairie potholes probably become very warm in late summer, corresponding with low DO levels. Toxicity and/or biological availability is increased by low DO and high temperatures for PCBs, PCP, and cyanide. These relationships will require further evaluation.

Step 3: Comparisons of Species Lists

Semi-permanent prairie pothole wetlands are generally shallow and eutrophic. Water levels fluctuate greatly, as does salinity. The cold winters can cause some of the wetlands to freeze to the bottom. Both winterkill and summerkill, caused by the effects of lack of oxygen, can occur. Fish can survive only in semipermanent wetlands that have connections to deeper water habitat. The only native fishes known to occur in semi-permanent prairie potholes are fathead minnow (Pimephales promelas) and brook stickleback (Culaea inconstans).[20]

The invertebrate taxa of prairie potholes are typical of other eutrophic, alkaline systems in the United States. Macroinvertebrate species assemblages are highly influenced by hydroperiod and salinity in these systems, and species diversity drops as salinity increases.[20] Care must be taken in aggregating large salinity ranges into one wetland type (i.e. "oligosaline" may be too broad a class in terms of species representativeness). Comparisons of species typical of the wetlands with the criteria species lists reveals some major differences. For example, a large proportion of the aquatic insects tested for each criterion are found in flowing water, and therefore might not be characteristic of prairie pothole aquatic insects. Although many species of aquatic insects are found in these wetlands[20], there are not many suitable aquatic insects on the criteria species lists to compare to resident wetland species. Prairie pothole wetlands do not harbor Decapods (crayfish and shrimp), another common group for testing. Eubranchiopods (fairy, tadpole, and clam shrimp) are commonly found in prairie pothole wetlands[20], but only one representative of this group has been used to establish criteria, and that species was not on the list for any of the criteria used as examples here. Except for PCBs, for which no plant value is available, plant species tested do overlap with species occurring in the wetlands. Appendix B contains sources used in making comparisons.

The above discussion has obvious implications for determining applicability of criteria based on suitability of species. As Table 7 shows, the percentages of species tested for each criterion that have not been found in prairie potholes are rather high (up to 67%). There are more gaps in the minimum family requirements for fish and chordates (Table 9) than were found for the Minnesota marsh example. The lack of fish in these wetlands dictates that amphibians or other chordates be used to fill these family requirements. The paucity of fish in these wetlands again has relevance to the protectiveness of the criteria. Fish are the most sensitive group tested for PCP and cyanide, so these criteria may have an added "buffer" of protection (in relation to the other criteria used as examples) when applied with no modifications to this wetland type.

SUMMARY OF THE EXAMPLE ANALYSES

The conclusions discussed below should be considered as examples only. They should not be considered final for these wetland types.

Cofactor Effects

Based on this simple analysis, the only cofactors that potentially could cause criteria to be underprotective were DO and temperature. The low DO and high temperatures common in both wetland types in mid to late summer could cause increased toxicity and/or biological availability for PCBs, PCP, and cyanide. Cofactor effects for chromium(VI), zinc, and parathion were either not important under the chemical conditions encountered in these wetlands or should result in criteria being more, rather than less, protective for the wetland biota. Based on water quality characteristics, it can be concluded that chromium(VI), zinc, and parathion criteria are probably adequately protective of these wetland types with no acute modification.

The importance of the DO and temperature relationship requires further evaluation for PCBs, PCP, and cyanide. Chemists and wetlands experts should be consulted and further literature reviews should be completed to evaluate the need for additional toxicity tests. If it is determined that a modification to a criterion is warranted, seasonal site-specific criteria might be appropriate in this case. The indicator species procedure could be used, requiring toxicity tests using site water on one fish and one invertebrate. The tests could be done at the high temperatures and low DO found in late summer in the wetlands.

Species Comparisons

The Salmonidae are a required family group for establishing a Final Acute Value and yet are not present in either of the wetland types used as examples. This evaluation is most concerned with ensuring that criteria are adequately protective, so the absence of this family in the wetlands should only be considered a problem if the unmodified criterion (which includes the Salmonidae) might be underprotective. This would most likely be true for parathion and chromium(VI).

For several criteria, some family requirements are not fulfilled because the available toxicity data for that taxonomic group do not include wetland species or genera ("NT" in Tables 6 and 9). While this document made comparisons at the genus level, others have made comparisons at the family level to determine if the species listed in the criteria document is a member of a family that exists at the site.[16] Issues related to species comparisons should be addressed through discussion with criteria experts and wetlands ecologists and through further literature review.

The absence of fish in prairie potholes to fill the "other chordates" category for cyanide, zinc, chromium(VI), and PCBs may warrant additional toxicity tests and site-specific modifications. The only other fish likely to be present in these wetlands is the brook stickleback (*Culaea inconstans*)[20] which was not tested for any of the six criteria. No non-fish chordates were tested either, so no evaluation of the probable sensitivity of other chordates to these criteria can be made based on the criteria documents.

If it is decided upon more rigorous evaluation that these differences in taxonomic groups warrant additional efforts and development of site-specific criteria, the recalculation procedure can be used. A suitable family, resident in the wetlands, can be added to the list to replace the Salmonidae and/or other missing groups, either through additional toxicity tests or by including additional available data.

Further Evaluation

This approach helps to prioritize wetland types and criteria for further evaluation. It was concluded that zinc, chromium(VI), and parathion criteria require no modification with regard to cofactor effects. PCBs, PCP, and cyanide, however, should be evaluated further in regard to the effects of high temperatures and low DO on toxicity, for both wetland types. The absence of salmonids may be most important for parathion and chromium(VI) in both wetland types. Further consideration should be given to the need for additional tests with chordates from prairie pothole wetlands for cyanide, zinc, chromium(VI) and PCBs, although there is no evidence to suggest that the absence of representative wetland chordates from the test species will result in underprotective criteria.

This type of evaluation, done for a number of wetland types and criteria, can be combined with information on the types of pollutants that threaten particular wetland types. In this way wetland types requiring additional evaluation and perhaps eventually some additional toxicity testing for particular pollutants can be prioritized based on adequacy of existing criteria, potential threats to the system, and resources available for testing. These examples illustrate the need for wetland scientists to work closely with criteria experts. Expert judgement is needed to evaluate the significance of the gaps in the available data.

SECTION 6

CONCLUSIONS

The efficient use of limited resources dictates that criteria and standards for wetlands be developed by making good use of the wealth of data that has been accumulated for other surface waters. This report focused on the application of numeric aquatic life criteria to wetlands. The numeric aquatic life criteria are designed to protect aquatic life and their uses. The criteria are conservative, and for most wetland types are probably protective or overprotective.

A simple, inexpensive evaluation technique has been proposed in this document for detecting wetland types that might be underprotected for some chemicals by existing criteria. The approach relies on information contained in criteria documents, data regarding species composition and water quality characteristics for the wetland types of interest, and consultation with experts. It is intended to be used as a screening tool for prioritizing those wetland types that require additional evaluations and research.

Two tests of the approach demonstrated that it can be used to identify cases in which criteria might be underprotective, but further evaluation and close coordination among regulatory agencies, wetland scientists, and criteria experts are needed to determine when actual modifications to the criteria are necessary.

Site-specific guidelines for modifying the numeric criteria should be appropriate for use on wetlands in cases where additional evaluations reveal that modifications are needed. The approach described in this document can be used to compile lists of the most commonly under-represented species and the most frequently encountered chemicals. Aquatic toxicity tests can then be conducted which would apply to a number of wetland types.

Information obtained with this approach can be used to prioritize further evaluations and research, identify gaps in data, and make further testing more efficient, but has some limitations. It does not adequately address the importance of plants in wetland systems and applies only to the aquatic component of wetlands. It relies on species assemblage and water quality data that are not available for some wetland types. For these reasons, a meeting of wetland scientists and criteria experts is recommended to discuss the need for this type of evaluation, the utility of this approach, and possible alternative approaches.

The application of numeric criteria to wetlands is just one part of a large effort to develop wetland standards and criteria. The development of biocriteria, sediment criteria, and wildlife criteria will help to ensure that all components of the wetland resource are adequately protected.

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APPENDIX A

SOURCES USED IN SPECIES HABITAT IDENTIFICATION FOR MINNESOTA MARSHES

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Personal Communications:
P. DeVore and C. Richards of the Natural Resources Research Institute, Duluth, Minnesota.
G. Montz, Minnesota Dept. of Natural Resources.

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P. DeVore and A. Hershey of the Natural Resources Research Institute, Duluth, Minnesota.
P. Mickelson of the University of Minnesota, Duluth.

APPENDIX B

SOURCES USED IN SPECIES HABITAT IDENTIFICATION FOR PRAIRIE POTHOLES

Fishes:

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TABLE 1. FRESHWATER NUMERIC AQUATIC LIFE CRITERIA*

Chemical	H, T, or pH** Dependent	Chemical	H, T, or pH** Dependent
Organochlorines:		Metals:	
Aldrin		Aluminum	
Chlordane		Arsenic(III)	
DDT		Cadmium	H
Dieldrin		Chromium(III)	H
Endosulfan		Chromium(VI)	
Endrin		Copper	H
Heptachlor		Lead	H
Lindane		Mercury	
PCBs		Nickel	H
Pentachlorophenol	pH	Selenium	
		Silver	H
Organophosphates:		Zinc	H
Chlorpyrifos			
Parathion		Others:	
		Ammonia	pH, T
		Chloride	
		Chlorine	
		Cyanide	
		Dissolved oxygen	T

* Summarized from individual criteria documents. Chemicals that have adjustment factors built into the criteria are indicated.

** H = Hardness, T = Temperature.

TABLE 2. SUITABILITY OF WETLAND SPECIES TO FILL MINIMUM FAMILY REQUIREMENTS FOR SIX CRITERION CHEMICALS

Required Taxonomic Group	PCBs	Parathion	PCP	Cyanide	Zinc	Chromium(VI)
Salmonid	NP*	NP	NP	NP	NP	NP
Other Fish	Y**	Y	Y	Y	Y	Y
Other Chordate	Y	Y	Y	Y	Y	Y
Planktonic Crustacean	Y	Y	Y	Y	Y	Y
Benthic Crustacean	Y	Y	Y	Y	Y	Y
Insect	Y	Y	Y	Y	Y	Y
Nonarthropod-Nonchordate	NT***	Y	Y	Y	Y	Y
Another Insect or New Phylum	Y	Y	Y	NT	Y	Y

*NP Not present: Taxonomic group not present in most wetland types.
**Y Wetland genera represented adequately.
***NT Not tested: Available toxicity data does not include sufficient wetland species.

TABLE 3. SOME CONDITIONS RECOMMENDED FOR DILUTION WATER FOR WATER QUALITY CRITERIA TESTING[17]

Characteristic	Freshwater	Saltwater
Total organic carbon	<5 mg/L	<20 mg/L[a]
Particulate matter	<5 mg/L	<20 mg/L[a]
pH	6.5-9.0	Stenohaline 8.0 Euryhaline 7.7 Range <0.2
Hardness (mg/L as $CaCO_3$)	Soft water 40-48 Range <5 mg/L[b]	
Salinity		Stenohaline 34 g/kg Euryhaline 17 g/kg Range <2 g/kg[c]
Dissolved oxygen	60-100% saturation[d]	60-100% saturation[d]
Temperature	+/- 5 °C of water[e] of origin	

[a] <5 mg/L for tests other than saltwater bivalve molluscs.
[b] Or 10% of average, whichever is higher.
[c] Or 20% of average, whichever is higher.
[d] For flow-through tests (40-100% for static tests).
[e] For invertebrates only.

TABLE 4. EFFECTS OF COFACTORS ON CRITERION CHEMICAL TOXICITY

	COFACTORS: Effect of Greater Value								
	pH	TOC	TURB	TEMP	DO	H	IONIC	S	NUTR/ORG
Organochlorines:									
Aldrin									
Chlordane									
DDT		+					+	+	
Dieldrin									
Endrin									
Heptachlor									
Lindane									
Endosulfan				+		O			
PCBs				+	-				
Pentachlorophenol	-			+	-				
Toxaphene		-		+					
Organophosphates:									
Parathion		-							
Chlorpyrifos									
Metals:									
Arsenic (III)		-		+	+				
ium		-		?	0	-	-	-	?
nium (VI)	-					-			
C. mium (III)						-			
Copper	-	-		+		-			
Lead		-?	-			-			
Mercury	-			+	-	0			
Nickel	-			+		-		-	
Selenium		-		-		-			
Silver		-				-			
Zinc	-	-			+	-	-	+	
Aluminum	-	-							-
Other:									
Chlorine	0					0	?		
Cyanide	-			±	-		0		
Ammonia	+			+	-		-		
Chloride				+?		-?	0		
DO									

+: increased toxicity/mobility
O: no effect on toxicity/mobility
-: decreased toxicity/mobility
TOC: total organic carbon
TURB: turbidity
C: ionic strength/cations

?: tested and found inconclusive
: not discussed in criteria document
±: short-term increase/long-term decrease
DO: dissolved oxygen H: hardness
NUTR/ORG: nutrients/organic acids
S: salinity

TABLE 5. WATER CHEMISTRY FOR SELECTED MINNESOTA MARSHES*

Water Quality Characteristic	Mean Value	Range	Comparison with Standard Testing Conditions
pH (pH units)	7.1	6.1 - 7.6	Within range
Total organic carbon (mg/L)	20	5 - 60	High
Dissolved oxygen (mg/L)	8.2	0.4 - 15.4	Seasonally low
Hardness (mg/L as $CaCO_3$)	No data	-	-
Alkalinity (mg/L as $CaCO_3$)	8	4 - 14	-
Temperature (°C)	11.9	0.3 - 31.0	Seasonal extremes
Turbidity (NTU)	33	1 - 412	-

* Data taken from Detenbeck (1990), n=42 wetlands.[18]

TABLE 6. COMPARISON OF TEST SPECIES WITH MINNESOTA MARSH BIOTA FOR SIX CRITERIA

Required Taxonomic Group	PCBs	Parathion	PCP
Salmonid	NP[d]	NP	NP
Other Fish[a]	Micropterus	Lepomis	Micropterus
Other Chordate	Pimephales	Pimephales	Rana
Planktonic Crustacean	Daphnia	Daphnia	Daphnia
Benthic Crustacean	unknown amphipod	Orconectes	Orconectes
Insect	Ishnura[b]	Chironomus	Tanytarsus
Nonarthropod-Nonchordate	NT[e]	unknown[c] nematodes/ annelids	unknown[c] nematodes/ annelids
Another Insect or New Phylum	Tanytarsus	Ishnura	unknown amphipod/ isopod
Aquatic Plant	NT	alga	Lemna

continued

TABLE 6, CONTINUED

Required Taxonomic Group	Cyanide	Zinc	Chromium(VI)
Salmonid	NP	NP	NP
Other Fish[a]	Perca	Lepomis	Lepomis
Other Chordate	Lepomis	Pimephales	Pimephales
Planktonic Crustacean	Daphnia	Daphnia	Daphnia
Benthic Crustacean	unknown[c] amphipod/ isopod	unknown[c] amphipod/ isopod	Orconectes
Insect	Tanytarsus	Argia[b]	Chironomus
Nonarthropod-Nonchordate	Physa	Physa	Physa
Another Insect or New Phylum	NT	unknown[c] annelid/ nematode	NT
Aquatic Plant	Lemna	Lemna	alga

a Fish were sampled in water bodies associated with some of the wetlands, not in the wetlands themselves.
b Probable or seen as an adult.
c Unknown species from these taxa found in wetlands. May or may not be similar in terms of habitat requirements, etc. to species used in toxicity tests.
d Not present: Taxonomic group not present in wetland type.
e Not tested: Available toxicity data does not include sufficient wetland species.

TABLE 7. NUMBER OF SPECIES TESTED FOR ACUTE CRITERIA AND PERCENTAGE OF TEST SPECIES THAT ARE NOT FOUND IN MINNESOTA MARSHES OR OLIGOSALINE PRAIRIE POTHOLES*

Chemical	Species Used to Establish FAV** (Total Number)	Not Present in Marshes (Per cent)	Not Present in Prairie Potholes (Per cent)
PCBs	10	30%	40%
Parathion	37	43%	64%
PCP	37	22%	43%
Cyanide	17	29%	65%
Zinc	45	45%	67%
Chromium(VI)	33	27%	64%

* Remainder of percentage includes both those species that are known to occur in these wetlands and those species that may occur in the wetlands, but insufficient data are available.

** Final Acute Value.

TABLE 8. WATER QUALITY CHARACTERISTICS FOR OLIGOSALINE PRAIRIE POTHOLES[a]

Water Quality Characteristic	Mean Value	Range	Comparison with Standard Testing Conditions
pH (pH units)	8.9	7.4 - 10.3[b]	High
Total organic carbon (mg/L)	No data[c]	-	-
Dissolved oxygen (ppm)	No data[d]	-	-
Hardness (mg/L as $CaCO_3$)	No data[e]	-	-
Alkalinity (mg/L as $CaCO_3$)	650	230 - 1300	High
Temperature (°C)	No data[f]	-	-
Specific conductance (µS/cm at 25°C)	3568	750 - 8000	-

a Data summarized from Swanson et al. (1988).[19]
b N=27 wetlands.
c Dissolved organic carbon data for Manitoba prairie potholes ranged from 0.4-102 mg/L, and for Nebraska, from 20-60 mg/L in one study and 139-440 mg/L in another study.[22]
d Winterkill, caused by low dissolved oxygen under ice, occurs in many of these lakes.
e An estimate of hardness based on alkalinity values gives a mean of 760 mg/L as $CaCO_3$.
f Region is characterized by very cold winters and warm summers.

TABLE 9. COMPARISON OF TEST SPECIES WITH PRAIRIE POTHOLE BIOTA FOR SIX CRITERIA

Required Taxonomic Group	PCBs	Parathion	PCP
Salmonid	NP	NP	NP
Other Fish	Pimephales	Pimephales	Pimephales
Other Chordate	NT	Pseudacris[a]	Rana[a]
Planktonic Crustacean	Daphnia	Daphnia	Daphnia
Benthic Crustacean	Gammarus[a]	Gammarus[a]	Hyalella
Insect	damselfly[b]	Peltodytes	Tanytarsus[b]
Nonarthropod-Nonchordate	NT	tubificid worm[b]	tubificid worm[b]
Another Insect or New Phylum	Tanytarsus[b]	Chironomus	Physa
Aquatic Plant	NT	Microcystis	Lemna

TABLE 9, CONTINUED

Required Taxonomic Group	Cyanide	Zinc	Chromium(VI)
Salmonid	NP	NP	NP
Other Fish	Pimephales	Pimephales	Pimephales
Other Chordate	NT	NT	NT
Planktonic Crustacean	Daphnia	Daphnia	Daphnia
Benthic Crustacean	Gammarus[a]	Gammarus[a]	Hyalella
Insect	Tanytarsus[b]	Argia[b]	Chironomus[a]
Nonarthropod-Nonchordate	Physa[a]	Physa[a]	Physa[a]
Another Insect or New Phylum	NT	tubificid worm[b]	damselfly[b]
Aquatic Plant	Lemna	Lemna	Nitzschia

a Genus is present in the wetlands; may not be same species.
b Species representative of that taxonomic group from criteria testing lists probably present in prairie potholes, but no actual data available.

APPENDIX F

COORDINATION BETWEEN THE
ENVIORNMENTAL PROTECTION AGENCY,
FISH AND WILDLIFE SERVICE AND NATIONAL
MARINE FISHERIES SERVICE REGARDING
DEVELOPMENT OF WATER QUALITY CRITERIA AND
WATER QUALITY STANDARDS UNDER
THE CLEAN WATER ACT

July 27, 1992

Signed by:

Ralph Morgenweck, Assistant Director
Fish and Wildlife Enhancement
U.S. Fish and Wildlife Service

Dr. Tudor Davies, Director
Office of Science and Technology
U.S. Environmental Protection Agency

Dr. Nancy Foster, Director
Office of Protected Resources
National Marine Fisheries Service

Coordination Between the Environmental Protection Agency, the Fish and Wildlife Service and the National Marine Fisheries Service Regarding the Development of Water Quality Criteria and Water Quality Standards Under the Clean Water Act

PURPOSE

This memorandum sets forth the procedures to be followed by Fish and Wildlife Service (FWS), the National Marine Fisheries Service (NMFS), and the Environmental Protection Agency (EPA) to insure compliance with Section 7 of the ESA in the development of water quality criteria published pursuant to Section 304(a) of the Clean Water Act (CWA) and the adoption of water quality standards under Section 303(c) of the CWA. Consultation will be conducted pursuant to 50 C.F.R. Part 402. Regional Offices of EPA and the Services may establish agreements, consistent with these procedures, specifying how they will implement this Memorandum.

I. BACKGROUND

A. Guiding Principles

The agencies recognize that EPA's water quality criteria and standards program has the express goal of ensuring the protection of the biological integrity of U.S. waterbodies and associated aquatic life. The agencies also recognize that implementation of the CWA in general, and the water quality standards program in particular, is primarily the responsibility of states. EPA's role in this program is primarily to provide scientific guidance to states to aid in their development of water quality standards and to oversee state adoption and revision of standards to insure that they meet the requirements of the CWA.

In view of the decentralized nature of EPA's water quality standards program responsibilities, and the agencies' desire to carry out their respective statutory obligations in the most efficient manner possible, the agencies believe that consultation should occur, to the maximum extent possible, at the national level. Should additional coordination be necessary on the regional level, the procedures outlined below are designed to insure that the Services are integrated early into EPA's oversight of the states' standards adoption process so that threatened and endangered species concerns can be addressed in the most efficient manner possible.

B. Legal Authorities

1. Section 7 of the ESA

Section 7 of the ESA contains several provisions which require federal agencies to take steps to conserve endangered and threatened species, and which impose the responsibility on agencies to insure, in consultation with the appropriate Service, that certain actions are not likely to jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of their critical habitat. Section 7 also requires agencies to confer with the appropriate Service regarding actions affecting species or critical habitat that have been proposed for listing or designation under section 4, but for which no final rule has been issued.

In particular, section 7(a)(1) provides that federal agencies shall "utilize their authorities in furtherance of the purposes of [the ESA] by carrying out programs for the conservation of endangered species and threatened species . . ." Section 7(a)(2) requires federal agencies to insure, in consultation with the appropriate Service, that actions which they authorize, fund or carry out are "not likely to jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of habitat of such species which is determined . . . to be critical." Section 7(a)(4) requires a conference for actions that are "likely to jeopardize the continued existence" of species proposed for listing or that are likely to "result in the destruction or adverse modification" of proposed critical habitat.

The procedures for consultation between federal agencies and the Services under section 7 of the ESA are contained in 50 C.F.R. Part 402. Section 402.14 of these regulations requires that agencies engage in formal consultation with the appropriate Service where any action of that agency may affect listed species or critical habitat. Formal consultation is not required if the action agency prepares a biological assessment or consults informally with the appropriate Service and obtains the written concurrence of the Service that the action is not likely to adversely affect listed species or critical habitat. Formal consultation culminates in the issuance of a biological opinion by the Service which concludes whether the agency action is likely to jeopardize the continued existence of a listed species or result in the destruction or adverse modification of critical habitat.[1] If the Service makes a jeopardy finding, the opinion shall include reasonable and prudent alternatives, if any, to avoid jeopardy. If the Service anticipates that an action would result in an incidental take of a listed species (defined in 50 C.F.R. 402.02), the Service shall include an incidental take statement and reasonable and prudent measures that the Director considers necessary or appropriate to minimize such impact. Such measures cannot alter the basic design, location, scope, duration or timing of the action and may involve only minor changes.

Evaluation of the potential effects of an agency action on listed species or their habitat is to be based upon the best scientific and commercial data available or which can be obtained prior to or during the consultation. 50 C.F.R. 402.14(d).

2. Water Quality Standards Development Under the CWA

Section 303 of the Clean Water Act provides for the development by states of water quality standards which are designed to protect the public health or welfare, enhance the quality of water and serve the purposes of the CWA. Such standards consist of designated uses of waterways (e.g., protection and propagation of fish, shellfish, and wildlife) and criteria which will insure the protection of designated uses.

Under the CWA, the development of water quality standards is primarily the responsibility of States. However, pursuant to section 304(a) of the CWA, EPA from time to time publishes water quality criteria which serve as scientific guidance to be used by states in establishing and revising water quality standards. These EPA criteria are not enforceable requirements, but are recommended criteria levels which states may adopt as part of their legally enforceable water quality standards; states may adopt other scientifically defensible criteria in lieu of EPA's recommended criteria. See 40 C.F.R. 131.11(b).

Standards adopted by states constitute enforceable requirements with which permits issued by States or EPA under section 402 of the Clean Water Act must assure compliance. CWA section 301(b)(1)(C). Under section 303(c) of the CWA, EPA must review water quality standards adopted by states and either approve them if the standards meet the requirements of the CWA or disapprove them if the standards fail to do so. However, EPA's disapproval of state water quality standards does not alter the enforceable requirements with which CWA section 402 permits must comply, because the state standards remain in full force and

[1] Any reference in this document to "jeopardy" for purposes of section 7 of the ESA is intended also to include the concept of destruction or adverse modification of critical habitat.

effect under state law. The state-adopted standards remain effective for all purposes of the CWA until they are revised by the state or EPA promulgates federal water quality standards applicable to the state.

II. PROCEDURES

A. Development of Water Quality Criteria Guidance Under Section 304(a) of the CWA

EPA will integrate the Services into its criteria development process by consulting with the Services regarding the effect EPA's existing aquatic life criteria (and any new or revised criteria) may have on listed endangered or threatened species. References below to endangered or threatened species include species proposed to be listed by the Services. In addition, EPA will include the Service(s) on the aquatic life criteria guidelines revision committee which is currently revising the methodological guidelines that will form the technical basis for future criteria adopted by EPA.

1. Consultation on Existing Criteria

EPA has developed and published aquatic life criteria documents explaining the scientific basis for aquatic life criteria that EPA has published. EPA will consult with the appropriate Service regarding the aquatic life criteria as described below.

Step 1: Services' Identification of Species that May Be Affected By Water Quality Degradation

The Services and EPA will request their regional offices to identify the endangered and threatened species within their jurisdictions that may be affected by degraded water quality. Each Service will provide EPA with a consolidated list of these species. To facilitate this process, the initial species list will include information identifying the areas where such species are located, a description of the pollutants causing the water quality problems affecting the species (if known) and any other relevant information provided by the Services' regional offices. In future consultations, the Services will provide a species list, as required in 50 C.F.R. Part 402, and access to any relevant data concerning identified species.

Step 2: EPA Initiation of Informal Consultation and Performance of Biological Assessment

Based upon a review of information provided by the Services under Step 1, above, and any other information available to EPA (as described by 50 C.F.R. 402.12(f)(1)-(5)), EPA will determine what species may be affected by the aquatic life criteria and will request informal consultation with the appropriate Service regarding such species. EPA will submit to the appropriate Service a biological assessment that evaluates the potential effects of the criteria levels on those species. The biological assessment will be developed in an iterative process between EPA and the Service (initially involving submission of a "pilot" assessment addressing 2 or 3 chemicals), and is expected to contain the information listed in the Appendix of this Memorandum.

Step 3: Further steps Based on Results of Biological Assessment

Based upon the findings made by EPA in the Biological Assessment, the consultation will proceed as follows (see 50 C.F.R. 402.12(k)):

- For those criteria/species where EPA determines that there is no effect, EPA will not initiate formal consultation.

- For those criteria/species where there is a "may affect" situation, and EPA determines that the species is not likely to be adversely affected, the appropriate Service will either concur or nonconcur with this finding under Step 4, below.

- Where EPA finds that a species is likely to be adversely affected, formal consultation will occur between the agencies under Step 5, below.

Step 4: Service Reviews Biological Assessment and Responds to EPA

Within 30 days after EPA submits a complete biological assessment to the Service, the Service will provide EPA with a written response that concurs or does not concur with any findings by EPA that species are not likely to be adversely affected by EPA's criteria. For those species/criteria where the Service concurs in EPA's finding, consultation is concluded and no formal consultation will be necessary. For any species/criteria where the Service does not concur in EPA's finding, formal consultation on the criteria/species will occur under step 5, below (see 50 C.F.R. 402.14).

Step 5: Formal Consultation

Formal consultation will occur between the agencies (coordinated by the agencies' headquarters' offices) beginning on the date the Service receives a written consultation request from EPA regarding those species where EPA or the Service believe there is likely to be an adverse affect, as determined under steps 3 and 4, above. The consultation will be based on the information supplied by EPA in the biological assessment and other relevant information that is available or which can feasibly be collected during the consultation period (see 50 C.F.R 402.14(d)). The Service will issue a biological opinion regarding whether any of the species are likely to be jeopardized by the pollutant concentrations contained in EPA's criteria. Any jeopardy conclusion will specify the specific pollutant(s), specie(s) and geographic area(s) which the Service believes is covered by such conclusion. If the Service makes a jeopardy finding, it will identify any available reasonable and prudent alternatives, which may include, but are not limited to, those specified below. EPA will notify the Service of its action regarding acceptance and implementation of all reasonable and prudent alternatives.

1. EPA works with the relevant State during its pending triennial review period to insure adoption (or revision) of water quality standards for the specific pollutants and water bodies that will avoid jeopardy. Such adoption or revision may include adoption of site-specific criteria in accordance with EPA's site-specific criteria guidance, or other basis for establishing more stringent criteria.

2. EPA disapproves relevant portions of state water quality standards (see 40 C.F.R. 131.21) and initiates promulgation of federal standards for the relevant water body (see 40 C.F.R. 131.22) that will avoid jeopardy. Where appropriate, EPA will promulgate such standards on an expedited basis.

2. Service Participation in Committee Revising Criteria's Methodological Guidelines

An EPA committee is currently charged with revising and updating the methodological guidelines which will in the future be followed by EPA when it issues new 304(a) water quality criteria. The Service(s) will become a member of the workgroup as an observer/advisor to insure that the methodological guidelines take into account the need to protect endangered and threatened species. The guidelines will be subject to peer review and public notice and comment prior to being finalized. During the public comment period, the Services will provide the agencies' official position on the guidelines.

3. Consultation with the Services on New or Revised Aquatic Life Criteria and New Wildlife and Sediment Criteria

When EPA develops and publishes new or revised aquatic life criteria and new wildlife and sediment criteria under section 304(a), EPA will request consultation with the Services on such criteria, which will proceed in accordance with the procedures outlined in section II.A.1 of this Memorandum.

B. EPA Review of State Water Quality Standards Under Section 303 of the CWA

In order to insure timely resolution of issues related to protection of endangered or threatened species, EPA and the Services will coordinate in the following manner with regard to state water quality standards that are subject to EPA review and approval under section 303(c) of the CWA.

1. Participation of the Services in EPA/State Planning Meetings

Unless other procedures ensuring adequate coordination are agreed to by the regional offices of EPA and the Service(s), EPA regional offices will request in writing that the Services attend EPA/state meetings where the state's plan for reviewing and possibly revising water quality standards is discussed. The invitation will include any preliminary plans submitted by the state and any suggestions offered by EPA to the state that will be discussed at the planning meeting, as well as a request for the Services to suggest any additional topics of concern to them.

Service staff will attend the planning session and be prepared to identify areas where threatened and endangered species that may be affected by the proposed action may be present in the state and to provide access to any data available to the Services in the event additional discussions will need to occur. If the Service does not intend to attend the planning meeting, it will notify the EPA regional office in writing. If threatened and endangered species may be present in the waters subject to the standards, such notice will include a species list.

2. Consultation on EPA Review of State Water Quality Standards Where Federally Listed Species Are Present

Except in those cases where the Service's Director, at the Washington Office level, requests consultation, EPA may complete its review and approval of state water quality standards without requesting consultation where (1) the state's criteria are as stringent as EPA's section 304(a) aquatic life criteria and consultation between EPA and the appropriate Service on EPA's criteria has resulted in a Service concurrence with an EPA finding of "not likely to adversely affect," a "no jeopardy" biological opinion (or EPA's implementation of a reasonable and prudent alternative contained in the Service's "jeopardy" biological opinion), and EPA's adherence to the terms and conditions of any incidental take statement and (2) the state has designated use classifications for the protection and propagation of fish and shellfish.

However, if a State adopts water quality standards consistent with the provisions of the preceding paragraph, but the Service believes that consultation may be necessary in either of the circumstances described below, only the Service's Director, at the Washington Office level, may request consultation with EPA. Such consultation may be necessary (1) where review of a state water quality standard identifies factors not considered during the relevant water quality criterion review under this Memorandum which indicate that the standard may affect an endangered or threatened species, or (2) where new scientific information not available during the earlier consultation indicates that the criterion, as implemented through the state water quality standard, may affect endangered or threatened species in a manner or to an extent not considered in the earlier consultation.

If a state submits water quality standards containing aquatic life criteria that are less stringent than EPA's section 304(a) aquatic life criteria, or use designations that do not provide for the protection and propagation of fish and shellfish, EPA will consult with the appropriate Service regarding the state's standards. EPA's request for formal or informal consultation (as appropriate) shall be made as early as possible in the standards development process (e.g., when standards regulation are under development by the state). The EPA region should not wait until standards are formally submitted by the state to request such consultation.

If a state water quality standard under review by EPA relates to specie(s), pollutant(s) and geographic area(s) that were the subject of a jeopardy opinion issued by the Service under section II.A. of this Memorandum, EPA will consider the opinion (and any reasonable and prudent alternatives specified by the Service) and take action that, in EPA's judgment, will insure that water quality standards applicable to the state are not likely to jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of species' critical habitat. EPA will notify the Service that issued the biological opinion of its action, in accordance with 50 C.F.R. 402.15.

Except in those cases where the Service's Director, at the Washington Office level, requests consultation, EPA may take action pursuant to CWA section 303(c)(4) to promulgate federal standards applicable to a water of the state without requesting consultation where (1) the aquatic life criteria promulgated by EPA are no less stringent than EPA's section 304(a) criteria guidance and consultation between EPA and the Service on EPA's criteria has resulted in a Service concurrence with an EPA finding of "not likely to adversely affect," a "no jeopardy" biological opinion (or EPA's implementation of a reasonable and prudent alternative contained in the Service's "jeopardy" biological opinion), and EPA's adherence to the terms and conditions of any incidental take statement and (2) the applicable use classifications provide for the protection and propagation of fish and shellfish.

However, if EPA promulgates water quality standards consistent with the provisions of the previous paragraph, but the Service believes that consultation may be necessary in either of the circumstances described below, only the Service's Director, at the Washington Office level, may request consultation with EPA. Such consultation may be necessary (1) where review of the water quality standard identifies factors not considered during the relevant water quality criterion review under this Memorandum which indicate that the standard may affect an endangered or threatened species, or (2) where new scientific information not available during the earlier consultation indicates that the criterion, as implemented through the water quality standard, may affect endangered or threatened species in a manner or to an extent not considered in the earlier consultation.

III. Revisions to Agreement

EPA and the Services may jointly revise the procedures agreed to in this document based upon the experience gained in the pilot consultation on EPA's aquatic life criteria or other experience in the implementation of the above procedures.

IV. Third Party Enforcement

The terms of this Memorandum are not intended to be enforceable by any party other than the signatories hereto.

V. Reservation of Agency Positions

No party to this Memorandum waives any administrative claims, positions or interpretations it may have with respect to the applicability or the enforceability of the ESA.

VI. Effective Date; Termination

This Memorandum will become effective upon signature by each of the parties hereto. Any of the parties may withdraw from this Memorandum upon 60 days' written notice to the other parties;

provided that any Section 7 consultation covered by the terms of this Memorandum that is pending at the time notice of withdrawal is received by the parties, and those activities covered by this Memorandum that begin the consultation process with the 60-day notice period, will continue to be governed by the procedures in this Memorandum.

Ralph Morgenweck, Assistant Director — Date
Fish and Wildlife Enhancement
U.S. Fish and Wildlife Service

Dr. Tudor T. Davies, Director — Date
Office of Science and Technology
U.S Environmental Protection Agency

Dr. Nancy Foster, Director — Date
Office of Protected Resources
National Marine Fisheries Service

APPENDIX

Expected Contents of EPA's Biological Assessment

I. Introduction/Overview

A. Benefits of pollution reduction relative to endangered and threatened species/description of the ESA

B. Role of Water Quality Standards under the CWA

C. Overview of water quality criteria (philosophy, objectives, methodology)

D. Discussion of comparative sensitivity of listed species (and surrogates) with criteria database

E. Description of Fact Sheet contents

- data included
- description of how specific criteria derived
- description of logic/thought processes supporting findings of effect on listed species

II. Fact Sheets

Pollutant-specific fact sheets will be compiled which evaluate the available data and reach conclusions regarding the findings of effect of the criteria on endangered and threatened species. The fact sheets will be presented largely in tabular, graph form.

A. Summary of toxicological relationships (from water quality criteria documents)

1. acute (acute lethality)
2. chronic (life processes at risk)
3. plants
4. residues
5. other key data
6. updated information through review of ACQUIRE database and other key data

B. Taxa at risk vis-a-vis listed species (through use of surrogates, where appropriate)

C. Impact of other water quality factors -- describe effects such as environmental variability, ph, hardness, temperature, etc.

D. **Assessment of impact on listed species**

Findings to be made regarding whether each criteria (1) "may affect" and/or (2) is likely to adversely affect, listed species.

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APPENDIX G

Questions and Answers on: Antidegradation

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

United States
Environmental Protection
Agency

Office of Water
Regulations and Standards
Washington, DC 20460

August 1985

Water

EPA

Questions & Answers on: Antidegradation

QUESTIONS AND ANSWERS ON ANTIDEGRADATION

INTRODUCTION

This document provides guidance on the antidegradation policy component of water quality standards and its application. The document begins with the text of the policy as stated in the water quality standards regulation, 40 CFR 131.12 (40 FR 51400, November 8, 1983), the portion of the Preamble discussing the antidegradation policy, and the response to comments generated during the public comment period on the regulation.

The document then uses a question and answer format to present information about the origin of the policy, the meaning of various terms, and its application in both general terms and in specific examples. A number of the questions and answers are closely related; the reader is advised to consider the document in its entirety, for a maximum understanding of the policy, rather than to focus on particular answers in isolation. While this document obviously does not address every question which could arise concerning the policy, we hope that the principles it sets out will aid the reader in applying the policy in other situations. Additional guidance will be developed concerning the application of the antidegradation policy as it affects pollution from nonpoint sources. Since Congress is actively considering amending the Clean Water Act to provide additional programs for the control of nonpoint sources, EPA will await the outcome of congressional action before proceeding further.

EPA also has available, for public information, a summary of each State's antidegradation policy. For historical interest, limited copies are available of a Compendium of Department of the Interior Statements on Non-Degradation of Interstate Waters, August, 1968. Information on any aspect of the water quality standards program and copies of these documents may be obtained from:

David Sabock, Chief
Standards Branch (WH-585)
Office of Water Regulations and Standards
Environmental Protection Agency
401 M. Street, S.W.
Washington, D.C. 20460

This document is designated as Appendix A to Chapter 2 - General Program Guidance (antidegradation) of the Water Quality Standards Handbook, December 1983.

James M. Conlon, Acting Director
Office of Water Regulations
and Standards

Federal Register / Vol. 48, No. 217 / Tuesday, November 8, 1983 / Rules and Regulations 51407

§ 131.12 Antidegradation policy.

(a) The State shall develop and adopt a statewide antidegradation policy and identify the methods for implementing such policy pursuant to this subpart. The antidegradation policy and implementation methods shall, at a minimum, be consistent with the following:

(1) Existing instream water uses and the level of water quality necessary to protect the existing uses shall be maintained and protected.

(2) Where the quality of the waters exceed levels necessary to support propagation of fish, shellfish, and wildlife and recreation in and on the water, that quality shall be maintained and protected unless the State finds, after full satisfaction of the intergovernmental coordination and public participation provisions of the State's continuing planning process, that allowing lower water quality is necessary to accommodate important economic or social development in the area in which the waters are located. In allowing such degradation or lower water quality, the State shall assure water quality adequate to protect existing uses fully. Further, the State shall assure that there shall be achieved the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best management practices for nonpoint source control.

(3) Where high quality waters constitute an outstanding National resource, such as waters of National and State parks and wildlife refuges and waters of exceptional recreational or ecological significance, that water quality shall be maintained and protected.

(4) In those cases where potential water quality impairment associated with a thermal discharge is involved, the antidegradation policy and implementing method shall be consistent with section 316 of the Act.

51402 Federal Register / Vol. 48, No. 217 / Tuesday, November 8, 1983 / Rules and Regulations

Antidegradation Policy

The preamble to the proposed rule discussed three options for changing the existing antidegradation policy. Option 1, the proposed option, provided simply that uses attained would be maintained. Option 2 stated that not only would uses attained be maintained but that high quality waters, i.e. waters with quality better than that needed to protect fish and wildlife, would be maintained (that is, the existing antidegradation policy minus the "outstanding natural resource waters" provision). Option 3 would have allowed changes in an existing use if maintaining that use would effectively prevent any future growth in the community or if the benefits of maintaining the use do not bear a reasonable relationship to the costs.

Although there was support for Option 2, there was greater support for retaining the full existing policy, including the provision on outstanding National resource waters. Therefore, EPA has retained the existing antidegradation policy (Section 131.12) because it more accurately reflects the degree of water quality protection desired by the public, and is consistent with the goals and purposes of the Act.

In retaining the policy EPA made four changes. First, the provisions on maintaining and protecting existing instream uses and high quality waters were retained, but the sentences stating that no further water quality degradation which would interfere with or becon .njurious to existing instream uses i' .owed were deleted. The delet .s were made because the terms "in! ere" and "injurious" were subject tc .sinterpretation as precluding any ivity which might even momentarily add pollutants to the water. Moreover, we believe the deleted sentence was intended merely as a restatement of the basic policy. Since the rewritten provision, with the addition of a phrase on water quality described in the next sentence, stands alone as expressing the basic thrust and intent of the antidegradation policy, we deleted the confusing phrases. Second, in § 131.12(a)(1) a phrase was added requiring that the level of water quality necessary to protect an existing use be maintained and protected. The previous policy required only that an existing use be maintained. In § 131.12(a)(2) a phrase was added that "In allowing such degradation or lower water quality, the State shall assure water quality adequate to protect existing uses fully". This means that the full use must continue to exist even if some change in water quality may be permitted. Third, in the first sentence of § 131.12(a)(2) the wording was changed from ". . . significant economic or social development . . ." to ". . . important economic or social development. . . ." In the context of t antidegradation policy the word .mportant" strengthens the intent of protecting higher quality waters. Although common usage of the words may imply otherwise, the correct definitions of the two terms indicate that the greater degree of environmental protection is afforded by the word "important."

Fourth, § 131.12(a)(3) dealing with the designation of outstanding National resource waters (ONRW) was changed to provide a limited exception to the absolute "no degradation" requirement. EPA was concerned that waters which properly could have been designated as ONRW were not being so designated because of the flat no degradation provision, and therefore were not being given special protection. The no degradation provision was sometimes interpreted as prohibiting *any* activity (including temporary or short-term) from being conducted. States may allow some limited activities which result in temporary and short-term changes in water quality. Such activities are considered to be consistent with the intent and purpose of an ONRW. Therefore, EPA has rewritten the provision to read ". . . that water quality shall be maintained and protected," and removed the phrase "No degradation shall be allowed. . . ."

In its entirety, the antidegradation policy represents a three-tiered approach to maintaining and protecting various levels of water quality and uses. At its base (Section 131.12(a)(1)), all existing uses and the level of water quality necessary to protect those uses must be maintained and protected. This provision establishes the absolute floor of water quality in all waters of the United States. The second level (Section 131.12(a)(2)) provides protection of actual water quality in areas where the quality of the waters exceed levels necessary to support propagation of fish, shellfish, and wildlife and recreation in and on the water ("fishable/swimmable"). There are provisions contained in this subsection to allow some limited water quality degradation after extensive public involvement, as long as the water quality remains adequate to be "fishable/swimmable." Finally § 131.23(a)(3) provides special protection of waters for which the ordinary use classifications and water quality criteria do not suffice, denoted "outstanding National resource water." Ordinarily most people view this subsection as protecting and maintaining the highest quality waters of the United States: that is clearly the thrust of the provision. It does, however, also offer special protection for waters of "ecological significance." These are water bodies which are important, unique, or sensitive ecologically, but whose water quality as measured by the traditional parameters (dissolved oxygen, pH, etc.) may not be particularly high or whose character cannot be adequately described by these parameters.

Antidegradation Policy

EPA's proposal, which would have limited the antidegradation policy to the maintenance of existing uses, plus three alternative policy statements described in the preamble to the proposal notice, generated extensive public comment. EPA's response is described in the Preamble to this final rule and includes a response to both the substantive and philosophical comments offered. Public comments overwhelmingly supported retention of the existing policy and EPA did so in the final rule.

EPA's response to several comments dealing with the antidegradation policy, which were not discussed in the Preamble are discussed below.

Option three contained in the Agency's proposal would have allowed the possibility of exceptions to maintaining existing uses. This option was either criticized for being illegal or was supported because it provided additional flexibility for economic growth. The latter commenters believed that allowances should be made for carefully defined exceptions to the absolute requirement that uses attained must be maintained. EPA rejects this contention as being totally inconsistent with the spirit and intent of both the Clean Water Act and the underlying philosophy of the antidegradation policy. Moreover, although the Agency specifically asked for examples of where the existing antidegradation policy had precluded growth, no examples were provided. Therefore, wholly apart from technical legal concerns, there appears to be no justification for adopting Option 3.

Most critics of the proposed antidegradation policy objected to removing the public's ability to affect decisions on high quality waters and outstanding national resource waters. In attempting to explain how the proposed antidegradation policy would be implemented, the Preamble to the proposed rule stated that no public participation would be necessary in certain instances because no change was being made in a State's water quality standard. Although that statement was technically accurate, it left the mistaken impression that all public participation was removed from the discussions on high quality waters and that is not correct. A NPDES permit would have to be issued or a 208 plan amended for any deterioration in water quality to be "allowed". Both actions require notice and an opportunity for public comment. However, EPA retained the existing policy so this issue is moot. Other changes in the policy affecting ONRW are discussed in the Preamble.

QUESTIONS AND ANSWERS ON ANTIDEGRADATION

1. WHAT IS THE ORIGIN OF THE ANTIDEGRADATION POLICY?

The basic policy was established on February 8, 1968, by the Secretary of the U.S. Department of the Interior. It was included in EPA's first water quality standards regulation 40 CFR 130.17, 40 FR 55340-41, November 28, 1975. It was slightly refined and repromulgated as part of the current program regulation published on November 8, 1983 (48 FR 51400, 40 CFR §131.12). An antidegradation policy is one of the minimum elements required to be included in a State's water quality standards.

2. WHERE IN THE CLEAN WATER ACT (CWA) IS THERE A REQUIREMENT FOR AN ANTIDEGRADATION POLICY OR SUCH A POLICY EXPRESSED?

There is no explicit requirement for such a policy in the Act. However, the policy is consistent with the spirit, intent, and goals of the Act, especially the clause "... restore and <u>maintain</u> the chemical, physical and biological integrity of the Nation's waters" (§101(a)) and arguably is covered by the provision of 303(a) which made water quality standard requirements under prior law the "starting point" for CWA water quality requirements.

3. CAN A STATE JUSTIFY NOT HAVING AN ANTIDEGRADATION POLICY IN ITS WATER QUALITY STANDARDS?

EPA's water quality standards regulation requires each State to adopt an antidegradation policy and specifies the minimum requirements for a policy. If not included in the standards regulation of a State, the policy must be specifically referenced in the water quality standards so that the functional relationship between the policy and the standards is clear. Regardless of the location of the policy, it must meet all applicable requirements.

4. WHAT HAPPENS IF A STATE'S ANTIDEGRADATION POLICY DOES NOT MEET THE REGULATORY REQUIREMENTS?

If this occurs either through State action to revise its policy or through revised Federal requirements, the State would be given an opportunity to make its policy consistent with the regulation. If this is not done, EPA has the authority to promulgate the policy for the State pursuant to Section 303(c)(4) of the Clean Water Act.

5. WHAT COULD HAPPEN IF A STATE FAILED TO IMPLEMENT ITS ANTI-DEGRADATION POLICY PROPERLY?

If a State issues an NPDES permit which violates the required antidegradation policy, it would be subject to a discretionary EPA veto under Section 402(d) or to a citizen challenge. In addition to actions on permits, any wasteload allocations and total maximum daily loads violating the antidegradation policy are subject to EPA disapproval and EPA promulgation of a new wasteload allocation/total maximum daily load under Section 303(d) of the Act. If a significant pattern of violation was evident, EPA could constrain the award of grants or possibly revoke any Federal permitting capability that had been delegated to the State. If the State issues a §401 certification (for an EPA-issued NPDES permit) which fails to reflect the requirements of the antidegradation policy, EPA will, on its own initiative, add any additional or more stringent effluent limitations required to ensure compliance with Section 301(b)(1)(C). If the faulty §401 certification related to permits issued by other Federal agencies (e.g. a Corp of Engineers Section 404 permit), EPA could comment unfavorably upon permit issuance. The public, of course, could bring pressure upon the permit issuing agency.

6. WILL THE APPLICATION OF THE ANTIDEGRADATION POLICY ADVERSELY IMPACT ECONOMIC DEVELOPMENT?

This concern has been raised since the inception of the antidegradation policy. The answer remains the same. The policy has been carefully structured to minimize adverse effects on economic development while protecting the water quality goals of the Act. As Secretary Udall put it in 1968, the policy serves "...the dual purpose of carrying out the letter and spirit of the Act without interfering unduly with further economic development" (Secretary Udall, February 8, 1968). Application of the policy could affect the levels and/or kinds of waste treatment necessary or result in the use of alternate sites where the environmental impact would be less damaging. These effects could have economic implications as do all other environmental controls.

7. WHAT IS THE PROPER INTERPRETATION OF THE TERM "AN EXISTING USE"?

An existing use can be established by demonstrating that fishing, swimming, or other uses have actually occurred since November 28, 1975, <u>or</u> that the water quality is suitable to allow such uses to occur (unless there are physical problems which prevent the use regardless of water quality). An example of the latter is an area where shellfish are propagating and surviving in a biologically suitable habitat and are available and suitable for harvesting. Such facts clearly establish that shellfish harvesting is an "existing" use, not one dependent on improvements in water quality. To argue otherwise would be to say that the only time an aquatic protection use "exists" is if someone succeeds in catching fish.

8. THE WATER QUALITY STANDARDS REGULATION STATES THAT "EXISTING USES AND THE LEVEL OF WATER QUALITY NECESSARY TO PROTECT THE EXISTING USES SHALL BE MAINTAINED AND PROTECTED." HOW FULLY AND AT WHAT LEVEL OF PROTECTION IS AN EXISTING USE TO BE PROTECTED IN ORDER TO SATISFY THE ABOVE REQUIREMENT?

No activity is allowable under the antidegradation policy which would partially or completely eliminate any existing use whether or not that use is designated in a State's water quality standards. The aquatic protection use is a broad category requiring further explanation. Species that are in the water body and which are consistent with the designated use (i.e., not aberrational) must be protected, even if not prevalent in number or importance. Nor can activity be allowed which would render the species unfit for maintaining the use. Water quality should be such that it results in no mortality and no significant growth or reproductive impairment of resident species. (See Question 16 for situation where an aberrant sensitive species may exist.) Any lowering of water quality below this full level of protection is not allowed. A State may develop subcategories of aquatic protection uses but cannot choose different levels of protection for like uses. The fact that sport or commercial fish are not present does not mean that the water may not be supporting an aquatic life protection function. An existing aquatic community composed entirely of invertebrates and plants, such as may be found in a pristine alpine tributary stream, should still be protected whether or not such a stream supports a fishery. Even though the shorthand expression "fishable/swimmable" is often used, the actual objective of the act is to "restore and maintain the chemical, physical, and biological integrity of our Nation's waters (section 101(a)).[1/] The term "aquatic life" would more accurately reflect the protection of the aquatic community that was intended in Section 101(a)(2) of the Act.

9. IS THERE ANY SITUATION WHERE AN EXISTING USE CAN BE REMOVED?

In general, no. Water quality may sometimes be affected, but an existing use, and the level of water quality to protect it must be maintained (§131.12(a)(1) and (2) of the regulation). However, the State may limit or not designate such a use if the reason for such action is non-water quality related. For example, a State may wish to impose a temporary shellfishing ban to prevent overharvesting and ensure an abundant population over the long run, or may wish to restrict swimming from heavily trafficked areas. If the State chooses,

1/ Note: "Fishable/swimmable" is a term of convenience used in the standards program in lieu of constantly repeating the entire text of Section 101(a)(2) goal of the Clean Water Act. As a short-hand expression it is potentially misleading.

for non-water quality reasons, to limit use designations, it must still adopt criteria to protect the use if there is a reasonable likelihood it will actually occur (e.g. swimming in a prohibited water). However, if the State's action is based on a recognition that water quality is likely to be lowered to the point that it no longer is sufficient to protect and maintain an existing use, then such action is inconsistent with the antidegradation policy.

10. HOW DOES THE REQUIREMENT THAT THE LEVEL OF WATER QUALITY NECESSARY TO PROTECT THE EXISTING USE(S) BE MAINTAINED AND PROTECTED, WHICH APPEARS IN §131.12(a)(1),(2), AND (3) OF THE WATER QUALITY STANDARDS REGULATION, ACTUALLY WORK?

Section 131.12(a)(1), as described in the Preamble to the regulation, provides the absolute floor of water quality in all waters of the United States. This paragraph applies a minimum level of protection to all waters. However, it is most pertinent to waters having beneficial uses that are less than the Section 101(a)(2) goals of the Act. If it can be proven, in that situation, that water quality exceeds that necessary to fully protect the existing use(s) and exceeds water quality standards but is not of sufficient quality to cause a better use to be achieved, then that water quality may be lowered to the level required to fully protect the existing use as long as existing water quality standards and downstream water quality standards are not affected. If this does not involve a change in standards, no public hearing would be required under Section 303(c). However, public participation would still be provided in connection with the issuance of a NPDES permit or amendment of a 208 plan. If, however, analysis indicates that the higher water quality does result in a better use, even if not up to the Section 101(a)(2) goals, then the water quality standards must be upgraded to reflect the uses presently being attained (§131.10(i)).

Section 131.12(a)(2) applies to waters whose quality exceeds that necessary to protect the Section 101(a)(2) goals of the Act. In this case, water quality may not be *lowered to less than the level necessary to fully protect* the "fishable /swimmable" uses and other existing uses and may be lowered even to those levels only after following all the provisions described in §131.12(a)(2). This requirement applies to individual water quality parameters.

Section 131.12(a)(3) applies to so-called outstanding National Resource (ONRW) waters where the ordinary use classifications and supporting criteria are not appropriate. As described in the Preamble to the water quality standards regulation "States may allow some limited activities which result in temporary and short-term changes in water quality," but such changes in water quality should not alter the essential character or special use which makes the water an ONRW. (See also pages 2-14,-15 of the Water Quality Standards Handbook.)

Any one or a combination of several activities may trigger the antidegradation policy analysis as discussed above. Such activities include a scheduled water quality standards review, the establishment of new or revised wasteload allocations NPDES permits, the demonstration of need for advanced treatment or request by private or public agencies or individuals for a special study of the water body.

11. WILL AN ACTIVITY WHICH WILL DEGRADE WATER QUALITY, AND PRECLUDE AN EXISTING USE IN ONLY A PORTION OF A WATER BODY (BUT ALLOW IT TO REMAIN IN OTHER PARTS OF THE WATER BODY) SATISFY THE ANTIDEGRADATION REQUIREMENT THAT EXISTING USES SHALL BE MAINTAINED AND PROTECTED?

No. Existing uses must be maintained in <u>all</u> parts of the water body segment in question other than in restricted mixing zones. For example, an activity which lowers water quality such that a buffer zone must be established within a previous shellfish harvesting area is inconsistent with the antidegradation policy. (However, a slightly different approach is taken for fills in wetlands, as explained in Question 13.)

12. DOES ANTIDEGRADATION APPLY TO POTENTIAL USES?

No. The focus of the antidegradation policy is on protecting existing uses. Of course, insofar as existing uses and water quality are protected and maintained by the policy the eventual improvement of water quality and attainment of new uses may be facilitated. The use attainability requirements of §131.10 also help ensure that attainable potential uses are actually attained. (See also questions 7 and 10.)

13. FILL OPERATIONS IN WETLANDS AUTOMATICALLY ELIMINATE ANY EXISTING USE IN THE FILLED AREA. HOW IS THE ANTIDEGRADATION POLICY APPLIED IN THAT SITUATION?

Since a literal interpretation of the antidegradation policy could result in preventing the issuance of any wetland fill permit under Section 404 of the Clean Water Act, and it is logical to assume that Congress intended some such permits to be granted within the framework of the Act, EPA interprets §131.12 (a)(1) of the antidegradation policy to be satisfied with regard to fills in wetlands if the discharge did not result in "significant degradation" to the aquatic ecosystem as defined under Section 230.10(c) of the Section 404(b)(1) guidelines. If any wetlands were found to have better water quality than "fishable/ swimmable", the State would be allowed to lower water quality to the no significant degradation level as long as the requirements of Section 131.12(a)(2) were followed. As for the ONRW provision of antidegradation (131.(a)(2)(3)), there is no difference in the way it applies to wetlands and other water bodies.

14. IS POLLUTION RESULTING FROM NONPOINT SOURCE ACTIVITIES SUBJECT TO PROVISIONS OF THE ANTIDEGRADATION POLICY?

Nonpoint source activities are not exempt from the provisions of the antidegradation policy. The language of Section 131.12 (a)(2) of the regulation: "Further, the State shall assure that there shall be achieved the highest statutory and regulatory requirements for all new and existing point sources and all cost-effective and reasonable best mangement practices for nonpoint source control" reflects statutory provisions of the Clean Water Act. While it is true that the Act does not establish a regulatory program for nonpoint sources, it clearly intends that the BMPs developed and approved under sections 205(j), 208 and 303(e) be agressively implemented by the States. As indicated in the introduction, EPA will be developing additional guidance in this area.

15. IN HIGH QUALITY WATERS, ARE NEW DISCHARGERS OR EXPANSION OF EXISTING FACILITIES SUBJECT TO THE PROVISIONS OF ANTIDEGRADATION?

Yes. Since such activities would presumably lower water quality, they would not be permissible unless the State finds that it is necessary to accommodate important economic or social development (Section 131.12(a)(2). In addition the minimum technology based requirements must be met, including new source performance standards. This standard would be implemented through the wasteload and NPDES permit process for such new or expanded sources.

16. A STREAM, DESIGNATED AS A WARM WATER FISHERY, HAS BEEN FOUND TO CONTAIN A SMALL, APPARENTLY NATURALLY OCCURRING POPULATION OF A COLD-WATER GAME FISH. THESE FISH APPEAR TO HAVE ADAPTED TO THE NATURAL WARM WATER TEMPERATURES OF THE STREAM WHICH WOULD NOT NORMALLY ALLOW THEIR GROWTH AND REPRODUCTION. WHAT IS THE EXISTING USE WHICH MUST BE PROTECTED UNDER SECTION 131.12(a)(1)?

Section 131.12(a)(1) states that "Existing instream water uses and level of water quality necessary to protect the existing uses shall be maintained and protected." While sustaining a small cold-water fish population, the stream does not support an existing use of a "cold-water fishery." The existing stream temperatures are unsuitable for a thriving cold-water fishery. The small marginal population is an artifact and should not be employed to mandate a more stringent use (true cold-water fishery) where natural conditions are not suitable for that use.

A use attainability analysis or other scientific assessment should be used to determine whether the aquatic life population is in fact an artifact or is a stable population requiring water quality protection. Where species appear in areas not normally expected, some adaptation may have occurred and site-specific criteria may be appropriately developed. Should the cold-water fish population consist of a threatened or endangered species, it may require protection under the Endangered Species Act. Otherwise the stream need only be protected as a warm water fishery.

17. HOW DOES EPA'S ANTIDEGRADATION POLICY APPLY TO A WATERBODY WHERE A CHANGE IN MAN'S ACTIVITIES IN OR AROUND THAT WATERBODY WILL PRECLUDE AN EXISTING USE FROM BEING FULLY MAINTAINED?

If a planned activity will forseeably lower water quality to the extent that it no longer is sufficient to protect and maintain the existing uses in that waterbody, such an activity is inconsistent with EPA's antidegradation policy which requires that existing uses are to be maintained. In such a circumstance the planned activity must be avoided or adequate mitigation or preventive measures must be taken to ensure that the existing uses and the water quality to protect them will be maintained.

In addition, in "high quality waters", under §131.12(a)(2), before any lowering of water quality occurs, there must be: 1) a finding that it is necessary in order to accommodate important economical or social development in the area in which the waters are located, (2) full satisfaction of all intergovernmental coordination and public participation provisions and (3) assurance that the highest statutory and regulatory requirements and best management practices for pollutant controls are achieved. This provision can normally be satisfied by the completion of Water Quality Management Plan updates or by a similar process that allows for public participation and intergovernmental coordination. This provision is intended to provide relief only in a few extra-ordinary circumstances where the economic and social <u>need</u> for the activity clearly outweighs the benefit of maintaining water quality above that required for "fishable/swimmable" water, and the two cannot both be achieved. The burden of demonstration on the individual proposing such activity will be very high. In any case, moreover, the existing use must be maintained and the activity shall not preclude the maintenance of a "fishable/swimmable" level of water quality protection.

18. WHAT DOES EPA MEAN BY "...THE STATE SHALL ENSURE THAT THERE SHALL BE ACHIEVED THE HIGHEST STATUTORY AND REGULATORY REQUIREMENTS FOR ALL NEW AND EXISTING POINT SOURCES AND ALL COST EFFECTIVE AND REASONABLE BEST MANAGEMENT PRACTICES FOR NON-POINT SOURCE CONTROL" (§131.12(a)(2)?

This requirement ensures that the limited provision for lowering water quality of high quality waters down to "fish-able /swimmable" levels will not be used to undercut the Clean Water Act requirements for point source and non-point source pollution control. Furthermore, by ensuring compliance with such statutory and regulatory controls, there is less chance that a lowering of water quality will be sought in order to accommodate new economic and social development.

19. WHAT DOES EPA MEAN BY "...IMPORTANT ECONOMIC OR SOCIAL DEVELOPMENT IN THE AREA IN WHICH THE WATERS ARE LOCATED" IN 131.1 2(a)(2)?

This phrase is simply intended to convey a general concept regarding what level of social and economic development could be used to justify a change in high quality waters. Any more exact meaning will evolve through case-by-case application under the State's continuing planning process. Although EPA has issued suggestions on what might be considered in determining economic or social impacts, the Agency has no predetermined level of activity that is defined as "important".

20. IF A WATER BODY WITH A PUBLIC WATER SUPPLY DESIGNATED USE IS, FOR NON-WATER QUALITY REASONS, NO LONGER USED FOR DRINKING WATER MUST THE STATE RETAIN THE PUBLIC WATER SUPPLY USE AND CRITERIA IN ITS STANDARDS?

Under 40 CFR 131.10(h)(1), the State may delete the public water supply use designation and criteria if the State adds or retains other use designations for the waterbodies which have more stringent criteria. The State may also delete the use and criteria if the public water supply is not an "existing use" as defined in 131.3 (i.e., achieved on or after November 1975), as long as one of the §131.10(g) justifications for removal is met.

Otherwise, the State must maintain the criteria even if it restricts the actual use on non-water quality grounds, as long as there is any possibility the water could actually be used for drinking. (This is analogous to the swimming example in the preamble.)

21. WHAT IS THE RELATIONSHIP BETWEEN WASTELOAD ALLOCATIONS, TOTAL MAXIMUM DAILY LOADS, AND THE ANTIDEGRADATION POLICY?

Wasteload allocations distribute the allowable pollutant loadings to a stream between dischargers. Such allocations also consider the contribution to pollutant loadings from non-point sources. Wasteload allocations must reflect applicable State water quality standards including the antidegradation policy. No wasteload allocation can be develped or NPDES permit issued that would result in standard being violated, or, in the case of waters whose quality exceeds that necessary for the Section 101(a)(2) goals of the Act, can result a lowering of water quality unless the applicable public participation, intergovernmental review and baseline control requirements of the antidegradation policy have been met.

22. DO THE INTERGOVERNMENTAL COORDINATION AND PUBLIC PARTICIPATION REQUIREMENTS WHICH ESTABLISH THE PROCEDURES FOR DETERMINING THAT WATER QUALITY WHICH EXCEEDS THAT NECESSARY TO SUPPORT THE SECTION 101(a)(2) GOAL OF THE ACT MAY BE LOWERED APPLY TO CONSIDERING ADJUSTMENTS TO THE WASTELOAD ALLOCATIONS DEVELOPED FOR THE DISCHARGERS IN THE AREA?

Yes. Section 131.12(a)(2) of the water quality standards regulation is directed towards changes in water quality *per se*, not just towards changes in standards. The intent is to ensure that no activity which will cause water quality to decline in existing high quality waters is undertaken without adequate public review. Therefore, if a change in wasteload allocation could alter water quality in high quality waters, the public participation and coordination requirements apply.

23. IS THE ANSWER TO THE ABOVE QUESTION DIFFERENT IF THE WATER QUALITY IS LESS THAN THAT NEEDED TO SUPPORT "FISHABLE/SWIMMABLE" USES?

Yes. Nothing in either the water quality standards or the wasteload allocation regulations requires the same degree of public participation or intergovernmental coordination for such waters as is required for high quality waters. However, as discussed in question 10, public participation would still be provided in connection with the issuance of a NPDES permit or amendment of a 208 plan. Also, if the action which causes reconsideration of the existing wasteloads (such as dischargers withdrawing from the area) will result in an improvement in water quality which makes a better use attainable, even if not up to the "fishable/swimmable" goal, then the water quality standards must be upgraded and full public review is required for any action affecting changes in standards. Although not specifically required by the standards regulation between the triennial reviews, we recommend that the State conduct a use attainability analysis to determine if water quality improvement will result in attaining higher uses than currently designated in situations where significant changes in wasteloads are expected (see question 10).

24. SEVERAL FACILITIES ON A STREAM SEGMENT DISCHARGE PHOSPHORUS-CONTAINING WASTES. AMBIENT PHOSPHORUS CONCENTRATIONS MEET CLASS B STANDARDS, BUT BARELY. THREE DISCHARGERS ACHIEVE ELIMINATION OF DISCHARGE BY DEVELOPING A LAND TREATMENT SYSTEM. AS A RESULT, ACTUAL WATER QUALITY IMPROVES (I.E., PHOSPHORUS LEVELS DECLINE) BUT NOT QUITE TO THE LEVEL NEEDED TO MEET CLASS A (FISHABLE/SWIMMABLE) STANDARDS. CAN THE THREE REMAINING DISCHARGERS NOW INCREASE THEIR PHOSPHORUS DISCHARGE WITH THE RESULT THAT WATER QUALITY DECLINES (PHOSPHORUS LEVELS INCREASE) TO PREVIOUS LEVELS?

Nothing in the water quality standards regulation explicitly prohibits this (see answer to questions 10 and 23). Of course, changes in their NPDES permit limits may be subject to non-water quality constraints, such as BPT or BAT, which may restrict this.

25. SUPPOSE IN THE ABOVE SITUATION WATER QUALITY IMPROVES TO THE POINT THAT ACTUAL WATER QUALITY NOW MEETS CLASS A REQUIREMENTS. IS THE ANSWER DIFFERENT?

Yes. The standards must be upgraded (see answer to question 10).

26. AS AN ALTERNATIVE CASE, SUPPOSE PHOSPHORUS LOADINGS GO DOWN AND WATER QUALITY IMPROVES BECAUSE OF A CHANGE IN FARMING PRACTICES, E.G., INITIATION OF A SUCCESSFUL NON-POINT PROGRAM. ARE THE ABOVE ANSWERS THE SAME?

Yes. Whether the improvement results from a change in point or nonpoint source activity is immaterial to how any aspect of the standards regulation operates. Section 131.10(d) clearly indicates that uses are deemed attainable if they can be achieved by "... cost-effective and reasonable best management practices for nonpoint source control". Section 131.12(a)(2) of the anti-degradation policy contains essentially the same wording.

27. WHEN A POLLUTANT DISCHARGE CEASES FOR ANY REASON, MAY THE WASTELOAD ALLOCATIONS FOR THE OTHER DISCHARGES IN THE AREA BE ADJUSTED TO REFLECT THE ADDITIONAL LOADING AVAILABLE?

This may be done consistent with the antidegradation policy only under two circumstances: (1) In "high quality waters" where after the full satisfaction of all public participation and intergovernmental review requirements, such adjustments are considered necessary to accomodate important economic or social development, and the "threshold" level requirements are met; or (2) in less than "high quality waters", when the expected improvement in water quality will not cause a better use to be achieved, the adjusted loads still meet water quality standards, and the new wasteload allocations are at least as stringent as technology-based limitations. Of course, all applicable requirements of the Section 402 permit regulations would have to be satisfied before a permittee could increase its discharge.

28. HOW MAY THE PUBLIC PARTICIPATION REQUIREMENTS BE SATISFIED?

This requirement may be satisfied in several ways. The State may obviously hold a public hearing or hearings. The State may also satisfy the requirement by providing the opportunity for the public to request a hearing. Activities which may affect several water bodies in a river basin or sub-basin may be considered in a single hearing. To ease the resource burden on both the State and public, standards issues may be combined with hearings on environmental impact statements, water management plans, or permits. However, if this is done, the public must be clearly informed that possible changes in water quality standards are being considered along with other activities. In other words, it is inconsistent with the water quality standards regulation to "back-door" changes in standards through actions on EIS's, wasteload allocations, plans, or permits.

29. WHAT IS MEANT BY THE REQUIREMENT THAT, WHERE A THERMAL DISCHARGE IS INCLUDED, THE ANTIDEGRADATION POLICY SHALL BE CONSISTENT WITH SECTION 316 OF THE ACT?

This requirement is contained in Section 131.12 (a)(4) of the regulation and is intended to coordinate the requirements and procedures of the antidegadation policy with those established in the Act for setting thermal discharge limitations. Regulations implementing Section 316 may be found at 40 CFR 124.66. The statutory scheme and legislative history indicate that limitations developed under Section 316 take precedence over other requirements of the Act.

30. WHAT IS THE RELATIONSHIP BETWEEN THE ANTIDEGRADATION POLICY, STATE WATER RIGHTS USE LAWS AND SECTION 101(g) OF THE CLEAN WATER ACT WHICH DEALS WITH STATE AUTHORITY TO ALLOCATE WATER QUANTITIES?

The exact limitations imposed by section 101(g) are unclear; however, the legislative history and the courts interpreting it do indicate that it does not nullify water quality measures authorized by CWA (such as water quality standards and their upgrading, and NPDES and 402 permits) even if such measures incidentally affect individual water rights; those authorities also indicate that if there is a way to reconcile water quality needs and water quantity allocations, such accomodation should be be pursued. In other words, where there are alternate ways to meet the water quality requirements of the Act, the one with least disruption to water quantity allocations should be chosen. Where a planned diversion would lead to a violation of water quality standards (either the antidegradation policy or a criterion), a 404 permit associated with the diversion should be suitably conditioned if possible and/or additional nonpoint and/or point source controls should be imposed to compensate.

31. AFTER READING THE REGULATION, THE PREAMBLE, AND ALL THESE QUESTIONS AND ANSWERS, I STILL DON'T UNDERSTAND ANTIDEGRADATION. WHOM CAN I TALK TO?

Call the Standards Branch at: (202) 245-3042. You can also call the water quality standards coordinators in each of our EPA Regional offices.

APPENDIX H

Derivation of the 1985 Aquatic Life Criteria

APPENDIX H

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

Derivation of the 1985 Aquatic Life Critera

The following is a summary of the Guidelines for Derivation of Criteria for Aquatic Life. The complete text is found in "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses," available from National Technical Information Service – PB85-227049.

Derivation of numerical national water quality criteria for the protection of aquatic organisms and their uses is a complex process that uses information from many areas of aquatic toxicology. When a national criterion is needed for a particular material, all available information concerning toxicity to and bioaccumulation by aquatic organisms is collected, reviewed for acceptability, and sorted. If enough acceptable data on acute toxicity to aquatic animals are available, they are used to estimate the highest one-hour average concentration that should not result in unacceptable effects on aquatic organisms and their uses. If justified, this concentration is made a function of water quality characteristics such as pH, salinity, or hardness. Similarly, data on the chronic toxicity of the material to aquatic animals are used to estimate the highest four-day average concentration that should not cause unacceptable toxicity during a long-term exposure. If appropriate, this concentration is also related to a water quality characteristic.

Data on toxicity to aquatic plants are examined to determine whether plants are likely to be unacceptably affected by concentrations that should not cause unacceptable effects on animals. Data on bioaccumulation by aquatic organisms are used to determine if residues might subject edible species to restrictions by the U.S. Food and Drug Administration (FDA), or if such residues might harm wildlife that consumes aquatic life. All other available data are examined for adverse effects that might be biologically important.

If a thorough review of the pertinent information indicates that enough acceptable data exists, numerical national water quality criteria are derived for fresh water or salt water or both to protect aquatic organisms and their uses from unacceptable effects due to exposures to high concentrations for short periods of time, lower concentrations for longer periods of time, and combinations of the two.

I. Definition of Material of Concern

A. Each separate chemical that does not ionize substantially in most natural bodies of water should usually be considered a separate material, except possibly for structurally similar organic compounds that exist only in large quantities as commercial mixtures of the various compounds and apparently have similar biological, chemical, physical, and toxicological properties.

B. For chemicals that do ionize substantially in most natural waterbodies (e.g., some phenols and organic acids, some salts of phenols and organic acids, and most inorganic salts and coordination complexes of metals), all forms in chemical equilibrium should usually be considered one material. Each different oxidation state of a metal and each different non-ionizable covalently bonded organometallic compound should usually be considered a separate material.

C. The definition of the material should include an operational analytical component. Identification of a material simply, for example, as "sodium" obviously implies "total sodium" but leaves room for doubt. If "total" is meant, it should be explicitly stated. Even "total" has different operational definitions, some of which do not necessarily measure "all that is there" in all sample. Thus, it is also necessary to reference or describe one analytical method that is intended. The operational analytical component should take into account the analytical and environmental chemistry of the material, the desirability of using the same analytical method on samples from laboratory tests, ambient water and aqueous effluents, and various practical considerations such as labor and equipment requirements and whether the method would require measurement in the field or would allow measurement after samples are transported to a laboratory.

The primary requirements of the operational analytical component are that it be appropriate for use on samples of receiving water, compatible with the available toxicity and bioaccumulation data without making overly hypothetical extrapolations, and rarely result in underprotection or overprotection of aquatic organisms and their uses. Because an ideal analytical measurement will rarely be available, a compromise measurement will usually be used. This compromise measurement must fit with the general approach: if an ambient concentration is lower than the national criterion, unacceptable effects will probably not occur (i.e., the compromise measurement must not err on the side of underprotection when measurements are made on a surface water). Because the chemical and physical properties of an effluent are usually quite different from those of the receiving water, an analytical method acceptable for analyzing an effluent might not be appropriate for analyzing a receiving water, and vice versa. If the ambient concentration *calculated* from a measured concentration in an effluent is higher than the national criterion, an additional option is to *measure* the concentration after dilution of the effluent with receiving water to determine if the measured concentration is lowered by such phenomena as complexation or sorption. A further option, of course, is to derive a site-specific criterion (1,2,3). Thus, the criterion should be based on an appropriate analytical measurement, but the criterion is not rendered useless if an ideal measurement either is not available or is not feasible.

The analytical chemistry of the material might need to be considered when defining the material or when judging the acceptability of some toxicity tests, but a criterion should not be based on the sensitivity of an analytical method. When aquatic organisms are more sensitive than routine analytical methods, the proper solution is to develop better analytical methods, not to underprotect aquatic life.

II. Collection of Data

A. Collect all available data on the material concerning toxicity to, and bioaccumulation by, aquatic animals and plants; FDA action levels (compliance Policy Guide, U.S. Food & Drug Admin. 1981) and chronic feeding studies and long-term field studies with wildlife species that regularly consume aquatic organisms.

B. All data that are used should be available in typed, dated, and signed hard copy (publication, manuscript, letter, memorandum) with enough supporting information to indicate that acceptable test procedures were used and that the results are probably reliable. In some cases, additional written information from the investigator may be needed. Information that is confidential, privileged, or otherwise not available for distribution should not be used.

C. Questionable data, whether published or unpublished, should not be used. Examples would be data from tests that did not contain a control treatment, tests in which too many organisms in the control treatment died or showed signs of stress or disease, and tests in which distilled or deionized water was used as the dilution water without addition of appropriate salts.

D. Data on technical grade materials may be used, if appropriate; but data on formulated mixtures and emulsifiable concentrates of the material may not be used.

E. For some highly volatile, hydrolyzable, or degradable materials, only use data from flow-through tests in which the concentrations of test material were measured often enough with acceptable analytical methods.

F. Data should be rejected if obtained by using:

- Brine shrimp — because they usually occur naturally only in water with salinity greater than 35 g/kg;
- Species that do not have reproducing wild populations in North America; or
- Organisms that were previously exposed to substantial concentrations of the test material or other contaminants.

G. Questionable data, data on formulated mixtures and emulsifiable concentrates, and data obtained with nonresident species or previously exposed organisms may be used to provide auxiliary information but should not be used in the derivation of criteria.

III. Required Data

A. Certain data should be available to help ensure that each of the four major kinds of possible adverse effects receives adequate consideration: results of acute and chronic toxicity tests with representative species of aquatic animals are necessary to indicate the sensitivities of appropriate untested species. However, since procedures for conducting tests with aquatic plants and interpreting the results are not as well developed, fewer data concerning toxicity are required. Finally, data concerning bioaccumulation by aquatic organisms are required only with relevant information on the significance of residues in aquatic organisms.

B. To derive a criterion for freshwater aquatic organisms and their uses, the following should be available:

1. Results of acceptable acute tests (see section IV) with at least one species of freshwater animal in at least eight different families including all of the following:
   - The family Salmonidae in the class Osteichthyes.
   - A second family in the class Osteichthyes, preferably a commercially or recreationally important warmwater species, such as bluegill or channel catfish.
   - A third family in the phylum Chordata (may be in the class Osteichthyes or may be an amphibian, etc.).
   - A planktonic crustacean such as a cladoceran or copepod.
   - A benthic crustacean (ostracod, isopod, amphipod, crayfish, etc.).
   - An insect (mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge, etc.).
   - A family in a phylum other than Arthropoda or Chordata, such as Rotifera, Annelida, Mollusca.
   - A family in any order of insect or any phylum not already represented.
2. Acute-chronic ratios (see section VI) with species of aquatic animals in at least three different families, provided that:
   - At least one is a fish;
   - At least one is an invertebrate; and
   - At least one is an acutely sensitive freshwater species (the other two may be saltwater species).
3. Results of at least one acceptable test with a freshwater alga or vascular plant (see section VIII). If the plants are among the aquatic organisms that are most sensitive to the material, test data on a plant in another phylum (division) should also be available.

4. At least one acceptable bioconcentration factor determined with an appropriate freshwater species, if a maximum permissible tissue concentration is available (see section IX).

C. To derive a criterion for saltwater aquatic organisms and their uses, the following should be available:

   1. Results of acceptable acute tests (see section IV) with at least one species of saltwater animal in at least eight different families, including all of the following:
      - Two families in the phylum Chordata;
      - A family in a phylum other than Arthropoda or Chordata;
      - Either the Mysidae or Penaeidae family;
      - Three other families not in the phylum Chordata (may include Mysidae or Penaeidae, whichever was not used previously); and
      - Any other family.

   2. Acute-chronic ratios (see section VI) with species of aquatic animals in at least three different families, provided that of the three species:
      - At least one is a fish;
      - At least one is an invertebrate; and
      - At least one is an acutely sensitive saltwater species (the other may be an acutely sensitive freshwater species).

   3. Results of at least one acceptable test with a saltwater alga or vascular plant (see section VIII). If plants are among the aquatic organisms most sensitive to the material, results of a test with a plant in another phylum (division) should also be available.

   4. At least one acceptable bioconcentration factor determined with an appropriate saltwater species, if a maximum permissible tissue concentration is available (see section IX).

D. If all required data are available, a numerical criterion can usually be derived, except in special cases. For example, derivation of a criterion might not be possible if the available acute-chronic ratios vary by more than a factor of 10 with no apparent pattern. Also, if a criterion is to be related to a water quality characteristic T (see sections V and VII), more data will be necessary.

   Similarly, if all required data are not available, a numerical criterion should not be derived except in special cases. For example, even if not enough acute and chronic data are available, it might be possible to derive a criterion if the available data clearly indicate that the Final Residue Value should be much lower than either the Final Chronic Value or the Final Plant Value.

E. Confidence in a criterion usually increases as the amount of available pertinent data increases. Thus, additional data are usually desirable.

IV. Final Acute Value

A. Appropriate measures of the acute (short-term) toxicity of the material to a variety of species of aquatic animals are used to calculate the Final Acute Value. The Final Acute Value is an estimate of the concentration of the material, corresponding to a cumulative probability of 0.05 in the acute toxicity values for genera used in acceptable acute tests conducted on the material. However, in some cases, if the Species Mean Acute Value of a commercially or recreationally important species is lower than the calculated Final Acute Value, then that Species Mean Acute Value replaces the calculated Final Acute Value to protect that important species.

B. Acute toxicity tests should have been conducted using acceptable procedures (ASTM Standards E 729 and 724).

C. Except for tests with saltwater annelids and mysids, do not use results of acute tests during which test organisms were fed, unless data indicate that the food did not affect the toxicity of the test material.

D. Results of acute tests conducted in unusual dilution water (dilution water in which total organic carbon or particulate matter exceeded 5 mg/L) should not be used unless a relationship is developed between acute toxicity and organic carbon or particulate matter or unless data show that the organic carbon or particulate matter does not affect toxicity.

E. Acute values should be based on endpoints that reflect the total severe acute adverse impact of the test material on the organisms used in the test. Therefore, only the following kinds of data on acute toxicity to aquatic animals should be used:

1. Tests with daphnids and other cladocerans should be started with organisms less than 24-hours old, and tests with midges should be stressed with second- or third-instar larvae. The result should be the 48-hour EC_{50} based on percentage of organisms immobilized plus percentage of organisms killed. If such an EC_{50} is not available from a test, the 48-hour LC_{50} should be used in place of the desired 48-hour EC_{50}. An EC_{50} or LC_{50} of longer than 48 hours can be used as long as the animals were not fed and the control animals were acceptable at the end of the test.

2. The result of a test with embryos and larvae of barnacles, bivalve molluscs (clams, mussels, oysters, and scallops), sea urchins, lobsters, crabs, shrimp, and abalones should be the 96-hour EC_{50} based on the percentage of organisms with incompletely developed shells plus the percentage of organisms killed. If such an EC_{50} is not available from a test, the lower of the 96-hour EC_{50}, based on the percentage of organisms with incompletely developed shells and the 96-hour LC_{50} should be used in place of the desired 96-hour EC_{50}. If the duration of the test was between 48 and 96 hours, the EC_{50} or LC_{50} at the end of the test should be used.

3. The acute values from tests with all other freshwater and saltwater animal species and older life stages of barnacles, bivalve molluscs, sea urchins, lobsters, crabs, shrimps, and abalones should be the 96-hour EC_{50} based on the percentage of organisms exhibiting loss of equilibrium, plus the percentage of organisms immobilized, plus the percentage of organisms killed. If such an EC_{50} is not available from a test, the 96-hour LC_{50} should be used in place of the desired 96-hour EC_{50}.

4. Tests with single-celled organisms are not considered acute tests, even if the duration was 96 hours or less.

5. If the tests were conducted properly, acute values reported as "greater than" values and those above the solubility of the test material should be used because rejection of such acute values would unnecessarily lower the Final Acute Value by eliminating acute values for resistant species.

F. If the acute toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Acute Equation should be derived based on that water quality characteristic. (Go to section V.)

G. If the available data indicate that one or more life stages are at least a factor of 2 more resistant than one or more other life stages of the same species, the data for the more resistant life stages should not be used in the calculation of the Species Mean Acute Value because a species can be considered protected from acute toxicity only if all life stages are protected.

H. The agreement of the data within and between species should be considered. Acute values that appear to be questionable in comparison with other acute and chronic data for the same species and for other species in the same genus probably should not be used in calculation of a Species Mean Acute Value. For example, if the acute values available for a species or genus differ by more than a factor of 10, some or all of the values probably should not be used in calculations.

I. For each species for which at least one acute value is available, the Species Mean Acute Value should be calculated as the geometric mean of the results of all flow-through tests in which the concentrations of test material were measured. For a species for which no such result is available, the Species Mean Acute Value should be calculated as the geometric mean of all available acute values — i.e., results of flow-through tests in which the concentrations were not measured and results of static and renewal tests based on initial concentrations of test material. (Nominal concentrations are acceptable for most test materials if measured concentrations are not available.)

NOTE: Data reported by original investigators should not be rounded off. Results of all intermediate calculations should be rounded to four significant digits.

NOTE: The geometric mean of N numbers is the N^{th} root of the product of the N numbers. Alternatively, the geometric mean can be calculated by adding the logarithms of the N numbers, dividing the sum by N, and taking the antilog of the quotient. The geometric mean of two numbers is the square root of the product of the two numbers, and the geometric mean of one number is that number. Either natural (base 0) or common (base 10) logarithms can be used to calculate geometric means as long as they are used consistently within each set of data (i.e., the antilog used must match the logarithm used).

NOTE: Geometric means rather than arithmetic means are used here because the distributions of individual organisms' sensitivities in toxicity tests on most materials, and the distributions of species' sensitivities within a genus, are more likely to be lognormal than normal. Similarly, geometric means are used for acute-chronic ratios and bioconcentration factors because quotients are likely to be closer to lognormal than normal distributions. In addition, division of the geometric mean of a set of numerators by the geometric mean of the set of corresponding denominators will result in the geometric mean of the set of corresponding quotients.

J. The Genus Mean Acute Value should be calculated as the geometric mean of the Species Mean Acute Values available for each genus.

K. Order the Genus Mean Acute Value from high to low.

L. Assign ranks, R, to the Genus Mean Acute Value from "1" for the lowest to "N" for the highest. If two or more Genus Mean Acute Values are identical, arbitrarily assign them successive ranks.

M. Calculate the cumulative probability, P, for each Genus Mean Acute Value as R/(N+1).

N. Select the four Genus Mean Acute Values that have cumulative probabilities closest to 0.05. (If there are less than 59 Genus Mean Acute Values, these will always be the four lowest Genus Mean Acute Values).

O. Using the selected Genus Mean Acute Values and Ps, calculate:

$$S^2 = \frac{\Sigma((\ln GMAV)^2) - ((\Sigma(\ln GMAV))^2/4)}{\Sigma(P) - ((\Sigma(\sqrt{P}))^2/4)}$$

$$L = (\Sigma(\ln GMAV) = S(\Sigma(\sqrt{P})))/4$$

$$A = S(\sqrt{0.05}) + L$$

$$FAV = e^A$$

(See original document, referenced at beginning of this appendix, for development of the calculation procedure and Appendix 2 for example calculation and computer program.)

NOTE: Natural logarithms (logarithms to base e, denoted as ln) are used herein merely because they are easier to use on some hand calculators and computers than common (base 10) logarithms. Consistent use of either will produce the same result.

P. If for a commercially or recreationally important species the geometric mean of the acute values from flow-through tests in which the concentrations of test material were measured is lower than the calculated Final Acute Value, then that geometric mean should be used as the Final Acute Value instead of the calculated Final Acute Value.

Q. Go to section VI.

V. Final Acute Equation

A. When enough data are available to show that acute toxicity to two or more species is similarly related to a water quality characteristic, the relationship should be taken into account as described in section IV, steps B through G, or using analysis of covariance. The two methods are equivalent and produce identical results. The manual method described below provides an understanding of this application of covariance analysis, but computerized versions of covariance analysis are much more convenient for analyzing large data tests. If two or more factors affect toxicity, multiple regression analysis should be used.

B. For each species for which comparable acute toxicity values are available at two or more different values of the water quality characteristic, perform a least squares regression of the acute toxicity values on the corresponding values of the water quality characteristic to obtain the slope and its 95 percent confidence limits for each species.

NOTE: Because the best documented relationship fitting these data is that between hardness and acute toxicity of metals in freshwater and a log-log relationship, geometric means and natural logarithms of both toxicity and water quality are used in the rest of this section. For relationships based on other water quality characteristics such as pH, temperature, or salinity, no transformation or a different transformation might fit the data better, and appropriate changes will be necessary.

C. Decide whether the data for each species are useful, taking into account the range and number of the tested values of the water quality characteristic and the degree of agreement within and between species. For example, a slope based on six data points might be of limited value if based only on data for a very narrow range of water quality characteristic values. A slope based on only two data points, however, might be useful if consistent with other information and if the two points cover a broad enough range of the water quality characteristic.

In addition, acute values that appear to be questionable in comparison with other acute and chronic data available for the same species and for other species in the same genus probably should not be used. For example, if after adjustment for the water quality characteristic the acute values available for a species or genus differ by more than a factor of 10, probably some or all of the values should be rejected. If useful slopes are not available for at least one fish and one invertebrate, or if the available slopes are too dissimilar, or if too few data are available to adequately define the relationship between acute toxicity and the water quality characteristic, return to section IV.G, using the results of tests conducted under conditions and in waters similar to those commonly used for toxicity tests with the species.

D. Individually for each species, calculate the geometric mean of the available acute values and then divide each of these acute values by the mean for the species. This normalizes the values so that the geometric mean of the normalized values for each species, individually, and for any combination of species is 1.0.

E. Similarly normalize the values of the water quality characteristic for each species, individually.

F. Individually for each species, perform a least squares regression of the normalized acute toxicity values on the corresponding normalized values of the water quality characteristic. The resulting slopes and 95 percent confidence limits will be identical to those obtained in step B. However, now, if the data are actually plotted, the line of best fit for each individual species will go through the point 1,1 in the center of the graph.

G. Treat normalized data as if they were all for the same species and perform a least squares regression of all the normalized acute values on the corresponding normalized values of the water quality characteristic to obtain the pooled acute slope, V, and its 95 percent confidence limits. If all the normalized data are actually plotted, the line of best fit will go through the point 1,1 in the center of the graph.

H. For each species, calculate the geometric mean, W, of the acute toxicity values and the geometric mean, X, of the values of the water quality characteristic. (These were calculated in steps D and E.)

I. For each species, calculate the logarithm, Y, of the Species Mean Acute Value at a selected value, Z, of the water quality characteristic using the equation:

$$Y = \ln W - V(\ln X - \ln Z).$$

J. For each species, calculate the SMAV at Z using the equation:

$$SMAV = e^{Y}.$$

NOTE: Alternatively, the Species Mean Acute Values at Z can be obtained by skipping step H using the equations in steps I and J to adjust each acute value individually to Z, and then calculating the geometric mean of the adjusted values for each species individually.

This alternative procedure allows an examination of the range of the adjusted acute values for each species.

K. Obtain the Final Acute Value at Z by using the procedure described in section IV, steps J through O.

L. If the Species Mean Acute Value at Z of a commercially or recreationally important species is lower than the calculated Final Acute Value at Z, then that Species Mean Acute Value should be used as the Final Acute Value at Z instead of the calculated Final Acute Value.

M. The Final Acute Equation is written as:

$$\text{Final Acute Value} = e^{(V[\ln(\text{water quality characteristic})] + \ln A - V[\ln Z])}$$

where

V = pooled acute slope
A = Final Acute Value at Z.

Because V, A, and Z are known, the Final Acute Value can be calculated for any selected value of the water quality characteristic.

VI. Final Chronic Value

A. Depending on the data that are available concerning chronic toxicity to aquatic animals, the Final Chronic Value might be calculated in the same manner as the Final Acute Value or by dividing the Final Acute Value by the Final Acute-Chronic Ratio. In some cases, it may not be possible to calculate a Final Chronic Value.

NOTE: As the name implies, the Acute-Chronic Ratio is a way of relating acute and chronic toxicities. The Acute-Chronic Ratio is basically the inverse of the application factor, but this new name is better because it is more descriptive and should help prevent confusion between "application factors" and "safety factors." Acute-Chronic Ratios and application factors are ways of relating the acute and chronic toxicities of a material to aquatic organisms. Safety factors are used to provide an extra margin of safety beyond the known or estimated sensitivities of aquatic organisms. Another advantage of the Acute-Chronic Ratio is that it will usually be greater than 1; this should avoid the confusion as to whether a large application factor is one that is close to unity or one that has a denominator that is much greater than the numerator.

B. Chronic values should be based on results of flow- through chronic tests in which the concentrations of test material in the test solutions were properly measured at appropriate times during the test. (Exception: renewal, which is acceptable for daphnids.)

C. Results of chronic tests in which survival, growth, or reproduction in the control treatment was unacceptably low should not be used. The limits of acceptability will depend on the species.

D. Results of chronic tests conducted in unusual dilution water (dilution water in which total organic carbon or particulate matter exceeded 5 mg/L) should not be used, unless a relationship is developed between chronic toxicity and organic carbon or particulate matter, or unless data show that organic carbon, particulate matter (and so forth) do not affect toxicity.

E. Chronic values should be based on endpoints and lengths of exposure appropriate to the species. Therefore, only results of the following kinds of chronic toxicity tests should be used:

1. Life-cycle toxicity tests consisting of exposures of each of two or more groups of individuals of a species to a different concentration of the test material throughout a life cycle. To ensure that all life stages and life processes are exposed, tests with fish should begin with embryos or newly hatched young less than 48-hours old, continue through maturation and reproduction, and end not less than 24 days (90 days for salmonids) after the hatching of the next generation. Tests with daphnids should begin with young less than 24-hours old and last for not less than 21 days. Tests with mysids should begin with young less than 24-hours old and continue until seven days past the median time of first brood release in the controls.

   For fish, data should be obtained and analyzed on survival and growth of adults and young, maturation of males and females, eggs spawned per female, embryo viability (salmonids only), and hatchability. For daphnids, data should be obtained and analyzed on survival and young per female. For mysids, data should be obtained and analyzed on survival, growth, and young per female.

2. Partial life-cycle toxicity tests consisting of exposures of each of two or more groups of individuals in a fish species to a concentration of the test material through most portions of a life cycle. Partial life-cycle tests are allowed with fish species that require more than a year to reach sexual maturity so that all major life stages can be exposed to the test material in less than 15 months.

   Exposure to the test material should begin with immature juveniles at least two months prior to active gonad development, continue through maturation and reproduction, and end not less than 24 days (90 days for salmonids) after the hatching of the next generation. Data should be obtained and analyzed on survival and growth of adults and young, maturation of males and females, eggs spawned per female, embryo viability (salmonids only), and hatchability.

3. Early life stage toxicity tests consisting of 28- to 32-day (60 days post hatch for salmonids) exposures of the early life stages of a fish species from shortly after fertilization through embryonic, larval, and early juvenile development. Data should be obtained and analyzed on survival and growth.

   NOTE: Results of an early life stage test are used as predictions of results of life-cycle and partial life-cycle tests with the same species. Therefore, when results of a total or partial life-cycle test are available, results of an early life stage test with the same species should not be used. Also, results of early life stage tests in which the incidence of mortalities or abnormalities increased substantially near the end should not be used because these results are possibly not good predictions of the results of comparable total or partial life cycle or partial life cycle tests.

F. A chronic value can be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis. A lower chronic limit is the highest tested concentration in an acceptable chronic test that did not cause an unacceptable amount of adverse effect on any of the specified biological measurements and below which no tested concentration caused an unacceptable effect. An upper chronic limit is the lowest tested concentration in an acceptable chronic test that did cause an unacceptable amount of adverse effect on one or more of the specified biological measurements and above which all tested concentrations also caused such an effect.

NOTE: Because various authors have used a variety of terms and definitions to interpret and report results of chronic tests, reported results should be reviewed carefully. The amount of effect that is considered unacceptable is often based on a statistical hypothesis test but might also be defined in terms of a specified percent reduction from the controls. A small percent reduction (e.g., 3 percent) might be considered acceptable even if it is statistically significantly different from the control, whereas a large percent reduction (e.g., 30 percent) might be considered unacceptable even if it is not statistically significant.

G. If the chronic toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Chronic Equation should be derived based on that water quality characteristic. Go to section VII.

H. If chronic values are available for species in eight families as described in sections III.B.1 or III.C.1, a Species Mean Chronic Value should also be calculated for each species for which at least one chronic value is available by calculating the geometric mean of all chronic values available for the species; appropriate Genus Mean Chronic Values should also be calculated. The Final Chronic Value should then be obtained using the procedure described in section III, steps J through O. Then go to section VI.M.

I. For each chronic value for which at least one corresponding appropriate acute value is available, calculate an acute-chronic ratio using for the numerator the geometric mean of the results of all acceptable flow-through acute tests in the same dilution water and in which the concentrations were measured. (Exception: static is acceptable for daphnids.)

For fish, the acute test(s) should have been conducted with juveniles and should have been part of the same study as the chronic test. If acute tests were not conducted as part of the same study, acute tests conducted in the same laboratory and dilution water but in a different study may be used. If no such acute tests are available, results of acute tests conducted in the same dilution water in a different laboratory may be used. If no such acute tests are available, an acute-chronic ratio should not be calculated.

J. For each species, calculate the species mean acute-chronic ratio as the geometric mean of all acute-chronic ratios available for that species.

K. For some materials, the acute-chronic ratio seems to be the same for all species, but for other materials, the ratio seems to increase or decrease as the Species Mean Acute Value increases. Thus the Final Acute-Chronic Ratio can be obtained in four ways, depending on the data available:

1. If the Species Mean Acute-Chronic ratio seems to increase or decrease as the Species Mean Acute Value increases, the Final Acute-Chronic Ratio should be calculated as the geometric mean of the acute-chronic ratios for species whose Species Mean Acute Values are close to the Final Acute Value.

2. If no major trend is apparent, and the acute-chronic ratios for a number of species are within a factor of 10, the Final Acute-Chronic Ratio should be calculated as the geometric mean of all the Species Mean Acute-Chronic Ratios available for both freshwater and saltwater species.

3. For acute tests conducted on metals and possibly other substances with embryos and larvae of barnacles, bivalve molluscs, sea urchins, lobsters, crabs, shrimp, and abalones (see section IV.E.2), it is probably appropriate to assume that the

acute-chronic ratio is 2. Chronic tests are very difficult to conduct with most such species, but the sensitivities of embryos and larvae would likely determine the results of life cycle tests. Thus, if the lowest available Species Mean Acute Values were determined with embryos and larvae of such species, the Final Acute-Chronic Ratio should probably be assumed to be 2, so that the Final Chronic Value is equal to the Criterion Maximum Concentration (see section XI.B)

4. If the most appropriate Species Mean Acute-Chronic Ratios are less than 2.0, and especially if they are less than 1.0, acclimation has probably occurred during the chronic test. Because continuous exposure and acclimation cannot be assured to provide adequate protection in field situations, the Final Acute-Chronic Ratio should be assumed to be 2, so that the Final Chronic Value is equal to the Criterion Maximum Concentration (see section XI.B).

   If the available Species Mean Acute-Chronic Ratios do not fit one of these cases, a Final Acute-Chronic Ratio probably cannot be obtained, and a Final Chronic Value probably cannot be calculated.

L. Calculate the Final Chronic Value by dividing the Final Acute Value by the Final Acute-Chronic Ratio. If there was a Final Acute Equation rather than a Final Acute Value, see also section VII.A.

M. If the Species Mean Chronic Value of a commercially or recreationally important species is lower than the calculated Final Chronic Value, then that Species Mean Chronic Value should be used as the Final Chronic Value instead of the calculated Final Chronic Value.

N. Go to section VIII.

VII. Final Chronic Equation

A. A Final Chronic Equation can be derived in two ways. The procedure described here will result in the chronic slope being the same as the acute slope. The procedure described in steps B through N usually will result in the chronic slope being different from the acute slope.

1. If acute-chronic ratios are available for enough species at enough values of the water quality characteristic to indicate that the acute-chronic ratio is probably the same for all species and is probably independent of the water quality characteristic, calculate the Final Acute-Chronic Ratio as the geometric mean of the available Species Mean Acute-Chronic Ratios.

2. Calculate the Final Chronic Value at the selected value Z of the water quality characteristic by dividing the Final Acute Value at Z (see section V.M) by the Final Acute-Chronic Ratio.

3. Use V = pooled acute slope (see section V.M) as L = pooled chronic slope.

4. Go to section VII.M.

B. When enough data are available to show that chronic toxicity to at least one species is related to a water quality characteristic, the relationship should be taken into account as described in steps B through G or using analysis of covariance. The two methods are equivalent and produce identical results. The manual method described in the next paragraph provides an understanding of this application of covariance analysis, but computerized versions of covariance analysis are much more convenient for analyzing large data sets. If two or more factors affect toxicity, multiple regression analysis should be used.

C. For each species for which comparable chronic toxicity values are available at two or more different values of the water quality characteristic, perform a least squares regression of the chronic toxicity values on the corresponding values of the water quality characteristic to obtain the slope and its 95 percent confidence limits for each species.

NOTE: Because the best-documented relationship fitting these data is that between hardness and acute toxicity of metals in fresh water and a log-log relationship, geometric means and natural logarithms of both toxicity and water quality are used in the rest of this section. For relationships based on other water quality characteristics such as pH, temperature, or salinity, no transformation or a different transformation might fit the data better, and appropriate changes will be necessary throughout this section. It is probably preferable, but not necessary, to use the same transformation that was used with the acute values in section V.

D. Decide whether the data for each species are useful, taking into account the range and number of the tested values of the water quality characteristic and the degree of agreement within and between species. For example, a slope based on six data points might be of limited value if founded only on data for a very narrow range of values of the water quality characteristic. A slope based on only two data points, however, might be useful if it is consistent with other information and if the two points cover a broad enough range of the water quality characteristic. In addition, chronic values that appear to be questionable in comparison with other acute and chronic data available for the same species and for other species in the same genus probably should not be used. For example, if after adjustment for the water quality characteristic the chronic values available for a species or genus differ by more than a factor of 10, probably some or all of the values should be rejected.

If a useful chronic slope is not available for at least one species, or if the available slopes are too dissimilar, or if too few data are available to adequately define the relationship between chronic toxicity and the water quality characteristic, the chronic slope is probably the same as the acute slope, which is equivalent to assuming that the acute-chronic ratio is independent of the water quality characteristic. Alternatively, return to section VI.H, using the results of tests conducted under conditions and in waters similar to those commonly used for toxicity tests with the species.

E. Individually for each species, calculate the geometric mean of the available chronic values and then divide each chronic value for a species by its mean. This normalizes the chronic values so that the geometric mean of the normalized values for each species individually, and for any combination of species, is 1.0.

F. Similarly normalize the values of the water quality characteristic for each species, individually.

G. Individually for each species, perform a least squares regression of the normalized chronic toxicity values on the corresponding normalized values of the water quality characteristic. The resulting slopes and the 95 percent confidence limits will be identical to those obtained in section B. Now, however, if the data are actually plotted, the line of best fit for each individual species will go through the point 1,1 in the center of the graph.

H. Treat all the normalized data as if they were all for the same species and perform a least squares regression of all the normalized chronic values on the corresponding normalized values of the water quality characteristic to obtain the pooled chronic slope, L, and its 95 percent confidence limits. If all the normalized data are actually plotted, the line of best fit will go through the point 1,1 in the center of the graph.

I. For each species, calculate the geometric mean, M, of the toxicity values and the geometric mean, P, of the values of the water quality characteristic. (These were calculated in steps E and F.)

J. For each species, calculate the logarithm, Q, of the Species Mean Chronic Value at a selected value, Z, of the water quality characteristic using the equation:

$$Q = \ln M - L(\ln P - \ln Z).$$

NOTE: Although it is not necessary, it will usually be best to use the same value of the water quality characteristic here as was used in section V.I.

K. For each species, calculate a Species Mean Chronic Value at Z using the equation:

$$SMCV = e^{Q}.$$

NOTE: Alternatively, the Species Mean Chronic Values at Z can be obtained by skipping step J, using the equations in steps J and K to adjust each acute value individually to Z, and then calculating the geometric means of the adjusted values for each species individually. This alternative procedure allows an examination of the range of the adjusted chronic values for each species.

L. Obtain the Final Chronic Value at Z by using the procedure described in section IV, steps J through O.

M. If the Species Mean Chronic Value at Z of a commercially or recreationally important species is lower than the calculated Final Chronic Value at Z, then that Species Mean Chronic Value should be used as the Final Chronic Value at Z instead of the calculated Final Chronic Value.

N. The Final Chronic Equation is written as:

$$\text{Final Chronic Value} = e^{(L[\ln(\text{water quality characteristic})] + \ln S - L[\ln Z])}$$

where

L = pooled chronic slope

S = Final Chronic Value at Z.

Because L, S, and Z are known, the Final Chronic Value can be calculated for any selected value of the water quality characteristic.

VIII. Final Plant Value

A. Appropriate measures of the toxicity of the material to aquatic plants are used to compare the relative sensitivities of aquatic plants and animals. Although procedures for conducting and interpreting the results of toxicity tests with plants are not well developed, results of tests with plants usually indicate that criteria which adequately protect aquatic animals and their uses will probably also protect aquatic plants and their uses.

B. A plant value is the result of a 96-hour test conducted with an alga, or a chronic test conducted with an aquatic vascular plant.

NOTE: A test of the toxicity of a metal to a plant usually should not be used if the medium contained an excessive amount of a complexing agent, such as EDTA, that might affect the toxicity of the metal. Concentrations of EDTA above about 200 µg/L should probably be considered excessive.

C. The Final Plant Value should be obtained by selecting the lowest result from a test with an important aquatic plant species in which the concentrations of test material were measured, and the endpoint was biologically important.

IX. Final Residue Value

A. The Final Residue Value is intended to prevent concentrations in commercially or recreationally important aquatic species from affecting marketability because they exceed applicable FDA action levels and to protect wildlife (including fishes and birds) that consume aquatic organisms from demonstrated unacceptable effects. The Final Residue Value is the lowest of the residue values that are obtained by dividing maximum permissible tissue concentrations by appropriate bioconcentration or bioaccumulation factors. A maximum permissible tissue concentration is either (a) an FDA action level (Compliance Policy Guide, U.S. Food & Drug Admin. 1981) for fish oil or for the edible portion of fish or shellfish, or a maximum acceptable dietary intake based on observations on survival, growth, or reproduction in a chronic wildlife feeding study or a long-term wildlife field study. If no maximum permissible tissue concentration is available, go to section X because no Final Residue Value can be derived.

B. Bioconcentration Factors (BCFs) and bioaccumulation factors (BAFs) are quotients of the concentration of a material in one or more tissues of an aquatic organism, divided by the average concentration in the solution in which the organism had been living. A BCF is intended to account only for net uptake directly from water and thus almost must be measured in a laboratory test. Some uptake during the bioconcentration test might not be directly from water if the food sorbs some of the test material before it is eaten by the test organisms. A BAF is intended to account for net uptake from both food and water in a real-world situation. A BAF almost must be measured in a field situation in which predators accumulate the material directly from water and by consuming prey that could have accumulated the material from both food and water.

The BCF and BAF are probably similar for a material with a low BCF, but the BAF is probably higher than the BCF for materials with high BCFs. Although BCFs are not too difficult to determine, very few BAFs have been measured acceptably because adequate measurements must be made of the material's concentration in water to ascertain if it was reasonably constant for a long enough time over the range of territory inhabited by the organisms. Because so few acceptable BAFs are available, only BCFs will be discussed further. However, if an acceptable BAF is available for a material, it should be used instead of any available BCFs.

C. If a maximum permissible tissue concentration is available for a substance (e.g., parent material, parent material plus metabolites, etc.), the tissue concentration used in the calculation of the BCF should be for the same substance. Otherwise, the tissue concentration used in the calculation of the BCF should derive from the material and its metabolites that are structurally similar and are not much more soluble in water than the parent material.

1. A BCF should be used only if the test was flow-through, the BCF was calculated based on measured concentrations of the test material in tissue and in the test solution, and the exposure continued at least until either apparent steady state or 28 days was reached. Steady state is reached when the BCF does not change significantly over a period of time, such as 2 days or 16 percent of the length of the exposure, whichever is longer. The BCF used from a test should be the highest of the apparent steady-state BCF, if apparent steady state was reached; the highest BCF obtained, if apparent steady state was not reached; and the projected steady state BCF, if calculated.

2. Whenever a BCF is determined for a lipophilic material, the percent lipids should also be determined in the tissue(s) for which the BCF was calculated.

3. A BCF obtained from an exposure that adversely affected the test organisms may be used only if it is similar to a BCF obtained with unaffected organisms of the same species at lower concentrations that did not cause adverse effects.

4. Because maximum permissible tissue concentrations are almost never based on dry weights, a BCF calculated using dry tissue weights must be converted to a wet tissue weight basis. If no conversion factor is reported with the BCF, multiply the dry weight BCF by 0.1 for plankton and by 0.2 for individual species of fishes and invertebrates.

5. If more than one acceptable BCF is available for a species, the geometric mean of the available values should be used; however, the BCFs are from different lengths of exposure and the BCF increases with length of exposure, then the BCF for the longest exposure should be used.

E. If enough pertinent data exists, several residue values can be calculated by dividing maximum permissible tissue concentrations by appropriate BCFs:

1. For each available maximum acceptable dietary intake derived from a chronic feeding study or a long-term field study with wildlife (including birds and aquatic organisms), the appropriate BCF is based on the whole body of aquatic species that constitutes or represents a major portion of the diet of the tested wildlife species.

2. For an FDA action level for fish or shellfish, the appropriate BCF is the highest geometric mean species BCF for the edible portion (muscle for decapods, muscle with or without skin for fishes, adductor muscle for scallops, and total soft tissue for other bivalve molluscs) of a consumed species. The highest species BCF is used because FDA action levels are applied on a species-by-species basis.

F. For lipophilic materials, calculating additional residue values is possible. Because the steady-state BCF for a lipophilic material seems to be proportional to percent lipids from one tissue to another and from one species to another, extrapolations can be made from tested tissues, or species to untested tissues, or species on the basis of percent lipids.

1. For each BCF for which the percent lipids is known for the same tissue for which the BCF was measured, normalize the BCF to a 1 percent lipid basis by dividing it by the percent lipids. This adjustment to a 1 percent lipid basis is intended to make all the measured BCFs for a material comparable regardless of the species or tissue with which the BCF was measured.
2. Calculate the geometric mean-normalized BCF. Data for both saltwater and freshwater species should be used to determine the mean-normalized BCF unless they show that the normalized BCFs are probably not similar.
3. Calculate all possible residue values by dividing the available maximum permissible tissue concentrations by the mean-normalized BCF and by the percent lipids values appropriate to the maximum permissible tissue concentrations, i.e.,

$$\textit{Residue value} = \frac{(\textit{maximum permissible tissue concentration})}{(\textit{mean normalized BCF})(\textit{appropriate percent lipids})}$$

- For an FDA action level for fish oil, the appropriate percent lipids value is 100.
- For an FDA action level for fish, the appropriate percent lipids value is 11 for freshwater criteria and 10 for saltwater criteria because FDA action levels are applied species-by-species to commonly consumed species. The highest lipid contents in the edible portions of important consumed species are about 11 percent for both the freshwater chinook salmon and lake trout and about 10 percent for the saltwater Atlantic herring.
- For a maximum acceptable dietary intake derived from a chronic feeding study or a long-term field study with wildlife, the appropriate percent lipids is that of an aquatic species or group of aquatic species that constitute a major portion of the diet of the wildlife species.

G. The Final Residue Value is obtained by selecting the lowest of the available residue values.

NOTE: In some cases, the Final Residue Value will not be low enough. For example, a residue value calculated from a FDA action level will probably result in an average concentration in the edible portion of a fatty species at the action level. Some individual organisms and possibly some species will have residue concentrations higher than the mean value, but no mechanism has been devised to provide appropriate additional protection. Also, some chronic feeding studies and long-term field studies with wildlife identify concentrations that cause adverse effects but do not identify concentrations that do not cause adverse effects; again, no mechanism has been devised to provide appropriate additional protection. These are some of the species and uses that are not protected at all times in all places.

X. Other Data

Pertinent information that could not be used in earlier sections might be available concerning adverse effects on aquatic organisms and their uses. The most important of these are data on cumulative and delayed toxicity, flavor impairment, reduction in survival, growth, or reproduction, or any other adverse effect shown to be biologically important. Especially important are data for species for which no other data are available. Data from behavioral, biochemical, physiological, microcosm, and field studies might also be available. Data might be available from tests conducted in unusual dilution water (see IV.D and VI.D), from chronic tests in which the concentrations were not measured (see VI.B), from tests with previously exposed organisms (see II.F), and from tests on formulated mixtures or emulsifiable concentrates (see II.D). Such data might affect a criterion if they were obtained with an important species, the test concentrations were measured, and the endpoint was biologically important.

XI. Criterion

A. A criterion consists of two concentrations: the Criterion Maximum Concentration and the Criterion Continuous Concentration.

B. The Criterion Maximum Concentration (CMC) is equal to one-half the Final Acute Value.

C. The Criterion Continuous Concentration (CCC) is equal to the lowest of the Final Chronic Value, the Final Plant Value, and the Final Residue Value, unless other data (see section X) show that a lower value should be used. If toxicity is related to a water quality characteristic, the Criterion Continuous Concentration is obtained from the Final Chronic Equation, the Final Plant Value, and the Final Residue Value by selecting the one, or the combination, that results in the lowest concentrations in the usual range of the water quality characteristic, unless other data (see section X) show that a lower value should be used.

D. Round both the Criterion Maximum Concentration and the Criterion Continuous Concentration to two significant digits.

E. The criterion is stated as follows:

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, *(1) aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of (2) does not exceed (3) μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed (4) μg/L more than once every three years on the average.

*where (1) = insert freshwater or saltwater

(2) = insert name of material

(3) = insert the Criterion Continuous Concentration

(4) = insert the Criterion Maximum Concentration.

XII. Final Review

A. The derivation of the criterion should be carefully reviewed by rechecking each step of the guidelines. Items that should be especially checked are

1. If unpublished data are used, are they well documented?
2. Are all required data available?
3. Is the range of acute values for any species greater than a factor of 10?
4. Is the range of Species Mean Acute Values for any genus greater than a factor of 10?
5. Is there more than a factor of 10 difference between the four lowest Genus Mean Acute Values?
6. Are any of the four lowest Genus Mean Acute Values questionable?
7. Is the Final Acute Value reasonable in comparison with the Species Mean Acute Values and Genus Mean Acute Values?
8. For any commercially or recreationally important species, is the geometric mean of the acute values from flow-through tests in which the concentrations of test material were measured lower than the Final Acute Value?

9. Are any of the chronic values questionable?
10. Are chronic values available for acutely sensitive species?
11. Is the range of acute-chronic ratios greater than a factor of 10?
12. Is the Final Chronic Value reasonable in comparison with the available acute and chronic data?
13. Is the measured or predicted chronic value for any commercially or recreationally important species below the Final Chronic Value?
14. Are any of the other data important?
15. Do any data look like they might be outliers?
16. Are there any deviations from the guidelines? Are they acceptable?

B. On the basis of all available pertinent laboratory and field information, determine if the criterion is consistent with sound scientific evidence. If not, another criterion — either higher or lower — should be derived using appropriate modifications of these guidelines.

APPENDIX I

List of EPA
Water Quality Criteria Documents

APPENDIX I

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

Water Quality Criteria Documents

The U.S. Environmental Protection Agency has published water quality criteria for toxic pollutant(s) categories. Copies of water quality criteria documents are available from the National Technical Information Service (NTIS), 5285 Front Royal Road, Springfield, VA 22161, (703) 487-4650. Prices of individual documents may be obtained by contacting NTIS. Order numbers are listed below. Where indicated, documents may be obtained from the Water Resource Center, 401 M St., S.W. RC-4100, Washington, DC 20460, (202) 260-7786.

Chemical	NTIS Order No.	EPA Document No.
Acenaphthene	PB 81-117269	EPA 440/5-80-015
Acrolein	PB 81-117277	EPA 440/5-80-016
Acrylonitrile	PB 81-117285	EPA 440/5-80-017
Aesthetics	PB 263943	EPA 440/9-76-023
Aldrin/Dieldrin	PB 81-117301	EPA 440/5-80-019
Alkalinity	PB 263943	EPA 440/9-76-023
Aluminum	PB 88-245998	EPA 440/5-86-008
Ammonia	PB 85-227114	EPA 440/5-85-001
Ammonia (saltwater)	PB 89-195242	EPA 440/5-88-004
Antimony	PB 81-117319	EPA 440/5-80-020
Antimony (III) — aquatic (draft)	resource center	
Arsenic — 1980	PB 81-117327	EPA 440/5-80-021
— 1984	PB 85-227445	EPA 440/5-84-033
Asbestos	PB 81-117335	EPA 440/5-80-022
Bacteria — 1976	PB 263943	EPA 440/9-76-023
— 1984	PB 86-158045	EPA 440/5-84-002
Barium	PB 263943	EPA 440/9-76-023
Benzene	PB 81-117293	EPA 440/5-80-018
Benzidine	PB 81-117343	EPA 440/5-80-023
Beryllium	PB 81-117350	EPA 440/5-80-024
Boron	PB 263943	EPA 440/9-76-023
Cadmium — 1980	PB 81-117368	EPA 440/5-80-025
— 1984	PB 85-224031	EPA 440/5-84-032
Carbon Tetrachloride	PB 81-117376	EPA 440/5-80-026
Chlordane	PB 81-117384	EPA 440/5-80-027
Chloride	PB 88-175047	EPA 440/5-88-001
Chlorinated Benzenes	PB 81-117392	EPA 440/5-80-028
Chlorinated Ethanes	PB 81-117400	EPA 440/5-80-029
Chlorinated Naphthalene	PB 81-117426	EPA 440/5-80-031
Chlorinated Phenols	PB 81-117434	EPA 440/5-80-032

Chemical	NTIS Order No.	EPA Document No.
Chlorine	PB 85-227429	EPA 440/5-84-030
Chloroalkyl Ethers	PB 81-117418	EPA 440/5-80-030
Chloroform	PB 81-117442	EPA 440/5-80-033
2-Chlorophenol	PB 81-117459	EPA 440/5-80-034
Chlorophenoxy Herbicides	PB 263943	EPA 440/9-76-023
Chlorpyrifos	PB 87-105359	EPA 440/5-86-005
Chromium — 1980	PB 81-117467	EPA 440/5-80-035
—1984	PB 85-227478	EPA 440/5-84-029
Color	PB 263943	EPA 440/9-76-023
Copper — 1980	PB 81-117475	EPA 440/5-80-036
— 1984	PB 85-227023	EPA 440/5-84-031
Cyanide	PB 85-227460	EPA 440/5-84-028
Cyanides	PB 81-117483	EPA 440/5-80-037
DDT and Metabolites	PB 81-117491	EPA 440/5-80-038
Demeton	PB 263943	EPA 440/9-76-023
Dichlorobenzenes	PB 81-117509	EPA 440/5-80-039
Dichlorobenzidine	PB 81-117517	EPA 440/5-80-040
Dichloroethylenes	PB 81-117525	EPA 440/5-80-041
2,4-Dichlorophenol	PB 81-117533	EPA 440/5-80-042
Dichloropropane/ Dichloropropene	PB 81-117541	EPA 440/5-80-043
2,4-Dimethylphenol	PB 81-117558	EPA 440/5-80-044
Dinitrotoluene	PB 81-117566	EPA 440/5-80-045
Diphenylhydrazine	PB 81-117731	EPA 440/5-80-062
Di-2-Ethylhexyl Phthalate — aquatic (draft)	resource center	
Dissolved Oxygen	PB 86-208253	EPA 440/5-86-003
Endosulfan	PB 81-117574	EPA 440/5-80-046
Endrin	PB 81-117582	EPA 440/5-80-047
Ethylbenzene	PB 81-117590	EPA 440/5-80-048
Fluoranthene	PB 81-117608	EPA 440/5-80-049
Gasses, Total Dissolved	PB 263943	EPA 440/9-76-023
Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses	PB 85-227049	
Guthion	PB 263943	EPA 440/9-76-023
Haloethers	PB 81-117616	EPA 440/5-80-050
Halomethanes	PB 81-117624	EPA 440/5-80-051
Hardness	PB 263943	EPA 440/9-76-023
Heptachlor	PB 81-117632	EPA 440/5-80-052
Hexachlorobenzene — aquatic (draft)	resource center	
Hexachlorobutadiene	PB 81-117640	EPA 440/5-80-053
Hexachlorocyclohexane	PB 81-117657	EPA 440/5-80-054

Chemical	NTIS Order No.	EPA Document No.
Hexachlorocyclopentadiene	PB 81-117665	EPA 440/5-80-055
Iron	PB 263943	EPA 440/9-76-023
Isophorone	PB 81-117673	EPA 440/5-80-056
Lead — 1980	PB 81-117681	EPA 440/5-80-057
— 1984	PB 85-227437	EPA 440/5-84-027
Malathion	PB 263943	EPA 440/9-76-023
Manganese	PB 263943	EPA 440/9-76-023
Mercury — 1980	PB 81-117699	EPA 440/5-80-058
— 1984	PB 85-227452	EPA 440/5-84-026
Methoxychlor	PB 263943	EPA 440/9-76-023
Mirex	PB 263943	EPA 440/9-76-023
Naphthalene	PB 81-117707	EPA 440/5-80-059
Nickel — 1980	PB 81-117715	EPA 440/5-80-060
— 1986	PB 87-105359	EPA 440/5-86-004
Nitrates/Nitrites	PB 263943	EPA 440/9-76-023
Nitrobenzene	PB 81-117723	EPA 440/5-80-061
Nitrophenols	PB 81-117749	EPA 440/5-80-063
Nitrosamines	PB 81-117756	EPA 440/5-80-064
Oil and Grease	PB 263943	EPA 440/9-76-023
Parathion	PB 87-105383	EPA 440/5-86-007
Pentachlorophenol — 1980	PB 81-117764	EPA 440/5-80-065
— 1986	PB 87-105391	EPA 440/5-85-009
pH	PB 263943	EPA 440/9-76-023
Phenanthrene — aquatic (draft)	resource center	
Phenol	PB 81-117772	EPA 440/5-80-066
Phosphorus	PB 263943	EPA 440/9-76-023
Phthalate Esters	PB 81-117780	EPA 440/5-80-067
Polychlorinated Biphenyls	PB 81-117798	EPA 440/5-80-068
Polynuclear Aromatic Hydrocarbons	PB 81-117806	EPA 440/5-80-069
Selenium — 1980	PB 81-117814	EPA 440/5-80-070
— 1987	PB 88-142239	EPA 440/5-87-008
Silver	PB 81-117822	EPA 440/5-80-071
Silver — aquatic (draft)	resource center	
Solids (dissolved) and Salinity	PB 263943	EPA 440/9-76-023
Solids (suspended) and Turbidity	PB 263943	EPA 440/9-76-023
Sulfides/Hydrogen Sulfide	PB 263943	EPA 440/9-76-023
Tainting Substances	PB 263943	EPA 440/9-76-023
Temperature	PB 263943	EPA 440/9-76-023
2,3,7,8-Tetrachlorodibenzo-P-Dioxin	PB 89-169825	EPA 440/5-84-007
Tetrachloroethylene	PB 81-117830	EPA 440/5-80-073
Thallium	PB 81-117848	EPA 440/5-80-074
Toluene	PB 81-117863	EPA 440/5-80-075

Chemical	NTIS Order No.	EPA Document No.
Toxaphene — 1980	PB 81-117863	EPA 440/5-80-076
— 1986	PB 87-105375	EPA 440/5-86-006
Tributyltin — aquatic (draft)	resource center	
Trichloroethylene	PB 81-117871	EPA 440/5-80-077
2,4,5-Trichlorophenol — aquatic (draft)	resource center	
Vinyl Chloride	PB 81-117889	EPA 440/5-80-078
Zinc — 1980	PB 81-117897	EPA 440/5-80-079
— 1987	PB 87-143581	EPA 440/5-87-003

APPENDIX J

Attachments to ***Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria***

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

ATTACHMENT #2

GUIDANCE DOCUMENT
ON DISSOLVED CRITERIA
Expression of Aquatic Life Criteria
October 1993

Percent Dissolved in Aquatic Toxicity Tests on Metals

The attached table contains all the data that were found concerning the percent of the total recoverable metal that was dissolved in aquatic toxicity tests. This table is intended to contain the available data that are relevant to the conversion of EPA's aquatic life criteria for metals from a total recoverable basis to a dissolved basis. (A factor of 1.0 is used to convert aquatic life criteria for metals that are expressed on the basis of the acid-soluble measurement to criteria expressed on the basis of the total recoverable measurement.) Reports by Grunwald (1992) and Brungs et al. (1992) provided references to many of the documents in which pertinent data were found. Each document was obtained and examined to determine whether it contained useful data.

"Dissolved" is defined as metal that passes through a 0.45-μm membrane filter. If otherwise acceptable, data that were obtained using 0.3-μm glass fiber filters and 0.1-μm membrane filters were used, and are identified in the table; these data did not seem to be outliers.

Data were used only if the metal was in a dissolved inorganic form when it was added to the dilution water. In addition, data were used only if they were generated in water that would have been acceptable for use as a dilution water in tests used in the derivation of water quality criteria for aquatic life; in particular, the pH had to be between 6.5 and 9.0, and the concentrations of total organic carbon (TOC) and total suspended solids (TSS) had to be below 5 mg/L. Thus most data generated using river water would not be used.

Some data were not used for other reasons. Data presented by Carroll et al. (1979) for cadmium were not used because 9 of the 36 values were above 150%. Data presented by Davies et al. (1976) for lead and Holcombe and Andrew (1978) for zinc were not used because "dissolved" was defined on the basis of polarography, rather than filtration.

Beyond this, the data were not reviewed for quality. Horowitz et al. (1992) reported that a number of aspects of the filtration procedure might affect the results. In addition, there might be concern about use of "clean techniques" and adequate QA/QC.

Each line in the table is intended to represent a separate piece of information. All of the data in the table were determined in fresh water, because no saltwater data were found. Data are becoming available for copper in salt water from the New York

Harbor study; based on the first set of tests, Hansen (1993) suggested that the average percent of the copper that is dissolved in sensitive saltwater tests is in the range of 76 to 82 percent.

A thorough investigation of the percent of total recoverable metal that is dissolved in toxicity tests might attempt to determine if the percentage is affected by test technique (static, renewal, flow-through), feeding (were the test animals fed and, if so, what food and how much), water quality characteristics (hardness, alkalinity, pH, salinity), test organisms (species, loading), etc.

The attached table also gives the freshwater criteria concentrations (CMC and CCC) because percentages for total recoverable concentrations much (e.g., more than a factor of 3) above or below the CMC and CCC are likely to be less relevant. When a criterion is expressed as a hardness equation, the range given extends from a hardness of 50 mg/L to a hardness of 200 mg/L.

The following is a summary of the available information for each metal:

Arsenic(III)

The data available indicate that the percent dissolved is about 100, but all the available data are for concentrations that are much higher than the CMC and CCC.

Cadmium

Schuytema et al. (1984) reported that "there were no real differences" between measurements of total and dissolved cadmium at concentrations of 10 to 80 ug/L (pH = 6.7 to 7.8, hardness = 25 mg/L, and alkalinity = 33 mg/L); total and dissolved concentrations were said to be "virtually equivalent".

The CMC and CCC are close together and only range from 0.66 to 8.6 ug/L. The only available data that are known to be in the range of the CMC and CCC were determined with a glass fiber filter. The percentages that are probably most relevant are 75, 92, 89, 78, and 80.

Chromium(III)

The percent dissolved decreased as the total recoverable concentration increased, even though the highest concentrations reduced the pH substantially. The percentages that are probably most relevant to the CMC are 50-75, whereas the percentages that are probably most relevant to the CCC are 86 and 61.

Chromium(VI)

The data available indicate that the percent dissolved is about 100, but all the available data are for concentrations that are much higher than the CMC and CCC.

Copper

Howarth and Sprague (1978) reported that the total and dissolved concentrations of copper were "little different" except when the total copper concentration was above 500 ug/L at hardness = 360 mg/L and pH = 8 or 9. Chakoumakos et al. (1979) found that the percent dissolved depended more on alkalinity than on hardness, pH, or the total recoverable concentration of copper.

Chapman (1993) and Lazorchak (1987) both found that the addition of daphnid food affected the percent dissolved very little, even though Chapman used yeast-trout chow-alfalfa whereas Lazorchak used algae in most tests, but yeast-trout chow-alfalfa in some tests. Chapman (1993) found a low percent dissolved with and without food, whereas Lazorchak (1987) found a high percent dissolved with and without food. All of Lazorchak's values were in high hardness water; Chapman's one value in high hardness water was much higher than his other values.

Chapman (1993) and Lazorchak (1987) both compared the effect of food on the total recoverable LC50 with the effect of food on the dissolved LC50. Both authors found that food raised both the dissolved LC50 and the total recoverable LC50 in about the same proportion, indicating that food did not raise the total recoverable LC50 by sorbing metal onto food particles; possibly the food raised both LC50s by (a) decreasing the toxicity of dissolved metal, (b) forming nontoxic dissolved complexes with the metal, or (c) reducing uptake.

The CMC and CCC are close together and only range from 6.5 to 34 ug/L. The percentages that are probably most relevant are 74, 95, 95, 73, 57, 53, 52, 64, and 91.

Lead

The data presented in Spehar et al. (1978) were from Holcombe et al. (1976). Both Chapman (1993) and Holcombe et al. (1976) found that the percent dissolved increased as the total recoverable concentration increased. It would seem reasonable to expect more precipitate at higher total recoverable concentrations and therefore a lower percent dissolved at higher concentrations. The increase in percent dissolved with increasing concentration might be due to a lowering of the pH as more metal is added if the stock solution was acidic.

The percentages that are probably most relevant to the CMC are 9, 18, 25, 10, 62, 68, 71, 75, 81, and 95, whereas the percentages that are probably most relevant to the CCC are 9 and 10.

Mercury

The only percentage that is available is 73, but it is for a concentration that is much higher than the CMC.

Nickel

The percentages that are probably most relevant to the CMC are 88, 93, 92, and 100, whereas the only percentage that is probably relevant to the CCC is 76.

Selenium

No data are available.

Silver

There is a CMC, but not a CCC. The percentage dissolved seems to be greatly reduced by the food used to feed daphnids, but not by the food used to feed fathead minnows. [illegible]he percentages that are probably most relevant to the CMC are 4[illegible] 79, 79, 73, 91, 90, and 93.

Zinc

The CMC and CCC are close together and only range from 59 to 210 ug/L. The percentages that are probably most relevant are 31, 77, 77, 99, 94, 100, 103, and 96.

Recommended Values (%)[A] and Ranges of Measured Percent Dissolved Considered Most Relevant in Fresh Water

Metal	CMC		CCC	
	Recommended Value (%)	(Range %)	Recommended Value (%)	(Range %)
Arsenic(III)	95	100-104[B]	95	100-104[B]
Cadmium	85	75-92	85	75-92
Chromium(III)	85	50-75	85	61-86
Chromium(VI)	95	100[B]	95	100[B]
Copper	85	52-95	85	52-95
Lead	50	9-95	25	9-10
Mercury	35	73[B]	NA[E]	NA[E]
Nickel	85	88-100	85	76
Selenium	NA[E]	NA[C]	NA[E]	NA[C]
Silver	85	41-93	YY[D]	YY[D]
Zinc	85	31-103	85	31-103

[A] The recommended values are based on current knowledge and are subject to change as more data becomes available.

[B] All available data are for concentrations that are much higher than the CMC.

[C] NA = No data are available.

[D] YY = A CCC is not available, and therefore cannot be adjusted.

[E] NA = Bioaccumulative chemical and not appropriate to adjust to percent dissolved.

Concn.[A] (ug/L)	Percent Diss.[B]	n[C]	Species[D]	SRF[E]	Food	Hard.	Alk.	pH	Ref.
ARSENIC(III) (Freshwater: CCC = 190 ug/L; CMC = 360 ug/L)									
600-15000	104	5	?	?	?	48	41	7.6	Lima et al. 1984
12600	100	3	FM	F	No	44	43	7.4	Spehar and Fiandt 1986
CADMIUM (Freshwater: CCC = 0.66 to 2.0 ug/L; CMC = 1.8 to 8.6 ug/L)[F]									
0.16	41	?	DM	R	Yes	53	46	7.6	Chapman 1993
0.28	75	?	DM	R	Yes	103	83	7.9	Chapman 1993
0.4-4.0	92[G]	?	CS	F	No	21	19	7.1	Finlayson and Verrue 1982
13	89	3	FM	F	No	44	43	7.4	Spehar and Fiandt 1986
15-21	96	8	FM	S	No	42	31	7.5	Spehar and Carlson 1984
42	84	4	FM	S	No	45	41	7.4	Spehar and Carlson 1984
10	78	?	DM	S	No	51	38	7.5	Chapman 1993
35	77	?	DM	S	No	105	88	8.0	Chapman 1993
51	59	?	DM	S	No	209	167	8.4	Chapman 1993
6-80	80	8	?	S	No	47	44	7.5	Call et al. 1982
3-232	90[H]	5	?	F	?	46	42	7.4	Spehar et al. 1978
450-6400	70	5	FM	F	No	202	157	7.7	Pickering and Gast 1972

CHROMIUM(III) (Freshwater: CCC = 120 to 370 ug/L; CMC = 980 to 3100 ug/L)[F]

5-13	94	?	SG	F	?	25	24	7.3	Stevens and Chapman 1984
19-495	86	?	SG	F	?	25	24	7.2	Stevens and Chapman 1984
>1100	50-75	?	SG	F	No	25	24	7.0	Stevens and Chapman 1984
42	54	?	DM	R	Yes	206	166	8.2	Chapman 1993
114	61	?	DM	R	Yes	52	45	7.4	Chapman 1993
16840	26	?	DM	S	No	<51	9	6.3[I]	Chapman 1993
26267	32	?	DM	S	No	110	9	6.7	Chapman 1993
27416	27	?	DM	S	No	96	10	6.0[I]	Chapman 1993
58665	23	?	DM	S	No	190	25	6.2[I]	Chapman 1993

CHROMIUM(VI) (Freshwater: CCC = 11 ug/L; CMC = 16 ug/L)

>25,000	100	1	FM,GF	F	Yes	220	214	7.6	Adelman and Smith 1976
43,300	99.5	4	FM	F	No	44	43	7.4	Spehar and Fiandt 1986

COPPER (Freshwater: CCC = 6.5 to 21 ug/L; CMC = 9.2 to 34 ug/L)[F]

10-30	74	?	CT	F	No	27	20	7.0	Chakoumakos et al. 1979
40-200	78	?	CT	F	No	154	20	6.8	Chakoumakos et al. 1979
30-100	79	?	CT	F	No	74	23	7.6	Chakoumakos et al. 1979
100-200	82	?	CT	F	No	192	72	7.0	Chakoumakos et al. 1979
20-200	86	?	CT	F	No	31	78	8.3	Chakoumakos et al. 1979
40-300	87	?	CT	F	No	83	70	7.4	Chakoumakos et al. 1979
10-80	89	?	CT	F	No	25	169	8.5	Chakoumakos et al. 1979

300-1300	92	?	CT	F	No	195	160	7.0	Chakoumakos et al. 1979
100-400	94	?	CT	F	No	70	174	8.5	Chakoumakos et al. 1979
3-4[J]	125-167	2	CD	R	Yes	31	38	7.2	Carlson et al. 1986a,b
12-91[J]	79-84	3	CD	R	Yes	31	38	7.2	Carlson et al. 1986a,b
18-19	95	2	DA	S	No	52	55	7.7	Carlson et al. 1986b
20[J]	95	1	DA	R	No	31	38	7.2	Carlson et al. 1986b
50	96	2	FM	S	No	52	55	7.7	Carlson et al. 1986b
175[J]	91	2	FM	R	No	31	38	7.2	Carlson et al. 1986b
5-52	>82[K]	?	FM	F	Yes[L]	47	43	8.0	Lind et al. 1978
6-80	83[G]	?	CS	F	No	21	19	7.1	Finlayson and Verrue 1982
6.7	57	?	DM	S	No	49	37	7.7	Chapman 1993
35	43	?	DM	S	Yes	48	39	7.4	Chapman 1993
13	73	?	DM	R	Yes	211	169	8.1	Chapman 1993
16	57	?	DM	R	Yes	51	44	7.6	Chapman 1993
51	39	?	DM	R	Yes	104	83	7.8	Chapman 1993
32	53	?	DM	S	No	52	45	7.6	Chapman 1993
33	52	?	DM	S	No	105	79	7.9	Chapman 1993
39	64	?	DM	S	No	106	82	8.1	Chapman 1993
25-84	96	14	FM,GM	S	No	50	40	7.0	Hammermeister et al. 1983
17	91	6	DM	S	No	52	43	7.3	Hammermeister et al. 1983
120	88	14	SG	S	No	48	47	7.3	Hammermeister et al. 1983
15-90	74	19	?	S	No	48	47	7.7	Call et al. 1982
12-162	80[H]	?	BG	F	Yes[L]	45	43	7-8	Benoit 1975
28-58	85	6	DM	R	No	168	117	8.0	Lazorchak 1987
26-59	79	7	DM	R	Yes[M]	168	117	8.0	Lazorchak 1987
56,101	86	2	DM	R	Yes[N]	168	117	8.0	Lazorchak 1987

96	86	4	FM	F	No	44	43	7.4	Spehar and Fiandt 1986
160	94	1	FM	S	No	203	171	8.2	Geckler et al. 1976
230-3000	>69->79	?	CR	F	No	17	13	7.6	Rice and Harrison 1983

LEAD (Freshwater: CCC = 1.3 to 7.7 ug/L; CMC = 34 to 200 ug/L)[F]

17	9	?	DM	R	Yes	52	47	7.6	Chapman 1993
181	18	?	DM	R	Yes	102	86	7.8	Chapman 1993
193	25	?	DM	R	Yes	151	126	8.1	Chapman 1993
612	29	?	DM	S	No	50	--	---	Chapman 1993
952	33	?	DM	S	No	100	--	---	Chapman 1993
1907	~38	?	DM	S	No	150	--	---	Chapman 1993
7-29	10	?	EZ	R	No	22	--	---	JRB Associates 1983
34	62[H]	?	BT	F	Yes	44	43	7.2	Holcombe et al. 1976
58	68[H]	?	BT	F	Yes	44	43	7.2	Holcombe et al. 1976
119	71[H]	?	BT	F	Yes	44	43	7.2	Holcombe et al. 1976
235	75[H]	?	BT	F	Yes	44	43	7.2	Holcombe et al. 1976
474	81[H]	?	BT	F	Yes	44	43	7.2	Holcombe et al. 1976
4100	82[H]	?	BT	F	No	44	43	7.2	Holcombe et al. 1976
2100	79	7	FM	F	No	44	43	7.4	Spehar and Fiandt 1986
220-2700	96	14	FM,GM,DM	S	No	49	44	7.2	Hammermeister et al. 1983
580	95	14	SG	S	No	51	48	7.2	Hammermeister et al. 1983

MERCURY(II) (Freshwater: CMC = 2.4 ug/L)

172	73	1	FM	F	No	44	43	7.4	Spehar and Fiandt 1986

NICKEL (Freshwater: CCC = 88 to 280 ug/L; CMC = 790 to 2500 ug/L)[F]

21	81	?	DM	R	Yes	51	49	7.4	Chapman 1993
150	76	?	DM	R	Yes	107	87	7.8	Chapman 1993
578	87	?	DM	R	Yes	205	161	8.1	Chapman 1993
645	88	?	DM	S	No	54	43	7.7	Chapman 1993
1809	93	?	DM	S	No	51	44	7.7	Chapman 1993
1940	92	?	DM	S	No	104	84	8.2	Chapman 1993
2344	100	?	DM	S	No	100	84	7.9	Chapman 1993
4000	90	?	PK	R	No	21	--	---	JRB Associates 1983

SELENIUM (FRESHWATER: CCC = 5 ug/L; CMC = 20 ug/L)

No data are available.

SILVER (Freshwater: CMC = 1.2 to 13 ug/L; a CCC is not available)

0.19	74	?	DM	S	No	47	37	7.6	Chapman 1993
9.98	13	?	DM	S	Yes	47	37	7.5	Chapman 1993
4.0	41	?	DM	S	No	36	25	7.0	Nebeker et al. 1983
4.0	11	?	DM	S	Yes	36	25	7.0	Nebeker et al. 1983
3	79	?	FM	S	No	51	49	8.1	UWS 1993
2-54	79	?	FM	S	Yes[O]	49	49	7.9	UWS 1993
2-32	73	?	FM	S	No	50	49	8.1	UWS 1993
4-32	91	?	FM	S	No	48	49	8.1	UWS 1993
5-89	90	?	FM	S	No	120	49	8.2	UWS 1993
6-401	93	?	FM	S	No	249	49	8.1	UWS 1993

ZINC (Freshwater: CCC = 59 to 190 ug/L; CMC 65 to 210 ug/L)[F]

52	31	?	DM	R	Yes	211	169	8.2	Chapman 1993
62	77	?	DM	R	Yes	104	83	7.8	Chapman 1993
191	77	?	DM	R	Yes	52	47	7.5	Chapman 1993
356	74	?	DM	S	No	54	47	7.6	Chapman 1993
551	78	?	DM	S	No	105	85	8.1	Chapman 1993
741	76	?	DM	S	No	196	153	8.2	Chapman 1993
7[J]	71-129	2	CD	R	Yes	31	38	7.2	Carlson et al. 1986b
18-273[J]	81-107	2	CD	R	Yes	31	38	7.2	Carlson et al. 1986b
167[J]	99	2	CD	R	No	31	38	7.2	Carlson et al. 1986b
180	94	1	CD	S	No	52	55	7.7	Carlson et al. 1986b
188-393[J]	100	2	FM	R	No	31	38	7.2	Carlson et al. 1986b
551	100	1	FM	S	No	52	55	7.7	Carlson et al. 1986b
40-500	95[G]	?	CS	F	No	21	19	7.1	Finlayson and Verrue 1982
1940	100	?	AS	F	No	20	12	7.1	Sprague 1964
5520	83	?	AS	F	No	20	12	7.9	Sprague 1964
<4000	90	?	FM	F	No	204	162	7.7	Mount 1966
>4000	70	?	FM	F	No	204	162	7.7	Mount 1966
160-400	103	13	FM,GM,DM	S	No	52	43	7.5	Hammermeister et al. 1983
240	96	13	SG	S	No	49	46	7.2	Hammermeister et al. 1983

[A] Total recoverable concentration.

[B] Except as noted, a 0.45-μm membrane filter was used.

[C] Number of paired comparisons.

[D] The abbreviations used are:

AS = Atlantic salmon
BT = Brook trout
CD = *Ceriodaphnia dubia*
CR = Crayfish
CS = Chinook salmon
CT = Cutthroat trout
DA = Daphnids
DM = *Daphnia magna*
EZ = *Elassoma zonatum*
FM = Fathead minnow
GF = Goldfish
GM = Gammarid
PK = *Palaemonetes kadiakensis*
SG = *Salmo gairdneri*

[E] The abbreviations used are:

S = static
R = renewal
F = flow-through

[F] The two numbers are for hardnesses of 50 and 200 mg/L, respectively.

[G] A 0.3-μm glass fiber filter was used.

[H] A 0.10-μm membrane filter was used.

[I] The pH was below 6.5.

[J] The dilution water was a clean river water with TSS and TOC below 5 mg/L.

[K] Only limited information is available concerning this value.

[L] It is assumed that the solution that was filtered was from the test chambers that contained fish and food.

[M] The food was algae.

[N] The food was yeast-trout chow-alfalfa.

[O] The food was frozen adult brine shrimp.

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ATTACHMENT #3

GUIDANCE DOCUMENT
ON DYNAMIC MODELING AND TRANSLATORS
August 1993

Total Maximum Daily Loads (TMDLs) and Permits

o Dynamic Water Quality Modeling

Although not specifically part of the reassessment of water quality criteria for metals, dynamic or probabilistic models are another useful tool for implementing water quality criteria, especially those for protecting aquatic life. Dynamic models make best use of the specified magnitude, duration, and frequency of water quality criteria and thereby provide a more accurate calculation of discharge impacts on ambient water quality. In contrast, steady-state modeling is based on various simplifying assumptions which makes it less complex and less accurate than dynamic modeling. Building on accepted practices in water resource engineering, ten years ago OW devised methods allowing the use of probability distributions in place of worst-case conditions. The description of these models and their advantages and disadvantages is found in the 1991 Technical Support Document for Water Quality-based Toxic Control (TSD).

Dynamic models have received increased attention in the last few years as a result of the perception that static modeling is over-conservative due to environmentally conservative dilution assumptions. This has led to the misconception that dynamic models will always justify less stringent regulatory controls (e.g. NPDES effluent limits) than static models. In effluent dominated waters where the upstream concentrations are relatively constant, however, a dynamic model will calculate a more stringent wasteload allocation than will a steady state model. The reason is that the critical low flow required by many State water quality standards in effluent dominated streams occurs more frequently than once every three years. When other environmental factors (e.g. upstream pollutant concentrations) do not vary appreciably, then the overall return frequency of the steady state model may be greater than once in three years. A dynamic modeling approach, on the other hand, would be more stringent, allowing only a once in three year return frequency. As a result, EPA considers dynamic models to be a more accurate rather than a less stringent approach to implementing water quality criteria.

The 1991 TSD provides recommendations on the use of steady state and dynamic water quality models. The reliability of any modeling technique greatly depends on the accuracy of the data used in the analysis. Therefore, the selection of a model also depends upon the data. EPA recommends that steady state wasteload allocation analyses generally be used where few or no whole effluent toxicity or specific chemical measurements are available, or where daily receiving water flow records are not available. Also, if staff resources are insufficient to use and defend the use of dynamic models, then steady state models may be necessary. If adequate receiving water flow and effluent concentration data are available to estimate frequency distributions, EPA recommends that one of the dynamic wasteload allocation modeling techniques be used to derive wasteload allocations which will more exactly maintain water quality standards. The minimum data required for input into dynamic models include at least 30 years of river flow data and one year of effluent and ambient pollutant concentrations.

o Dissolved-Total Metal Translators

When water quality criteria are expressed as the dissolved form of a metal, there is a need to translate TMDLs and NPDES permits to and from the dissolved form of a metal to the total recoverable form. TMDLs for toxic metals must be able to calculate 1) the dissolved metal concentration in order to ascertain attainment of water quality standards and 2) the total recoverable metal concentration in order to achieve mass balance. In meeting these requirements, TMDLs consider metals to be conservative pollutants and quantified as total recoverable to preserve conservation of mass. The TMDL calculates the dissolved or ionic species of the metals based on factors such as total suspended solids (TSS) and ambient pH. (These assumptions ignore the complicating factors of metals interactions with other metals.) In addition, this approach assumes that ambient factors influencing metal partitioning remain constant with distance down the river. This assumption probably is valid under the low flow conditions typically used as design flows for permitting of metals (e.g., 7Q10, 4B3, etc) because erosion, resuspension, and wet weather loadings are unlikely to be significant and river chemistry is generally stable. In steady-state dilution modeling, metals releases may be assumed to remain fairly constant (concentrations exhibit low variability) with time.

EPA's NPDES regulations require that metals limits in permits be stated as total recoverable in most cases (see 40 CFR §122.45(c)). Exceptions occur when an effluent guideline specifies the limitation in another form of the metal or the approved analytical methods measure only the dissolved form. Also, the permit writer may express a metals limit in another form (e.g., dissolved, valent, or total) when required, in highly unusual cases, to carry out the provisions of the CWA.

The preamble to the September 1984 National Pollutant Discharge Elimination System Permit Regulations states that the total recoverable method measures dissolved metals plus that portion of solid metals that can easily dissolve under ambient conditions (see 49 Federal Register 38028, September 26, 1984). This method is intended to measure metals in the effluent that are or may easily become environmentally active, while not measuring metals that are expected to settle out and remain inert.

The preamble cites, as an example, effluent from an electroplating facility that adds lime and uses clarifiers. This effluent will be a combination of solids not removed by the clarifiers and residual dissolved metals. When the effluent from the clarifiers, usually with a high pH level, mixes with receiving water having significantly lower pH level, these solids instantly dissolve. Measuring dissolved metals in the effluent, in this case, would underestimate the impact on the receiving water. Measuring with the total metals method, on the other hand, would measure metals that would be expected to disperse or settle out and remain inert or be covered over. Thus, measuring total recoverable metals in the effluent best approximates the amount of metal likely to produce water quality impacts.

However, the NPDES rule does not require in any way that State water quality standards be in the total recoverable form; rather, the rule requires permit writers to consider the translation between differing metal forms in the calculation of the permit limit so that a total recoverable limit can be established. Therefore, both the TMDL and NPDES uses of water quality criteria require the ability to translate from the dissolved form and the total recoverable form.

Many toxic substances, including metals, have a tendency to leave the dissolved phase and attach to suspended solids. The partitioning of toxics between solid and dissolved phases can be determined as a function of a pollutant-specific partition coefficient and the concentration of solids. This function is expressed by a linear partitioning equation:

$$C = \frac{C_{Tf}}{1 + K_d \cdot TSS \cdot 10^{-6}}$$

where,

C = dissolved phase metal concentration,
C_{Tf} = total metal concentration,
TSS = total suspended solids concentration, and
K_d = partition coefficient.

A key assumption of the linear partitioning equation is that the sorption reaction reaches dynamic equilibrium at the point of application of the criteria; that is, after allowing for initial mixing the partitioning of the pollutant between the adsorbed and dissolved forms can be used at any location to predict the fraction of pollutant in each respective phase.

Successful application of the linear partitioning equation relies on the selection of the partition coefficient. The use of a partition coefficient to represent the degree to which toxics adsorb to solids is most readily applied to organic pollutants; partition coefficients for metals are more difficult to define. Metals typically exhibit more complex speciation and complexation reactions than organics and the degree of partitioning can vary greatly depending upon site-specific water chemistry. Estimated partition coefficients can be determined for a number of metals, but waterbody or site-specific observations of dissolved and adsorbed concentrations are preferred.

EPA suggests three approaches for instances where a water quality criterion for a metal is expressed in the dissolved form in a State's water quality standards:

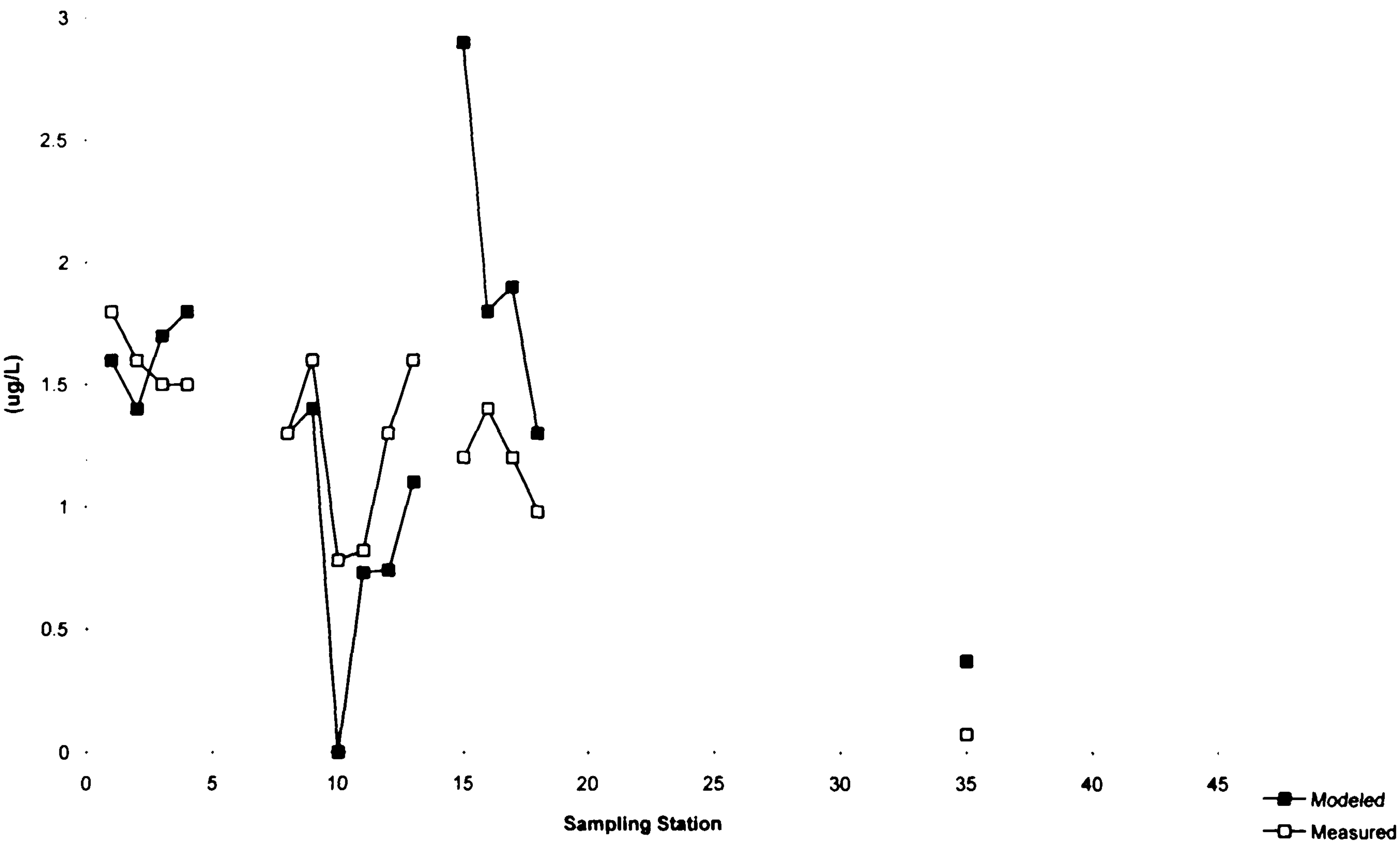
1. Using clean analytical techniques and field sampling procedures with appropriate QA/QC, collect receiving water samples and determine site specific values of K_d for each metal. Use these K_d values to "translate" between total recoverable and dissolved metals in receiving water. This approach is more difficult to apply because it relies upon the availability of good quality measurements of ambient metal concentrations. This approach provides an accurate assessment of the dissolved metal fraction providing sufficient samples are collected. EPA's initial recommendation is that at least four pairs of total recoverable and dissolved ambient metal measurements be made during low flow conditions or 20 pairs over all flow conditions. EPA suggests that the average of data collected during low flow or the 95th percentile highest dissolved fraction for all flows be used. The low flow average provides a representative picture of conditions during the rare low flow events. The 95th percentile highest dissolved fraction for all flows provides a critical condition approach analogous to the approach used to identify low flows and other critical environmental conditions.

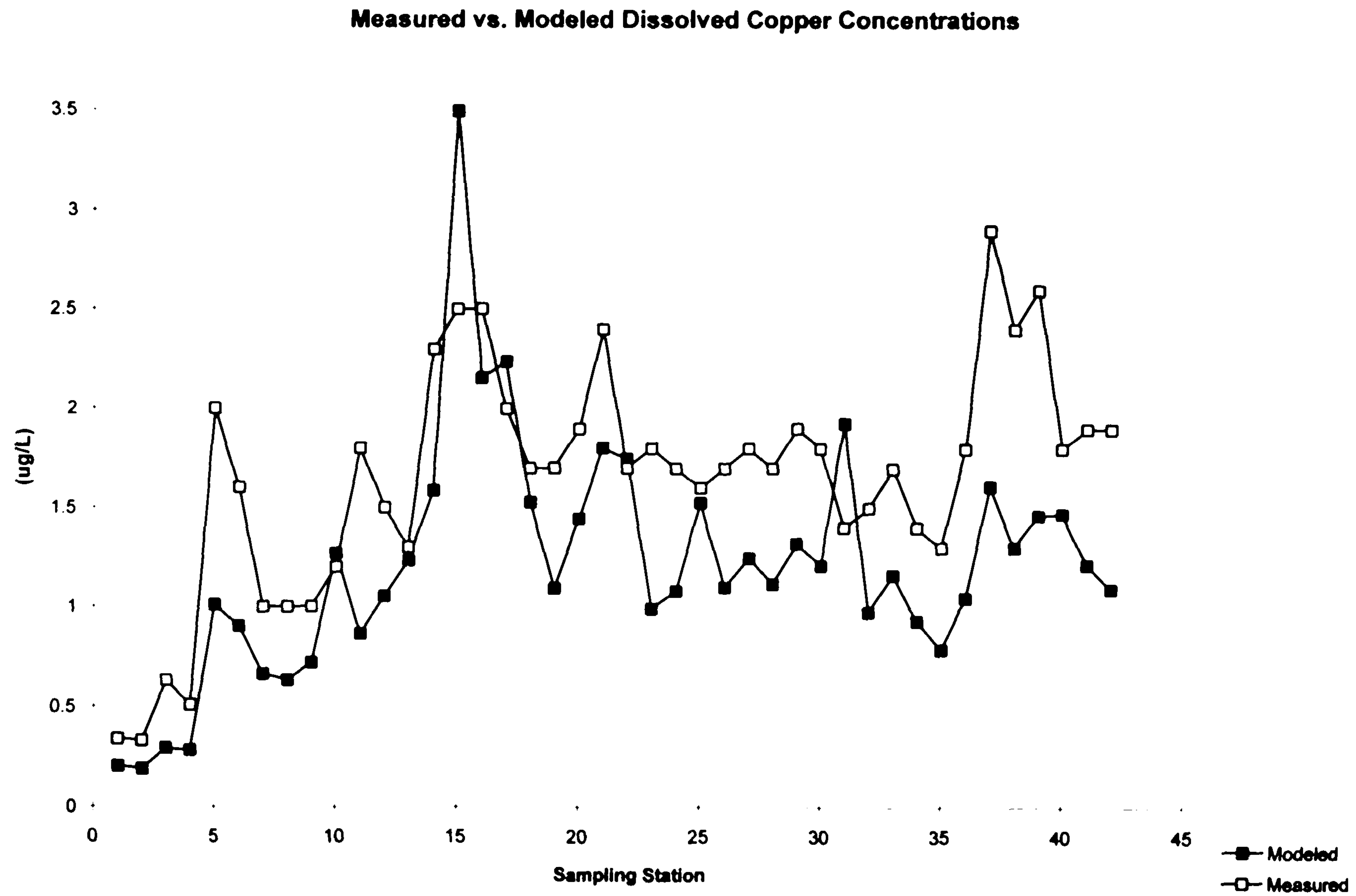
2. Calculate the total recoverable concentration for the purpose of setting the permit limit. Use a value of 1 unless the permittee has collected data (see #1 above) to show that a different ratio should be used. The value of 1 is conservative and will not err on the side of violating standards. This approach is very simple to apply because it places the entire burden of data collection and analysis solely upon permitted facilities. In terms of technical merit, it has the same characteristics of the previous approach. However, permitting authorities may be faced with difficulties in negotiating with facilities on the amount of data necessary to determine the ratio and the necessary quality control methods to assure that the ambient data are reliable.

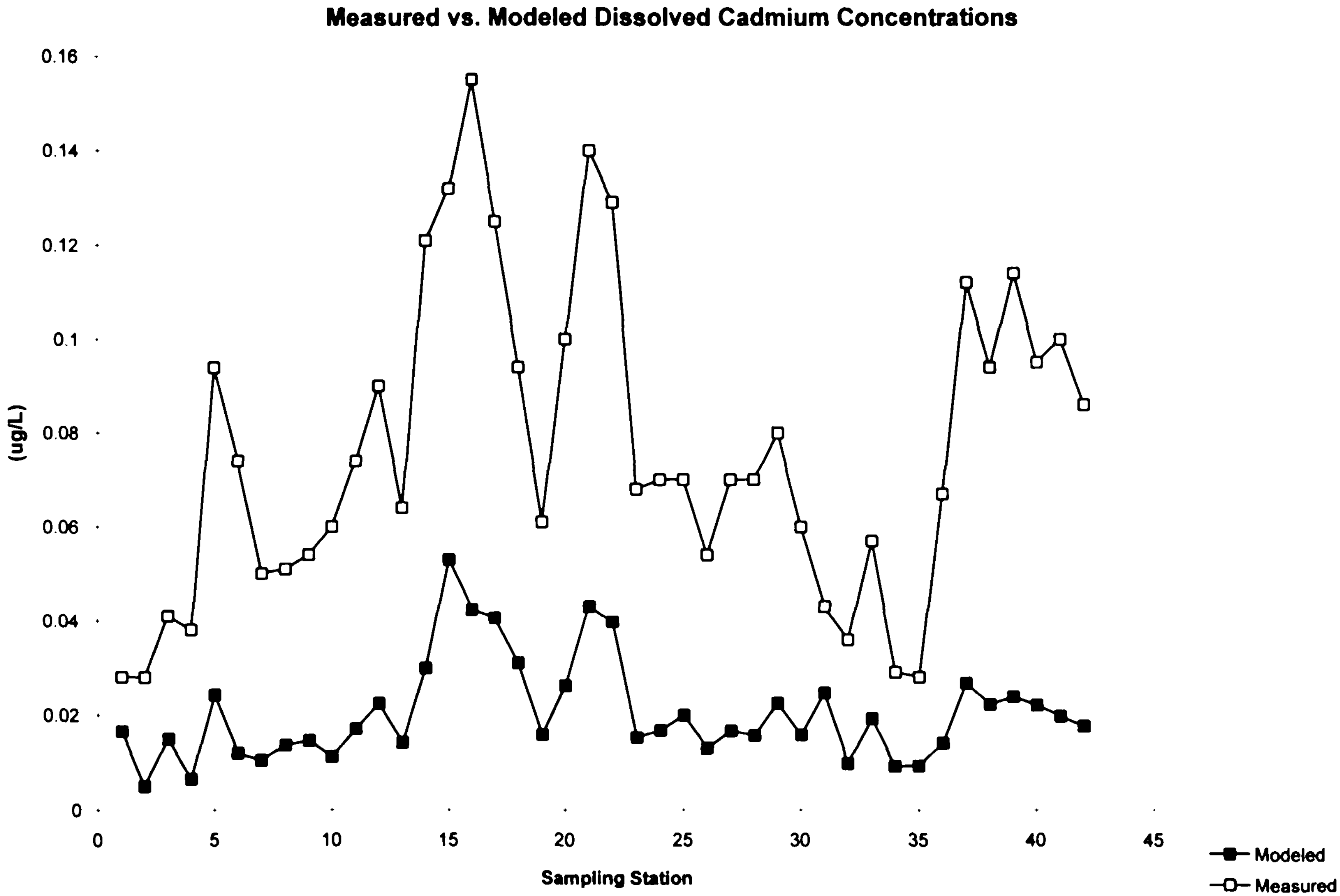
3. Use the historical data on total suspended solids (TSS) in receiving waterbodies at appropriate design flows and K_d values presented in the Technical Guidance Manual for Performing Waste Load Allocations. Book II. Streams and Rivers. EPA-440/4-84-020 (1984) to "translate" between (total recoverable) permits limits and dissolved metals in receiving water. This approach is fairly simple to apply. However, these K_d values are suspect due to possible quality assurance problems with the data used to develop the values. EPA's initial analysis of this approach and these values in one site indicates that these K_d values generally over-estimate the dissolved fraction of metals in ambient waters (see Figures following). Therefore, although this approach may not provide an accurate estimate of the dissolved fraction, the bias in the estimate is likely to be a conservative one.

EPA suggests that regulatory authorities use approaches #1 and #2 where States express their water quality standards in the dissolved form. In those States where the standards are in the total recoverable or acid soluble form, EPA recommends that no translation be used until the time that the State changes the standards to the dissolved form. Approach #3 may be used as an interim measure until the data are collected to implement approach #1.

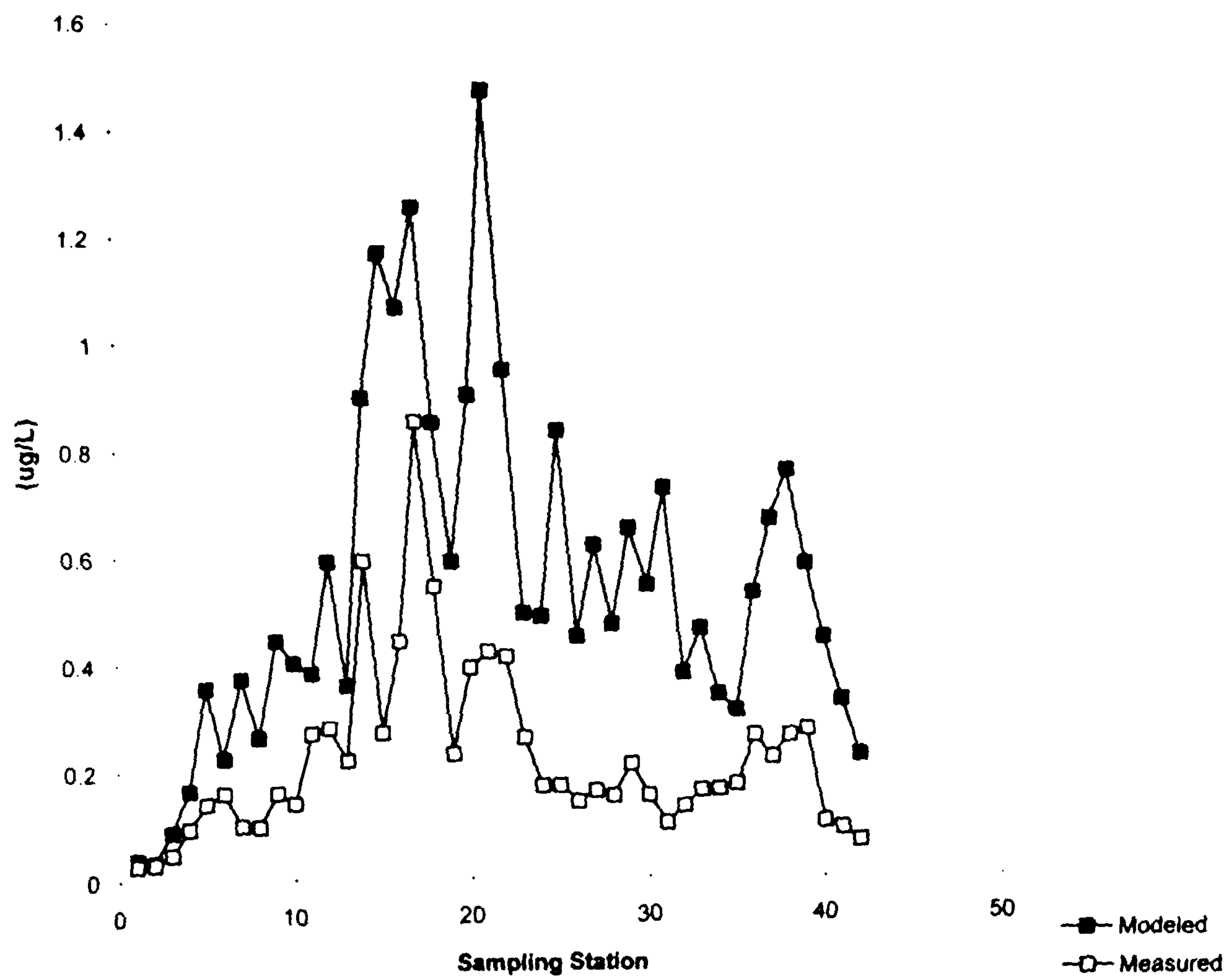
Measured vs. Modeled Dissolved Arsenic Concentrations



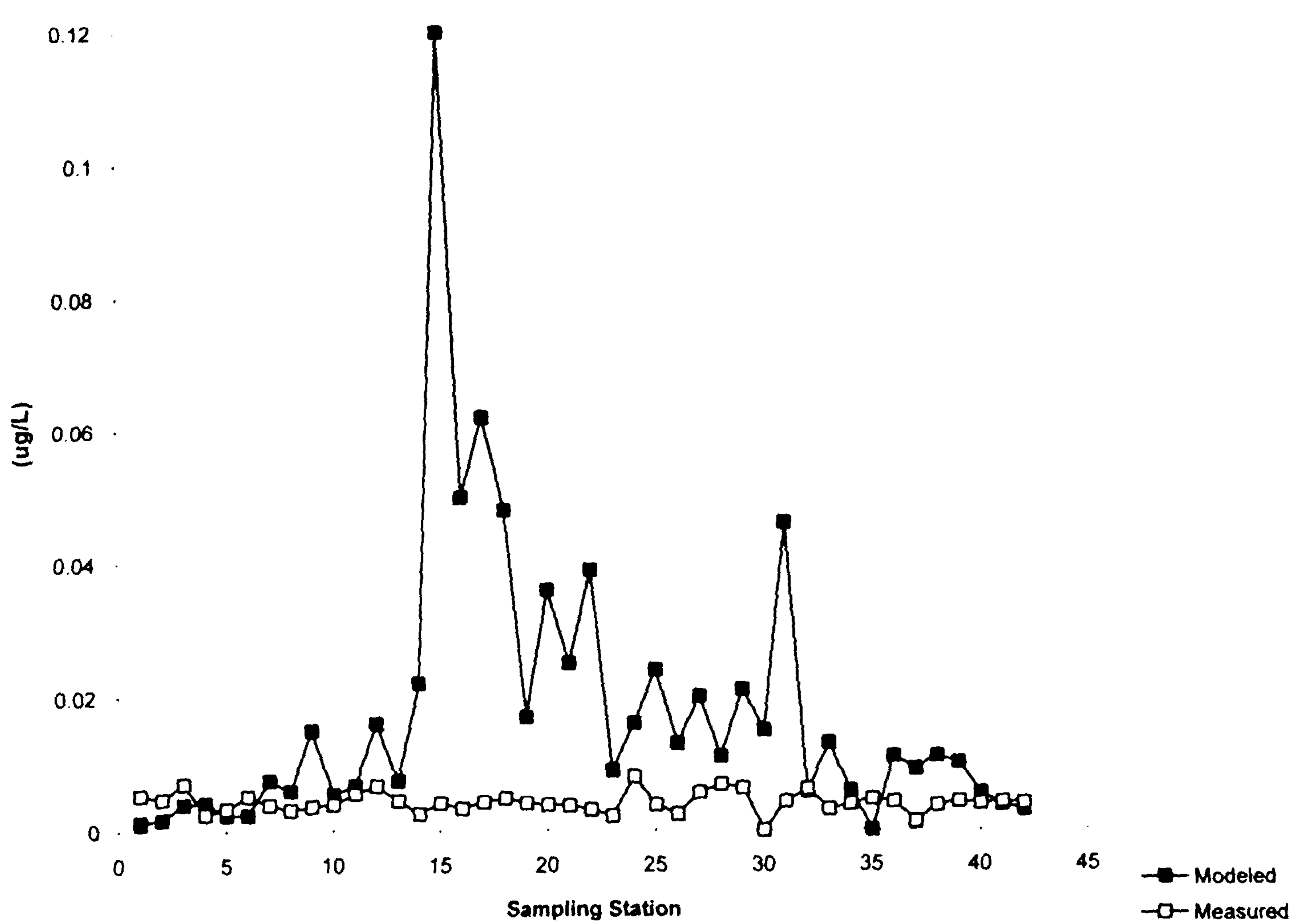

Measured vs. Modeled Dissolved Copper Concentrations
3.5
3
2.5
2
1.5
1
0.5
0
(ug/L)
0
5
10
15
20
25
30
35
40
45
Sampling Station
Modeled
Measured


Measured vs. Modeled Dissolved Cadmium Concentrations
0.16
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
(ug/L)
0
5
10
15
20
25
30
35
40
45
Sampling Station
Modeled
Measured

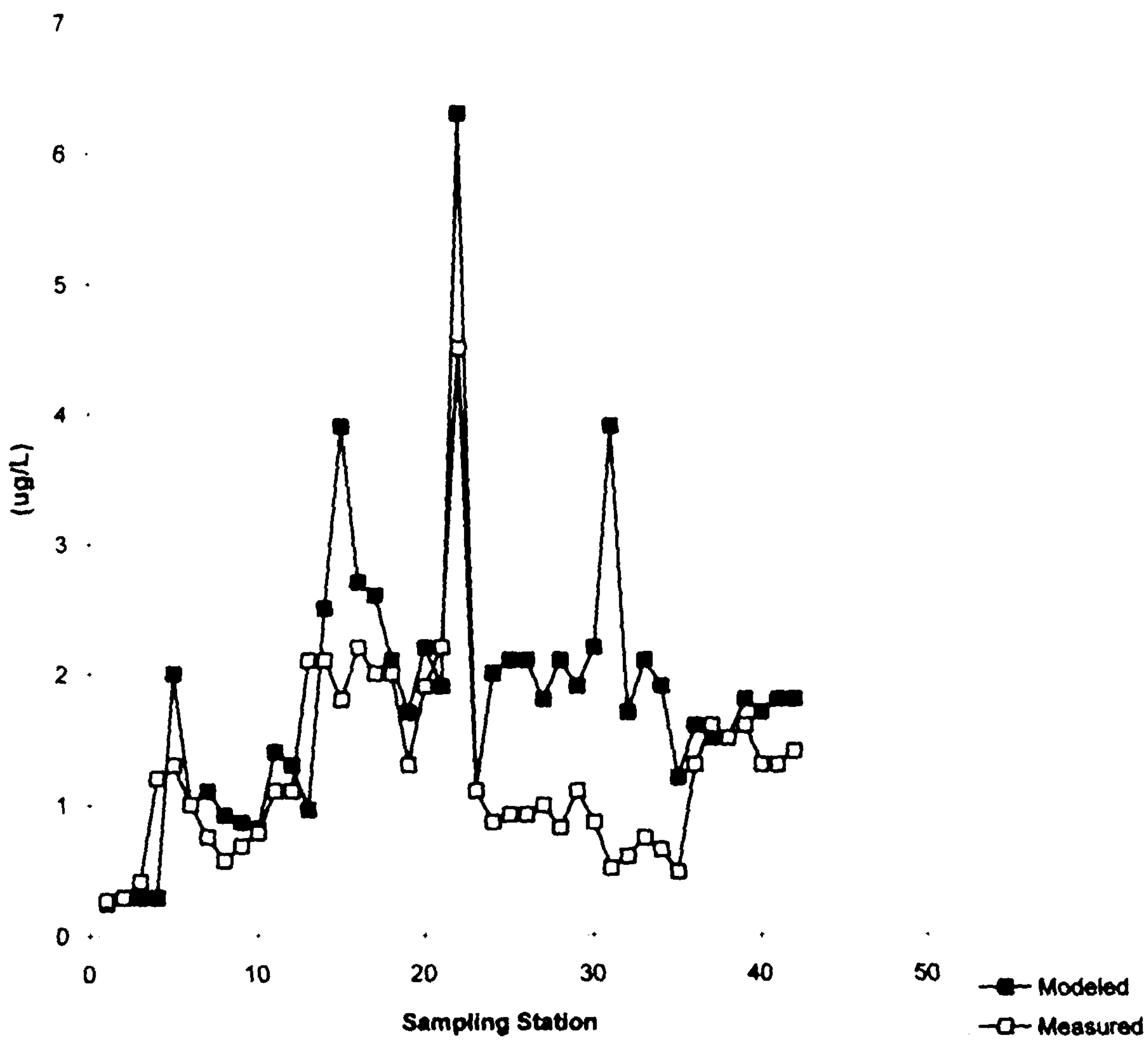
Measured vs. Modeled Dissolved Lead Concentrations



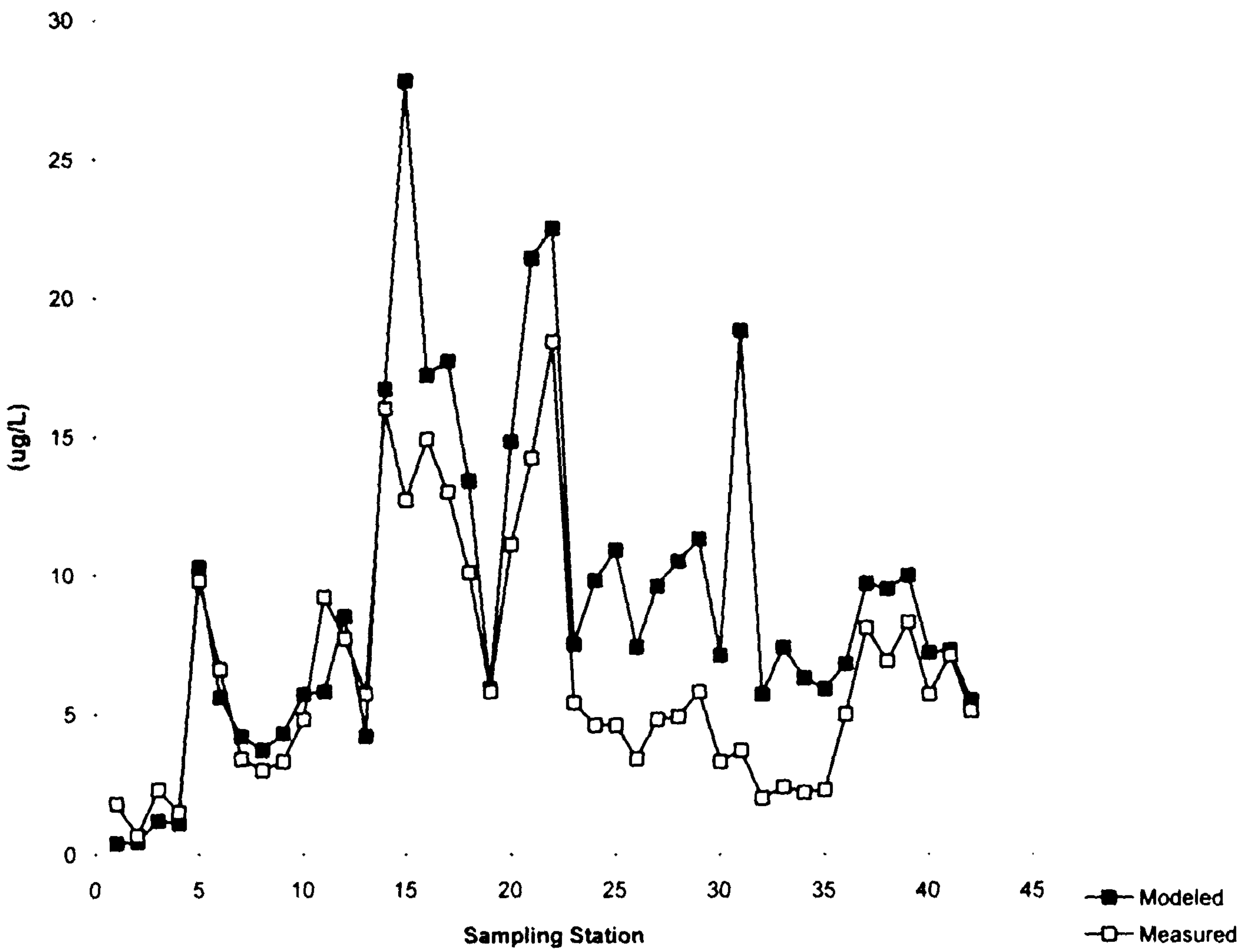
Measured vs. Modeled Dissolved Mercury Concentrations



Measured vs. Modeled Dissolved Nickel Concentrations



Measured vs. Modeled Dissolved Zinc Concentrations



ATTACHMENT #4

GUIDANCE DOCUMENT
ON CLEAN ANALYTICAL TECHNIQUES AND MONITORING

October 1993

Guidance on Monitoring

o Use of Clean Sampling and Analytical Techniques

Pages 98-108 of the WER guidance document (Appendix L of the *Water Quality Standards Handbook-Second Edition*) provides some general guidance on the use of clean techniques. The Office of Water recommends that this guidance be used by States and Regions as an interim step while the Office of Water prepares more detailed guidance.

o Use of Historical DMR Data

With respect to effluent or ambient monitoring data reported by an NPDES permittee on a Discharge Monitoring Report (DMR), the certification requirements place the burden on the permittee for collecting and reporting quality data. The certification regulation at 40 CFR 122.22(d) requires permittees, when submitting information, to state: "I certify under penalty of law that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to assure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, the information submitted is, to the best of my knowledge and belief, true, accurate, and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations."

Permitting authorities should continue to consider the information reported in DMRs to be true, accurate, and complete as certified by the permittee. Under 40 CFR 122.41(l)(8), however, as soon as the permittee becomes aware of new information specific to the effluent discharge that calls into question the accuracy of the DMR data, the permittee must submit such information to the permitting authority. Examples of such information include a new finding that the reagents used in the laboratory analysis are contaminated with trace levels of metals, or a new study that the sampling equipment imparts trace metal contamination. This information must be specific to the discharge and based on actual measurements rather than extrapolations from reports from other facilities. Where a permittee submits information supporting the contention that the previous data are questionable and the permitting authority agrees with the findings of the information, EPA expects that permitting authorities will consider such information in determining appropriate enforcement responses.

In addition to submitting the information described above, the permittee also must develop procedures to assure the collection and analysis of quality data that are true, accurate, and complete. For example, the permittee may submit a revised quality assurance plan that describes the specific procedures to be undertaken to reduce or eliminate trace metal contamination.

APPENDIX K

Procedures for the Initiation of Narrative Biological Criteria

APPENDIX K

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

United States
Environmental Protection
Agency

Office of Science and Technology
Office of Water
Washington, D.C. 20460

EPA-822-B-92-002
October 1992

PROCEDURES FOR INITIATING NARRATIVE BIOLOGICAL CRITERIA

Procedures for Initiating Narrative Biological Criteria

By

George R. Gibson, Jr., Coordinator
Biological Criteria Program
Health and Ecological Criteria Division
Office of Water
U.S. Environmental Protection Agency
Washington, DC

October 1992

ACKNOWLEDGMENTS

Appreciation is extended to all the specialists in the States, EPA Headquarters program offices, and the ten EPA Regional Offices for their suggestions and review comments in the preparation of this document.

Fred Leutner, Kent Ballentine, and Robert Shippen of the Standards and Applied Sciences Division contributed advice and citations pertinent to the proper application of these criteria to EPA regulatory standards.

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF WATER

MEMORANDUM

To: Users of "Procedures for Initiating Narrative Biological Criteria"

Regarding: Guidance for the development of narrative biological criteria

From: Margarete Stasikowski, Director
Health and Ecological Criteria Division
Office of Science and Technology
U.S. EPA

This guidance was written in response to requests from many State water resource agencies for specific information about EPA expectations of them as they prepare narrative biological criteria for the assessment of their surface water resources.

The array of State experiences with this form of water quality evaluation extends from almost no experience in some cases to national leadership roles in others. It may therefore, be that some readers will find this information too involved, while others will feel it is too basic. To the latter we wish to express the sincere hope that this material is a fair approximation of their good examples. To the former, we emphasize that there is no expectation that a State just entering the process will develop a full blown infrastructure overnight. The intent is to outline both the initiation and the subsequent implementation and application of a State program based on commonly collected data as a starting point. User agencies are encouraged to progress through this material at their own best pace as needs and resources determine.

Specific advice, clarification and assistance may be obtained from the U.S. EPA Regional Offices by consultation with the designated resource personnel listed in the appendix to this document.

Attachment

Pnnted on Recycled Paper

Purpose of this Paper

The Biological Criteria Program was initiated by EPA in response to research and interest generated over the last several years by Agency, State, and academic investigators. This interest has been documented in several reports and conference proceedings that were the basis for creation of the program and for the preparation of *Biological Criteria National Program Guidance for Surface Waters* (U.S. Environ. Prot. Agency, 1990a). The overall concept and "narrative biological criteria" are described in that guide.

Because establishing narrative criteria is an important first step in the process, the material that follows here is intended to be an elaboration upon and clarification of the term narrative biological criteria as used in the guide. The emphasis here is on a practical, applied approach with particular attention to cost considerations and the need to introduce the material to readers who may not be familiar with the program.

Introduction and Background

Biological monitoring, assessment and the resultant biological criteria represent the current and increasingly sophisticated process of an evolving water quality measurement technology. This process spans almost 200 years in North America and the entire 20 years of EPA responsibility.

The initial efforts in the 1700's to monitor and respond to human impacts on watercourses were based on physical observations of sediments and debris discharged by towns, commercial operations, and ships in port (Capper, et al. 1983).

Later, chemical analyses were developed to measure less directly observable events. With industrialization, increasing technology, and land development pressures, both types of monitoring were incorporated into the body of our State and Federal public health and environmental legislation.

Valuable as these methods were, early investigations and compliance with water quality standards relied primarily on water column measurements reflecting only conditions at a given time of sampling. Investigators and managers have long recognized this limitation and have used sampling of resident organisms in the streams, rivers, lakes, or estuaries to enhance their understanding of water resource quality over a greater span of time. During the past 20 years, this biological technique has become increasingly sophisticated and reliable and is now a necessary adjunct to the established physical and chemical measures of water resources quality. In fact, the Clean Water Act states in Section 101 (a) that the objective of the law is to *restore and maintain the chemical, physical, and biological integrity of the Nation's waters.*

EPA has therefore concluded that biological assessment and consequent biological criteria are an appropriate and valuable complement to the Nation's surface water management programs. This added approach not only expands and refines this management effort, it is also consistent with the country's growing concern that the environment must be protected and managed for more than the legitimate interests of human health and welfare. The protection of healthy ecosystems is part of EPA's responsibility and is indeed related to the public's welfare. Fish, shellfish, wildlife, and other indigenous flora and fauna of our surface waters require protection as intrinsic components of the natural system. Inherent to the Biological Criteria Program is the restoration and protection of this "biological integrity" of our waters.

A carefully completed survey and subsequent assessment of these resident organisms in relatively undisturbed areas reveal not only the character, e.g., biological integrity, of a natural, healthy waterbody, they also provide a benchmark or biological criterion against which similar systems may be compared where degradation is suspected. Biological measurements also help record waterbody changes over time with less potential temporal variation than physical or chemical approaches to water quality measurement. Thus, they can be used to help determine "existing aquatic life uses" of waterbodies requiring protection under State management programs.

This document elaborates on the initiation of narrative biological criteria as described in *Biological Criteria National Program Guidance for Surface Waters*. Future guidance documents will provide additional technical information to facilitate development and implementation of both narrative and numerical criteria for each of the surface water types.

Narrative Biological Criteria

The first phase of the program is the development of "narrative biological criteria". These are essentially statements of intent incorporated in State water laws to formally consider the fate and status of aquatic biological communities. Officially stated, biological criteria are ". . . numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use" (U.S. Environ. Prot. Agency, 1990a).

While a narrative criterion does not stipulate that numerical indices or other population parameters be used to indicate a particular level of water quality, it does rely upon the use of standard measures and data analyses to make qualitative determinations of the resident communities.

The State, Territory, or Reservation should not only carefully compose the narrative biological criteria statement but should also indicate how its application is to be accomplished. The determination of text (how the narrative biological criteria are written) and measurement procedures (how the criteria will be applied) is up to the individual States in consultation with EPA. Some degree of standardization among States sharing common regions and waters will be in their best interests. This regional coordination and cooperation could help improve efficiency, reduce costs, and expand the data base available to each State so that management determinations can be made with greater certainty.

Attributes of A Sound Narrative Criteria Statement

A narrative biological criterion should:

1. Support the goals of the Clean Water Act to provide for the protection and propagation of fish, shellfish and wildlife, and to restore and maintain the chemical, physical, and biological integrity of the Nation's waters;

2. Protect the most natural biological community possible by emphasizing the protection of its most sensitive components.

3. Refer to specific aquatic, marine, and estuarine community characteristics that must be present for the waterbody to meet a particular designated use, e.g., natural diverse systems with their respective communities or taxa indicated; and then,

4. Include measures of the community characteristics, based on sound scientific principles, that are quantifiable and written to protect and or enhance the designated use;

5. In no case should impacts degrading existing uses or the biological integrity of the waters be authorized.

An Example of A Narrative Biocriteria Statement

The State will preserve, protect, and restore the water resources of [name of State] in their most natural condition. The condition of these waterbodies shall be determined from the measures of physical, chemical, and biological characteristics of each surface waterbody type, according to its designated use. As a component of these measurements, the biological quality of any given water system shall be assessed by comparison to a reference condition(s) based upon similar hydrologic and watershed characteristics that represent the optimum natural condition for that system.

Such reference conditions or reaches of water courses shall be those observed to support the greatest variety and abundance of aquatic life in the region as is expected to be or has been historically found in natural settings essentially undisturbed or minimally disturbed by human impacts, development, or discharges. This condition shall be determined by consistent sampling and reliable measures of selected indicative communities of flora and/or fauna as established by . . . [appropriate State agency or agencies] . . . and may be used in conjunction with acceptable chemical, physical, and microbial water quality measurements and records judged to be appropriate to this purpose.

Regulations and other management efforts relative to these criteria shall be consistent with the objective of preserving, protecting, and restoring the most natural communities of fish, shellfish, and wildlife attainable in these waters; and in all cases shall protect against degradation of the highest existing or subsequently attained uses or biological conditions pursuant to State antidegradation requirements.

Data Gathering to Establish and Support Narrative Biological Criteria

A State need not specifically list in the narrative statement the sampling procedures and parameters to be employed, but it should identify and charge the appropriate administrative authority with this responsibility as indicated parenthetically in the preceding example.

The selection and sampling process, certainly at the outset, should be simple, reliable, and cost effective. In many instances existing data and State procedures will be adequate to initiate a biological criteria program, but there is no limitation on the sophistication or rigor of a State's procedures.

In reviewing existing procedures and in designing new ones, it is important that the planning group include the water resource managers, biologists, and chemists directly involved with the resource base. They should be the primary participants from the outset to help ensure that the data base and derived information adequately support the decisions to be made. .

The State may choose to create procedures and regulations more complex and complete than are indicated here; however, the basic design and methodology should include the following elements:

■ **1. Resource Inventory.** A field review of State water resource conditions and a first hand documentation of the status of water quality relative to the use designation categories ("305(b)" reports) are essential to provide reliable data for the selections of reference sites, test sites, and for setting program priorities.

■ **2. Specific Objectives and Sampling Design.** States will need to design a system identifying "natural, unimpacted" reference sources appropriate to each surface waterbody type in each of the designated use categories in the State (e.g., streams, lakes and reservoirs, rivers, wetlands, estuaries and coastal waters) and the use categories (see example, Page 8) for each grouping of these waterbody types. Sources for defining reference condition may include historical data sets, screening surveys, or a consensus of experts in the region of interest, particularly in significantly disrupted areas as discussed later (see item 6, page 7).

Because natural water courses do not always follow political boundaries, the most effective approach may be a joint or group effort between two or more States. Where this coordination and cooperation is possible, it may produce a superior data base at less cost than any individual State effort. EPA is working through its regional offices to assist in the development of such joint operations through the use of ecoregions and subregions (Gallant et al. 1988). Regional EPA biologists and water quality or standards coordinators can advise and assist with these interstate cooperative efforts.

In any case, reference sites or sources for each waterbody type, subcategory of similar waters, and designated use category will be needed. These may be drawn from "upstream" locations, "far field" transects or selected nearby or "ecoregional" sites representative of relatively unimpacted, highest quality natural settings (U.S. Environ. Prot. Agency, 1990a).

Care must be taken to equate comparable physical characteristics when selecting reference sites for the waterbodies to be evaluated. For example, a site on a piedmont stream cannot be the reference source against which sites on a coastal plain stream are compared; similarly, coastal tidal and nontidal wetlands should not be compared.

The organisms to be collected and communities sampled should represent an array of sensitivities to be as responsive and informative as possible. An example would be to collect fish, invertebrates representing both insects and shellfish, and perhaps macrophytes as elements of the sampling scheme.

■ **3. Collection Methods.** The same sampling techniques should always be employed at both the reference sites and test sites and should be consistent as much as possible for both spatial and temporal conditions. For example, a consistent seining or electroshocking technique should always be used in collecting fish over the same length of stream and with the same degree of effort using the same gear. In addition, the sampling area must be representative of the entire reach or waterbody segment. The temporal conditions to be considered include not only such factors as the length of time spent towing a trawl at a constant speed but also extend to the times of year when data are gathered.

Seasonality of life cycles and natural environmental pressures must be addressed to make legitimate evaluations. For example, the spring hatch of aquatic insects is usually avoided as a sampling period in favor of more stable community conditions later in the summer. Conversely, low nutrient availability in mid-summer may temporarily but cyclically reduce the abundance of estuarine or marine benthos. Dissolved oxygen cycles are another seasonal condition to consider as are migratory patterns of some fish and waterfowl. The entire array of temporal and spatial patterns must be accommodated to avoid inconsistent and misleading data gathering.

Processing and analysis of the collected specimens is usually based on the number and identity of taxa collected and the number of individuals per taxon. This preliminary information is the foundation of most of the subsequent analytical processes used to evaluate community composition. In the course of examining and sorting the plants or animals, notations should be made of any abnormal gross morphological or pathological conditions such as deformities, tumors or lesions. This information on disease and deformities in itself can be an important assessment variable.

Taxonomic sorting can also be the basis for functional groupings of the data, and preservation of the specimens allows for the option of additional analyses after the field season is concluded.

Table 1 is not all inclusive in the sense of a thorough biological investigation, but it does represent an initial approach to the selection of parameters for biological assessment to support the narrative criteria.

Table 1.—Indicator communities and reference sources for biological criteria.

WATERBODY	FLORA / FAUNA INDICATORS	REFERENCE STATIONS
Freshwater Streams	Fish, periphyton & macroinvertebrates, incl. insects & shellfish	Ecoregion, upstream and downstream stations
Lakes & Reservoirs	Same, also macrophytes	May need to start with trophic groups; far- and near-field transects, ecoregions*
Rivers	Same as lake & reservoirs	Upstream and downstream stations; where appropriate, far- and near-field transects, ecoregions*
Wetlands	All of above, plus emergent and terrestrial vegetation & perhaps wildlife & avian spp.	Ecoregion;* far- and near-field transects
Estuarine & near-coastal Waters	Fish, periphyton & macroinvertebrates, esp. shellfish, echinoderms, polychaetes	Far- and near-field transects; ecoregion* or physiographic province

* Where appropriate; ecoregions that are heterogeneous may need to be subdivided into cohesive subregions or these subregions aggregated where financial resources are limited or aquatic systems are large (tidal rivers, estuaries, near-coastal marine waters). Also, major basins and watersheds could be considered for "keystone indicators" for fish and shellfish

■ **4. Quality Control.** Much of the analytical potential and strength of any conclusions reached will depend upon the precision and accuracy of sampling techniques and data handling procedures. Rigorous attention should therefore be given to the design and consistency of data gathering techniques and to the training and evaluation of field and laboratory staff. Data cataloging and record keeping procedures also must be carefully designed and strictly adhered to by all parties involved. EPA Regional Office personnel can provide advice and Agency guidance manuals on this subject; an example is the 1990 field and laboratory manual by the U.S. Environmental Protection Agency, (1990b). Similarly, many States already have excellent quality assurance procedures that can be used as a foundation for their biological criteria program.

■ **5. Analytical Procedures.** The usual approach to biological analyses is to identify the presence of impairment and establish the probability of being certain in that judgment.

For example, if there is a significant increase in the number of deformed or diseased organisms, and a significant decrease in the taxa and/or individuals and in sensitive or intolerant taxa — given that the physical habitats and collection techniques are equivalent — then the study site may be presumed to be degraded. This conclusion will have further support if the trend holds true over time; is also supported by applicable chemical or physical data; or if probable sources are identified. The apparent source or sources of perturbation should then be investigated and further specific diagnostic tests conducted to establish cause. Remedial action may then follow through regulatory or other appropriate management procedures.

■ **6. Reference Condition and Criteria for Significantly Disrupted Areas.** In regions of significantly disrupted land use such as areas of intensive agricultural or urban/suburban development, the only data base available to serve as a reference condition might be simply "the best of what is left." To establish criteria on this basis would mean an unacceptable lowering of water quality objectives and de facto acceptance of degraded conditions as the norm; or worse, as the goal of water quality management. The alternative would be to establish perhaps impossible goals to restore the water system to pristine, pre-development conditions.

A rational solution avoiding these two pitfalls is to establish the reference condition from the body of historical research for the region and the consensus opinion of a panel of qualified water resource experts. The panel, selected in consultation with EPA, should be required to establish an objective and reasonable expectation of the restorable (achievable) water resource quality for the region. The determination would become the basis of the biological criteria selected.

Consistent with State antidegradation requirements, the best existing conditions achieved since November 28, 1975 [see 40 CFR 131.3(c) and 131.12(a)(1)] must be the lowest acceptable status for interim consideration while planning, managing, and regulating to meet the higher criteria established above. In this way reasonable progress can be made to improve water quality without making unrealistic demands upon the community.

Application of Biological Criteria to State Surface Water Use Attainability Procedures

Another application of the data collected is in helping define the designated uses to be achieved by comparing all test sites relative to the benchmark of reference conditions established per designated use category. Biological criteria can be used to help define the level of protection for "aquatic life use" designated uses for surface waters. These criteria also help determine relative improvement or decline of water resource quality, and should be equated to appropriate reference site conditions as closely as possible. Determinations of attainable uses and biological conditions should be made in accordance with the requirements stipulated in Section 131.10 of the *EPA Water Quality Standards Regulations* (40 CFR 131). A hypothetical State-designated use category system might be as follows:

- **Class A: Highest quality or Special Category State waters.** Includes those designated as unique aesthetic or habitat resources and fisheries, especially protected shellfish waters. No discharges of any kind and no significant landscape alterations are permitted in the drainage basins of these waters. Naturally occurring biological life shall be attained, maintained, and protected in all respects. (Indicator sensitive resident species might be designated to help define each class, e.g., trout, some darters, mayflies, oysters, or clams, etc.)

- **Class B: High quality waters suitable for body contact.** Only highly treated nonimpacting discharges and land development with well established riparian vegetative buffer zones are allowed. Naturally occurring biological life shall be protected and no degradation of the aquatic communities of these waters is allowed. (Indicator sensitive species might be suckers and darters, stoneflies, or soft-shelled clams, etc.)

- **Class C: Good quality water but affected by runoff from prevailing developed land uses.** Shore zones are protected, but buffer zones are not as extensive as Class B. Highly treated, well-diluted final effluent permitted. Existing aquatic life and community composition shall be protected and no further degradation of the aquatic communities is allowed. (Indicator sensitive species might be sunfish, caddisflies, or blue crabs, etc.)

- **Class D: Lowest quality water in State's designated use system.** Ambient water quality must be or become sufficient to support indigenous aquatic life and no further degradation of the aquatic community is allowed. Structure and function of aquatic community must be preserved, but species composition may differ from Class C waters.

Since all States have some form of designated use classification system, bioassessment procedures can be applied to each surface water type by class and the information used to help determine relative management success or failure. In concert with other measurements, bioassessments and biocriteria help determine designated use attainment under the Clean Water Act. This attainment or nonattainment in turn determines the need for or the conditions of such regulatory requirements as total maximum daily loads (TMDLs) and National Pollutant Discharge Elimination System (NPDES) permits. In addition, biological assessments based on these biological criteria can be used to help meet section 305(b) of the Clean Water Act, which requires periodic reports from the States on the status of their surface water resources. The procedure also can be used to support regulatory actions, detect previously unidentified problems, and help establish priorities for management projects (see "Additional Applications of Biological Criteria," Page 10).

Table 2 is a simplified illustration of this approach to evaluating comprehensive surface water quality conditions by each designated use to help determine and report "designated use attainment" status.

It is important to construct and calibrate <u>each</u> table according to consistent regional and habitat conditions.

Using quantitative parameters or metrics derived from the data base and the reference condition, standings in the tables can be established from which relative status can be defined. This material can eventually serve as the basis for numeric biological criteria.

A well-refined quantitative approach to the narrative process can be administratively appended to the States' preexisting narrative criteria to meet future needs for numeric criteria. This can be accomplished fairly easily by amending the narrative statement, as illustrated on page 3, to include a designated regulatory responsibility for the appropriately identified agency. The advantage of this approach is as changes in the supportive science evolve, the criteria can be appropriately adjusted.

Table 2.—Data display to facilitate evaluating waterbody condition and relative designated use attainment.

DESIGNATED USE (per St. water type)	BIOLOGICAL ASSESSMENT PARAMETERS (by number) Taxa Inverts	Taxa Fish	Invertebrates Intolerants	Fish Intolerants	Diseased
Highest quality in designated use	high	high	high	high	low
Good quality in designated use					
Adequate to designated use					
Marginal for designated use					
Poor quality	low	low	low	low	high

DESIGNATED USE (per St. water type)	PUBLIC HEALTH, CHEMICAL, PHYSICAL DATA T. Coli	E. Coli	D.O.	pH	PO4	NO3	Turb.
Highest quality in designated use	low	low	high		Usually low	Usually low	Usually low
Good quality in designated use				Vbl by region			
Adequate to designated use							
Marginal for designated use							
Poor quality	high	high	low		Usually high	Usually high	Usually high

Further, the compiling of physical and chemical data with the biological data facilitates comprehensive evaluations and aids in the investigation of causes of evident water quality declines. Having the numbers all in one place helps the water resource manager assess conditions. However, it is important to note that none of these parameters should supercede the others in management or regulations because they have unique as well as overlapping attributes. Failure of a designated site to meet any one of a State's physical, chemical, or biological criteria should be perceived as sufficient justification for corrective action.

One other note on the use of biological criteria is important. The data gathered should be comprehensively evaluated on a periodic basis. This gives the manager an opportunity to assess relative monitoring and management success, monitor the condition of the reference sites, and adjust procedures accordingly. As conditions improve, it will also be important to reassess and adjust the biological criteria. This may be particularly appropriate in the case of "significantly disrupted areas" discussed earlier.

Additional Applications of Biological Criteria

As shown in the previous illustrations, narrative biological criteria can have many applications to the management and enhancement of surface water quality.

- **Refinement and augmentation of existing waterbody monitoring procedures.** With between 200 and 500 new chemicals entering the market annually, it is impossible to develop chemical criteria that address them all. Further, synergism between even regulated chemicals meeting existing standards may create degraded conditions downstream that are identifiable only by using biological monitoring and criteria. Thus, the approach may help identify and correct problems not previously recognized.

- **Non-chemical impairments** (e.g., degradation of physical habitats, changes in hydrologic conditions, stocking, and harvesting) can be identified. Remediation of these impairments, when they are the primary factor, can be less expensive and more relevant than some point source abatements.

- **Waterbody management decisionmaking.** By reviewing an array of diverse parameters in a comprehensive manner, the decisionmaker is able to make better judgments. The strengths of this diversity can be used to determine with greater confidence the resources to assign to a given waterbody or groups of waterbodies in the allocation of scarce manpower or funds. The information can also be used to set priorities where required by law, such as section 303(d) of the Clean Water Act, or to help guide regulatory decisions.

  In conjunction with nutrient, chemical, and sediment parameters, biological information and criteria are an important tool for watershed investigations. The combined data helps the manager select areas of likely nonpoint as well as point sources of pertebation and makes it possible to focus remedial efforts on key subbasins.

- **Regulatory aspect.** Once established to the satisfaction of the State and EPA, the biocriteria process may be incorporated in the State's system of regulations as part of its surface water quality protection and management program. Biological assessment and criteria can become an important additional tool in this context as the Nation increasingly upgrades the quality of our water resources.

Perspective of the Future: Implementing Biological Criteria

This guide to narrative biological criteria was composed with the fiscal and technical constraints of all the States, Territories, and Reservations in mind. The array of scientific options available to biological assessment and criteria illustrated here is by no means exhaustive, and many jurisdictions will prefer a more involved approach. In no way is this guide intended to restrain States from implementing more detailed or rigorous programs. In fact, we welcome comments and suggestions for additional techniques and parameters to consider.

The basic approach discussed here, while compiled to be the least demanding on State budgets, equipment, and manpower pools, consists of a reliable, reproducible scientific method. The metrics considered should not be restricted to those illustrated in this guide. Rather, they should be developed from the expertise of State biologists and water resource managers — perhaps in concert with colleagues in neighboring States for a coordinated regional approach to waterbodies and natural biological regions that cross political boundaries. Good science should be applied to a realistic appraisal of what can actually be accomplished, and the EPA regional office specialists, listed on the following pages, can assist in such assessments and coordination. For more detailed discussions of sampling and analytical methods, the reader is also referred to the references appended to this text.

The structure for narrative biological criteria described here is an appropriate interim step for the eventual development of numeric biological criteria. The infrastructure developed now may be expanded and refined to meet future needs.

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U.S. EPA Regional Sources of Technical Assistance

REGION 1: JFK Federal Building, Boston, MA 02203
Regional Biologist: Pete Nolan/Celeste Barr (617) 860-4343
Monitoring Coordinator: Diane Switzer (617) 860-4377
Water Quality Standards Coordinator: Eric Hall (617) 565-3533

REGION 2: 26 Federal Plaza, New York, NY 10278
Regional Biologist: Jim Kurtenbach (908) 321-6716
Monitoring Coordinator: Randy Braun (908) 321-6692
Water Quality Standards Coordinator: Felix Locicero (212) 264-5691

REGION 3: 841 Chestnut Street, Philadelphia, PA 19107
Regional Biologist: Ron Preston (304) 233-2315
Monitoring Coordinator: Chuck Kanetsky (215) 597-8176
Water Quality Standards Coordinator: Helene Drago (215) 597-9911

REGION 4: 345 Courtland Street, NE, Atlanta, GA 30365
Regional Biologist: Hoke Howard/Jerry Stober/William Peltier (706) 546-2296
Monitoring Coordinator: Larinda Tervelt (706) 347-2126
Water Quality Standards Coordinator: Fritz Wagener/Jim Harrison (706) 347-3396

REGION 5: 230 South Dearborn Street, Chicago, IL 60604
Regional Biologist: Charles Steiner (312) 353-9070
Monitoring Coordinator: Donna Williams (312) 886-6233
Water Quality Standards Coordinators: David Pfiefer (312) 353-9024
Tom Simon (312) 353-8341

REGION 6: 1445 Ross Avenue, Suite 1200, Dallas, TX 75202
Regional Biologist: Evan Hornig/Philip Crocker/Terry Hollister (214) 655-2289
Monitoring Coordinator: Charles Howell (214) 655-2289
Water Quality Standards Coordinator: Cheryl Overstreet (214) 655-7145

REGION 7: 726 Minnesota Avenue, Kansas City, KS 66101
Regional Biologist: Michael Tucker/Gary Welker (913) 551-5000
Monitoring Coordinator: John Helvig (913) 551-5002
Water Quality Standards Coordinator: Lawrence Shepard (913) 551-7441

REGION 8: 999 18th Street, Suite 500, Denver, CO 80202-2405
Regional Biologist: Loys Parrish (303) 236-5064
Monitoring Coordinator: Phil Johnson (303) 293-1581
Water Quality Standards Coordinator: Bill Wuerthele (303) 293-1586

REGION 9: 75 Hawthorne Street, San Francisco, CA 94105
Regional Biologist: Peter Husby (415) 744-1488
Monitoring Coordinator: Ed Liu (415) 744-2006
Water Quality Standards Coordinator: Phillip Woods (415) 744-1997

REGION 10: 1200 Sixth Avenue, Seattle, WA 98101
Regional Biologist: Joseph Cummins (206) 871-0748, ext. 1247
Monitoring Coordinator: Gretchen Hayslip (206) 553-1685
Water Quality Standards Coordinators: Sally Marquis (206) 553-2116
Marica Lagerloeff (206) 553-0176

HEADQUARTERS: 401 M Street SW, Biocriteria Program (WH 586), Washington, DC 20640
Program Coordinators: George Gibson (202) 260-7580
Susan Jackson (202) 260-1800

NOTE: *Address provided is the EPA Regional Office; personnel indicated may be located at satellite facilities.*

APPENDIX L

Interim Guidance on Determination and Use of Water-Effect Ratios for Metals

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

FEB 22 1994

EPA-823-B-94-001

MEMORANDUM

SUBJECT: Use of the Water-Effect Ratio in Water Quality Standards

FROM: Tudor T. Davies, Director
Office of Science and Technology

TO: Water Management Division Directors, Regions I - X
State Water Quality Standards Program Directors

PURPOSE

There are two purposes for this memorandum.

The first is to transmit the Interim Guidance on the Determination and Use of Water-Effect Ratios for Metals. EPA committed to developing this guidance to support implementation of federal standards for those States included in the National Toxics Rule.

The second is to provide policy guidance on whether a State's application of a water-effect ratio is a site-specific criterion adjustment subject to EPA review and approval/disapproval.

BACKGROUND

In the early 1980's, members of the regulated community expressed concern that EPA's laboratory-derived water quality criteria might not accurately reflect site-specific conditions because of the effects of water chemistry and the ability of species to adapt over time. In response to these concerns, EPA created three procedures to derive site-specific criteria. These procedures were published in the Water Quality Standards Handbook, 1983.

Site-specific criteria are allowed by regulation and are subject to EPA review and approval. The Federal water quality standards regulation at section 131.11(b)(1) provides States with the opportunity to adopt water quality criteria that are "...modified to reflect site-specific conditions." Under section 131.5(a)(2), EPA reviews standards to determine "whether a State has adopted criteria to protect the designated water uses."

On December 22, 1992, EPA promulgated the National Toxics Rule which established Federal water quality standards for 14 States which had not met the requirements of Clean Water Act Section 303(c)(2)(B). As part of that rule, EPA gave the States discretion to adjust the aquatic life criteria for metals to reflect site-specific conditions through use of a water-effect ratio. A water-effect ratio is a means to account for a difference between the toxicity of the metal in laboratory dilution water and its toxicity in the water at the site.

In promulgating the National Toxics Rule, EPA committed to issuing updated guidance on the derivation of water-effect ratios. The guidance reflects new information since the previous guidance and is more comprehensive in order to provide greater clarity and increased understanding. This new guidance should help standardize procedures for deriving water-effect ratios and make results more comparable and defensible.

Recently, an issue arose concerning the most appropriate form of metals upon which to base water quality standards. On October 1, 1993, EPA issued guidance on this issue which indicated that measuring the dissolved form of metal is the recommended approach. This new policy however, is prospective and does not affect the criteria in the National Toxics Rule. Dissolved metals criteria are not generally numerically equal to total recoverable criteria and the October 1, 1993 guidance contains recommendations for correction factors for fresh water criteria. The determination of site-specific criteria is applicable to criteria expressed as either total recoverable metal or as dissolved metal.

DISCUSSION

Existing guidance and practice are that EPA will approve site- specific criteria developed using appropriate procedures. That policy continues for the options set forth in the interim guidance transmitted today, regardless of whether the resulting criterion is equal to or more or less stringent than the EPA national 304(a) guidance. This interim guidance supersedes all guidance concerning water-effect ratios previously issued by the Agency.

Each of the three options for deriving a final water-effect ratio presented in this interim guidance meets the scientific and technical acceptability test for deriving site-specific criteria.

Option 3 is the simplest, least restrictive and generally the least expensive approach for situations where simulated downstream water appropriately represents a "site." It is a fully acceptable approach for deriving the water-effect ratio although it will generally provide a lower water-effect ratio than the other 2 options. The other 2 options may be more costly and time consuming if more than 3 sample periods and water effect ratio measurements are made, but are more accurate, and may yield a larger, but more scientifically defensible site specific criterion.

Site-specific criteria, properly determined, will fully protect existing uses. The waterbody or segment thereof to which the site-specific criteria apply must be clearly defined. A site can be defined by the State and can be any size, small or large, including a watershed or basin. However, the site-specific criteria must protect the site as a whole. It is likely to be more cost-effective to derive any site-specific criteria for as large an area as possible or appropriate. It is emphasized that site-specific criteria are ambient water quality criteria applicable to a site. They are not intended to be direct modifications to National Pollutant Discharge Elimination System (NPDES) permit limits. In most cases the "site" will be synonymous with a State's "segment" in its water quality standards. By defining sites on a larger scale, multiple dischargers can collaborate on water-effect ratio testing and attain appropriate site-specific criteria at a reduced cost.

More attention has been given to water-effect ratios recently because of the numerous discussions and meetings on the entire question of metals policy and because WERs were specifically applied in the National Toxics Rule. In comments on the proposed National Toxics Rule, the public questioned whether the EPA promulgation should be based solely on the total recoverable form of a metal. For the reasons set forth in the final preamble, EPA chose to promulgate the criteria based on the total recoverable form with a provision for the application of a water-effect ratio. In addition, this approach was chosen because of the unique difficulties of attempting to authorize site-specific criteria modifications for nationally promulgated criteria.

EPA now recommends the use of dissolved metals for States revising their water quality standards. Dissolved criteria may also be modified by a site-specific adjustment.

While the regulatory application of the water-effect ratio applied only to the 10 jurisdictions included in the final National Toxics Rule for aquatic life metals criteria, we understood that other States would be interested in applying WERs to their adopted water quality standards. The guidance upon which to base the judgment of the acceptability of the water-effect ratio applied by the State is contained in the attached Interim Guidance on The Determination and Use of Water-Effect Ratios for Metals. It should be noted that this guidance also provides additional information on the recalculation procedure for site-specific criteria modifications.

Status of the Water-effect Ratio (WER) in non-National Toxics Rule States

A central question concerning WERs is whether their use by a State results in a site-specific criterion subject to EPA review and approval under Section 303(c) of the Clean Water Act?

Derivation of a water-effect ratio by a State is a site-specific criterion adjustment subject to EPA review and approval/disapproval under Section 303(c). There are two options by which this review can be accomplished.

Option 1: A State may derive and submit each individual water-effect ratio determination to EPA for review and approval. This would be accomplished through the normal review and revision process used by a State.

Option 2: A State can amend its water quality standards to provide a formal procedure which includes derivation of water-effect ratios, appropriate definition of sites, and enforceable monitoring provisions to assure that designated uses are protected. Both this procedure and the resulting criteria would be subject to full public participation requirements. Public review of a site-specific criterion could be accomplished in conjunction with the public review required for permit issuance. EPA would review and approve/disapprove this protocol as a revised standard once. For public information, we recommend that once a year the State publish a list of site-specific criteria.

An exception to this policy applies to the waters of the jurisdictions included in the National Toxics Rule. The EPA review is not required for the jurisdictions included in the National Toxics Rule where EPA established the procedure for the State for application to the criteria promulgated. The National Toxics Rule was a formal rulemaking process with notice and comment by which EPA pre-authorized the use of a correctly applied water-effect ratio. That same process has not yet taken place in States not included in the National Toxics Rule.

However, the National Toxics Rule does not affect State authority to establish scientifically defensible procedures to determine Federally authorized WERs, to certify those WERs in NPDES permit proceedings, or to deny their application based on the State's risk management analysis.

As described in Section 131.36(b)(iii) of the water quality standards regulation (the official regulatory reference to the National Toxics Rule), the water-effect ratio is a site-specific calculation. As indicated on page 60866 of the preamble to the National Toxics Rule, the rule was constructed as a rebuttable presumption. The water-effect ratio is assigned a value of 1.0 until a different water-effect ratio is derived from suitable tests representative of conditions in the affected waterbody. It is the responsibility of the State to determine whether to rebut the assumed value of 1.0 in the National Toxics Rule and apply another value of the water-effect ratio in order to establish a site-specific criterion. The site-specific criterion is then used to develop appropriate NPDES permit limits. The rule thus provides a State with the flexibility to derive an appropriate site-specific criterion for specific waterbodies.

As a point of emphasis, although a water-effect ratio affects permit limits for individual dischargers, it is the State in all cases that determines if derivation of a site-specific criterion based on the water-effect ratio is allowed and it is the State that ensures that the calculations and data analysis are done completely and correctly.

CONCLUSION

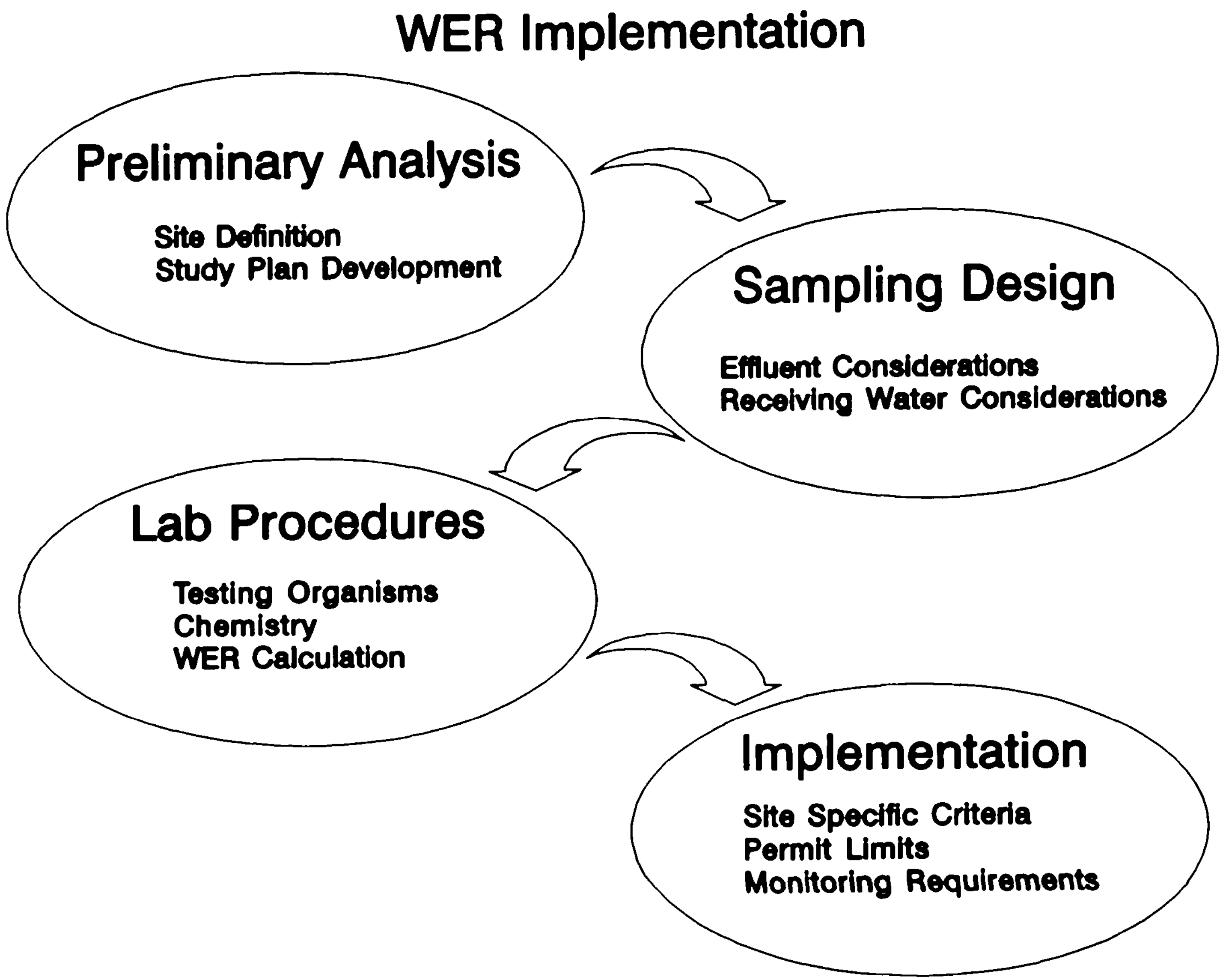
This interim guidance explains and clarifies the use of site-specific criteria. It is issued as interim guidance because it will be included as part of the process underway for review and possible revision of the national aquatic life criteria development methodology guidelines. As part of that review, this interim guidance is subject to amendment based on comments, especially those from the users of the guidance. At the end of the guidelines revision process the guidance will be issued as "final."

EPA is interested in and encourages the submittal of high quality datasets that can be used to provide insights into the use of these guidelines and procedures. Such data and technical comments should be submitted to Charles E. Stephan at EPA's Environmental Research Laboratory at Duluth, MN. A complete address, telephone number and fax number for Mr. Stephan are included in the guidance itself. Other questions or comments should be directed to the Standards and Applied Science Division (mail code 4305, telephone 202-260-1315).

There is attached to this memorandum a simplified flow diagram and an implementation procedure. These are intended to aid a user by placing the water-effect ratio procedure in the context of proceeding from at site-specific criterion to a permit limit. Following these attachments is the guidance itself.

Attachments

cc: Robert Perciasepe, OW
Martha G. Prothro, OW
William Diamond, SASD
Margaret Stasikowski, HECD
Mike Cook, OWEC
Cynthia Dougherty, OWEC
Lee Schroer, OGC
Susan Lepow, OGC
Courtney Riordan, ORD
ORD (Duluth and Narragansett Laboratories)
ESD Directors, Regions I - VIII, X
ESD Branch, Region IX
Water Quality Standards Coordinators, Regions I - X


WER Implementation
Preliminary Analysis
Site Definition
Study Plan Development
Sampling Design
Effluent Considerations
Receiving Water Considerations
Lab Procedures
Testing Organisms
Chemistry
WER Calculation
Implementation
Site Specific Criteria
Permit Limits
Monitoring Requirements

WATER-EFFECT RATIO IMPLEMENTATION

PRELIMINARY ANALYSIS & PLAN FORMULATION

- Site definition

  - How many discharges must be accounted for? Tributaries? See page 17.
  - What is the waterbody type? (i.e., stream, tidal river, bay, etc.). See page 44 and Appendix A.
  - How can these considerations best be combined to define the relevant geographic "site"? See Appendix A @ page 82.

- Plan Development for Regulatory Agency Review

  - Is WER method 1 or 2 appropriate? (e.g., Is design flow a meaningful concept or are other considerations paramount?). See page 6.
  - Define the effluent & receiving water sample locations
  - Describe the temporal sample collection protocols proposed. See page 48.
  - Can simulated site water procedure be done, or is downstream sampling required? See Appendix A.
  - Describe the testing protocols - test species, test type, test length, etc. See page 45, 50; Appendix I.
  - Describe the chemical testing proposed. See Appendix C.
  - Describe other details of study - flow measurement, QA/QC, number of sampling periods proposed, to whom the results are expected to apply, schedule, etc.

SAMPLING DESIGN FOR STREAMS

- Discuss the quantification of the design streamflow (e.g., 7Q10) - USGS gage directly, by extrapolation from USGS gage, or ?

- Effluents

  - measure flows to determine average for sampling day
  - collect 24 hour composite using "clean" equipment and appropriate procedures; avoid the use of the plant's daily composite sample as a shortcut.

- Streams

  - measure flow (use current meter or read from gage if available) to determine dilution with effluent; and to check if within acceptable range for use of the data (i.e., design flow to 10 times the design flow).
  - collect 24 hour composite of upstream water.

LABORATORY PROCEDURES (NOTE: These are described in detail in interim guidance).

- Select appropriate primary & secondary tests

- Determine appropriate cmcWER and/or cccWER

- Perform chemistry using clean procedures, with methods that have adequate sensitivity to measure low concentrations, and use appropriate QA/QC

- Calculate final water-effect ratio (FWER) for site. See page 36.

IMPLEMENTATION

- Assign FWERs and the site specific criteria for each metal to each discharger (if more than one).

- perform a waste load allocation and total maximum daily load (if appropriate) so that each discharger is provided a permit limit.

- establish monitoring condition for periodic evaluation of instream biology (recommended)

- establish a permit condition for periodic testing of WER to verify site-specific criterion (NTR recommendation)

Interim Guidance on

Determination and Use of

Water-Effect Ratios for Metals

February 1994

U.S. Environmental Protection Agency

Office of Water
Office of Science and Technology
Washington, D.C.

Office of Research and Development
Environmental Research Laboratories
Duluth, Minnesota
Narragansett, Rhode Island

NOTICES

This document has been reviewed by the Environmental Research Laboratories, Duluth, MN and Narragansett, RI (Office of Research and Development) and the Office of Science and Technology (Office of Water), U.S. Environmental Protection Agency, and approved for publication.

Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

FOREWORD

This document provides interim guidance concerning the experimental determination of water-effect ratios (WERs) for metals; some aspects of the use of WERs are also addressed. It is issued in support of EPA regulations and policy initiatives involving the application of water quality criteria and standards for metals. This document is agency guidance only. It does not establish or affect legal rights or obligations. It does not establish a binding norm or prohibit alternatives not included in the document. It is not finally determinative of the issues addressed. Agency decisions in any particular case will be made by applying the law and regulations on the basis of specific facts when regulations are promulgated or permits are issued.

This document is expected to be revised periodically to reflect advances in this rapidly evolving area. Comments, especially those accompanied by supporting data, are welcomed and should be sent to: Charles E. Stephan, U.S. EPA, 6201 Congdon Boulevard, Duluth MN 55804 (TEL: 218-720-5510; FAX: 218-720-5539).

FEB 22 1994

OFFICE OF SCIENCE AND TECHNOLOGY POSITION STATEMENT

Section 131.11(b)(ii) of the water quality standards regulation (40 CFR Part 131) provides the regulatory mechanism for a State to develop site-specific criteria for use in water quality standards. Adopting site-specific criteria in water quality standards is a State option--not a requirement. The Environmental Protection Agency (EPA) in 1983 provided guidance on scientifically acceptable methods by which site-specific criteria could be developed.

The interim guidance provided in this document supersedes all guidance concerning water-effect ratios and the Indicator Species Procedure given in Chapter 4 of the *Water Quality Standards Handbook* issued by EPA in 1983 and in *Guidelines for Deriving Numerical Aquatic Site-Specific Water Quality Criteria by Modifying National Criteria, 1984*. Appendix B also supersedes the guidance in these earlier documents for the Recalculation Procedure for performing site-specific criteria modifications.

This interim guidance fulfills a commitment made in the final rule to establish numeric criteria for priority toxic pollutants (57 FR 60848, December 22, 1992, also known as the "National Toxics Rule"). This guidance also is applicable to pollutants other than metals with appropriate modifications, principally to chemical analyses.

Except for the jurisdictions subject to the aquatic life criteria in the national toxics rule, water-effect ratios are site-specific criteria subject to review and approval by the appropriate EPA Regional Administrator. Site-specific criteria are new or revised criteria subject to the normal EPA review requirements established in Clean Water Act § 303(c). For the States in the National Toxics Rule, EPA has established that site-specific water-effect ratios may be applied to the criteria promulgated in the rule to establish site-specific criteria. The water-effect ratio portion of these criteria would still be subject to State review before the development of total maximum daily loads, waste load allocations or translation into NPDES permit limits. EPA would only review these water-effect ratios during its oversight review of these State programs or review of State-issued permits.

Each of the three options for deriving a final water-effect ratio presented on page 36 of this interim guidance meets the scientific and technical acceptability test for deriving site-specific criteria specified in the water quality standards regulation (40 CFR 131.11(a)). Option 3 is the simplest, least restrictive and generally the least expensive approach for situations where simulated downstream water appropriately represents a "site." Option 3 requires experimental determination of three water-effect ratios with the primary test species that are determined during any season (as long as the downstream flow is between 2 and 10 times design flow conditions.) The final WER is generally (but not always) the lowest experimentally determined WER. Deriving a final water-effect ratio using option 3 with the use of simulated downstream water for a situation where this simulation appropriately represents a "site", is a fully acceptable approach for deriving a water-effect ratio for use in determining a site-specific criterion, although it will generally provide a lower water-effect ratio than the other 2 options.

As indicated in the introduction to this guidance, the determination of a water-effect ratio may require substantial resources. A discharger should consider cost-effective, preliminary measures described in this guidance (e.g., use of "clean" sampling and chemical analytical techniques or in non-NTR States, a recalculated criterion) to determine if an indicator species site-specific criterion is really needed. It may be that an appropriate site-specific criterion is actually being attained. In many instances, use of these other measures may eliminate the need for deriving final water-effect ratios. The methods described in this interim guidance should be sufficient to develop site-specific criteria that resolve concerns of dischargers when there appears to be no instream toxicity from a metal but, where (a) a discharge appears to exceed existing or proposed water quality-based permit limits, or (b) an instream concentration appears to exceed an existing or proposed water quality criterion.

This guidance describes 2 different methods for determining water-effect ratios. Method 1 has 3 options each of which may only require 3 sampling periods. However options 1 and 2 may be expanded and require a much greater effort. While this position statement has discussed the simplest, least expensive option for method 1 (the single discharge to a stream) to illustrate that site specific criteria are feasible even when only small dischargers are affected, water-effect ratios may be calculated using any of the other options described in the guidance if the State/discharger believe that there is reason to expect that a more accurate site-specific criterion will result from the increased cost and complexity inherent in conducting the additional tests and analyzing the results. Situations where this could be the case include, for example, where seasonal effects in receiving water quality or in discharge quality need to be assessed.

In addition, EPA will consider other scientifically defensible approaches in developing final water-effect ratios as authorized in 40 CFR 131.11. However, EPA strongly recommends that before a State/discharger implements any approach other than one described in this interim guidance, discussions be held with appropriate EPA regional offices and Office of Research and Development's scientists before actual testing begins. These discussions would be to ensure that time and resources are not wasted on scientifically and technically unacceptable approaches. It remains EPA's responsibility to make final decisions on the scientific and technical validity of alternative approaches to developing site-specific water quality criteria.

EPA is fully cognizant of the continuing debate between what constitutes guidance and what is a regulatory requirement. Developing site-specific criteria is a State regulatory option. Using the methodology correctly as described in this guidance assures the State that EPA will accept the result. Other approaches are possible and logically should be discussed with EPA prior to implementation.

The Office of Science and Technology believes that this interim guidance advances the science of determining site-specific criteria and provides policy guidance that States and EPA can use in this complex area. It reflects the scientific advances in the past 10 years and the experience gained from dealing with these issues in real world situations. This guidance will help improve implementation of water quality standards and be the basis for future progress.

Tudor T. Davies, Director
Office of Science And Technology
Office of Water

CONTENTS

APPENDICES

FIGURES

ACKNOWLEDGMENTS

This document was written by:

Charles E. Stephan, U.S. EPA, ORD, Environmental Research Laboratory, Duluth, MN.

William H. Peltier, U.S. EPA, Region IV, Environmental Services Division, Athens, GA.

David J. Hansen, U.S. EPA, ORD, Environmental Research Laboratory, Narragansett, RI.

Charles G. Delos, U.S. EPA, Office of Water, Health and Ecological Criteria Division, Washington, DC.

Gary A. Chapman, U.S. EPA, ORD, Environmental Research Laboratory (Narragansett), Pacific Ecosystems Branch, Newport, OR.

The authors thank all the people who participated in the open discussion of the experimental determination of water-effect ratios on Tuesday evening, January 26, 1993 in Annapolis, MD. Special thanks go to Herb Allen, Bill Beckwith, Ken Bruland, Lee Dunbar, Russ Erickson, and Carlton Hunt for their technical input on this project, although none of them necessarily agree with everything in this document. Comments by Kent Ballentine, Karen Gourdine, Mark Hicks, Suzanne Lussier, Nelson Thomas, Bob Spehar, Fritz Wagener, Robb Wood, and Phil Woods on various drafts, or portions of drafts, were also very helpful, as were discussions with several other individuals.

EXECUTIVE SUMMARY

A variety of physical and chemical characteristics of both the water and the metal can influence the toxicity of a metal to aquatic organisms in a surface water. When a site-specific aquatic life criterion is derived for a metal, an adjustment procedure based on the toxicological determination of a water-effect ratio (WER) may be used to account for a difference between the toxicity of the metal in laboratory dilution water and its toxicity in the water at the site. If there is a difference in toxicity and it is not taken into account, the aquatic life criterion for the body of water will be more or less protective than intended by EPA's Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses. After a WER is determined for a site, a site-specific aquatic life criterion can be calculated by multiplying an appropriate national, state, or recalculated criterion by the WER. Most WERs are expected to be equal to or greater than 1.0, but some might be less than 1.0. Because most aquatic life criteria consist of two numbers, i.e., a Criterion Maximum Concentration (CMC) and a Criterion Continuous Concentration (CCC), either a cmcWER or a cccWER or both might be needed for a site. The cmcWER and the cccWER cannot be assumed to be equal, but it is not always necessary to determine both.

In order to determine a WER, side-by-side toxicity tests are performed to measure the toxicity of the metal in two dilution waters. One of the waters has to be a water that would be acceptable for use in laboratory toxicity tests conducted for the derivation of national water quality criteria for aquatic life. In most situations, the second dilution water will be a simulated downstream water that is prepared by mixing upstream water and effluent in an appropriate ratio; in other situations, the second dilution water will be a sample of the actual site water to which the site-specific criterion is to apply. The WER is calculated by dividing the endpoint obtained in the site water by the endpoint obtained in the laboratory dilution water. A WER should be determined using a toxicity test whose endpoint is close to, but not lower than, the CMC and/or CCC that is to be adjusted.

A total recoverable WER can be determined if the metal in both of the side-by-side toxicity tests is analyzed using the total recoverable measurement, and a dissolved WER can be determined if the metal is analyzed in both tests using the dissolved measurement. Thus four WERs can be determined:

- Total recoverable cmcWER.
- Total recoverable cccWER.
- Dissolved cmcWER.
- Dissolved cccWER.

A total recoverable WER is used to calculate a total recoverable site-specific criterion from a total recoverable national, state, or recalculated aquatic life criterion, whereas a dissolved WER is used to calculate a dissolved site-specific criterion from a dissolved criterion. WERs are determined individually for each metal at each site; WERs cannot be extrapolated from one metal to another, one effluent to another, or one site water to another.

Because determining a WER requires substantial resources, the desirability of obtaining a WER should be carefully evaluated:
1. Determine whether use of "clean techniques" for collecting, handling, storing, preparing, and analyzing samples will eliminate the reason for considering determination of a WER, because existing data concerning concentrations of metals in effluents and surface waters might be erroneously high.
2. Evaluate the potential for reducing the discharge of the metal.
3. Investigate possible constraints on the permit limits, such as antibacksliding and antidegradation requirements and human health and wildlife criteria.
4. Consider use of the Recalculation Procedure.
5. Evaluate the cost-effectiveness of determining a WER.

If the determination of a WER is desirable, a detailed workplan for should be submitted to the appropriate regulatory authority (and possibly to the Water Management Division of the EPA Regional Office) for comment. After the workplan is completed, the initial phase should be implemented, the data should be evaluated, and the workplan should be revised if appropriate.

Two methods are used to determine WERs. Method 1, which is used to determine cccWERs that apply near plumes and to determine all cmcWERs, uses data concerning three or more distinctly separate sampling events. It is best if the sampling events occur during both low-flow and higher-flow periods. When sampling does not occur during both low and higher flows, the site-specific criterion is derived in a more conservative manner due to greater uncertainty. For each sampling event, a WER is determined using a selected toxicity test; for at least one of the sampling events, a confirmatory WER is determined using a different test.

Method 2, which is used to determine a cccWER for a large body of water outside the vicinities of plumes, requires substantial site-specific planning and more resources than Method 1. WERs are determined using samples of actual site water obtained at various times, locations, and depths to identify the range of WERs in the body of water. The WERs are used to determine how many site-specific CCCs should be derived for the body of water and what the one or more CCCs should be.

The guidance contained herein replaces previous agency guidance concerning (a) the determination of WERs for use in the derivation of site-specific aquatic life criteria for metals and (b) the Recalculation Procedure. This guidance is designed to apply to metals, but the principles apply to most pollutants.

ABBREVIATIONS

ACR: Acute-Chronic Ratio

CCC: Criterion Continuous Concentration

CMC: Criterion Maximum Concentration

CRM: Certified Reference Material

FAV: Final Acute Value

FCV: Final Chronic Value

FW: Freshwater

FWER: Final Water-Effect Ratio

GMAV: Genus Mean Acute Value

HCME: Highest Concentration of the Metal in the Effluent

MDR: Minimum Data Requirement

NTR: National Toxics Rule

QA/QC: Quality Assurance/Quality Control

SMAV: Species Mean Acute Value

SW: Saltwater

TDS: Total Dissolved Solids

TIE: Toxicity Identification Evaluation

TMDL: Total Maximum Daily Load

TOC: Total Organic Carbon

TRE: Toxicity Reduction Evaluation

TSD: Technical Support Document

TSS: Total Suspended Solids

WER: Water-Effect Ratio

WET: Whole Effluent Toxicity

WLA: Wasteload Allocation

GLOSSARY

Acute-chronic ratio - an appropriate measure of the acute toxicity of a material divided by an appropriate measure of the chronic toxicity of the same material under the same conditions.

Appropriate regulatory authority - Usually the State water pollution control agency, even for States under the National Toxics Rule; if, however, a State were to waive its section 401 authority, the Water Management Division of the EPA Regional Office would become the appropriate regulatory authority.

Clean techniques - a set of procedures designed to prevent contamination of samples so that concentrations of trace metals can be measured accurately and precisely.

Critical species - a species that is commercially or recreationally important at the site, a species that exists at the site and is listed as threatened or endangered under section 4 of the Endangered Species Act, or a species for which there is evidence that the loss of the species from the site is likely to cause an unacceptable impact on a commercially or recreationally important species, a threatened or endangered species, the abundances of a variety of other species, or the structure or function of the community.

Design flow - the flow used for steady-state wasteload allocation modeling.

Dissolved metal - defined here as "metal that passes through either a 0.45-μm or a 0.40-μm membrane filter".

Endpoint - the concentration of test material that is expected to cause a specified amount of adverse effect.

Final Water-Effect Ratio - the WER that is used in the calculation of a site-specific aquatic life criterion.

Flow-through test - a test in which test solutions flow into the test chambers either intermittently (every few minutes) or continuously and the excess flows out.

Labile metal - metal that is in water and will readily convert from one form to another when in a nonequilibrium condition.

Particulate metal - metal that is measured by the total recoverable method but not by the dissolved method.

Primary test - the toxicity test used in the determination of a Final Water-Effect Ratio (FWER); the specification of the test includes the test species, the life stage of the species, the duration of the test, and the adverse effect on which the endpoint is based.

Refractory metal - metal that is in water and will not readily convert from one form to another when in a nonequilibrium condition, i.e., metal that is in water and is not labile.

Renewal test - a test in which either the test solution in a test chamber is renewed at least once during the test or the test organisms are transferred into a new test solution of the same composition at least once during the test.

Secondary test - a toxicity test that is usually conducted along with the primary test only once to test the assumptions that, within experimental variation, (a) similar WERs will be obtained using tests that have similar sensitivities to the test material, and (b) tests that are less sensitive to the test material will usually give WERs that are closer to 1.

Simulated downstream water - a site water prepared by mixing effluent and upstream water in a known ratio.

Site-specific aquatic life criterion - a water quality criterion for aquatic life that has been derived to be specifically appropriate to the water quality characteristics and/or species composition at a particular location.

Site water - upstream water, actual downstream water, or simulated downstream water in which a toxicity test is conducted side-by-side with the same toxicity test in a laboratory dilution water to determine a WER.

Static test - a test in which the solution and organisms that are in a test chamber at the beginning of the test remain in the chamber until the end of the test.

Total recoverable metal - metal that is in aqueous solution after the sample is appropriately acidified and digested and insoluble material is separated.

Water-effect ratio - an appropriate measure of the toxicity of a material obtained in a site water divided by the same measure of the toxicity of the same material obtained simultaneously in a laboratory dilution water.

PREFACE

Several issues need consideration when guidance such as this is written:

1. Degrees of importance: Procedures and methods are series of instructions, but some of the instructions are more important than others. Some instructions are so important that, if they are not followed, the results will be questionable or unacceptable; other instructions are less important, but definitely desirable. Possibly the best way to express various degrees of importance is the approach described in several ASTM Standards, such as in section 3.6 of Standard E729 (ASTM 1993a), which is modified here to apply to WERs:

   The words "must", "should", "may", "can", and "might" have specific meanings in this document. "Must" is used to express an instruction that is to be followed, unless a site-specific consideration requires a deviation, and is used only in connection with instructions that directly relate to the validity of toxicity tests, WERs, FWERs, and the Recalculation Procedure. "Should" is used to state instructions that are recommended and are to be followed if reasonably possible. Deviation from one "should" will not invalidate a WER, but deviation from several probably will. Terms such as "is desirable", "is often desirable", and "might be desirable" are used in connection with less important instructions. "May" is used to mean "is (are) allowed to", "can" is used to mean "is (are) able to", and "might" is used to mean "could possibly". Thus the classic distinction between "may" and "can" is preserved, and "might" is not used as a synonym for either "may" or "can".

   This does not eliminate all problems concerning the degree of importance, however. For example, a small deviation from a "must" might not invalidate a WER, whereas a large deviation would. (Each "**must**" and "**must not**" is in bold print for convenience, not for emphasis, in this document.)

2. Educational and explanatory material: Many people have asked for much detail in this document to ensure that as many WERs as possible are determined in an acceptable manner. In addition, some people want justifications for each detail. Much of the detail that is desired by some people is based on "best professional judgment", which is rarely considered an acceptable justification by people who disagree with a specified detail. Even if details are taken from an EPA method or an ASTM standard, they were often included in those documents on the basis of best professional judgment. In contrast, some people want detailed methodology presented without explanatory material. It was decided to include as much detail as is feasible, and to provide rationale and explanation for major items.

3. Alternatives: When more than one alternative is both scientifically sound and appropriately protective, it seems reasonable to present the alternatives rather than presenting the one that is considered best. The reader can then select one based on cost-effectiveness, personal preference, details of the particular situation, and perceived advantages and disadvantages.

4. Separation of "science", "best professional judgment" and "regulatory decisions": These can never be completely separated in this kind of document; for example, if data are analyzed for a statistically significant difference, the selection of alpha is an important decision, but a rationale for its selection is rarely presented, probably because the selection is not a scientific decision. In this document, an attempt has been made to focus on good science, best professional judgment, and presentation of the rationale; when possible, these are separated from "regulatory decisions" concerning margin of safety, level of protection, beneficial use, regulatory convenience, and the goal of zero discharge. Some "regulatory decisions" relating to implementation, however, should be integrated with, not separated from, "science" because the two ought to be carefully considered together wherever science has implications for implementation.

5. Best professional judgment: Much of the guidance contained herein is qualitative rather than quantitative, and much judgment will usually be required to derive a site-specific water quality criterion for aquatic life. In addition, although this version of the guidance for determining and using WERs attempts to cover all major questions that have arisen during use of the previous version and during preparation of this version, it undoubtedly does not cover all situations, questions, and extenuating circumstances that might arise in the future. All necessary decisions should be based on both a thorough knowledge of aquatic toxicology and an understanding of this guidance; each decision should be consistent with the spirit of this guidance, which is to make best use of "good science" to derive the most appropriate site-specific criteria. This guidance should be modified whenever sound scientific evidence indicates that a site-specific criterion produced using this guidance will probably substantially underprotect or overprotect the aquatic life at the site of concern. Derivation of site-specific criteria for aquatic life is a complex process and requires knowledge in many areas of aquatic toxicology; any deviation from this guidance should be carefully considered to ensure that it is consistent with other parts of this guidance and with "good science".

6. Personal bias: Bias can never be eliminated, and some decisions are at the fine line between "bias" and "best

professional judgment". The possibility of bias can be eliminated only by adoption of an extreme position such as "no regulation" or "no discharge". One way to deal with bias is to have decisions made by a team of knowledgeable people.

7. Teamwork: The determination of a WER should be a cooperative team effort beginning with the completion of the initial workplan, interpretation of initial data, revision of the workplan, etc. The interaction of a variety of knowledgeable, reasonable people will help obtain the best results for the expenditure of the fewest resources. Members of the team should acknowledge their biases so that the team can make best use of the available information, taking into account its relevancy to the immediate situation and its quality.

INTRODUCTION

National aquatic life criteria for metals are intended to protect the aquatic life in almost all surface waters of the United States (U.S. EPA 1985). This level of protection is accomplished in two ways. First, the national dataset is required to contain aquatic species that have been found to be sensitive to a variety of pollutants. Second, the dilution water and the metal salt used in the toxicity tests are required to have physical and chemical characteristics that ensure that the metal is at least as toxic in the tests as it is in nearly all surface waters. For example, the dilution water is to be low in suspended solids and in organic carbon, and some forms of metal (e.g., insoluble metal and metal bound by organic complexing agents) cannot be used as the test material. (The term "metal" is used herein to include both "metals" and "metalloids".)

Alternatively, a national aquatic life criterion might not adequately protect the aquatic life at some sites. An untested species that is important at a site might be more sensitive than any of the tested species. Also, the metal might be more toxic in site water than in laboratory dilution water because, for example, the site water has a lower pH and/or hardness than most laboratory waters. Thus although a national aquatic life criterion is intended to be lower than necessary for most sites, a national criterion might not adequately protect the aquatic life at some sites.

Because a national aquatic life criterion might be more or less protective than intended for the aquatic life in most bodies of water, the U.S. EPA provided guidance (U.S. EPA 1983a,1984) concerning three procedures that may be used to derive a site-specific criterion:

1. The Recalculation Procedure is intended to take into account relevant differences between the sensitivities of the aquatic organisms in the national dataset and the sensitivities of organisms that occur at the site.
2. The Indicator Species Procedure provides for the use of a water-effect ratio (WER) that is intended to take into account relevant differences between the toxicity of the metal in laboratory dilution water and in site water.
3. The Resident Species Procedure is intended to take into account both kinds of differences simultaneously.

A site-specific criterion is intended to come closer than the national criterion to providing the intended level of protection to the aquatic life at the site, usually by taking into account the biological and/or chemical conditions (i.e., the species composition and/or water quality characteristics) at the site. The fact that the U.S. EPA has made these procedures available should not be interpreted as implying that the agency advocates that states derive site-specific criteria before setting state standards. Also, derivation of a site-specific criterion does not change the intended level of protection of the aquatic life at the site. Because a WER is expected to appropriately take into account (a) the site-specific toxicity of the metal, and (b) synergism, antagonism, and additivity with other constituents of the site water, using a WER is more likely to provide the intended level of protection than not using a WER.

Although guidance concerning site-specific criteria has been available since 1983 (U.S. EPA 1983a,1984), interest has increased in recent years as states have devoted more attention to chemical-specific water quality criteria for aquatic life. In addition, interest in water-effect ratios (WERs) increased when the "Interim Guidance" concerning metals (U.S. EPA 1992) made a fundamental change in the way that WERs are experimentally determined (see Appendix A), because the change is expected to substantially increase the magnitude of many WERs. Interest was further focused on WERs when they were integrated into some of the aquatic life criteria for metals that were promulgated by the National Toxics Rule (57 FR 60848, December 22, 1992). The newest guidance issued by the U.S. EPA (Prothro 1993) concerning aquatic life criteria for metals affected the determination and use of WERs only insofar as it affected the use of total recoverable and dissolved criteria.

The early guidance concerning WERs (U.S. EPA 1983a,1984) contained few details and needs revision, especially to take into account newer guidance concerning metals (U.S. EPA 1992; Prothro 1993). The guidance presented herein supersedes all guidance concerning WERs and the Indicator Species Procedure given in Chapter 4 of the Water Quality Standards Handbook (U.S. EPA 1983a) and in U.S. EPA (1984). All guidance presented in U.S. EPA (1992) is superseded by that presented by Prothro (1993) and by this document. Metals are specifically addressed herein because of the National Toxics Rule (NTR) and because of current interest in aquatic life criteria for metals; although most of this guidance also applies to other pollutants, some obviously applies only to metals.

Even though this document was prepared mainly because of the NTR, the guidance contained herein concerning WERs is likely to have impact beyond its use with the NTR. Therefore, it is appropriate to also present new guidance concerning the Recalculation Procedure (see Appendix B) because the previous guidance (U.S. EPA 1983a,1984) concerning this procedure also contained few details and needs revision. The NTR does not allow use of the Recalculation Procedure in jurisdictions subject to the NTR.

The previous guidance concerning site-specific procedures did not allow the Recalculation Procedure and the WER procedure to be used together in the derivation of a site-specific aquatic life criterion; the only way to take into account both species composition and water quality characteristics in the determination of a site-specific criterion was to use the Resident Species Procedure. A specific change contained herein is that, except in jurisdictions that are subject to the NTR, the Recalculation Procedure and the WER Procedure may now be used together. Additional reasons for addressing both the Recalculation Procedure and the WER Procedure in this document are that both procedures are based directly on the guidelines for deriving national aquatic life criteria (U.S. EPA 1985) and, when the two are used together, use of the Recalculation Procedure has specific implications concerning the determination of the WER.

This guidance is intended to produce WERs that may be used to derive site-specific aquatic life criteria for metals from most national and state aquatic life criteria that were derived from laboratory toxicity data. Except in jurisdictions that are subject to the NTR, the WERs may also be used with site-specific aquatic life criteria that are derived for metals using the Recalculation Procedure described in Appendix B. WERs obtained using the methods described herein should not be used to adjust aquatic life criteria that were derived for metals in other ways. For example, because they are designed to be applied to criteria derived on the basis of laboratory toxicity tests, WERs determined using the methods described herein cannot be used to adjust the residue-based mercury Criterion Continuous Concentration (CCC) or the field-based selenium freshwater criterion. For the purposes of the NTR, WERs may be used with the aquatic life criteria for arsenic, cadmium, chromium(III), chromium(VI), copper, lead, nickel, silver, and zinc and with the Criterion Maximum Concentration (CMC) for mercury. WERs may also be used with saltwater criteria for selenium.

The concept of a WER is rather simple:

> Two side-by-side toxicity tests are conducted - one test using laboratory dilution water and the other using site water. The endpoint obtained using site water is divided by the endpoint obtained using laboratory dilution water. The quotient is the WER, which is multiplied times the national, state, or recalculated aquatic life criterion to calculate the site-specific criterion.

Although the concept is simple, the determination and use of WERs involves many considerations.

The primary purposes of this document are to:

1. Identify steps that should be taken before the determination of a WER is begun.
2. Describe the methods recommended by the U.S. EPA for the determination of WERs.
3. Address some issues concerning the use of WERs.
4. Present new guidance concerning the Recalculation Procedure.

Before Determining a WER

Because a national criterion is intended to protect aquatic life in almost all bodies of water and because a WER is intended to account for a difference between the toxicity of a metal in a laboratory dilution water and its toxicity in a site water, dischargers who want higher permit limits than those derived on the basis of an existing aquatic life criterion will probably consider determining a WER. Use of a WER should be considered only as a last resort for at least three reasons:

a. Even though some WERs will be substantially greater than 1.0, some will be about 1.0 and some will be less than 1.0.
b. The determination of a WER requires substantial resources.
c. There are other things that a discharger can do that might be more cost-effective than determining a WER.

The two situations in which the determination of a WER might appear attractive to dischargers are when (a) a discharge appears to exceed existing or proposed water quality-based permit limits, and (b) an instream concentration appears to exceed an existing or proposed aquatic life criterion. Such situations result from measurement of the concentration of a metal in an effluent or a surface water. It would therefore seem reasonable to ensure that such measurements were not subject to contamination. Usually it is much easier to verify chemical measurements by using "clean techniques" for collecting, handling, storing, preparing, and analyzing samples, than to determine a WER. Clean techniques and some related QA/QC considerations are discussed in Appendix C.

In addition to investigating the use of "clean techniques", other steps that a discharger should take prior to beginning the experimental determination of a WER include:

1. Evaluate the potential for reducing the discharge of the metal.
2. Investigate such possible constraints on permit limits as antibacksliding and antidegradation requirements and human health and wildlife criteria.
3. Obtain assistance from an aquatic toxicologist who understands the basics of WERs (see Appendix D), the U.S. EPA's national aquatic life guidelines (U.S. EPA 1985), the guidance presented by Prothro (1993), the national criteria document for the metal(s) of concern (see Appendix E), the procedures described by the U.S. EPA (1993a,b,c) for acute and chronic toxicity tests on effluents and surface waters, and the procedures described by ASTM (1993a,b,c,d,e) for acute and chronic toxicity tests in laboratory dilution water.
4. Develop an initial definition of the site to which the site-specific criterion is to apply.
5. Consider use of the Recalculation Procedure (see Appendix B).
6. Evaluate the cost-effectiveness of the determination of a WER. Comparative toxicity tests provide the most useful data, but chemical analysis of the downstream water might be helpful

because the following are often true for some metals:

a. The lower the percent of the total recoverable metal in the downstream water that is dissolved, the higher the WER.
b. The higher the concentration of total organic carbon (TOC) and/or total suspended solids (TSS), the higher the WER.

It is also true that the higher the concentration of nontoxic dissolved metal, the higher the WER. Although some chemical analyses might provide useful information concerning the toxicities of some metals in water, at the present only toxicity tests can accurately reflect the toxicities of different forms of a metal (see Appendix D).

7. Submit a workplan for the experimental determination of the WER to the appropriate regulatory authority (and possibly to the Water Management Division of the EPA Regional Office) for comment. The workplan should include detailed descriptions of the site; existing criterion and standard; design flows; site water; effluent; sampling plan; procedures that will be used for collecting, handling, and analyzing samples of site water and effluent; primary and secondary toxicity tests; quality assurance/quality control (QA/QC) procedures; Standard Operating Procedures (SOPs); and data interpretation.

After the workplan is completed, the initial phase should be implemented; then the data obtained should be evaluated, and the workplan should be revised if appropriate. Developing and modifying the workplan and analyzing and interpreting the data should be a cooperative effort by a team of knowledgeable people.

Two Kinds of WERs

Most aquatic life criteria contain both a CMC and a CCC, and it is usually possible to determine both a cmcWER and a cccWER. The two WERs cannot be assumed to be equal because the magnitude of a WER will probably depend on the sensitivity of the toxicity test used and on the percent effluent in the site water (see Appendix D), both of which can depend on which WER is to be determined. In some cases, it is expected that a larger WER can be applied to the CCC than to the CMC, and so it would be environmentally conservative to apply cmcWERs to CCCs. In such cases it is possible to determine a cmcWER and apply it to both the CMC and the CCC in order to derive a site-specific CMC, a site-specific CCC, and new permit limits. If these new permit limits are controlled by the new site-specific CCC, a cccWER could be determined using a more sensitive test, possibly raising the site-specific CCC and the permit limits again. A cccWER may, of course, be determined whenever desired. Unless the experimental variation is increased, use of a cccWER will usually improve the accuracy of the resulting site-specific CCC.

In some cases, a larger WER cannot be applied to the CCC than to the CMC and so it might not be environmentally conservative to apply a cmcWER to a CCC (see section A.4 of Method 1).

Steady-state and Dynamic Models

Some of the guidance contained herein specifically applies to situations in which the permit limits were calculated using steady-state modeling; in particular, some samples are to be obtained when the actual stream flow is close to the design flow. If permit limits were calculated using dynamic modeling, the guidance will have to be modified, but it is unclear at present what modifications are most appropriate. For example, it might be useful to determine whether the magnitude of the WER is related to the flow of the upstream water and/or the effluent.

Two Methods

Two methods are used to determine WERs. Method 1 will probably be used to determine all cmcWERs and most cccWERs because it can be applied to situations that are in the vicinities of plumes. Because WERs are likely to depend on the concentration of effluent in the water and because the percent effluent in a water sample obtained in the immediate vicinity of a plume is unknown, simulated downstream water is used so that the percent effluent in the sample is known. For example, if a sample that was supposed to represent a complete-mix situation was accidently taken in the plume upstream of complete mix, the sample would probably have a higher percent effluent and a higher WER than a sample taken downstream of complete mix; use of the higher WER to derive a site-specific criterion for the complete-mix situation would result in underprotection. If the sample were accidently taken upstream of complete mix but outside the plume, overprotection would probably result.

Method 1 will probably be used to determine all cmcWERs and most cccWERs in flowing fresh waters, such as rivers and streams. Method 1 is intended to apply not only to ordinary rivers and streams but also to streams that some people might consider extraordinary, such as streams whose design flows are zero and streams that some state and/or federal agencies refer to as "effluent-dependent", "habitat-creating", or "effluent-dominated". Method 1 is also used to determine cmcWERs in such large sites as oceans and large lakes, reservoirs, and estuaries (see Appendix F).

Method 2 is used to determine WERs that apply outside the area of plumes in large bodies of water. Such WERs will be cccWERs and will be determined using samples of actual site water obtained at various times, locations, and depths in order to identify the range of WERs that apply to the body of water. These experimentally determined WERs are then used to decide how many site-specific criteria should be derived for the body of water and what the criterion (or criteria) should be. Method 2 requires substantially more resources than Method 1.

The complexity of each method increases when the number of metals and/or the number of discharges is two or more:

a. The simplest situation is when a WER is to be determined for only one metal and only one discharge has permit limits for that metal. (This is the single-metal single-discharge situation.)
b. A more complex situation is when a WER is to be determined for only one metal, but more than one discharge has permit limits for that metal. (This is the single-metal multiple-discharge situation.)
c. An even more complex situation is when WERs are to be determined for more than one metal, but only one discharge has permit limits for any of the metals. (This is the multiple-metal single-discharge situation.)
d. The most complex situation is when WERs are to be determined for more than one metal and more than one discharge has permit limits for some or all of the metals. (This is the multiple-metal multiple-discharge situation.)

WERs need to be determined for each metal at each site because extrapolation of a WER from one metal to another, one effluent to another, or one surface water to another is too uncertain.

Both methods work well in multiple-metal situations, but special tests or additional tests will be necessary to show that the resulting combination of site-specific criteria will not be too toxic. Method 2 is better suited to multiple-discharge situations than is Method 1. Appendix F provides additional guidance concerning multiple-metal and multiple-discharge situations, but it does not discuss allocation of waste loads, which is performed when a wasteload allocation (WLA) or a total maximum daily load (TMDL) is developed (U.S. EPA 1991a).

Two Analytical Measurements

A total recoverable WER can be determined if the metal in both of the side-by-side toxicity tests is analyzed using the total recoverable measurement; similarly, a dissolved WER can be determined if the metal in both tests is analyzed using the dissolved measurement. A total recoverable WER is used to calculate a total recoverable site-specific criterion from an aquatic life criterion that is expressed using the total recoverable measurement, whereas a dissolved WER is used to calculate a dissolved site-specific criterion from a criterion that is expressed in terms of the dissolved measurement. Figure 1 illustrates the relationships between total recoverable and dissolved criteria, WERs, and the Recalculation Procedure.

Both Method 1 and Method 2 can be used to determine a total recoverable WER and/or a dissolved WER. The only difference in the experimental procedure is whether the WER is based on measurements of total recoverable metal or dissolved metal in the test solutions. Both total recoverable and dissolved measurements are to be performed for all tests to help judge the quality of the tests, to provide a check on the analytical chemistry, and to help understand the results; performing both measurements also increases the alternatives available for use of the results. For example, a dissolved WER that is not useful with a total recoverable criterion might be useful in the future if a dissolved criterion becomes available. Also, as explained in Appendix D, except for experimental variation, use of a total recoverable WER with a total recoverable criterion should produce the same total recoverable permit limits as use of a dissolved WER with a dissolved criterion; the internal consistency of the approaches and the data can be evaluated if both total recoverable and dissolved criteria and WERs are determined. It is expected that in many situations total recoverable WERs will be larger and more variable than dissolved WERs.

The Quality of the Toxicity Tests

Traditionally, for practical reasons, the requirements concerning such aspects as acclimation of test organisms to test temperature and dilution water have not been as stringent for toxicity tests on surface waters and effluents as for tests using laboratory dilution water. Because a WER is a ratio calculated from the results of side-by-side tests, it might seem that acclimation is not important for a WER as long as the organisms and conditions are identical in the two tests. Because WERs are used to adjust aquatic life criteria that are derived from results of laboratory tests, the tests conducted in laboratory dilution water for the determination of WERs should be conducted in the same way as the laboratory toxicity tests used in the derivation of aquatic life criteria. In the WER process, the tests in laboratory dilution water provide the vital link between national criteria and site-specific criteria, and so it is important to compare at least some results obtained in the laboratory dilution water with results obtained in at least one other laboratory.

Three important principles for making decisions concerning the methodology for the side-by-side tests are:

1. The tests using laboratory dilution water should be conducted so that the results would be acceptable for use in the derivation of national criteria.
2. As much as is feasible, the tests using site water should be conducted using the same procedures as the tests using the laboratory dilution water.
3. All tests should follow any special requirements that are necessary because the results are to be used to calculate a WER. Some such special requirements are imposed because the criterion for a rather complex situation is being changed based on few data, so more assurance is required that the data are high quality.

The most important special requirement is that the concentrations of the metal are to be measured using both the total recoverable and dissolved methods in all toxicity tests used for the determination of a WER. This requirement is necessary because half of the tests conducted for the determination of WERs use a site water in which the concentration of metal probably is not negligible. Because it is likely that the concentration of metal in the laboratory dilution water is negligible, assuming that the concentration in both waters is negligible and basing WERs on the amount of metal added would produce an unnecessarily low value for the WER. In addition, WERs are based on too few data to assume that nominal concentrations are accurate. Nominal concentrations obviously cannot be used if a dissolved WER is to be determined. Measured dissolved concentrations at the beginning and end of the test are used to judge the acceptability of the test, and it is certainly reasonable to measure the total recoverable concentration when the dissolved concentration is measured. Further, measuring the concentrations might lead to an interpretation of the results that allows a substantially better use of the WERs.

Conditions for Determining a WER

The appropriate regulatory authority might recommend that one or more conditions be met when a WER is determined in order to reduce the possibility of having to determine a new WER later:

1. Requirements that are in the existing permit concerning WET testing, Toxicity Identification Evaluation (TIE), and/or Toxicity Reduction Evaluation (TRE) (U.S. EPA 1991a).
2. Implementation of pollution prevention efforts, such as pretreatment, waste minimization, and source reduction.
3. A demonstration that applicable technology-based requirements are being met.

<u>If one or more of these is not satisfied when the WER is determined and is implemented later, it is likely that a new WER will have to be determined because of the possibility of a change in the composition of the effluent.</u>

Even if all recommended conditions are satisfied, determination of a WER might not be possible if the effluent, upstream water, and/or downstream water are toxic to the test organisms. In some such cases, it might be possible to determine a WER, but remediation of the toxicity is likely to be required anyway. <u>It is unlikely that a WER determined before remediation would be considered acceptable for use after remediation.</u> If it is desired to determine a WER before remediation and the toxicity is in the upstream water, it might be possible to use a laboratory dilution water or a water from a clean tributary in place of the upstream water; if a substitute water is used, its water quality characteristics should be similar to those of the upstream water (i.e., the pH should be within 0.2 pH units and the hardness, alkalinity, and concentrations of TSS and TOC should be within 10 % or 5 mg/L, whichever is greater, of those in the upstream water). If the upstream water is chronically toxic, but not acutely toxic, it might be possible to determine a cmcWER even if a cccWER cannot be determined; a cmcWER might not be useful, however, if the permit limits are controlled by the CCC; in such a case, it would probably not be acceptable to assume that the cmcWER is an environmentally conservative estimate of the cccWER. If the WER is determined using downstream water and the toxicity is due to the effluent, tests at lower concentrations of the effluent might give an indication of the amount of remediation needed.

Conditions for Using a WER

Besides requiring that the WER be valid, the appropriate regulatory authority might consider imposing other conditions for the approval of a site-specific criterion based on the WER:

1. 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 using downstream water are influenced more by the effluent than 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 metal, hardness, alkalinity, pH, suspended solids, organic carbon, or other toxic materials. A special concern is the possibility of a shift from discharge of nontoxic metal to discharge of toxic metal such that the concentration of the metal does not increase; analytical chemistry might not detect the change but toxicity tests would.

   Even if no changes are known to have occurred, WERs should be reevaluated periodically. (The NTR recommends that NPDES permits include periodic determinations of WERs in 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 there should be a reasonably long period of time during which the reevaluation can be performed. Periodic determination of WERs should be designed to answer questions, not just generate data.
2. Increased chemical monitoring of the upstream water, effluent, and/or downstream water, as appropriate, for water quality characteristics that probably affect the toxicity of the metal

(e.g., hardness, alkalinity, pH, TOC, and TSS) to determine whether conditions change. The conditions at the times the samples were obtained should be kept 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.

3. Periodic reevaluation of the environmental fate of the metal in the effluent (see Appendix A).
4. WET testing.
5. Instream bioassessments.

Decisions concerning the possible imposition of such conditions should take into account:

a. The ratio of the new and old criteria. The greater the increase in the criterion, the more concern there should be about (1) the fate of any nontoxic metal that contributes to the WER and (2) changes in water quality that might occur within 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. The significance of the magnitude of the ratio can be judged by comparison with the acute-chronic ratio, the factor of two that is the ratio of the FAV to the CMC, and the range of sensitivities of species in the criteria document for the metal (see Appendix E).
b. The size of the site.
c. The size of the discharge.
d. The rate of downstream dilution.
e. Whether the CMC or the CCC controls the permit limits.

When WERs are determined using upstream water, conditions on the use of a WER are more likely when the water contains an effluent that increases the WER by adding TOC and/or TSS, because the WER will be larger and any decrease in the discharge of such TOC and/or TSS might decrease the WER and result in underprotection. A WER determined using downstream water is likely to be larger and quite dependent on the composition of the effluent; there should be concern about whether a change in the effluent might result in underprotection at some time in the future.

Implementation Considerations

In some situations a discharger might not want to or might not be allowed to raise a criterion as much as could be justified by a WER:

1. The maximum possible increase is not needed and raising the criterion more than needed might greatly raise the cost if a greater increase would require more tests and/or increase the conditions imposed on approval of the site-specific criterion.
2. Such other constraints as antibacksliding or antidegradation requirements or human health or wildlife criteria might limit the amount of increase regardless of the magnitude of the WER.

3. The permit limits might be limited by an aquatic life criterion that applies outside the site. It is EPA policy that permit limits cannot be so high that they inadequately protect a portion of the same or a different body of water that is outside the site; nothing contained herein changes this policy in any way.

If no increase in the existing discharge is allowed, the only use of a WER will be to determine whether an existing discharge needs to be reduced. Thus a major use of WERs might be where technology-based controls allow concentrations in surface waters to exceed national, state, or recalculated aquatic life criteria. In this case, it might only be necessary to determine that the WER is greater than a particular value; it might not be necessary to quantify the WER. When possible, it might be desirable to show that the maximum WER is greater than the WER that will be used in order to demonstrate that a margin of safety exists, but again it might not be necessary to quantify the maximum WER.

In jurisdictions not subject to the NTR, WERs should be used to derive site-specific criteria, not just to calculate permit limits, because data obtained from ambient monitoring should be interpreted by comparison with ambient criteria. (This is not a problem in jurisdictions subject to the NTR because the NTR defines the ambient criterion as "WER x the EPA criterion".) If a WER is used to adjust permit limits without adjusting the criterion, the permit limits would allow the criterion to be exceeded. Thus the WER should be used to calculate a site-specific criterion, which should then be used to calculate permit limits. In some states, site-specific criteria can only be adopted as revised criteria in a separate, independent water quality standards review process. In other states, site-specific criteria can be developed in conjunction with the NPDES permitting process, as long as the adoption of a site-specific criterion satisfies the pertinent water quality standards procedural requirements (i.e., a public notice and a public hearing). In either case, site-specific criteria are to be adopted prior to NPDES permit issuance. Moreover, the EPA Regional Administrator has authority to approve or disapprove all new and revised site-specific criteria and to review NPDES permits to verify compliance with the applicable water quality criteria.

Other aspects of the use of WERs in connection with permit limits, WLAs, and TMDLs are outside the scope of this document. The Technical Support Document (U.S. EPA 1991a) and Prothro (1993) provide more information concerning implementation procedures. Nothing contained herein should be interpreted as changing the three-part approach that EPA uses to protect aquatic life: (1) numeric chemical-specific water quality criteria for individual pollutants, (2) whole effluent toxicity (WET) testing, and (3) instream bioassessments.

Even though there are similarities between WET testing and the determination of WERs, there are important differences. For example, WERs can be used to derive site-specific criteria for individual pollutants, but WET testing cannot. The difference between WET testing and the determination of WERs is less when the toxicity tests used in the determination of the WER are ones that are used in WET testing. If a WER is used to make a large change in a criterion, additional WET testing and/or instream bioassessments are likely to be recommended.

The Sample-Specific WER Approach

A major problem with the determination and use of aquatic life criteria for metals is that no analytical measurement or combination of measurements has yet been shown to explain the toxicity of a metal to aquatic plants, invertebrates, amphibians, and fishes over the relevant range of conditions in surface waters (see Appendix D). It is not just that insufficient data exist to justify a relationship; rather, existing data possibly contradict some ideas that could possibly be very useful if true. For example, the concentration of free metal ion could possibly be a useful basis for expressing water quality criteria for metals if it could be feasible and could be used in a way that does not result in widespread underprotection of aquatic life. Some available data, however, might contradict the idea that the toxicity of copper to aquatic organisms is proportional to the concentration or the activity of the cupric ion. Evaluating the usefulness of any approach based on metal speciation is difficult until it is known how many of the species of the metal are toxic, what the relative toxicities are, whether they are additive (if more than one is toxic), and the quantitative effects of the factors that have major impacts on the bioavailability and/or toxicity of the toxic species. Just as it is not easy to find a useful quantitative relationship between the analytical chemistry of metals and the toxicity of metals to aquatic life, it is also not easy to find a qualitative relationship that can be used to provide adequate protection for the aquatic life in almost all bodies of water without providing as much overprotection for some bodies of water as results from use of the total recoverable and dissolved measurements.

The U.S. EPA cannot ignore the existence of pollution problems and delay setting aquatic life criteria until all scientific issues have been adequately resolved. In light of uncertainty, the agency needs to derive criteria that are environmentally conservative in most bodies of water. Because of uncertainty concerning the relationship between the analytical chemistry and the toxicity of metals, aquatic life criteria for metals are expressed in terms of analytical measurements that result in the criteria providing more protection than necessary for the aquatic life in most bodies of water. The agency has provided for the use of WERs to address the general conservatism, but expects that some WERs will be less than 1.0 because national, state, and recalculated criteria are not necessarily environmentally conservative for all bodies of water.

It has become obvious, however, that the determination and use of WERs is not a simple solution to the existing general conservatism. It is likely that a permanent solution will have to be based on an adequate quantitative explanation of how metals and aquatic organisms interact. In the meantime, the use of total recoverable and dissolved measurements to express criteria and the use of site-specific criteria are intended to provide adequate protection for almost all bodies of water without excessive overprotection for too many bodies of water. Work needs to continue on the permanent solution and, just in case, on improved alternative approaches.

Use of WERs to derive site-specific criteria is intended to allow a reduction or elimination of the general overprotection associated with application of a national criterion to individual bodies of water, but a major problem is that a WER will rarely be constant over time, location, and depth in a body of water due to plumes, mixing, and resuspension. It is possible that dissolved concentrations and WERs will be less variable than total recoverable ones. It might also be possible to reduce the impact of the heterogeneity if WERs are additive across time, location, and depth (see Appendix G). Regardless of what approaches, tools, hypotheses, and assumptions are utilized, variation will exist and WERs will have to be used in a conservative manner. Because of variation between bodies of water, national criteria are derived to be environmentally conservative for most bodies of water, whereas the WER procedure, which is intended to reduce the general conservatism of national criteria, has to be conservative because of variation among WERs within a body of water.

The conservatism introduced by variation among WERs is due not to the concept of WERs, but to the way they are used. The reason that national criteria are conservative in the first place is the uncertainty concerning the linkage of analytical chemistry and toxicity; the toxicity of solutions can be measured, but toxicity cannot be modelled adequately using available chemical measurements. Similarly, the current way that WERs are used depends on a linkage between analytical chemistry and toxicity because WERs are used to derive site-specific criteria that are expressed in terms of chemical measurements.

Without changing the amount or kind of toxicity testing that is performed when WERs are determined using Method 2, a different way of using the WERs could avoid some of the problems introduced by the dependence on analytical chemistry. The "sample-specific WER approach" could consist of sampling a body of water at a number of locations, determining the WER for each sample, and measuring the concentration of the metal in each sample. Then for each individual sample, a quotient would be calculated by dividing the concentration of metal in the sample by the product of the national criterion times the WER obtained for that sample. Except for experimental variation, when the quotient for a sample is less than 1, the concentration of metal in that sample is acceptable; when the quotient for a sample is greater than 1, the concentration of metal in that sample is too high. As a check, both the total recoverable measurement and the dissolved measurement should be used because they should provide the same answer if everything is done correctly and accurately. This approach can also be used whenever Method 1 is used; although Method 1 is used with simulated downstream water, the sample-specific WER approach can be used with either simulated downstream water or actual downstream water.

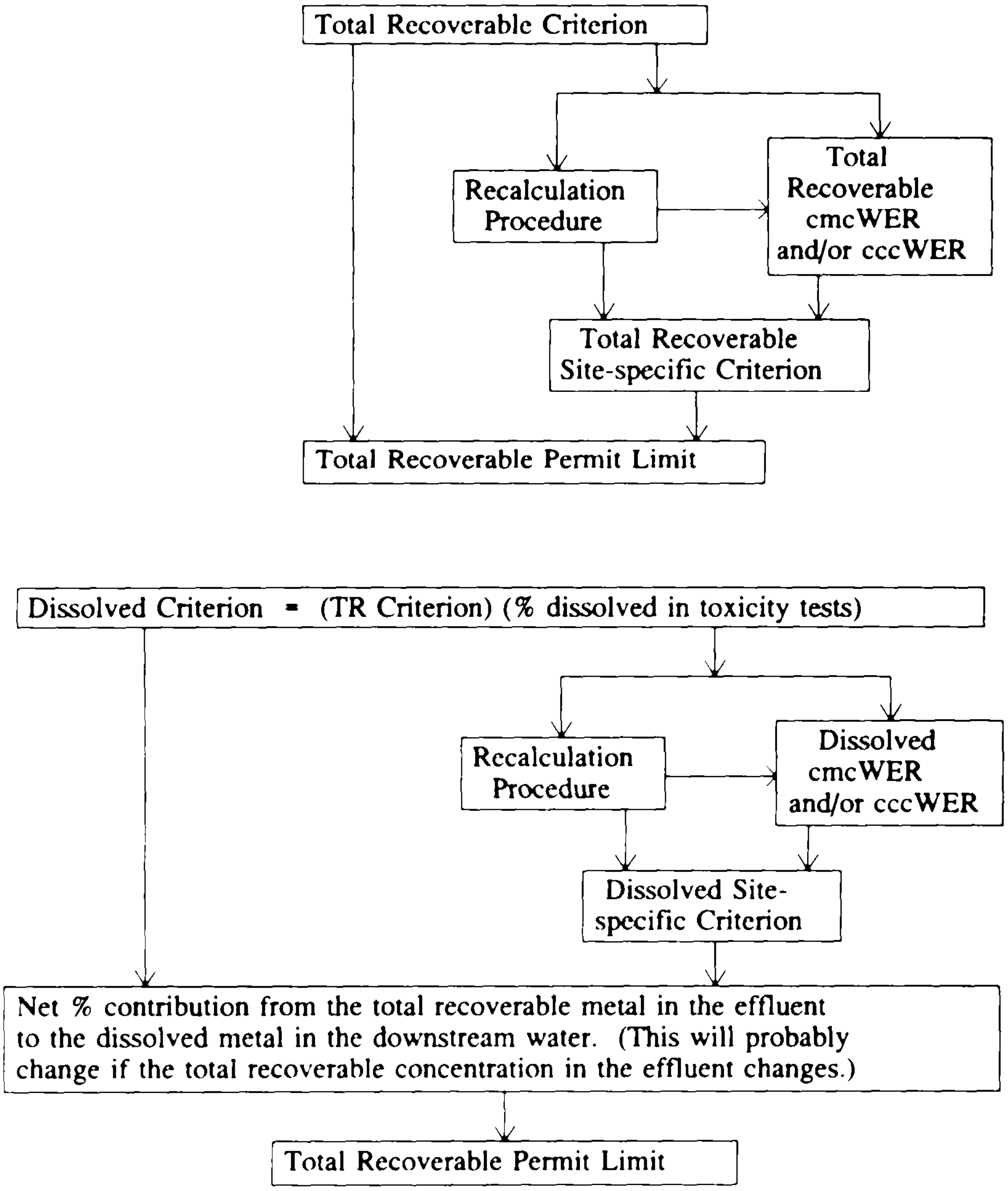
This sample-specific WER approach has several interesting features:

1. It is not a different way of determining WERs; it is merely a different way of using the WERs that are determined.
2. Variation among WERs within a body of water is not a problem.
3. It eliminates problems concerning the unknown relationship between toxicity and analytical chemistry.
4. It works equally well in areas that are in or near plumes and in areas that are away from plumes.
5. It works equally well in single-discharge and multiple-discharge situations.
6. It automatically accounts for synergism, antagonism, and additivity between toxicants.

This way of using WERs is equivalent to expressing the national criterion for a pollutant in terms of toxicity tests whose endpoints equal the CMC and the CCC; if the site water causes less adverse effect than is defined to be the endpoint, the concentration of that pollutant in the site water does not exceed the national criterion. This sample-specific WER approach does not directly fit into the current framework wherein criteria are derived and then permit limits are calculated from the criteria.

If the sample-specific WER approach were to produce a number of quotients that are greater than 1, it would seem that the concentration of metal in the discharge(s) should be reduced enough that the quotient is not greater than 1. Although this might sound straightforward, the discharger(s) would find that a substantial reduction in the discharge of a metal would not achieve the intended result if the reduction was due to removal of nontoxic metal. A chemical monitoring approach that cannot differentiate between toxic and nontoxic metal would not detect that only nontoxic metal had been removed, but the sample-specific WER approach would.

Figure 1: Four Ways to Derive a Permit Limit



For both the total recoverable and dissolved measurements, derivation of an optional site-specific criterion is described on the right. If both the Recalculation Procedure and the WER procedure are used, the Recalculation Procedure must be performed first. (The Recalculation Procedure cannot be used in jurisdictions that are subject to the National Toxics Rule.)

METHOD 1: DETERMINING WERs FOR AREAS IN OR NEAR PLUMES

Method 1 is based on the determination of WERs using simulated downstream water and so it can be used to determine a WER that applies in the vicinity of a plume. Use of simulated downstream water ensures that the concentration of effluent in the site water is known, which is important because the magnitude of the WER will often depend on the concentration of effluent in the downstream water. Knowing the concentration of effluent makes it possible to quantitatively relate the WER to the effluent. Method 1 can be used to determine either cmcWERs or cccWERs or both in single-metal, flowing freshwater situations, including streams whose design flow is zero and "effluent-dependent" streams (see Appendix F). As is also explained in Appendix F, Method 1 is used when cmcWERs are determined for "large sites", although Method 2 is used when cccWERs are determined for "large sites". In addition, Appendix F addresses special considerations regarding multiple-metal and/or multiple-discharge situations.

Neither Method 1 nor Method 2 covers all important methodological details for conducting the side-by-side toxicity tests that are necessary in order to determine a WER. Many references are made to information published by the U.S. EPA (1993a,b,c) concerning toxicity tests on effluents and surface waters and by ASTM (1993a,b,c,d,e,f) concerning tests in laboratory dilution water. Method 1 addresses aspects of toxicity tests that (a) need special attention when determining WERs and/or (b) are usually different for tests conducted on effluents and tests conducted in laboratory dilution water. Appendix H provides additional information concerning toxicity tests with saltwater species.

A. Experimental Design

Because of the variety of considerations that have important implications for the determination of a WER, decisions concerning experimental design should be given careful attention and need to answer the following questions:

1. Should WERs be determined using upstream water, actual downstream water, and/or simulated downstream water?
2. Should WERs be determined when the stream flow is equal to, higher than, and/or lower than the design flow?
3. Which toxicity tests should be used?
4. Should a cmcWER or a cccWER or both be determined?
5. How should a FWER be derived?
6. For metals whose criteria are hardness-dependent, at what hardness should WERs be determined?

The answers to these questions should be based on the reason that WERs are determined, but the decisions should also take into account some practical considerations.

1. Should WERs be determined using upstream water, actual downstream water, and/or simulated downstream water?

   a. Upstream water provides the least complicated way of determining and using WERs because plumes, mixing zones, and effluent variability do not have to be taken into account. Use of upstream water provides the least useful WERs because it does not take into account the presence of the effluent, which is the source of the metal. It is easy to assume that upstream water will give smaller WERs than downstream water, but in some cases downstream water might give smaller WERs (see Appendix G). Regardless of whether upstream water gives smaller or larger WERs, a WER should be determined using the water to which the site-specific criterion is to apply (see Appendix A).

   b. Actual downstream water might seem to be the most pertinent water to use when WERs are determined, but whether this is true depends on what use is to be made of the WERs. WERs determined using actual downstream water can be quantitatively interpreted using the sample-specific WER approach described at the end of the Introduction. If, however, it is desired to understand the quantitative implications of a WER for an effluent of concern, use of actual downstream water is problematic because the concentration of effluent in the water can only be known approximately.

   Sampling actual downstream water in areas that are in or near plumes is especially difficult. The WER obtained is likely to depend on where the sample is taken because the WER will probably depend on the percent effluent in the sample (see Appendix D). The sample could be taken at the end of the pipe, at the edge of the acute mixing zone, at the edge of the chronic mixing zone, or in a completely mixed situation. If the sample is taken at the edge of a mixing zone, the composition of the sample will probably differ from one point to another along the edge of the mixing zone.

   If samples of actual downstream water are to be taken close to a discharge, the mixing patterns and plumes should be well known. Dye dispersion studies (Kilpatrick 1992) are commonly used to determine isopleths of effluent concentration and complete mix; dilution models (U.S. EPA 1993d) might also be helpful when selecting sampling locations. The most useful samples of actual downstream water are probably those taken just downstream of the point at which complete mix occurs or at the most distant point that is within the site to which the site-specific criterion is to apply. When samples are collected from a complete-mix situation, it might be appropriate to composite samples taken over a cross section of the stream. Regardless of where it is decided conceptually that a sample should be taken, it might be difficult to identify where the point exists in the stream and how it changes with flow and over time. In addition, if it is not known exactly what the sample actually represents, there is no way to know how reproducible the sample is. These problems make it difficult to relate WERs determined in actual downstream water to an effluent of concern because the concentration of effluent in the sample is not known; this is not a problem, however, if the sample-specific WER approach is used to interpret the results.

c. Simulated downstream water would seem to be the most unnatural of the three kinds of water, but it offers several important advantages because effluent and upstream water are mixed at a known ratio. This is important because the magnitude of the WER will often depend on the concentration of effluent in the downstream water. Mixtures can be prepared to simulate the ratio of effluent and upstream water that exists at the edge of the acute mixing zone, at the edge of the chronic mixing zone, at complete mix, or at any other point of interest. If desired, a sample of effluent can be mixed with a sample on upstream water in different ratios to simulate different points in a stream. Also, the ratio used can be one that simulates conditions at design flow or at any other flow.

The sample-specific WER approach can be used with both actual and simulated downstream water. Additional quantitative uses can be made of WERs determined using simulated downstream water because the percent effluent in the water is known, which allows quantitative extrapolations to the effluent. In addition, simulated downstream water can be used to determine the variation in the WER that is due to variation in the effluent. It also allows comparison of two or more effluents and determination of the interactions of two or more effluents. Additivity of WERs can be studied using simulated downstream water (see Appendix G); studies of toxicity within plumes and studies of whether increased flow of upstream water can increase toxicity are both studies of additivity of WERs. Use of simulated downstream water also makes it possible to conduct controlled studies of changes in WERs due to aging and changes in pH.

In Method 1, therefore, WERs are determined using simulated downstream water that is prepared by mixing samples of effluent and upstream water in an appropriate ratio. Most importantly, Method 1 can be used to determine a WER that applies in the vicinity of a plume and can be quantitatively extrapolated to the effluent.

2. Should WERs be determined when the stream flow is equal to, higher than, and/or lower than the design flow?

   WERs are used in the derivation of site-specific criteria when it is desired that permit limits be based on a criterion that takes into account the characteristics of the water and/or the metal at the site. In most cases, permit limits are calculated using steady-state models and are based on a design flow. It is therefore important that WERs be adequately protective under design-flow conditions, which might be expected to require that some sets of samples of effluent and upstream water be obtained when the actual stream flow is close to the design flow. Collecting samples when the stream flow is close to the design flow will limit a WER determination to the low-flow season (e.g., from mid-July to mid October in some places) and to years in which the flow is sufficiently low.

   It is also important, however, that WERs that are applied at design flow provide adequate protection at higher flows. Generalizations concerning the impact of higher flows on WERs are difficult because such flows might (a) reduce hardness, alkalinity, and pH, (b) increase or decrease the concentrations of TOC and TSS, (c) resuspend toxic and/or nontoxic metal from the sediment, and (d) wash additional pollutants into the water. Acidic snowmelt, for example, might lower the WER both by diluting the WER and by reducing the hardness, alkalinity, and pH; if substantial labile metal is present, the WER might be lowered more than the concentration of the metal, possibly resulting in increased toxicity at flows higher than design flow. Samples taken at higher flows might give smaller WERs because the concentration of the effluent is more dilute; however, total recoverable WERs might be larger if the sample is taken just after an event that greatly increases the concentration of TSS and/or TOC because this might increase both (1) the concentration of nontoxic particulate metal in the water and (2) the capacity of the water to sorb and detoxify metal.

   WERs are not of concern when the stream flow is lower than the design flow because these are acknowledged times of reduced protection. Reduced protection might not occur, however, if the WER is sufficiently high when the flow is lower than design flow.

3. Which toxicity tests should be used?

   a. As explained in Appendix D, the magnitude of an experimentally determined WER is likely to depend on the sensitivity of the toxicity test used. This relationship between the magnitude of the WER and the sensitivity of the toxicity test is due to the aqueous chemistry of metals and is not related to the test organisms or the type of test. The available data indicate that WERs determined with different tests do not differ greatly if the tests have about the same sensitivities, but the data also support the generalization that less sensitive toxicity tests usually give smaller WERs than more sensitive tests (see Appendix D).
   b. When the CCC is lower than the CMC, it is likely that a larger WER will result from tests that are sensitive at the CCC than from tests that are sensitive at the CMC.
   c. The considerations concerning the sensitivities of two tests should also apply to two endpoints for the same test. For any lethality test, use of the LC25 is likely to result in a larger WER than use of the LC50, although the difference might not be measurable in most cases and the LC25 is likely to be more variable than the LC50. Selecting the percent effect to be used to define the endpoint might take into account (a) whether the endpoint is above or below the CMC and/or the CCC and (b) the data obtained when tests are conducted. Once the percent effect is selected for a particular test (e.g., a 48-hr LC50 with 1-day-old fathead minnows), the same percent effect **must** be used whenever that test is used to determine a WER for that effluent. Similarly, if two different tests with the same species (e.g., a lethality test and a sublethal test) have substantially different sensitivities, both a cmcWER and a cccWER could be obtained with the same species.
   d. The <u>primary</u> toxicity test used in the determination of a WER should have an endpoint in laboratory dilution water that is close to, but <u>not lower</u> than, the CMC and/or CCC to which the WER is to be applied.
   e. Because the endpoint of the primary test in laboratory dilution water cannot be lower than the CMC and/or CCC, the magnitude of the WER is likely to become closer to 1 as the endpoint of the primary test becomes closer to the CMC and/or CCC (see Appendix D).
   f. The WER obtained with the primary test should be confirmed with a <u>secondary</u> test that uses a species that is taxonomically different from the species used in the primary test.
      1) The endpoint of the secondary test may be <u>higher or lower</u> than the CMC, the CCC, or the endpoint of the primary test.

2) Because of the limited number of toxicity tests that have sensitivities near the CMC or CCC for a metal, it seems unreasonable to require that the two species be further apart taxonomically than being in different orders.

Two different endpoints with the same species **must not** be used as the primary and secondary tests, even if one endpoint is lethal and the other is sublethal.

g. If more sensitive toxicity tests generally give larger WERs than less sensitive tests, the maximum value of a WER will usually be obtained using a toxicity test whose endpoint in laboratory dilution water equals the CMC or CMC. If such a test is not used, the maximum possible WER probably will not be obtained.

h. No rationale exists to support the idea that different species or tests with the same sensitivity will produce different WERs. Because the mode of action might differ from species to species and/or from effect to effect, it is easy to speculate that in some cases the magnitude of a WER will depend to some extent on the species, life stage, and/or kind of test, but no data are available to support conclusions concerning the existence and/or magnitude of any such differences.

i. If the tests are otherwise acceptable, both cmcWERs and cccWERs may be determined using acute and/or chronic tests and using lethal and/or sublethal endpoints. The important consideration is the sensitivity of the test, not the duration, species, life stage, or adverse effect used.

j. There is no reason to use species that occur at the site; they may be used in the determination of a WER if desired, but:
   1) It might be difficult to determine which of the species that occur at the site are sensitive to the metal and are adaptable to laboratory conditions.
   2) Species that occur at the site might be harder to obtain in sufficient numbers for conducting toxicity tests over the testing period.
   3) Additional QA tests will probably be needed (see section C.3.b) because data are not likely to be available from other laboratories for comparison with the results in laboratory dilution water.

k. Because a WER is a ratio of results obtained with the same test in two different dilution waters, toxicity tests that are used in WET testing, for example, may be used, even if the national aquatic life guidelines (U.S. EPA 1985) do not allow use of the test in the derivation of an aquatic life criterion. Of course, a test whose endpoint in laboratory dilution water is below the CMC and/or CCC that is to be adjusted cannot be used as a primary test.

l. Because there is no rationale that suggest that it makes any difference whether the test is conducted with a species that is warmwater or coldwater, a fish or an invertebrate, or resident or nonresident at the site, other than the fact that less sensitive tests are likely to give smaller WERs, such considerations as the availability of test organisms might be important in the selection of the test. Information in Appendix I, a criteria document for the metal of concern (see Appendix E), or any other pertinent source might be useful when selecting primary and secondary tests.

m. A test in which the test organisms are not fed might give a different WER than a test in which the organisms are fed just because of the presence of the food (see Appendix D). This might depend on the metal, the type and amount of food, and whether a total recoverable or dissolved WER is determined.

Different tests with similar sensitivities are expected to give similar WERs, except for experimental variation. The purpose of the secondary test is to provide information concerning this assumption and the validity of the WER.

4. Should a cmcWER or a cccWER or both be determined?

This question does not have to be answered if the criterion for the site contains either a CMC or a CCC but not both. For example, a body of water that is protected for put-and-take fishing might have only a CMC, whereas a stream whose design flow is zero might have only a CCC.

When the criterion contains both a CMC and a CCC, the simplistic way to answer the question is to determine whether the CMC or the CCC controls the existing permit limits; which one is controlling depends on (a) the ratio of the CMC to the CCC, (b) whether the number of mixing zones is zero, one, or two, and (c) which steady-state or dynamic model was used in the calculation of the permit limits. A better way to answer the question would be to also determine how much the controlling value would have to be changed for the other value to become controlling; this might indicate that it would not be cost-effective to derive, for example, a site-specific CMC (ssCMC) without also deriving a site-specific CCC (ssCCC). There are also other possibilities: (1) It might be appropriate to use a phased approach, i.e., determine either the cmcWER or the cccWER and then decide whether to determine the other. (2) It might be appropriate and environmentally conservative to determine a WER that can be applied to both the CMC and the CCC. (3) It is always allowable to determine and use both a cmcWER and a cccWER, although both can be determined only if toxicity tests with appropriate sensitivities are available.

Because the phased approach can always be used, it is only important to decide whether to use a different approach when its use might be cost-effective. Deciding whether to use a different approach and selecting which one to use is complex because a number of considerations need to be taken into account:

a. Is the CMC equal to or higher than the CCC?

   If the CMC equals the CCC, two WERs cannot be determined if they would be determined using the same site water, but two WERs could be determined if the cmcWER and the cccWER would be determined using different site waters, e.g., waters that contain different concentrations of the effluent.

b. If the CMC is higher than the CCC, is there a toxicity test whose endpoint in laboratory dilution water is between the CMC and the CCC?

   If the CMC is higher than the CCC and there is a toxicity test whose endpoint in laboratory dilution water is between the CMC and the CCC, both a cmcWER and a cccWER can be determined. If the CMC is higher than the CCC but no toxicity test has an endpoint in laboratory dilution water between the CMC and the CCC, two WERs cannot be determined if they would be determined using the same site water; two WERs could be determined if they were determined using different site waters, e.g., waters that contain different concentrations of the effluent.

c. Was a steady-state or a dynamic model used in the calculation of the permit limits?

   It is complex, but reasonably clear, how to make a decision when a steady-state model was used, but it is not clear how a decision should be made when a dynamic model was used.

d. If a steady-state model was used, were one or two design flows used, i.e., was the hydrologically based steady-state method used or was the biologically based steady-state method used?

   When the hydrologically based method is used, one design flow is used for both the CMC and the CCC, whereas when the biologically based method is used, there is a CMC design flow and a CCC design flow. When WERs are determined using downstream water, use of the biologically based method will probably cause the percent effluent in the site water used in the determination of the cmcWER to be different from the percent effluent in the site water used in the determination of the cccWER; thus the two WERs should be determined using two different site waters. This does not impact WERs determined using upstream water.

e. Is there an acute mixing zone? Is there a chronic mixing zone?
   1. When WERs are determined using upstream water, the presence or absence of mixing zones has no impact; the cmcWER and the cccWER will both be determined using site water that contains zero percent effluent, i.e., the two WERs will be determined using the same site water.
   2. Even when downstream water is used, whether there is an acute mixing zone affects the point of application of the CMC or ssCMC, but it does not affect the determination of any WER.
   3. The existence of a chronic mixing zone has important implications for the determination of WERs when downstream water is used (see Appendix A). When WERs are determined using downstream water, the cmcWER should be determined using water at the edge of the chronic mixing zone, whereas the cccWER should be determined using water from a complete-mix situation. (If the biologically based method is used, the two different design flows should also be taken into account when determining the percent effluent that should be in the simulated downstream water.) Thus the percent effluent in the site water used in the determination of the cmcWER will be different from the percent effluent in the site water used in the determination of the cccWER; this is important because the magnitude of a WER will often depend substantially on the percent effluent in the water (see Appendix D).

f. In what situations would it be environmentally conservative to determine one WER and use it to adjust both the cmcWER and the cccWER?

   Because (1) the CMC is never lower than the CCC and (2) a more sensitive test will generally give a WER closer to 1, it will be environmentally conservative to use a cmcWER to adjust a CCC when there are no contradicting considerations. In this case, a cmcWER can be determined and used to adjust both the CMC and the CCC. Because water quality can affect the WER, this approach is necessarily valid only if the cmcWER and the cccWER are determined in the same site water. Other situations in which it would be environmentally conservative to use one WER to adjust both the CMC and the CCC are described below.

These considerations have one set of implications when both the cmcWER and cccWER are to be determined using the same site water, and another set of implications when the two WERs are to be determined using different site waters, e.g., when the site waters contain different concentrations of effluent.

When WERs are determined using <u>upstream</u> water, the same site water is used in the determination of both the cmcWER and the cccWER. Whenever the two WERs are determined in the same site water, any difference in the magnitude of the cmcWER and the cccWER will probably be due to the sensitivities of the toxicity tests used. Therefore:

a. If more sensitive toxicity tests generally give larger WERs than less sensitive tests, the maximum cccWER (a cccWER determined with a test whose endpoint equals the CCC) will usually be larger than the maximum cmcWER because the CCC is never higher than the CMC.
b. Because the CCC is never higher than the CMC, the maximum cmcWER will usually be smaller than the maximum cccWER and it will be environmentally conservative to use the cmcWER to adjust the CCC.
c. A cccWER can be determined separately from a cmcWER only if there is a toxicity test with an endpoint in laboratory dilution water that is between the CMC and the CCC. If no such test exists or can be devised, only a cmcWER can be determined, but it can be used to adjust both the CMC and the CCC.
d. Unless the experimental variation is increased, use of a cccWER, instead of a cmcWER, to adjust the CCC will usually improve the accuracy of the resulting site-specific CCC. Thus a cccWER may be determined and used whenever desired, if a toxicity test has an endpoint in laboratory dilution water between the CMC and the CCC.
e. A cccWER cannot be used to adjust a CMC if the cccWER was determined using an endpoint that was lower than the CMC in laboratory dilution water because it will probably reduce the level of protection.
f. Even if there is a toxicity test that has an endpoint in laboratory dilution water that is between the CMC and the CCC, it is not necessary to decide initially whether to determine a cmcWER and/or a cccWER. When upstream water is used, it is always allowable to determine a cmcWER and use it to derive a site-specific CMC and a site-specific CCC and then decide whether to determine a cccWER.
g. If there is a toxicity test whose endpoint in laboratory dilution water is between the CCC and the CMC, and if this test is used as the secondary test in the determination of the cmcWER, this test will provide information that should be very useful for deciding whether to determine a cccWER in addition to a cmcWER. Further, if it is decided to determine a cccWER, the same two tests used in the determination of the cmcWER could then be used in the determination of the cccWER, with a reversal of their roles as primary and secondary tests. Alternatively, a cmcWER and a cccWER could be determined simultaneously if both tests are conducted on each sample of site water.

When WERs are determined using downstream water, the magnitude of each WER will probably depend on the concentration of effluent in the downstream water used (see Appendix D). The first important consideration is whether the design flow is greater than zero, and the second is whether there is a chronic mixing zone.

a. If the design flow is zero, cmcWERs and/or cccWERs that are determined for design-flow conditions will both be determined in 100 percent effluent. Thus this case is similar to using upstream water in that both WERs are determined in the same site water. When WERs are determined for high-flow conditions, it will make a difference whether a chronic mixing zone needs to be taken into account, which is the second consideration.
b. If there is no chronic mixing zone, both WERs will be determined for the complete-mix situation; this case is similar to using upstream water in that both WERs are determined using the same site water. If there is a chronic mixing zone, cmcWERs should be determined in the site water that exists at the edge of the chronic mixing zone, whereas cccWERs should be determined for the complete-mix situation (see Appendix A). Thus the percent effluent will be higher in the site water used in the determination of the cmcWER than in the site water used in the determination of the cccWER. Because a site water with a higher percent effluent will probably give a larger WER than a site water with a lower percent effluent, both a cmcWER and a cccWER can be determined even if there is no test whose endpoint in laboratory dilution water is between the CMC and the CCC. There are opposing considerations, however:
   1) The site water used in the determination of the cmcWER will probably have a higher percent effluent than the site water used in the determination of the cccWER, which will tend to cause the cmcWER to be larger than the cccWER.
   2) If there is a toxicity test whose endpoint in laboratory dilution water is between the CMC and the CCC, use of a more sensitive test in the determination of the cccWER will tend to cause the cccWER to be larger than the cmcWER.

One consequence of these opposing considerations is that it is not known whether use of the cmcWER to adjust the CCC would be environmentally conservative; if this simplification is not known to be conservative, it should not be used. Thus it is important whether there is a toxicity test whose endpoint in laboratory dilution water is between the CMC and the CCC:

a. If no toxicity test has an endpoint in laboratory dilution water between the CMC and the CCC, the two WERs have to be determined with the same test, in which case the cmcWER will probably be larger because the percent effluent in the site water will be higher. Because of the difference in percent effluent in the site waters that should be used in the determinations of the two WERs, use of the cmcWER to adjust the CCC would not be environmentally conservative, but use of the cccWER to adjust the CMC would be environmentally conservative. Although both WERs could be determined, it would also be acceptable to determine only the cccWER and use it to adjust both the CMC and the CCC.

b. If there is a toxicity test whose endpoint in laboratory dilution water is between the CMC and the CCC, the two WERs could be determined using different toxicity tests. An environmentally conservative alternative to determining two WERs would be to determine a hybrid WER by using (1) a toxicity test whose endpoint is above the CMC (i.e., a toxicity test that is appropriate for the determination of a cmcWER) and (2) site water for the complete-mix situation (i.e., site water appropriate for the determination of cccWER). It would be environmentally conservative to use this hybrid WER to adjust the CMC and it would be environmentally conservative to use this hybrid WER to adjust the CCC. Although both WERs could be determined, it would also be acceptable to determine only the hybrid WER and use it to adjust both the CMC and the CCC. (This hybrid WER described here in paragraph b is the same as the cccWER described in paragraph a above in which no toxicity test had an endpoint in laboratory dilution water between the CMC and the CCC.)

5. How should a FWER be derived?

<u>Background</u>

Because of experimental variation and variation in the composition of surface waters and effluents, a single determination of a WER does not provide sufficient information to justify adjustment of a criterion. After a sufficient number of WERs have been determined in an acceptable manner, a Final Water-Effect Ratio (FWER) is derived from the WERs, and the FWER is then used to calculate the site-specific criterion. If both a site-specific CMC and a site-specific CCC are to be derived, both a cmcFWER and a cccFWER have to be derived, unless an environmentally conservative estimate is used in place of the cmcFWER and/or the cccFWER.

When a WER is determined using upstream water, the two major sources of variation in the WER are (a) variability in the quality of the upstream water, much of which might be related to season and/or flow, and (b) experimental variation. When a WER is determined in downstream water, the four major sources of variation are (a) variability in the quality of the upstream water, much of which might be related to season and/or flow, (b) experimental variation, (c) variability in the composition of the effluent, and (d) variability in the percent effluent in the downstream water. Variability and the possibility of mistakes and rare events make it necessary to try to compromise between (1) providing a high probability of adequate protection and (2) placing too much reliance on the smallest experimentally determined WER, which might reflect experimental variation, a mistake, or a rare event rather than a meaningful difference in the WER.

Various ways can be employed to address variability:

a. Replication can be used to reduce the impact of some sources of variation and to verify the importance of others.
b. Because variability in the composition of the effluent might contribute substantially to the variability of the WER, it might be desirable to obtain and store two or more samples of the effluent at slightly different times, with the selection of the sampling times depending on such characteristics of the discharge as the average retention time, in case an unusual WER is obtained with the first sample used.
c. Because of the possibility of mistakes and rare events, samples of effluent and upstream water should be large enough that portions can be stored for later testing or analyses if an unusual WER is obtained.
d. It might be possible to reduce the impact of the variability in the percent effluent in the downstream water by establishing a relationship between the WER and the percent effluent.

Confounding of the sources can be a problem when more than one source contributes substantial variability.

When permit limits are calculated using a steady-state model, the limits are based on a design flow, e.g., the 7Q10. It is usually assumed that a concentration of metal in an effluent that does not cause unacceptable effects at the design flow will not cause unacceptable effects at higher flows because the metal is diluted by the increased flow of the upstream water. Decreased protection might occur, however, if an increase in flow increases toxicity more than it dilutes the concentration of metal. When permit limits are based on a national criterion, it is often assumed that the criterion is sufficiently conservative that an increase in toxicity will not be great enough to overwhelm the combination of dilution and the assumed conservatism, even though it is likely that the national criterion is not overprotective of all bodies of water. When WERs are used to reduce the assumed conservatism, there is more concern about the possibility of increased toxicity at flows higher than the design flow and it is important to (1) determine some WERs that correspond to higher flows or (2) provide some conservatism. If the concentration of effluent in the downstream water decreases as flow increases, WERs determined at higher flows are likely to be smaller than WERs determined at design flow but the concentration of metal will also be lower. If the concentration of TSS increases at high flows, however, both the WER and the concentration of metal might increase. If they are determined in an appropriate manner, WERs determined at flows higher than the design flow can be used in two ways:

a. As environmentally conservative estimates of WERs determined at design flow.
b. To assess whether WERs determined at design flow will provide adequate protection at higher flows.

In order to appropriately take into account seasonal and flow effects and their interactions, both ways of using high-flow WERs require that the downstream water used in the determination of the WER be similar to that which actually exists during the time of concern. In addition, high-flow WERs can be used in the second way only if the composition of the downstream water is known. To satisfy the requirements that (a) the downstream water used in the determination of a WER be similar to the actual water and (b) the composition of the downstream water be known, it is necessary to obtain samples of effluent and upstream water at the time of concern and to prepare a simulated downstream water by mixing the samples at the ratio of the flows of the effluent and the upstream water that existed when the samples were obtained.

For the first way of using high-flow WERs, they are used directly as environmentally conservative estimates of the design-flow WER. For the second way of using high-flow WERs, each is used to calculate the highest concentration of metal that could be in the effluent without causing the concentration of metal in the downstream water to exceed the site-specific criterion that would be derived for that water using the experimentally determined WER. This highest concentration of metal in the effluent (HCME) can be calculated as:

$$HCME = \frac{[(CCC)(WER)(eFLOW + uFLOW)] - [(uCONC)(uFLOW)]}{eFLOW},$$

where:

CCC = the national, state, or recalculated CCC (or CMC) that is to be adjusted.

eFLOW = the flow of the effluent that was the basis of the preparation of the simulated downstream water. This should be the flow of the effluent that existed when the samples were taken.

uFLOW = the flow of the upstream water that was the basis of the preparation of the simulated downstream water. This should be the flow of the upstream water that existed when the samples were taken.

uCONC = the concentration of metal in the sample of upstream water used in the preparation of the simulated downstream water.

In order to calculate a HCME from an experimentally determined WER, the only information needed besides the flows of the effluent and the upstream water is the concentration of metal in the upstream water, which should be measured anyway in conjunction with the determination of the WER.

When a steady-state model is used to derive permit limits, the limits on the effluent apply at all flows; thus, each HCME can be used to calculate the highest WER (hWER) that could be used to derive a site-specific criterion for the downstream water at design flow so that there would be adequate protection at the flow for which the HCME was determined. The hWER is calculated as:

$$hWER = \frac{(HCME)(eFLOWdf) + (uCONCdf)(uFLOWdf)}{(CCC)(eFLOWdf + uFLOWdf)}$$

The suffix "df" indicates that the values used for these quantities in the calculation of the hWER are those that exist at design-flow conditions. The additional datum needed in order to calculate the hWER is the concentration of metal in upstream water at design-flow conditions; if this is assumed to be zero, the hWER will be environmentally conservative. If a WER is determined when uFLOW equals the design flow, hWER = WER.

The two ways of using WERs determined at flows higher than design flow can be illustrated using the following examples. These examples were formulated using the concept of additivity of WERs (see Appendix G). A WER determined in downstream water consists of two components, one due to the effluent (the eWER) and one due to the upstream water (the uWER). If the eWER and uWER are strictly additive, when WERs are determined at various upstream flows, the downstream WERs can be calculated from the composition of the downstream water (the % effluent and the % upstream water) and the two WERs (the eWER and the uWER) using the equation:

$$WER = \frac{(\%\ effluent)(eWER) + (\%\ upstream\ water)(uWER)}{100}\ .$$

In the examples below, it is assumed that:
a. A site-specific CCC is being derived.
b. The national CCC is 2 ug/L.
c. The eWER is 40.
d. The eWER and uWER are constant and strictly additive.
e. The flow of the effluent (eFLOW) is always 10 cfs.
f. The design flow of the upstream water (uFLOWdf) is 40 cfs.

Therefore:

$$HCME = \frac{[(2\ ug/L)(WER)(10\ cfs + uFLOW)] - [(uCONC)(uFLOW)]}{10\ ug/L}\ .$$

$$hWER = \frac{(HCME)(10\ cfs) + (uCONCdf)(40\ cfs)}{(2\ ug/L)(10\ cfs + 40\ cfs)}\ .$$

In the first example, the uWER is assumed to be 5 and so the upstream site-specific CCC (ussCCC) = (CCC)(uWER) = (2 ug/L)(5) = 10 ug/L. uCONC is assumed to be 0.4 ug/L, which means that the assimilative capacity of the upstream water is 9.6 ug/L.

eFLOW	uFLOW	At Complete Mix			HCME	hWER
(cfs)	(cfs)	% Eff.	% Ups.	WER	(ug/L)	
10	40	20.0	80.0	12.000	118.4	12.00
10	63	13.7	86.3	9.795	140.5	14.21
10	90	10.0	90.0	8.500	166.4	16.80
10	190	5.0	95.0	6.750	262.4	26.40
10	490	2.0	98.0	5.700	550.4	55.20
10	990	1.0	99.0	5.350	1030.4	103.20
10	1990	0.5	99.5	5.175	1990.4	199.20

As the flow of the upstream water increases, the WER decreases to a limiting value equal to uWER. Because the assimilative capacity is greater than zero, the HCMEs and hWERs increase due to the increased dilution of the effluent. The increase in hWER at higher flows will not allow any use of the assimilative capacity of the upstream water because the allowed concentration of metal in the effluent is controlled by the lowest hWER, which is the design-flow hWER in this example. Any WER determined at a higher flow can be used as an environmentally conservative estimate of the design-flow WER, and the hWERs show that the WER of 12 provides adequate protection at all flows. When uFLOW equals the design flow of 40 cfs, WER = hWER.

In the second example, uWER is assumed to be 1, which means that ussCCC = 2 ug/L. uCONC is assumed to be 2 ug/L, so that uCONC = ussCCC. The assimilative capacity of the upstream water is 0 ug/L.

eFLOW (cfs)	uFLOW (cfs)	At Complete Mix % Eff.	% Ups.	WER	HCME (ug/L)	hWER
10	40	20.0	80.0	8.800	80.00	8.800
10	63	13.7	86.3	6.343	80.00	8.800
10	90	10.0	90.0	4.900	80.00	8.800
10	190	5.0	95.0	2.950	80.00	8.800
10	490	2.0	98.0	1.780	80.00	8.800
10	990	1.0	99.0	1.390	80.00	8.800
10	1990	0.5	99.5	1.195	80.00	8.800

All the WERs in this example are lower than the comparable WERs in the first example because the uWER dropped from 5 to 1; the limiting value of the WER at very high flow is 1. Also, the HCMEs and hWERs are independent of flow because the increased dilution does not allow any more metal to be discharged when uCONC = ussCCC, i.e., when the assimilative capacity is zero. As in the first example, any WER determined at a flow higher than design flow can be used as an environmentally conservative estimate of the design-flow WER and the hWERs show that the WER of 8.8 determined at design flow will provide adequate protection at all flows for which information is available. When uFLOW equals the design flow of 40 cfs, WER = hWER.

In the third example, uWER is assumed to be 2, which means that ussCCC = 4 ug/L. uCONC is assumed to be 1 ug/L; thus the assimilative capacity of the upstream water is 3 ug/L.

eFLOW (cfs)	uFLOW (cfs)	At Complete Mix % Eff.	% Ups.	WER	HCME (ug/L)	hWER
10	40	20.0	80.0	9.600	92.0	9.60
10	63	13.7	86.3	7.206	98.9	10.29
10	90	10.0	90.0	5.800	107.0	11.10
10	190	5.0	95.0	3.900	137.0	14.10
10	490	2.0	98.0	2.760	227.0	23.10
10	990	1.0	99.0	2.380	377.0	38.10
10	1990	0.5	99.5	2.190	677.0	68.10

All the WERs in this example are intermediate between the comparable WERs in the first two examples because the uWER is now 2, which is between 1 and 5; the limiting value of the WER at very high flow is 2. As in the other examples, any WER determined at a flow higher than design flow can be used as an environmentally conservative estimate of the design-flow WER and the hWERs show that the WER of 9.6 determined at design flow will provide adequate protection at all flows for which information is available. When uFLOW equals the design flow of 40 cfs, WER = hWER.

If this third example is assumed to be subject to acidic snowmelt in the spring so that the eWER and uWER are less-than-additive and result in a WER of 4.8 (rather than 5.8) at a uFLOW of 90 cfs, the third HCME would be 87 ug/L, and the third hWER would be 9.1. This hWER is lower than the design-flow WER of 9.6, so the site-specific criterion would have to be derived using the WER of 9.1, rather than the design-flow WER of 9.6, in order to provide the intended level of protection. If the eWER and uWER were less-than-additive only to the extent that the third WER was 5.3, the third HCME would be 97 ug/L and the third hWER would be 10.1. In this case, dilution by the increased flow would more than compensate for the WERs being less-than-additive, so that the design-flow WER of 9.6 would provide adequate protection at a uFLOW of 90 cfs. Auxiliary information might indicate whether an unusual WER is real or is an accident; for example, if the hardness, alkalinity, and pH of snowmelt are all low, this information would support a low WER.

If the eWER and uWER were more-than-additive so that the third WER was 10, this WER would not be an environmentally conservative estimate of the design-flow WER. If a WER determined at a higher flow is to be used as an estimate of the design-flow WER and there is reason to believe that the eWER and the uWER might be more-than-additive, a test for additivity can be performed (see Appendix G).

Calculating HCMEs and hWERs is straightforward if the WERs are based on the total recoverable measurement. If they are based on the dissolved measurement, it is necessary to take into account the percent of the total recoverable metal in the effluent that becomes dissolved in the downstream water.

To ensure adequate protection, a group of WERs should include one or more WERs corresponding to flows near the design flow, as well as one or more WERs corresponding to higher flows.

a. Calculation of hWERs from WERs determined at various flows and seasons identifies the highest WER that can be used in the derivation of a site-specific criterion and still provide adequate protection at all flows for which WERs are available. Use of hWERs eliminates the need to assume that WERs determined at design flow will provide adequate protection at higher flows. Because hWERs are calculated to apply at design flow, they apply to the flow on which the permit limits are based. The lowest of the hWERs ensures adequate protection at all flows, if hWERs are available for a sufficient range of flows, seasons, and other conditions.

b. Unless additivity is assumed, a WER cannot be extrapolated from one flow to another and therefore it is not possible to predict a design-flow WER from a WER determined at other conditions. The largest WER is likely to occur at design flow because, of the flows during which protection is to be provided, the design flow is the flow at which the highest concentration of effluent will probably occur in the downstream water. This largest WER has to be experimentally determined; it cannot be predicted.

The examples also illustrate that if the concentration of metal in the upstream water is below the site-specific criterion for that water, in the limit of infinite dilution of the effluent with upstream water, there will be adequate protection. The concern, therefore, is for intermediate levels of dilution. Even if the assimilative capacity is zero, as in the second example, there is more concern at the lower or intermediate flows, when the effluent load is still a major portion of the total load, than at higher flows when the effluent load is a minor contribution.

The Options

To ensure adequate protection over a range of flows, two types of WERs need to be determined:

Type 1 WERs are determined by obtaining samples of effluent and upstream water when the downstream flow is between one and two times higher than what it would be under design-flow conditions.

Type 2 WERs are determined by obtaining samples of effluent and upstream water when the downstream flow is between two and ten times higher than what it would be under design-flow conditions.

The only difference between the two types of samples is the downstream flow at the time the samples are taken. For both types of WERs, the samples should be mixed at the ratio of the flows that existed when the samples were taken so that seasonal and flow-related changes in the water quality characteristics of the upstream water are properly related to the flow at which they occurred. The ratio at which the samples are mixed does not have to be the exact ratio that existed when the samples were taken, but the ratio has to be known, which is why simulated downstream water is used. For each Type 1 WER and each Type 2 WER that is determined, a hWER is calculated.

Ideally, sufficient numbers of both types of WERs would be available and each WER would be sufficiently precise and accurate and the Type 1 WERs would be sufficiently similar that the FWER could be the geometric mean of the Type 1 WERs, unless the FWER had to be lowered because of one or more hWERs. If an adequate number of one or both types of WERs is not available, an environmentally conservative WER or hWER should be used as the FWER.

Three Type 1 and/or Type 2 WERs, which were determined using acceptable procedures and for which there were at least three weeks between any two sampling events, **must** be available in order for a FWER to be derived. If three or more are available, the FWER should be derived from the WERs and hWERs using the lowest numbered option whose requirements are satisfied:

1. If there are two or more Type 1 WERs:
   a. If at least nineteen percent of all of the WERs are Type 2 WERs, the derivation of the FWER depends on the properties of the Type 1 WERs:
      1) If the range of the Type 1 WERs is not greater than a factor of 5 and/or the range of the ratios of the Type 1 WER to the concentration of metal in the simulated downstream water is not greater than a factor of 5, the FWER is the lower of (a) the adjusted geometric mean (see Figure 2) of all of the Type 1 WERs and (b) the lowest hWER.
      2) If the range of the Type 1 WERs is greater than a factor of 5 and the range of the ratios of the Type 1 WER to the concentration of metal in the simulated downstream water is greater than a factor of 5, the FWER is the lowest of (a) the lowest Type 1 WER, (b) the lowest hWER, and (c) the geometric mean of all the Type 1 and Type 2 WERs, unless an analysis of the joint probabilities of the occurrences of WERs and metal concentrations indicates that a higher WER would still provide the level of protection intended by the criterion. (EPA intends to provide guidance concerning such an analysis.)
   b. If less than nineteen percent of all of the WERs are Type 2 WERs, the FWER is the lower of (1) the lowest Type 1 WER and (2) the lowest hWER.
2. If there is one Type 1 WER, the FWER is the lowest of (a) the Type 1 WER, (b) the lowest hWER, and (c) the geometric mean of all of the Type 1 and Type 2 WERs.
3. If there are no Type 1 WERs, the FWER is the lower of (a) the lowest Type 2 WER and (b) the lowest hWER.

If fewer than three WERs are available and a site-specific criterion is to be derived using a WER or a FWER, the WER or FWER has to be assumed to be 1. Examples of deriving FWERs using these options are presented in Figure 3.

The options are designed to ensure that:
a. The options apply equally well to ordinary flowing waters and to streams whose design flow is zero.
b. The requirements for deriving the FWER as something other than the lowest WER are not too stringent.
c. The probability is high that the criterion will be adequately protective at all flows, regardless of the amount of data that are available.
d. The generation of both types of WERs is encouraged because environmental conservatism is built in if both types of WERs are not available in acceptable numbers.
e. The amount of conservatism decreases as the quality and quantity of the available data increase.

The requirement that three WERs be available is based on a judgment that fewer WERs will not provide sufficient information. The requirement that at least nineteen percent of all of the available WERs be Type 2 WERs is based on a judgment concerning what constitutes an adequate mix of the two types of WERs: when there are five or more WERs, at least one-fifth should be Type 2 WERs.

Because each of these options for deriving a FWER is expected to provide adequate protection, anyone who desires to determine a FWER can generate three or more appropriate WERs and use the option that corresponds to the WERs that are available. The options that utilize the least useful WERs are expected to provide adequate protection because of the way the FWER is derived from the WERs. It is intended that, on the average, Option 1a will result in the highest FWER, and so it is recommended that data generation should be designed to satisfy the requirements of this option if possible. For example, if two Type 1 WERs have been determined, determining a third Type 1 WER will require use of Option 1b, whereas determining a Type 2 WER will require use of Option 1a.

Calculation of the FWER as an adjusted geometric mean raises three issues:
a. The level of protection would be greater if the lowest WER, rather than an adjusted mean, were used as the FWER. Although true, the intended level of protection is provided by the national aquatic life criterion derived according to the national guidelines; when sufficient data are available and it is clear how the data should be used, there is no reason to add a substantial margin of safety and thereby change the intended level of protection. Use of an adjusted geometric mean is acceptable if sufficient data are available concerning the WER to demonstrate that the adjusted geometric mean will provide the intended level of protection. Use of the lowest of three or more WERs would be justified, if, for example, the criterion had

been lowered to protect a commercially important species and a WER determined with that species was lower than WERs determined with other species.

b. The level of protection would be greater if the adjustment was to a probability of 0.95 rather than to a probability of 0.70. As above, the intended level of protection is provided by the national aquatic life criterion derived according to the national guidelines. There is no need to substantially increase the level of protection when site-specific criteria are derived.

c. It would be easier to use the more common arithmetic mean, especially because the geometric mean usually does not provide much more protection than the arithmetic mean. Although true, use of the geometric mean rather than the arithmetic mean is justified on the basis of statistics and mathematics; use of the geometric mean is also consistent with the intended level of protection. Use of the arithmetic mean is appropriate when the values can range from minus infinity to plus infinity. The geometric mean (GM) is equivalent to using the arithmetic mean of the logarithms of the values. WERs cannot be negative, but the logarithms of WERs can. The distribution of the logarithms of WERs is therefore more likely to be normally distributed than is the distribution of the WERs. Thus, it is better to use the GM of WERs. In addition, when dealing with quotients, use of the GM reduces arguments about the correct way to do some calculations because the same answer is obtained in different ways. For example, if WER1 = (N1)/(D1) and WER2 = (N2)/(D2), then the GM of WER1 and WER2 gives the same value as [(GM of N1 and N2)/(GM of D1 and D2)] and also equals the square root of {[(N1)(N2)]/[(D1)(D2)]}.

Anytime the FWER is derived as the lowest of a series of experimentally determined WERs and/or hWERs, the magnitude of the FWER will depend at least in part on experimental variation. There are at least three ways that the influence of experimental variation on the FWER can be reduced:

a. A WER determined with a primary test can be replicated and the geometric mean of the replicates used as the value of the WER for that determination. Then the FWER would be the lowest of a number of geometric means rather than the lowest of a number of individual WERs. To be true replicates, the replicate determinations of a WER should not be based on the same test in laboratory dilution water, the same sample of site water, or the same sample of effluent.

b. If, for example, Option 3 is to be used with three Type 2 WERs and the endpoints of both the primary and secondary tests in laboratory dilution water are above the CMC and/or CCC to which the WER is to apply, WERs can be determined with both the primary and secondary tests for each of the three sampling times. For each sampling time, the geometric mean of the WER obtained with the primary test and the WER obtained with the secondary test could be calculated; then the lowest of these three geometric means could be used as the FWER. The three WERs cannot consist of some WERs determined with one of the tests and some WERs determined with the other test; similarly the three WERs cannot consist of a combination of individual WERs obtained with the primary and/or secondary tests and geometric means of results of primary and secondary tests.

c. As mentioned above, because the variability of the effluent might contribute substantially to the variability of the WERs, it might be desirable to obtain and store more than one sample of the effluent when a WER is to be determined in case an unusual WER is obtained with the first sample used.

Examples of the first and second ways of reducing the impact of experimental variation are presented in Figure 4. The availability of these alternatives does not mean that they are necessarily cost-effective.

6. For metals whose criteria are hardness-dependent, at what hardness should WERs be determined?

The issue of hardness bears on such topics as acclimation of test organisms to the site water, adjustment of the hardness of the site water, and how an experimentally determined WER should be used. If all WERs were determined at design-flow conditions, it might seem that all WERs should be determined at the design-flow hardness. Some permit limits, however, are not based on the hardness that is most likely to occur at design flow; in addition, conducting all tests at design-flow conditions provides no information concerning whether adequate protection will be provided at other flows. Thus, unless the hardnesses of the upstream water and the effluent are similar and do not vary with flow, the hardness of the site water will not be the same for all WER determinations.

Because the toxicity tests should be begun within 36 hours after the samples of effluent and upstream water are collected, there is little time to acclimate organisms to a sample-specific hardness. One alternative would be to acclimate the organisms to a preselected hardness and then adjust the hardness of the site water, but adjusting the hardness of the site water might have various effects on the toxicity of the metal due to competitive binding and ionic impacts on the test organisms and on the speciation of the metal; lowering hardness without also diluting the WER is especially problematic. The least objectionable approach is to acclimate the organisms to a laboratory dilution water with a hardness in the range of 50 to 150 mg/L and then use this water as the laboratory dilution water when the WER is determined. In this way, the test organisms will be acclimated to the laboratory dilution water as specified by ASTM (1993a,b,c,d,e).

Test organisms may be acclimated to the site water for a short time as long as this does not cause the tests to begin more than 36 hours after the samples were collected. Regardless of what acclimation procedure is used, the organisms used for the toxicity test conducted using site water are unlikely to be acclimated as well as would be desirable. This is a general problem with toxicity tests conducted in site water (U.S. EPA 1993a,b,c; ASTM 1993f), and its impact on the results of tests is unknown.

For the practical reasons given above, an experimentally determined WER will usually be a ratio of endpoints determined at two different hardnesses and will thus include contributions from a variety of differences between the two waters, including hardness. The disadvantages of differing hardnesses are that (a) the test organisms probably will not be adequately acclimated to site water and (b) additional calculations will be needed to account for the differing hardnesses; the advantages are that it allows the generation of data concerning the adequacy of protection at various flows of upstream water and it provides a way of overcoming two problems with the hardness equations: (1) it is not known how applicable they are to hardnesses outside the range of 25 to 400 mg/L and (2) it is not known how applicable they are to unusual combinations of hardness, alkalinity, and pH or to unusual ratios of calcium and magnesium.

The additional calculations that are necessary to account for the differing hardnesses will also overcome the shortcomings of the hardness equations. The purpose of determining a WER is to determine how much metal can be in a site water without lowering the intended level of protection. Each experimentally determined WER is inherently referenced to the hardness of the laboratory dilution water that was used in the determination of the WER, but the hardness equation can be used to calculate adjusted WERs that are referenced to other hardnesses for the laboratory dilution water. When used to adjust WERs, a hardness equation for a CMC or CCC can be used to reference a WER to any hardness for a laboratory dilution water, whether it is inside or outside the range of 25 to 400 mg/L, because any inappropriateness in the equation will be automatically compensated for when the adjusted WER is used in the derivation of a FWER and permit limits.

For example, the hardness equation for the freshwater CMC for copper gives CMCs of 9.2, 18, and 34 ug/L at hardnesses of 50, 100, and 200 mg/L, respectively. If acute toxicity tests with Ceriodaphnia reticulata gave an EC50 of 18 ug/L using a laboratory dilution water with a hardness of 100 mg/L and an EC50 of 532.2 ug/L in a site water, the resulting WER would be 29.57. It can be assumed that, within experimental variation, EC50s of 9.2 and 34 ug/L and WERs of 57.85 and 15.65 would have been obtained if laboratory dilution waters with hardnesses of 50 and 200 mg/L, respectively, had been used, because the EC50 of 532.2 ug/L obtained in the site water does not depend on what water is used for the laboratory dilution water. The WERs of 57.85 and 15.65 can be considered to be adjusted WERs that were extrapolated from the experimentally determined WER using the hardness equation for the copper CMC. If used correctly, the experimentally determined WER and all of the adjusted WERs will result in the same permit limits because they are internally consistent and are all based on the EC50 of 532.2 ug/L that was obtained in site water.

A hardness equation for copper can be used to adjust the WER if the hardness of the laboratory dilution water used in the determination of the WER is in the range of 25 to 400 mg/L (preferably in the range of about 40 to 250 mg/L because most of the data used to derive the equation are in this range). However, the hardness equation can be used to adjust WERs to hardnesses outside the range of 25 to 400 mg/L because the basis of the adjusted WER does not change the fact that the EC50 obtained in site water was 532.2 ug/L. If the hardness of the site water was 16 mg/L, the hardness equation would predict an EC50 of 3.153 ug/L, which would result in an adjusted WER of 168.8. This use of the hardness equation outside the range of 25 to 400 mg/L is valid only if the calculated CMC is used with the corresponding adjusted WER. Similarly, if the hardness of the site water had been 447 mg/L, the hardness equation would predict an EC50 of 72.66 ug/L, with a corresponding adjusted WER of 7.325. If the hardness of 447 mg/L were due to an effluent that contained calcium chloride and the alkalinity and pH of the site water were what would usually occur at a hardness of 50 mg/L rather than 400 mg/L, any inappropriateness in the calculated EC50 of 72.66 ug/L will be compensated for in the adjusted WER of 7.325, because the adjusted WER is based on the EC50 of 532.2 ug/L that was obtained using the site water.

In the above examples it was assumed that at a hardness of 100 mg/L the EC50 for C. reticulata equalled the CMC, which is a very reasonable simplifying assumption. If, however, the WER had been determined with the more resistant Daphnia pulex and EC50s of 50 ug/L and 750 ug/L had been obtained using a laboratory dilution water and a site water, respectively, the CMC given by the hardness equation could not be used as the predicted EC50. A new equation would have to be derived by changing the intercept so that the new equation gives an EC50 of 50 ug/L at a hardness of 100 mg/L; this new equation could then be used to calculate adjusted EC50s, which could then be used to calculate corresponding adjusted WERs:

Hardness (mg/L)	EC50 (ug/L)	WER
16	8.894	84.33
50	26.022	28.82
100	50.000*	15.00*
200	96.073	7.81
447	204.970	3.66

The values marked with an asterisk are the assumed experimentally determined values; the others were calculated from these values. At each hardness the product of the EC50 times the WER equals 750 ug/L because all of the WERs are based on the same EC50 obtained using site water. Thus use of the WER allows application of the hardness equation for a metal to conditions to which it otherwise might not be applicable.

HCMEs can then be calculated using either the experimentally determined WER or an adjusted WER as long as the WER is applied to the CMC that corresponds to the hardness on which the WER is based. For example, if the concentration of copper in the upstream water was 1 ug/L and the flows of the effluent and upstream water were 9 and 73 cfs, respectively, when the samples were collected, the HCME calculated from the WER of 15.00 would be:

$$HCME = \frac{(17.73\ ug/L)(15)(9 + 73\ cfs) - (1\ ug/L)(73\ cfs)}{9\ cfs} = 2415\ ug/L$$

because the CMC is 17.73 ug/L at a hardness of 100 mg/L. (The value of 17.73 ug/L is used for the CMC instead of 18 ug/L to reduce roundoff error in this example.) If the hardness of the site water was actually 447 ug/L, the HCME could also be calculated using the WER of 3.66 and the CMC of 72.66 ug/L that would be obtained from the CMC hardness equation:

$$HCME = \frac{(72.66\ ug/L)(3.66)(9 + 73\ cfs) - (1\ ug/L)(73\ cfs)}{9\ cfs} = 2415\ ug/L\ .$$

Either WER can be used in the calculation of the HCME as long as the CMC and the WER correspond to the same hardness and therefore to each other, because:

$$(17.73\ ug/L)(15) = (72.66\ ug/L)(3.66)\ .$$

Although the HCME will be correct as long as the hardness, CMC, and WER correspond to each other, the WER used in the derivation of the FWER **must** be the one that is calculated using a hardness equation to be compatible with the hardness of the site water. If the hardness of the site water was 447 ug/L, the WER used in the derivation of the FWER has to be 3.66; therefore, the simplest approach is to calculate the HCME using the WER of 3.66 and the corresponding CMC of 72.66 ug/L, because these correspond to the hardness of 447 ug/L, which is the hardness of the site water.

In contrast, the hWER should be calculated using the CMC that corresponds to the design hardness. If the design hardness is 50 mg/L, the corresponding CMC is 9.2 ug/L. If the design flows of the effluent and the upstream water are 9 and 20 cfs, respectively, and the concentration of metal in upstream water at design conditions is 1 ug/L, the hWER obtained from the WER determined using the site water with a hardness of 447 mg/L would be:

$$hWER = \frac{(2415\ ug/L)(9\ cfs) + (1\ ug/L)(20\ cfs)}{(9.2\ ug/L)(9\ cfs + 20\ cfs)} = 81.54\ .$$

None of these calculations provides a way of extrapolating a WER from one site-water hardness to another. The only extrapolations that are possible are from one hardness of laboratory dilution water to another; the adjusted WERs are based on predicted toxicity in laboratory dilution water, but they are all based on measured toxicity in site water. If a WER is to apply to the design flow and the design hardness, one or more toxicity tests have to be conducted using samples of effluent and upstream water obtained under design-flow conditions and mixed at the design-flow ratio to produce the design hardness. A WER that is specifically appropriate to design conditions cannot be based on predicted toxicity in site water; it has to be based on measured toxicity in site water that corresponds to design-flow conditions. The situation is more complicated if the design hardness is not the hardness that is most likely to occur when effluent and upstream water are mixed at the ratio of the design flows.

B. Background Information and Initial Decisions

1. Information should be obtained concerning the effluent and the operating and discharge schedules of the discharger.

2. The spatial extent of the site to which the WER and the site-specific criterion are intended to apply should be defined (see Appendix A). Information concerning tributaries, the plume, and the point of complete mix should be obtained. Dilution models (U.S. EPA 1993d) and dye dispersion studies (Kilpatrick 1992) might provide information that is useful for defining sites for cmcWERs.

3. If the Recalculation Procedure (see Appendix B) is to be used, it should be performed.

4. Pertinent information concerning the calculation of the permit limits should be obtained:
   a. What are the design flows, i.e., the flow of the upstream water (e.g., 7Q10) and the flow of the effluent that are used in the calculation of the permit limits? (The design flows for the CMC and CCC might be the same or different.)
   b. Is there a CMC (acute) mixing zone and/or a CCC (chronic) mixing zone?
   c. What are the dilution(s) at the edge(s) of the mixing zone(s)?
   d. If the criterion is hardness-dependent, what is the hardness on which the permit limits are based? Is this a hardness that is likely to occur under design-flow conditions?

5. It should be decided whether to determine a cmcWER and/or a cccWER.

6. The water quality criteria document (see Appendix E) that serves as the basis of the aquatic life criterion should be read to identify any chemical or toxicological properties of the metal that are relevant.

7. If the WER is being determined by or for a discharger, it will probably be desirable to decide what is the smallest WER that is desired by the discharger (e.g., the smallest WER that would not require a reduction in the amount of metal discharged). This "smallest desired WER" might be useful when deciding whether to determine a WER. If a WER is determined, this "smallest desired WER" might be useful when selecting the range of concentrations to be tested in the site water.

8. Information should be read concerning health and safety considerations regarding collection and handling of

effluent and surface water samples and conducting toxicity tests (U.S. EPA 1993a; ASTM 1993a). Information should also be read concerning safety and handling of the metallic salt that will be used in the preparation of the stock solution.

9. The proposed work should be discussed with the appropriate regulatory authority (and possibly the Water Management Division of the EPA Regional Office) before deciding how to proceed with the development of a detailed workplan.

10. Plans should be made to perform one or more rangefinding tests in both laboratory dilution water and site water (see section G.7).

C. Selecting Primary and Secondary Tests

1. For each WER (cmcWER and/or cccWER) to be determined, the primary and secondary tests should be selected using the rationale presented in section A.3, the information in Appendix I, the information in the criteria document for the metal (see Appendix E), and any other pertinent information that is available. When a specific test species is not specified, also select the species. Because at least three WERs **must** be determined with the primary test, but only one **must** be determined with the secondary test, selection of the tests might be influenced by the availability of the species (and the life stage in some cases) during the planned testing period.
   a. The description of a "test" specifies not only the test species and the duration of the test but also the life stage of the species and the adverse effect on which the results are to be based, all of which can have a major impact on the sensitivity of the test.
   b. The endpoint (e.g., LC50, EC50, IC50) of the primary test in laboratory dilution water should be as close as possible, but it **must not** be below, the CMC and/or CCC to which the WER is to be applied, because for any two tests, the test that has the lower endpoint is likely to give the higher WER (see Appendix D).
      NOTE: If both the Recalculation Procedure and a WER are to be used in the derivation of the site-specific criterion, the Recalculation Procedure **must** be completed first because the recalculated CMC and/or CCC **must** be used in the selection of the primary and secondary tests.
   c. The endpoint (e.g., LC50, EC50, IC50) of the secondary test in laboratory dilution water should be as close as possible, but may be above or below, the CMC and/or CCC to which the WER is to be applied.

1) Because few toxicity tests have endpoints close to the CMC and CCC and because the major use of the secondary test is confirmation (see section I.7.b), the endpoint of the secondary test may be below the CMC or CCC. If the endpoint of the secondary test in laboratory dilution water is above the CMC and/or CCC, it might be possible to use the results to reduce the impact of experimental variation (see Figure 4). If the endpoint of the primary test in laboratory dilution water is above the CMC and the endpoint of the secondary test is between the CMC and CCC, it should be possible to determine both a cccWER and a cmcWER using the same two tests.
2) It is often desirable to conduct the secondary test when the first primary test is conducted in case the results are surprising; conducting both tests the first time also makes it possible to interchange the primary and secondary tests, if desired, without increasing the number of tests that need to be conducted. (If results of one or more rangefinding tests are not available, it might be desirable to wait and conduct the secondary test when more information is available concerning the laboratory dilution water and the site water.)

2. The primary and secondary tests **must** be conducted with species in different taxonomic orders; at least one species **must** be an animal and, when feasible, one species should be a vertebrate and the other should be an invertebrate. A plant cannot be used if nutrients and/or chelators need to be added to either or both dilution waters in order to determine the WER. It is desirable to use a test and species for which the rate of success is known to be high and for which the test organisms are readily available. (If the WER is to be used with a recalculated CMC and/or CCC, the species used in the primary and secondary tests do not have to be on the list of species that are used to obtain the recalculated CMC and/or CCC.)

3. There are advantages to using tests suggested in Appendix I or other tests of comparable sensitivity for which data are available from one or more other laboratories.
   a. A good indication of the sensitivity of the test is available. This helps ensure that the endpoint in laboratory dilution water is close to the CMC and/or CCC and aids in the selection of concentrations of the metal to be used in the rangefinding and/or definitive toxicity tests in laboratory dilution water. Tests with other species such as species that occur at the site may be used, but it is sometimes more difficult to obtain, hold, and test such species.

b. When a WER is determined and used, the results of the tests in laboratory dilution water provide the connection between the data used in the derivation of the national criterion and the data obtained in site water, i.e., the results in laboratory dilution water are a vital link in the derivation and use of a WER. It is, therefore, important to be able to judge the quality of the results in laboratory dilution water. Comparison of results with data from other laboratories evaluates all aspects of the test methodology simultaneously, but for the determination of WERs, the most important aspect is the quality of the laboratory dilution water because the dilution water is the most important difference between the two side-by-side tests from which the WER is calculated. Thus, two tests **must** be conducted for which data are available on the metal of concern in a laboratory dilution water from at least one other laboratory. If both the primary and secondary tests are ones for which acceptable data are available from at least one other laboratory, these are the only two tests that have to be conducted. If, however, the primary and/or secondary tests are ones for which no results are already available for the metal of concern from another laboratory, the first or second time a WER is determined at least two additional tests **must** be conducted in the laboratory dilution water in addition to the tests that are conducted for the determination of WERs (see sections F.5 and I.5).
   1) For the determination of a WER, data are not required for a reference toxicant with either the primary test or the secondary test because the above requirement provides similar data for the metal for which the WER is actually being determined.
   2) See Section I.5 concerning interpretation of the results of these tests before additional tests are conducted.

D. Acquiring and Acclimating Test Organisms

1. The test organisms should be obtained, cultured, held, acclimated, fed, and handled as recommended by the U.S. EPA (1993a,b,c) and/or by ASTM (1993a,b,c,d,e). All test organisms **must** be acceptably acclimated to a laboratory dilution water that satisfies the requirements given in sections F.3 and F.4; an appropriate number of the organisms may be randomly or impartially removed from the laboratory dilution water and placed in the site water when it becomes available in order to acclimate the organisms to the site water for a while just before the tests are begun.

2. The organisms used in a pair of side-by-side tests **must** be drawn from the same population and tested under identical conditions.

E. Collecting and Handling Upstream Water and Effluent

1. Upstream water will usually be mixed with effluent to prepare simulated downstream water. Upstream water may also be used as a site water if a WER is to be determined using upstream water in addition to or instead of determining a WER using downstream water. The samples of upstream water **must** be representative; they **must not** be unduly affected by recent runoff events (or other erosion or resuspension events) that cause higher levels of TSS than would normally be present, unless there is particular concern about such conditions.

2. The sample of effluent used in the determination of a WER **must** be representative; it **must** be collected during a period when the discharger is operating normally. Selection of the date and time of sampling of the effluent should take into account the discharge pattern of the discharger. It might be appropriate to collect effluent samples during the middle of the week to allow for reestablishment of steady-state conditions after shutdowns for weekends and holidays; alternatively, if end-of-the-week slug discharges are routine, they should probably be evaluated. As mentioned above, because the variability of the effluent might contribute substantially to the variability of the WERs, it might be desirable to obtain and store more than one sample of the effluent when WERs are to be determined in case an unusual WER is obtained with the first sample used.

3. When samples of site water and effluent are collected for the determination of the WERs with the primary test, there **must** be <u>at least three weeks</u> between one sampling event and the next. It is desirable to obtain samples in at least two different seasons and/or during times of probable differences in the characteristics of the site water and/or effluent.

4. Samples of upstream water and effluent **must** be collected, transported, handled, and stored as recommended by the U.S. EPA (1993a). For example, samples of effluent should usually be composites, but grab samples are acceptable if the residence time of the effluent is sufficiently long. A sufficient volume should be obtained so that some can be stored for additional testing or analyses if an unusual WER is obtained. Samples **must** be stored at 0 to 4°C in the dark with no air space in the sample container.

5. At the time of collection, the flow of both the upstream water and the effluent **must** be either measured or estimated by means of correlation with a nearby U.S.G.S. gauge, the pH of both upstream water and effluent **must** be measured, and samples of both upstream water and effluent should be filtered for measurement of dissolved metals. Hardness, TSS, TOC, and total recoverable and dissolved metal **must** be measured in both the effluent and the upstream water. Any other water quality characteristics, such as total dissolved solids (TDS) and conductivity, that are monitored monthly or more often by the permittee and reported in the Discharge Monitoring Report **must** also be measured. These and the other measurements provide information concerning the representativeness of the samples and the variability of the upstream water and effluent.

6. "Chain of custody" procedures (U.S. EPA 1991b) should be used for all samples of site water and effluent, especially if the data might be involved in a legal proceeding.

7. Tests **must** be begun within 36 hours after the collection of the samples of the effluent and/or the site water, except that tests may be begun more than 36 hours after the collection of the samples if it would require an inordinate amount of resources to transport the samples to the laboratory and begin the tests within 36 hours.

8. If acute and/or chronic tests are to be conducted with daphnids and if the sample of the site water contains predators, the site water **must** be filtered through a 37-μm sieve or screen to remove predators.

F. Laboratory Dilution Water

1. The laboratory dilution water **must** satisfy the requirements given by U.S. EPA (1993a,b,c) or ASTM (1993a,b,c,d,e). The laboratory dilution water **must** be a ground water, surface water, reconstituted water, diluted mineral water, or dechlorinated tap water that has been demonstrated to be acceptable to aquatic organisms. If a surface water is used for acute or chronic tests with daphnids and if predators are observed in the sample of the water, it **must** be filtered through a 37-μm sieve or screen to remove the predators. Water prepared by such treatments as deionization and reverse osmosis **must not** be used as the laboratory dilution water unless salts, mineral water, hypersaline brine, or sea salts are added as recommended by U.S. EPA (1993a) or ASTM (1993a).

2. The concentrations of both TOC and TSS **must** be less than 5 mg/L.

3. The hardness of the laboratory dilution water should be between 50 and 150 mg/L and **must** be between 40 and 220 mg/L. If the criterion for the metal is hardness-dependent, the hardness of the laboratory dilution water **must not** be above the hardness of the site water, unless the hardness of the site water is below 50 mg/L.

4. The alkalinity and pH of the laboratory dilution water **must** be appropriate for its hardness; values for alkalinity and pH that are appropriate for some hardnesses are given by U.S. EPA (1993a) and ASTM (1993a); other corresponding values should be determined by interpolation. Alkalinity should be adjusted using sodium bicarbonate, and pH should be adjusted using aeration, sodium hydroxide, and/or sulfuric acid.

5. It would seem reasonable that, before any samples of site water or effluent are collected, the toxicity tests that are to be conducted in the laboratory dilution water for comparison with results of the same tests from other laboratories (see sections C.3.b and I.5) should be conducted. These should be performed at the hardness, alkalinity, and pH specified in sections F.3 and F.4.

G. Conducting Tests

1. There **must** be no differences between the side-by-side tests other than the composition of the dilution water, the concentrations of metal tested, and possibly the water in which the test organisms are acclimated just prior to the beginning of the tests.

2. More than one test using site water may be conducted side-by-side with a test using laboratory dilution water; the one test in laboratory dilution water will be used in the calculation of several WERs, which means that it is very important that that one test be acceptable.

3. Facilities for conducting toxicity tests should be set up and test chambers should be selected and cleaned as recommended by the U.S. EPA (1993a,b,c) and/or ASTM (1993a,b,c,d,e).

4. A stock solution should be prepared using an inorganic salt that is highly soluble in water.
   a. The salt does not have to be one that was used in tests that were used in the derivation of the national criterion. Nitrate salts are generally acceptable;

chloride and sulfate salts of many metals are also acceptable (see Appendix J). It is usually desirable to avoid use of a hygroscopic salt. The salt used should meet A.C.S. specifications for reagent-grade, if such specifications are available; use of a better grade is usually not worth the extra cost. No salt should be used until information concerning safety and handling has been read.

b. The stock solution may be acidified (using metal-free nitric acid) only as necessary to get the metal into solution.

c. The same stock solution **must** be used to add metal to all tests conducted at one time.

5. For tests suggested in Appendix I, the appendix presents the recommended duration and whether the static or renewal technique should be used; additional information is available in the references cited in the appendix. Regardless of whether or not or how often test solutions are renewed when these tests are conducted for other purposes, the following guidance applies to all tests that are conducted for the determination of WERs:
   a. The renewal technique **must** be used for tests that last longer than 48 hr.
   b. If the concentration of dissolved metal decreases by more than 50 % in 48 hours in static or renewal tests, the test solutions **must** be renewed every 24 hours. Similarly, if the concentration of dissolved oxygen becomes too low, the test solutions **must** be renewed every 24 hours. If one test in a pair of tests is a renewal test, both tests **must** be renewal tests.
   c. When test solutions are to be renewed, the new test solutions **must** be prepared from the original unspiked effluent and water samples that have been stored at 0 to 4°C in the dark with no air space in the sample container.
   d. The static technique may be used for tests that do not last longer than 48 hours unless the above specifications require use of the renewal technique.

   If a test is used that is not suggested in Appendix I, the duration and technique recommended for a comparable test should be used.

6. Recommendations concerning temperature, loading, feeding, dissolved oxygen, aeration, disturbance, and controls given by the U.S. EPA (1993a,b,c) and/or ASTM (1993a,b,c,d,e) **must** be followed. The procedures that are used **must** be used in both of the side-by-side tests.

7. To aid in the selection of the concentrations of metals that should be used in the test solutions in site water, a static rangefinding test should be conducted for 8 to 96

hours, using a dilution factor of 10 (or 0.1) or 3.2 (or 0.32) increasing from about a factor of 10 below the value of the endpoint given in the criteria document for the metal or in Appendix I of this document for tests with newly hatched fathead minnows. If the test is not in the criteria document and no other data are available, a mean acute value or other data for a taxonomically similar species should be used as the predicted value. This rangefinding test will provide information concerning the concentrations that should be used to bracket the endpoint in the definitive test and will provide information concerning whether the control survival will be acceptable. If dissolved metal is measured in one or more treatments at the beginning and end of the rangefinding test, these data will indicate whether the concentration should be expected to decrease by more than 50 % during the definitive test. The rangefinding test may be conducted in either of two ways:

a. It may be conducted using the samples of effluent and site water that will be used in the definitive test. In this case, the duration of the rangefinding test should be as long as possible within the limitation that the definitive test **must** begin within 36 hours after the samples of effluent and/or site water were collected, except as per section E.7.
b. It may be conducted using one set of samples of effluent and upstream water with the definitive tests being conducted using samples obtained at a later date. In this case the rangefinding test might give better results because it can last longer, but there is the possibility that the quality of the effluent and/or site water might change. Chemical analyses for hardness and pH might indicate whether any major changes occurred from one sample to the next.

Rangefinding tests are especially desirable before the first set of toxicity tests. It might be desirable to conduct rangefinding tests before each individual determination of a WER to obtain additional information concerning the effluent, dilution water, organisms, etc., before each set of side-by-side tests are begun.

8. Several considerations are important in the selection of the dilution factor for definitive tests. Use of concentrations that are close together will reduce the uncertainty in the WER but will require more concentrations to cover a range within which the endpoints might occur. Because of the resources necessary to determine a WER, it is important that endpoints in both dilution waters be obtained whenever a set of side-by-side tests are conducted. Because static and renewal tests can be used to determine WERs, it is relatively easy to use more treatments than would be used in flow-through tests.

The dilution factor for total recoverable metal **must** be between 0.65 and 0.99, and the recommended factor is 0.7. Although factors between 0.75 and 0.99 may be used, their use will probably not be cost-effective. Because there is likely to be more uncertainty in the predicted value of the endpoint in site water, 6 or 7 concentrations are recommended in the laboratory dilution water, and 8 or 9 in the simulated downstream water, at a dilution factor of 0.7. It might be desirable to use even more treatments in the first of the WER determinations, because the design of subsequent tests can be based on the results of the first tests if the site water, laboratory dilution water, and test organisms do not change too much. The cost of adding treatments can be minimized if the concentration of metal is measured only in samples from treatments that will be used in the calculation of the endpoint.

9. Each test **must** contain a dilution-water control. The number of test organisms intended to be exposed to each treatment, including the controls, **must** be at least 20. It is desirable that the organisms be distributed between two or more test chambers per treatment. If test organisms are not randomly assigned to the test chambers, they **must** be assigned impartially (U.S. EPA 1993a; ASTM 1993a) between all test chambers for a pair of side-by-side tests. For example, it is not acceptable to assign 20 organisms to one treatment, and then assign 20 organisms to another treatment, etc. Similarly, it is not acceptable to assign all the organisms to the test using one of the dilution waters and then assign organisms to the test using the other dilution water. The test chambers should be assigned to location in a totally random arrangement or in a randomized block design.

10. For the test using site water, one of the following procedures should be used to prepare the test solutions for the test chambers and the "chemistry controls" (see section H.1):
    a. Thoroughly mix the sample of the effluent and place the same known volume of the effluent in each test chamber; add the necessary amount of metal, which will be different for each treatment; mix thoroughly; let stand for 2 to 4 hours; add the necessary amount of upstream water to each test chamber; mix thoroughly; let stand for 1 to 3 hours.
    b. Add the necessary amount of metal to a large sample of the effluent and also maintain an unspiked sample of the effluent; perform serial dilution using a graduated cylinder and the well-mixed spiked and unspiked samples of the effluent; let stand for 2 to 4 hours; add the necessary amount of upstream water to each test chamber; mix thoroughly; let stand for 1 to 3 hours.

c. Prepare a large volume of simulated downstream water by mixing effluent and upstream water in the desired ratio; place the same known volume of the simulated downstream water in each test chamber; add the necessary amount of metal, which will be different for each treatment; mix thoroughly and let stand for 1 to 3 hours.
d. Prepare a large volume of simulated downstream water by mixing effluent and upstream water in the desired ratio; divide it into two portions; prepare a large volume of the highest test concentration of metal using one portion of the simulated downstream water; perform serial dilution using a graduated cylinder and the well-mixed spiked and unspiked samples of the simulated downstream water; let stand for 1 to 3 hours.

Procedures "a" and "b" allow the metal to equilibrate somewhat with the effluent before the solution is diluted with upstream water.

11. For the test using the laboratory dilution water, either of the following procedures may be used to prepare the test solutions for the test chambers and the "chemistry controls" (see section H.1):
    a. Place the same known volume of the laboratory dilution water in each test chamber; add the necessary amount of metal, which will be different for each treatment; mix thoroughly; let stand for 1 to 3 hours.
    b. Prepare a large volume of the highest test concentration in the laboratory dilution water; perform serial dilution using a graduated cylinder and the well-mixed spiked and unspiked samples of the laboratory dilution water; let stand for 1 to 3 hours.

12. The test organisms, which have been acclimated as per section D.1, **must** be added to the test chambers for the site-by-side tests at the same time. The time at which the test organisms are placed in the test chambers is defined as the beginning of the tests, which **must** be within 36 hours of the collection of the samples, except as per section E.7.

13. Observe the test organisms and record the effects and symptoms as specified by the U.S. EPA (1993a,b,c) and/or ASTM (1993a,b,c,d,e). Especially note whether the effects, symptoms, and time course of toxicity are the same in the side-by-side tests.

14. Whenever solutions are renewed, sufficient solution should be prepared to allow for chemical analyses.

H. Chemical and Other Measurements

1. To reduce the possibility of contamination of test solutions before or during tests, thermometers and probes for measuring pH and dissolved oxygen **must not** be placed in test chambers that will provide data concerning effects on test organisms or data concerning the concentration of the metal. Thus measurements of pH, dissolved oxygen, and temperature before or during a test **must** be performed either on "chemistry controls" that contain test organisms and are fed the same as the other test chambers or on aliquots that are removed from the test chambers. The other measurements may be performed on the actual test solutions at the beginning and/or end of the test or the renewal.

2. Hardness (in fresh water) or salinity (in salt water), pH, alkalinity, TSS, and TOC **must** be measured on the upstream water, the effluent, the simulated and/or actual downstream water, and the laboratory dilution water. Measurement of conductivity and/or total dissolved solids (TDS) is recommended in fresh water.

3. Dissolved oxygen, pH, and temperature **must** be measured during the test at the times specified by the U.S. EPA (1993a,b,c) and/or ASTM (1993a,b,c,d,e). The measurements **must** be performed on the same schedule for both of the side-by-side tests. Measurements **must** be performed on both the chemistry controls and actual test solutions at the end of the test.

4. Both total recoverable and dissolved metal **must** be measured in the upstream water, the effluent, and appropriate test solutions for each of the tests.
   a. The analytical measurements should be sufficiently sensitive and precise that variability in analyses will not greatly increase the variability of the WERs. If the detection limit of the analytical method that will be used to determine the metal is greater than one-tenth of the CCC or CMC that is to be adjusted, the analytical method should probably be improved or replaced (see Appendix C). If additional sensitivity is needed, it is often useful to separate the metal from the matrix because this will simultaneously concentrate the metal and remove interferences. Replicate analyses should be performed if necessary to reduce the impact of analytical variability.
      1) EPA methods (U.S. EPA 1983b,1991c) should usually be used for both total recoverable and dissolved measurements, but in some cases alternate methods might have to be used in order to achieve the necessary sensitivity. Approval for use of

alternate methods is to be requested from the appropriate regulatory authority.

b. All measurements of metals **must** be performed using appropriate QA/QC techniques. <u>Clean techniques for obtaining, handling, storing, preparing, and analyzing the samples should be used when necessary to achieve blanks that are sufficiently low (see Appendix C).</u>

c. Rather than measuring the metal in all test solutions, it is often possible to store samples and then analyze only those that are needed to calculate the results of the toxicity tests. For dichotomous data (e.g., either-or data; data concerning survival), the metal in the following **must** be measured:
   1) all concentrations in which some, but not all, of the test organisms were adversely affected.
   2) the highest concentration that did not adversely affect any test organisms.
   3) the lowest concentration that adversely affected all of the test organisms.
   4) the controls.

   For data that are not dichotomous (i.e., for count and continuous data), the metal in the controls and in the treatments that define the concentration-effect curve **must** be measured; measurement of the concentrations of metals in other treatments is desirable.

d. In each treatment in which the concentration of metal is to be measured, <u>both the total recoverable and dissolved concentrations **must** be measured</u>:
   1) Samples **must** be taken for measurement of total recoverable metal once for a static test, and once for each renewal for renewal tests; in renewal tests, the samples are to be taken after the organisms have been transferred to the new test solutions. When total recoverable metal is measured in a test chamber, the whole solution in the chamber **must** be mixed before the sample is taken for analysis; the solution in the test chamber **must not** be acidified before the sample is taken. The sample **must** be acidified after it is placed in the sample container.
   2) Dissolved metal **must** be measured at the beginning and end of each static test; in a renewal test, the dissolved metal **must** be measured at the beginning of the test and just before the solution is renewed the first time. When dissolved metal is measured in a test chamber, the whole solution in the test chamber **must** be mixed before a sufficient amount is removed for filtration; the solution in the test chamber **must not** be acidified before the sample is taken. The sample **must** be filtered within one hour after it is taken, and the filtrate **must** be acidified after filtration.

5. Replicates, matrix spikes, and other QA/QC checks **must** be performed as required by the U.S. EPA (1983a,1991c).

I. Calculating and Interpreting the Results

1. To prevent roundoff error in subsequent calculations, at least four significant digits **must** be retained in all endpoints, WERs, and FWERs. This requirement is not based on mathematics or statistics and does not reflect the precision of the value; its purpose is to minimize concern about the effects of rounding off on a site-specific criterion. All of these numbers are intermediate values in the calculation of permit limits and should not be rounded off as if they were values of ultimate concern.

2. Evaluate the acceptability of each toxicity test individually.
   a. If the procedures used deviated from those specified above, particularly in terms of acclimation, randomization, temperature control, measurement of metal, and/or disease or disease-treatment, the test should be rejected; if deviations were numerous and/or substantial, the test **must** be rejected.
   b. Most tests are unacceptable if more than 10 percent of the organisms in the controls were adversely affected, but the limit is higher for some tests; for the tests recommended in Appendix I, the references given should be consulted.
   c. If an LC50 or EC50 is to be calculated:
      1) The percent of the organisms that were adversely affected **must** have been less than 50 percent, and should have been less than 37 percent, in at least one treatment other than the control.
      2) In laboratory dilution water the percent of the organisms that were adversely affected **must** have been greater than 50 percent, and should have been greater than 63 percent, in at least one treatment. In site water the percent of the organisms that were adversely affected should have been greater than 63 percent in at least one treatment. (The LC50 or EC50 may be a "greater than" or "less than" value in site water, but not in laboratory dilution water.)
      3) If there was an inversion in the data (i.e., if a lower concentration killed or affected a greater percentage of the organisms than a higher concentration), it **must not** have involved more than two concentrations that killed or affected between 20 and 80 percent of the test organisms.

      If an endpoint other than an LC50 or EC50 is used or if Abbott's formula is used, the above requirements will have to be modified accordingly.

d. Determine whether there was anything unusual about the test results that would make them questionable.
e. If solutions were not renewed every 24 hours, the concentration of dissolved metal **must not** have decreased by more than 50 percent from the beginning to the end of a static test or from the beginning to the end of a renewal in a renewal test in test concentrations that were used in the calculation of the results of the test.

3. Determine whether the effects, symptoms, and time course of toxicity was the same in the side-by-side tests in the site water and the laboratory dilution water. For example, did mortality occur in one acute test, but immobilization in the other? Did most deaths occur before 24 hours in one test, but after 24 hours in the other? In sublethal tests, was the most sensitive effect the same in both tests? If the effects, symptoms, and/or time course of toxicity were different, it might indicate that the test is questionable or that additivity, synergism, or antagonism occurred in site water. Such information might be particularly useful when comparing tests that produced unusually low or high WERs with tests that produced moderate WERs.

4. Calculate the results of each test:
   a. If the data for the most sensitive effect are dichotomous, the endpoint **must** be calculated as a LC50, EC50, LC25, EC25, etc., using methods described by the U.S. EPA (1993a) or ASTM (1993a). If two or more treatments affected between 0 and 100 percent in both tests in a side-by-side pair, probit analysis **must** be used to calculate results of both tests, unless the probit model is rejected by the goodness of fit test in one or both of the acute tests. If probit analysis cannot be used, either because fewer than two percentages are between 0 and 100 percent or because the model does not fit the data, computational interpolation **must** be used (see Figure 5); graphical interpolation **must not** be used.
      1) The same endpoint (LC50, EC25, etc.) and the same computational method **must** be used for both tests used in the calculation of a WER.
      2) The selection of the percentage used to define the endpoint might be influenced by the percent effect that occurred in the tests and the correspondence with the CCC and/or CMC.
      3) If no treatment killed or affected more than 50 percent of the test organisms and the test was otherwise acceptable, the LC50 or EC50 should be reported to be greater than the highest test concentration.

4) If no treatment other than the control killed or affected less than 50 percent of the test organisms and the test was otherwise acceptable, the LC50 or EC50 should be reported to be less than the lowest test concentration.

b. If the data for the most sensitive effect are not dichotomous, the endpoint **must** be calculated using a regression-type method (Hoekstra and Van Ewijk 1993; Stephan and Rogers 1985), such as linear interpolation (U.S. EPA 1993b,c) or a nonlinear regression method (Barnthouse et al. 1987; Suter et al. 1987; Bruce and Versteeg 1992). The selection of the percentage used to define the endpoint might be influenced by the percent effect that occurred in the tests and the correspondence with the CCC and/or CMC. The endpoints in the side-by-side tests **must** be based on the same amount of the same adverse effect so that the WER is a ratio of identical endpoints. The same computational method **must** be used for both tests used in the calculation of the WER.

c. Both total recoverable and dissolved results should be calculated for each test.

d. Results should be based on the time-weighted average measured metal concentrations (see Figure 6).

5. The acceptability of the laboratory dilution water **must** be evaluated by comparing results obtained with two sensitive tests using the laboratory dilution water with results that were obtained using a comparable laboratory dilution water in one or more other laboratories (see sections C.3.b and F.5).

a. If, after taking into account any known effect of hardness on toxicity, the new values for the endpoints of <u>both</u> of the tests are (1) more than a factor of 1.5 higher than the respective means of the values from the other laboratories or (2) more than a factor of 1.5 lower than the respective means of values from the other laboratories or (3) lower than the respective lowest values available from other laboratories or (4) higher than the respective highest values available from other laboratories, the new and old data **must** be carefully evaluated to determine whether the laboratory dilution water used in the WER determination was acceptable. For example, there might have been an error in the chemical measurements, which might mean that the results of all tests performed in the WER determination need to be adjusted and that the WER would not change. It is also possible that the metal is more or less toxic in the laboratory dilution water used in the WER determination. Further, if the new data were based on measured concentrations but the old data were based on nominal concentrations, the new data should probably be considered to be better than the old. Evaluation of results of any other toxicity tests on the same or a different metal using the same laboratory dilution water might be useful.

b. If, after taking into account any known effect of hardness on toxicity, the new values for the endpoints of the two tests are not either both higher or both lower in comparison than data from other laboratories (as per section a above) and if both of the new values are within a factor of 2 of the respective means of the previously available values or are within the ranges of the values, the laboratory dilution water used in the WER determination is acceptable.
c. A control chart approach may be used if sufficient data are available.
d. If the comparisons do not indicate that the laboratory dilution water, test method, etc., are acceptable, the tests probably should be considered unacceptable, unless other toxicity data are available to indicate that they are acceptable.

Comparison of results of tests between laboratories provides a check on all aspects of the test procedure; the emphasis here is on the quality of the laboratory dilution water because all other aspects of the side-by-side tests on which the WER is based **must** be the same, except possibly for the concentrations of metal used and the acclimation just prior to the beginning of the tests.

6. If all the necessary tests and the laboratory dilution water are acceptable, a WER **must** be calculated by dividing the endpoint obtained using site water by the endpoint obtained using laboratory dilution water.
   a. If both a primary test and a secondary test were conducted using both waters, WERs **must** be calculated for both tests.
   b. Both total recoverable and dissolved WERs **must** be calculated.
   c. If the detection limit of the analytical method used to measure the metal is above the endpoint in laboratory dilution water, the detection limit **must** be used as the endpoint, which will result in a lower WER than would be obtained if the actual concentration had been measured. If the detection limit of the analytical method used is above the endpoint in site water, a WER cannot be determined.

7. Investigation of the WER.
   a. The results of the chemical measurements of hardness, alkalinity, pH, TSS, TOC, total recoverable metal, dissolved metal, etc., on the effluent and the upstream water should be examined and compared with previously available values for the effluent and upstream water,

respectively, to determine whether the samples were representative and to get some indication of the variability in the composition, especially as it might affect the toxicity of the metal and the WER, and to see if the WER correlates with one or more of the measurements.

b. The WERs obtained with the primary and secondary tests should be compared to determine whether the WER obtained with the secondary test confirmed the WER obtained with the primary test. Equally sensitive tests are expected to give WERs that are similar (e.g., within a factor of 3), whereas a test that is less sensitive will probably give a smaller WER than a more sensitive test (see Appendix D). Thus a WER obtained with a primary test is considered confirmed if either or both of the following are true:
   1) the WERs obtained with the primary and secondary tests are within a factor of 3.
   2) the test, regardless of whether it is the primary or secondary test, that gives a higher endpoint in the laboratory dilution water also gives the larger WER.

   If the WER obtained with the secondary test does not confirm the WER obtained with the primary test, the results should be investigated. In addition, WERs probably should be determined using both tests the next time samples are obtained and it would be desirable to determine a WER using a third test. It is also important to evaluate what the results imply about the protectiveness of any proposed site-specific criterion.

c. If the WER is larger than 5, it should be investigated.
   1) If the endpoint obtained using the laboratory dilution water was lower than previously reported lowest value or was more than a factor of two lower than an existing Species Mean Acute Value in a criteria document, additional tests in the laboratory dilution water are probably desirable.
   2) If a total recoverable WER was larger than 5 but the dissolved WER was not, is the metal one whose WER is likely to be affected by TSS and/or TOC and was the concentration of TSS and/or TOC high? Was there a substantial difference between the total recoverable and dissolved concentrations of the metal in the downstream water?
   3) If both the total recoverable and dissolved WERs were larger than 5, is it likely that there is nontoxic dissolved metal in the downstream water?

d. The adverse effects and the time-course of effects in the side-by-side tests should be compared. If they are different, it might indicate that the site-water test is questionable or that additivity, synergism, or antagonism occurred in the site water. This might be especially important if the WER obtained with the secondary test did not confirm the WER obtained with the primary test or if the WER was very large or small.

8. If at least one WER determined with the primary test was confirmed by a WER that was simultaneously determined with the secondary test, the cmcFWER and/or the cccFWER should be derived as described in section A.5.

9. All data generated during the determination of the WER should be examined to see if there are any implications for the national or site-specific aquatic life criterion.
   a. If there are data for a species for which data were not previously available or unusual data for a species for which data were available, the national criterion might need to be revised.
   b. If the primary test gives an LC50 or EC50 in laboratory dilution water that is the same as the national CMC, the resulting site-specific CMC should be similar to the LC50 that was obtained with the primary test using downstream water. Such relationships might serve as a check on the applicability of the use of WERs.
   c. If data indicate that the site-specific criterion would not adequately protect a critical species, the site-specific criterion probably should be lowered.

J. Reporting the Results

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

1. Name(s) of the investigator(s), name and location of the laboratory, and dates of initiation and termination of the tests.
2. A description of the laboratory dilution water, including source, preparation, and any demonstrations that an aquatic species can survive, grow, and reproduce in it.
3. The name, location, and description of the discharger, a description of the effluent, and the design flows of the effluent and the upstream water.
4. A description of each sampling station, date, and time, with an explanation of why they were selected, and the flows of the upstream water and the effluent at the time the samples were collected.
5. The procedures used to obtain, transport, and store the samples of the upstream water and the effluent.
6. Any pretreatment, such as filtration, of the effluent, site water, and/or laboratory dilution water.
7. Results of all chemical and physical measurements on upstream water, effluent, actual and/or simulated downstream water, and laboratory dilution water, including hardness (or salinity), alkalinity, pH, and concentrations of total recoverable metal, dissolved metal, TSS, and TOC.

8. 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. Source and grade of the metallic salt, and how the stock solution was prepared, including any acids or bases used.
10. Source of the test organisms, scientific name and how verified, age, life stage, means and ranges of weights and/or lengths, observed diseases, treatments, holding and acclimation procedures, and food.
11. The average and range of the temperature, pH, hardness (or salinity), and the concentration of dissolved oxygen (as % saturation and as mg/L) during acclimation, and the method used to measure them.
12. The following **must** be presented for each toxicity test:
    a. The average and range of the measured concentrations of dissolved oxygen, as % saturation and as mg/L.
    b. The average and range of the test temperature and the method used to measure it.
    c. The schedule for taking samples of test solutions and the methods used to obtain, prepare, and store them.
    d. A summary table of the total recoverable and dissolved concentrations of the metal in each treatment, including all controls, in which they were measured.
    e. A summary table of the values of the toxicological variable(s) for each treatment, including all controls, in sufficient detail to allow an independent statistical analysis of the data.
    f. The endpoint and the method used to calculate it.
    g. Comparisons with other data obtained by conducting the same test on the same metal using laboratory dilution water in the same and different laboratories; such data may be from a criteria document or from another source.
    h. Anything unusual about the test, any deviations from the procedures described above, and any other relevant information.
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. Comparison of results obtained with the primary and secondary tests.
15. The WER and an explanation of its calculation.

A report of the derivation of a FWER **must** include the following:

1. A report of the determination of each WER that was determined for 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 hWER calculated from the WER.
   c. the test 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.
4. A detailed explanation of how the FWER was derived from the WERs that are in the summary table.

METHOD 2: DETERMINING cccWERs FOR AREAS AWAY FROM PLUMES

Method 2 might be viewed as a simple process wherein samples of site water are obtained from locations within a large body of fresh or salt water (e.g., an ocean or a large lake, reservoir, or estuary), a WER is determined for each sample, and the FWER is calculated as the geometric mean of some or all of the WERs. In reality, Method 2 is not likely to produce useful results unless substantial resources are devoted to planning and conducting the study. Most sites to which Method 2 is applied will have long retention times, complex mixing patterns, and a number of dischargers. Because metals are persistent, the long retention times mean that the sites are likely to be defined to cover rather large areas; thus such sites will herein be referred to generically as "large sites". Despite the differences between them, all large sites require similar special considerations regarding the determination of WERs. Because Method 2 is based on samples of actual surface water (rather than simulated surface water), no sample should be taken in the vicinity of a plume and the method should be used to determine cccWERs, not cmcWERs. If WERs are to be determined for more than one metal, Appendix F should be read.

Method 2 uses many of the same methodologies as Method 1, such as those for toxicity tests and chemical analyses. Because the sampling plan is crucial to Method 2 and the plan has to be based on site-specific considerations, this description of Method 2 will be more qualitative than the description of Method 1.

Method 2 is based on use of actual surface water samples, but use of simulated surface water might provide information that is useful for some purposes:

1. It might be desirable to compare the WERs for two discharges that contain the same metal. This might be accomplished by selecting an appropriate dilution water and preparing two simulated surface waters, one that contains a known concentration of one effluent and one that contains a known concentration of the other effluent. The relative magnitude of the two WERs is likely to be more useful than the absolute values of the WERs themselves.
2. It might be desirable to determine whether the eWER for a particular effluent is additive with the WER of the site water (see Appendix G). This can be studied by determining WERs for several different known concentrations of the effluent in site water.
3. An event such as a rain might affect the WER because of a change in the water quality, but it might also reduce the WER just by dilution of refractory metal or TSS. A proportional decrease in the WER and in the concentration of the metal (such as by dilution of refractory metal) will not result in underprotection; if, however, dilution decreases the WER

proportionally more than it decreases the concentration of metal in the downstream water, underprotection is likely to occur. This is essentially a determination of whether the WER is additive when the effluent is diluted with rain water (see Appendix G).

4. An event that increases TSS might increase the total recoverable concentration of the metal and the total recoverable WER without having much effect on either the dissolved concentration or the dissolved WER.

In all four cases, the use of simulated surface water is useful because it allows for the determination of WERs using known concentrations of effluent.

An important step in the determination of any WER is to define the area to be included in the site. The major principle that should be applied when defining the area is the same for all sites: The site should be neither too small nor too large. If the area selected is too small, permit limits might be unnecessarily controlled by a criterion for an area outside the site, whereas too large an area might unnecessarily incorporate spatial complexities that are not relevant to the discharge(s) of concern and thereby unnecessarily increase the cost of determining the WER. Applying this principle is likely to be more difficult for large sites than for flowing-water sites.

Because WERs for large sites will usually be determined using actual, rather than simulated, surface water, there are five major considerations regarding experimental design and data analysis:

1. Total recoverable WERs at large sites might vary so much across time, location, and depth that they are not very useful. An assumption should be developed that an appropriately defined WER will be much more similar across time, location, and depth within the site than will a total recoverable WER. If such an assumption cannot be used, it is likely that either the FWER will have to be set equal to the lowest WER and be overprotective for most of the site or separate site-specific criteria will have to be derived for two or more sites.
   a. One assumption that is likely to be worth testing is that the dissolved WER varies much less across time, location, and depth within a site than the total recoverable WER. If the assumption proves valid, a dissolved WER can be applied to a dissolved national water quality criterion to derive a dissolved site-specific water quality criterion that will apply to the whole site.
   b. A second assumption that might be worth testing is that the WER correlates with a water quality characteristic such as TSS or TOC across time, location, and depth.
   c. Another assumption that might be worth testing is that the dissolved and/or total recoverable WER is mostly due to

nontoxic metal rather than to a water quality characteristic that reduces toxicity. If this is true and if there is variability in the WER, the WER will correlate with the concentration of metal in the site water. This is similar to the first assumption, but this one can allow use of both total recoverable and dissolved WERs, whereas the first one only allows use of a dissolved WER.

If WERs are too variable to be useful and no way can be found to deal with the variability, additional sampling will probably be required in order to develop a WER and/or a site-specific water quality criterion that is either (a) spatially and/or temporally dependent or (b) constant and environmentally conservative for nearly all conditions.

2. An experimental design should be developed that tests whether the assumption is of practical value across the range of conditions that occur at different times, locations, and depths within the site. Each design has to be formulated individually to fit the specific site. The design should try to take into account the times, locations, and depths at which the extremes of the physical, chemical, and biological conditions occur within the site, which will require detailed information concerning the site. In addition, the experimental design should balance available resources with the need for adequate sampling.
   a. Selection of the number and timing of sampling events should take into account seasonal, weekly, and daily considerations. Intensive sampling should occur during the two most extreme seasons, with confirmatory sampling during the other two seasons. Selection of the day and time of sample collection should take into account the discharge schedules of the major industrial and/or municipal discharges. For example, it might be appropriate to collect samples during the middle of the week to allow for reestablishment of steady-state conditions after shutdowns for weekends and holidays; alternatively, end-of-the-week slug discharges are routine in some situations. In coastal sites, the tidal cycle might be important if facilities discharge, for example, over a four-hour period beginning at slack high tide. Because the highest concentration of effluent in the surface water probably occurs at ebb tide, determination of WERs using site water samples obtained at this time might result in inappropriately large WERs that would result in underprotection at other times; samples with unusually large WERs might be especially useful for testing assumptions. The importance of each consideration should be determined on a case-by-case basis.
   b. Selection of the number and locations of stations to be sampled within a sampling event should consider the site as a whole and take into account sources of water and discharges, mixing patterns, and currents (and tides in coastal areas). If the site has been adequately

characterized, an acceptable design can probably be developed using existing information concerning (1) sources of the metal and other pollutants and (2) the spatial and temporal distribution of concentrations of the metal and water quality factors that might affect the toxicity of the metal. Samples should not be taken within or near mixing zones or plumes of dischargers; dilution models (U.S. EPA 1993) and dye dispersion studies (Kilpatrick 1992) can indicate areas that should definitely be avoided. Maps, current charts, hydrodynamic models, and water quality models used to allocate waste loads and derive permit limits are likely to be helpful when determining when and where to obtain site-water samples. Available information might provide an indication of the acceptability of site water for testing selected species. The larger and more complex the site, the greater the number of sampling locations that will be needed.

c. In addition to determining the horizontal location of each sampling station, the vertical location (i.e., depth) of the sampling point needs to be selected. Known mixing regimes, the presence of vertical stratification of TSS and/or salinity, concentration of metal, effluent plumes, tolerance of test species, and the need to obtain samples of site water that span the range of site conditions should be considered when selecting the depth at which the sample is to be taken. Some decisions concerning depth cannot be made until information is obtained at the time of sampling; for example, a conductivity meter, salinometer, or transmissometer might be useful for determining where and at what depth to collect samples. Turbidity might correlate with TSS and both might relate to the toxicity of the metal in site water; salinity can indicate whether the test organisms and the site water are compatible.

Because each site is unique, specific guidance cannot be given here concerning either the selection of the appropriate number and locations of sampling stations within a site or the frequency of sampling. All available information concerning the site should be utilized to ensure that the times, locations, and depths of samples span the range of water quality characteristics that might affect the toxicity of the metal:

a. High and low concentrations of TSS.
b. High and low concentrations of effluents.
c. Seasonal effects.
d. The range of tidal conditions in saltwater situations.

The sampling plan should provide the data needed to allow an evaluation of the usefulness of the assumption(s) that the experimental design is intended to test. Statisticians should play a key role in experimental design and data analysis, but professional judgment that takes into account pertinent biological, chemical, and toxicological considerations is at least as important as rigorous statistical analysis when interpreting the data and determining the degree to which the data correspond to the assumption(s).

3. The details of each sampling design should be formulated with the aid of people who understand the site and people who have a working knowledge of WERs. Because of the complexity of designing a WER study for large sites, the design team should utilize the combined expertise and experience of individuals from the appropriate EPA Region, states, municipalities, dischargers, environmental groups, and others who can constructively contribute to the design of the study. Building a team of cooperating aquatic toxicologists, aquatic chemists, limnologists, oceanographers, water quality modelers, statisticians, individuals from other key disciplines, as well as regulators and those regulated, who have knowledge of the site and the site-specific procedures, is central to success of the derivation of a WER for a large site. Rather than submitting the workplan to the appropriate regulatory authority (and possibly the Water Management Division of the EPA Regional Office) for comment at the end, they should be members of the team from the beginning.

4. Data from one sampling event should always be analyzed prior to the next sampling event with the goal of improving the sampling design as the study progresses. For example, if the toxicity of the metal in surface water samples is related to the concentration of TSS, a water quality characteristic such as turbidity might be measured at the time of collection of water samples and used in the selection of the concentrations to be used in the WER toxicity tests in site water. At a minimum, the team that interprets the results of one sampling event and plans the next should include an aquatic toxicologist, a metals chemist, a statistician, and a modeler or other user of the data.

5. The final interpretation of the data and the derivation of the FWER(s) should be performed by a team. Sufficient data are likely to be available to allow a quantitative estimate of experimental variation, differences between species, and seasonal differences. It will be necessary to decide whether one site-specific criterion can be applied to the whole area or whether separate site-specific criteria need to be derived for two or more sites. The interpretation of the data might produce two or more alternatives that the appropriate regulatory authority could subject to a cost-benefit analysis.

Other aspects of the determination of a WER for a large site are likely to be the same as described for Method 1. For example:

a. WERs should be determined using two or more sensitive species; the suggestions given in Appendix I should be considered when selecting the tests and species to be used.

b. Chemical analyses of site water, laboratory dilution water, and test solutions should follow the requirements for the specific test used and those given in this document.
c. If tests in many surface water samples are compared to one test in a laboratory dilution water, it is very important that that one test be acceptable. Use of (1) rangefinding tests, (2) additional treatments beyond the standard five concentrations plus controls, and (3) dilutions that are functions of the known concentration-effect relationships obtained with the toxicity test and metal of concern will help ensure that the desired endpoints and WERs can be calculated.
d. Measurements of the concentrations of both total recoverable and dissolved metal should be targeted to the test concentrations whose data will be used in the calculation of the endpoints.
e. Samples of site water and/or effluent should be collected, handled, and transported so that the tests can begin as soon as is feasible.
f. If the large site is a saltwater site, the considerations presented in Appendix H ought to be given attention.

Figure 2: Calculating an Adjusted Geometric Mean

Where n = the number of experimentally determined WERs in a set, the "adjusted geometric mean" of the set is calculated as follows:

a. Take the logarithm of each of the WERs. The logarithms can be to any base, but natural logarithms (base e) are preferred for reporting purposes.
b. Calculate $\bar{x}$ = the arithmetic mean of the logarithms.
c. Calculate s = the sample standard deviation of the logarithms:

$$s = \sqrt{\frac{(x - \bar{x})^2}{n - 1}} \ .$$

d. Calculate SE = the standard error of the arithmetic mean: $SE = s/\sqrt{n}$.
e. Calculate $A = \bar{x} - (t_{0.7})(SE)$, where $t_{0.7}$ is the value of Student's t statistic for a one-sided probability of 0.70 with $n - 1$ degrees of freedom. The values of $t_{0.7}$ for some common degrees of freedom (df) are:

df	$t_{0.7}$
1	0.727
2	0.617
3	0.584
4	0.569
5	0.559
6	0.553
7	0.549
8	0.546
9	0.543
10	0.542
11	0.540
12	0.539

The values of $t_{0.7}$ for more degrees of freedom are available, for example, on page T-5 of Natrella (1966).
f. Take the antilogarithm of A.

This adjustment of the geometric mean accounts for the fact that the means of fifty percent of the sets of WERs are expected to be higher than the actual mean; using the one-sided value of t for 0.70 reduces the percentage to thirty.

Figure 3: An Example Derivation of a FWER

This example assumes that cccWERs were determined monthly using simulated downstream water that was prepared by mixing upstream water with effluent at the ratio that existed when the samples were obtained. Also, the flow of the effluent is always 10 cfs, and the design flow of the upstream water is 40 cfs. (Therefore, the downstream flow at design-flow conditions is 50 cfs.) The concentration of metal in upstream water at design flow is 0.4 ug/L, and the CCC is 2 ug/L. Each FWER is derived from the WERs and hWERs that are available through that month.

Month	eFLOW (cfs)	uFLOW (cfs)	uCONC (ug/L)	WER	HCME (ug/L)	hWER	FWER
March	10	850	0.8	5.2[a]	826.4	82.80	1.0[b]
April	10	289	0.6	6.0[c]	341.5	34.31	1.0[b]
May	10	300	0.6	5.8[c]	341.6	34.32	1.0[b]
June	10	430	0.6	5.7[c]	475.8	47.74	5.7[d]
July	10	120	0.4	7.0[c]	177.2	17.88	5.7[d]
Aug.	10	85	0.4	10.5[e]	196.1	19.77	6.80[f]
Sept.	10	40	0.4	12.0[e]	118.4	12.00	10.69[g]
Oct.	10	45	0.4	11.0[e]	119.2	12.08	10.88[g]
Nov.	10	150	0.4	7.5[c]	234.0	23.56	10.88[g]
Dec.	10	110	0.4	3.5[c]	79.6	8.12	8.12[h]
Jan.	10	180	0.6	6.9[c]	251.4	25.30	8.12[h]
Feb.	10	244	0.6	6.1[c]	295.2	29.68	8.12[h]

[a] Neither Type 1 nor Type 2; the downstream flow (i.e., the sum of the eFLOW and the uFLOW) is > 500 cfs.
[b] The total number of available Type 1 and Type 2 WERs is less than 3.
[c] A Type 2 WER; the downstream flow is between 100 and 500 cfs.
[d] No Type 1 WER is available; the FWER is the lower of the lowest Type 2 WER and the lowest hWER.
[e] A Type 1 WER; the downstream flow is between 50 and 100 cfs.
[f] One Type 1 WER is available; the FWER is the geometric mean of all Type 1 and Type 2 WERs.
[g] Two or more Type 1 WERs are available and the range is less than a factor of 5; the FWER is the adjusted geometric mean (see Figure 2) of the Type 1 WERs, because all the hWERs are higher.
[h] Two or more Type 1 WERs are available and the range is not greater than a factor of 5; the FWER is the lowest hWER because the lowest hWER is lower than the adjusted geometric mean of the Type 1 WERs.

Figure 4: Reducing the Impact of Experimental Variation

When the FWER is the lowest of, for example, three WERs, the impact of experimental variation can be reduced by conducting additional primary tests. If the endpoint of the secondary test is above the CMC or CCC to which the FWER is to be applied, the additional tests can also be conducted with the secondary test.

Month	Case 1	Case 2		
	(Primary Test)	(Primary Test)	(Primary Test)	Geometric Mean
April	4.801	4.801	3.565	4.137
May	2.552	2.552	4.190	3.270
June	9.164	9.164	6.736	7.857
Lowest	2.552			3.270

Month	Case 3			Case 4		
	(Primary Test)	(Second. Test)	Geo. Mean	(Primary Test)	(Second. Test)	Geo. Mean
April	4.801	3.163	3.897	4.801	3.163	3.897
May	2.552	5.039	3.586	2.552	2.944	2.741
June	9.164	7.110	8.072	9.164	7.110	8.072
Lowest			3.586			2.741

Case 1 uses the individual WERs obtained with the primary test for the three months, and the FWER is the lowest of the three WERs. In Case 2, duplicate primary tests were conducted in each month, so that a geometric mean could be calculated for each month; the FWER is the lowest of the three geometric means.

In Cases 3 and 4, both a primary test and a secondary test were conducted each month and the endpoints for both tests in laboratory dilution water are above the CMC or CCC to which the FWER is to be applied. In both of these cases, therefore, the FWER is the lowest of the three geometric means.

The availability of these alternatives does not mean that they are necessarily cost-effective.

Figure 5: Calculating an LC50 (or EC50) by Interpolation

When fewer than two treatments kill some but not all of the exposed test organisms, a statistically sound estimate of an LC50 cannot be calculated. Some programs and methods produce LC50s when there are fewer than two "partial kills", but such results are obtained using interpolation, not statistics. If (a) a test is otherwise acceptable, (b) a sufficient number of organisms are exposed to each treatment, and (c) the concentrations are sufficiently close together, a test with zero or one partial kill can provide all the information that is needed concerning the LC50. An LC50 calculated by interpolation should probably be called an "approximate LC50" to acknowledge the lack of a statistical basis for its calculation, but this does not imply that such an LC50 provides no useful toxicological information. If desired, the binomial test can be used to calculate a statistically sound probability that the true LC50 lies between two tested concentrations (Stephan 1977).

Although more complex interpolation methods can be used, they will not produce a more useful LC50 than the method described here. Inversions in the data between two test concentrations should be removed by pooling the mortality data for those two concentrations and calculating a percent mortality that is then assigned to both concentrations. Logarithms to a base other than 10 can be used if desired. If P1 and P2 are the percentages of the test organisms that died when exposed to concentrations C1 and C2, respectively, and if $C1 < C2$, $P1 < P2$, $0 \leq P1 \leq 50$, and $50 \leq P2 \leq 100$, then:

$$P = \frac{50 - P1}{P2 - P1}$$

$$C = Log\ C1 + P(Log\ C2 - Log\ C1)$$

$$LC50 = 10^{C}$$

If $P1 = 0$ and $P2 = 100$, $LC50 = \sqrt{(C1)(C2)}$.
If $P1 = P2 = 50$, $LC50 = \sqrt{(C1)(C2)}$.
If $P1 = 50$, $LC50 = C1$.
If $P2 = 50$, $LC50 = C2$.
If C1 = 4 mg/L, C2 = 7 mg/L, P1 = 15 %, and P2 = 100 %,
then LC50 = 5.036565 mg/L.

Besides the mathematical requirements given above, the following toxicological recommendations are given in sections G.8 and I.2:
a. $0.65 < C1/C2 < 0.99$.
b. $0 \leq P1 < 37$.
c. $63 < P2 \leq 100$.

Figure 6: Calculating a Time-Weighted Average

If a sampling plan (e.g., for measuring metal in a treatment in a toxicity test) is designed so that a series of values are obtained over time in such a way that each value contains the same amount of information (i.e., represents the same amount of time), then the most meaningful average is the arithmetic average. In most cases, however, when a series of values is obtained over time, some values contain more information than others; in these cases the most meaningful average is a time-weighted average (TWA). If each value contains the same amount of information, the arithmetic average will equal the TWA.

A TWA is obtained by multiplying each value by a weight and then dividing the sum of the products by the sum of the weights. The simplest approach is to let each weight be the duration of time that the sample represents. Except for the first and last samples, the period of time represented by a sample starts halfway to the previous sample and ends halfway to the next sample. The period of time represented by the first sample starts at the beginning of the test, and the period of time represented by the last sample ends at the end of the test. Thus for a 96-hr toxicity test, the sum of the weights will be 96 hr.

The following are hypothetical examples of grab samples taken from 96-hr flow-through tests for two common sampling regimes:

Sampling time (hr)	Conc. (mg/L)	Weight (hr)	Product (hr)(mg/L)	Time-weighted average (mg/L)
0	12	48	576	
96	14	48	672	
		96	1248	1248/96 = 13.00
0	8	12	96	
24	6	24	144	
48	7	24	168	
72	9	24	216	
96	8	12	96	
		96	720	720/96 = 7.500

When all the weights are the same, the arithmetic average equals the TWA. Similarly, if only one sample is taken, both the arithmetic average and the TWA equal the value of that sample.

The rules are more complex for composite samples and for samples from renewal tests. In all cases, however, the sampling plan can be designed so that the TWA equals the arithmetic average.

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Appendix A: Comparison of WERs Determined Using Upstream and Downstream Water

The "Interim Guidance" concerning metals (U.S. EPA 1992) made a fundamental change in the way WERs should be experimentally determined because it changed the source of the site water. The earlier guidance (U.S. EPA 1983,1984) required that upstream water be used as the site water, whereas the newer guidance (U.S. EPA 1992) recommended that downstream water be used as the site water. The change in the source of the site water was merely an acknowledgement that the WER that applies at a location in a body of water should, when possible, be determined using the water that occurs at that location.

Because the change in the source of the dilution water was expected to result in an increase in the magnitude of many WERs, interest in and concern about the determination and use of WERs increased. When upstream water was the required site water, it was expected that WERs would generally be low and that the determination and use of WERs could be fairly simple. After downstream water became the recommended site water, the determination and use of WERs was examined much more closely. It was then realized that the determination and use of upstream WERs was more complex than originally thought. It was also realized that the use of downstream water greatly increased the complexity and was likely to increase both the magnitude and the variability of many WERs. Concern about the fate of discharged metal also increased because use of downstream water might allow the discharge of large amounts of metal that has reduced or no toxicity at the end of the pipe. The probable increases in the complexity, magnitude, and variability of WERs and the increased concern about fate, increased the importance of understanding the relevant issues as they apply to WERs determined using both upstream water and downstream water.

A. Characteristics of the Site Water

The idealized concept of an upstream water is a pristine water that is relatively unaffected by people. In the real world, however, many upstream waters contain naturally occurring ligands, one or more effluents, and materials from nonpoint sources; all of these might impact a WER. If the upstream water receives an effluent containing TOC and/or TSS that contributes to the WER, the WER will probably change whenever the quality or quantity of the TOC and/or TSS changes. In such a case, the determination and use of the WER in upstream water will have some of the increased complexity associated with use of downstream water and some of the concerns associated with multiple-discharge situations (see Appendix F). The amount of complexity will depend greatly on the number and type of upstream point and nonpoint sources, the frequency and magnitude of fluctuations, and whether the WER is being determined above or below the point of complete mix of the upstream sources.

Downstream water is a mixture of effluent and upstream water, each of which can contribute to the WER, and so there are two components to a WER determined in downstream water: the effluent component and the upstream component. The existence of these two components has the following implications:

1. WERs determined using downstream water are likely to be larger and more variable than WERs determined using upstream water.
2. The effluent component should be applied only where the effluent occurs, which has implications concerning implementation.
3. The magnitude of the effluent component of a WER will depend on the concentration of effluent in the downstream water. (A consequence of this is that the effluent component will be zero where the concentration of effluent is zero, which is the point of item 2 above.)
4. The magnitude of the effluent component of a WER is likely to vary as the composition of the effluent varies.
5. Compared to upstream water, many effluents contain higher concentrations of a wider variety of substances that can impact the toxicity of metals in a wider variety of ways, and so the effluent component of a WER can be due to a variety of chemical effects in addition to such factors as hardness, alkalinity, pH, and humic acid.
6. Because the effluent component might be due, in whole or in part, to the discharge of refractory metal (see Appendix D), the WER cannot be thought of simply as being caused by the effect of water quality on the toxicity of the metal.

Dealing with downstream WERs is so much simpler if the effluent WER (eWER) and the upstream WER (uWER) are additive that it is desirable to understand the concept of additivity of WERs, its experimental determination, and its use (see Appendix G).

B. The Implications of Mixing Zones.

When WERs are determined using upstream water, the presence or absence of mixing zones has no impact; the cmcWER and the cccWER will both be determined using site water that contains zero percent of the effluent of concern, i.e., the two WERs will be determined using the same site water.

When WERs are determined using downstream water, the magnitude of each WER will probably depend on the concentration of effluent in the downstream water used (see Appendix D). The concentration of effluent in the site water will depend on where the sample is taken, which will not be the same for the cmcWER and the cccWER if there are mixing zone(s). Most, if not all, discharges have a chronic (CCC) mixing zone; many, but not all, also have an acute (CMC) mixing zone. The CMC applies at all points except those inside a CMC mixing zone; thus if there is no CMC mixing zone, the CMC applies at the end of the pipe. The CCC applies at all points outside the CCC mixing zone. It is generally assumed that if permit limits are based on a point in a stream at which both the CMC and the CCC apply, the CCC will control the permit limits, although the CMC might control if different averaging periods are appropriately taken into account. For this discussion, it will be assumed that the same design flow (e.g., 7Q10) is used for both the CMC and the CCC.

If the cmcWER is to be appropriate for use inside the chronic mixing zone, but the cccWER is to be appropriate for use outside the chronic mixing zone, the concentration of effluent that is appropriate for use in the determination of the two WERs will not be the same. Thus even if the same toxicity test is used in the determination of the cmcWER and the cccWER, the two WERs will probably be different because the concentration of effluent will be different in the two site waters in which the WERs are determined.

If the CMC is only of concern within the CCC mixing zone, the highest relevant concentration of metal will occur at the edge of the CMC mixing zone if there is a CMC mixing zone; the highest concentration will occur at the end of the pipe if there is no CMC mixing zone. In contrast, within the CCC mixing zone, the lowest cmcWER will probably occur at the outer edge of the CCC mixing zone. Thus the greatest level of protection would be provided if the cmcWER is determined using water at the outer edge of the CCC mixing zone, and then the calculated site-specific CMC is applied at the edge of the CMC mixing zone or at the end of the pipe, depending on whether there is an acute mixing zone. The cmcWER is likely to be lowest at the outer edge of the CCC mixing zone because of dilution of the effluent, but this dilution will also dilute the metal. If the cmcWER is determined at the outer edge of the CCC mixing zone but the resulting site-specific CMC is applied at the end of the pipe or at the edge of the CMC mixing zone, dilution is allowed to reduce the WER but it is not allowed to reduce the concentration of the metal. This approach is environmentally conservative, but it is probably necessary given current implementation procedures. (The situation might be more complicated if the uWER is higher than the eWER or if the two WERs are less-than-additive.)

A comparable situation applies to the CCC. Outside the CCC mixing zone, the CMC and the CCC both apply, but it is assumed that the CMC can be ignored because the CCC will be more restrictive. The cccWER should probably be determined for the complete-mix situation, but the site-specific CCC will have to be met at the edge of the CCC mixing zone. Thus dilution of the WER from the edge of the CCC mixing zone to the point of complete mix is taken into account, but dilution of the metal is not.

If there is neither an acute nor a chronic mixing zone, both the CMC and the CCC apply at the end of the pipe, but the CCC should still be determined for the complete-mix situation.

C. Definition of site.

In the general context of site-specific criteria, a "site" may be a state, region, watershed, waterbody, segment of a waterbody, category of water (e.g., ephemeral streams), etc., but the site-specific criterion is to be derived to provide adequate protection for the entire site, however the site is defined. Thus, when a site-specific criterion is derived using the Recalculation Procedure, all species that "occur at the site" need to be taken into account when deciding what species, if any, are to be deleted from the dataset. Similarly, when a site-specific criterion is derived using a WER, the WER is to be adequately protective of the entire site. If, for example, a site-specific criterion is being derived for an estuary, WERs could be determined using samples of the surface water obtained from various sampling stations, which, to avoid confusion, should not be called "sites". If all the WERs were sufficiently similar, one site-specific criterion could be derived to apply to the whole estuary. If the WERs were sufficiently different, either the lowest WER could be used to derive a site-specific criterion for the whole estuary, or the data might indicate that the estuary should be divided into two or more sites, each with its own criterion.

The major principle that should be applied when defining the area to be included in the site is very simplistic: The site should be neither too small nor too large.

1. Small sites are probably appropriate for cmcWERs, but usually are not appropriate for cccWERs because metals are persistent, although some oxidation states are not persistent and some metals are not persistent in the water column. For cccWERs, the smaller the defined site, the more likely it is that the permit limits will be controlled by a criterion for an area that is outside the site, but which could have been included in the site without substantially changing the WER or increasing the cost of determining the WER.
2. Too large an area might unnecessarily increase the cost of determining the WER. As the size of the site increases,

the spatial and temporal variability is likely to increase, which will probably increase the number of water samples in which WERs will need to be determined before a site-specific criterion can be derived.

3. Events that import or resuspend TSS and/or TOC are likely to increase the total recoverable concentration of the metal and the total recoverable WER while having a much smaller effect on the dissolved concentration and the dissolved WER. Where the concentration of dissolved metal is substantially more constant than the concentration of total recoverable metal, the site can probably be much larger for a dissolved criterion than for a total recoverable criterion. If one criterion is not feasible for the whole area, it might be possible to divide it into two or more sites with separate total recoverable or dissolved criteria or to make the criterion dependent on a water quality characteristic such as TSS or salinity.
4. Unless the site ends where one body of water meets another, at the outer edge of the site there will usually be an instantaneous decrease in the allowed concentration of the metal in the water column due to the change from one criterion to another, but there will not be an instantaneous decrease in the actual concentration of metal in the water column. The site has to be large enough to include the transition zone in which the actual concentration decreases so that the criterion outside the site is not exceeded.

It is, of course, possible in some situations that relevant distant conditions (e.g., a lower downstream pH) will necessitate a low criterion that will control the permit limits such that it is pointless to determine a WER.

When a WER is determined in upstream water, it is generally assumed that a downstream effluent will not decrease the WER. It is therefore assumed that the site can usually cover a rather large geographic area.

When a site-specific criterion is derived based on WERs determined using downstream water, the site should not be defined in the same way that it would be defined if the WER were determined using upstream water. The eWER should be allowed to affect the site-specific criterion wherever the effluent occurs, but it should not be allowed to affect the criterion in places where the effluent does not occur. In addition, insofar as the magnitude of the effluent component at a point in the site depends on the concentration of effluent, the magnitude of the WER at a particular point will depend on the concentration of effluent at that point. To the extent that the eWER and the uWER are additive, the WER and the concentration of metal in the plume will decrease proportionally (see Appendix G).

When WERs are determined using downstream water, the following considerations should be taken into account when the site is defined:

1. If a site-specific criterion is derived using a WER that applies to the complete-mix situation, the upstream edge of the site to which this criterion applies should be the point at which complete mix actually occurs. If the site to which the complete-mix WER is applied starts at the end of the pipe and extends all the way across the stream, there will be an area beside the plume that will not be adequately protected by the site-specific criterion.
2. Upstream of the point of complete mix, it will usually be protective to apply a site-specific criterion that was derived using a WER that was determined using upstream water.
3. The plume might be an area in which the concentration of metal could exceed a site-specific criterion without causing toxicity because of simultaneous dilution of the metal and the eWER. The fact that the plume is much larger than the mixing zone might not be important if there is no toxicity within the plume. As long as the concentration of metal in 100 % effluent does not exceed that allowed by the additive portion of the eWER, from a toxicological standpoint neither the size nor the definition of the plume needs to be of concern because the metal will not cause toxicity within the plume. If there is no toxicity within the plume, the area in the plume might be like a traditional mixing zone in that the concentration of metal exceeds the site-specific criterion, but it would be different from a traditional mixing zone in that the level of protection is not reduced.

Special considerations are likely to be necessary in order to take into account the eWER when defining a site related to multiple discharges (see Appendix F).

D. The variability in the experimental determination of a WER.

When a WER is determined using upstream water, the two major sources of variation in the WER are (a) variability in the quality of the site water, which might be related to season and/or flow, and (b) experimental variation. Ordinary day-to-day variation will account for some of the variability, but seasonal variation is likely to be more important.

As explained in Appendix D, variability in the concentration of nontoxic dissolved metal will contribute to the variability of both total recoverable WERs and dissolved WERs; variability in the concentration of nontoxic particulate metal will contribute to the variability in a total recoverable WER, but not to the variability in a dissolved WER. Thus, dissolved

WERs are expected to be less variable than total recoverable WERs, especially where events commonly increase TSS and/or TOC. In some cases, therefore, appropriate use of analytical chemistry can greatly increase the usefulness of the experimental determination of WERs. The concerns regarding variability are increased if an upstream effluent contributes to the WER.

When a WER is determined in downstream water, the four major sources of variability in the WER are (a) variability in the quality of the upstream water, which might be related to season and/or flow, (b) experimental variation, (c) variability in the composition of the effluent, and (d) variability in the ratio of the flows of the upstream water and the effluent. The considerations regarding the first two are the same as for WERs determined using upstream water; because of the additional sources of variability, WERs determined using downstream water are likely to be more variable than WERs determined using upstream water.

It would be desirable if a sufficient number of WERs could be determined to define the variable factors in the effluent and in the upstream water that contribute to the variability in WERs that are determined using downstream water. Not only is this likely to be very difficult in most cases, but it is also possible that the WER will be dependent on interactions between constituents of the effluent and the upstream water, i.e., the eWER and uWER might be additive, more-than-additive, or less-than-additive (see Appendix G). When interaction occurs, in order to completely understand the variability of WERs determined using downstream water, sufficient tests would have to be conducted to determine the means and variances of:

a. the effluent component of the WER.
b. the upstream component of the WER.
c. any interaction between the two components.

An interaction might occur, for example, if the toxicity of a metal is affected by pH, and the pH and/or the buffering capacity of the effluent and/or the upstream water vary considerably.

An increase in the variability of WERs decreases the usefulness of any one WER. Compensation for this decrease in usefulness can be attempted by determining WERs at more times; although this will provide more data, it will not necessarily provide a proportionate increase in understanding. Rather than determining WERs at more times, a better use of resources might be to obtain more information concerning a smaller number of specially selected occasions.

It is likely that some cases will be so complex that achieving even a reasonable understanding will require unreasonable resources. In contrast, some WERs determined using the methods presented herein might be relatively easy to understand if appropriate chemical measurements are performed when WERs are determined.

1. If the variation of the total recoverable WER is substantially greater than the variation of the comparable dissolved WER, there is probably a variable and substantial concentration of particulate nontoxic metal. It might be advantageous to use a dissolved WER just because it will have less variability than a total recoverable WER.
2. If the total recoverable and/or dissolved WER correlates with the total recoverable and/or dissolved concentration of metal in the site water, it is likely that a substantial percentage of the metal is nontoxic. In this case the WER will probably also depend on the concentration of effluent in the site water and on the concentration of metal in the effluent.

These approaches are more likely to be useful when WERs are determined using downstream water, rather than upstream water, unless both the magnitude of the WER and the concentration of the metal in the upstream water are elevated by an upstream effluent and/or events that increase TSS and/or TOC.

Both of these approaches can be applied to WERs that are determined using actual downstream water, but the second can probably provide much better information if it is used with WERs determined using simulated downstream water that is prepared by mixing a sample of the effluent with a sample of the upstream water. In this way the composition and characteristics of both the effluent and the upstream water can be determined, and the exact ratio in the downstream water is known.

Use of simulated downstream water is also a way to study the relation between the WER and the ratio of effluent to upstream water at one point in time, which is the most direct way to test for additivity of the eWER and the uWER (see Appendix G). This can be viewed as a test of the assumption that WERs determined using downstream water will decrease as the concentration of effluent decreases. If this assumption is true, as the flow increases, the concentration of effluent in the downstream water will decrease and the WER will decrease. Obtaining such information at one point in time is useful, but confirmation at one or more other times would be much more useful.

E. The fate of metal that has reduced or no toxicity.

Metal that has reduced or no toxicity at the end of the pipe might be more toxic at some time in the future. For example, metal that is in the water column and is not toxic now might become more toxic in the water column later or might move into the sediment and become toxic. If a WER allows a surface water to contain as much toxic metal as is acceptable, the WER would not be adequately protective if metal that was nontoxic when the WER was determined became toxic in the water column, unless a compensating change occurred. Studies of the fate of metals need to address not only the changes that take place, but also the rates of the changes.

Concern about the fate of discharged metal justifiably raises concern about the possibility that metals might contaminate sediments. The possibility of contamination of sediment by toxic and/or nontoxic metal in the water column was one of the concerns that led to the establishment of EPA's sediment quality criteria program, which is developing guidelines and criteria to protect sediment. A separate program was necessary because ambient water quality criteria are not designed to protect sediment. Insofar as technology-based controls and water quality criteria reduce the discharge of metals, they tend to reduce the possibility of contamination of sediment. Conversely, insofar as WERs allow an increase in the discharge of metals, they tend to increase the possibility of contamination of sediment.

When WERs are determined in upstream water, the concern about the fate of metal with reduced or no toxicity is usually small because the WERs are usually small. In addition, the factors that result in upstream WERs being greater than 1.0 usually are (a) natural organic materials such as humic acids and (b) water quality characteristics such as hardness, alkalinity, and pH. It is easy to assume that natural organic materials will not degrade rapidly, and it is easy to monitor changes in hardness, alkalinity, and pH. Thus there is usually little concern about the fate of the metal when WERs are determined in upstream water, especially if the WER is small. If the WER is large and possibly due at least in part to an upstream effluent, there is more concern about the fate of metal that has reduced or no toxicity.

When WERs are determined in downstream water, effluents are allowed to contain virtually unlimited amounts of nontoxic particulate metal and nontoxic dissolved metal. It would seem prudent to obtain some data concerning whether the nontoxic metal might become toxic at some time in the future whenever (1) the concentration of nontoxic metal is large, (2) the concentration of dissolved metal is below the dissolved national criterion but the concentration of total recoverable metal is substantially above the total recoverable national criterion, or (3) the site-specific criterion is substantially above the national criterion. It would seem appropriate to:

a. Generate some data concerning whether "fate" (i.e., environmental processes) will cause any of the nontoxic metal to become toxic due to oxidation of organic matter, oxidation of sulfides, etc. For example, a WER could be determined using a sample of actual or simulated downstream water, the sample aerated for a period of time (e.g., two weeks), the pH adjusted if necessary, and another WER determined. If aeration reduced the WER, shorter and longer periods of aeration could be used to study the rate of change.

b. Determine the effect of a change in water quality characteristics on the WER; for example, determine the effect of lowering the pH on the WER if influent lowers the pH of the downstream water within the area to which the site-specific criterion is to apply.
c. Determine a WER in actual downstream water to demonstrate whether downstream conditions change sufficiently (possibly due to degradation of organic matter, multiple dischargers, etc.) to lower the WER more than the concentration of the metal is lowered.

If environmental processes cause nontoxic metal to become toxic, it is important to determine whether the time scale involves days, weeks, or years.

Summary

When WERs are determined using downstream water, the site water contains effluent and the WER will take into account not only the constituents of the upstream water, but also the toxic and nontoxic metal and other constituents of the effluent as they exist after mixing with upstream water. The determination of the WER automatically takes into account any additivity, synergism, or antagonism between the metal and components of the effluent and/or the upstream water. The effect of calcium, magnesium, and various heavy metals on competitive binding by such organic materials as humic acid is also taken into account. Therefore, a site-specific criterion derived using a WER is likely to be more appropriate for a site than a national, state, or recalculated criterion not only because it takes into account the water quality characteristics of the site water but also because it takes into account other constituents in the effluent and upstream water.

Determination of WERs using downstream water causes a general increase in the complexity, magnitude, and variability of WERs, and an increase in concern about the fate of metal that has reduced or no toxicity at the end of the pipe. In addition, there are some other drawbacks with the use of downstream water in the determination of a WER:

1. It might serve as a disincentive for some dischargers to remove any more organic carbon and/or particulate matter than required, although WERs for some metals will not be related to the concentration of TOC or TSS.

2. If conditions change, a WER might decrease in the future. This is not a problem if the decrease is due to a reduction in nontoxic metal, but it might be a problem if the decrease is due to a decrease in TOC or TSS or an increase in competitive binding.
3. If a WER is determined when the effluent contains refractory metal but a change in operations results in the discharge of toxic metal in place of refractory metal, the site-specific criterion and the permit limits will not provide adequate protection. In most cases chemical monitoring probably will not detect such a change, but toxicological monitoring probably will.

Use of WERs that are determined using downstream water rather than upstream water increases:

1. The importance of understanding the various issues involved in the determination and use of WERs.
2. The importance of obtaining data that will provide understanding rather than obtaining data that will result in the highest or lowest WER.
3. The appropriateness of site-specific criteria.
4. The resources needed to determine a WER.
5. The resources needed to use a WER.
6. The resources needed to monitor the acceptability of the downstream water.

A WER determined using upstream water will usually be smaller, less variable, and simpler to implement than a WER determined using downstream water. Although in some situations a downstream WER might be smaller than an upstream WER, the important consideration is that a WER should be determined using the water to which it is to apply.

References

U.S. EPA. 1983. Water Quality Standards Handbook. Office of Water Regulations and Standards, Washington, DC.

U.S. EPA. 1984. Guidelines for Deriving Numerical Aquatic Site-Specific Water Quality Criteria by Modifying National Criteria. EPA-600/3-84-099 or PB85-121101. National Technical Information Service, Springfield, VA.

U.S. EPA. 1992. Interim Guidance on Interpretation and Implementation of Aquatic Life Criteria for Metals. Office of Science and Technology, Health and Ecological Criteria Division, Washington, DC.

Appendix B: The Recalculation Procedure

NOTE: The National Toxics Rule (NTR) does not allow use of the Recalculation Procedure in the derivation of a site-specific criterion. Thus nothing in this appendix applies to jurisdictions that are subject to the NTR.

The Recalculation Procedure is intended to cause a site-specific criterion to appropriately differ from a national aquatic life criterion if justified by demonstrated pertinent toxicological differences between the aquatic species that occur at the site and those that were used in the derivation of the national criterion. There are at least three reasons why such differences might exist between the two sets of species. First, the national dataset contains aquatic species that are sensitive to many pollutants, but these and comparably sensitive species might not occur at the site. Second, a species that is critical at the site might be sensitive to the pollutant and require a lower criterion. (A critical species is a species that is commercially or recreationally important at the site, a species that exists at the site and is listed as threatened or endangered under section 4 of the Endangered Species Act, or a species for which there is evidence that the loss of the species from the site is likely to cause an unacceptable impact on a commercially or recreationally important species, a threatened or endangered species, the abundances of a variety of other species, or the structure or function of the community.) Third, the species that occur 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. The procedure presented here 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 do and do not occur at the site; in effect, this procedure makes it possible to update the national aquatic life criterion.

The phrase "occur at the site" includes the species, genera, families, orders, classes, and phyla that:

a. are usually present at the site.
b. are present at the site only seasonally due to migration.
c. are present intermittently because they periodically return to or extend their ranges into the site.
d. 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.
e. 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 taxa that "occur at the site" cannot be determined merely by sampling downstream and/or upstream of the site at one point in time. "Occur at the site" does not include taxa that were once present at the site but cannot exist at the site now due to permanent physical alteration of the habitat at the site resulting from dams, etc.

The definition of the "site" can be extremely important when using the Recalculation Procedure. For example, the number of taxa that occur at the site 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 that applies outside (e.g., downstream of) the site.

Note: If the variety of aquatic invertebrates, amphibians, and fishes is so limited that species in <u>fewer than eight families</u> occur at the site, the general Recalculation Procedure is not applicable and the following special version of the Recalculation Procedure **must** be used:

1. Data **must** be available for at least one species in each of the families that occur at the site.
2. The lowest Species Mean Acute Value that is available for a species that occurs at the site **must** be used as the FAV.
3. The site-specific CMC and CCC **must** be calculated as described below in part 2 of step E, which is titled "Determination of the CMC and/or CCC".

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 in some or all of three ways:

a. Correction of data that are in the national dataset.
b. Addition of data to the national dataset.
c. Deletion of data that are in the national dataset.

All corrections and additions that have been approved by U.S. EPA 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.

The Recalculation Procedure consists of the following steps:

A. Corrections are made in the national dataset.
B. Additions are made to the national dataset.
C. The deletion process may be applied if desired.
D. If the new dataset does not satisfy the applicable Minimum Data Requirements (MDRs), additional pertinent data **must** be generated; if the new data are approved by the U.S. EPA, the Recalculation Procedure **must** be started again at step B with the addition of the new data.
E. The new CMC or CCC or both are determined.
F. A report is written.

Each step is discussed in more detail below.

A. Corrections

1. Only corrections approved by the U.S. EPA may be made.
2. 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 the datum is claimed to be suspect. If additional data are available for the same species, the U.S. EPA will decide which data should be used, based on the available guidance (U.S. EPA 1985); also, data based on measured concentrations are usually preferable to those based on nominal concentrations.
3. Two kinds of corrections are possible:
   a. The first includes those corrections that are known to and have been approved by the U.S. EPA; a list of these will be available from the U.S. EPA.
   b. The second includes those corrections that are submitted to the U.S. EPA for approval. If approved, these will be added to EPA's list of approved corrections.
4. Selective corrections are not allowed. All corrections on EPA's newest list **must** be made.

B. Additions

1. Only additions approved by the U.S. EPA may be made.
2. Two kinds of additions are possible:
   a. The first includes those additions that are known to and have been approved by the U.S. EPA; a list of these will be available from the U.S. EPA.
   b. The second includes those additions that are submitted to the U.S. EPA for approval. If approved, these will be added to EPA's list of approved additions.
3. Selective additions are not allowed. All additions on EPA's newest list **must** be made.

C. The Deletion Process

The basic principles are:
1. Additions and corrections **must** be made as per steps A and B above, before the deletion process is performed.
2. Selective deletions are not allowed. If any species is to be deleted, the deletion process described below **must** be applied to all species in the national dataset, after any necessary corrections and additions have been made to the national dataset. The deletion process specifies which species **must** be deleted and which species **must not** be deleted. Use of the deletion process is optional, but no deletions are optional when the deletion process is used.
3. Comprehensive information **must** be available concerning what species occur at the site; a species cannot be deleted based

on incomplete information concerning the species that do and do not satisfy the definition of "occur at the site".

4. Data might have to be generated before the deletion process is begun:
   a. Acceptable pertinent toxicological data **must** be available for at least one species in each class of aquatic plants, invertebrates, amphibians, and fish that contains a species that is a critical species at the site.
   b. For each aquatic plant, invertebrate, amphibian, and fish species that occurs at the site and is listed as threatened or endangered under section 4 of the Endangered Species Act, data **must** be available or be generated for an acceptable surrogate species. Data for each surrogate species **must** be used as if they are data for species that occur at the site.

   If additional data are generated using acceptable procedures (U.S. EPA 1985) and they are approved by the U.S. EPA, the Recalculation Procedure **must** be started again at step B with the addition of the new data.
5. Data might have to be generated after the deletion process is completed. Even if one or more species are deleted, there still are MDRs (see step D below) that **must** be satisfied. If the data remaining after deletion do not satisfy the applicable MDRs, additional toxicity tests **must** be conducted using acceptable procedures (U.S. EPA 1985) so that all MDRs are satisfied. If the new data are approved by the U.S. EPA, the Recalculation Procedure **must** be started again at step B with the addition of new data.
6. Chronic tests do not have to be conducted because the national Final Acute-Chronic Ratio (FACR) may be used in the derivation of 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 species that occur at the site, a site-specific FACR may be derived and used in place of the national FACR. Because a FACR was not used in the derivation of the freshwater CCC for cadmium, this CCC can only be modified the same way as a FAV; what is acceptable will depend on which species are deleted.

If any species are to be deleted, the following deletion process **must** be applied:

   a. Obtain a copy of the national dataset, i.e., tables 1, 2, and 3 in the national criteria document (see Appendix E).
   b. Make corrections in and/or additions to the national dataset as described in steps A and B above.
   c. Group all the species in the dataset taxonomically by phylum, class, order, family, genus, and species.
   d. Circle each species that satisfies the definition of "occur at the site" as presented on the first page of this appendix, and including any data for species that are surrogates of threatened or endangered species that occur at the site.

e. Use the following step-wise process to determine which of the uncircled species **must** be deleted and which **must not** be deleted:

1. Does the genus occur at the site?
    - If "No", go to step 2.
    - If "Yes", are there one or more species in the genus that occur at the site but are not in the dataset?
        - If "No", go to step 2.
        - If "Yes", retain the uncircled species.*

2. Does the family occur at the site?
    - If "No", go to step 3.
    - If "Yes", are there one or more genera in the family that occur at the site but are not in the dataset?
        - If "No", go to step 3.
        - If "Yes", retain the uncircled species.*

3. Does the order occur at the site?
    - If "No", go to step 4.
    - If "Yes", does the dataset contain a circled species that is in the same order?
        - If "No", retain the uncircled species.*
        - If "Yes", delete the uncircled species.*

4. Does the class occur at the site?
    - If "No", go to step 5.
    - If "Yes", does the dataset contain a circled species that is in the same class?
        - If "No", retain the uncircled species.*
        - If "Yes", delete the uncircled species.*

5. Does the phylum occur at the site?
    - If "No", delete the uncircled species.*
    - If "Yes", does the dataset contain a circled species that is in the same phylum?
        - If "No", retain the uncircled species.*
        - If "Yes", delete the uncircled species.*

* = Continue the deletion process by starting at step 1 for another uncircled species unless 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 Figure B1.)

This deletion process is designed to ensure that:

a. Each species that occurs both in the national dataset and at the site also occurs in the site-specific dataset.

b. Each species that occurs at the site but does not occur in the national dataset is represented in the site-specific dataset by <u>all</u> species in the national dataset that are in the same genus.
c. Each genus that occurs at the site but does not occur in the national dataset is represented in the site-specific dataset by <u>all</u> genera in the national dataset that are in the same family.
d. Each order, class, and phylum that occurs both in the national dataset and at the site is represented in the site-specific dataset by the one or more species in the national dataset that are most closely related to a species that occurs at the site.

D. Checking the Minimum Data Requirements

The initial MDRs for the Recalculation Procedure are the same as those for the derivation of a national criterion. If a specific requirement cannot be satisfied after deletion because that kind of species does not occur at the site, a taxonomically similar species **must** be substituted in order to meet the eight MDRs:

> If no species of the kind required occurs at the site, but a species in the same order does, the MDR can only be satisfied by data for a species that occurs at the site and is in that order; if no species in the order occurs at the site, but a species in the class does, the MDR can only be satisfied by data for a species that occurs at the site and is in that class. If no species in the same class occurs at the site, but a species in the phylum does, the MDR can only be satisfied by data for a species that occurs at the site and is in that phylum. If no species in the same phylum occurs at the site, any species that occurs at the site and is not used to satisfy a different MDR can be used to satisfy the MDR. If additional data are generated using acceptable procedures (U.S. EPA 1985) and they are approved by the U.S. EPA, the Recalculation Procedure **must** be started again at step B with the addition of the new data.

If fewer than eight families of aquatic invertebrates, amphibians, and fishes occur at the site, a Species Mean Acute Value **must** be available for at least one species in each of the families and the special version of the Recalculation Procedure described on the second page of this appendix **must** be used.

E. Determining the CMC and/or CCC

1. Determining the FAV:
   a. If the eight family MDRs are satisfied, the site-specific FAV **must** be calculated from Genus Mean Acute Values using

the procedure described in the national aquatic life guidelines (U.S. EPA 1985).

b. If fewer than eight families of aquatic invertebrates, amphibians, and fishes occur at the site, the lowest Species Mean Acute Value that is available for a species that occurs at the site **must** be used as the FAV, as per the special version of the Recalculation Procedure described on the second page of this appendix.

2. The site-specific CMC **must** be calculated by dividing the site-specific FAV by 2. 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). (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.)
3. The calculated FAV, CMC, and/or CCC **must** be lowered, if necessary, to (1) protect an aquatic plant, invertebrate, amphibian, or fish species that is a critical species at the site, and (2) 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.

F. Writing the Report

The report of the results of use of the Recalculation Procedure **must** include:

1. A list of all species of aquatic invertebrates, amphibians, and fishes that are known to "occur at the site", along with the source of the information.
2. A list of all aquatic plant, invertebrate, amphibian, and fish species that are critical species at the site, including all species that occur at the site and are listed as threatened or endangered under section 4 of the Endangered Species Act.
3. A site-specific version of Table 1 from a criteria document produced by the U.S. EPA after 1984.
4. A site-specific version of Table 3 from a criteria document produced by the U.S. EPA after 1984.
5. A list of all species that were deleted.
6. The new calculated FAV, CMC, and/or CCC.
7. The lowered FAV, CMC, and/or CCC, if one or more were lowered to protect a specific species.

Reference

U.S. EPA. 1985. Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses. PB85-227049. National Technical Information Service, Springfield, VA.

Figure B1: An Example of the Deletion Process Using Three Phyla

SPECIES THAT ARE IN THE THREE PHYLA AND OCCUR AT THE SITE

Phylum	Class	Order	Family	Species
Annelida	Hirudin.	Rhynchob.	Glossiph.	Glossip. complanata
Bryozoa	(No species in this phylum occur at the site.)			
Chordata	Osteich.	Cyprinif.	Cyprinid.	Carassius auratus
Chordata	Osteich.	Cyprinif.	Cyprinid.	Notropis anogenus
Chordata	Osteich.	Cyprinif.	Cyprinid.	Phoxinus eos
Chordata	Osteich.	Cyprinif.	Catostom.	Carpiodes carpio
Chordata	Osteich.	Salmonif.	Osmerida.	Osmerus mordax
Chordata	Osteich.	Percifor.	Centrarc.	Lepomis cyanellus
Chordata	Osteich.	Percifor.	Centrarc.	Lepomis humilis
Chordata	Amphibia	Caudata	Ambystom.	Ambystoma gracile

SPECIES THAT ARE IN THE THREE PHYLA AND IN THE NATIONAL DATASET

Phylum	Class	Order	Family	Species	Code
Annelida	Oligoch.	Haplotax.	Tubifici.	Tubifex tubifex	P
Bryozoa	Phylact.	---	Lophopod.	Lophopod. carteri	D
Chordata	Cephala.	Petromyz.	Petromyz.	Petromyzon marinus	D
Chordata	Osteich.	Cyprinif.	Cyprinid.	Carassius auratus	S
Chordata	Osteich.	Cyprinif.	Cyprinid.	Notropis hudsonius	G
Chordata	Osteich.	Cyprinif.	Cyprinid.	Notropis stramineus	G
Chordata	Osteich.	Cyprinif.	Cyprinid.	Phoxinus eos	S
Chordata	Osteich.	Cyprinif.	Cyprinid.	Phoxinus oreas	D
Chordata	Osteich.	Cyprinif.	Cyprinid.	Tinca tinca	D
Chordata	Osteich.	Cyprinif.	Catostom.	Ictiobus bubalus	F
Chordata	Osteich.	Salmonif.	Salmonid.	Oncorhynchus mykiss	O
Chordata	Osteich.	Percifor.	Centrarc.	Lepomis cyanellus	S
Chordata	Osteich.	Percifor.	Centrarc.	Lepomis macrochirus	G
Chordata	Osteich.	Percifor.	Percidae	Perca flavescens	D
Chordata	Amphibia	Anura	Pipidae	Xenopus laevis	C

Explanations of Codes:

- S = retained because this Species occurs at the site.
- G = retained because there is a species in this Genus that occurs at the site but not in the national dataset.
- F = retained because there is a genus in this Family that occurs at the site but not in the national dataset.
- O = retained because this Order occurs at the site and is not represented by a lower taxon.
- C = retained because this Class occurs at the site and is not represented by a lower taxon.
- P = retained because this Phylum occurs at the site and is not represented by a lower taxon.
- D = deleted because this species does not satisfy any of the requirements for retaining species.

Appendix C: Guidance Concerning the Use of "Clean Techniques" and QA/QC when Measuring Trace Metals

Note: This version of this appendix contains more information than the version that was Appendix B of Prothro (1993).

Recent information (Shiller and Boyle 1987; Windom et al. 1991) has raised questions concerning the quality of reported concentrations of trace metals in both fresh and salt (estuarine and marine) surface waters. A lack of awareness of true ambient concentrations of metals in fresh and salt surface waters can be both a cause and a result of the problem. The ranges of dissolved metals that are typical in surface waters of the United States away from the immediate influence of discharges (Bruland 1983; Shiller and Boyle 1985,1987; Trefry et al. 1986; Windom et al. 1991) are:

Metal	Salt water (μg/L)	Fresh water (μg/L)
Cadmium	0.01 to 0.2	0.002 to 0.08
Copper	0.1 to 3.	0.4 to 4.
Lead	0.01 to 1.	0.01 to 0.19
Nickel	0.3 to 5.	1. to 2.
Silver	0.005 to 0.2	-------------
Zinc	0.1 to 15.	0.03 to 5.

The U.S. EPA (1983,1991) has published analytical methods for monitoring metals in waters and wastewaters, but these methods are inadequate for determination of ambient concentrations of some metals in some surface waters. Accurate and precise measurement of these low concentrations requires appropriate attention to seven areas:

1. Use of "clean techniques" during collecting, handling, storing, preparing, and analyzing samples to avoid contamination.
2. Use of analytical methods that have sufficiently low detection limits.
3. Avoidance of interference in the quantification (instrumental analysis) step.
4. Use of blanks to assess contamination.
5. Use of matrix spikes (sample spikes) and certified reference materials (CRMs) to assess interference and contamination.
6. Use of replicates to assess precision.
7. Use of certified standards.

In a strict sense, the term "clean techniques" refers to techniques that reduce contamination and enable the accurate and precise measurement of trace metals in fresh and salt surface waters. In a broader sense, the term also refers to related issues concerning detection limits, quality control, and quality assurance. Documenting data quality demonstrates the amount of confidence that can be placed in the data, whereas increasing the sensitivity of methods reduces the problem of deciding how to interpret results that are reported to be below detection limits.

<u>This appendix is written for those analytical laboratories that want guidance concerning ways to lower detection limits, increase accuracy, and/or increase precision.</u> The ways to achieve these goals are to increase the sensitivity of the analytical methods, decrease contamination, and decrease interference. Ideally, validation of a procedure for measuring concentrations of metals in surface water requires demonstration that agreement can be obtained using completely different procedures beginning with the sampling step and continuing through the quantification step (Bruland et al. 1979), but few laboratories have the resources to compare two different procedures. Laboratories can, however, (a) use techniques that others have found useful for improving detection limits, accuracy, and precision, and (b) document data quality through use of blanks, spikes, CRMs, replicates, and standards.

<u>Nothing contained or not contained in this appendix adds to or subtracts from any regulatory requirement set forth in other EPA documents concerning analyses of metals.</u> A WER can be acceptably determined without the use of clean techniques as long as the detection limits, accuracy, and precision are acceptable. No QA/QC requirements beyond those that apply to measuring metals in effluents are necessary for the determination of WERs. The word "must" is not used in this appendix. Some items, however, are considered so important by analytical chemists who have worked to increase accuracy and precision and lower detection limits in trace-metal analysis that "**should**" is in bold print to draw attention to the item. Most such items are emphasized because they have been found to have received inadequate attention in some laboratories performing trace-metal analyses.

In general, in order to achieve accurate and precise measurement of a particular concentration, both the detection limit and the blanks should be less than one-tenth of that concentration. Therefore, the term "metal-free" can be interpreted to mean that the total amount of contamination that occurs during sample collection and processing (e.g., from gloves, sample containers, labware, sampling apparatus, cleaning solutions, air, reagents, etc.) is sufficiently low that blanks are less than one-tenth of the lowest concentration that needs to be measured.

Atmospheric particulates can be a major source of contamination (Moody 1982; Adeloju and Bond 1985). The term "class-100" refers to a specification concerning the amount of particulates in air (Moody 1982); although the specification says nothing about the composition of the particulates, generic control of particulates can greatly reduce trace-metal blanks. Except during collection of samples, initial cleaning of equipment, and handling of samples containing high concentrations of metals, all handling of samples, sample containers, labware, and sampling apparatus should be performed in a class-100 bench, room, or glove box.

Neither the "ultraclean techniques" that might be necessary when trace analyses of mercury are performed nor safety in analytical laboratories is addressed herein. Other documents should be consulted if one or both of these topics are of concern.

Avoiding contamination by use of "clean techniques"

Measurement of trace metals in surface waters should take into account the potential for contamination during each step in the process. Regardless of the specific procedures used for collection, handling, storage, preparation (digestion, filtration, and/or extraction), and quantification (instrumental analysis), the general principles of contamination control should be applied. Some specific recommendations are:

a. Powder-free (non-talc, class-100) latex, polyethylene, or polyvinyl chloride (PVC, vinyl) gloves **should** be worn during all steps from sample collection to analysis. (Talc seems to be a particular problem with zinc; gloves made with talc cannot be decontaminated sufficiently.) Gloves should only contact surfaces that are metal-free; gloves should be changed if even suspected of contamination.
b. The acid used to acidify samples for preservation and digestion and to acidify water for final cleaning of labware, sampling apparatus, and sample containers **should** be metal-free. The quality of the acid used should be better than reagent-grade. Each lot of acid **should** be analyzed for the metal(s) of interest before use.
c. The water used to prepare acidic cleaning solutions and to rinse labware, sample containers, and sampling apparatus may be prepared by distillation, deionization, or reverse osmosis, and **should** be demonstrated to be metal-free.
d. The work area, including bench tops and hoods, should be cleaned (e.g., washed and wiped dry with lint-free, class-100 wipes) frequently to remove contamination.
e. All handling of samples in the laboratory, including filtering and analysis, **should** be performed in a class-100 clean bench or a glove box fed by particle-free air or nitrogen; ideally the clean bench or glove box should be located within a class-100 clean room.
f. Labware, reagents, sampling apparatus, and sample containers **should** never be left open to the atmosphere; they should be stored in a class-100 bench, covered with plastic wrap, stored in a plastic box, or turned upside down on a clean surface. Minimizing the time between cleaning and using will help minimize contamination.

g. Separate sets of sample containers, labware, and sampling apparatus should be dedicated for different kinds of samples, e.g., surface water samples, effluent samples, etc.
h. To avoid contamination of clean rooms, samples that contain very high concentrations of metals and do not require use of "clean techniques" **should not** be brought into clean rooms.
i. Acid-cleaned plastic, such as high-density polyethylene (HDPE), low-density polyethylene (LDPE), or a fluoroplastic, **should** be the only material that ever contacts a sample, except possibly during digestion for the total recoverable measurement.
   1. Total recoverable samples can be digested in some plastic containers.
   2. HDPE and LDPE might not be acceptable for mercury.
   3. Even if acidified, samples and standards containing silver should be in amber containers.
j. All labware, sample containers, and sampling apparatus **should** be acid-cleaned before use or reuse.
   1. Sample containers, sampling apparatus, tubing, membrane filters, filter assemblies, and other labware **should** be soaked in acid until metal-free. The amount of cleaning necessary might depend on the amount of contamination and the length of time the item will be in contact with samples. For example, if an acidified sample will be stored in a sample container for three weeks, ideally the container should have been soaked in an acidified metal-free solution for at least three weeks.
   2. It might be desirable to perform initial cleaning, for which reagent-grade acid may be used, before the items are taken into a clean room. For most metals, items should be either (a) soaked in 10 percent concentrated nitric acid at 50°C for at least one hour, or (b) soaked in 50 percent concentrated nitric acid at room temperature for at least two days; for arsenic and mercury, soaking for up to two weeks at 50°C in 10 percent concentrated nitric acid might be required. For plastics that might be damaged by strong nitric acid, such as polycarbonate and possibly HDPE and LDPE, soaking in 10 percent concentrated hydrochloric acid, either in place of or before soaking in a nitric acid solution, might be desirable.
   3. Chromic acid **should not** be used to clean items that will be used in analysis of metals.
   4. Final soaking and cleaning of sample containers, labware, and sampling apparatus **should** be performed in a class-100 clean room using metal-free acid and water. The solution in an acid bath **should** be analyzed periodically to demonstrate that it is metal-free.
k. Labware, sampling apparatus, and sample containers **should** be stored appropriately after cleaning:
   1. After the labware and sampling apparatus are cleaned, they may be stored in a clean room in a weak acid bath prepared using metal-free acid and water. Before use, the items

should be rinsed at least three times with metal-free water. After the final rinse, the items should be moved immediately, with the open end pointed down, to a class-100 clean bench. Items may be dried on a class-100 clean bench; items **should not** be dried in an oven or with laboratory towels. The sampling apparatus should be assembled in a class-100 clean room or bench and double-bagged in metal-free polyethylene zip-type bags for transport to the field; new bags are usually metal-free.

2. After sample containers are cleaned, they should be filled with metal-free water that has been acidified to a pH of 2 with metal-free nitric acid (about 0.5 mL per liter) for storage until use.

l. Labware, sampling apparatus, and sample containers **should** be rinsed and not rinsed with sample as necessary to prevent high and low bias of analytical results because acid-cleaned plastic will sorb some metals from unacidified solutions.
   1. Because samples for the dissolved measurement are not acidified until after filtration, all sampling apparatus, sample containers, labware, filter holders, membrane filters, etc., that contact the sample before or during filtration **should** be rinsed with a portion of the solution and then that portion discarded.
   2. For the total recoverable measurement, labware, etc., that contact the sample <u>only</u> before it is acidified **should** be rinsed with sample, whereas items that contact the sample after it is acidified **should not** be rinsed. For example, the sampling apparatus should be rinsed because the sample will not be acidified until it is in a sample container, but the sample container should not be rinsed <u>if</u> the sample will be acidified in the sample container.
   3. If the total recoverable and dissolved measurements are to be performed on the same sample (rather than on two samples obtained at the same time and place), all the apparatus and labware, including the sample container, should be rinsed before the sample is placed in the sample container; then an unacidified aliquot should be removed for the total recoverable measurement (and acidified, digested, etc.) and an unacidified aliquot should be removed for the dissolved measurement (and filtered, acidified, etc.) (If a container is rinsed and filled with sample and an unacidified aliquot is removed for the dissolved measurement and then the solution in the container is acidified before removal of an aliquot for the total recoverable measurement, the resulting measured total recoverable concentration might be biased high because the acidification might desorb metal that had been sorbed onto the walls of the sample container; the amount of bias will depend on the relative volumes involved and on the amount of sorption and desorption.)

m. Field samples **should** be collected in a manner that eliminates the potential for contamination from sampling platforms, probes, etc. Exhaust from boats and the direction of wind and water currents should be taken into account. The people who collect the samples **should** be specifically trained on how to collect field samples. After collection, all handling of samples in the field that will expose the sample to air **should** be performed in a portable class-100 clean bench or glove box.

n. Samples **should** be acidified (after filtration if dissolved metal is to be measured) to a pH of less than 2, except that the pH **should** be less than 1 for mercury. Acidification should be done in a clean room or bench, and so it might be desirable to wait and acidify samples in a laboratory rather than in the field. If samples are acidified in the field, metal-free acid can be transported in plastic bottles and poured into a plastic container from which acid can be removed and added to samples using plastic pipettes. Alternatively, plastic automatic dispensers can be used.
o. Such things as probes and thermometers **should not** be put in samples that are to be analyzed for metals. In particular, pH electrodes and mercury-in-glass thermometers **should not** be used if mercury is to be measured. If pH is measured, it **should** be done on a separate aliquot.
p. Sample handling should be minimized. For example, instead of pouring a sample into a graduated cylinder to measure the volume, the sample can be weighed after being poured into a tared container, which is less likely to be subject to error than weighing the container from which the sample is poured. (For saltwater samples, the salinity or density should be taken into account if weight is converted to volume.)
q. Each reagent used **should** be verified to be metal-free. If metal-free reagents are not commercially available, removal of metals will probably be necessary.
r. For the total recoverable measurement, samples should be digested in a class-100 bench, not in a metallic hood. If feasible, digestion should be done in the sample container by acidification and heating.
s. The longer the time between collection and analysis of samples, the greater the chance of contamination, loss, etc.
t. Samples should be stored in the dark, preferably between 0 and 4°C with no air space in the sample container.

Achieving low detection limits

a. Extraction of the metal from the sample can be extremely useful if it simultaneously concentrates the metal and eliminates potential matrix interferences. For example, ammonium 1-pyrrolidinedithiocarbamate and/or diethylammonium diethyldithiocarbamate can extract cadmium, copper, lead, nickel, and zinc (Bruland et al. 1979; Nriagu et al. 1993).
b. The detection limit should be less than ten percent of the lowest concentration that is to be measured.

Avoiding interferences

a. Potential interferences **should** be assessed for the specific instrumental analysis technique used and for each metal to be measured.
b. If direct analysis is used, the salt present in high-salinity saltwater samples is likely to cause interference in most instrumental techniques.
c. As stated above, extraction of the metal from the sample is particularly useful because it simultaneously concentrates the metal and eliminates potential matrix interferences.

Using blanks to assess contamination

a. A laboratory (procedural, method) blank consists of filling a sample container with analyzed metal-free water and processing (filtering, acidifying, etc.) the water through the laboratory procedure in exactly the same way as a sample. A laboratory blank **should** be included in each set of ten or fewer samples to check for contamination in the laboratory, and **should** contain less than ten percent of the lowest concentration that is to be measured. Separate laboratory blanks **should** be processed for the total recoverable and dissolved measurements, if both measurements are performed.
b. A field (trip) blank consists of filling a sample container with analyzed metal-free water in the laboratory, taking the container to the site, processing the water through tubing, filter, etc., collecting the water in a sample container, and acidifying the water the same as a field sample. A field blank **should** be processed for each sampling trip. Separate field blanks **should** be processed for the total recoverable measurement and for the dissolved measurement, if filtrations are performed at the site. Field blanks **should** be processed in the laboratory the same as laboratory blanks.

Assessing accuracy

a. A calibration curve **should** be determined for each analytical run and the calibration should be checked about every tenth sample. Calibration solutions **should** be traceable back to a certified standard from the U.S. EPA or the National Institute of Science and Technology (NIST).
b. A blind standard or a blind calibration solution **should** be included in each group of about twenty samples.
c. At least one of the following **should** be included in each group of about twenty samples:
   1. A matrix spike (spiked sample; the method of known additions).

2. A CRM, if one is available in a matrix that closely approximates that of the samples. Values obtained for the CRM **should** be within the published values.

The concentrations in blind standards and solutions, spikes, and CRMs **should not** be more than 5 times the median concentration expected to be present in the samples.

Assessing precision

a. A sampling replicate **should** be included with each set of samples collected at each sampling location.
b. If the volume of the sample is large enough, replicate analysis of at least one sample **should** be performed along with each group of about ten samples.

Special considerations concerning the dissolved measurement

Whereas total recoverable measurements are especially subject to contamination during digestion, dissolved measurements are subject to both loss and contamination during filtration.

a. Because acid-cleaned plastic sorbs metal from unacidified solutions and because samples for the dissolved measurement are not acidified before filtration, all sampling apparatus, sample containers, labware, filter holders, and membrane filters that contact the sample before or during filtration **should** be conditioned by rinsing with a portion of the solution and discarding that portion.
b. Filtrations **should** be performed using acid-cleaned plastic filter holders and acid-cleaned membrane filters. Samples **should not** be filtered through glass fiber filters, even if the filters have been cleaned with acid. If positive-pressure filtration is used, the air or gas **should** be passed through a 0.2-μm in-line filter; if vacuum filtration is used, it **should** be performed on a class-100 bench.
c. Plastic filter holders **should** be rinsed and/or dipped between filtrations, but they do not have to be soaked between filtrations if all the samples contain about the same concentrations of metal. It is best to filter samples from low to high concentrations. A membrane filter **should not** be used for more than one filtration. After each filtration, the membrane filter **should** be removed and discarded, and the filter holder **should** be either rinsed with metal-free water or dilute acid and dipped in a metal-free acid bath or rinsed at least twice with metal-free dilute acid; finally, the filter holder **should** be rinsed at least twice with metal-free water.
d. For each sample to be filtered, the filter holder and membrane filter **should** be conditioned with the sample, i.e., an initial portion of the sample **should** be filtered and discarded.

The accuracy and precision of the dissolved measurement **should** be assessed periodically. A large volume of a buffered solution (such as aerated 0.05 N sodium bicarbonate for analyses in fresh water and a combination of sodium bicarbonate and sodium chloride for analyses in salt water) should be spiked so that the concentration of the metal of interest is in the range of the low concentrations that are to be measured. Sufficient samples should be taken alternately for (a) acidification in the same way as after filtration in the dissolved method and (b) filtration and acidification using the procedures specified in the dissolved method until ten samples have been processed in each way. The concentration of metal in each of the twenty samples should then be determined using the same analytical procedure. The means of the two groups of ten measurements should be within 10 percent, and the coefficient of variation for each group of ten should be less than 20 percent. Any values deleted as outliers **should** be acknowledged.

Reporting results

To indicate the quality of the data, reports of results of measurements of the concentrations of metals **should** include a description of the blanks, spikes, CRMs, replicates, and standards that were run, the number run, and the results obtained. All values deleted as outliers **should** be acknowledged.

Additional information

The items presented above are some of the important aspects of "clean techniques"; some aspects of quality assurance and quality control are also presented. This is not a definitive treatment of these topics; additional information that might be useful is available in such publications as Patterson and Settle (1976), Zief and Mitchell (1976), Bruland et al. (1979), Moody and Beary (1982), Moody (1982), Bruland (1983), Adeloju and Bond (1985), Berman and Yeats (1985), Byrd and Andreae (1986), Taylor (1987), Sakamoto-Arnold (1987), Tramontano et al. (1987), Puls and Barcelona (1989), Windom et al. (1991), U.S. EPA (1992), Horowitz et al. (1992), and Nriagu et al. (1993).

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Appendix D: Relationships between WERs and the Chemistry and Toxicology of Metals

The aquatic toxicology of metals is complex in part because the chemistry of metals in water is complex. Metals usually exist in surface water in various combinations of particulate and dissolved forms, some of which are toxic and some of which are nontoxic. In addition, all toxic forms of a metal are not necessarily equally toxic, and various water quality characteristics can affect the relative concentrations and/or toxicities of some of the forms.

The toxicity of a metal has sometimes been reported to be proportional to the concentration or activity of a specific species of the metal. For example, Allen and Hansen (1993) summarized reports by several investigators that the toxicity of copper is related to the free cupric ion, but other data do not support a correlation (Erickson 1993a). For example, Borgmann (1983), Chapman and McCrady (1977), and French and Hunt (1986) found that toxicity expressed on the basis of cupric ion activity varied greatly with pH, and Cowan et al. (1986) concluded that at least one of the copper hydroxide species is toxic. Further, chloride and sulfate salts of calcium, magnesium, potassium, and sodium affect the toxicity of the cupric ion (Nelson et al. 1986). Similarly for aluminum, Wilkinson et al. (1993) concluded that "mortality was best predicted not by the free Al^{3+} activity but rather as a function of the sum $\Sigma([Al^{3+}] + [AlF^{2+}])$" and that "no longer can the reduction of Al toxicity in the presence of organic acids be interpreted simply as a consequence of the decrease in the free Al^{3+} concentration".

Until a model has been demonstrated to explain the quantitative relationship between chemical and toxicological measurements, aquatic life criteria should be established in an environmentally conservative manner with provision for site-specific adjustment. Criteria should be expressed in terms of feasible analytical measurements that provide the necessary conservatism without substantially increasing the cost of implementation and site-specific adjustment. Thus current aquatic life criteria for metals are expressed in terms of the total recoverable measurement and/or the dissolved measurement, rather than a measurement that would be more difficult to perform and would still require empirical adjustment. The WER is operationally defined in terms of chemical and toxicological measurements to allow site-specific adjustments that account for differences between the toxicity of a metal in laboratory dilution water and in site water.

Forms of Metals

Even if the relationship of toxicity to the forms of metals is not understood well enough to allow setting site-specific water quality criteria without using empirical adjustments, appropriate use and interpretation of WERs requires an understanding of how changes in the relative concentrations of different forms of a metal might affect toxicity. Because WERs are defined on the basis of relationships between measurements of toxicity and measurements of total recoverable and/or dissolved metal, the toxicologically relevant distinction is between the forms of the metal that are toxic and nontoxic whereas the chemically relevant distinction is between the forms that are dissolved and particulate. "Dissolved metal" is defined here as "metal that passes through either a 0.45-μm or a 0.40-μm membrane filter" and "particulate metal" is defined as "total recoverable metal minus dissolved metal". Metal that is in or on particles that pass through the filter is operationally defined as "dissolved".

In addition, some species of metal can be converted from one form to another. Some conversions are the result of reequilibration in response to changes in water quality characteristics whereas others are due to such fate processes as oxidation of sulfides and/or organic matter. Reequilibration usually occurs faster than fate processes and probably results in any rapid changes that are due to effluent mixing with receiving water or changes in pH at a gill surface. To account for rapid changes due to reequilibration, the terms "labile" and "refractory" will be used herein to denote metal species that do and do not readily convert to other species when in a nonequilibrium condition, with "readily" referring to substantial progression toward equilibrium in less than about an hour. Although the toxicity and lability of a form of a metal are not merely yes/no properties, but rather involve gradations, a simple classification scheme such as this should be sufficient to establish the principles regarding how WERs are related to various operationally defined forms of metal and how this affects the determination and use of WERs.

Figure D1 presents the classification scheme that results from distinguishing forms of metal based on analytical methodology, toxicity tests, and lability, as described above. Metal that is not measured by the total recoverable measurement is assumed to be sufficiently nontoxic and refractory that it will not be further considered here. Allowance is made for toxicity due to particulate metal because some data indicate that particulate metal might contribute to toxicity and bioaccumulation, although other data imply that little or no toxicity can be ascribed to particulate metal (Erickson 1993b). Even if the toxicity of particulate metal is not negligible in a particular situation, a dissolved criterion will not be underprotective if the dissolved criterion was derived using a dissolved WER (see below) or if there are sufficient compensating factors.

Figure D1: A Scheme for Classifying Forms of Metal in Water

- Total recoverable metal
  - Dissolved
    - Nontoxic
      - Labile
      - Refractory
    - Toxic
      - Labile
  - Particulate
    - Nontoxic
      - Labile
      - Refractory
    - Toxic
      - Labile
- Metal not measured by the total recoverable measurement

Not only can some changes in water quality characteristics shift the relative concentrations of toxic and nontoxic labile species of a metal, some changes in water quality can also increase or decrease the toxicities of the toxic species of a metal and/or the sensitivities of aquatic organisms. Such changes might be caused by (a) a change in ionic strength that affects the activity of toxic species of the metal in water, (b) a physiological effect whereby an ion affects the permeability of a membrane and thereby alters both uptake and apparent toxicity, and (c) toxicological additivity, synergism, or antagonism due to effects within the organism.

Another possible complication is that a form of metal that is toxic to one aquatic organism might not be toxic to another. Although such differences between organisms have not been demonstrated, the possibility cannot be ruled out.

The Importance of Lability

The only common metal measurement that can be validly extrapolated from the effluent and the upstream water to the downstream water merely by taking dilution into account is the total recoverable measurement. A major reason this measurement is so useful is because it is the only measurement that obeys the law of mass balance (i.e., it is the only measurement that is conservative). Other metal measurements usually do not obey the law of mass balance because they measure some, but not all, of the labile species of metals. A measurement of refractory metal would be conservative in terms of changes in water quality characteristics, but not necessarily in regards to fate processes; such a measurement has not been developed, however.

Permit limits apply to effluents, whereas water quality criteria apply to surface waters. If permit limits and water quality criteria are both expressed in terms of total recoverable metal, extrapolations from effluent to surface water only need to take dilution into account and can be performed as mass balance calculations. If either permit limits or water quality criteria or both are expressed in terms of any other metal measurement, lability needs to be taken into account, even if both are expressed in terms of the same measurement.

Extrapolations concerning labile species of metals from effluent to surface water depend to a large extent on the differences between the water quality characteristics of the effluent and those of the surface water. Although equilibrium models of the speciation of metals can provide insight, the interactions are too complex to be able to make useful nonempirical extrapolations from a wide variety of effluents to a wide variety of surface waters of either (a) the speciation of the metal or (b) a metal measurement other than total recoverable.

Empirical extrapolations can be performed fairly easily and the most common case will probably occur when permit limits are based on the total recoverable measurement but water quality criteria are based on the dissolved measurement. The empirical extrapolation is intended to answer the question "What percent of the total recoverable metal in the effluent becomes dissolved in the downstream water?" This question can be answered by:

a. Collecting samples of effluent and upstream water.
b. Measuring total recoverable metal and dissolved metal in both samples.
c. Combining aliquots of the two samples in the ratio of the flows when the samples were obtained and mixing for an appropriate period of time under appropriate conditions.
d. Measuring total recoverable metal and dissolved metal in the mixture.

An example is presented in Figure D2. This percentage cannot be extrapolated from one metal to another or from one effluent to another. The data needed to calculate the percentage will be obtained each time a WER is determined using simulated downstream water if both dissolved and total recoverable metal are measured in the effluent, upstream water, and simulated downstream water.

The interpretation of the percentage is not necessarily as straightforward as might be assumed. For example, some of the metal that is dissolved in the upstream water might sorb onto particulate matter in the effluent, which can be viewed as a detoxification of the upstream water by the effluent. Regardless of the interpretation, the described procedure provides a simple way of relating the total recoverable concentration in the effluent to the concentration of concern in the downstream water. Because this empirical extrapolation can be used with any analytical measurement that is chosen as the basis for expression of aquatic life criteria, use of the total recoverable measurement to express permit limits on effluents does not place any restrictions on which analytical measurement can be used to express criteria. Further, even if both criteria and permit limits are expressed in terms of a measurement such as dissolved metal, an empirical extrapolation would still be necessary because dissolved metal is not likely to be conservative from effluent to downstream water.

Merits of Total Recoverable and Dissolved WERs and Criteria

A WER is operationally defined as the value of an endpoint obtained with a toxicity test using site water divided by the value of the same endpoint obtained with the same toxicity test using a laboratory dilution water. Therefore, just as aquatic life criteria can be expressed in terms of either the total recoverable measurement or the dissolved measurement, so can WERs. A pair of side-by-side toxicity tests can produce both a total recoverable WER and a dissolved WER if the metal in the test solutions in both of the tests is measured using both methods. A total recoverable WER is obtained by dividing endpoints that were calculated on the basis of total recoverable metal, whereas a dissolved WER is obtained by dividing endpoints that were calculated on the basis of dissolved metal. Because of the way they are determined, a total recoverable WER is used to calculate a total recoverable site-specific criterion from a national, state, or recalculated aquatic life criterion that is expressed using the total recoverable measurement, whereas a dissolved WER is used to calculate a dissolved site-specific criterion from a national, state, or recalculated criterion that is expressed in terms of the dissolved measurement.

In terms of the classification scheme given in Figure D1, the basic relationship between a total recoverable national water quality <u>criterion</u> and a total recoverable <u>WER</u> is:

- A total recoverable <u>criterion</u> treats all the toxic and nontoxic metal in the site water as if its average toxicity were the same as the average toxicity of all the toxic and nontoxic metal in the toxicity tests in laboratory dilution water on which the criterion is based.
- A total recoverable <u>WER</u> is a measurement of the actual ratio of the average toxicities of the total recoverable metal and replaces the assumption that the ratio is 1.

Similarly, the basic relationship between a dissolved national criterion and a dissolved WER is:

- A dissolved criterion treats all the toxic and nontoxic dissolved metal in the site water as if its average toxicity were the same as the average toxicity of all the toxic and nontoxic dissolved metal in the toxicity tests in laboratory dilution water on which the criterion is based.
- A dissolved WER is a measurement of the actual ratio of the average toxicities of the dissolved metal and replaces the assumption that the ratio is 1.

In both cases, use of a criterion without a WER involves measurement of toxicity in laboratory dilution water but only prediction of toxicity in site water, whereas use of a criterion with a WER involves measurement of toxicity in both laboratory dilution water and site water.

When WERs are used to derive site-specific criteria, the total recoverable and dissolved approaches are inherently consistent. They are consistent because the toxic effects caused by the metal in the toxicity tests do not depend on what chemical measurements are performed; the same number of organisms are killed in the acute lethality tests regardless of what, if any, measurements of the concentration of the metal are made. The only difference is the chemical measurement to which the toxicity is referenced. Dissolved WERs can be derived from the same pairs of toxicity tests from which total recoverable WERs are derived, if the metal in the tests is measured using both the total recoverable and dissolved measurements. Both approaches start at the same place (i.e., the amount of toxicity observed in laboratory dilution water) and end at the same place (i.e., the amount of toxicity observed in site water). The combination of a total recoverable criterion and WER accomplish the same thing as the combination of a dissolved criterion and WER. By extension, whenever a criterion and a WER based on the same measurement of the metal are used together, they will end up at the same place. Because use of a total recoverable criterion with a total recoverable WER ends up at exactly the same place as use of a dissolved criterion with a dissolved WER, whenever one WER is determined, both should be determined to allow (a) a check on the analytical chemistry, (b) use of the inherent internal consistency to check that the data are used correctly, and (c) the option of using either approach in the derivation of permit limits.

An examination of how the two approaches (the total recoverable approach and the dissolved approach) address the four relevant forms of metal (toxic and nontoxic particulate metal and toxic and nontoxic dissolved metal) in laboratory dilution water and in site water further explains why the two approaches are inherently consistent. Here, only the way in which the two approaches address each of the four forms of metal in site water will be considered:

a. Toxic dissolved metal:
   This form contributes to the toxicity of the site water and is measured by both chemical measurements. If this is the only form of metal present, the two WERs will be the same.
b. Nontoxic dissolved metal:
   This form does not contribute to the toxicity of the site water, but it is measured by both chemical measurements. If this is the only form of metal present, the two WERs will be the same. (Nontoxic dissolved metal can be the only form present, however, only if all of the nontoxic dissolved metal present is refractory. If any labile nontoxic dissolved metal is present, equilibrium will require that some toxic dissolved metal also be present.)
c. Toxic particulate metal:
   This form contributes to the toxicological measurement in both approaches; it is measured by the total recoverable measurement, but not by the dissolved measurement. Even though it is not measured by the dissolved measurement, its presence is accounted for in the dissolved approach because it increases the toxicity of the site water and thereby decreases the dissolved WER. It is accounted for because it makes the dissolved metal appear to be more toxic than it is. Most toxic particulate metal is probably not toxic when it is particulate; it becomes toxic when it is dissolved at the gill surface or in the digestive system; in the surface water, however, it is measured as particulate metal.
d. Nontoxic particulate metal:
   This form does not contribute to the toxicity of the site water; it is measured by the total recoverable measurement, but not by the dissolved measurement. Because it is measured by the total recoverable measurement, but not by the dissolved measurement, it causes the total recoverable WER to be higher than the dissolved WER.

In addition to dealing with the four forms of metal similarly, the WERs used in the two approaches comparably take synergism, antagonism, and additivity into account. Synergism and additivity in the site water increase its toxicity and therefore decrease the WER; in contrast, antagonism in the site water decreases toxicity and increases the WER.

Each of the four forms of metal is appropriately taken into account because use of the WERs makes the two approaches internally consistent. In addition, although experimental variation will cause the measured WERs to deviate from the actual WERs, the measured WERs will be internally consistent with the data from which they were generated. If the percent dissolved is the same at the test endpoint in the two waters, the two WERs will be the same. If the percent of the total recoverable metal that is dissolved in laboratory dilution water is less than 100 percent, changing from the total recoverable measurement to the dissolved measurement will lower the criterion but it will comparably lower the denominator in the WER, thus increasing the WER. If the percent of the total recoverable metal that is dissolved in the site water is less than 100 percent, changing from the total recoverable measurement to the dissolved measurement will lower the concentration in the site water that is to be compared with the criterion, but it also lowers the numerator in the WER, thus lowering the WER. Thus when WERs are used to adjust criteria, the total recoverable approach and the dissolved approach result in the same interpretations of concentrations in the site water (see Figure D3) and in the same maximum acceptable concentrations in effluents (see Figure D4).

Thus, if WERs are based on toxicity tests whose endpoints equal the CMC or CCC and if both approaches are used correctly, the two measurements will produce the same results because each WER is based on measurements on the site water and then the WER is used to calculate the site-specific criterion that applies to the site water when the same chemical measurement is used to express the site-specific criterion. The equivalency of the two approaches applies if they are based on the same sample of site water. When they are applied to multiple samples, the approaches can differ depending on how the results from replicate samples are used:

a. If an appropriate averaging process is used, the two will be equivalent.
b. If the lowest value is used, the two approaches will probably be equivalent only if the lowest dissolved WER and the lowest total recoverable WER were obtained using the same sample of site water.

There are several advantages to using a dissolved <u>criterion</u> even when a dissolved WER is not used. In some situations use of a dissolved criterion to interpret results of measurements of the concentration of dissolved metal in site water might demonstrate that there is no need to determine either a total recoverable WER or a dissolved WER. This would occur when so much of the total recoverable metal was nontoxic particulate metal that even though the total recoverable criterion was exceeded, the corresponding dissolved criterion was not exceeded. The particulate metal might come from an effluent, a resuspension event, or runoff that washed particulates into the body of water. In such a situation the total recoverable WER would also show that the site-specific criterion was not exceeded, but there would be no need to determine a WER if the criterion were expressed on the basis of the dissolved measurement. If the variation over time in the concentration of particulate metal is much greater than the variation in the concentration of dissolved metal, both the total recoverable concentration and the total recoverable WER are likely to vary so much over time that a dissolved criterion would be much more useful than a total recoverable criterion.

Use of a dissolved criterion <u>without</u> a dissolved WER has three disadvantages, however:

1. Nontoxic dissolved metal in the site water is treated as if it is toxic.
2. Any toxicity due to particulate metal in the site water is ignored.
3. Synergism, antagonism, and additivity in the site water are not taken into account.

Use of a dissolved criterion <u>with</u> a dissolved WER overcomes all three problems. For example, if (a) the total recoverable concentration greatly exceeds the total recoverable criterion, (b) the dissolved concentration is below the dissolved criterion, and (c) there is concern about the possibility of toxicity of particulate metal, the determination of a dissolved WER would demonstrate whether toxicity due to particulate metal is measurable.

Similarly, use of a total recoverable criterion <u>without</u> a total recoverable WER has three comparable disadvantages:

1. Nontoxic dissolved metal in site water is treated as if it is toxic.
2. Nontoxic particulate metal in site water is treated as if it is toxic.
3. Synergism, antagonism, and additivity in site water are not taken into account.

Use of a total recoverable criterion <u>with</u> a total recoverable WER overcomes all three problems. For example, determination of a total recoverable WER would prevent nontoxic particulate metal (as well as nontoxic dissolved metal) in the site water from being treated as if it is toxic.

<u>Relationships between WERs and the Forms of Metals</u>

Probably the best way to understand what WERs can and cannot do is to understand the relationships between WERs and the forms of metals. A WER is calculated by dividing the concentration of a metal that corresponds to a toxicity endpoint in a site water by the concentration of the same metal that corresponds to the same toxicity endpoint in a laboratory dilution water. Therefore, using the classification scheme given in Figure D1:

$$WER = \frac{R_S + N_S + T_S + \Delta N_S + \Delta T_S}{R_L + N_L + T_L + \Delta N_L + \Delta T_L} .$$

The subscripts "S" and "L" denote site water and laboratory dilution water, respectively, and:

R = the concentration of <u>R</u>efractory metal in a water. (By definition, all refractory metal is nontoxic metal.)

N = the concentration of Nontoxic labile metal in a water.

T = the concentration of Toxic labile metal in a water.

ΔN = the concentration of metal added during a WER determination that is Nontoxic labile metal after it is added.

ΔT = the concentration of metal added during a WER determination that is Toxic labile metal after it is added.

For a total recoverable WER, each of these five concentrations includes both particulate and dissolved metal, if both are present; for a dissolved WER only dissolved metal is included.

Because the two side-by-side tests use the same endpoint and are conducted under identical conditions with comparable test organisms, $T_S + \Delta T_S = T_L + \Delta T_L$ when the toxic species of the metal are equally toxic in the two waters. If a difference in water quality causes one or more of the toxic species of the metal to be more toxic in one water than the other, or causes a shift in the ratios of various toxic species, we can define

$$H = \frac{T_S + \Delta T_S}{T_L + \Delta T_L} .$$

Thus H is a multiplier that accounts for a proportional increase or decrease in the toxicity of the toxic forms in site water as compared to their toxicities in laboratory dilution water. Therefore, the general WER equation is:

$$WER = \frac{R_S + N_S + \Delta N_S + H(T_L + \Delta T_L)}{R_L + N_L + \Delta N_L + (T_L + \Delta T_L)} .$$

Several things are obvious from this equation:

1. A WER should not be thought of as a simple ratio such as H. H is the ratio of the toxicities of the toxic species of the metal, whereas the WER is the ratio of the sum of the toxic and the nontoxic species of the metal. Only under a very specific set of conditions will $WER = H$. If these conditions are satisfied and if, in addition, $H = 1$, then $WER = 1$. Although it might seem that all of these conditions will rarely be satisfied, it is not all that rare to find that an experimentally determined WER is close to 1.
2. When the concentration of metal in laboratory dilution water is negligible, $R_L = N_L = T_L = 0$ and

$$WER = \frac{R_S + N_S + \Delta N_S + H(\Delta T_L)}{\Delta N_L + \Delta T_L}$$

Even though laboratory dilution water is low in TOC and TSS, when metals are added to laboratory dilution water in toxicity tests, ions such as hydroxide, carbonate, and chloride react with some metals to form some particulate species and some dissolved species, both of which might be toxic or nontoxic. The metal species that are nontoxic contribute to ΔN_L, whereas those that are toxic contribute to ΔT_L. Hydroxide, carbonate, chloride, TOC, and TSS can increase ΔN_S. Anything that causes ΔN_S to differ from ΔN_L will cause the WER to differ from 1.

3. Refractory metal and nontoxic labile metal in the site water above that in the laboratory dilution water will increase the WER. Therefore, if the WER is determined in downstream water, rather than in upstream water, the WER will be increased by refractory metal and nontoxic labile metal in the effluent.

Thus there are three major reasons why WERs might be larger or smaller than 1:

a. The toxic species of the metal might be more toxic in one water than in the other, i.e., $H \neq 1$.
b. ΔN might be higher in one water than in the other.
c. R and/or N might be higher in one water than in the other.

The last reason might have great practical importance in some situations. When a WER is determined in downstream water, if most of the metal in the effluent is nontoxic, the WER and the endpoint in site water will correlate with the concentration of metal in the site water. In addition, they will depend on the concentration of metal in the effluent and the concentration of effluent in the site water. This correlation will be best for refractory metal because its toxicity cannot be affected by water quality characteristics; even if the effluent and upstream water are quite different so that the water quality characteristics of the site water depend on the percent effluent, the toxicity of the refractory metal will remain constant at zero and the portion of the WER that is due to refractory metal will be additive.

The Dependence of WERs on the Sensitivity of Toxicity Tests

It would be desirable if the magnitude of the WER for a site water were independent of the toxicity test used in the determination of the WER, so that any convenient toxicity test could be used. It can be seen from the general WER equation that the WER will be independent of the toxicity test only if:

$$WER = \frac{H(T_L + \Delta T_L)}{(T_L + \Delta T_L)} = H ,$$

which would require that $R_S = N_S = \Delta N_S = R_L = N_L = \Delta N_L = 0$. (It would be easy to assume that $T_L = 0$, but it can be misleading in some situations to make more simplifications than are necessary.)

This is the simplistic concept of a WER that would be advantageous if it were true, but which is not likely to be true very often. Any situation in which one or more of the terms is greater than zero can cause the WER to depend on the sensitivity of the toxicity test, although the difference in the WERs might be small.

Two situations that might be common can illustrate how the WER can depend on the sensitivity of the toxicity test. For these illustrations, there is no advantage to assuming that $H = 1$, so H will be retained for generality.

1. The simplest situation is when $R_S > 0$, i.e., when a substantial concentration of refractory metal occurs in the site water. If, for simplification, it is assumed that $N_S = \Delta N_S = R_L = N_L = \Delta N_L = 0$, then:

$$WER = \frac{R_S + H(T_L + \Delta T_L)}{(T_L + \Delta T_L)} = \frac{R_S}{(T_L + \Delta T_L)} + H \; .$$

   The quantity $T_L + \Delta T_L$ obviously changes as the sensitivity of the toxicity test changes. When $R_S = 0$, then $WER = H$ and the WER is independent of the sensitivity of the toxicity test. When $R_S > 0$, then the WER will decrease as the sensitivity of the test decreases because $T_L + \Delta T_L$ will increase.

2. More complicated situations occur when $(N_S + \Delta N_S) > 0$. If, for simplification, it is assumed that $R_S = R_L = N_L = \Delta N_L = 0$, then:

$$WER = \frac{(N_S + \Delta N_S) + H(T_L + \Delta T_L)}{(T_L + \Delta T_L)} = \frac{(N_S + \Delta N_S)}{(T_L + \Delta T_L)} + H \; .$$

   a. If $(N_S + \Delta N_S) > 0$ because the site water contains a substantial concentration of a complexing agent that has an affinity for the metal and if complexation converts toxic metal into nontoxic metal, the complexation reaction will control the toxicity of the solution (Allen 1993). A complexation curve can be graphed in several ways, but the S-shaped curve presented in Figure D5 is most convenient here. The vertical axis is "% uncomplexed", which is assumed to correlate with "% toxic". The "% complexed" is then the "% nontoxic". The ratio of nontoxic metal to toxic metal is:

$$\frac{\%nontoxic}{\%toxic} = \frac{\%complexed}{\%uncomplexed} = V \; .$$

   For the complexed nontoxic metal:

$$V = \frac{concentration\ of\ nontoxic\ metal}{concentration\ of\ toxic\ metal} \; .$$

In the site water, the concentration of complexed nontoxic metal is $(N_S + \Delta N_S)$ and the concentration of toxic metal is $(T_S + \Delta T_S)$, so that:

$$V_S = \frac{(N_S + \Delta N_S)}{(T_S + \Delta T_S)} = \frac{(N_S + \Delta N_S)}{H(T_L + \Delta T_L)} \ .$$

and

$$WER = \frac{V_S H(T_L + \Delta T_L) + H(T_L + \Delta T_L)}{(T_L + \Delta T_L)} = V_S H + H = H(V_S + 1) \ .$$

If the WER is determined using a sensitive toxicity test so that the % uncomplexed (i.e., the % toxic) is 10 %, then $V_S = (90\ \%)/(10\ \%) = 9$, whereas if a less sensitive test is used so that the % uncomplexed is 50 %, then $V_S = (50\ \%)/(50\ \%) = 1$. Therefore, if a portion of the WER is due to a complexing agent in the site water, the magnitude of the WER can decrease as the sensitivity of the toxicity test decreases because the % uncomplexed will decrease. In these situations, the largest WER will be obtained with the most sensitive toxicity test; progressively smaller WERs will be obtained with less sensitive toxicity tests. The magnitude of a WER will depend not only on the sensitivity of the toxicity test but also on the concentration of the complexing agent and on its binding constant (complexation constant, stability constant). In addition, the binding constants of most complexing agents depend on pH.

If the laboratory dilution water contains a low concentration of a complexing agent,

$$V_L = \frac{N_L + \Delta N_L}{T_L + \Delta T_L}$$

and

$$WER = \frac{V_S H(T_L + \Delta T_L) + H(T_L + \Delta T_L)}{V_L(T_L + \Delta T_L) + (T_L + \Delta T_L)} = \frac{V_S H + H}{V_L + 1} = \frac{H(V_S + 1)}{V_L + 1} \ .$$

The binding constant of the complexing agent in the laboratory dilution water is probably different from that of the complexing agent in the site water. Although changing from a more sensitive test to a less sensitive test will decrease both V_S and V_L, the amount of effect is not likely to be proportional.

If the change from a more sensitive test to a less sensitive test were to decrease V_L proportionately more than V_S, the change could result in a larger WER, rather than a smaller WER, as resulted in the case above when it was assumed that the laboratory dilution water did not contain any complexing agent. This is probably most likely to occur if $H = 1$ and if $V_S < V_L$, which would mean that $WER < 1$. Although this is likely to be a rare situation, it does demonstrate again the importance of determining WERs using toxicity tests that have endpoints in laboratory dilution water that are close to the CMC or CCC to which the WER is to be applied.

b. If $(N_S + \Delta N_S) > 0$ because the site water contains a substantial concentration of an ion that will precipitate the metal of concern and if precipitation converts toxic metal into nontoxic metal, the precipitation reaction will control the toxicity of the solution. The "precipitation curve" given in Figure D6 is analogous to the "complexation curve" given in Figure D5; in the precipitation curve, the vertical axis is "% dissolved", which is assumed to correlate with "% toxic". If the endpoint for a toxicity test is below the solubility limit of the precipitate, $(N_S + \Delta N_S) = 0$, whereas if the endpoint for a toxicity test is above the solubility limit, $(N_S + \Delta N_S) > 0$. If WERs are determined with a series of toxicity tests that have increasing endpoints that are above the solubility limit, the WER will reach a maximum value and then decrease. The magnitude of the WER will depend not only on the sensitivity of the toxicity test but also on the concentration of the precipitating agent, the solubility limit, and the solubility of the precipitate.

Thus, depending on the composition of the site water, a WER obtained with an insensitive test might be larger, smaller, or similar to a WER obtained with a sensitive test. Because of the range of possibilities that exist, the best toxicity test to use in the experimental determination of a WER is one whose endpoint in laboratory dilution water is close to the CMC or CCC that is to be adjusted. This is the rationale that was used in the selection of the toxicity tests that are suggested in Appendix I.

The available data indicate that a less sensitive toxicity test usually gives a smaller WER than a more sensitive test (Hansen 1993a). Thus, use of toxicity tests whose endpoints are higher than the CMC or CCC probably will not result in underprotection; in contrast, use of tests whose endpoints are substantially below the CMC or CCC might result in underprotection.

The factors that cause R_S and $(N_S + \Delta N_S)$ to be greater than zero are all external to the test organisms; they are chemical effects that affect the metal in the water. The magnitude of the WER is therefore expected to depend on the toxicity test used only in regard to the sensitivity of the test. If the endpoints for two different tests occur at the same concentration of the metal, the magnitude of the WERs obtained with the two tests should be the same; they should not depend on (a) the duration of the test, (b) whether the endpoint is based on a lethal or sublethal effect, or (c) whether the species is a vertebrate or an invertebrate.

Another interesting consequence of the chemistry of complexation is that the % uncomplexed will increase if the solution is diluted (Allen and Hansen 1993). The concentration of total metal will decrease with dilution but the % uncomplexed will increase. The increase will not offset the decrease and so the concentration of uncomplexed metal will decrease. Thus the portion of a WER that is due to complexation will not be strictly additive (see Appendix G), but the amount of nonadditivity might be difficult to detect in toxicity studies of additivity. A similar effect of dilution will occur for precipitation.

The illustrations presented above were simplified to make it easier to understand the kinds of effects that can occur. The illustrations are qualitatively valid and demonstrate the direction of the effects, but real-world situations will probably be so much more complicated that the various effects cannot be dealt with separately.

<u>Other Properties of WERs</u>

1. Because of the variety of factors that can affect WERs, no rationale exists at present for extrapolating WERs from one metal to another, from one effluent to another, or from one surface water to another. Thus WERs should be individually determined for each metal at each site.

2. The most important information that the determination of a WER provides is whether simulated and/or actual downstream water adversely affects test organisms that are sensitive to the metal. A WER cannot indicate how much metal needs to be removed from or how much metal can be added to an effluent.
   a. If the site water already contains sufficient metal that it is toxic to the test organisms, a WER cannot be determined with a sensitive test and so an insensitive test will have to be used. Even if a WER could be determined with a sensitive test, the WER cannot indicate how much metal has to be removed. For example, if a WER indicated that there was 20 percent too much metal in an effluent, a 30 percent reduction by the discharger would not reduce toxicity if only nontoxic metal was removed. The next WER determination would show that the effluent still contained too much metal. Removing metal is useful only if the metal removed is toxic metal. Reducing the total recoverable concentration does not necessarily reduce toxicity.

b. If the simulated or actual downstream water is not toxic, a WER can be determined and used to calculate how much additional metal the effluent could contain and still be acceptable. Because an unlimited amount of refractory metal can be added to the effluent without affecting the organisms, what the WER actually determines is how much additional toxic metal can be added to the effluent.

3. The effluent component of nearly all WERs is likely to be due mostly to either (a) a reduction in toxicity of the metal by TSS or TOC, or (b) the presence of refractory metal. For both of these, if the percentage of effluent in the downstream water decreases, the magnitude of the WER will usually decrease. If the water quality characteristics of the effluent and the upstream water are quite different, it is possible that the interaction will not be additive; this can affect the portion of the WER that is due to reduced toxicity caused by sorption and/or binding, but it cannot affect the portion of the WER that is due to refractory metal.

4. Test organisms are fed during some toxicity tests, but not during others; it is not clear whether a WER determined in a fed test will differ from a WER determined in an unfed test. Whether there is a difference is likely to depend on the metal, the type and amount of food, and whether a total recoverable or dissolved WER is determined. This can be evaluated by determining two WERs using a test in which the organisms usually are not fed - one WER with no food added to the tests and one with food added to the tests. Any effect of food is probably due to an increase in TOC and/or TSS. If food increases the concentration of nontoxic metal in both the laboratory dilution water and the site water, the food will probably decrease the WER. Because complexes of metals are usually soluble, complexation is likely to lower both total recoverable and dissolved WERs; sorption to solids will probably reduce only total recoverable WERs. The food might also affect the acute-chronic ratio. Any feeding during a test should be limited to the minimum necessary.

Ranges of Actual Measured WERs

The acceptable WERs found by Brungs et al. (1992) were total recoverable WERs that were determined in relatively clean fresh water. These WERs ranged from about 1 to 15 for both copper and cadmium, whereas they ranged from about 0.7 to 3 for zinc. The few WERs that were available for chromium, lead, and nickel ranged from about 1 to 6. Both the total recoverable and dissolved WERs for copper in New York harbor range from about 0.4 to 4 with most of the WERs being between 1 and 2 (Hansen 1993b).

Figure D2: An Example of the Empirical Extrapolation Process

Assume the following hypothetical effluent and upstream water:

Effluent:

T_E:	100 ug/L	
D_E:	10 ug/L	(10 % dissolved)
Q_E:	24 cfs	

Upstream water:

T_U:	40 ug/L	
D_U:	38 ug/L	(95 % dissolved)
Q_U:	48 cfs	

Downstream water:

T_D:	60 ug/L	
D_D:	36 ug/L	(60 % dissolved)
Q_D:	72 cfs	

where:

T = concentration of total recoverable metal.
D = concentration of dissolved metal.
Q = flow.

The subscripts E, U, and D signify effluent, upstream water, and downstream water, respectively.

By conservation of flow: $Q_D = Q_E + Q_U$.

By conservation of total recoverable metal: $T_D Q_D = T_E Q_E + T_U Q_U$.

If P = the percent of the total recoverable metal in the effluent that becomes dissolved in the downstream water,

$$P = \frac{100\,(D_D Q_D - D_U Q_U)}{T_E Q_E} \ .$$

For the data given above, the percent of the total recoverable metal in the effluent that becomes dissolved in the downstream water is:

$$P = \frac{100[(36\ ug/L)(72\ cfs) - (38\ ug/L)(48\ cfs)]}{(100\ ug/L)(24\ cfs)} = 32\ \%\ ,$$

which is greater than the 10 % dissolved in the effluent and less than the 60 % dissolved in the downstream water.

Figure D3: The Internal Consistency of the Two Approaches

The internal consistency of the total recoverable and dissolved approaches can be illustrated by considering the use of WERs to interpret the total recoverable and dissolved concentrations of a metal in a site water. For this hypothetical example, it will be assumed that the national CCCs for the metal are:

200 ug/L as total recoverable metal.
160 ug/L as dissolved metal.

It will also be assumed that the concentrations of the metal in the site water are:

300 ug/L as total recoverable metal.
120 ug/L as dissolved metal.

The total recoverable concentration in the site water exceeds the national CCC, but the dissolved concentration does not.

The following results might be obtained if WERs are determined:

In Laboratory Dilution Water

Total recoverable LC50 = 400 ug/L.
% of the total recoverable metal that is dissolved = 80.
(This is based on the ratio of the national CCCs, which were determined in laboratory dilution water.)
Dissolved LC50 = 320 ug/L.

In Site Water

Total recoverable LC50 = 620 ug/L.
% of the total recoverable metal that is dissolved = 40.
(This is based on the data given above for site water).
Dissolved LC50 = 248 ug/L.

WERs

Total recoverable WER = (620 ug/L)/(400 ug/L) = 1.55
Dissolved WER = (248 ug/L)/(320 ug/L) = 0.775

Checking the Calculations

$$\frac{\textit{Total recoverable WER}}{\textit{Dissolved WER}} = \frac{1.55}{0.775} = \frac{\textit{lab water \% dissolved}}{\textit{site water \% dissolved}} = \frac{80}{40} = 2$$

Site-specific CCCs (ssCCCs)

Total recoverable ssCCC = (200 ug/L)(1.55) = 310 ug/L.
Dissolved ssCCC = (160 ug/L)(0.775) = 124 ug/L.

Both concentrations in site water are below the respective ssCCCs.

In contrast, the following results might have been obtained when the WERs were determined:

In Laboratory Dilution Water
Total recoverable LC50 = 400 ug/L.
% of the total recoverable metal that is dissolved = 80.
Dissolved LC50 = 320 ug/L.

In Site Water
Total recoverable LC50 = 580 ug/L.
% of the total recoverable metal that is dissolved = 40.
Dissolved LC50 = 232 ug/L.

WERs
Total recoverable WER = (580 ug/L)/(400 ug/L) = 1.45
Dissolved WER = (232 ug/L)/(320 ug/L) = 0.725

Checking the Calculations

$$\frac{\textit{Total recoverable WER}}{\textit{Dissolved WER}} = \frac{1.45}{0.725} = \frac{\textit{lab water \% dissolved}}{\textit{site water \% dissolved}} = \frac{80}{40} = 2$$

Site-specific CCCs (ssCCCs)

Total recoverable ssCCC = (200 ug/L)(1.45) = 290 ug/L.
Dissolved ssCCC = (160 ug/L)(0.725) = 116 ug/L.

In this case, both concentrations in site water are above the respective ssCCCs.

In each case, both approaches resulted in the same conclusion concerning whether the concentration in site water exceeds the site-specific criterion.

The two key assumptions are:
1. The ratio of total recoverable metal to dissolved metal in laboratory dilution water when the WERs are determined equals the ratio of the national CCCs.
2. The ratio of total recoverable metal to dissolved metal in site water when the WERs are determined equals the ratio of the concentrations reported in the site water.

Differences in the ratios that are outside the range of experimental variation will cause problems for the derivation of site-specific criteria and, therefore, with the internal consistency of the two approaches.

Figure D4: The Application of the Two Approaches

Hypothetical upstream water and effluent will be used to demonstrate the equivalence of the total recoverable and dissolved approaches. The upstream water and the effluent will be assumed to have specific properties in order to allow calculation of the properties of the downstream water, which will be assumed to be a 1:1 mixture of the upstream water and effluent. It will also be assumed that the ratios of the forms of the metal in the upstream water and in the effluent do not change when the total recoverable concentration changes.

<u>Upstream water</u> (Flow = 3 cfs)

Total recoverable:	400 ug/L	
Refractory particulate:	200 ug/L	
Toxic dissolved:	200 ug/L	(50 % dissolved)

<u>Effluent</u> (Flow = 3 cfs)

Total recoverable:	440 ug/L	
Refractory particulate:	396 ug/L	
Labile nontoxic particulate:	44 ug/L	
Toxic dissolved:	0 ug/L	(0 % dissolved)

(The labile nontoxic particulate, which is 10 % of the total recoverable in the effluent, becomes toxic dissolved in the downstream water.)

<u>Downstream water</u> (Flow = 6 cfs)

Total recoverable:	420 ug/L	
Refractory particulate:	298 ug/L	
Toxic dissolved:	122 ug/L	(29 % dissolved)

The values for the downstream water are calculated from the values for the upstream water and the effluent:

Total recoverable:	[3(400) + 3(440)]/6	= 420 ug/L
Dissolved:	[3(200) + 3(44+0)]/6	= 122 ug/L
Refractory particulate:	[3(200) + 3(396)]/6	= 298 ug/L

<u>Assumed National CCC (nCCC)</u>

Total recoverable = 300 ug/L

Dissolved = 240 ug/L

Upstream site-specific CCC (ussCCC)

Assume: Dissolved cccWER = 1.2
Dissolved ussCCC = (1.2)(240 ug/L) = 288 ug/L
By calculation: TR ussCCC = (288 ug/L)/(0.5) = 576 ug/L
Total recoverable cccWER = (576 ug/L)/(300 ug/L) = 1.92

	nCCC	cccWER	ussCCC	Conc.
Total recoverable:	300 ug/L	1.92	576 ug/L	400 ug/L
Dissolved:	240 ug/L	1.2	288 ug/L	200 ug/L
% dissolved	80 %	----	50 %	50 %

Neither concentration exceeds its respective ussCCC.

$$\frac{\textit{Total recoverable WER}}{\textit{Dissolved WER}} = \frac{1.92}{1.2} = \frac{\textit{lab water \% dissolved}}{\textit{site water \% dissolved}} = \frac{80}{50} = 1.6$$

Downstream site-specific CCC (dssCCC)

Assume: Dissolved cccWER = 1.8
Dissolved dssCCC = (1.8)(240 ug/L) = 432 ug/L
By calculation: TR dssCCC =
{(432 ug/L-[(200 ug/L)/2])/0.1}+{(400 ug/L)/2} = 3520 ug/L
This calculation determines the amount of dissolved metal contributed by the effluent, accounts for the fact that ten percent of the total recoverable metal in the effluent becomes dissolved, and adds the total recoverable metal contributed by the upstream flow.
Total recoverable cccWER = (3520 ug/L)/(300 ug/L) = 11.73

	nCCC	cccWER	dssCCC	Conc.
Total recoverable:	300 ug/L	11.73	3520 ug/L	420 ug/L
Dissolved:	240 ug/L	1.80	432 ug/L	122 ug/L
% dissolved	80 %	----	12.27 %	29 %

Neither concentration exceeds its respective dssCCC.

$$\frac{\textit{Total recoverable WER}}{\textit{Dissolved WER}} = \frac{11.73}{1.80} = \frac{\textit{lab water \% dissolved}}{\textit{site water \% dissolved}} = \frac{80}{12.27} = 6.52$$

Calculating the Maximum Acceptable Concentration in the Effluent

Because neither the total recoverable concentration nor the dissolved concentration in the downstream water exceeds its respective site-specific CCC, the concentration of metal in the effluent could be increased. Under the assumption that the ratios of the two forms of the metal in the effluent do not change when the total recoverable concentration changes, the maximum acceptable concentration of total recoverable metal in the effluent can be calculated as follows:

Starting with the total recoverable dssCCC of 3520 ug/L

$$\frac{(6\ cfs)(3520\ ug/L)\ -\ (3\ cfs)(400\ ug/L)}{3\ cfs} = 6640\ ug/L$$

Starting with the dissolved dssCCC of 432 ug/L

$$\frac{(6\ cfs)(432\ ug/L)\ -\ (3\ cfs)(400\ ug/L)(0.5)}{(3\ cfs)(0.10)} = 6640\ ug/L$$

Checking the Calculations

Total recoverable:

$$\frac{(3\ cfs)(6640\ ug/L)\ +\ (3\ cfs)(400\ ug/L)}{6\ cfs} = 3520\ ug/L\ .$$

Dissolved:

$$\frac{(3\ cfs)(6640\ ug/L)(0.10)\ +\ (3\ cfs)(400\ ug/L)(0.50)}{6\ cfs} = 432\ ug/L\ .$$

The value of 0.10 is used because this is the percent of the total recoverable metal in the effluent that becomes dissolved in the downstream water.

The values of 3520 ug/L and 432 ug/L equal the downstream site-specific CCCs derived above.

Another Way to Calculate the Maximum Acceptable Concentration

The maximum acceptable concentration of total recoverable metal in the effluent can also be calculated from the dissolved dssCCC of 432 ug/L using a partition coefficient to convert from the dissolved dssCCC of 432 ug/L to the total recoverable dssCCC of 3520 ug/L:

$$\frac{[6\ cfs]\left[\frac{432\ ug/L}{0.1227}\ -\ (3\ cfs)(400\ ug/L)\right]}{3\ cfs} = 6640\ ug/L\ .$$

Note that the value used for the partition coefficient in this calculation is 0.1227 (the one that applies to the downstream water when the total recoverable concentration of metal in the effluent is 6640 ug/L), not 0.29 (the one that applies when the concentration of metal in the effluent is only 420 ug/L). The three ways of calculating the maximum acceptable concentration give the same result if each is used correctly.

Figure D5: A Generalized Complexation Curve

The curve is for a constant concentration of the complexing ligand and an increasing concentration of the metal.

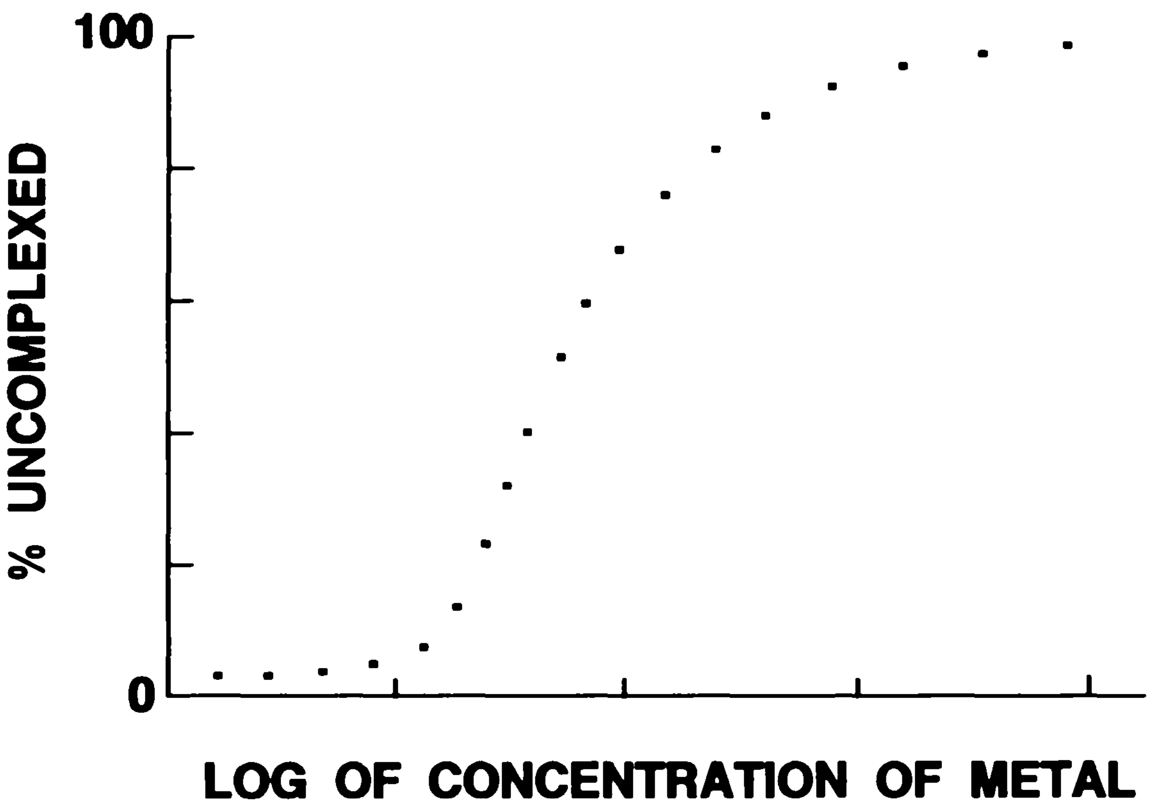
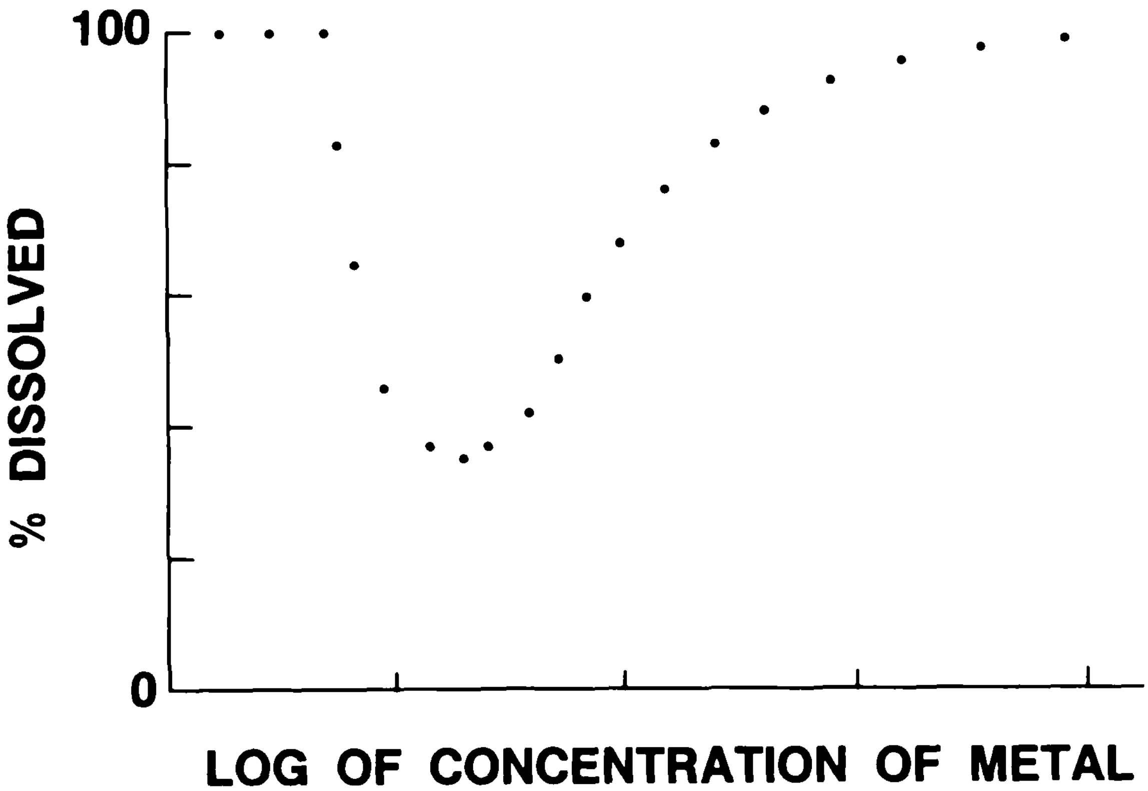


Figure D6: A Generalized Precipitation Curve

The curve is for a constant concentration of the precipitating ligand and an increasing concentration of the metal.



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Appendix E: U.S. EPA Aquatic Life Criteria Documents for Metals

Metal	EPA Number	NTIS Number
Aluminum	EPA 440/5-86-008	PB88-245998
Antimony	EPA 440/5-80-020	PB81-117319
Arsenic	EPA 440/5-84-033	PB85-227445
Beryllium	EPA 440/5-80-024	PB81-117350
Cadmium	EPA 440/5-84-032	PB85-227031
Chromium	EPA 440/5-84-029	PB85-227478
Copper	EPA 440/5-84-031	PB85-227023
Lead	EPA 440/5-84-027	PB85-227437
Mercury	EPA 440/5-84-026	PB85-227452
Nickel	EPA 440/5-86-004	PB87-105359
Selenium	EPA 440/5-87-006	PB88-142237
Silver	EPA 440/5-80-071	PB81-117822
Thallium	EPA 440/5-80-074	PB81-117848
Zinc	EPA 440/5-87-003	PB87-153581

All are available from:

National Technical Information Service (NTIS)
5285 Port Royal Road
Springfield, VA 22161
TEL: 703-487-4650

Appendix F: Considerations Concerning Multiple-Metal, Multiple-Discharge, and Special Flowing-Water Situations

Multiple-Metal Situations

Both Method 1 and Method 2 work well in multiple-metal situations, although the amount of testing required increases as the number of metals increases. The major problem is the same for both methods: even when addition of two or more metals individually is acceptable, simultaneous addition of the two or more metals, each at its respective maximum acceptable concentration, might be unacceptable for at least two reasons:

1. Additivity or synergism might occur between metals.
2. More than one of the metals might be detoxified by the same complexing agent in the site water. When WERs are determined individually, each metal can utilize all of the complexing capacity; when the metals are added together, however, they cannot simultaneously utilize all of the complexing capacity.

Thus a discharger might feel that it is cost-effective to try to justify the lowest site-specific criterion that is acceptable to the discharger rather than trying to justify the highest site-specific criterion that the appropriate regulatory authority might approve.

There are two options for dealing with the possibility of additivity and synergism between metals:

a. WERs could be developed using a mixture of the metals but it might be necessary to use several primary toxicity tests depending on the specific metals that are of interest. Also, it might not be clear what ratio of the metals should be used in the mixture.
b. If a WER is determined for each metal individually, one or more additional toxicity tests **must** be conducted at the end to show that the combination of all metals at their proposed new site-specific criteria is acceptable. Acceptability **must** be demonstrated with each toxicity test that was used as a primary toxicity test in the determination of the WERs for the individual metals. Thus if a different primary test was used for each metal, the number of acceptability tests needed would equal the number of metals. It is possible that a toxicity test used as the primary test for one metal might be more sensitive than the CMC (or CCC) for another metal and thus might not be usable in the combination test unless antagonism occurs. When a primary test cannot be used, an acceptable alternative test **must** be used.

The second option is preferred because it is more definitive; it provides data for each metal individually and for the mixture. The first option leaves the possibility that one of the metals is antagonistic towards another so that the toxicity of the mixture would increase if the metal causing the antagonism were not present.

Multiple-Discharge Situations

Because the National Toxics Rule (NTR) incorporated WERs into the aquatic life criteria for some metals, it might be envisioned that more than one criterion could apply to a metal at a site if different investigators obtained different WERs for the same metal at the site. In jurisdictions subject to the NTR, as well as in all other jurisdictions, EPA intends that there should be no more than one criterion for a pollutant at a point in a body of water. Thus whenever a site-specific criterion is to be derived using a WER at a site at which more than one discharger has permit limits for the same metal, it is important that all dischargers work together with the appropriate regulatory authority to develop a workplan that is designed to derive a site-specific criterion that adequately protects the entire site.

Method 2 is ideally suited for taking into account more than one discharger.

Method 1 is straightforward if the dischargers are sufficiently far downstream of each other that the stream can be divided into a separate site for each discharger. Method 1 can also be fairly straightforward if the WERs are additive, but it will be complex if the WERs are not additive. Deciding whether to use a simulated downstream water or an actual downstream water can be difficult in a flowing-water multiple-discharge situation. Use of actual downstream water can be complicated by the existence of multiple mixing zones and plumes and by the possibility of varying discharge schedules; these same problems exist, however, if effluents from two or more discharges are used to prepare simulated downstream water. Dealing with a multiple-discharge situation is much easier if the WERs are additive, and use of simulated downstream water is the best way to determine whether the WERs are additive. Taking into account all effluents will take into account synergism, antagonism, and additivity. If one of the discharges stops or is modified substantially, however, it will usually be necessary to determine a new WER, except possibly if the metal being discharged is refractory. Situations concerning intermittent and batch discharges need to be handled on a case-by-case basis.

Special Flowing-Water Situations

Method 1 is intended to apply not only to ordinary rivers and streams but also to streams that some people might consider "special", such as streams whose design flows are zero and streams that some state and/or federal agencies might refer to as "effluent-dependent", "habitat-creating", "effluent-dominated", etc. (Due to differences between agencies, some streams whose design flows are zero are not considered "effluent-dependent", etc., and some "effluent-dependent" streams have design flows that are greater than zero.) The application of Method 1 to these kinds of streams has the following implications:

1. If the design flow is zero, at least some WERs ought to be determined in 100% effluent.
2. If thunderstorms, etc., occasionally dilute the effluent substantially, at least one WER should be determined in diluted effluent to assess whether dilution by rainwater might result in underprotection by decreasing the WER faster than it decreases the concentration of the metal. This might occur, for example, if rainfall reduces hardness, alkalinity, and pH substantially. This might not be a concern if the WER demonstrates a substantial margin of safety.
3. If the site-specific criterion is substantially higher than the national criterion, there should be increased concern about the fate of the metal that has reduced or no toxicity. Even if the WER demonstrates a substantial margin of safety (e.g., if the site-specific criterion is three times the national criterion, but the experimentally determined WER is 11), it might be desirable to study the fate of the metal.
4. If the stream merges with another body of water and a site-specific criterion is desired for the merged waters, another WER needs to be determined for the mixture of the waters.
5. Whether WET testing is required is not a WER issue, although WET testing might be a condition for determining and/or using a WER.
6. A concern about what species should be present and/or protected in a stream is a beneficial-use issue, not a WER issue, although resolution of this issue might affect what species should be used if a WER is determined. (If the Recalculation Procedure is used, determining what species should be present and/or protected is obviously important.)
7. Human health and wildlife criteria and other issues might restrict an effluent more than an aquatic life criterion.

Although there are no scientific reasons why "effluent-dependent", etc., streams and streams whose design flows are zero should be subject to different guidance than other streams, a regulatory decision (for example, see 40 CFR 131) might require or allow some or all such streams to be subject to different guidance. For example, it might be decided on the basis of a use attainability analysis that one or more constructed streams do not have to comply with usual aquatic life criteria because it is decided that the water quality in such streams does not need to protect sensitive aquatic species. Such a decision might eliminate any further concern for site-specific aquatic life criteria and/or for WET testing for such streams. The water quality might be unacceptable for other reasons, however.

In addition to its use with rivers and streams, Method 1 is also appropriate for determining cmcWERs that are applicable to near-field effects of discharges into large bodies of fresh or salt water, such as an ocean or a large lake, reservoir, or estuary:

a. The near-field effects of a pipe that extends far into a large body of fresh or salt water that has a current, such as an ocean, can probably best be treated the same as a single discharge into a flowing stream. For example, if a mixing zone is defined, the concentration of effluent at the edge of the mixing zone might be used to define how to prepare a simulated site water. A dye dispersion study (Kilpatrick 1992) might be useful, but a dilution model (U.S. EPA 1993) is likely to be a more cost-effective way of obtaining information concerning the amount of dilution at the edge of the mixing zone.
b. The near-field effects of a single discharge that is near a shore of a large body of fresh or salt water can also probably best be treated the same as a single discharge into a flowing stream, especially if there is a definite plume and a defined mixing zone. The potential point of impact of near-field effects will often be an embayment, bayou, or estuary that is a nursery for fish and invertebrates and/or contains commercially important shellfish beds. Because of their importance, these areas should receive special consideration in the determination and use of a WER, taking into account sources of water and discharges, mixing patterns, and currents (and tides in coastal areas). The current and flushing patterns in estuaries can result in increased pollutant concentrations in confined embayments and at the terminal up-gradient portion of the estuary due to poor tidal flushing and exchange. Dye dispersion studies (Kilpatrick 1992) can be used to determine the spatial concentration of the effluent in the receiving water, but dilution models (U.S. EPA 1993) might not be sufficiently accurate to be useful. Dye studies of discharges in near-shore tidal areas are especially complex. Dye injection into the discharge should occur over at least one, and preferably two or three, complete tidal cycles; subsequent dispersion patterns should be monitored in the ambient water on consecutive tidal cycles using an intensive sampling regime over time, location, and depth. Information concerning dispersion and the community at risk can be used to define the appropriate mixing zone(s), which might be used to define how to prepare simulated site water.

References

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Appendix G: Additivity and the Two Components of a WER Determined Using Downstream Water

The Concept of Additivity of WERs

In theory, whenever samples of effluent and upstream water are taken, determination of a WER in 100 % effluent would quantify the effluent WER (eWER) and determination of a WER in 100 % upstream water would quantify the upstream WER (uWER); determination of WERs in known mixtures of the two samples would demonstrate whether the eWER and the uWER are additive. For example, if eWER = 40, uWER = 5, and the two WERs are additive, a mixture of 20 % effluent and 80 % upstream water would give a WER of 12, except possibly for experimental variation, because:

$$\frac{20(eWER)\ +\ 80(uWER)}{100} = \frac{20(40)\ +\ 80(5)}{100} = \frac{800\ +\ 400}{100} = \frac{1200}{100} = 12\ .$$

Strict additivity of an eWER and an uWER will probably be rare because one or both WERs will probably consist of a portion that is additive and a portion that is not. The portions of the eWER and uWER that are due to refractory metal will be strictly additive, because a change in water quality will not make the metal more or less toxic. In contrast, metal that is nontoxic because it is complexed by a complexing agent such as EDTA will not be strictly additive because the % uncomplexed will decrease as the solution is diluted; the amount of change in the % uncomplexed will usually be small and will depend on the concentration and the binding constant of the complexing agent (see Appendix D). Whether the nonrefractory portions of the uWER and eWER are additive will probably also depend on the differences between the water quality characteristics of the effluent and the upstream water, because these will determine the water quality characteristics of the downstream water. If, for example, 85 % of the eWER and 30 % of the uWER are due to refractory metal, the WER obtained in the mixture of 20 % effluent and 80 % upstream water could range from 8 to 12. The WER of 8 would be obtained if the only portions of the eWER and uWER that are additive are those due to refractory metal, because:

$$\frac{20(0.85)(eWER)\ +\ 80(0.30)(uWER)}{100} = \frac{20(0.85)(40)\ +\ 80(0.30)(5)}{100} = 8\ .$$

The WER could be as high as 12 depending on the percentages of the other portions of the WERs that are also additive. Even if the eWER and uWER are not strictly additive, the concept of additivity of WERs can be useful insofar as the eWER and uWER are partially additive, i.e., insofar as a portion of at least one of the WERs is additive. In the example given above, the WER determined using downstream water that consisted of 20 % effluent and 80 % upstream water would be 12 if the eWER and uWER were strictly additive; the downstream WER would be less than 12 if the eWER and uWER were partially additive.

The Importance of Additivity

The major advantage of additivity of WERs can be demonstrated using the effluent and upstream water that were used above. To simplify this illustration, the acute-chronic ratio will be assumed to be large, and the eWER of 40 and the uWER of 5 will be assumed to be cccWERs that will be assumed to be due to refractory metal and will therefore be strictly additive. In addition, the complete-mix downstream water at design-flow conditions will be assumed to be 20 % effluent and 80 % upstream water, so that the downstream WER will be 12 as calculated above for strict additivity.

Because the eWER and the uWER are cccWERs and are strictly additive, this metal will cause neither acute nor chronic toxicity in downstream water if (a) the concentration of metal in the effluent is less than 40 times the CCC and (b) the concentration of metal in the upstream water is less than 5 times the CCC. As the effluent is diluted by mixing with upstream water, both the eWER and the concentration of metal will be diluted simultaneously; proportional dilution of the metal and the eWER will prevent the metal from causing acute or chronic toxicity at any dilution. When the upstream flow equals the design flow, the WER in the plume will decrease from 40 at the end of the pipe to 12 at complete mix as the effluent is diluted by upstream water; because this WER is due to refractory metal, neither fate processes nor changes in water quality characteristics will affect the WER. When stream flow is higher or lower than design flow, the complete-mix WER will be lower or higher, respectively, than 12, but toxicity will not occur because the concentration of metal will also be lower or higher.

If the eWER and the uWER are strictly additive and if the national CCC is 1 mg/L, the following conclusions are valid when the concentration of the metal in 100 % effluent is less than 40 mg/L and the concentration of the metal in 100 % upstream water is less than 5 mg/L:

1. This metal will not cause acute or chronic toxicity in the upstream water, in 100 % effluent, in the plume, or in downstream water.
2. There is no need for an acute or a chronic mixing zone where a lesser degree of protection is provided.
3. If no mixing zone exists, there is no discontinuity at the edge of a mixing zone where the allowed concentration of metal decreases instantaneously.

These results also apply to partial additivity as long as the concentration of metal does not exceed that allowed by the amount of additivity that exists. It would be more difficult to take into account the portions of the eWER and uWER that are not additive.

The concept of additivity becomes unimportant when the ratios, concentrations of the metals, or WERs are very different. For example, if eWER = 40, uWER = 5, and they are additive, a mixture of 1 % effluent and 99 % upstream water would have a WER of 5.35. Given the reproducibility of toxicity tests and WERs, it would be extremely difficult to distinguish a WER of 5 from a WER of 5.35. In cases of extreme dilution, rather than experimentally determining a WER, it is probably acceptable to use the limiting WER of 5 or to calculate a WER if additivity has been demonstrated.

Traditionally it has been believed that it is environmentally conservative to use a WER determined in upstream water (i.e., the uWER) to derive a site-specific criterion that applies downstream (i.e., that applies to areas that contain effluent). This belief is probably based on the assumption that a larger WER would be obtained in downstream water that contains effluent, but the belief could also be based on the assumption that the uWER is additive. It is possible that in some cases neither assumption is true, which means that using a uWER to derive a downstream site-specific criterion might result in underprotection. It seems likely, however, that WERs determined using downstream water will usually be at least as large as the uWER.

Several kinds of concerns about the use of WERs are actually concerns about additivity:

1. Do WERs need to be determined at higher flows in addition to being determined at design flow?
2. Do WERs need to be determined when two bodies of water mix?
3. Do WERs need to be determined for each additional effluent in a multiple-discharge situation.

In each case, the best use of resources might be to test for additivity of WERs.

Mixing Zones

In the example presented above, there would be no need for a regulatory mixing zone with a reduced level of protection if:

1. The eWER is always 40 and the concentration of the metal in 100 % effluent is always less than 40 mg/L.
2. The uWER is always 5 and the concentration of the metal in 100 % upstream water is always less than 5 mg/L.
3. The WERs are strictly additive.

If, however, the concentration exceeded 40 mg/L in 100 % effluent, but there is some assimilative capacity in the upstream water, a regulatory mixing zone would be needed if the discharge were to be allowed to utilize some or all of the assimilative capacity. The concept of additivity of WERs can be used to calculate the maximum allowed concentration of the metal in the effluent if the eWER and the uWER are strictly additive.

If the concentration of metal in the upstream water never exceeds 0.8 mg/L, the discharger might want to determine how much above 40 mg/L the concentration could be in 100 % effluent. If, for example, the downstream water at the edge of the chronic mixing zone under design-flow conditions consists of 70 % effluent and 30 % upstream water, the WER that would apply at the edge of the mixing zone would be:

$$\frac{70(eWER) + 30(uWER)}{100} = \frac{70(40) + 30(5)}{100} = \frac{2800 + 150}{100} = 29.5 \ .$$

Therefore, the maximum concentration allowed at this point would be 29.5 mg/L. If the concentration of the metal in the upstream water was 0.8 mg/L, the maximum concentration allowed in 100 % effluent would be 41.8 mg/L because:

$$\frac{70(41.8 \ mg/L) + 30(0.8 \ mg/L)}{100} = \frac{2926 \ mg/L + 24 \ mg/L}{100} = 29.5 \ mg/L \ .$$

Because the eWER is 40, if the concentration of the metal in 100 % effluent is 41.8 mg/L, there would be chronic toxicity inside the chronic mixing zone. If the concentration in 100 % effluent is greater than 41.8 mg/L, there would be chronic toxicity past the edge of the chronic mixing zone. Thus even if the eWER and the uWER are taken into account and they are assumed to be completely additive, a mixing zone is necessary if the assimilative capacity of the upstream water is used to allow discharge of more metal.

If the complete-mix downstream water consists of 20 % effluent and 80 % upstream water at design flow, the complete-mix WER would be 12 as calculated above. The complete-mix approach to determining and using downstream WERs would allow a maximum concentration of 12 mg/L at the edge of the chronic mixing zone, whereas the alternative approach resulted in a maximum allowed concentration of 29.5 mg/L. The complete-mix approach would allow a maximum concentration of 16.8 mg/L in the effluent because:

$$\frac{70(16.8 \ mg/L) + 30(0.8 \ mg/L)}{100} = \frac{1176 \ mg/L + 24 \ mg/L}{100} = 12 \ mg/L \ .$$

In this example, the complete-mix approach limits the concentration of the metal in the effluent to 16.8 mg/L, even though it is known that as long as the concentration in 100 % effluent is less than 40 mg/L, chronic toxicity will not occur inside or outside the mixing zone. If the WER of 12 is used to derive a site-specific CCC of 12 mg/L that is applied to a site that starts at the edge of the chronic mixing zone and extends all the way across the stream, there would be overprotection at the edge of the chronic mixing zone (because the maximum allowed concentration is 12 mg/L, but a concentration of 29.5 mg/L will not cause chronic toxicity), whereas there would be underprotection on the other side of the stream (because the maximum allowed concentration is 12 mg/L, but concentrations above 5 mg/L can cause chronic toxicity.)

The Experimental Determination of Additivity

Experimental variation makes it difficult to quantify additivity without determining a large number of WERs, but the advantages of demonstrating additivity might be sufficient to make it worth the effort. It should be possible to decide whether the eWER and uWER are strictly additive based on determination of the eWER in 100 % effluent, determination of the uWER in 100 % upstream water, and determination of WERs in 1:3, 1:1, and 3:1 mixtures of the effluent and upstream water, i.e., determination of WERs in 100, 75, 50, 25, and 0 % effluent. Validating models of partial additivity and/or interactions will probably require determination of more WERs and more sophisticated data analysis (see, for example, Broderius 1991).

In some cases chemical measurements or manipulations might help demonstrate that at least some portion of the eWER and/or the uWER is additive:

1. If the difference between the dissolved WER and the total recoverable WER is explained by the difference between the dissolved and total recoverable concentrations, the difference is probably due to particulate refractory metal.
2. If the WERs in different samples of the effluent correlate with the concentration of metal in the effluent, all, or nearly all, of the metal in the effluent is probably nontoxic.
3. A WER that remains constant as the pH is lowered to 6.5 and raised to 9.0 is probably additive.

The concentration of refractory metal is likely to be low in upstream water except during events that increase TSS and/or TOC; the concentration of refractory metal is more likely to be substantial in effluents. Chemical measurements might help identify the percentages of the eWER and the uWER that are due to refractory metal, but again experimental variation will limit the usefulness of chemical measurements when concentrations are low.

Summary

The distinction between the two components of a WER determined using downstream water has the following implications:

1. The magnitude of a WER determined using downstream water will usually depend on the percent effluent in the sample.

2. Insofar as the eWER and uWER are additive, the magnitude of a downstream WER can be calculated from the eWER, the uWER, and the ratio of effluent and upstream water in the downstream water.
3. The derivation and implementation of site-specific criteria should ensure that each component is applied only where it occurs.
   a. Underprotection will occur if, for example, any portion of the eWER is applied to an area of a stream where the effluent does not occur.
   b. Overprotection will occur if, for example, an unnecessarily small portion of the eWER is applied to an area of a stream where the effluent occurs.
4. Even though the concentration of metal might be higher than a criterion in both a regulatory mixing zone and a plume, a reduced level of protection is allowed in a mixing zone, whereas a reduced level of protection is not allowed in the portion of a plume that is not inside a mixing zone.
5. Regulatory mixing zones are necessary if, and only if, a discharger wants to make use of the assimilative capacity of the upstream water.
6. It might be cost-effective to quantify the eWER and uWER, determine the extent of additivity, study variability over time, and then decide how to regulate the metal in the effluent.

Reference

Broderius, S.J. 1991. Modeling the Joint Toxicity of Xenobiotics to Aquatic Organisms: Basic Concepts and Approaches. In: Aquatic Toxicology and Risk Assessment: Fourteenth Volume. (M.A. Mayes and M.G. Barron, eds.) ASTM STP 1124. American Society for Testing and Materials, Philadelphia, PA. pp. 107-127.

Appendix H: Special Considerations Concerning the Determination of WERs with Saltwater Species

1. The test organisms should be compatible with the salinity of the site water, and the salinity of the laboratory dilution water should match that of the site water. Low-salinity stenohaline organisms should not be tested in high-salinity water, whereas high-salinity stenohaline organisms should not be tested in low-salinity water; it is not known, however, whether an incompatibility will affect the WER. If the community to be protected principally consists of euryhaline species, the primary and secondary toxicity tests should use the euryhaline species suggested in Appendix I (or taxonomically related species) whenever possible, although the range of tolerance of the organisms should be checked.
   a. When Method 1 is used to determine cmcWERs at saltwater sites, the selection of test organisms is complicated by the fact that most effluents are freshwater and they are discharged into salt waters having a wide range of salinities. Some state water quality standards require a permittee to meet an LC50 or other toxicity limit at the end of the pipe using a freshwater species. However, the intent of the site-specific and national water quality criteria program is to protect the communities that are at risk. Therefore, freshwater species should not be used when WERs are determined for saltwater sites unless such freshwater species (or closely related species) are in the community at risk. The addition of a small amount of brine and the use of salt-tolerant freshwater species is inappropriate for the same reason. The addition of a large amount of brine and the use of saltwater species that require high salinity should also be avoided when salinity is likely to affect the toxicity of the metal. Salinities that are acceptable for testing euryhaline species can be produced by dilution of effluent with sea water and/or addition of a commercial sea salt or a brine that is prepared by evaporating site water; small increases in salinity are acceptable because the effluent will be diluted with salt water wherever the communities at risk are exposed in the real world. Only as a last resort should freshwater species that tolerate low levels of salinity and are sensitive to metals, such as Daphnia magna and Hyalella azteca, be used.
   b. When Method 2 is used to determine cccWERs at saltwater sites:
      1) If the site water is low-salinity but all the sensitive test organisms are high-salinity stenohaline organisms, a commercial sea salt or a brine that is prepared by evaporating site water may be added in order to increase the salinity to the minimum level that is acceptable to the test organisms; it should be determined whether the

salt or brine reduces the toxicity of the metal and thereby increases the WER.

2) If the site water is high-salinity, selecting test organisms should not be difficult because many of the sensitive test organisms are compatible with high-salinity water.

2. It is especially important to consider the availability of test organisms when saltwater species are to be used, because many of the commonly used saltwater species are not cultured and are only available seasonally.

3. Many standard published methodologies for tests with saltwater species recommend filtration of dilution water, effluent, and/or test solutions through a 37-μm sieve or screen to remove predators. Site water should be filtered only if predators are observed in the sample of the water because filtration might affect toxicity. Although recommended in some test methodologies, ultraviolet treatment is often not needed and generally should be avoided.

4. If a natural salt water is to be used as the laboratory dilution water, the samples should probably be collected at slack high tide ($\pm$ 2 hours). Unless there is stratification, samples should probably be taken at mid-depth; however, if a water quality characteristic, such as salinity or TSS, is important, the vertical and horizontal definition of the point of sampling might be important. A conductivity meter, salinometer, and/or transmissometer might be useful for determining where and at what depth to collect the laboratory dilution water; any measurement of turbidity will probably correlate with TSS.

5. The salinity of the laboratory dilution water should be within $\pm$ 10 percent or 2 mg/L (whichever is higher) of that of the site water.

Appendix I: Suggested Toxicity Tests for Determining WERs for Metals

Selecting primary and secondary toxicity tests for determining WERs for metals should take into account the following:

1. WERs determined with more sensitive tests are likely to be larger than WERs determined with less sensitive tests (see Appendix D). Criteria are derived to protect sensitive species and so WERs should be derived to be appropriate for sensitive species. The appropriate regulatory authority will probably accept WERs derived with less sensitive tests because such WERs are likely to provide at least as much protection as WERs determined with more sensitive tests.
2. The species used in the primary and secondary tests **must** be in different orders and should include a vertebrate and an invertebrate.
3. The test organism (i.e., species and life stage) should be readily available throughout the testing period.
4. The chances of the test being successful should be high.
5. The relative sensitivities of test organisms vary substantially from metal to metal.
6. The sensitivity of a species to a metal usually depends on both the life stage and kind of test used.
7. Water quality characteristics might affect chronic toxicity differently than they affect acute toxicity (Spehar and Carlson 1984; Chapman, unpublished; Voyer and McGovern 1991).
8. The endpoint of the primary test in laboratory dilution water should be as close as possible (but **must not** be below) the CMC or CCC to which the WER is to be applied; the endpoint of the secondary test should be as close as possible (and should not be below) the CMC or CCC.
9. Designation of tests as acute and chronic has no bearing on whether they may be used to determine a cmcWER or a cccWER.

The suggested toxicity tests should be considered, but the actual selection should depend on the specific circumstances that apply to a particular WER determination.

Regardless of whether test solutions are renewed when tests are conducted for other purposes, if the concentrations of dissolved metal and dissolved oxygen remain acceptable when determining WERs, tests whose duration is not longer than 48 hours may be static tests, whereas tests whose duration is longer than 48 hours **must** be renewal tests. If the concentration of dissolved metal and/or the concentration of dissolved oxygen does not remain acceptable, the test solutions **must** be renewed every 24 hours. If one test in a pair of side-by-side tests is a renewal test, both of the tests **must** be renewed on the same schedule.

Appendix H should be read if WERs are to be determined with saltwater species.

Suggested Tests[1] for Determining cmcWERs and cccWERs[2].
(Concentrations are to be measured in all tests.)

Metal	Water[3]	cmcWERs[4]		cccWERs[4]	
Aluminum	FW	DA	X	CDC	X
Arsenic(III)	FW	DA	GM	CDC	FMC
	SW	BM	CR	MYC	BM
Cadmium	FW	DA	SL[5] or FM	CDC	FMC
	SW	MY	CR	MYC	X
Chrom(III)	FW	GM	SL or DA	FMC	CDC
Chrom(VI)	FW	DA	GM	CDC	GM
	SW	MY	NE	MYC	NEC
Copper	FW	DA	FM or GM	CDC	FM
	SW	BM	AR	BMC	AR
Lead	FW	DA	GM	CDC	X
	SW	BM	MYC	MYC	X
Mercury	FW	DA	GM	Y	Y
	SW	MY	BM	Y	Y
Nickel	FW	DA	FX	CDC	FMC
	SW	MY	BM	MYC	BMC
Selenium	FW	Y	Y	Y	Y
	SW	CR	MYC	MYC	X
Silver	FW	DA	FMC	CDC	FMC
	SW	BM	CR	MYC	BMC
Zinc	FW	DA	FM	CDC	FMC
	SW	BM	MY	MYC	BMC

1 The description of a test specifies not only the test species and the duration of the test but also the life stage of the species and the adverse effect(s) on which the endpoint is to be based.

2 Some tests that are sensitive and are used in criteria documents are not suggested here because the chances of the test organisms being available and the test being successful might be low. Such tests may be used if desired.

[3] FW = Fresh Water; SW = Salt Water.

[4] Two-letter codes are used for acute tests, whereas codes for chronic tests contain three letters and end in "C". One-letter codes are used for comments.

[5] In acute tests on cadmium with salmonids, substantial numbers of fish usually die after 72 hours. Also, the fish are sensitive to disturbance, and it is sometimes difficult to determine whether a fish is dead or immobilized.

ACUTE TESTS

AR. A 48-hr EC50 based on mortality and abnormal development from a static test with embryos and larvae of sea urchins of a species in the genus Arbacia (ASTM 1993a) or of the species Strongylocentrotus purpuratus (Chapman 1992).

BM. A 48-hr EC50 based on mortality and abnormal larval development from a static test with embryos and larvae of a species in one of four genera (Crassostrea, Mulinia, Mytilus, Mercenaria) of bivalve molluscs (ASTM 1993b).

CR. A 48-hr EC50 (or LC50 if there is no immobilization) from a static test with Acartia or larvae of a saltwater crustacean; if molting does not occur within the first 48 hours, renew at 48 hours and continue the test to 96 hours (ASTM 1993a).

DA. A 48-hr EC50 (or LC50 if there is no immobilization) from a static test with a species in one of three genera (Ceriodaphnia, Daphnia, Simocephalus) in the family Daphnidae (U.S. EPA 1993a; ASTM 1993a).

FM. A 48-hr LC50 from a static test at 25°C with fathead minnow (Pimephales promelas) larvae that are 1 to 24 hours old (ASTM 1993a; U.S. EPA 1993a). The embryos **must** be hatched in the laboratory dilution water, except that organisms to be used in the site water may be hatched in the site water. The larvae **must not** be fed before or during the test and at least 90 percent **must** survive in laboratory dilution water for at least six days after hatch.

Note: The following 48-hr LC50s were obtained at a hardness of 50 mg/L with fathead minnow larvae that were 1 to 24 hours old. The metal was measured using the total recoverable procedure (Peltier 1993):

Metal	LC50 (μg/L)
Cadmium	13.87
Copper	6.33
Zinc	100.95

FX. A 96-hr LC50 from a renewal test (renew at 48 hours) at 25°C with fathead minnow (Pimephales promelas) larvae that are 1 to 24 hours old (ASTM 1993a; U.S. EPA 1993a). The embryos **must** be hatched in the laboratory dilution water, except that organisms to be used in the site water may be hatched in the site water. The larvae **must not** be fed before or during the test and at least 90 percent **must** survive in laboratory dilution water for at least six days after hatch.

Note: A 96-hr LC50 of 188.14 μg/L was obtained at a hardness of 50 mg/L in a test on nickel with fathead minnow larvae that were 1 to 24 hours old. The metal was measured using the total recoverable procedure (Peltier 1993). A 96-hr LC50 is used for nickel because substantial mortality occurred after 48 hours in the test on nickel, but not in the tests on cadmium, copper, and zinc.

GM. A 96-hr EC50 (or LC50 if there is no immobilization) from a renewal test (renew at 48 hours) with a species in the genus Gammarus (ASTM 1993a).

MY. A 96-hr EC50 (or LC50 if there is no immobilization) from a renewal test (renew at 48 hours) with a species in one of two genera (Mysidopsis, Holmesimysis [nee Acanthomysis]) in the family Mysidae (U.S. EPA 1993a; ASTM 1993a). Feeding is required during all acute and chronic tests with mysids; for determining WERs, mysids should be fed four hours before the renewal at 48 hours and minimally on the non-renewal days.

NE. A 96-hr LC50 from a renewal test (renew at 48 hours) using juvenile or adult polychaetes in the genus Nereidae (ASTM 1993a).

SL. A 96-hr EC50 (or LC50 if there is no immobilization) from a renewal test (renew at 48 hours) with a species in one of two genera (Oncorhynchus, Salmo) in the family Salmonidae (ASTM 1993a).

CHRONIC TESTS

BMC. A 7-day IC25 from a survival and development renewal test (renew every 48 hours) with a species of bivalve mollusc, such as a species in the genus Mulinia. One such test has been described by Burgess et al. 1992. [Note: When determining WERs, sediment **must not** be in the test chamber.] [Note: This test has not been widely used.]

CDC. A 7-day IC25 based on reduction in survival and/or reproduction in a renewal test with a species in the genus Ceriodaphnia in the family Daphnidae (U.S. EPA 1993b). The test solutions **must** be renewed every 48 hours. (A 21-day life-cycle test with Daphnia magna is also acceptable.)

FMC. A 7-day IC25 from a survival and growth renewal test (renew every 48 hours) with larvae (≤ 48-hr old) of the fathead minnow (Pimephales promelas) (U.S. EPA 1993b). When determining WERs, the fish **must** be fed four hours before each renewal and minimally during the non-renewal days.

MYC. A 7-day IC25 based on reduction in survival, growth, and/or reproduction in a renewal test with a species in one of two genera (Mysidopsis, Holmesimysis [nee Acanthomysis]) in the family Mysidae (U.S. EPA 1993c). Mysids **must** be fed during all acute and chronic tests; when determining WERs, they **must** be fed four hours before each renewal. The test solutions **must** be renewed every 24 hours.

NEC. A 20-day IC25 from a survival and growth renewal test (renew every 48 hours) with a species in the genus Neanthes (Johns et al. 1991). [Note: When determining WERs, sediment **must not** be in the test chamber.] [Note: This test has not been widely used.]

COMMENTS

X. Another sensitive test cannot be identified at this time, and so other tests used in the criteria document should be considered.

Y. Because neither the CCCs for mercury nor the freshwater criterion for selenium is based on laboratory data concerning toxicity to aquatic life, they cannot be adjusted using a WER.

REFERENCES

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U.S. EPA. 1993c. Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Marine and Estuarine Organisms. Second Edition. EPA/600/4-91/003. National Technical Information Service, Springfield, VA.

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Appendix J: Recommended Salts of Metals

The following salts are recommended for use when determining a WER for the metal listed. If available, a salt that meets American Chemical Society (ACS) specifications for reagent-grade should be used.

Aluminum
*Aluminum chloride 6-hydrate: $AlCl_3 \bullet 6H_2O$
Aluminum sulfate 18-hydrate: $Al_2(SO_4)_3 \bullet 18H_2O$
Aluminum potassium sulfate 12-hydrate: $AlK(SO_4)_2 \bullet 12H_2O$

Arsenic(III)
*Sodium arsenite: $NaAsO_2$

Arsenic(V)
Sodium arsenate 7-hydrate, dibasic: $Na_2HAsO_4 \bullet 7H_2O$

Cadmium
Cadmium chloride 2.5-hydrate: $CdCl_2 \bullet 2.5H_2O$
Cadmium sulfate hydrate: $3CdSO_4 \bullet 8H_2O$

Chromium(III)
*Chromic chloride 6-hydrate (Chromium chloride): $CrCl_3 \bullet 6H_2O$
*Chromic nitrate 9-hydrate (Chromium nitrate): $Cr(NO_3)_3 \bullet 9H_2O$
Chromium potassium sulfate 12-hydrate: $CrK(SO_4)_2 \bullet 12H_2O$

Chromium(VI)
Potassium chromate: K_2CrO_4
Potassium dichromate: $K_2Cr_2O_7$
*Sodium chromate 4-hydrate: $Na_2CrO_4 \bullet 4H_2O$
Sodium dichromate 2-hydrate: $Na_2Cr_2O_7 \bullet 2H_2O$

Copper
*Cupric chloride 2-hydrate (Copper chloride): $CuCl_2 \bullet 2H_2O$
Cupric nitrate 2.5-hydrate (Copper nitrate): $Cu(NO_3)_2 \bullet 2.5H_2O$
Cupric sulfate 5-hydrate (Copper sulfate): $CuSO_4 \bullet 5H_2O$

Lead
*Lead chloride: $PbCl_2$
Lead nitrate: $Pb(NO_3)_2$

Mercury
Mercuric chloride: $HgCl_2$
Mercuric nitrate monohydrate: $Hg(NO_3)_2 \bullet H_2O$
Mercuric sulfate: $HgSO_4$

Nickel
*Nickelous chloride 6-hydrate (Nickel chloride): $NiCl_2 \bullet 6H_2O$
*Nickelous nitrate 6-hydrate (Nickel nitrate): $Ni(NO_3)_2 \bullet 6H_2O$
Nickelous sulfate 6-hydrate (Nickel sulfate): $NiSO_4 \bullet 6H_2O$

Selenium(IV)
*Sodium selenite 5-hydrate: $Na_2SeO_3 \bullet 5H_2O$

Selenium(VI)
*Sodium selenate 10-hydrate: $Na_2SeO_4 \bullet 10H_2O$

Silver
Silver nitrate: $AgNO_3$
(Even if acidified, standards and samples containing silver **must** be in amber containers.)

Zinc
Zinc chloride: $ZnCl_2$
*Zinc nitrate 6-hydrate: $Zn(NO_3)_2 \bullet 6H_2O$
Zinc sulfate 7-hydrate: $ZnSO_4 \bullet 7H_2O$

*Note: ACS reagent-grade specifications might not be available for this salt.

No salt should be used until information concerning the safety and handling of that salt has been read.

APPENDIX M

Reserved

APPENDIX M

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

APPENDIX N

Integrated Risk Information System Background Paper

IRIS

BACKGROUND PAPER

Integrated Risk Information System

Office of Health and Environmental Assessment

Office of Research and Development

FEBRUARY, 1993

VERSION 1.0

IRIS Background Paper

On February 25, 1993, a FEDERAL REGISTER notice (58 FR 11490) was published on the Integrated Risk Information System (IRIS). This background paper is a companion piece to that notice.

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For further information on IRIS, please contact:

IRIS User Support
(Operated by Computer Sciences Corporation)
26 W. Martin Luther King Drive (MS-190)
Cincinnati, OH 45268

Telephone (513) 569-7254 Facsimile (513) 569-7916

Introduction

This background paper provides the history, purposes, and goals of the Integrated Risk Information System (IRIS) and a detailed description of the current processes used by the two Agency scientific work groups responsible for developing the health hazard information in IRIS. This background will help interested persons to better understand the focus and contents of the companion FEDERAL REGISTER notice.

The February 25, 1993 FEDERAL REGISTER notice (58 FR 11490): (1) announces the availability of this paper that describes IRIS, its contents, and the current processes used by the two Agency work groups responsible for developing IRIS information; (2) discusses an Agency activity to review IRIS processes and solicits comments on this review; (3) highlights points in the current process where public input, including information submissions, is encouraged; (4) describes how to access IRIS; and (5) announces a new process to publish regularly a list of the substances scheduled for IRIS work group review and to solicit pertinent data, studies, and comments on these substances.

General Background

IRIS is an EPA data base, updated monthly, containing Agency consensus positions on the potential adverse human health effects of approximately 500 specific substances. It contains summaries of EPA qualitative and quantitative human health information that support two of the four major steps of the risk assessment process outlined in the National Research Council's (NRC) 1983 publication, "*Risk Assessment in the Federal Government: Managing the Process.*"

The risk assessment process described in the 1983 NRC publication consists of four major steps: hazard identification, dose-response evaluation, exposure assessment, and risk characterization. IRIS includes information in support of the first two of those steps, hazard identification and dose-response evaluation. Hazard identification is the qualitative determination of how likely it is that a substance will increase the incidence and/or severity of an adverse health effect. Dose-response evaluation is the quantitative relationship between the magnitude of the effect and the dose inducing such an effect. IRIS information supporting risk characterization consists of brief statements on the quality of data and very general statements on confidence in the dose-response evaluation. IRIS consensus information does not include exposure assessment information. Combined with specific situational exposure assessment information, the summary health hazard information in IRIS may be used as one source in evaluating potential public health risks of or from environmental contaminants.

Many EPA program offices and program support offices, including the Office of Research and Development, both at Headquarters and in EPA's ten Regional offices, are involved in assessment activities in support of various legislative mandates. In the 1980s, as health risk assessment became a more widespread practice across Agency programs, the need became clear for greater consensus and consistency in the areas of hazard identification and dose-response assessment. It was determined that an internal process should be established for reaching an Agency-wide judgment on the potential health effects of substances of common interest to these offices, and a system developed for communicating that Agency judgment to EPA risk assessors and risk managers. These would provide the needed consistency and coordination. In 1986, two EPA work groups with representation from program offices involved in risk assessment were convened to carry out such an internal process to reach consensus Agency positions on a chemical-by-chemical basis. In 1986, the IRIS data base was created for EPA staff as the official repository of that consensus information.

On June 2, 1988, a FEDERAL REGISTER notice (53 FR 20162-20164) of public availability of IRIS was published. That notice described IRIS, the types of risk information it contains, and how to get access to the system. It informed the public about the establishment of the IRIS Information Submission Desk. The submission desk was intended to provide opportunity for public input. The notice explained the procedures for submission of data or comments by interested parties on substances either on IRIS or scheduled for review by the work groups. As stated in the June 1988 notice, a list of the substances scheduled for work group review has been a separate file on IRIS since it became publicly available. It was hoped that users would submit pertinent information to the IRIS Information Submission Desk. In fact, few users have taken advantage of the opportunity to submit data and comments.

Therefore, data submission procedures are reiterated in the FEDERAL REGISTER notice (58 FR 11490) related to this paper and a list of the substances scheduled for review by specific work groups is included. The data submission procedures will be reprinted in the FEDERAL REGISTER every 6 months with a new or revised list of substances scheduled for work group review. For the latest status of the substances scheduled for review, interested persons should first check the IRIS data base itself or contact:

IRIS User Support (Operated by Computer Sciences Corporation)
U.S. EPA
26 W. Martin Luther King Drive (MS-190)
Cincinnati, OH 45268
Telephone: (513) 569-7254 Facsimile: (513) 569-7916

Data Base Contents

The core of IRIS is the three consensus health hazard information summary sections: the reference dose for noncancer health effects resulting from oral exposure, the reference concentration for noncancer health effects resulting from inhalation exposure, and the carcinogen assessment for both oral and inhalation exposure. All of these terms are commonly used for judging the effects of lifetime exposure to a given substance or mixture. Citations for the scientific methodologies that are the basis for the consensus health hazard sections on IRIS are included on page 10 of this paper.

In addition, an IRIS substance file may include supplemental information such as summaries of health advisories, regulatory actions, and physical/chemical properties.

Noncancer Health Effects Information

An oral reference dose (RfD) is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily oral exposure to the human population (including sensitive subgroups) that is believed likely to be without an appreciable risk of certain deleterious effects during a lifetime ("Reference Dose [RfD]; Description and Use in Health Risk Assessment" *Regulatory Toxicology and Pharmacology* 8:471-486, 1988). RfDs are developed by an assessment method that assumes that there is a dose threshold below which adverse effects will not occur. An RfD, which is expressed in milligrams per kilogram per day (mg/kg-day), is based on the determination of a critical effect from a review of all toxicity data and a judgment of the necessary uncertainty and modifying factors based on a review of available data. IRIS substance files contain the following information pertaining to the oral RfD: reference dose summary tables, principal and supporting studies, uncertainty and modifying factors used in calculating the RfD, a statement of confidence in the RfD, EPA documentation and review, EPA scientific contacts, and complete bibliographies for references cited.

The inhalation reference concentration (RfC) is analogous to the oral RfD (*Interim Methods for Development of Inhalation Concentrations*, EPA/600/8-90/066A). It is also based on the assumption that thresholds exist for noncancer toxic effects. The RfC considers toxic effects for both the respiratory system (portal-of-entry) and for effects peripheral to the respiratory system (extra-respiratory). The inhalation RfC is expressed in milligrams per cubic meter (mg/cu.m). The RfC method departs from that used to determine the oral RfD primarily by the integration of the anatomical and physiological dynamics of the respiratory system (i.e., portal-of-entry) with the physicochemical properties of the substance or substances entering the system. Different dosimetric adjustments are made according to whether the substance is a particle or gas and whether the observed toxicity is respiratory or extra-respiratory. These adjustments scale the concentration of the substance that causes an observed effect in laboratory animals (or in humans, when available from occupational epidemiology studies) to a human equivalent concentration for ambient exposures.

IRIS substance files contain the following inhalation RfC information: reference concentration summary tables, description of dosimetric adjustment, principal and supporting studies, uncertainty and modifying factors used to calculate the RfC, a statement of confidence in the RfC, EPA documentation and review, EPA scientific contacts, and complete bibliographies for references cited.

Cancer Health Effects Information

The carcinogen assessment of an IRIS substance file contains health hazard identification and dose-response assessments developed from procedures outlined in the EPA Guidelines for Carcinogen Risk Assessment (51 FR 33992-43003, September 24, 1986). Each cancer assessment, as a rule, is based on an Agency document that has received external peer review. The hazard identification involves a judgment in the form of a weight-of-evidence classification of the likelihood that the substance is a human carcinogen. It includes the type of data used as the basis of the classification. This judgment is made independently of considerations of the strength of the possible response. The dose-response assessment is a quantitative estimate of the potential activity or magnitude of a substance's carcinogenic effect, usually expressed as a cancer unit risk. A cancer unit risk is an upper-bound estimate on the increased likelihood that an individual will develop cancer when exposed to a substance over a lifetime at a concentration of either 1 microgram per liter (1 μg/L) in drinking water for oral exposure or 1 microgram per cubic meter (1 μg/cu.m) in air for continuous inhalation exposure. Generally, a slope factor for dietary use is also given. It is an upper-bound estimate of cancer risk for humans per milligram of agent per kilogram of body weight per day.

IRIS contains the following information in the cancer assessment section: EPA weight-of-evidence classification and its basis, a summary of human carcinogenicity studies when available, a summary of animal carcinogenicity studies, a summary of other data supporting the classification, oral and/or inhalation quantitative estimates, dose-response data used to derive these estimates and the method of calculation, statements of confidence in magnitude of unit risk, documentation and review, EPA scientific contacts, and complete bibliographies for references cited.

Scientific Contacts

It is important to note that in each of the three sections described above, EPA staff names and telephone numbers are included as scientific contacts for further information. The Agency believes that the inclusion of Agency scientific contacts able to discuss the basis for the Agency's position, has been very valuable. These individuals play a major role in providing public access to IRIS and a conduit for valued public comment.

Bibliographies

IRIS contains full bibliographic citations for each substance file, directing the user to the primary cited studies and pertinent scientific literature. One of the major intents of IRIS was to encourage users to evaluate the primary literature used to develop the IRIS information in light of the assumptions and uncertainties underlying the risk assessment process.

Supplementary Information

In addition to the RfD, RfC, and carcinogenicity sections, IRIS substance files may contain one or more of three supplementary information sections: a summary of an Office of Water's Drinking Water Health Advisory, a summary of EPA regulatory actions, and a summary of physical/chemical properties. The only purpose of these supplemental sections is to serve as accessory information to the consensus health hazard information. Since the primary intent of the IRIS data base is to communicate EPA consensus health hazard information, these other sections are only included as auxiliary material to provide a broader profile of a substance and are never added until at least one of the consensus health hazard sections described above (namely, the RfD section, RfC section, or carcinogenicity section) is prepared and approved for final inclusion on the data base. These supplemental sections should not be used as the sole or primary source of information on the current status of EPA substance-specific regulations.

Use and Development of Health Hazard Information

The type of substance-specific consensus health hazard information on IRIS may become part of the supporting materials used to develop site-specific EPA health hazard assessments. These assessments may in turn lead to EPA risk management decisions, generally resulting in the formal Agency rulemaking process. This rulemaking process often includes FEDERAL REGISTER publication of a proposed rule where the public is encouraged to comment. These comments may be directed at both the proposed rule and the scientific basis of the decision, including information obtained from IRIS and thus offer a further opportunity for comment on the risk information in the context of its use.

The area of human health risk assessment has evolved over the past several years. As the risk assessment community has grown and the field itself has matured, new approaches to the assessment and use of human health risk information have been developed. The evolving nature of risk assessment has also resulted in changes to IRIS. The development of methodologies such as those for the inhalation RfC determination illustrates the ability of the IRIS information development process to grow with the changing science. Areas of future growth may include less-than-lifetime risk information and developmental toxicity risk information and other endpoint-specific health hazard information. Also, on several occasions, the information in IRIS has been reevaluated and modified to reflect new information and approaches. New studies on individual substances are continually being conducted by Federal, private, and academic institutions and may have significant impact on IRIS information. In those cases, the IRIS substance information is reevaluated in light of the new data; any changes resulting from that reevaluation are included on the system.

Management of the Data Base

The IRIS data base is managed and maintained by the Office of Health and Environmental Assessment (OHEA), Office of Research and Development (ORD). IRIS is an Agency system primarily funded by OHEA with additional significant support from EPA program offices.

Oversight

Oversight activities for IRIS are conducted by the IRIS Oversight Committee, a subgroup of the Agency's Risk Assessment Council. Committee membership consists of senior Agency risk assessors. The main purpose of the IRIS Oversight Committee is to serve as a forum for discussion and advice on significant scientific or science policy issues involving IRIS. The Council, which is chaired by EPA's Deputy Administrator, receives periodic status reports on IRIS and related work group activities.

Information Development Process

There are two EPA work groups, the Carcinogen Risk Assessment Verification Endeavor (CRAVE) and the Oral Reference Dose/Inhalation Reference Concentration (RfD/RfC) Work Group, that develop consensus health hazard information for IRIS. Each group consists of EPA scientists from a mix of pertinent disciplines and represents intra-Agency membership. The work groups serve as the Agency's final review for EPA risk assessment information. When the work groups reach consensus on the health effects information and the dose-response assessment for a particular substance, the descriptive summary is added to IRIS.

CRAVE: Information Development Procedures

The goals of the CRAVE are to reach Agency consensus on Agency carcinogen risk assessments; to arrive at a unified view on potential cancer risk from exposure to specific substances across Agency programs; and to identify, discuss, and resolve general issues associated with methods used to estimate carcinogenic risks for specific agents. The major outputs of the work group are summaries of risk information that have been previously developed and documented by scientific experts in Agency program and program support offices, and results of discussions of general issues in carcinogen risk assessment.

Scientists are selected by executive appointment from respective member offices. Membership is open to all major Agency program and regional offices, ORD, and the Office of Policy, Planning, and Evaluation (OPPE). Substances are discussed at the request of Agency offices or regions according to an established timetable. The CRAVE priorities are determined by the member offices. The office requesting review prepares a summary describing both a judgment on the weight-of-evidence for potential health hazard effects and any dose-response information for the substances according to an established format. Literature files on the substances including critical studies, pertinent EPA documents, and other relevant supporting documentation are made available to work group members in advance of the meeting. Generally, the judgment and the dose-response assessment are expected to have appeared in a publicly available document of some sort.

The CRAVE usually meets bimonthly for two days. Work group members normally receive draft summaries for pre-meeting review at least one week prior to the scheduled meeting. At the meeting, data and documentation are examined, and there is discussion of the basis for the risk information and the methods by which it was derived. In addition, the nature and extent of previous internal and external peer review, including the comments received, are reviewed by the work group. The summary is revised by the office originating the review to reflect the meeting discussion and accurately express the consensus view of the work group. After the process of revision is completed, the summary is circulated again to the work group for final approval prior to its inclusion on IRIS.

Consensus means that no member office is aware either of information that would conflict with the final carcinogenicity summary, or of analyses that would suggest that a different view is more credible. Such assurance rests on the capabilities of the individuals who represent their offices; thus, every effort is made to seek scientists who are both expert in the area of human health assessment and who can represent their office.

Peer review has generally been part of the IRIS information development processes from the beginning of the system. In the preparation of summaries, emphasis has been placed on the use of peer-reviewed EPA assessments. These have included Office of Pesticide Programs assessments that have received both program office peer review and Science Advisory Panel review. Other EPA documentation includes assessments prepared by OHEA such as Health Assessment Documents, Health and Environmental Effects Documents, and Health Effects Assessments. These documents receive OHEA review and program office review and some receive Science Advisory Board (SAB) or other external review. Assessments developed by or for the Office of Ground Water and Drinking Water and incorporated in either Drinking Water or Ambient Water Criteria Documents, or in Drinking Water Health Advisories generally receive extensive Agency review and SAB review prior to discussion by CRAVE.

On occasion, risk assessments that were contained in draft documents have been discussed by CRAVE. In these instances, results of the work group deliberations have been incorporated into the document development process at the program office or program support office level. Loading of the information on IRIS is delayed pending completion of the document.

If consensus is not reached at the meeting it is generally because an issue is raised that requires resolution. Work group deliberations continue until consensus is achieved. In the case of substance-specific issues, the substance is referred back to the member office that initiated the review for more information and clarification. In some instances, it has been necessary for more than one program office to engage in a dialogue to resolve the issue.

For general issues, CRAVE practice has been to form a subcommittee to prepare an issue paper that is subsequently discussed at a special meeting. As examples of this process, issue papers have been developed for (1) issues relating to accuracy and precision of quantitative dose-response information, (2) factors involving confidence in quantitative estimates, and (3) use of split classifications and combining estimates.

When consensus is not achieved on a particular substance at a meeting of the CRAVE, it is considered to have "under review" status. If after three months, there is no further activity to bring the substance back to the work group for additional review, the substance loses its "under review" status. The substance is then dropped from the work group review list after notifying the responsible office. Any office may resubmit the substance for further discussion at any time.

<u>Reference Dose (RfD)/Reference Concentration (RfC): Information Development Procedures</u>

The purpose of the RfD/RfC Work Group is to reach consensus on oral RfDs and inhalation RfCs for noncancer chronic human health effects developed by or in support of program offices and the regions. The work group also works to resolve inconsistent RfDs or RfCs among program offices and to identify, discuss, and resolve generic issues associated with methods used to estimate RfDs and RfCs.

Scientists are selected by executive appointment from respective member offices. Membership is open to all major Agency program and regional offices. There are two work group co-chairs. In addition, scientists from the Agency for Toxic Substances and Disease Registry and the Food and Drug Administration are invited to work group meetings as observers to assist the Agency in the information gathering process. Their involvement fosters better communication and coordination among federal agencies regarding assessment approaches and data evaluation. Members reflect a variety of pertinent scientific disciplines including expertise in the fields of general and inhalation human toxicology.

Member offices schedule substances for discussion through the work group co-chairs for specific meetings, usually one or two months in advance. Regional requests for specific substance discussions are routed through the co-chairs, who then either schedule these substances in the usual manner or, if the region has not prepared a file, requests an appropriate office to undertake that task.

The RfD/RfC Work Group usually meets once a month for two days. Substances are discussed at the request of any Agency office or region. The requesting office generally prepares a file that consists of a summary sheet, a copy of the critical study and supporting documentation, and distributes these to work group members prior to the meeting.

Consensus generally means that no member office is aware either of information that would conflict with the RfD or RfC, or of analyses that would suggest a different value that is more credible. Such assurance rests on the capabilities of the individuals who represent their offices; thus, a large effort is conducted biannually to seek scientists who are both expert in this area of assessment and can represent their offices.

RfD or RfC summaries are not always based on existing EPA assessment documents but may be based on assessments prepared specifically for the work group. This is a fundamental difference between the usual processes of the RfD/RfC Work Group and those of CRAVE. As stated previously, the general rule has been that for a substance to be brought to the CRAVE Work Group for review there should be an existing peer-reviewed Agency health effects document. However, for RfDs there may or may not be an existing EPA document on which to base work group deliberations and in the case of RfCs, there have not, to date, been any existing peer-reviewed EPA documents. Thus, RfC deliberations are based on extensive assessment summaries prepared expressly for the work group. Therefore, when an Agency peer-reviewed document is not available, as with RfCs and some RfDs, extensive assessment summaries are included on IRIS once the work group has completed verification and reached consensus.

The work group co-chairs assure that the final summary accurately expresses the consensus view of the group at the meeting as specified in the meeting notes. Once unanimous consensus is reached, the substance-specific summary for either an RfD or RfC is prepared for inclusion on IRIS. In some cases, the work group agrees that adequate information is not available to derive an RfD or RfC. A message is then put on IRIS to that effect and the reasons for the "not verifiable" status. In most cases the message states that the health effects data for a specific substance were reviewed by the work group and determined to be inadequate for derivation of an RfD or RfC.

Conflicts that arise during a meeting regarding a given RfD or RfC generally are resolved outside the meeting by scientists from the appropriate offices, and then brought back to the work group for clarification and subsequent consensus. Conflicts that arise regarding the methods by which RfDs or RfCs are estimated, or the incorporation of new methods, are generally taken up at separately scheduled meetings of the work group, for which the sponsoring office prepares the appropriate material for review.

While, as discussed above, the RfD/RfC Work Group process is somewhat different from that of the CRAVE, they both use generally the same consensus procedures. Other procedural similarities are discussed in the following paragraphs.

On occasion, scientific issues on individual substances, methods, or on a general question cannot be resolved at the work group level. In the event that an issue is unresolvable in the work group processes, the issue is referred to the Risk Assessment Council. In some cases, the issue is brought to the IRIS Oversight Subcommittee for review and discussion, prior to consideration by the full Council. If an issue is raised to the Council, it may be referred by the Council to the Risk Assessment Forum for consultation.

Both the CRAVE and RfD/RfC Work Groups, through the IRIS Information Submission Desk, discussed in the companion FEDERAL REGISTER notice, have received comments and studies from interested parties outside of the Agency that were either pertinent to the work group's initial review or resulted in reconsideration of a particular substance assessment. Further, the work groups often contact the authors of a primary study if clarifications are necessary, and consult with outside experts on scientific issues that require expertise that is not present in the work group. Also, through professional societies and other private sector organizations, the work groups have fostered discussions and exchanges regarding new and innovative approaches to human health assessment methodologies.

Methods and Guidelines

Both Agency work groups responsible for the development of the health hazard information on IRIS use Agency scientific methods documents and EPA's risk assessment guidelines as the basis for their work. These guidelines and methodologies used to develop the RfD or RfC have been peer reviewed by the SAB.

Summaries of methods used for development of oral RfDs and carcinogenicity information on IRIS are contained in IRIS background documents that are available on the system. A paper copy of the oral RfD and CRAVE background documents, "Reference Dose (RfD); Description and Use in Health Risk Assessment" (*Regulatory Toxicology and Pharmacology* 8:471-486, 1988) and *The U.S. EPA Approach for Assessing the Risks Associated with Chronic Exposures to Carcinogens*, respectively, is also available from IRIS User Support by calling: (513) 569-7254.

The draft methods document, *Interim Methods for Development of Inhalation Concentrations* (EPA/600/8-90/066A), is the basis for the inhalation RfCs. A copy of the document is available from the Center for Environmental Research Information (CERI) by calling: (513) 569-7562. Please cite the EPA document number (EPA/600/8-90/066A) when requesting a copy. A revised RfC methodology document based on SAB peer-review comments will undergo a second SAB review and will be available later this year.

The CRAVE background document is based on EPA's 1986 Guidelines for Carcinogen Risk Assessment (51 FR 33992-34003). A copy of the EPA risk assessment guidelines (EPA/600/8-87/045) is also available by calling CERI.

Public Involvement

The section in the companion FEDERAL REGISTER notice (February 25, 1993, 58 FR 11490) on **Current Opportunities for Public Involvement in the IRIS Process** elaborates on opportunities for public input and dialogue.

APPENDIX O

Reserved

APPENDIX O

APPENDIX P

List of 126
CWA Section 307(a)
Priority Toxic Pollutants

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

126 Priority Pollutants

A. Chlorinated Benzenes
- Chlorobenzene
- 1,2-dichlorobenzene
- 1,3-dichlorobenzene
- 1,4-dichlorobenzene
- 1,2,4-trichlorobenzene
- Hexachlorobenzene

B. Chlorinated Ethanes
- Chloroethane
- 1,1-dichloroethane
- 1,2-dichloroethane
- 1,1,1-trichloroethane
- 1,1,2-trichloroethane
- 1,1,2,2-tetrachloroethane
- Hexachloroethane

C. Chlorinated Phenols
- 2-chlorophenol
- 2,4-dichlorophenol
- 2,4,6-trichlorophenol
- Parametachlorocresol (4-chloro-3-methyl phenol)

D. Other Chlorinated Organics
- Chloroform (trichloromethane)
- Carbon tetrachloride (tetrachloromethane)
- Bis(2-chloroethoxy)methane
- Bis(2-chloroethyl)ether
- 2-chloroethyl vinyl ether (mixed)
- 2-chloronaphthalene
- 3,3-dichlorobenzidine
- 1,1-dichloroethylene
- 1,2-trans-dichloroethylene
- 1,2-dichloropropane
- 1,2-dichloropropylene (1,3-dichloropropene)
- Tetrachloroethylene
- Trichloroethylene
- Vinyl chloride (chloroethylene)
- Hexachlorobutadiene
- Hexachlorocyclopentadiene
- 2,3,7,8-tetrachloro-dibenzo-p-dioxin (TCDD)

E. Haloethers
- 4-chlorophenyl phenyl ether
- 2-bromophenyl phenyl ether
- Bis(2-chloroisopropyl) ether

F. Halomethanes
- Methylene chloride (dichloromethane)
- Methyl chloride (chloromethane)

Methyl Bromide (bromomethane)
Bromoform (tribromomethane)
Dichlorobromomethane
Chlorodibromomethane

G. Nitrosamines
N-nitrosodimethylamine
N-nitrosodiphenylamine
N-nitrosodi-n-propylamine

H. Phenols (other than chlorinated)
2-nitrophenol
4-nitrophenol
2,4-dinitrophenol
4,6-dinitro-o-cresol (4,6-dinitro-2-methylphenol)
Pentachlorophenol
Phenol
2,4-dimethylphenol

I. Phthalate Esters
Bis(2-ethylhexyl)phthalate
Butyl benzyl phthalate
Di-N-butyl phthalate
Di-n-octyl phthalate
Diethyl phthalate
Dimethyl phthalate

J. Polnuclear Aromatic Hydrocarbons (PAHs)
Acenaphthene
1,2-benzanthracene (benzo(a) anthracene)
Benzo(a)pyrene (3,4-benzo-pyrene)
3,4-benzofluoranthene (benzo(b) fluoranthene)
11,12-benzofluoranthene (benzo(k) fluoranthene)
Chrysene
Acenaphthalene
Anthracene
1,12-benzoperylene (bonze(ghi) perylene)
Fluorene
Fluoranthene
Phenanthrene
1,2,5,6-bibenzanthracene (dibenzo(ah) anthracene)
Indeno (1,2,3-cd) pyrene (2,3-o-phenylene pyrene)
Pyrene

K. Pesticides and Metabolites
Aldrin
Dieldrin
Chlordane (technical mixture and metobolites)
Alpha-endosulfan
Beta-endosulfan
Endosulfan sulfate
Endrin
Endrin aldehyde
Heptachlor
Heptachlor epoxide (BHC-hexachlorocyclohexane)

Alpha-BHC
Beta-BHC
Gamma-BHC (Lindane)
Delta-BHC
Toxaphene

L. DDT and Metabolites
4,4-DDT
4,4-DDE (p,p-DDX)
4,4-DDD (p,p-TDE)

M. Polychlorinated Biphenyls (PCBs)
PCB-1242 (Arochlor 1242)
PCB-1254 (Arochlor 1254)
PCB-1221 (Arochlor 1221)
PCB-1232 (Arochlor 1232)
PCB-1248 (Arochlor 1248)
PCB-1260 (Arochlor 1260)
PCB-1016 (Arochlor 1016)

N. Other Organics
Acrolein
Acrylonitrile
Benzene
Benzidine
2,4-dinitrotoluene
2,6-dinitrotoluene
1,2-diphenylhydrazine
Ethylbenzene
Isophorone
Naphthalene
Nitrobenzene
Toluene

O. Inorganics
Antimony
Arsenic
Asbestos
Beryllium
Cadmium
Chromium
Copper
Cyanide, total
Lead
Mercury
Nickel
Selenium
Silver
Thallium
Zinc

APPENDIX Q

Wetlands and 401 Certification: Opportunities and Guidelines for States and Eligible Indian Tribes

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

United States
Environmental Protection
Agency

Office of Water
(A-104F)

April 1989

EPA

Wetlands And 401 Certification

Opportunities And Guidelines For States And Eligible Indian Tribes

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

JUN 28 1989

OFFICE OF
WATER

NOTE TO THE READER

I am pleased to introduce this handbook, "Wetlands and 401 Certification," developed by EPA's Office of Wetlands Protection. This document examines the Section 401 State water quality certification process and how it applies to wetlands. We strongly encourage States to use this handbook as one reference when establishing a wetlands protection program or improving wetlands protection tools.

Protection of wetland resources has become an important national priority as evidenced by President Bush's 1990 Budget statement calling for "no net loss" of wetlands. In addition, the National Wetlands Policy Forum included a recommendation in their 1988 report which says that States should "make more aggressive use of their certification authorities under Section 401 of the Clean Water Act, to protect wetlands from chemical and other types of alterations". This handbook is intended to help States do just that.

EPA would like to work with States who wish to delve into 401 certification for wetlands. You will find EPA Regional contacts listed in Appendix A of the document. The Office of Wetlands Protection plans to provide additional technical support including guidance focused on wetland-specific water quality standards.

It is very important to begin now to address the loss and degradation of this nation's wetlands. That is why 401 certification is a perfect tool, already in place, for States just getting started. It can also help States fill some gaps in their own statutory authorities protecting wetlands. States can make great strides using their existing 401 certification authorities, while developing the capability and the complementary programs to provide more comprehensive protection for wetlands in the future.

Sincerely,

David G. Davis
Director
Office of Wetlands Protection

ENDNOTES

1. The state water quality certification process is authorized by Section 401 of the Clean Water Act, 33 U.S.C. §1341.

2. A Tribe is eligible for treatment as a State if it meets the following criteria: 1) it is federally recognized; 2) it carries out substantial government duties and powers over a Federal Indian Reservation; 3) it has appropriate regulatory authority over surface waters of the reservation; and 4) it is reasonably expected to be capable of administering the relevant Clean Water Act program. EPA is currently developing regulations to implement Section 518(e) for programs including Section 401 certification which will provide further explanation of the process tribes must go through to achieve state status. In addition, the term "state" also includes the District of Columbia, the Commonwealth of Puerto Rico, the Virgin Islands, Guam, American Samoa, the Commonwealth of the Northern Mariana Islands, and the Trust Territory of the Pacific Islands.

3. The National Wetlands Policy Forum, chaired by Governor Kean of New Jersey, represents a very diverse group of perspectives concerned with policy issues to protect and manage the nation's wetland resources. The goal of the Forum was to develop sound, broadly supported recommendations to improve federal, state, and local wetlands policy. The Forum released its recommendations in a report, "Protecting America's Wetlands: An Action Agenda" which can be obtained from The Conservation Foundation, 1250 24th Street, NW, Washington, D.C. 20037.

4. 33 U.S.C. §4.1313 (c)(2)(A).

5. Section 301(b)(1)(c) of the Clean Water Act.

6. If the applicant is a federal agency, however, at least one federal court has ruled that the state's certification decision may be reviewed by the federal courts.

7. 33 C.F.R. §328.3 (Corps regulations); 40 C.F.R. §232.2(q) (EPA regulations).

8. For instance, except for wetlands designated as having unusual local importance, New York's freshwater wetlands law regulates only those wetlands over 12.4 acres in size.

9. Alaska Administrative Code, Title 6, Chapter 50.

10. Kentucky Environmental Protection Act, KRS 224.005(28).

11. Tennessee Water Quality Control Act, §69-3-103(29).

12. Massachusetts Clean Waters Act, Chapter 21, §26.

13. K.R.S. 224.005(28)(Kentucky enabling legislation defining waters of the state); 401 K.A.R. 5:029(1)(bb)(Kentucky water quality standards defining surface waters); Ohio Water Pollution Control Act, §6111.01(H)(enabling legislation defining waters of the state); Ohio Administrative Code, §3745-1-02(DDD) (water quality standards defining surface waters of the state).

14. Massachusetts Clean Waters Act, Chapter 21, §26 (enabling legislation defining waters of the state); 314 Code of Mass. Regs. 4.01(5)(water quality standards defining surface waters).

15. Ohio Administrative Code, 3745-32-01(N).

16. 40 C.F.R. §131.

17. A use attainability analysis (40 C.F.R. §131.10(g)) must show at least one of six factors in order to justify not meeting the minimum "fishable/swimmable" designated uses or to remove such a designated use. The analysis must show that attaining a use is not feasible because of: naturally occurring pollutant concentrations; natural flow conditions or water levels that cannot be made up by effluent discharges without violating state water conservation requirements; human caused pollution that cannot be remedied or that would cause more environmental damage if corrected; hydrologic modifications, if it is not feasible to restore the water to its original conditions or operate the modification to attain the use; natural non-water quality physical conditions precluding attainment of aquatic life protection uses; or controls more stringent than those required by §301(b) and §306 would result in substantial and widespread economic and social impact.

18. Questions and Answers on Antidegradation (EPA, 1985). this document is designated as Appendix A of Chapter 2 of EPA's Water Quality Standards Handbook.

19. The regulations implementing Section 404(b)(1) of the Clean Water Act are known as the "(b)(1) Guidelines" and are located at 40 C.F.R. §230.

20. 40 C.F.R. §230.1(d)

21. 40 C.F.R. §230.10(c).

22. Code of Maryland Regulations Title 10, §10.50.01.02(B)(2)(a).

23. Minnesota Rules, §7050.0170. The rule states in full:

> The waters of the state may, in a state of nature, have some characteristics or properties approaching or exceeding the limits specified in the water quality standards. The standards shall be construed as limiting the addition of pollutants of human activity to those of natural origin, where such be present, so that in total the specified limiting concentrations will not be exceeded in the waters by reason of such controllable additions. Where the background level of the natural origin is reasonably definable and normality is higher than the specified standards the natural level may be used as the standard for controlling the addition of pollutants of human activity which are comparable in nature and significance with those of natural origin. The natural background level may be used instead of the specified water quality standard as a maximum limit of the addition of pollutants, in those instances where the natural level is lower than the specified standard and reasonable justification exists for preserving the quality to that found in a state of nature.

24. No. 83-1352-I (Chancery Court, 7th Division, Davidson County, 1984)(unpublished opinion).

25. These criteria are at 401 K.A.R. 5:031, §2(4) and §4(1)(c), respectively.

26. Ohio Admin. Code, §3745-32-05.

27. Ohio Admin. Code, §3745-1-05(C).

28. Copies of Ohio's review guidelines are available from Ohio EPA, 401 Coordinator, Division of Water Quality Monitoring and Assessment, P.O. Box 1049, Columbus, Ohio 43266-0149.

29. 40 CFR §131.12.

30. 48 Fed. Reg. 51,400, 51,403 (1983)(preamble).

31. Kentucky Water Quality Standards, Title 401 K.A.R. 5:031, §7.

32. Minnesota Rules, §7050.0180, Subpart 7.

33. 314 Code of Massachusetts Regulation, §4.04(4).

34. Minnesota Rules, §7050.0180, Subpart 9.

35. H.R. Rep. No. 91-127, 91st Cong., 1st Sess. 6 (1969).

36. 115 Cong. Rec. H9030 (April 15, 1969)(House debate); 115 Cong. Rec. S28958-59 (Oct. 7, 1969)(Senate debate).

37. C.F.R. §323.2(d). However, in Reid v. Marsh, a case predating these regulations, the U.S. District Court for the Northern Corps District of Ohio ruled that "even minimal discharges of dredged material are not exempt from Section 404 review". In this district, the Corps treats all dredging projects under Section 404.

38. West Virginia Code, §47-5A-1 (emphasis added).

39. Clean Water Act, §401(a)(2).

40. 40 C.F.R. §230.10(a).

41. 40 C.F.R. §230.10(d).

42. Arnold Irrigation District v. Department of Environmental Quality, 717 Pac.Rptr.2d 1274 (Or.App. 1986).

43. Marmac Corporation v. Department of Natural Resources of the State of West Virginia, C.A. No. CA-81-1792 (Cir. Ct., Kanawha County 1982).

44. 33 U.S.C. §1313(c)(2)(A).

45. West Va. Admin. Code, §47-5A-9.3 (a).

46. Unpublished paper by Dr. Paul Hill of West Virginia's Department of Natural Resources. Prepared for EPA-sponsored December 1987 workshop on "The Role of Section 401 Certification in Wetlands Protection".

47. 33 C.F.R. §325.2(b)(ii).

48. 18 C.F.R. §4.38(e)(2).

49. 40 C.F.R. §124.53(c)(3).

50. Wisconsin Administrative Code, NR 299.04.

51. West Va. Admin. Code, §47-5A-4.3.

52. Id.

53. 40 C.F.R. §121.2. EPA's regulations implementing Section 401 were issued under the 1970 Water Pollution Control Act, (not the later Clean Water Act) and thus, may have some anomalies as a result.

54. This is a reference to Section 10 of the Rivers and Harbors Act.

55. Ohio Admin. Code, §3745-32-05.

56. See, e.g., P. Adamus, Wetland Evaluation Technique (WET), Volume II: Methodology Y-87(U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, MS, 1987); L. Cowardin, Classification of Wetlands and Deepwater Habitats of the United States (U.S. Fish and Wildlife Service 1979). See also Lonard and Clairain, Identification of Wetland Functions and Values, in Proceedings: National Wetlands Assessment Symposium (Chester, VT: Association of State Wetland Managers, 1986)(list of twenty five methodologies).

57. See, e.g., R. Tiner, Wetlands of the United States: Current Status and Recent Trends (U.S. Govt. Printing Office 1984)(National Wetlands Inventory). The National Wetlands Inventory has mapped approximately 45 percent of the lower forty eight states and 12 percent of Alaska. A number of regional and state reports may be obtained from the National Wetlands Inventory of the U.S. Fish and Wildlife Service in Newton Corner, MA. Region 5 maps can also be ordered from the U.S. Geological Survey's National Cartographic Information Center in Reston, VA.

58. The new joint Federal Manual for Identifying and Delineating Jurisdictional Wetlands, can be obtained from the U.S. Government Printing Office 1989).

59. See, e.g., Chesapeake Bay Critical Areas Commission, Guidance Paper No. 3, Guidelines for Protecting Non-Tidal Wetlands in the Critical Area (Maryland Department of Natural Resources, April 1987).

60. For information on the Wetlands Values Data Base contact: Data Base Administrator, U.S. Fish and Wildlife Service, National Energy Center, 2627 Redwing Road, Creekside One, Fort Collins, Colorado, 80526. Phone: (303) 226-9411.

61. For example, Florida's Section 380 process designates "Areas of Critical State Concern" which often include wetlands. Florida Statutes §380.05.

62. 40 C.F.R. §230.80 (1987).

63. 16 U.S.C. §1452(3) (1980). See also, U.S.Army Corps of Engineers, Regulatory Guidance Letter No. 10 (1986).

64. See D. Burke, Technical and Programmatic Support for 401 Certification in Maryland, (Maryland Department of Natural Resources, Water Resources Administration, December 1987)(unpublished); A. Lam, Geographic Information Systems for River Corridor and Wetland Management in River Corridor Handbook (N.Y.Department of Environmental Conservation)(J. Kusler and E. Meyers eds., 1988).

The system described by Burke is called MIPS (Map and Image Processing System) and is capable of translating a myriad of information to the scale specified by the user.

65. See, e.g., [multiple authors], "Ecological Considerations in Wetlands Treatment of Municipal Wastewaters," (Van Nostrand Reinhold Co., New York, 1985); E. Stockdale, "The Use of Wetlands for Stormwater Management and Nonpoint Pollution Control: A Review of the Literature," (Dept. of Ecology, State of Washington 1986); "Viability of Freshwater Wetlands for Urban Surface Water Management and Nonpoint Pollution: An Annotated Bibliography," prepared by The Resource Planning Section of King County, Washington Department of Planning and Community Development (July, 1986).

66. The Warren S. Henderson Wetlands Protection Act of 1984, Fla. Stat. §403.91 - 403.938, required the Florida Department of Environmental Regulation to establish specific criteria for wetlands that receive and treat domestic wastewater treated to secondary standards. The rule is at Fla. Admin. Code, §17-6.

67. Maximization of sheet flow.

68. Hydrologic loading and retention rates.

69. Id.; See also L. Schwartz, Criteria for Wastewater Discharge to Florida Wetlands, (Florida Department of Environmental Regulation)(Dec. 1987)(unpublished report).

70. Copies of the draft, "Use of Advance Identification Authorities under Section 404 of the Clean Water Act: Guidance for Regional Offices", can be obtained from the Regulatory Actitivities Division of the Office of Wetlands Protection (A-104F), EPA, 401 M Street, SW, Washington, D.C. 20460.

Acknowledgements:

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For additional copies contact:

Wetlands Strategies and State Programs Division
Office of Wetlands Protection A-104F
Environmental Protection Agency
401 M Street, SW
Washington, D.C. 20460

Phone: (202) 382-5043

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APPENDICES

I. INTRODUCTION

This handbook has been developed by EPA's Office of Wetlands Protection (OWP) to highlight the potential of the State water quality certification process for protecting wetlands, and to provide information and guidance to the States.[1] Throughout this document, the term "State" includes those Indian Tribes which qualify for treatment as States under the federal Clean Water Act (CWA) Section 518(e).[2] We encourage Tribes who are interested in expanding their protection of wetlands and other waters under this new provision of the CWA to examine water quality certification as a readily available tool to begin their programs.

One of OWP's key mandates is to broaden EPA's wetlands protection efforts in areas which complement our authority under the Clean Water Act Section 404 regulatory program. Thus, we are exploring and working with other laws, regulations, and nonregulatory approaches to enhance their implementation to protect wetlands. In addition, the National Wetlands Policy Forum has recommended in its report issued in November 1988, that States "make more aggressive use of their certification authorities under Section 401 of the CWA, to protect their wetlands from chemical and other types of alterations."[3]

In light of these directives, we have examined the role of the Section 401 State water quality certification process and are working with States to improve its application to wetlands. This process offers the opportunity to fulfill many goals for wetland protection because:

- **It is a cooperative federal/State program and it increases the role of States in decisions regarding the protection of natural resources;**

- **It gives States extremely broad authority to review proposed activities in and/or affecting State waters (including wetlands) and, in effect, to deny or place conditions on federal permits or licenses that authorize such activities;**

- **It is an existing program which can be vastly improved to protect wetlands without major legislative initiatives;**

- **Its proper implementation for wetlands should integrate many State programs related to wetlands, water quality, and aquatic resource preservation and enhancement, to ensure consistency of activities with these State requirements. Examples of such programs include coastal zone management, floodplain management, and nonpoint source programs.**

The issues discussed in this handbook were identified through discussions with State 401 certification program personnel and through a workshop held in December 1987 with many of the States who actively apply 401 certification to wetlands. The handbook includes examples of how some States have successfully approached the issues discussed. Because the water quality certification process is continually evolving, we do not attempt to address all the issues here. This handbook is a first step towards clarifying how 401 certification applies to wetlands, and helping States use this tool more effectively.

EPA would like to work with the States to ensure that their authority under Section 401 is exercised in a manner that achieves the goals of the Clean Water Act and reflects the State role at the forefront in administering water quality programs. Clearly, the integrity of waters of the U.S. cannot be protected by an exclusive focus on wastewater effluents in open waters. While the federal Section 404 program addresses many discharges into wetlands, and other federal agencies have environmental review programs which benefit wetlands, these do not substitute for a State's responsibilities under Section 401. A State's authority under Section 401 includes consideration of a broad range of chemical, physical, and biological impacts. The State's responsibility includes acting upon the recognition that wetlands are critical components of healthy, functioning aquatic systems.

To help States implement the guidance provided in this handbook and to foster communication on 401 issues, you will find a list of State 401 certification contacts and federal EPA contacts in Appendix A. In order to keep this and other wetland contact lists current, EPA has asked the Council of State Governments to establish a computerized database of State wetland programs and contacts (See Appendix A for details.) EPA is also refining a list of Tribal contacts to foster communication with interested Tribes.

SUMMARY OF ACTIONS NEEDED

The following is a summary of the activities needed to make 401 certification a more effective tool to protect wetlands. States can undertake many of these activities right away, while also taking other actions which lay the groundwork for improving future 401 certification decisions. Tribes, who primarily are just beginning to develop wetlands programs, should consider these actions (along with developing water quality standards) as first steps to becoming more involved in wetlands regulatory efforts. The actions below are discussed throughout the handbook.

- **All states should begin by including wetlands in their definitions of state waters.**

- **States should develop or modify their existing 401 certification and water quality standard regulations and guidelines to accomodate special wetland considerations.**

- **States should make more effective use of their existing narrative water quality standards (including the antidegradation policy) to protect the integrity of wetlands.**

- **States should initiate or improve upon existing inventories of their wetland resources.**

- **States should designate uses for these wetlands based on wetland functions associated with each wetland type. Such estimated uses could be verified when needed for individual applications with an assessment tool such as the Wetlands Evaluation Technique, or Habitat Evaluation Procedure, or region-specific evaluation methods.**

- **States should tap into the potential of the outstanding resource waters designation of the antidegradation policy for their wetlands.**

- **States should incorporate 401 certification for wetlands into their water quality management planning process. This process can integrate wetland resource information with different water management programs affecting wetlands (including coastal zone management, nonpoint source and wastewater programs).**

II. WHAT IS WATER QUALITY CERTIFICATION AND HOW DOES IT WORK?

States may grant or deny "certification" for a federally permitted or licensed activity that may result in a discharge to the waters of the United States, if it is the State where the discharge will originate. The decision to grant or deny certification is based on a State's determination from data submitted by an applicant (and any other information available to the State) whether the proposed activity will comply with the requirements of certain sections of the Clean Water Act enumerated in Section 401(a)(1). These requirements address effluent limitations for conventional and nonconventional pollutants, water quality standards, new source performance standards, and toxic pollutants (Sections 301, 302, 303, 306 and 307). Also included are requirements of State law or regulation more stringent than those sections or their federal implementing regulations.

States adopt surface water quality standards pursuant to Section 303 of the Clean Water Act and have broad authority to base those standards on the waters' use and value for "public water supplies, propagation of fish and wildlife, recreational purposes, and . . . other purposes."[4] All permits must include effluent limitations at least as stringent as needed to maintain established beneficial uses and to attain the quality of water designated by States for their waters.[5] Thus, the States' water quality standards are a critical concern of the 401 certification process.

If a State grants water quality certification to an applicant for a federal license or permit, it is in effect saying that the proposed activity <u>will</u> comply with State water quality standards (and the other CWA and State law provisions enumerated above). The State may thus deny certification because the applicant has not demonstrated that the project will comply with those requirements. Or it may place whatever limitations or conditions on the certification it determines are necessary to assure compliance with those provisions, and with any other "appropriate" requirements of State law.

If a State denies certification, the federal permitting or licensing agency is prohibited from issuing a permit or license. While the procedure varies from State to State, a State's decision to grant or deny certification is ordinarily subject to an administrative appeal, with review in the State courts designated for appeals of agency decisions. Court review is typically limited to the question of whether the State agency's decision is supported by the record and is not arbitrary or capricious. The courts generally presume regularity in agency procedures and defer to agency expertise in their review.[6]

States may also waive water quality certification, either affirmatively or involuntarily. Under Section 401(a)(1), if the State fails to act on a certification request "within a reasonable time (which shall not exceed one year)" after the receipt of an application, it forfeits its authority to grant conditionally or to deny certification.

The most important regulatory tools for the implementation of 401 certification are the States' water quality standards regulations and their 401 certification implementing regulations and guidelines. While all of the States have some form of water quality standards, not all States have standards which can be easily applied to wetlands. Most Tribes do not yet have water quality standards, and developing them would be a first step prior to having the authority to conduct water quality certification. Also, many States have not adopted regulations implementing their authority to grant, deny and condition water quality certification. The remainder of this handbook discusses specific approaches, and elements of water quality standards and 401 certification regulations that OWP views as effective to implement the States' water quality certification authority, both generally, and specifically with regard to wetlands.

III. 401 CERTIFICATION CAN BE A POWERFUL TOOL TO PROTECT WETLANDS

In States without a wetlands regulatory program, the water quality certification process may be the only way in which a State can exert any direct control over projects in or affecting wetlands. It is thus critical for these States to develop a program that fully includes wetlands in their water quality certification process.

But even in States which have their own wetlands regulatory programs, the water quality certification process can be an extremely valuable tool to protect wetlands. First, most State wetland regulatory laws are more limited in the wetlands that are subject to regulation than is the Clean Water Act. The Clean Water Act covers all interstate wetlands; wetlands adjacent to other regulated waters; and all other wetlands, the use, degradation or destruction of which <u>could</u> affect interstate or foreign commerce.[7] This definition is extremely broad and one would be hard pressed to find a wetland for which it could be shown that its use or destruction clearly would <u>not</u> affect interstate commerce. Federal jurisdiction extends beyond that of States which regulate only coastal and/or shoreline wetlands, for instance. And in States that regulate inland wetlands, often size limitations prevent States from regulating wetlands that are subject to federal jurisdiction.[8]

Even if State jurisdiction is as encompassing or more so than federal jurisdiction, however, water quality certification may still be a valuable and essential wetlands protection device. In the State of Massachusetts, for instance, a 401 certification is <u>not</u> simply "rubber stamped" on the permitting decisions made pursuant to the Massachusetts Wetlands Protection Act. The State has denied certification to proposed projects requiring a federal permit even though the State wetlands permitting authority (in Massachusetts, permits are granted by local "conservation commissions") has granted authorization for a project.

There may be a number of reasons that a proposed activity may receive authorization under a State wetland regulatory program, but fail to pass muster under a 401 certification review. The most commonly cited reason, however, is that water quality personnel have a specialized understanding of the requirements and implementation of the State's water quality standards and the ways in which certain activities may interfere with their attainment.

It is important, however, to keep in mind the limitations of 401 certification when considering a comprehensive approach to protecting your wetland resources. The primary limitation is that if 401 certification is the only tool a State has to protect wetlands, it cannot place limits on activities which do not require a federal license or permit. Some activities such as drainage or groundwater pumping, can have severe impacts on the viability of wetlands, but may not require a permit or license. Ideally, 401 certification should be combined with other programs in the State offering wetlands protection opportunities (such as coastal management and floodplain management). For example, Alaska has integrated its 401 certification and coastal management consistency review processes so that the provisions of each program augment the other to provide more comprehensive protection. This approach not only strengthens protection, it reduces duplication of State efforts and coordinates permit review for applicants.[9]

IV. THE ROLE OF WATER QUALITY STANDARDS IN THE CERTIFICATION PROCESS

A. Wetlands Should be Specifically Designated as Surface Waters of the States

In order to bring wetlands fully into the State water quality certification process, a first step is to include the term "wetlands" in the State water quality standards' definition of surface waters. EPA will be working with all States through the triennial review process of State standards to ensure that their definitions are at least as comprehensive as the federal definitions for waters (see Appendix B for federal definitions of "Waters of the U.S." and the term "wetlands").

It may seem minor, but from every standpoint, it is important to have wetlands specifically designated as surface waters in State water quality standards. First, it precludes any arguments that somehow wetlands are not covered by water quality standards. Second, it predisposes decision makers (from 401 certification program managers, to the head of the agency or a water quality board, all the way to the judges on the courts that may review these decisions) to consider the importance of wetlands as part of the aquatic ecosystem. Third, it makes it clear that wetlands are to be treated as waters in and of themselves for purposes of compliance with water quality standards and not just as they relate to other surface waters.

The third point is critical and bears further explanation. When States include wetlands in the definition of surface waters covered by their water quality standards, they clarify that activities in or affecting wetlands are subject to the same analysis in the certification decision as are projects affecting lakes, rivers, or streams. This is not to say that a wetland project's effects on adjacent or downstream waters are not also part of the water quality certification analysis. Rather, it is to say that wetlands, either adjacent to or isolated from other waters, are waterbodies in and of themselves and an applicant for water quality certification must show that a proposed project will not violate water quality standards in those wetlands, as well as in other waters.

The States currently have a variety of definitions of "waters of the State" in the legislation that enables water quality standards (e.g., multi-media environmental protection acts, water quality acts, and the like). Only three States currently have the term "wetlands" explicitly listed as one of the types of waters in this enabling legislation (Nebraska, Rhode Island, West Virginia). These States need only to repeat that definition in their water quality standards and their 401 certification implementing regulations.

While most States do not have the term "wetlands" in their enabling legislation, many use the term "marshes" in a list of different types of waters to illustrate "waters of the State" in their enabling legislation. Kentucky, for example, defines waters of the State as:

> *. . . any and all rivers, streams, creeks, lakes, ponds, impounding reservoirs, springs, wells, marshes, and all other bodies of surface or underground water, natural or artificial, situated wholly or partly within or bordering upon the Commonwealth or within its jurisdiction.*[10]

When used in this way, the term "marshes" is typically understood to be generic in nature rather than being descriptive of a type of wetland, and can therefore be considered as the equivalent of the term "wetlands". In these States, however, in order to ensure that the term "marshes" is interpreted as the equivalent of wetlands, the best approach is to include the term "wetlands" in the definition of surface waters used in the State's water quality standards and in the 401 certification implementing regulations.

There is another group of States that has neither the term "wetlands" or "marshes" in the enabling legislation's definition of waters of the State. These definitions typically contain language that describes in some generic manner, however, all waters that exist in the State. They may not specifically designate any particular type of water body, as, for instance, Tennessee's Water Quality Control Act:

> *. . . any and all water, public or private, on or beneath the surface of the ground, which [is] contained within, flow[s] through, or border[s] upon Tennessee or any portion thereof . . . .*[11]

Or they may specify some types of surface waters and then generically include all others with a clause such as "and all other water bodies" or "without limitation", as does Massachusetts:

> *All waters within the jurisdiction of the Commonwealth, including, without limitation, rivers, streams, lakes, ponds, springs, impoundments, estuaries, and coastal waters and groundwaters.*[12]

In these States, as in the States with "marshes" in the enabling legislation's definition of waters, regulators should clarify that wetlands are part of the surface waters of the State subject to the States' water quality standards by including that term, and any others they deem appropriate, in a definition of surface waters in their water quality standards and in their 401 certification implementing regulations.

Both Kentucky and Ohio, for instance, which have the term "marshes," but not the term "wetlands" in their enabling legislation, have included the term "wetlands" in their surface water quality standards' definition of waters.[13] Massachusetts, which does not have the term "wetlands" or "marshes" in its enabling legislation, has put the term "wetlands" into its water quality standards also.[14] Additionally, Ohio's 401 certification implementing regulations include the term "wetlands" in the definition of waters covered by those regulations and specifically address activities affecting the integrity of wetlands.[15]

B. General Requirements of EPA's Water Quality Standards Regulations.[16]

When the States review their water quality standards for applicability to projects affecting wetlands, it is important to have in mind the basic concepts and requirements of water quality standards generally. Congress has given the States broad authority to adopt water quality standards, directing only that the States designate water uses that protect the public health and welfare and that take into account use of State waters for drinking water, the propagation of fish and wildlife, recreation, and agricultural, industrial and other purposes.

EPA's water quality standards regulations require States to adopt water quality standards which have three basic components: use designations, criteria to protect those uses, and an antidegradation policy.

EPA directs that, where attainable, designated uses must include, at a minimum, uses necessary to protect the goals of the CWA for the protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the waters. This baseline is commonly referred to as the "fishable/swimmable" designation. If the State does not designate these minimum uses, or wishes to remove such a designated use, it must justify it through a use attainability analysis based on at least one of six factors.[17] In no event, however, may a beneficial existing use (any use which is actually attained in the water body on or after November 28, 1975) be removed from a water body or segment.

Criteria, either pollutant-specific numerical criteria or narrative criteria, must protect the designated and existing uses. Many of the existing numeric criteria are not specifically adapted to the characteristics of wetlands (see last section of handbook for steps in this direction). However, almost all States have some form of the narrative standards (commonly known as the "free froms") which say that all waters shall be free from substances that: settle to form objectionable deposits; float as debris, scum, oil or other matter to form nuisances; produce objectionable color, odor, taste, or turbidity; injure, or are toxic,or produce adverse physiological responses in humans, animals, or plants; or produce undesirable or nuisance aquatic life. States have also used other narrative criteria to protect wetland quality. The use of criteria to protect wetlands is discussed in the following section.

In addition, EPA also requires that all States adopt an antidegradation policy. Several States have used their antidegradation policy effectively to protect the quality of their wetland resources. At a minimum, a State's antidegradation policy must be consistent with the following provisions:

(1) **Existing uses and the level of water quality necessary to protect existing uses in all segments of a water body must be maintained;**

(2) **if the quality of the water is higher than that necessary to support propagation of fish, shellfish, and wildlife, and recreation in and on the water, that quality shall be maintained and protected, unless the State finds that lowering the water quality is justified by overriding economic or social needs determined after full public involvement. In no event, however, may water quality fall below that necessary to protect the existing beneficial uses;**

(3) **if the waters have been designated as outstanding resource waters (ORWs) no degradation (except temporary) of water quality is allowed.**

In the case of wetland fills, however, EPA allows a slightly different interpretation of the antidegradation policy.[18] Because on the federal level, the Congress has anticipated the issuance of at least some permits by virtue of Section 404, it is EPA's policy that, except in the case of ORWs, the "existing use" requirements of the antidegradation policy are met if the wetland fill does not cause or contribute to "significant degradation" of the aquatic environment as defined by Section 230.10(c) of the Section 404(b)(1) Guidelines.[19]

These Guidelines lay a substantial foundation for protecting wetlands and other special aquatic sites from degradation or destruction. The purpose section of the Guidelines states that:

". . . from a national perspective, the degradation or destruction of special aquatic sites, such as filling operations in wetlands, is considered to be among the most severe environmental impacts covered by these Guidelines. The guiding principal should be that degradation or destruction of special sites may represent an irreversible loss of valuable aquatic resources."[20]

The Guidelines also state that the following effects contribute to significant degradation, either individually or collectively:

". . . significant adverse effects on (1) human health or welfare, including effects on municipal water supplies, plankton, fish, shellfish, wildlife, and special aquatic sites (e.g., wetlands); (2) on the life stages of aquatic life and other wildlife dependent on aquatic ecosystems, including the transfer, concentration or spread of pollutants or their byproducts beyond the site through biological, physical, or chemical process; (3) on ecosystem diversity, productivity and stability, including loss of fish and wildlife habitat or loss of the capacity of a wetland to assimilate nutrients, purify water or reduce wave energy; or (4) on recreational, aesthetic, and economic values."[21]

The Guidelines may be used by the States to determine "significant degradation" for wetland fills. Of course, the States are free to adopt stricter requirements for wetland fills in their own antidegradation policies, just as they may adopt more stringent requirements than federal law requires for their water quality standards in general.

C. Applying Water Quality Standards Regulations to Wetlands - What States are Doing Now

Some States have taken the lead in using 401 certification as a wetlands protection tool to protect them for their water quality and other irreplaceable functions, such as storage places for flood waters, erosion control, foodchain support and habitat for a wide variety of plants and animals. These States have taken several different approaches to wetlands protection in their water quality certification process.

1. Using Narrative Criteria

States have applied a variety of narrative criteria to projects in or affecting wetlands in the 401 certification determination. For example, Maryland's water quality standards contain a narrative directive, which the agency relied upon to deny certification for a non-tidal wetland fill. The standard provides that "[a]ll waters of this State shall be protected for the basic uses of water contact recreation, fish, other aquatic life, wildlife, and water supply."[22] In its denial, Maryland stated:

> *Storm waters are relieved of much of their sediment loads via overbanking into the adjacent wetland and a resultant decrease in nutrient and sediment loading to downstream receiving waters is occurring. To permit the fill of this area would eliminate these benefits and in the future, would leave the waterway susceptible to adverse increased volumes of storm waters and their associated pollutants. It is our determination that [a specified waterway] . . . requires protection of these wetland areas to assure that the waters of this State are protected for the basic uses of fish, other aquatic life, wildlife and water supply.*

Because wetlands vary tremendously in background levels of certain parameters measured by the traditional numerical/chemical criteria applied to surface waters, some States have relied on "natural water quality" criteria to protect wetlands in the 401 certification process. Minnesota, for instance, has taken this approach in denying certification for a flood control project because of the State's "primary concern . . . that the project would likely change Little Diann Lake from an acid bog to a fresh-circumneutral water chemistry type of wetland." The agency was concerned that "introduction of lake water into the closed acid system of Little Diann Lake would completely destroy the character of this natural resource." It relied on a provision of its water quality standards allowing the State to limit the addition of pollutants according to background levels instead of to the levels specified by criteria for that class of waters generally. The denial letter pointed out that this rule "States that the natural background level may be used instead of the specified water quality standards, where reasonable justification exists for preserving the quality found in the State of nature." According to the denial letter, because of the clear potential for impacts to the bog, the State was invoking that particular provision.[23]

Tennessee has relied on broad prohibitory language in its water quality standards to deny water quality certification for wetland fill projects and has been upheld in court. Hollis v. Tennessee Water Quality Control Board[24] was brought by a 401 certification applicant who proposed to place fill along the southeastern shoreline of a natural swamp lake. The court upheld the denial of 401 certification, explaining:

> *Reelfoot Lake is classified for fish and aquatic life, recreation, and livestock watering and wildlife uses. The [Water Quality] Board has established various standards for the waters in each classification. Among other things, these standards pertain to dissolved oxygen, pH, temperature, toxic substances, and other pollutants. The Permit Hearing Panel found the petitioner's activity will violate the "other pollutants" standard in each classification. Collectively, these ["other pollutants"] standards provide that other pollutants shall not be added to the water that will be detrimental to fish or aquatic life, to recreation, and to livestock watering and wildlife.*

The court found that while there was no evidence that the project in and of itself would "kill" Reelfoot Lake, there was evidence that the shoreline was important to recreation because tourists visit Reelfoot to view its natural beauty and the lacustrine wetlands function as a spawning ground for fish and produce food for both fish and wildlife. It found that although the evidence in the record did not quantify the damage to fish and aquatic life, recreation, and wildlife that would result from the proposed fill, the opinion of the State's expert that the activity would be detrimental to these uses was sufficient to uphold the denial of certification.

Kentucky has also relied on narrative criteria. It denied an application to place spoil from underground mine construction in a wetland area because wetlands are protected from pollution as "Waters of the Commonwealth" and because placing spoil or any fill material (pollutants under KRS 224.005(28)) in a wetland specifically violated at least two water quality criteria. One of Kentucky's criteria, applicable to all surface waters, provides that the waters "*shall not be aesthetically or otherwise degraded by substances that . . . [i]njure, [are] toxic to or produce adverse physiological or behavioral responses in humans, animals, fish and other aquatic life.*"

The other criterion, applicable to warm water aquatic habitat, provides that "*[f]low shall not be altered to a degree which will adversely affect the aquatic community.*"[25] This second criterion which addresses hydrological changes is a particularly important but often overlooked component to include in water quality standards to help maintain wetland quality. Changes in flow can severely alter the plant and animal species composition of a wetland, and destroy the entire wetland system if the change is great enough.

Ohio has adopted 401 certification regulations applicable to wetlands (and other waters) that, together with internal review guidelines, result in an approach to the 401 certification decision similar to that of the 404(b)(1) Guidelines. Its 401 certification regulations first direct that no certification may be issued unless the applicant has demonstrated that activities permitted by Section 404 or by Section 10 of the Rivers and Harbors Act (RHA) will not:

(1) prevent or interfere with the attainment or maintenance of applicable water quality standards;

(2) result in a violation of Sections 301, 302, 303, 306 or 307 of the CWA; additionally, the agency may deny a request notwithstanding the applicant's demonstration of the above if it concludes that the activity "will result in adverse long or short term impacts on water quality."[26]

Ohio has placed all of its wetlands as a class in the category of "State resource waters." For these waters, Ohio has proposed amendments to its standards to say that "[p]resent ambient water quality and uses shall be maintained and protected without exception." [27] The proposed standards also require that point source discharges to State resource waters be regulated according to Ohio's biological criteria for aquatic life.

However, Ohio has not yet developed biological indices specifically for wetlands. Thus, for projects affecting wetlands, it bases its certification decisions on internal review guidelines that are similar to the federal Section 404(b)(1) Guidelines. Ohio's guidelines are structured by type of activity. For instance, for fills, their requirements are as follows:

(a) if the project is not water dependent, certification is denied;

(b) if the project is water dependent, certification is denied if there is a viable alternative (e.g., available upland nearby is viable alternative);

(c) if no viable alternatives exist and impacts to wetland cannot be made acceptable through conditions on certification (e.g., fish movement criteria, creation of floodways to bypass oxbows, flow through criteria), certification is denied.

Ohio's internal review guidelines also call for (1) an historical overview and ecological evaluation of the site (including biota inventory and existing bioaccumulation studies); (2) a sediment physical characterization (to predict contaminant levels) and (3) a sediment analysis.[28]

Using these guidelines, Ohio frequently conditions or denies certification for projects that eliminate wetland uses. For instance, Ohio has issued a proposed denial of an application to fill a three acre wetland area adjacent to Lake Erie for a recreational and picnic area for a lakefront marina based on its classification of wetlands as "State resource waters:"

> *Wetlands serve a vital ecological function including food chain production, provision of spawning, nursery and resting habitats for various aquatic species, natural filtration of surface water runoff, ground water recharge, and erosion and flood abatement. The O.A.C. Section 3745-1-05(C) includes wetlands [in the] State Resource Waters category and allows no further water quality degradation which would interfere with or become injurious to the existing uses. The addition of fill material to the wetland would cause severe adverse effects to the wetland. This fill would eliminate valuable wetland habitat, thereby degrading the existing use.*

The justification for this denial, according to Ohio program managers, was not only that the project would interfere with existing uses, but in addition, the project was not water dependent as called for in Ohio's internal guidelines. Ohio 401 certification program personnel note that these review guidelines present the general approach to certification, but with regard to projects that are determined to be of public necessity, this approach may give way to other public interest concerns. For example, a highway is not water dependent per se; if, however, safety and financial considerations point to a certain route that necessitates filling wetlands, the agency may allow it. In that event, however, mitigation by wetland creation and/or restoration would be sought by the agency as a condition of certification.

2. Highest Tier of Protection: Wetlands as Outstanding Resource Waters

One extremely promising approach taken by some of the States has been to designate wetlands as outstanding resource waters (ORW), in which water quality must be maintained and protected according to EPA's regulations on antidegradation (i.e., no degradation for any purposes is allowed, except for short term changes which have no long term consequences).[29] This approach provides wetlands with significant protection if the States' antidegradation policies are at least as protective as that of EPA. EPA designed this classification not only for the highest quality waters, but also for water bodies which are "important, unique, or sensitive ecologically, but whose water quality as measured by the traditional parameters (dissolved oxygen, pH, etc.) may not be particularly high or whose character cannot be adequately described by these parameters."[30] This description is particularly apt for many wetland systems.

The designation of wetlands as outstanding resource waters has occurred in different ways in different States. Minnesota, for instance, has designated some of its rare, calcareous fens as ORWs and intends to deny fills in these fens.

Ohio has issued for comment, proposed revised water quality standards that include a newly created "outstanding State resource waters" category. Ohio intends to prohibit all point source discharges to these waters. Of fourteen specific water bodies proposed to be included in this category by the Ohio EPA at this time, ten are wetlands: four fens; three bogs; and three marshes.

Because the designation of wetlands as ORWs is such an appropriate classification for many wetland systems, it would behoove the States to adopt regulations which maximize the ability of State agencies and citizens to have wetlands and other waters placed in this category. The State of Kentucky has set out procedures for the designation of these waters in its water quality standards. Certain categories of waters automatically included as ORWs are: waters designated under the Kentucky Wild Rivers Act or the Federal Wild and Scenic Rivers Act; waters within a formally dedicated nature preserve or published in the registry of natural areas and concurred upon by the cabinet; and waters that support federally recognized endangered or threatened species. In addition, Kentucky's water quality standards include a provision allowing anyone to propose waters for the ORW classification.[31]

Minnesota has a section in its water quality standards that could be called an "emergency" provision for the designation of outstanding resource waters. Normally it is necessary under Minnesota's water quality standards for the agency to provide an opportunity for a hearing before identifying and establishing outstanding resource waters and before prohibiting or restricting any discharges to those waters. The "emergency" provision allows the agency to prohibit new or expanded discharges for unlisted waters "*to the extent . . . necessary to preserve the existing high quality, or to preserve the wilderness, scientific, recreational, or other special characteristics that make the water an outstanding resource value water.*"[32] This provision allows the agency to protect the waterbody while completing the listing process which could take several years.

Moreover, some States have improved on the formulation of the ORW classification by spelling out the protection provided by that designation more specifically than do EPA's regulations. For instance, Massachusetts' water quality standards state that for "National Resource Waters:"

> *Waters so designated may not be degraded and are not subject to a variance procedure. New discharges of pollutants to such waters are prohibited. Existing discharges shall be eliminated unless the discharger is able to demonstrate that: (a) Alternative means of disposal are not reasonably available or feasible; and (b) The discharge will not affect the quality of the water as a national resource.*[33]

This provision explicitly outlines how the State intends to maintain and protect the water quality of ORWs. Another provision which Minnesota uses to control discharges to waters that flow into ORWs for their effect on ORWs is that:

> *The agency shall require new or expanded discharges that flow into outstanding resource value waters [to] be controlled so as to assure no deterioration in the quality of the downstream outstanding resource value water.*[34]

V. USING 401 CERTIFICATION

A. The Permits/Licenses Covered and the Scope of Review

The language of Section 401(a)(1) is written very broadly with respect to the activities it covers. "[A]ny activity, including, but not limited to, the construction or operation of facilities, which may result in any discharge" requires water quality certification.

When the Congress first enacted the water quality certification provision in 1970, it spoke of the "wide variety of licenses and permits . . . issued by various Federal agencies," which "involve activities or operations potentially affecting water quality."[35] The purpose of the water quality certification requirement, the Congress said, was to ensure that no license or permit would be issued "for an activity that through inadequate planning or otherwise could in fact become a source of pollution."[36]

1. Federal Permits/Licenses Subject to Certification

The first consideration is which federal permits or licenses are subject to 401 certification. OWP has identified five federal permits and/or licenses which authorize activities which may result in a discharge to the waters. **These are: permits for point source discharges under Section 402 and discharges of dredged and fill material under Section 404 of the Clean Water Act; permits for activities in navigable waters which may affect navigation under Sections 9 and 10 of the Rivers and Harbors Act (RHA); and licenses required for hydroelectric projects issued under the Federal Power Act.**

There are likely other federal permits and licenses, such as permits for activities on public lands, and Nuclear Regulatory Commission licenses, which may result in a discharge and thus require 401 certification. Each State should work with EPA and the federal agencies active in its State to determine whether 401 certification is in fact applicable.

Indeed, it is not always clear when 401 certification should apply. For instance, there remains some confusion under Sections 9 and 10 of RHA concerning which projects may involve or result in a discharge, and thus require State certification. In many cases there is an overlap between Section 404 CWA and Sections 9 and 10 RHA. Where these permits overlap, 401 certification always applies. Under the Section 404 regulations, the question of whether dredging involves a discharge and is therefore subject to Section 404, depends on whether there is more than "de minimis, incidental soil movement occurring during normal dredging operations".[37]

Where only a Section 9 or 10 permit is required, 401 certification would apply if the activity may <u>lead</u> to a discharge. For example, in the case of pilings, which the Corps sometimes considers subject to Section 10 only, a 401 certification would be required for the Section 10 permit if structures on top of the pilings may result in a discharge.

States should notify the regional office of federal permitting or licensing agencies of their authority to review these permits and licenses (e.g., the Corps of Engineers for Section 404 in nonauthorized States, and Sections 9 and 10 of the RHA; EPA for Section 402 permits in nonauthorized States; and the Federal Energy Regulatory Commission (FERC) for hydropower licenses). In their 401 certification implementing regulations, States should also give notice to applicants for these particular federal permits and licenses, and for all other permits and licenses that may result in a discharge to waters of the State, of their obligation to obtain 401 certification from the State.

West Virginia's 401 certification implementing regulations, for instance, state that:

> *1.1. Scope. . . . Section 401 of the Clean Water Act requires that any applicant for a federal license or permit to conduct an activity which will or may discharge into waters of the United States (as defined in the Clean Water Act) must present the federal authority with a certification from the appropriate State agency. Federal permits and licenses issued by the federal government requiring certification <u>include</u> permits issued by the United States Army Corps of Engineers under Section 404 of the Clean Water Act, 33 U.S.C. 1344 and licenses issued by the Federal Energy Regulatory Commission under the Federal Power Act, 16 U.S.C. 1791 et seq.*[38]

Because West Virginia has been authorized to administer the NPDES permitting program under Section 402 of the Clean Water Act, applicants for NPDES permits do not have to apply for water quality certification separately. In addition, West Virginia has not specifically designated Rivers and Harbors Act permits in the above regulation. However, because the regulation States that such permits or licenses <u>include</u> Section

404 and FERC licenses, those and all other permits not specifically designated but which may result in a discharge to the waters would be covered by the regulation's language. The better approach would be to enumerate all such licenses and permits that are known to the State and include a phrase for all others generically.

2. Scope of Review Under Section 401

An additional issue is the scope of the States' review under Section 401. Congress intended for the States to use the water quality certification process to ensure that no federal license or permits would be issued that would violate State standards or become a source of pollution in the future. Also, because the States' certification of a construction permit or license also operates as certification for an operating permit (except for in certain instances specified in Section 401(a)(3)), it is imperative for a State review to consider all potential water quality impacts of the project, both direct and indirect, over the life of the project.

A second component of the scope of the review is when an activity requiring 401 certification in one State (i.e. the State in which the discharge originates) will have an impact on the water quality of another State.[39] The statute provides that after receiving notice of application from a federal permitting or licensing agency, EPA will notify any States whose water quality may be affected. Such States have the right to submit their objections and request a hearing. EPA may also submit its evaluation and recommendations. If the use of conditions cannot insure compliance with the affected State's water quality requirements, the federal permitting or licensing agency shall not issue such permit or license.

The following example of 401 certification denial by the Pennsylvania Department of Environmental Resources (DER) for a proposed FERC hydroelectric project illustrates the breadth of the scope of review under Section 401 (see Appendix C for full description of project and impacts addressed). The City of Harrisburg, Pennsylvania proposed to construct a hydroelectric power project on the Susquehanna River. The Pennsylvania DER considered a full range of potential impacts on the aquatic system in its review. The impacts included those on State waters located at the dam site, as well as those downstream and upstream from the site. The impacts considered were not just from the discharge initiating the certification review, but water quality impacts from the entire project. Thus, potential impacts such as flooding, changes in dissolved oxygen, loss of wetlands, and changes in groundwater, both from construction and future operation of the project, were all considered in the State's decision.

The concerns expressed by the Pennsylvania Department of Environmental Resources are not necessarily all those that a State should consider in a dam certification review; each project will have its own specific impacts and potential water quality problems. The point of the illustration is to show that **all of the potential effects of a proposed activity on water quality -- direct and indirect, short and long term, upstream and downstream, construction and operation -- should be part of a State's certification review.**

B. Conditioning 401 Certifications for Wetland Protection

In 401(d), the Congress has given the States the authority to place any conditions on a water quality certification that are necessary to assure that the applicant will comply with effluent limitations, water quality standards, standards of performance or pretreatment standards; with any State law provisions or regulations more stringent than those sections; and with "any other appropriate requirement of State law."

The legislative history of the subsection indicates that the Congress meant for the States to impose whatever conditions on the certification are necessary to ensure that an applicant complies with all State requirements that are related to water quality concerns.

1. What are Appropriate Conditions?

There are any number of possible conditions that could be placed on a certification that have as their purpose preventing water quality deterioration.

By way of example, the State of Maryland issued a certification with conditions for placement of fill to construct a 35-foot earthen dam located 200 feet downstream of an existing dam. Maryland used some general conditions applicable to many of the proposed projects it considers, along with specific conditions tailored to the proposed project. Examples of the conditions placed on this particular certification include:

> *The applicant shall obtain and certify compliance with a grading and sediment control plan which has been approved by the [county] Soil Conservation District. The approved plan shall be available at the project site during all phases of construction.*
>
> *Stormwater runoff from impervious surfaces shall be controlled to prevent the washing of debris into the waterway. The natural vegetation shall be maintained and restored when disturbed or eroded. Stormwater drainage facilities shall be designed, implemented, operated, and maintained in accordance with the requirements of the applicable approving authority.*

The applicant is required to provide a mixing tower release structure to achieve in-stream compliance with Class III trout temperature (20[degrees] C) and dissolved oxygen (5.0 mg/liter) standards prior to the Piney Run/Church Creek confluence. The design of this structure shall be approved by the Maryland Department of the Environment (MDE).

The applicant is required to provide a watershed management plan to minimize pollutant loadings into the reservoir. This plan shall be reviewed and approved by MDE prior to operation of the new dam facility. In conjunction with this plan's development any sources of pollutant loading identified during field surveys shall be eliminated or minimized to the extent possible given available technology.

The applicant is required to provide to MDE an operating and maintenance plan for the dam assuring minimum downstream flows in accordance with the requirements of the DNR and assuring removal of accumulated sediments with subsequent approved disposal of the materials removed.

The applicant is to provide mitigation for the wetlands lost as a result of the construction of this project and its subsequent operation. Wetland recreation should be located in the newly created headwaters areas to: a) assure adequate filtration of runoff prior to its entry into the reservoir and b) replace the aquatic resource being lost on an acre for acre basis.

See Appendix D for the full list of conditions placed on this certification. While few of these conditions are based directly on traditional water quality standards, all are valid and relate to the maintenance of water quality or the designated use of the waters in some way. Some of the conditions are clearly requirements of State or local law related to water quality other than those promulgated pursuant to the CWA sections enumerated in Section 401(a)(1). Other conditions were designed to minimize the project's adverse effects on water quality over the life of the project.

In addition, Appendix D contains a list of conditions which West Virginia and Alaska placed on the certification of some Section 404 nationwide permits. Many of the West Virginia conditions are typical of ones it uses on individual proposals as well. For any particular project, West Virginia will include more specific conditions designed to address the potential adverse effects of the project in addition to those enumerated in Appendix D. The conditions from Alaska are used on a nationwide permit (#26) regarding isolated waters and waters above headwaters. These conditions are discussed in Section V. C(1).

2. The Role of Mitigation in Conditioning Certification

Many States are trying to determine the role that mitigation should play in 401 certification decisions. We cannot answer this question definitively for each State, but offer as a guide EPA's general framework for mitigation under the Section 404(b)(1) Guidelines used to evaluate applications for Section 404 permits. In assuring compliance of a project with the Guidelines, **EPA's approach is to first, consider avoidance of adverse impacts, next, determine ways to minimize the impacts, and finally, require appropriate and practicable compensation for unavoidable impacts.**

The Guidelines provide for avoiding adverse impacts by selecting the least environmentally damaging practicable alternative. In addition, wetlands are "special aquatic sites." For such sites, if the proposed activity is not "water dependent," practicable alternatives with less adverse environmental impacts are presumed to be available unless the applicant clearly demonstrates otherwise.[40]

The Guidelines also require an applicant to take "appropriate and practicable" steps to minimize the impacts of the least environmentally damaging alternative selected.[41] Examples in the Guidelines for minimizing impacts through project modifications and best management practices are provided in Appendix E.

After these two steps are complete, appropriate compensation is required for the remaining unavoidable adverse impacts. Compensation would consist of restoration of previously altered wetlands or creation of wetlands from upland sites. In most cases, compensation on or adjacent to the project site is preferred over off-site locations. The restoration or creation should be functionally equivalent to the values which are lost. Finally, compensating with the same type of wetland lost is preferred to using another wetland type.

The States may choose to adopt mitigation policies which require additional replacement to help account for the uncertainty in the science of wetland creation and restoration. What is important from EPA's perspective is that mitigation not be used as a trade-off for <u>avoidable</u> losses of wetlands, and that mitigation compensate, to the fullest extent possible, for the functional values provided to the local ecosystem by the wetlands unavoidably lost by the project.

3. The Role of Other State Laws

Another question that has been asked is what State law or other requirements are appropriately used to condition a 401 certification. The legislative history of Section 401(d) indicates that Congress meant for the States to condition certifications on compliance with any State and local law requirements related to water quality preservation. The courts that have touched on the issue have also indicated that conditions that relate in any way to water quality maintenance are appropriate. Each State will have to make these determinations for itself, of course; there are any number of State and local programs that have components related to water quality preservation and enhancement.

One issue that has arisen in two court cases is whether a State may use State law requirements, other than those that are more stringent than the provisions of Sections 301, 302, 303, 306 and 307 of the CWA(401(a)(1)), to deny water quality certification. An Oregon State court has ruled that a State may, and indeed must, include conditions on certifications reflecting State law requirements "to the extent that they have any relationship to water quality." "Only to the extent that [a State law requirement] has absolutely no relationship to water quality," the court said, "would it not be an 'other appropriate requirement of State law.'"[42] State agencies must act in accord with State law, of course, and thus the decision to grant certification carries with it the obligation to condition certification to ensure compliance with such State requirements.

This State court decision struck down a State agency's denial of certification because it was based on the applicant's failure to certify compliance with a county's comprehensive plan and land use ordinances. The court held that such "other appropriate requirement[s] of State law" could not be the basis for denying certification. However, the court held that the agency should determine which of the provisions of the land use ordinances had any relation to the maintenance and preservation of water quality. Any such provisions, the court said, could and should be the basis for conditions placed on a certification.

Another State court, however, this one in West Virginia, has upheld the State's denial of certification on the basis of State law requirements unrelated to the implementation of the CWA provisions enumerated in Section 401(a)(1).[43] The court simply issued an order upholding the State's denial, however, and did not write an opinion on the subject. The questions raised by these two opinions are thorny. If States may not deny certification based on State law requirements other than those implementing the CWA, yet want to address related requirements of State law, they must walk a thin line between their State requirements and the limitations of their certification authority under federal law.

One way to avoid these difficulties and to ensure that 401 certification may properly be used to deny certification where the State has determined that the activity cannot be conditioned in such a way as to ensure compliance with State water quality related requirements, is to adopt water quality standards that include all State provisions related to water quality preservation. Congress has given the States great latitude to adopt water quality standards that take into consideration the waters' use for such things as "the propagation of fish and wildlife, recreational purposes, and . . . other purposes."[44] Because of the broad authority granted by the Congress to the States to adopt water quality standards pursuant to Section 303 of the CWA, and because compliance with Section 303 is clearly one of the bases on which a State can deny certification, the States can avoid the difficulty of the deny/condition dilemma by adopting water standards that include all the water quality related considerations it wishes to include in the 401 certification review.

For example, the State of Washington has included State water right permit flow requirements in its conditions for certification of a dam project. This is one means of helping to ensure that hydrological changes do not adversely affect the quality of a waterbody. However, a more direct approach is to include a narrative criterion in the State's water quality standards that requires maintenance of base flow necessary to protect the wetland's (or other waterbody's) living resources. The State of Kentucky has such a criterion in its water quality standards (see previous section IV. D(1) on "Using Narrative Criteria"). Placing the provision directly in the State standards might better serve the State if a certification is challenged because the requirement would be an explicit consideration of 401 certification.

C. Special Considerations for Review of Section 404 Permits: Nationwide and After-the-Fact Permits

1. Nationwide Permits.

Pursuant to Section 404(e) of the CWA, the Corps may issue general permits, after providing notice and an opportunity for a hearing, on a State, regional or nationwide basis for any category of activities involving discharges of dredged or fill material, where such activities are similar in nature and will cause only minimal adverse environmental effects both individually and cumulatively. These permits may remain in effect for 5 years, after which they must be reissued with notice and an opportunity for a hearing. If the activities authorized by general permits may result in a discharge, the permits are subject to the State water quality certification requirement when they are first proposed and when proposed for reissuance. States may either grant certification with appropriate conditions or deny certification of these permits.

Under the Corps' regulations, if a State has denied certification of any particular general permit, any person proposing to do work pursuant to such a permit must first obtain State water quality certification. If a State has conditioned the grant of certification upon some requirement of State review prior to the activity's commencing, such condition[s] must be satisfied before work can begin.

Some States have reported that for general permits for which they have denied water quality certification or on which they have imposed some condition of review, they are having difficulties ensuring that parties performing activities pursuant to these permits are applying to the State for water quality certification or otherwise fulfilling the conditions placed on the certification prior to the commencement of work under these permits.

At least one State is grappling with the problem through its 401 certification implementing regulations. The State of West Virginia denied certification for some nationwide permits issued by the Corps and conditioned the granting of certification for others. One of the conditions that West Virginia has imposed on those certifications that it granted (which thus apply to all nationwide permits in the State) is compliance with its 401 certification implementing regulations. The regulations in turn require that any person authorized to conduct an activity under a nationwide permit must, prior to conducting any activity authorized by a Corps general permit, publish a Class I legal advertisement in a qualified newspaper in the county where the activity is proposed to take place. The notice must describe the activity, advise the public of the scope of the conditionally granted certification, the public's right to comment on the proposed activity and its right to request a hearing. The applicant must forward a certificate of publication of this notice to the State agency prior to conducting any such activity.[45]

The regulation further provides that any person whose property, interest in property or "other constitutionally protected interest under [the West Virginia Constitution] [is] directly affected by the Department's certification" may request a hearing within 15 days of the publication of the notice given by the applicant. The agency will then decide whether to "uphold, modify or withdraw certification for the individual activity."

West Virginia program officers have described the reasons for this procedure:

> *Because of a long-standing concern . . . that untracked dredge and fill activities could prove disastrous on both individual and cumulative bases, the regulations require an authorized permittee [under federal law] to forward proof of publication and a copy of the newspaper advertisement. The information on the notice is logged into a computer system and a site specific inspection sheet is generated. Inspectors then may visit the site to determine compliance with permit conditions and to evaluate cumulative impacts.*[46]

Without such notice and a tracking system of activities performed under these permits, such as that adopted by West Virginia, it will be difficult for a State to evaluate whether or not to grant or deny water quality certification for these permits when they come up for reissuance by the Corps or to condition them in such a way as to avoid adverse impacts peculiar to each of these general permits. It is advisable for the States, regardless of whether they have granted or denied certification, to adopt as part of their 401 certification implementing regulations, provisions addressing these concerns for general permits.

Another way in which some States are attempting to minimize the potential environmental impact of nationwide permits is by stringently conditioning their certification. Alaska, for instance, placed conditions on nationwide permit 26 regarding isolated waters and waters above the headwaters. One of the conditions Alaska used excludes isolated or headwater wetlands of known or suspected high value. When there is uncertainty about a particular wetland, the Corps is required to send pre-discharge notification to designated State officials for a determination. (See Appendix D for a full description of conditions on nationwide permit 26).

2. Section 404 After-the-Fact Permits

The Corps of Engineers' regulations implementing Section 404 provide for the acceptance of after-the-fact permit applications for unauthorized discharges except under certain circumstances. Several States have expressed concern with after-the-fact permits, including the belief that once the discharges have taken place, the water quality certification process is moot. Because of that belief, many States report that they waive certification for after-the-fact permits. Such an approach frustrates law enforcement efforts generally and the water quality certification process in particular because it encourages illegal activity.

The evaluation of after-the-fact permit applications should be no different than for normal applications. Because the burden should be on the applicant to show compliance with water quality standards and other CWA requirements, rather than waiving certification, States could deny certification if the applicant cannot show from baseline data prior to its activity that the activity did not violate water quality standards. If data exist to determine compliance with water quality standards, the States' analysis should be no different merely because the work has already been partially performed or completed. Arkansas denied after-the-fact water quality certification of a wetland fill as follows:

> *[a certain slough] is currently classified as a warmwater fishery . . . . Draining and clearing of [its associated] wetlands will significantly alter the existing use by drastically reducing or eliminating the fishery habitat and spawning areas. This physical alteration of the lake will prevent it from being "water which is suitable for the propagation of indigenous warmwater species of fish" which is the definition of a warmwater fishery. Thus, the . . . project [violates] Section 3 (A) of the Arkansas Water Quality Standards, "Existing instream water uses and the level of water quality necessary to protect the*

existing uses shall be maintained and protected." The Department recommends the area be restored to as near original contours as possible.

With after-the-fact permits, just as with any other permit application, if the State denies certification, the Corps is prohibited from granting a permit. If the applicant refuses to restore the area and does not have a permit, the applicant is subject to a potential enforcement action for restoration and substantial penalties for the unpermitted discharge of pollutants by the EPA, the Corps, a citizen under the citizen suit provision of the CWA, or by the State, if the activity violates a prohibition of State law.

If the State determines that it will get a better environmental result by conditioning certification, it may choose to take that approach. The condition might require mitigation for the filled area (where restoration may cause more environmental harm than benefit, for instance) with restoration or creation of a potentially more valuable wetland area.

In any event, a State should not waive certification of an after-the-fact permit application simply because it is after-the-fact.

VI. DEVELOPING 401 CERTIFICATION IMPLEMENTING REGULATIONS: ADDITIONAL CONSIDERATIONS

A comprehensive set of 401 certification implementing regulations would have both procedural and substantive provisions which maximize the State agency's control over the process and which make its decisions defensible in court. The very fact of having 401 certification regulations goes a long way in providing the State agency that implements 401 certification with credibility in the courts. Currently, no State has "ideal" 401 certification implementing regulations, and many do not have them at all. When 401 certification regulations are carefully considered, they can be very effective not only in conserving the quality of the State's waters, but in providing the regulated sectors with some predictability of State actions, and in minimizing the State's financial and human resource requirements as well.

Everything in this handbook relates in some way to the development of sound water quality standards and 401 certification implementing regulations that will enhance wetland protection. This section addresses some very basic procedural considerations of 401 certification implementing regulations which have not been treated elsewhere. These include provisions concerning the contents of an application for certification; the agency's timeframe for review; and the requirements placed on the applicant in the certification process.

A. Review Timeframe and "Complete" Applications

Under Section 401(a)(1) a State will be deemed to have waived certification if it fails to act within "a reasonable period of time (which shall not exceed one year) after receipt of such request." Program managers should keep in mind that the federal permitting or license agency may have regulations of its own which provide a time limit for the State's certification decision. For instance, Corps regulations say that a waiver "will be deemed to occur if the certifying agency fails or refuses to act on a request for certification within sixty days after receipt . . . unless the district engineer determines a shorter or longer period is reasonable . . . ."[47] FERC rules state that a certifying agency "is deemed to have waived the certification requirements if . . . [it] has not denied or granted certification by one year after the date the certifying agency received the request".[48] EPA regulations for Section 402 in non-authorized States set a limit of 60 days unless the Regional Administrator finds that unusual circumstances require a longer time.[49]

States should coordinate closely with the appropriate federal agency on timing issues. For example, Alaska negotiated joint EPA/State procedures for coastal NPDES permit review. The agreement takes into account and coordinates EPA, Coastal Zone Management, and 401 certification time frames.

It is also advisable for the States to adopt rules which reasonably protect against an unintended waiver due, for example, to insufficient information to make a certification decision or because project plans have changed enough to warrant a reevaluation of the impacts on water quality. Thus, after taking the federal agencies' regulations into account, the State's 401 certification regulations should link the timing for review to what is considered receipt of a complete application.

Wisconsin, for instance, requires the applicant to submit a complete application for certification before the official agency review time begins. The State's regulations define the major components of a complete application, including the existing physical environment at the site, the size of the area affected, all environmental impact assessment information provided to the licensing or permitting agency, and the like. The rules State that the agency will review the application for completeness within 30 days of its receipt and notify the applicant of any additional materials reasonably necessary for review. Although the application will be deemed "complete" for purposes of review time if the agency does not request additional materials within 40 days of receipt of the application, the agency reserves the right to request additional information during the review process.[50]

In the case of FERC projects, West Virginia has taken additional precautions with regard to time for review:

> *If the project application is altered or modified during the FERC licensing process prior to FERC's final decision, the applicant shall inform the Department of such changes. The Department may review such alterations or modifications and, if the changes are deemed significant by the Director, the Department may require a new application for certification. The Department will have ninety (90) days to review such changes or until the end of the year review period . . . , whichever is longer, to determine whether to require a new application or to alter its original certification decision. If the department requires a new application because of a significant application modification, then the Department will have six (6) months to issue its certification decision from the date of submission of the application.*[51]

B. Requirements for the Applicant

It is very important, in particular for conserving the agency's resources and ensuring that there is sufficient information to determine that water quality standards and other provisions of the CWA will not be violated by the activity, to clarify that it is the applicant who is responsible for providing or proving particular facts or requirements.

For instance, Section 401(a)(1) requires that a State "establish procedures for public notice in the case of all applications for certification." West Virginia requires applicants for FERC licenses to be responsible for this notice. In the case of Section 404 permits, West Virginia has a joint notice process with the Corps to issue public notices for 404 applications which also notify the public of the State certification process. Thus, there is no need for West Virginia to require the applicant to do so for these permits.[52]

A second consideration is that States should require the <u>applicant</u> to demonstrate the project's compliance with applicable federal and State law and regulation. EPA's 401 certification regulations name the sources of information a State should use as that contained in the application and other information "furnished by the applicant" sufficient to allow the agency to make a statement that water quality standards will not be violated.[53] Of course in addition, the regulations also refer to other information the agency may choose to examine which is not furnished by the applicant.

Ohio, for instance, has written a requirement for the applicant to demonstrate compliance into its 401 certification implementing regulations:

> *(A) The director shall not issue a Section 401 water quality certification unless he determines that the applicant has demonstrated that the discharge of dredged or fill material to waters of the state or the creation of any obstruction or alteration in waters of the state will:*[54] *(1) Not prevent or interfere with the attainment or maintenance of applicable water quality standards; (2) Not result in a violation of any applicable provision of the following sections of the Federal Water Pollution Control Act [301, 302, 303, 306 and 307].*
>
> *(B) Notwithstanding an applicant's demonstration of the criteria in paragraph (A) . . . the director may deny an application for a Section 401 water quality certification if the director concludes that the discharge of dredged or fill material or obstructions or alterations in waters of the state will result in adverse long or short term impact on water quality.*[55]

C. Permit Fees

A very significant concern for all States who plan to initiate or expand their 401 certification program is the availability of funding. Application fee requirements are a potential funding source to supplement State program budgets. The State of California's Regional Water Quality Control Boards require filing fees for 401 certification applications unless a Board determines that certification is not required. The fee structure is spelled out in the California Water Code. The money collected from the fees goes into the State agency's general fund. The Regional Boards may recover some portion of the fees through the budget request process. The State of Ohio also has a fee structure for 401 certification applicants. In Ohio, however, fees go into the State's general fund, rather than back into the State agency. Neither State collects fees sufficient to support the 401 certification program fully. Despite these potential barriers, application fees could provide a much needed funding source which States should explore.

D. Basis for Certification Decisions

The regulations should also set out the grounds on which the decision to grant or deny certification will be based, the scope of the State's review, and the bases for conditioning a certification. If a State has denied water quality certification for a general permit or has conditioned such a permit on some requirement of State review, the State's 401 certification implementing regulations might also outline the obligations of a person proposing to accomplish work under such a permit. The following is a hypothetical example of regulatory language a State might use to define the grounds for the State's decision to grant, condition, or deny certification:

> In order to obtain certification of any proposed activity that may result in a discharge to waters of the United States, an applicant must demonstrate that the entire activity over its lifetime will not violate or interfere with the attainment of any limitations or standards contained in Section 301, 302, 303, 306, and 307, the federal regulations promulgated pursuant thereto, and any provisions of state law or regulation adopted pursuant to, or which are more stringent than, those provisions of the Clean Water Act.
>
> The agency may condition certification on any requirements consistent with ensuring the applicant's compliance with the provisions listed above, or with any other requirements of state law related to the maintenance, preservation, or enhancement of water quality.

This sample regulatory language provides the grounds for the certification decision, sets the scope of review (lifetime effects of the entire activity) and clearly States that the applicant must demonstrate compliance. For purposes of conditioning the certification in the event it is granted, the same standards can be applied, with the addition of any other requirements of State law that are related to water quality.

Regulations are not project specific. They must be generally applicable to all projects subject to 401 certification review, while at the same time providing reasonable notice to an applicant regarding the general standards employed by the agency in the certification process. (A State may choose to adopt license/permit-specific regulations for 401 certification, but such regulations will still have to be applicable to all activities that may occur pursuant to that license or permit).

There are other considerations that should be addressed in 401 certification implementing regulations, some of which have been mentioned in other parts of this handbook. These include provisions which require applicants for federal licenses and permits which may result in a discharge to apply for water quality certification; provisions which define waters of the State to include wetlands and which define other pertinent terms; and provisions addressing general permits.

VII. EXISTING AND EMERGING SOURCES OF DATA TO AID 401 CERTIFICATION AND STANDARDS DECISION MAKERS

According to a number of State program managers, more data on wetland functions, or "uses," would greatly assist the certification process. Wetland ecosystems not only perform a wide variety of functions but do so in varying degrees. Public agencies and private applicants currently employ a number of assessment methods such as the Wetlands Evaluation Technique and the Habitat Evaluation Procedure to determine what functions or uses exist in a particular wetland system.[56] In many States, however, water quality certification reviewers lack the resources to perform even a simple assessment of a wetland's boundaries, values and functions. Information about the location and types of wetland systems, and of the functions they may perform (such as flood storage, habitat, pollution attenuation, nutrient uptake, and sediment fixing) would aid standard writers in developing appropriate uses and criteria for wetlands, and allow 401 certification officials to conduct a more thorough review.

Several States already have extensive knowledge of their wetland resources, and data gathering efforts are also being undertaken by EPA, the U.S. Fish and Wildlife Service and other agencies.[57] Although these efforts to inventory and classify wetlands have not been closely tied to the 401 certification process in the past, these existing data can be valuable sources of information for 401 certification reviewers. It is important to remember, however, that wetland boundaries for regulatory purposes may differ from those identified by National Wetland Inventory maps for general inventory purposes. The EPA, Corps of Engineers, Fish and Wildlife Service, and Soil Conservation Service have adopted a joint manual for identifying and delineating wetlands in the United States. The manual will be available in June, 1989.[58]

There are several programs that offer technical support for 401 certification decisions. For example, approximately forty States have worked with the Nature Conservancy to establish "natural heritage programs," which identify the most critical species, habitats, plant communities, and other natural features within a State's territorial boundaries. Most States now have a State natural heritage office to coordinate this identification program. Inventory efforts such as the natural heritage program could give 401 certification managers some of the information they need to limit or prohibit adverse water quality impacts in important wetland areas. Specifically, the inventory process can identify existing wetland uses in order to maintain them. The information may also be used in identifying wetlands for Outstanding Resource Waters designation.[59]

The Fish and Wildlife Service maintains a Wetlands Values Data Base which may be very useful in identifying wetland functions and in designating wetland uses for water quality standards. The data base is on computer and contains an annotated bibliography of scientific literature on wetland functions and values.[60] Several States have established critical area programs to identify and protect unique and highly sensitive land and water resources. These programs can provide data to the State water quality certification office and thereby strengthen the scientific basis for 401 certification decision making.[61]

Another potential source of information which might identify wetlands appropriate for designation as Outstanding Resource Waters are the wetland plans which each State is required to develop to comply with the 1986 Emergency Wetlands Resources Act. Beginning in fiscal year 1988, Statewide Comprehensive Outdoor Recreation Plans (SCORP) must now contain a Wetlands Priority Conservation Plan approved by the Department of Interior. Although these plans are primarily focused on wetlands for acquisition, they are a potential source of data on wetland locations and functions. The wetlands identified may also be suitable for special protection under the Outstanding Resource Waters provisions of the antidegradation policy.

The Advance Identification program (ADID), conducted by EPA and the permitting authority, may also furnish a considerable amount of useful information. EPA's 404(b)(1) Guidelines contain a procedure for identifying in advance areas that are generally suitable or unsuitable for the deposit of dredged or fill material.[62] In recent years, EPA has made greater use of this authority. ADID is often used in wetland areas that are experiencing significant development or other conversion pressures. Many ADID efforts generate substantial data on the location and functions of wetlands within the study area such as wetland maps, and habitat, water quality, or hydrological studies.

Special Area Management Plans (SAMPs) are another planning process which may yield useful information. SAMPs refer to a process authorized by the 1980 amendments to the Coastal Zone Management Improvement Act, which provides grants to States to develop comprehensive plans for natural resource protection and "reasonable coastal-dependent economic growth."[63] The SAMP process implicitly recognizes the State water quality certification process, directing all relevant local, State, and federal authorities to coordinate permit programs in carrying out the completed SAMP. The Corps of Engineers has supported and initiated several of these processes. In addition, other SAMPs have been completed by several States.

Much of these data can be collected, combined, and used in decision making with the aid of geographic-based computer systems that can store, analyze, and present data related to wetlands in graphic and written forms.[64] A reviewing official can quickly access and overlay a range of different existing information bases such as flora and fauna inventories, soil surveys, remote sensing data, watershed and wetland maps, existing uses and criteria, and project proposal information.

Finally, data is presently emerging on the use of wetlands as treatment areas for wastewater, stormwater, and non-point discharges.[65] Florida, for instance, has adopted a rule on wastewater releases into wetlands.[66] Florida prohibits wastewater discharges into the following kinds of wetlands: those designated as outstanding waters of the State; wetlands within potable water supplies; shellfish propagation or harvesting waters; wetlands in areas of critical State concern; wetlands where herbaceous ground cover constitutes more than thirty percent of the uppermost stratum (unless seventy-five percent is cattail); and others. Wastewater discharges are permitted in certain wetlands dominated by woody vegetation, certain hydrologically altered wetlands, and artificially created wetlands; however, the State applies special effluent limitations to take account of a wetland's ability to assimilate nitrogen and phosphorus. It also applies qualitative[67] and quantitative[68] design criteria.

The rule establishes four "wetland biological quality" standards. First, the flora and fauna of the wetland cannot be changed so as to impair the wetland's ability to function in the propagation and maintenance of fish and wildlife populations or substantially reduce its effectiveness in wastewater treatment. Second, the Shannon-Weaver diversity index of benthic macroinvertebrates cannot be reduced below fifty percent of background levels. Third, fish populations must be monitored and maintained, and an annual survey of each species must be conducted. Fourth, the "importance value" of any dominant plant species in the canopy and subcanopy at any monitoring station cannot be reduced by more than fifty percent, and the average "importance value" of any dominant plant species cannot be reduced by more than twenty-five percent.[69]

These types of efforts, constantly being adjusted to take account of new information in a field where knowledge is rapidly expanding, are fertile sources of information for wetland standard writers and 401 certification decision makers.

VIII. SUMMARY OF ACTIONS NEEDED

This handbook has only scratched the surface of issues surrounding effective use of 401 certification to protect wetlands. The preceding discussion and examples from active States have highlighted possible approaches for all States to incorporate into their 401 certification programs. The handbook shows that there are many things that a State can act on right away to improve the effectiveness of 401 certification to protect the integrity of its wetlands. At the same time, there are improvements to water quality standards for wetlands which will have to take place within a longer timeframe.

A. Steps States Can Take Right Away

- **All states should begin by explicitly incorporating wetlands into their definitions of state waters in both state water quality standards regulations, and in state 401 certifications regulations.**

- **States should develop or modify their regulations and guidelines for 401 certification and water quality standards to clarify their programs, codify their decision process, and to incorporate special wetlands considerations into the more traditional water quality approaches.**

- **States should make more effective use of their existing narrative water quality standards (including the antidegradation policy) to protect wetlands.**

- **States should initiate or improve upon existing inventories of their wetland resources.**

- **States should designate uses for their wetlands based on estimates of wetland functions typically associated with given wetland types. Such potential uses could be verified for individual applications with an assessment tool such as the Wetlands Evaluation Technique or Habitat Evaluation Procedure.**

- **States should tap into the potential of the outstanding resource waters tier of the antidegradation policy for wetlands. It may not be an appropriate designation for all of a state's wetlands, but it can provide excellent protection to particularly valuable or ecologically sensitive wetlands from both physical and chemical degradation.**

- **States should incorporate wetlands and 401 certification into their other water quality management processes. Integrating this tool with other mechanisms such as coastal zone management programs, point and nonpoint source programs, and water quality management plans will help fill the gaps of each individual tool and allow better protection of wetlands systems from the whole host of physical, chemical, and biological impacts.**

Time and the courts may be needed to resolve some of the more complicated and contentious issues surrounding 401 certification such as which federal permits and licenses require 401 certification. EPA intends to support States in resolving such issues.

OWP, in cooperation with the Office of Water Regulations and Standards (OWRS), will build on this 401 certification handbook by developing guidance in FY 89-90 on water quality standards for wetlands. The guidance will provide the framework for States to incorporate wetlands into their water quality standards. The guidance will: require States to include wetlands as "waters of the State;" provide methods to designate wetland uses that recognize differences in wetland types and functions; address some chemical-specific and narrative biological criteria for wetlands; and discuss implementation of State antidegradation policies.

B: Laying the Groundwork for Future Decisions

Many States are successfully applying their existing narrative and, to a lesser extent, numeric water quality criteria to their wetland resources. Nevertheless, more work is needed to test the overall adequacy and applicability of these standards for wetlands, and to develop additional criteria where needed.

For example, existing criteria related to pH do not account for the extreme natural acidity of many peat bogs nor the extreme alkalinity of certain fens. Also, many existing criteria focus too extensively on the chemical quality of the water column without adequately protecting the other physical and biological components which are an integral part of wetland aquatic systems. Some numeric criteria for chemicals may not be protective enough of species (particularly bird species) which feed, breed, and/or spend a portion of their life cycle in wetlands. Hydrological changes can have severe impacts on wetland quality, but these changes are rarely addressed in traditional water quality standards.

Research of interest to State programs is being sponsored by the Wetlands Research Program of EPA's Office of Research and Development (ORD). Research covers three areas: Cumulative Effects, Water Quality, and Mitigation. Although these efforts will be developed over several years, interim products will be distributed to the States. States may find these products of use when developing criteria and standards, when identifying and designating wetlands as outstanding resource waters, and when making 401 certification decisions.

Cumulative Effects:

EPA's research on cumulative effects of wetlands takes a regional perspective. Through a series of regional pilot studies involving landscape analyses, ORD is correlating water quality conditions at the outlets of major watersheds with the percentage of wetlands in these watersheds. The types of wetlands, their position, and non-wetland factors are also being analyzed. The results will allow water quality managers in these regions to specify the optimal percentage and combination of various types of wetlands needed to maintain water quality of lakes and rivers. Such watershed criteria could be used to guide efforts to create or restore wetlands for the purpose of intercepting and improving the quality of nonpoint runoff.

The pilot studies will also determine which wetland features can be used to predict wetland functions. Once differences among wetlands can be identified based on their functions, it will be possible to classify particular wetlands with regard to specific designated uses.

The cumulative effects program is using the results of the pilot studies as technical support for developing a "Synoptic Assessment Method". This method has already been used to rank watersheds within certain regions, according to the likely cumulative benefits of their wetlands. Also, sources of information useful for designating uses of individual wetlands were described by ORD in EPA's draft guidance for Advance Identification Appendix D.[70] Information on regionally rare or declining wetland wildlife, which could be used as one basis for establishing "special aquatic areas" in selected wetlands, is also available from the ORD Wetlands Research Team at the Corvallis EPA Lab.

Water Quality:

Another ORD study, being implemented through the Duluth Lab, is examining impacts to the water quality and biota of 30 wetlands, before and after regional development. This study will be useful, as part of 401 certification, for developing performance standards for activities which may affect wetland water quality.

Several research projects being proposed by the Wetland Research Program could produce information very useful to water quality managers. These are described in ORD's publication, "Wetlands and Water Quality: A Research and Monitoring Implementation Plan for the Years 1989-1994". Many of these proposals are planned, but will hinge upon funding decisions in future budget years. Those which drew the most support from a 1988 EPA workshop of scientists and State program administrators were as follows:

- **Water Quality Criteria to Protect Wetland Function.** Existing quality criteria for surface waters would be reviewed for applicability to wetlands. Methods for biological and chemical monitoring of wetlands would be refined, and a field manual produced.

- **Ecological Status and Trends of the Wetland Resource.** A nationwide network would be established to monitor the wetland resource. Field surveys would define the expected range of numerical values within each region for particular chemicals and especially, for biological community metrics, across a gradient of sites ranging from nearly-pristine to severely disturbed.

- **Waste Assimilative Limits of Wetlands.** Observable features which determine the long-term ability of wetlands to retain contaminants and nutrients would be tested. "Safe" loading limits for various substances would be proposed for specific wetland types or regions. Similar kinds of information would also become available from a research effort focused specifically on artificial wetlands and coordinated by EPA-Cincinnati, in cooperation with the Corvallis and Duluth Labs. That study would recommend engineering design factors essential in wetlands constructed by municipalities for tertiary wastewater treatment.

Mitigation:

Information useful to 401 certification will also originate from ORD'S mitigation research. This research aims to determine if created and restored wetlands replace functions lost by wetland destruction permitted under Section 404. The research is organized to (1) synthesize current knowledge on wetland creation and restoration, (2) compile 404 permit information on created and restored wetlands, and (3) compare created and naturally occurring wetlands. Research results will be incorporated into a "Mitigation Handbook" useful for designing and evaluating mitigation projects. A literature synthesis being developed as a Provisional Guidance Document will be available in 1989. A provisional version of the handbook will be produced in 1990. This will assist States in identifying areas at greatest risk due to 404 permit activities and thus help target 401 certification and water quality standards activities.

APPENDIX A

Provided below are State 401 certifictation contacts and EPA wetlands contacts who can provide assistance in applying 401 to wetlands.

EPA has asked the Council of State Governments (CSG) to maintain a database of State wetland contacts and programs. In order to help keep the database up to date, please contact CSG when you have changes in your program or staff contacts, or if you come across inaccuracies in other State programs. You can access this database using virtually any computer with a modem. In order to obtain your free username and password contact:

The Council of State Governments
P.O. Box 11910, Iron Works Pike
Lexington, Kentucky 40578
phone: (606) 252-2291

FEDERAL 401 CERTIFICATION CONTACTS FOR WETLANDS

EPA Headquarters:

Dianne Fish
Wetlands Strategies Team
(A-104F)
Environmental Protection Agency
401 M Street, SW
Washington, D.C. 20460
Phone: (202) 382-7071

Jeanne Melanson
Outreach and State Programs Staff
(A-104F)
Environmental Protection Agency
401 M Street, SW
Washington, D.C. 20460
Phone: (202) 475-6745

EPA Region Contacts:

EPA Region I
Doug Thompson, Chief
Wetlands Protection Section (WPP-1900)
John F. Kennedy Federal Building
Boston, Massachusetts 02203
(617) 565-4421

EPA Region II
Mario del Vicario, Chief
Marine/Wetlands Prot. Branch (2WM-MWP)
26 Federal Plaza
New York, New York 10278
(212) 264-5170

EPA Region III
Barbara De Angelo, Chief
Marine & Wetlands Policy Sect. (3ES42)
841 Chestnut Street
Philadelphia, Pennsylvania 19107
(215) 597-1181

EPA Region IV
Tom Welborn, Acting Chief
Wetlands Section (4WM-MEB)
345 Courtland Street, N.E.
Atlanta, Georgia 30365
(404) 347-2126

EPA Region V
Doug Ehorn, Deputy Chief
Water Quality Branch (5WQ-TUB8)
230 South Dearborn Street
Chicago, Illinois 60604
(312) 886-0139

EPA Region VI
Jerry Saunders, Chief
Technical Assistance Sect. (6E-FT)
1445 Ross Avenue
12th Floor, Suite 1200
Dallas, Texas 75202
(214) 655-2260

EPA Region VII
B. Katherine Biggs, Chief
Environmental Review Branch (ENVR)
726 Minnesota Avenue
Kansas City, Kansas 66101
(913) 236-2823

EPA Region VIII
Gene Reetz, Chief
Water Quality Requirements Sect.
One Denver Place
Suite 1300
999 18th Street
Denver, Colorado 80202
(303) 293-1568

EPA Region IX
Phil Oshida, Chief
Wetlands Section (W-7)
215 Fremont Street
San Francisco, California 94105
(415) 974-7429

EPA Region X
Bill Riley, Chief
Water Resources Assessment (WD-138)
1200 Sixth Avenue
Seattle, Washington 98101
(206) 442-1412

C.D. Robison, Jr.
Alaska Operations Office, Region X
Federal Building Room E551
701 C Street, Box 19
Anchorage, Alaska 99513

EPA Wetlands Research
Eric Preston
Environmental Research Lab
Corvallis/ORD
200 S.W. 35 Street
Corvallis, OR 97333
(503) 757-4666

Bill Sanville
Environmental Research
Laboratory/ORD
6201 Congdon Blvd
Duluth, MN 55804
(218) 720-5723

State 401 CERTIFICATION CONTACTS

Brad Gane
Field Operation Division
Dept. of Enviromental Management
2204 Perimeter Road
Mobile, Alabama 36615
(205)479-2236

Walter Tatum
Field Operation Division
Dept. of Enviromental Management
2204 Perimeter Road
Mobile, Alabama 36615
(205) 968-7576

Doug Redburn
Dept. of Enviromental Conservation
3220 Hospital Drive
Juneau, Alaska 99811
(907) 465-2653

Mr. Dick Stokes
Southeast Office
Department of Environmental Conservation
P.O. Box 2420
9000 Old Glacier Highway
Juneau, Alaska 99803
(907) 789-3151

Mr. Tim Rumfelt
Southcentral Office
Department of Environmental Conservation
437 E Street, Second Floor
Anchorage, Alaska 99501
(907) 274-2533

Mr. Paul Bateman
Northern Office (Arctic)
Department of Environmental Conservation
1001 Noble Street, Suite 350
Fairbanks, Alaska 99701
(907) 452-1714

Ms. Joyce Beelman
Northern Office (Interior)
Department of Environmental Conservation
1001 Noble Street, Suite 350
Fairbanks, Alaska 99701
(907) 452-1714

Steve Drown
Dept. of Pollution Control and Ecology
8001 National Drive
Little Rock, Arkansas 72207
(501) 652-7444

Jack Hodges
State Water Resources Control Board
P.O. Box 100
Sacramento, California 95801-0100
(916) 322-0207

Jon Scherschligt
Water Quality Control Division
4210 E. 11th Avenue
Denver, Colorado 80220
(303) 320-8333

Douglas E. Cooper
Wetlands Management Section
Dept. of Env. Prot. Water Resources
Room 203, State Office Building
165 Capitol Avenue
Hartford, Connecticut 06106
(203) 566-7280

William F. Moyer
Dept. of Natural Resources and
Environmental Control
89 King's Highway
P.O. Box 1401
Dover, Delaware 19903
(302) 736-4691

Richmond Williams
Dept. of Natural Resources and
Environmental Control
Legal Office
89 King's Highway
P.O. Box 1401
Dover, Delaware 19903
(302) 736-4691

Randall L. Armstrong
Division of Environmental Permitting
Dept. of Env. Regulation
2600 Blairstone Road
Tallahassee, Florida 32399
(904) 488-0130

Mike Creason
Environmental Protection Division
Dept. of Natural Resources
205 Butler Street S.E.
Floyd Towers East
Atlanta, Georgia 30334
(404) 656-4887

James K. Ikeda
Environmental Protection & Health
Services Division
Department of Health
1250 Punchbowl Street
P.O. Box 3378
Honolulu, Hawaii 96801-9984
(808) 548-6455

John Winters
Water Quality and Standards Branch
Dept. of Env. Management
105 S. Meridian Street
Indianapolis, Indiana 46206-6015
(317) 243-5028

Al Keller
Environmental Protection Agency
2200 Churchill Road
Springfield, Illinois 62706
(217) 782-0610

Bruce Yurdin
Environmental Protection Agency
2200 Churchill Road
Springfield, Illinois 62706
(217) 782-0610

Jerry Yoder
Bureau of Water Quality
Division of Environmental Quality
450 West State Street
Boise, Idaho 83720
(208) 334-5860

Ralph Turkle
Department of Natural Resources
900 East Grand Avenue
Des Moines, Iowa 50319
(515) 281-7025

Lavoy Haage
Department of Natural Resources
900 East Grand Avenue
Henry A. Wallace Office Building
Des Moines, Iowa 50319
(515) 281-8877

Larry Hess
Dept. of Health and Environment
Building 740
Forbes Field
Topeka, Kansas 66620
(913) 862-9360

Paul Beckley
Division of Water
Dept. of Natural Resources
Fort Boone Plaza
Frankfort, Kentucky 40601
(502) 564-310, ext. 495

Dale Givens
Water Pollution Control
P.O. Box 44091
Baton Rouge, Louisiana 70804
(504) 342-6363

Donald T. Witherill
Dept. of Env. Protection
Division of Licensing
Augusta, Maine 04333
(207) 289-2111

Mary Jo Garries
Division of Standards
Department of the Environment
201 West Preston Street
Baltimore, Maryland 21201
(301) 225-6293

Jo Ann Watson
Division of Standards
Dept. of Health and Mental Hygiene
201 West Preston Street
Baltimore, Maryland 21201
(301) 225-6293

Ken Chrest
Water Quality Bureau
Cogswell Building
Helena, Montana 59620
(406) 444-2406

Bill Gaughan
Div. of Water Pollution
Dept. of Env. Quality Engineering
1 Winter Street
Boston, Massachusetts 02108
(617) 292-5658

Judy Perry
Regulatory Branch Div. of Water Pollution
Dept. of Env. Quality Engineering
1 Winter Street
Boston, Massachusetts 02108
(617) 292-5655

Les Thomas
Land and Water Management Div.
Dept. of Natural Resources
P.O. Box 30028
Lansing, Michigan 48909
(517) 373-9244

Robert Seyfarth
Bureau of Pollution Control
Dept. of Natural Resources
Box 10385
Jackson, Mississippi 39209
(601) 961-5171

Charles Chisolm
Bureau of Pollution control
Dept. of Natural Resources
Box 10385
Jackson, Mississippi 39209
(601) 961-5171

Jim Morris
Water Quality Management Section
Dept. of Natural Resources
Box 10385
Jackson, Mississippi 39209
(601) 961-5151

Louis Flynn
MPLA
1935 West County Road B-2
Roseville, Minnesota 55113
(612) 296-7355

Laurie K. Collerot
Water Supply and Pollution Control
Hazen Drive
P.O. Box 95
Concord, New Hampshire 03301
(603) 271-2358

Fred Elkind
Water Supply and Pollution Control
Dept. of Env. Services
Hazen Drive
P.O. Box 95
Concord, New Hampshire 03301
(603) 271-2358

Ray Carter
Water Supply and Pollution Control
Hazen Drive
P.O. Box 95
Concord, New Hampshire 03301
(603) 271-2358

George Danskin
Div. of Regulatory Affairs
Dept. of Env. Conservation
50 Wolf Road
Albany, New York 12233
(518) 457-2224

William Clarke
Div. of Regulatory Affairs
Dept. of Env. Conservation
50 Wolf Road
Albany, New York 12233
(518) 457-2224

U. Gale Hutton
Water Quality Division
Dept. of Env. Control
P.O. Box 94877
State House Station
Lincoln, Nebraska 68509-4877
(402) 471-2186

George Horzepa
Division of Water Resources
Dept. of Env. Protection
CN 029
Trenton, New Jersey 08625
(609) 633-7021

Barry Chalofsky
Division of Water Resources
Dept. of Env. Protection
CN 029
Trenton, New Jersey 08625
(609) 633-7021

Robert Piel
Div. of Coastal Resources
Dept. of Env. Protection
CN 401
Trenton, New Jersey 08625
(609) 633-7021

David Tague
Env. Improvement Division
P.O. Box 968
Sante Fe, New Mexico 87504-0968
(505) 827-2822

Michael T. Sauer
State Dept. of Health
1200 Missouri avenue
Bismarck, North Dakota 58505
(701) 224-2354

Paul Wilms
Div. of Env. Management
Department of Natural Resources
and Community Development
P.O. Box 27687
Raleigh, North Carolina 27611
(919) 733-7015

Bill Mills
Water Quality Section
Department of Natural Resources
P.O. Box 27687
Raleigh, North Carolina 27611
(919) 733-5083

Colleen Crook
Div. of Water Quality and
Ohio EPA
1800 Watermark Drive
P.O. Box 1049
Columbus, Ohio 43266-0149
(614) 981-7130

Brooks Kirlin
Water Resource Board
P.O. Box 53585
Oklahoma City, Oklahoma 73152
(405) 271-2541

Glen Carter
Dept. of Env. Quality
P.O. Box 1760
Portland, Oregon 97207
(503) 229-5358

Louis W. Bercheni
Bureau of Water Quality
Dept. of Env. Resources
P.O. Box 2063
Harrisburg, Pennsylvania 17120
(717) 787-2666

Peter Slack
Bureau of Water Quality
Dept. of Env. Resources
P.O. Box 2063
Harrisburg, Pennsylvania 17120
(717) 787-2666

Edward S. Szymanski
Dept. of Env. Management
Division of Water Resources
291 Promenade Street
Providence, Rhode Island 02908-5767
(401) 277-3961

Carolyn Weymouth
Office of Environmental Coordination
Department of Environmental
Management
83 Park Street
Providence, Rhode Island 02903
(401) 277-3434

Chester E. Sansbury
Division of Water Quality
Dept. of Health and Env. Control
2600 Bull Street
Columbia, South Carolina 29201
(803) 758-5496

Larry Bowers
Div. of Water Pollution Control
Dept. of Health and Env.
150 Ninth North Avenue
Nashville, Tennessee 37203
(615) 741-7883

Robert Sileus
Water Commission
P.O. Box 13087
Capitol Station
Austin, Texas 78711
(512) 463-8202

Dr. Donald Hilden
Bureau of Water Pollution Control
P.O. Box 45500
Salt Lake City, Utah 84145
(801) 533-6146

Carl Pagel
Agency of Natural Resources
Dept. of Environmental Conservation
103 S. Main Street
Waterbury, Vermont 05676
(802) 244-6951

Steve Syz
Agency of Natural Resources
Dept. of Env. Conservation
103 S. Main Street
Waterbury, Vermont 05676
(802) 244-6951

Jean Gregory
Office of Water Resources Management
Water Control Board
P.O. Box 11143
Richmond, Virginia 23230
(804) 367-6985

Mike Carnavale
Water Quality Division
State Dept. of Env. Quality
Herschler Building
Cheyenne, Wyoming 82202
(307) 777-7781

Mike Palko
Dept. of Ecology
Mail Stop PV-11
Olympia, Washington 98504
(206) 459-6289

John Schmidt
Water Resources Division
Dept. of Natural Resources
1201 Greenbrier Street
Charleston, West Virginia 25311
(304) 348-2108

Jim Rawson
Wildlife Division
Dept. of Natural Resources
P.O. Box 67
Elkins, West Virginia 26241
(304) 636-1767

Scott Hausmann
Bureau of Water Regulation and Zoning
Dept. of Natural Resources
P.O. Box 7921
Madison, Wisconsin 53701
(608) 266-7360

APPENDIX B
FEDERAL DEFINITIONS

The federal definition of "waters of the United States" is (40 CFR Section 232.2(q)):

(1) All waters which are currently used, were used in the past, or may be susceptible to use in interstate or foreign commerce, including all waters which are subject to the ebb and flow of the tide;

(2) All interstate waters including interstate wetlands;

(3) All other waters such as intrastate lakes, rivers, streams (including intermittent streams), mudflats, sandflats, wetlands, sloughs, prairie potholes, wet meadows, playa lakes, or natural ponds, the use, degradation or destruction of which would or could affect interstate or foreign commerce including any such waters:

   (i) Which are or could be used by interstate or foreign travelers for recreational or other purposes; or

   (ii) From which fish or shellfish could be taken and sold in interstate or foreign commerce;

   (iii) Which are used or could be used for industrial purposes by industries in interstate commerce;*

(4) All impoundments of waters otherwise defined as waters of the United States under this definition;

(5) Tributaries of waters identified in paragraphs 1-4.

(6) The territorial sea;

(7) Wetlands adjacent to waters (other than waters that are themselves wetlands) identified in 1-6; waste treatment systems, including treatment ponds or lagoons designed to meet the requirements of CWA (other than cooling ponds as defined in 40 CFR § 423.11(m) which also meet criteria in this definition) are not waters of the United States.

(* Note: EPA has clarified that waters of the U.S. under the commerce connection in (3) above also include, for example, waters:

Which are or would be used as habitat by birds protected by Migratory Bird Treaties or migratory birds which cross State lines;

Which are or would be used as habitat for endangered species;

Used to irrigate crops sold in interstate commerce.)

The federal definition of "wetlands" (40 CFR § 232.2(r)). Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.

APPENDIX C

SCOPE OF PROJECT REVIEW: PENNSYLVANIA DAM PROPOSAL EXAMPLE

The dam proposed by the City of Harrisburg was to be 3,000 feet long and 17 feet high. The dam was to consist of 32 bottom hinged flap gates. The dam would have created an impoundment with a surface area of 3,800 acres, a total storage capacity of 35,000 acre feet, and a pool elevation of 306.5 feet. The backwater would have extended approximately eight miles upstream on the Susquehanna River and approximately three miles upstream on the Conodoguinet Creek.

The project was to be a run-of-the-river facility, using the head difference created by the dam to create electricity. Maximum turbine flow would have been 10,000 cfs (at a nethead of 12.5) and minimum flow would have been 2,000 cfs. Under normal conditions, all flows up to 40,000 cfs would have passed through the turbines.

The public notice denying 401 certification for this project stated as follows:

1. The construction and operation of the project will result in the significant loss of wetlands and related aquatic habitat and acreage. More specifically:

    a. The destruction of the wetlands will have an adverse impact on the local river ecosystem because of the integral role wetlands play in maintaining that ecosystem.

    b. The destruction of the wetlands will cause the loss of beds of emergent aquatic vegetation that serve as habitat for juvenile fish. Loss of this habitat will adversely affect the relative abundance of juvenile and adult fish (especially smallmouth bass).

    c. The wetlands which will be lost are critical habitat for, among other species, the yellow crowned night heron, black crowned night heron, marsh wren and great egret. In addition, the yellow crowned night heron is a proposed State threatened species, and the marsh wren and great egret are candidate species of special concern.

    d. All affected wetlands areas are important and, to the extent that the loss of these wetlands can be mitigated, the applicant has failed to demonstrate that the mitigation proposed is adequate. To the extent that adequate mitigation is possible, mitigation must include replacement in the river system.

e. Proposed riprapping of the shoreline could further reduce wetland acreage. The applicant has failed to demonstrate that there will not be an adverse water quality and related habitat impact resulting from riprapping.

f. Based upon information received by the Department, the applicant has underestimated the total wetland acreage affected.

2. The applicant has failed to demonstrate that there will be no adverse water quality impacts from increased groundwater levels resulting from the project. The ground water model used by the applicant is not acceptable due to erroneous assumptions and the lack of a sensitivity analysis. The applicant has not provided sufficient information concerning the impact of increased groundwater levels on existing sites of subsurface contamination, adequacy of subsurface sewage system replacement areas and the impact of potential increased surface flooding. Additionally, information was not provided to adequately assess the effect of raised groundwater on sewer system laterals, effectiveness of sewer rehabilitation measures and potential for increased flows at the Harrisburg wastewater plant.

3. The applicant has failed to demonstrate that there will not be a dissolved oxygen problem as a result of the impoundment. Present information indicates the existing river system in the area is sensitive to diurnal, dissolved oxygen fluctuation. Sufficient information was not provided to allow the Department to conclude that dissolved oxygen standards will be met in the pool area. Additionally, the applicant failed to adequately address the issue of anticipated dissolved oxygen levels below the dam.

4. The proposed impoundment will create a backwater on the lower three miles of the Conodoguinet Creek. Water quality in the Creek is currently adversely affected by nutrient problems. The applicant has failed to demonstrate that there will not be water qualitv degradation as a result of the impoundment.

5. The applicant has failed to demonstrate that there will not be an adverse water quality impact resulting from combined sewer overflows.

6. The applicant has failed to demonstrate that there will not be an adverse water quality impact to the 150 acre area downstream of the proposed dam and upstream from the existing Dock Street dam.

7. The applicant has failed to demonstrate that the construction and operation of the proposed dam will not have an adverse impact on the aquatic resources upstream from the proposed impoundment. For example, the suitability of the impoundment for smallmouth bass spawning relative to the frequency of turbid

conditions during spawning was not adequately addressed and construction of the dam and impoundment will result in a decrease in the diversity and density of the macroinvertebrate community in the impoundment area.

8. Construction of the dam will have an adverse impact on upstream and downstream migration of migratory fish (especially shad). Even with the construction of fish passageways for upstream and downstream migration, significant declines in the numbers of fish successfully negotiating the obstruction are anticipated.

9. The applicant has failed to demonstrate that there will not be an adverse water quality impact related to sedimentation within the pool area.

APPENDIX D

EXAMPLES OF CERTIFICATION CONDITIONS

MARYLAND

Maryland certified with conditions the fill/alteration of 6.66 acres of non-tidal wetlands as part of the construction of an 18 hole golf course and a residential subdivision. Approximately three-fourths of the entire site of 200 acres had been cleared for cattle grazing and agricultural activities in the past. As a result, a stream on the east side of the property with no buffer had been severely degraded. An unbuffered tractor crossing had also degraded the stream. A palustrine forested wetland area on the southeast side of the property received stormwater runoff from a highway bordering the property and served as a flood storage and ground water recharge area. Filling this area for construction of a fairway would eliminate some 4.5 acres of wetlands. Additionally, other smaller wetland areas on the property, principally around an old farm pond that was to be fashioned into four separate ponds for water traps, were proposed to be altered or lost as a result of the development.

The Corps did not exercise its discretionary authority to require an individual permit and thus the project was permitted under a nationwide permit (26). The State decided to grant certification, conditioned on a number of things that it believed would improve the water quality of the stream in the long run.

The filled wetland areas had to be replaced on an acre-for-acre basis on the property and in particular, the 4.5 acre forested palustrine wetland had to be replaced onsite with a wetland area serving the same functions regarding stormwater runoff from the highway.

Some of the other conditions placed on the certification were as follows:

1. The applicant must obtain and certify compliance with a grading and sediment control plan approved by the [name of county] Soil Conservation District;

2. Stormwater runoff from impervious surfaces shall be controlled to prevent the washing of debris into the waterway. Stormwater drainage facilities shall be designed, implemented, operated and maintained in accordance with the requirements of the [applicable county authority];

3. The applicant shall ensure that fish species are stocked in the ponds upon completion of the construction phase in accordance with the requirements of the [fisheries division of the natural resources department of the State];

4. The applicant shall ensure that all mitigation areas are inspected annually by a wetlands scientist to ensure that all wetlands are functioning properly;

5. A vegetated buffer shall be established around the existing stream and proposed ponds;

6. Biological control methods for weed, insects and other undesirable species are to be employed whenever possible on the greens, tees, and fairways located within or in close proximity to the wetland or waterways;

7. Fertilizers are to be used on greens, tees, and fairways only. From the second year of operation, all applications of fertilizers at the golf course shall be in the lower range dosage rates [specified]. The use of slow release compounds such as sulfur-coated urea is required. There shall be no application of fertilizers within two weeks of verticutting, coring or spiking operations.

**** WEST VIRGINIA ****

THE FOLLOWING GENERAL CONDITIONS APPLY TO ALL NATIONWIDE PERMITS IN WEST VIRGINIA:

1. Permittee will investigate for water supply intakes or other activities immediately downstream which may be affected by suspended solids and turbidity increases caused by work in the watercourse. He will give notice to operators of any such water supply intakes before beginning work in the watercourse in sufficient time to allow preparation for any change in water quality.

2. When no feasible alternative is available, excavation, dredging or filling in the watercourse will be done to the minimum extent practicable.

3. Spoil materials from the watercourse or onshore operations, including sludge deposits, will not be dumped into the water course or deposited in wetlands.

4. Permittee will employ measures to prevent or control spills from fuels, lubricants, or any other materials used in construction from entering the watercourse.

5. Upon completion of earthwork operations, all fills in the watercourse or onshore and other areas disturbed during construction, will be seeded, riprapped, or given some other type of protection from subsequent soil erosion. If riprap is utilized, it is to be of such weight and size that bank stress or slump conditions will not be created due to its placement. Fill is to be clean and of such composition that it will not adversely effect the biological, chemical or physical properties of the receiving waters.

6. Runoff from any storage areas or spills will not be allowed to enter storm sewers without acceptable removal of solids, oils and toxic compounds. All spills will promptly be reported to the appropriate Department of Natural Resources office.

7. Best Management Practices for sediment and erosion control as described in the 208 Construction Water Quality Management Plan are to be implemented.

8. Green concrete will not be permitted to enter the watercourse unless contained by tightly sealed forms or cells. Concrete handling equipment will not discharge waste washwater into the watercourse or wetlands without adequate wastewater treatment.

9. No instream work is permissible during the fish spawning season April through June.

10. Removal of mature riparian vegetation not directly associated with project construction is prohibited.

11. Instream equipment operation is to be minimized and should be accomplished during low flow periods.

12. Nationwide permits are not applicable for activities on Wild and Scenic Rivers or study streams, streams on the Natural Streams Preservation List or the New River Gorge National River. These streams include New River (confluence with Gauley to mouth of Greenbrier); Greenbrier River (mouth to Knapps Creek), Birch River (mouth to Cora Brown Barge in Nicholas County), Anthony Creek, Cranberry Run, Bluestone River, Gauley River, and Meadow River.

13. Each permittee shall follow the notice requirements contained in Section 9 of the Department of Natural Resources Regulations for State Certification of Activities Requiring Federal Licenses and Permits, Chapter 20-1, Series XIX (1984).

14. Each permittee shall, if he does not understand or is not aware of applicable Nationwide Permit conditions, contact the Corps of Engineers prior to conducting any activity authorized by a nationwide permit in order to be advised of applicable conditions.

** ALASKA **

EXAMPLES OF CERTIFICATION CONDITIONS REQUIRED FOR NATIONWIDE PERMIT 26 FROM ALASKA

(26) Discharges of dredged or fill material into the waters listed in subparagraph (i) and (ii) of this paragraph which do not cause the loss or substantial adverse modification of 10 acres or more of waters of the United States, including wetlands. For discharges which cause the loss or substantial adverse modification of 1 to 10 acres of such waters, including wetlands, notification of the District Engineer is required in accordance with 330.7 of this part (see Section 2 of this Public Notice).

(i) Non-tidal rivers, streams, and their lakes and impoundments, including adjacent wetlands, that are located above the headwaters.

(ii) Other non-tidal waters of the United States, including adjacent wetlands, that are not part of the surface tributary system to interstate waters or navigable waters of the United States (i.e., isolated waters).

REGIONAL CONDITION H: Work in a designated anadromous fish stream is subject to authorization from the Alaska Department of Fish and Game. (No change from REGIONAL CONDITION H previously published in SPN 84-7.)

REGIONAL CONDITION J:

a. If, during review of the pre-discharge notification, the Corps of Engineers or the designated State of Alaska reviewing officials determine that the proposed activity would occur in any of the following areas, the applicant will be advised that an individual 404 permit will be required. Where uncertainty exists, the Corps will send pre-discharge notification to the designated State officials for a determination.

1. National Wildlife Refuges
2. National Parks and Preserves
3. National Conservation Areas
4. National Wild and Scenic Rivers
5. National Experimental Areas
6. State Critical Habitat AReas
7. State Sanctuaries
8. State Ranges and Refuges
9. State Eagle Preserves
10. State Ecological Reserves and Experimental Areas
11. State Recreation Areas

12. Wetlands contiguous with designated anadromous fish streams
13. Headwaters and isolated wetlands in designated public water supply watersheds of Craig, Hoonah, Hydaburg, Anchorage, Cordova, Seldovia and Kodiak
14. Sitka Area: Wetlands in the Swan Lake Area Meriting Special Attention (AMSA) in the district Coastal Management Plan
15. Anchorage area: Designated Preservation and Conservation Wetlands in the Wetlands Management Plan
16. Bethel area: Designated Significant Wetlands in the district Coastal Management Plan not covered under General Permit 83-4
17. Hydaburg area: The six AMSA's of the district Coastal Management Plan
18. Bering Strait area: All designated conservation AMSA's of the district Coastal Management Plan
19. Juneau area: Designated Sensitive Wetlands of the district Coastal Management Plan
20. NANA: Designated Special Use Areas and Restricted/ Sensitive areas in the district Coastal Management Plan
21. Tanana Basin Area Plan: type A-1 wetlands in the Alaska Rivers Cooperative State/Federal Study
22. Susitna Area Plan: type A-1 wetlands in the Alaska Rivers Cooperative State/Federal Study
23. High value headwaters and isolated wetlands identified once the ongoing Wetlands Management Plans or Guides listed in b-5 (below) are completed
24. Alaska Natural Gas Pipeline Corridor designated type A and B wetlands
25. Headwaters and isolated waters which include identified bald eagle, peregrine falcon, and trumpeter swan nesting areas
26. ADF&G identified waterfowl use areas of statewide significance
27. Designated caribou calving areas.

Any individual permit issued in locations covered by district coastal management plans, State or Federal regional wetlands plans or local wetlands plans (numbers 14 through 23 above) will be consistent with the plan provisions for the specific wetland type and may require adding stipulations.

Oil and gas activities in the North Slope Borough which involve the discharge of dredged or fill material into waters including wetlands are not covered by the previous nationwide permit under 33 CFR 330.4(a) and (b) and are not covered under the nationwide permit 26. These activities require individual 404 permits or other general permits. These activities were previously excluded by the Corps of Engineers Special Public Notice 84-3 dated March 9, 1984.

b. Pre-discharge notification received by the Corps of Engineers for the discharge of dredged or fill material in the following areas will be provided to designated State agencies which include (1) the appropriate ADEC Regional Environmental Supervisor, (2) the appropriate ADF&G Regional Habitat Supervisor, (3) the appropriate DGC regional contact point, and (4) the appropriate DNR regional contact (should DNR indicate interest in receiving notices).

1. Headwater tributaries of designated anadromous fish streams and their adjacent contiguous wetlands
2. Open water areas of isolated wetlands greater than 10 acres and lakes greater than 10 acres above the headwaters
3. North Slope Borough wet and moist tundra areas not already covered by APP process
4. Wet and moist tundra areas outside the North Slope Borough
5. High value headwaters and isolated wetlands identified in the following ongoing State or Federal wetland management guides or plans: Mat-Su, Kenai Borough, Valdez, North Star Borough Yukon Delta and Copper River Basin
6. Headwater or isolated wetlands within local CZM district boundaries or the identified coastal zone boundary, whichever is geographically smaller (not withstanding the requirements under "a." 14.20 (above))
7. Anchorage Area: designated Special Study areas in the Wetlands Management Plan
8. Tanana Basin Area Plan: areas designated A-2, B-1, B-2 in the Alaska River Cooperative State/Federal Study
9. Susitna Area Plan: areas designated A-2, A-3, A-4 in the Alaska River Cooperative State/Federal Study

The designated officials of the State of Alaska, and the Corps will evaluate the notifications received for the areas listed "b." above under the provisions set forth in 33 CFR 330.7 (see Section 2 of this Public Notice) which includes an evaluation of the environmental effects using the guidelines set forth in Section 404(b)(1) of the Clean Water Act. Notices shall be screened against the nationwide conditions under 330.5(b) (See Section 4 of the Public Notice) using available resource information. Conditions 330.5(b)(1), (2), (3), (4), (6), and (7) and (9) will be focused on during the State review.

The State's review of these areas under "b." above will encompass the following:

1. After receiving pre-discharge notification from the Corps, the State of Alaska shall comment verbally, and/or if time permits, in writing to the Corps District Engineer through a single State agency concerning the need for an individual permit review.

2. Existing fish and wildlife atlases and field knowledge shall be used to evaluate notices. If significant resource values are not identified for the area in question or if insufficient resource information exists, State agencies will not request an individual permit unless:

(a) An on-site field evaluation will be conducted, weather permitting, during the extended review provided under the individual permit, or;

(b) Federal resource agencies plan a similar field evaluation that could provide identical information to State resource agencies.

Should either the State review or the Corps review determine that the nationwide permit is not applicable, an individual 404 permit will be required.

New categories may be added at a later date should either the Corps or the State of Alaska recognize a need. These changes will be made available for public review through a public notice and comment period at the appropriate time.

This REGIONAL CONDITION shall be effective for the period of time that nationwide permit 26 is in effect unless the REGIONAL CONDITION is sooner revoked by the Department of the Army with prior coordination with the State of Alaska.

APPENDIX E

Subpart H—Actions To Minimize Adverse Effects

Note.—There are many actions which can be undertaken in response to § 230.10(d) to minimize the adverse effects of discharges of dredged or fill material. Some of these, grouped by type of activity, are listed in this subpart.

§ 230.70 Actions concerning the location of the discharge.

The effects of the discharge can be minimized by the choice of the disposal site. Some of the ways to accomplish this are by:

(a) Locating and confining the discharge to minimize smothering of organisms;

(b) Designing the discharge to avoid a disruption of periodic water inundation patterns;

(c) Selecting a disposal site that has been used previously for dredged material discharge;

(d) Selecting a disposal site at which the substrate is composed of material similar to that being discharged, such as discharging sand on sand or mud on mud;

(e) Selecting the disposal site, the discharge point, and the method of discharge to minimize the extent of any plume;

(f) Designing the discharge of dredged or fill material to minimize or prevent the creation of standing bodies of water in areas of normally fluctuating water levels, and minimize or prevent the drainage of areas subject to such fluctuations.

§ 230.71 Actions concerning the material to be discharged.

The effects of a discharge can be minimized by treatment of, or limitations on the material itself, such as:

(a) Disposal of dredged material in such a manner that physiochemical conditions are maintained and the potency and availability of pollutants are reduced.

(b) Limiting the solid, liquid, and gaseous components of material to be discharged at a particular site;

(c) Adding treatment substances to the discharge material;

(d) Utilizing chemical flocculants to enhance the deposition of suspended particulates in diked disposal areas.

§ 230.72 Actions controlling the material after discharge.

The effects of the dredged or fill material after discharge may be controlled by:

(a) Selecting discharge methods and disposal sites where the potential for erosion, slumping or leaching of materials into the surrounding aquatic ecosystem will be reduced. These sites or methods include, but are not limited to:

(1) Using containment levees, sediment basins, and cover crops to reduce erosion;

(2) Using lined containment areas to reduce leaching where leaching of chemical constituents from the discharged material is expected to be a problem;

(b) Capping in-place contaminated material with clean material or selectively discharging the most contaminated material first to be capped with the remaining material;

(c) Maintaining and containing discharged material properly to prevent point and nonpoint sources of pollution;

(d) Timing the discharge to minimize impact, for instance during periods of unusual high water flows, wind, wave, and tidal actions.

§ 230.73 Actions affecting the method of dispersion.

The effects of a discharge can be minimized by the manner in which it is dispersed, such as:

(a) Where environmentally desirable, distributing the dredged material widely in a thin layer at the disposal site to maintain natural substrate contours and elevation;

(b) Orienting a dredged or fill material mound to minimize undesirable obstruction to the water current or circulation pattern, and utilizing natural bottom contours to minimize the size of the mound;

(c) Using silt screens or other appropriate methods to confine suspended particulate/turbidity to a small area where settling or removal can occur;

(d) Making use of currents and circulation patterns to mix, disperse and dilute the discharge;

(e) Minimizing water column turbidity by using a submerged diffuser system. A similar effect can be accomplished by submerging pipeline discharges or otherwise releasing materials near the bottom;

(f) Selecting sites or managing discharges to confine and minimize the release of suspended particulates to give decreased turbidity levels and to maintain light penetration for organisms;

(g) Setting limitations on the amount of material to be discharged per unit of time or volume of receiving water.

§ 230.74 Actions related to technology.

Discharge technology should be adapted to the needs of each site. In determining whether the discharge operation sufficiently minimizes adverse environmental impacts, the applicant should consider:

(a) Using appropriate equipment or machinery, including protective devices, and the use of such equipment or machinery in activities related to the discharge of dredged or fill material;

(b) Employing appropriate maintenance and operation on equipment or machinery, including adequate training, staffing, and working procedures;

(c) Using machinery and techniques that are especially designed to reduce damage to wetlands. This may include machines equipped with devices that scatter rather than mound excavated materials, machines with specially designed wheels or tracks, and the use of mats under heavy machines to reduce wetland surface compaction and rutting;

(d) Designing access roads and channel spanning structures using culverts, open channels, and diversions that will pass both low and high water flows, accommodate fluctuating water levels, and maintain circulation and faunal movement;

(e) Employing appropriate machinery and methods of transport of the material for discharge.

§ 230.75 Actions affecting plant and animal populations.

Minimization of adverse effects on populations of plants and animals can be achieved by:

(a) Avoiding changes in water current and circulation patterns which would interfere with the movement of animals;

(b) Selecting sites or managing discharges to prevent or avoid creating habitat conducive to the development of undesirable predators or species which have a competitive edge ecologically over indigenous plants or animals;

(c) Avoiding sites having unique habitat or other value, including habitat of threatened or endangered species;

(d) Using planning and construction practices to institute habitat development and restoration to produce a new or modified environmental state of higher ecological value by displacement of some or all of the existing environmental characteristics. Habitat development and restoration techniques can be used to minimize adverse impacts and to compensate for destroyed habitat. Use techniques that have been demonstrated to be effective in circumstances similar to those under consideration wherever possible. Where proposed development and restoration techniques have not yet advanced to the pilot demonstration stage, initiate their use on a small scale to allow corrective action if unanticipated adverse impacts occur;

(e) Timing discharge to avoid spawning or migration seasons and other biologically critical time periods;

(f) Avoiding the destruction of remnant natural sites within areas already affected by development.

§ 230.76 Actions affecting human use.

Minimization of adverse effects on human use potential may be achieved by:

(a) Selecting discharge sites and following discharge procedures to prevent or minimize any potential damage to the aesthetically pleasing features of the aquatic site (e.g. viewscapes), particularly with respect to water quality;

(b) Selecting disposal sites which are not valuable as natural aquatic areas;

(c) Timing the discharge to avoid the seasons or periods when human recreational activity associated with the aquatic site is most important;

(d) Following discharge procedures which avoid or minimize the disturbance of aesthetic features of an aquatic site or ecosystem.

(e) Selecting sites that will not be detrimental or increase incompatible human activity, or require the need for frequent dredge or fill maintenance activity in remote fish and wildlife areas;

(f) Locating the disposal site outside of the vicinity of a public water supply intake.

§ 230.77 Other actions.

(a) In the case of fills, controlling runoff and other discharges from activities to be conducted on the fill;

(b) In the case of dams, designing water releases to accommodate the needs of fish and wildlife.

(c) In dredging projects funded by Federal agencies other than the Corps of Engineers, maintain desired water quality of the return discharge through agreement with the Federal funding authority on scientifically defensible pollutant concentration levels in addition to any applicable water quality standards.

(d) When a significant ecological change in the aquatic environment is proposed by the discharge of dredged or fill material, the permitting authority should consider the ecosystem that will be lost as well as the environmental benefits of the new system.

APPENDIX R

Policy on the Use of Biological Assessments and Criteria in the Water Quality Program

APPENDIX R

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF WATER

MEMORANDUM

SUBJECT: Transmittal of Final Policy on Biological Assessments and Criteria

FROM: Tudor T. Davies, Director
Office of Science and Technology (WH-551)

TO: Water Management Division Directors
Regions I-X

Attached is EPA's "Policy on the Use of Biological Assessments and Criteria in the Water Quality Program" (Attachment A). This policy is a significant step toward addressing all pollution problems within a watershed. It is a natural outgrowth of our greater understanding of the range of problems affecting watersheds from toxic chemicals to physical habitat alteration, and reflects the need to consider the whole picture in developing watershed pollution control strategies.

This policy is the product of a broad-based workgroup chaired by Jim Plafkin and Chris Faulkner of the Office of Wetlands, Oceans and Watersheds. The workgroup was composed of representatives from seven EPA Headquarters offices, four EPA Research Laboratories, all 10 EPA Regions, U.S. Fish and Wildlife Service, U.S. Forest Service, and the States of New York and North Carolina (see Attachment B). This policy also reflects review comments to the draft policy statement issued in March of 1990. Comments were received from three EPA Headquarters offices, three EPA Research Laboratories, five EPA Regions and two States. The following sections of this memorandum provide a brief history of the policy development and additional information on relevant guidance.

Background

The Ecopolicy Workgroup was formed in response to several converging initiatives in EPA's national water program. In September 1987, a major management study entitled "Surface Water Monitoring: A Framework for Change" strongly emphasized the need to "accelerate development and application of promising biological monitoring techniques" in State and EPA monitoring programs. Soon thereafter, in December 1987, a National Workshop on Instream Biological Monitoring and Criteria reiterated this recommendation but also pointed out the importance of integrating the biological criteria and assessment methods with traditional chemical/physical methods (see Final Proceedings, EPA-905/9-89/003). Finally, at the June 1988 National Symposium on Water Quality Assessment, a workgroup of State and Federal representatives unanimously recommended the development of a national bioassessment policy that encouraged the expanded use of the new biological tools and directed their implementation across the water quality program.

Guided by these recommendations, the workgroup held three workshop-style meetings between July and December 1988. Two major questions emerged from the lengthy discussions as issues of general concern:

ISSUE 1 - How hard should EPA push for formal adoption of biological criteria (biocriteria) in State water quality standards?

ISSUE 2 - Despite the many beneficial uses of biomonitoring information, how do we guard against potentially inappropriate uses of such data in the permitting process?

Issue 1 turns on the means and relative priority of having biological criteria formally incorporated in State water quality standards. Because biological criteria must be related to local conditions, the development of quantitative national biological criteria is not ecologically appropriate. Therefore, the primary concern is how biological criteria should be promoted and integrated into State water quality standards.

Issue 2 addresses the question of how to reconcile potential apparent conflicts in the results obtained from different assessment methods (i.e., chemical-specific analyses, toxicity testing, and biosurveys) in a permitting situation. Should the relevance of each be judged strictly on a case-by-case basis? Should each method be applied independently?

These issues were discussed at the policy workgroup's last meeting in November 1988, and consensus recommendations were then presented to the Acting Assistant Administrator of Water on December 16, 1988. For Issue 1, it was determined that adapting biological criteria to State standards has significant advantages, and adoption of biological criteria should be strongly encouraged. Therefore, the current Agency Operating Guidance establishes the State adaptation of basic narrative biological criteria as a program priority.

With respect to Issue 2, the policy reflects a position of "independent application." Independent application means that any one of the three types of assessment information (i.e., chemistry, toxicity testing results, and ecological assessment) provides conclusive evidence of nonattainment of water quality standards regardless of the results from other types of assessment information. Each type of assessment is sensitive to different types of water quality impact. Although rare, apparent conflicts in the results from different approaches can occur. These apparent conflicts occur when one assessment approach detects a problem to which the other approaches are not sensitive. This policy establishes that a demonstration of water quality standards nonattainment using one assessment method does not require confirmation with a second method and that the failure of a second method to confirm impact does not negate the results of the initial assessment.

__Review of Draft Policy__

The draft was circulated to the Regions and States on March 23, 1990. The comments were mostly supportive and most of the suggested changes have been incorporated. Objections were raised by one State that using ecological measures would increase the magnitude of the pollution control workload. We expect that this will be one result of this policy but that our mandate under the Clean Water Act to ensure physical, chemical, and biological integrity requires that we adopt this policy. Another State objected to the independent application policy. EPA has carefully considered the merits of various approaches to integrating data in light of the available data, and we have concluded that independent application is the most appropriate policy at this time. Where there are concerns that the results from one approach are inaccurate, there may be opportunities to develop more refined information that would provide a more accurate conclusion (e.g., better monitoring or more sophisticated wasteload allocation modelling).

Additional discussion on this policy occurred at the Water Quality Standards for the 21st Century Symposium in December, 1990.

__What Actions Should States Take__

This policy does not require specific actions on the part of the States or the regulated community. As indicated under the Fiscal Year 1991 Agency Operating Guidance, States are required to adopt narrative biocriteria at a minimum during the 1991 to 1993 triennial review. More specific program guidance on developing biological criteria is scheduled to be issued within the next few months. Technical guidance documents on developing narrative and numerical biological criteria for different types of aquatic systems are also under development.

__Relevant Guidance__

There are several existing EPA documents which pertain to biological assessments and several others that are currently under development. Selected references that are likely to be important in implementing this policy are listed in Attachment C.

Please share this policy statement with your States and work with them to institute its provisions. If you have any questions, please call me at (FTS) 382-5400 or have your staff contact Geoffrey Grubbs of the Office of Wetlands, Oceans and Watersheds at (FTS) 382-7040 or Bill Diamond of the Office of Science and Technology at (FTS) 475-7301.

Attachments

cc: OW Office Directors
Environmental Services Division Directors, Regions I-X

Attachment A

Policy on the Use of Biological Assessments and Criteria in the Water Quality Program

May 1991

Contents

Statement of Policy

To help restore and maintain the biological integrity of the Nation's waters, it is the policy of the Environmental Protection Agency (EPA) that biological surveys shall be fully integrated with toxicity and chemical-specific assessment methods in State water quality programs. EPA recognizes that biological surveys should be used together with whole-effluent and ambient toxicity testing, and chemical-specific analyses to assess attainment/nonattainment of designated aquatic life uses in State water quality standards. EPA also recognizes that each of these three methods can provide a valid assessment of designated aquatic life use impairment. Thus, if any one of the three assessment methods demonstrate that water quality standards are not attained, it is EPA's policy that appropriate action should be taken to achieve attainment, including use of regulatory authority.

It is also EPA's policy that States should designate aquatic life uses that appropriately address biological integrity and adopt biological criteria necessary to protect those uses. Information concerning attainment/nonattainment of standards should be used to establish priorities, evaluate the effectiveness of controls, and make regulatory decisions.

Close cooperation among the States and EPA will be needed to carry out this policy. EPA will provide national guidance and technical assistance to the States; however, specific assessment methods and biological criteria should be adopted on a State-by-State basis. EPA, in its oversight role, will work with the States to ensure that assessment procedures and biological criteria reflect important ecological and geographical differences among the Nation's waters yet retain national consistency with the Clean Water Act.

Definitions

Ambient Toxicity: Is measured by a toxicity test on a sample collected from a waterbody.

Aquatic Community: An association of interacting populations of aquatic organisms in a given waterbody or habitat.

Aquatic Life Use: Is the water quality objective assigned to a waterbody to ensure the protection and propagation of a balanced, indigenous aquatic community.

Biological Assessment: An evaluation of the biological condition of a waterbody using biological surveys and other direct measurements of resident biota in surface waters.

Biological Criteria (or Biocriteria): Numerical values or narrative expressions that describe the reference biological integrity of aquatic communities inhabiting waters of a given designated aquatic life use.

Biological Integrity: Functionally defined as the condition of the aquatic community inhabiting unimpaired waterbodies of a specified habitat as measured by community structure and function.

Biological Monitoring: Use of a biological entity as a detector and its response as a measure to determine environmental conditions. Toxicity tests and biosurveys are common biomonitoring methods.

Biological Survey (or Biosurvey): Consists of collecting, processing, and analyzing a representative portion of the resident aquatic community to determine the community structure and function.

Community Component: Any portion of a biological community. The community component may pertain to the taxonomic group (fish, invertebrates, algae), the taxonomic category (phylum, order, family, genus, species), the feeding strategy (herbivore, omnivore, carnivore), or organizational level (individual, population, community association) of a biological entity within the aquatic community.

Habitat Assessment: An evaluation of the physical characteristics and condition of a waterbody (example parameters include the variety and quality of substrate, hydrological regime, key environmental parameters and surrounding land use.)

Toxicity Test: Is a procedure to determine the toxicity of a chemical or an effluent using living organisms. A toxicity test measures the degree of response of exposed test organisms to a specific chemical or effluent.

Whole-effluent Toxicity: Is the total toxic effect of an effluent measured directly with a toxicity test.

Background

Policy context

Monitoring data are applied toward water quality program needs such as identifying water quality problems, assessing their severity, and setting planning and management priorities for remediation. Monitoring data should also be used to help make regulatory decisions, develop appropriate controls, and evaluate the effectiveness of controls once they are implemented. This policy focuses on the use of a particular type of monitoring information that is derived from ambient biosurveys, and its proper integration with chemical-specific analyses, toxicity testing methods, and biological criteria in State water quality programs.

The distinction between biological surveys, assessments and criteria is an important one. Biological surveys, as stated in the section above, consist of the collection and analysis of the resident aquatic community data and the subsequent determination of the aquatic community's structure and function. A biological assessment is an evaluation of the biological condition of a waterbody using data gathered from biological surveys or other direct measures of the biota. Finally, biological criteria are the numerical values or narrative expressions used to describe the expected structure and function of the aquatic community.

Rationale for Conducting Biological Assessments

To more fully protect aquatic habitats and provide more comprehensive assessments of aquatic life use attainment/nonattainment, EPA expects States to fully integrate chemical-specific techniques, toxicity testing, biological surveys and biological criteria into their water quality programs. To date, EPA's activities have focused on the interim goal of the Clean Water Act (the Act), stated in Section 101(a)(2): To achieve; "...wherever attainable, an interim goal of water quality which provides for protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water...." However, the ultimate objective of the Act, stated in Section 101(a), goes further. Section 101(a) states: "The objective of this Act is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." Taken together, chemical, physical, and biological integrity define the overall ecological integrity of an aquatic ecosystem. Because biological integrity is a strong indicator of overall ecological integrity, it can serve as both a meaningful goal and a useful measure of environmental status that relates directly to the comprehensive objective of the Act.

Deviations from, and threats to, biological integrity can be estimated indirectly or directly. Traditional measures, such as chemical-specific analyses and toxicity tests, are indirect estimators of biological conditions. They assess the suitability of the waters to support a healthy community, but they do not directly assess the community itself. Biosurveys are used to directly evaluate the overall structural and/or functional characteristics of the aquatic community. Water quality programs should use both direct and indirect methods to assess biological conditions and to determine attainment/nonattainment of designated aquatic life uses.

Adopting an integrated approach to assessing aquatic life use attainment/nonattainment represents the next logical step in the evolution of the water quality program. Historically, water quality programs have focused on evaluating the impacts of specific chemicals discharged from discreet point sources. In 1984, the program scope was significantly broadened to include a combination of chemical-specific and whole-effluent toxicity testing methods to evaluate and predict the biological impacts of potentially toxic mixtures in wastewater and surface waters. Integration of these two indirect measures of biological impact into a unified assessment approach has been discussed in detail in national policy (49 FR 9016) and guidance (EPA-440/4-85-032). This approach has proven to be an effective means of assessing and controlling toxic pollutants and whole-effluent toxicity originating from point sources. Additionally, direct measures of biological impacts, such as biosurvey and bioassessment techniques, can be useful for regulating point sources. However, where pollutants and pollutant sources are difficult to characterize or aggregate impacts are difficult to assess (e.g., where discharges are multiple, complex, and variable; where point and nonpoint sources are both potentially important; where physical habitat is potentially limiting), direct measures of ambient biological conditions are also needed.

Biosurveys and biological criteria add this needed dimension to assessment programs because they focus on the resident community. The effects of multiple stresses and pollution sources on the numerous biological components of resident communities are integrated over a relatively long period of time. The community thus provides a useful indicator of both aggregate ecological impact and overall temporal trends in the condition of an aquatic ecosystem. Furthermore, biosurveys can detect aquatic life impacts that other available assessment methods may miss. Biosurveys detect impacts caused by: (1) pollutants that are difficult to identify chemically or characterize toxicologically (e.g., rare or unusual toxics [although biosurveys cannot themselves identify specific toxicants causing toxic impact], "clean" sediment, or nutrients); (2) complex or unanticipated exposures (e.g., combined point and non-point source loadings, storm events, spills); and perhaps most importantly, (3) habitat degradation (e.g., channelization, sedimentation, historical contamination), which disrupt the interactive balance among community components.

Biosurveys and biological criteria provide important information for a wide variety of water quality program needs. This data could be used to:

- o Refine use classifications among different types of aquatic ecosystems (e.g., rivers, streams, wetlands, lakes, estuaries, coastal and marine waters) and within a given type of use category such as warmwater fisheries;

- o Define and protect existing aquatic life uses and classify Outstanding National Resource Waters under State antidegradation policies as required by the Water Quality Standards Regulation (40 CFR 131.12);

- o Identify where site-specific criteria modifications may be needed to effectively protect a waterbody;

- o Improve use-attainability studies;

- o Fulfill requirements under Clean Water Act Sections 303(c), 303(d), 304(l), 305(b), 314, and 319;

- o Assess impacts of certain nonpoint sources and, together with chemical-specific and toxicity methods, evaluate the effectiveness of nonpoint source controls;

- o Develop management plans and conduct monitoring in estuaries of national significance under Section 320;

- o Monitor the overall ecological effects of regulatory actions under Sections 401, 402, and 301(h);

- o Identify acceptable sites for disposal of dredge and fill material under Section 404 and determine the effects of that disposal;

- o Conduct assessments mandated by other statutes (e.g., CERCLA/RCRA) that pertain to the integrity of surface waters; and

- o Evaluate the effectiveness and document the instream biological benefits of pollution controls.

Conduct of Biological Surveys

As is the case with all types of water quality monitoring programs, biosurveys should have clear data quality objectives, use standardized, validated laboratory and field methods, and include appropriate quality assurance and quality control practices. Biosurveys should be tailored to the particular type of waterbody being assessed (e.g., wetland, lake, stream, river, estuary, coastal or marine water) and should focus on community components and attributes that are both representative of the larger community and are practical to measure. Biosurveys should be routinely coupled with basic physicochemical measurements and an objective assessment of habitat quality. Due to the importance of the monitoring design and the intricate relationship between the biosurvey and the habitat assessment, well-trained and experienced biologists are essential to conducting an effective biosurvey program.

Integration of Assessment Methods and Regulatory Application

Site-specific Considerations

Although biosurveys provide direct information for assessing biological integrity, they may not always provide the most accurate or practical measure of water quality standards attainment/nonattainment. For example, biosurveys and measures of biological integrity do not directly assess nonaquatic life uses, such as agricultural, industrial, or drinking water uses, and may not predict potential impacts from pollutants that accumulate in sediments or tissues. These pollutants may pose a significant long-term threat to aquatic organisms or to humans and wildlife that consume these organisms, but may only minimally alter the structure and function of the ambient community. Furthermore, biosurveys can only indicate the presence of an impact; they cannot directly identify the stress agents causing that impact. Because chemical-specific and toxicity methods are designed to detect specific stressors, they are particularly useful for diagnosing the causes of impact and for developing source controls. Where a specific chemical or toxicity is likely to impact standards attainment/nonattainment, assessment methods that measure these stresses directly are often needed.

Independent Application

Because biosurvey, chemical-specific, and toxicity testing methods have unique as well as overlapping attributes, sensitivities, and program applications, no single approach for detecting impact should be considered uniformly superior to any other approach. EPA recognizes that each method can provide valid and independently sufficient evidence of aquatic life use impairment, irrespective of any evidence, or lack of it, derived from the other two approaches. The failure of one method to confirm an impact identified by another method would not negate the results of the initial assessment. This policy, therefore, states that appropriate action should be taken when any one of the three types of assessment determines that the standard is not attained. States are encouraged to implement and integrate all three approaches into their water quality programs and apply them in combination or independently as site-specific conditions and assessment objectives dictate.

In cases where an assessment result is suspected to be inaccurate, the assessment may be repeated using more intensive and/or accurate methods. Examples of more intensive assessment methods are dynamic modelling instead of steady state modelling, site specific criteria, dissolved metals analysis, and a more complete biosurvey protocol.

Biological Criteria

To better protect the integrity of aquatic communities, it is EPA's policy that States should develop and implement biological criteria in their water quality standards.

Biological criteria are numerical measures or narrative descriptions of biological integrity. Designated aquatic life use classifications can also function as narrative biological criteria. When formally adopted into State standards, biological criteria and aquatic life use designations serve as direct, legal endpoints for determining aquatic life use attainment/nonattainment. Per Section 131.11(b)(2) of the Water Quality Standards Regulation (40 CFR Part 131), biological criteria can supplement existing chemical-specific criteria and provide an alternative to chemical-specific criteria where such criteria cannot be established.

Biological criteria can be quantitatively developed by identifying unimpaired or least-impacted reference waters that operationally represent best attainable conditions. EPA recommends States use the ecoregion concept when establishing a list of reference waters. Once candidate references are identified, integrated assessments are conducted to substantiate the unimpaired nature of the reference and to characterize the resident community. Biosurveys cannot fully characterize the entire aquatic community and all its attributes. Therefore, State standards should contain biological criteria that consider various components (e.g., algae, invertebrates, fish) and attributes (measures of structure and/or function) of the larger aquatic community. In order to provide maximum protection of surface water quality, States should continue to develop water quality standards integrating all three assessment methods.

Statutory Basis

<u>Section 303(c)</u>

The primary statutory basis for this policy derives from Section 303 of the Clean Water Act. Section 303 requires that States adopt standards for their waters and review and revise these standards as appropriate, or at least once every three years. The Water Quality Standards Regulation (40 CFR 131) requires that such standards consist of the designated uses of the waters involved, criteria based upon such uses, and an antidegradation policy.

Each State develops its own use classification system based on the generic uses cited in the Act (e.g., protection and propagation of fish, shellfish, and wildlife). States may also subcategorize types of uses within the Act's general use categories. For example, aquatic life uses may be subcategorized on the basis of attainable habitat (e.g., cold- versus warm-water habitat), innate differences in community structure and function (e.g., high versus low species richness or productivity), or fundamental differences in important community components (e.g., warm-water fish communities naturally dominated by bass versus catfish). Special uses may also be designated to protect particularly unique, sensitive or valuable aquatic species, communities, or habitats.

Each State is required to "specify appropriate water uses to be achieved and protected" (40 CFR 131.10). If an aquatic life use is formally adopted for a waterbody, that designation becomes a formal component of the water quality standards. Furthermore, nonattainment of the use, as determined with either biomonitoring or chemical-specific assessment methods, legally constitutes nonattainment of the standard. Therefore, the more refined the use designation, the more precise the biological criteria (i.e., the more detailed the description of desired biological attributes), and the more complete the chemical-specific criteria for aquatic life, the more objective the assessment of standards attainment/nonattainment.

Section 304(a)

Section 304(a) requires EPA to develop and publish criteria and other scientific information regarding a number of water-quality-related matters, including:

- Effects of pollutants on aquatic community components ("Plankton, fish, shellfish, wildlife, plant life...") and community attributes ("diversity, productivity, and stability...");

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

- Appropriate "methods for establishing and measuring water quality criteria for toxic pollutants on other bases than pollutant-by-pollutant criteria, including biological monitoring and assessment methods."

This section of the Act has been historically cited as the basis for publishing national guidance on chemical-specific criteria for aquatic life, but is equally applicable to the development and use of biological monitoring and assessment methods and biological criteria.

State/EPA Roles in Policy Implementation

State Implementation

Because there are important qualitative differences among aquatic ecosystems (streams, rivers, lakes, wetlands, estuaries, coastal and marine waters), and there is significant geographical variation even among systems of a given type, no single set of assessment methods or numeric biological criteria is fully applicable nationwide. Therefore, States must take the primary responsibility for adopting their own standard biosurvey methods, integrating them with other techniques at the program level, and applying them in appropriate combinations on a case-by-case basis. Similarly, States should develop their own biological criteria and implement them appropriately in their water quality standards.

EPA Guidance and Technical Support

EPA will provide the States with national guidance on performing technically sound biosurveys, and developing and integrating biological criteria into a comprehensive water quality program. EPA will also supply guidance to the States on how to apply ecoregional concepts to reference site selection. In addition, EPA Regional Administrators will ensure that each Region has the capability to conduct fully integrated assessments and to provide technical assistance to the States.

Workgroup Members

Attachment B

NAME	LAST	OFFICE	ADDRESS	CITY
Rick	Albright	USEPA Reg. 10 WMD WD-139	1200 6th Avenue	Seattle, WA 98101
Ed	Bender	USEPA OWEP/ED (EN-338)	401 M. St. SW	Washington, DC 20460
David W.	Charters	Env. Resp. Team MS 101	Woodbridge Avenue	Edison, NJ 08837
Norm	Crisp	USEPA ESD Reg. 7	25 Funston Road	Kansas City, KS 66101
Philip	Crocker	USEPA Reg. 6 (6W-QT) WMD	1445 Ross Avenue	Dallas, TX 75202-2733
Wayne	Davis	USEPA ESD Reg. 5 (5-SMQA)	536 S. Clark St.	Chicago, IL 60605
Steve	Dressing	USEPA OWRS/AWPD WH-553	401 M. St. SW	Washington, DC 20460
Roland	Dubois	USEPA OGC (LE-132W)	401 M. St. SW	Washington, DC 20460
Bruce	Elliott	USEPA Reg. 6 WMD	1445 Ross Avenue	Dallas, TX 75202
Steve	Glomb	USEPA OWEP (EN-336)	401 M. St. SW	Washington, DC 20460
Warren	Harper	USDA Forest Serv. OPPE	P.O. Box 96090 Rm. 121	Washington, DC 20090-6090
Margarete	Heber	USEPA OWEP/PD (EN-336)	401 M. St. SW	Washington, DC 20460
Del	Hicks	USEPA Reg. 4 ESD	College Station Road	Athens, GA 30613-7799
John	Houlihan	USEPA WMD Reg. 7	726 Minnesota Avenue	Kansas City, KS 66101
Bob	Hughes	USEPA ERL-Corvallis	200 SW 35th Street	Corvallis, OR 97333
Jim	Kurtenbach	USEPA ESD/Reg. 2	Bldg. 209 Woodbridge Ave	Edison, NJ 07828
Jacques	Landy	USEPA WMD Reg. 9 W-3-2	215 Fremont Street	San Francisco, CA 94105
Jim	Lazorchak	USEPA EMSL-Cinn ABBranch	3411 Church Street	Newtown, OH 45244
Peter	Mack	NY State DEC Div. of Water	50 Wolf Road	Albany, NY 12233
Suzanne	Marcy	USEPA OWRS/CSD (WH-585)	401 M. St. SW	Washington, DC 20460
John	Maxted	USEPA OWP (A-104)	401 M. St. SW	Washington, DC 20460
Pete	Nolan	USEPA Reg. I ESD	60 Westview Street	Lexington, MA 02173
Bill	Painter	USEPA OPA/ERED PM-221	401 M. St. SW	Washington, DC 20460
Ronald	Preston	USEPA Reg. 3 ESD	303 Meth. Bldg. 11th & Chapl.	Wheeling, WV 26003
Jackie	Romney	USEPA OWEP/PD (EN-336)	401 M. St. SW	Washington, DC 20460
Mark	Sprenger	USEPA Reg. 2 ESD	Woodbridge Avenue	Edison, NJ 08837
Steve	Tedder	NC Dept. of Envir. Mgmt.	P.O. Box 27687	Raleigh, NC 27611
Nelson A.	Thomas	USEPA ERL-Duluth	6201 Condgon Blvd.	Duluth, MN 55804
Randall	Waite	USEPA Reg. 3 (3 WM 12)	841 Chestnut Blvd.	Philadelphia, PA 19107
Bill	Wuerthele	USEPA Reg. 8 WMD (8WM-SP)	999 18th Street, #500	Denver, CO 80202

Attachment C

Relevant Guidance

Existing documents

- Chemical-specific evaluations

  Guidance for Deriving National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses (45 FR 79342, November 28, 1990, as amended at 50 FR 30784, July 29, 1985)

  Quality Criteria for Water 1986 (EPA 440/5-86-001, May 1, 1987)

- Toxicity testing

  Short-Term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Freshwater Organisms, Second Edition (EPA/600-4-89-001), March 1989)

  Short-Term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Waters to Marine and Estuarine Organisms (EPA/600-4-87/028, May 1988)

  Methods for Measuring Acute Toxicity of Effluents to Freshwater and Marine Organisms (EPA/600-4-85-013, March 1985)

- Biosurveys and integrated assessments

  Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses: Volumes I-III (Office of Water Regulations and Standards, November 1983-1984)

  Technical Support Document for Water Quality-based Toxics Control (EPA/505/2-90/001, March 1991)

  Rapid Bioassessment Protocols for Streams and Rivers: Benthic Macro-invertebrates and Fish (EPA/444-4-89-001, May 1989)

  Hughes, Robert M. and David P. Larsen. 1988. Ecoregions: An Approach to Surface Water Protection. Journal of the Water Pollution Control Federation 60, No. 4: 486-93.

  Omerik, J.M. 1987. Ecoregions of the Coterminous United States. Annals of the Association of American Geographers 77, No. 1: 118-25.

Regionalization as a Tool for Managing Environmental Resources (EPA/600-3-89-060, July 1989)

EPA Biological Criteria - National Program Guidance for Surface Waters (EPA/440-5-90-004, April 1990)

<u>Documents being developed</u>

Technical Guidance on the Development of Biological Criteria

State Development of Biological Criteria (case studies of State implementation)

Monitoring Program Guidance

Sediment Classification Methods Compendium

Macroinvertebrate Field and Laboratory Manual for Evaluating the Biological Integrity of Surface Waters

Fish Field and Laboratory Manual for Determining the Biological Integrity of Surface Waters

APPENDIX S

Reserved

APPENDIX S

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

APPENDIX T

Use Attainability Analysis Case Studies

APPENDIX T

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION

CASE STUDIES

Introduction

The Water Body Survey and Assessment Guidance for Conducting Use Attainability Analyses provides guidance on the factors that may be examined to determine if an aquatic life protection use is attainable in a given stream or river system. The guidance proposed that States perform physical, chemical and biological evaluations in order to determine the existing and potential uses of a water body. The analyses suggested within this guidance represent the type of analyses EPA believes are sufficient for States to justify changes in uses designated in a water quality standard and to show in Advanced Treatment Project Justifications that the uses are attainable. States are also encouraged to use alternative analyses as long as they are scientifically and technically supportable. Furthermore, the guidance also encourages the use of existing data to perform the physical, chemical and biological evaluations and whenever possible States should consider grouping water bodies having similar physical and chemical characteristics to treat several water bodies or segments as a single unit.

Using the framework provided by this guidance, studies were conducted to (1) test the applicability of the guidance, (2) familiarize State and Regional personnel with the procedures and (3) identify situations where additional guidance is needed. The results of these case studies, which are summarized in this Handbook, pointed out the following:

(1) The Water Body Surveys and Assessment guidance can be applied and provides a good framework for conducting use attainability analyses;
(2) The guidance provides sufficient flexibility to the States in conducting such analyses; and,
(3) The case studies show that EPA and States can cooperatively agree to the data and analyses needed to evaluate the existing and potential uses.

Upon completion of the case studies, several States requested that EPA provide additional technical guidance on the techniques mentioned in the guidance document. In order to fulfill these requests, EPA has developed a technical support manual on conducting attainability analyses and is continuing research to develop new cost effective tools for conducting such analyses. EPA is striving to develop a partnership with States to improve the scientific and technical bases of the water quality standards decision-making process and will continue to provide technical assistance.

The summaries of the case studies provided in this Handbook illustrate the different methods States used in determining the existing and potential uses. As can be seen, the specific analyses used were dictated by (1) the characteristics of the site, (2) the

States capabilities and technical expertise using certain methods and (3) the availability of data. EPA is providing these summaries to show how use attainability analyses can be conducted. States will find these case studies informative on the technical aspects of use attainability analyses and will provide them with alternate views on how such analyses may be conducted.

WATER BODY SURVEY AND ASSESSMENT
Assabet River, Massachusetts

I. INTRODUCTION

A. Site Description

The drainage basin of the Assabet River comprises 175 square miles located in twenty towns in East-Central Massachusetts. The Assabet River begins as the outflow from a small wildlife preservation impoundment in the Town of Westborough and flows northeast through the urban centers of Northborough, Hudson, Maynard and Concord to its confluence with the Sudbury River, forming the Concord River. Between these urbanized centers, the river is bordered by stretches of rural and undeveloped land. Similarly, the vast majority of the drainage basin is characterized by rural development. Figure 1 presents a schematic diagram of the drainage basin.

The Assabet River provides the opportunity to study a repeating sequence of water quality degradation and recovery. One industrial and six domestic wastewater treatment plants (WWTP) discharge their effluents into this 31-mile long river. All of the treatment plants presently provide secondary or advanced secondary treatment, although many of them are not performing to their design specifications. Most of the treatment plants are scheduled to be upgraded in the near future.

Interspersed among the WWTP discharges are six low dams, all but one of which were built at least a half century ago. All are "run-of-the-river" structures varying in height from three to eleven feet. The last dam built on the river was a flood control structure completed in 1980.

The headwaters of the Assabet River are formed by the discharge from a wildlife preservation impoundment, and are relatively "clean" except for low dissolved oxygen (DO) and high biochemical oxygen demand (BOD) during winter and summer. Water is discharged from the preserve through the foot of the dam that forms the impoundment, and therefore, tends to be low in DO. DO and BOD problems in the impoundment are attributed to winter ice cover and peak algal growth in summer. After the discharge of effluents from the Westborough and Shrewsbury municipal wastewater treatment plants, the river enters its first degradation/recovery cycle. The cycle is repeated as the river receives effluent from the four remaining domestic treatment plants. Water quality problems in the river are magnified when the effluents are discharged into the head of an impoundment. However, the flow of water over the dams also serves as a primary means of reaeration in the river, and thus, the dams also become a major factor in the recovery segment of the cycle. Water quality surveys performed in 1979 showed violations of the fecal coliform, phosphorus, and dissolved oxygen criteria throughout the river.

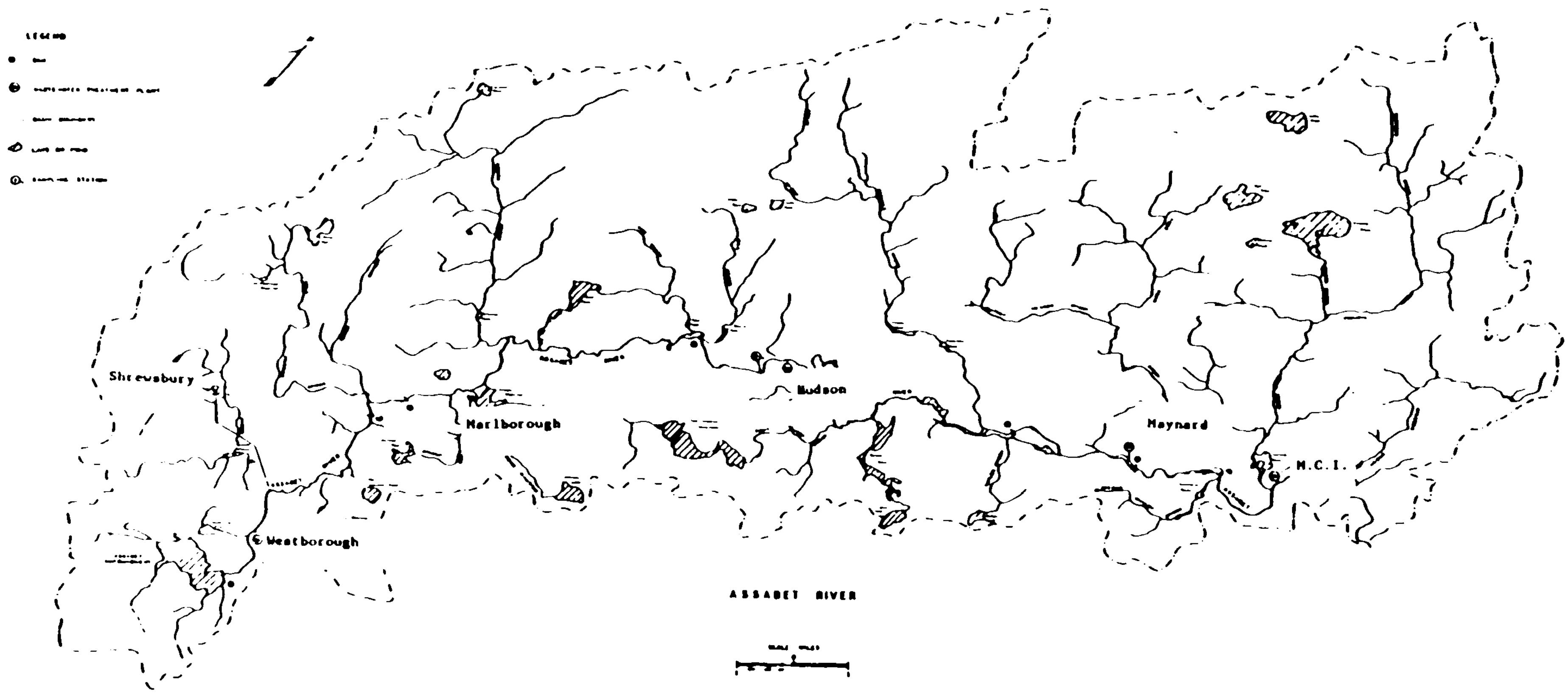

Figure 1 ASSABET RIVER DRAINAGE SYSTEM

At present, the entire length of the Assabet River is classified B, which is designated for the protection and propagation of fish, other aquatic life and wildlife, and for primary and secondary recreation. Two different uses have been designated for the Assabet River--from river mile 31.8 to 12.4 the designated use is "aquatic life" and from river mile 12.4 to the confluence with the Sudbury River the designated use is a "warm water fishery". The difference in these designated uses is that maintenance of a warm water fishery has a maximum temperature criterion of 83 degrees F, and a minimum DO of 5 mg/l. There are no temperature or DO criteria associated with the aquatic life use. These designations seem contrary to the existing data, which document violations of both criteria in the lower reaches of the river where warm water fishery is the designated use.

B. Problem Definition

The Assabet River was managed as a put and take trout fishery prior to the early 1970s when the practice was stopped on advisement of the MDWPC because of poor water quality conditions in the river. While the majority of the water quality problems are attributable to the wastewater treatment plant discharges, the naturally low velocities in the river, compounded by its impoundment in several places, led to the examination of both factors as contributors to the impairment of aquatic life uses. This combination of irreversible physical factors and wastewater treatment plant-induced water quality problems led to the selection of the Assabet River for this water body survey.

C. Approach to Use Attainability Analysis

Assessment of the Assabet River is based on the previously mentioned site visits and discussions among representatives of the Massachusetts Division of Water Pollution Control (MDWPC); the U.S. Environmental Protection Agency (EPA); and the Massachusetts Fish and Wildlife Division. This assessment is also based in part upon findings reported in the field and laboratory analyses on the Assabet River in early June, 1979, and again in early August, 1979. These surveys are part of the on-going MDWPC monitoring program, which included similar water quality assessments of the Assabet in 1969 and 1974. The water quality monitoring includes extensive information on the chemical characteristics of the Assabet River.

Analyses Conducted

A review of physical, chemical and biological information was conducted to determine which aquatic life use designations would be appropriate.

A. Physical Factors

The low flow condition of the river during the summer months may have an impact on the ability of certain fish species to survive. Various percentages of average annual flow (AAF) have been used to describe stream regimens for critical fisheries flow. As reported in

Cortell (1977), studies conducted by Tennant indicate that 10%, 30%, and 60% of AAF describe the range of fisheries flows from absolute minimum (10% AAF) to optimum (60% of AAF). The average annual flow of the Assabet River, as calculated from 39 years of record at the USGS gauge at river mile 7.7, is 183 cfs. Flow measurements taken at the USGS gauge on four consecutive days in early August, 1979, were 43, 34, 27, and 33 cfs. These flows average about 19 percent of the AAF indicating that some impairment of the protection of fish species may occur due to low flow in the river. The 7-day 10-year low flow for this reach of the river is approximately 18 to 20 cfs.

The outstanding physical features of the Assabet River are the dams, which have a significant influence on the aquatic life of the river. Most fish are incapable of migrating upstream of the dams, thus limiting their ability to find suitable (sufficient) habitats when critical water quality conditions occur. The low flow conditions downstream of the dams during dry periods also result in high water temperatures, further limiting fish survival in the river.

B. Biological Factors

As with data on the physical parameters for the Assabet River, biological data are sparse. The last fish survey of the Assabet River was conducted by the Massachusetts Fish and Wildlife Division in 1952. Yellow perch, bluegills, pickerel, sunfish, and bass were all observed. The Assabet River was sampled by the MDWPC for macroinvertebrates at five locations in June, 1979, as part of an intensive water quality survey.

The data were reviewed and analyses performed to determine whether conditions preclude macroinvertebrate habitats. The results were inconclusive.

C. Chemical Factors

Of all the chemical constituents measured in the June and August, 1979, water quality surveys, dissolved oxygen, ammonia nitrogen, and temperature have the greatest potential to limit the survival of aquatic life. Ammonia toxicity was investigated using the criteria outlined in Water Quality Criteria 1972. The results of this analysis indicate that the concentration of un-ionized ammonia would need to be increased approximately three times before acute mortality in the species of fish listed would occur. Therefore, ammonia is not a problem.

Temperatures in the lower reaches of the Assabet frequently exceed the maximum temperature criteria (83 degrees F) for maintenance of a warm water fishery. However, temperature readings were taken in early and late afternoon and are believed to be surface water measurements. They are short-term localized observations and should not preclude the maintenance of a warm water fishery in those reaches. Dissolved oxygen concentrations above Maynard are unsuitable for supporting cold or warm water fisheries, but are sufficient to support a fishery below this point.

The impoundments may exhibit water quality problems in the form of high surface temperatures and low bottom DO. Surface temperatures have been found to be similar to those in the remainder of the river. The only depth sample was at 13 feet in the wildlife impoundment, where the temperature was 63 degrees F, while 83 degrees F at the surface. While such bottom temperatures are likely to be sufficient to support a cold water fishery, it is likely that the DO at the bottom of the impoundments will be near zero due to benthic demands and lack of surface aeration, which would preclude the survival of any fish.

Findings

The data, observations, and analyses as presented herein lead to the conclusion that there are four possible uses for the Assabet: aquatic life, warm water fishery, cold water fishery, and seasonal cold water fishery. The seasonal fishery would be managed by stocking the river during the spring.

These uses were analyzed under three water quality conditions: existing, existing without the wastewater discharges, and inclusion of the wastewater effluent discharges with treatment at the levels stipulated in the 1981 Suasco Basin Water Quality Management Plan. The no discharge condition is included as a baseline that represents the quality under "natural" conditions.

A. Existing Uses

A limited number of warm water fish species predominate in the Assabet River under existing conditions. The species should not be different from those observed during the 1952 survey. The combination of numerous low-level dams and wastewater treatment plants with low flow conditions in the summer results in dissolved oxygen concentrations and temperatures which place severe stress on the metabolism of the fish.

The observed temperatures are most conducive to support the growth of coarse fish, including pike, perch, walleye, smallmouth and largemouth bass, sauger, bluegill and crappie.

The minimum observed DO concentrations are unacceptable for the protection of any fish. Water Quality Criteria establishes the values 6.8, 5.6, and 4.2 mg/l of DO for high, moderate, and low levels of protection of fish for rivers with the temperature characteristics of the Assabet. The Draft National Criteria for Dissolved Oxygen in Freshwater establishes criteria as 3.0 mg/l for survival, 4.0 mg/l for moderate production impairment, 5.0 mg/l for slight impairment, and 6.0 for no production impairment. The upper reaches will not even support a warm water fishery at the survival level, except in the uppermost reach. On the other hand, the lower reaches can support a warm water fishery under existing conditions.

B. Potential Uses

The potential aquatic life uses of the Assabet River would be restricted by temperature and low flow, and by physical barriers that would exist even if water quality (measured in terms of DO and bacteria) is significantly improved. Despite an overall improvement in treated effluent quality, the river would be suitable for aquatic life, as it is currently, and would continue to be too warm to support a cold water fishery in the summertime. The possibility of maintaining the cold water species in tributaries during the summer was investigated, but there are no data on which to draw conclusions. Water quality observations in the only tributary indicate temperatures similar to those in the mainstem. Therefore, the maintenance of a cold water fishery in the Assabet is considered unfeasible.

The attainable uses in the river without discharges or at planned levels of treatment are warm water fishery and seasonal cold water fishery. These uses are both attainable throughout the basin, but may be impaired in Reach 1, as the water naturally entering Reach 1 from the wildlife preservation impoundment is low in DO. The seasonal cold water fishery is attainable because the discharge limits are established to maintain a DO of 5 mg/l under 7Q10 conditions. If the DO is 5 mg/l under summer low flow conditions, it will certainly be 6 mg/l or greater during the colder, higher flow spring stocking period, and a seasonal cold water fishery would be attainable.

According to the Fish and Wildlife Division, the impoundments of the Assabet River have the potential to be a valuable warm water fishery. The reaches of the river that have a non-vegetated gravel bottom also have a high potential to support a significant fishery because these habitats allow the benthic invertebrates that comprise the food supply for the fish to flourish. It was further suggested that if the dissolved oxygen concentration could be maintained above 5 mg/l, the river could again be stocked as a put and take trout fishery in the spring.

Summary and Conclusions

The low flow conditions of the Assabet River have been exacerbated by the low dams which span its course. In the summer months, the flow in the river is slowed as the river passes through its impoundments and flow below the dams is often reduced to a relative trickle. When flow is reduced, temperatures in the shallow river (easily walkable in many places) can exceed the maximum temperature criterion for protection and propagation of a warm water fishery. Additionally, the dams limit the mobility of fish. At present, most of the river reaches also undergo extensive degradation due to the discharge of wastewater treatment plant effluent which is manifest in low dissolved oxygen concentrations. All of these factors impair the aquatic life potential of the Assabet River.

Three use levels corresponding with three alternative actions related to the wastewater discharges are possible in the Assabet. The no action alternative would result in very low dissolved oxygen concentrations in many reaches which are appropriate only for the use designation of aquatic life and warm water fishery. In this scenario, fish would only survive in the lowest river reaches, and aquatic life would be limited to sludge worms and similar invertebrates in the upper reaches. The remaining two alternatives are related to upgrading treatment plants in the basin. If the discharges are improved sufficiently to raise the instream DO to 5 mg/l throughout, as stipulated in the 1981 Water Quality Management Plan, it will be suitable as a warm water or seasonal cold water fishery. Should the discharge be elimInted altogether, the same uses would be attainable.

The treatment plant discharges inhibit the protection and propagation of aquatic life. Most of the treatment plants are scheduled to be upgraded in the near future, which would relieve the existing dissolved oxygen problems. Even if the river is returned to relatively pristine conditions, the type of fish that would be able to propagate there would not change, due to the existing physical conditions. However, the extent of their distribution, their abundance, and the health of the biota would be likely to increase.

The present use designations of the Assabet River are sufficient to characterize the aquatic life use it is capable of supporting, while physical barriers prevent the year-round attainment of a "higher" aquatic life use. The potential aquatic life uses could include extension of the warm water and seasonal cold water fishery classifications to the entire length of the river, should the planned improvements to the wastewater treatment plants be implemented.

WATER BODY SURVEY AND ASSESSMENT
Blackwater River
Franklin, Virginia

I. INTRODUCTION

A. Site Description

The area of the Blackwater River which was chosen for this study extends from Joyner's Bridge (Southampton County, Route 611) to Cobb's Wharf near its confluence with the Nottoway River (Table 1 and Figure 1). In addition, data from the USGS gaging station near Burdette (river mile 24.57) provided information on some physical characteristics of the system.

TABLE 1

Sampling Locations for Blackwater River Use Attainability Survey

Station No.	Location	River Mile
1	Vicinity Joyner's Bridge, Route 611	20.90
2	Below Franklin Sewage Treatment Plant Discharge	13.77
3	Vicinity Cobb's Wharf, Route 687	2.59

The mean annual rainfall is 48 inches, much of which occurs in the summer in the form of thunderstorms. The SCS has concluded that approximately 41,000 tons of soil are transported to streams in the watershed due to rainfall induced erosion. Seventy (70) percent of this originates from croplands, causing a potential pollution problem from pesticides and from fertilizer based nutrients. In addition, 114,000 pounds of animal waste are produced annually, constituting the only other major source of non-point pollution.

There are two primary point source discharges on the Blackwater River. The Franklin Sewage Treatment Plant at Station 2 discharges an average of 1.9 mgd of municipal effluent. The discharge volume exceeds NPDES permit levels due to inflow and infiltration problems. The plant has applied for a federal grant to upgrade treatment. The second discharge is from Union Camp Corporation, an integrated kraft mill that produces bleached paper and bleached board products. The primary by-products are crude tall oil and crude sulfate turpentine. Union Camp operates at 36.6 mgd but retains its treated waste in lagoons until the winter months when it is discharged. The

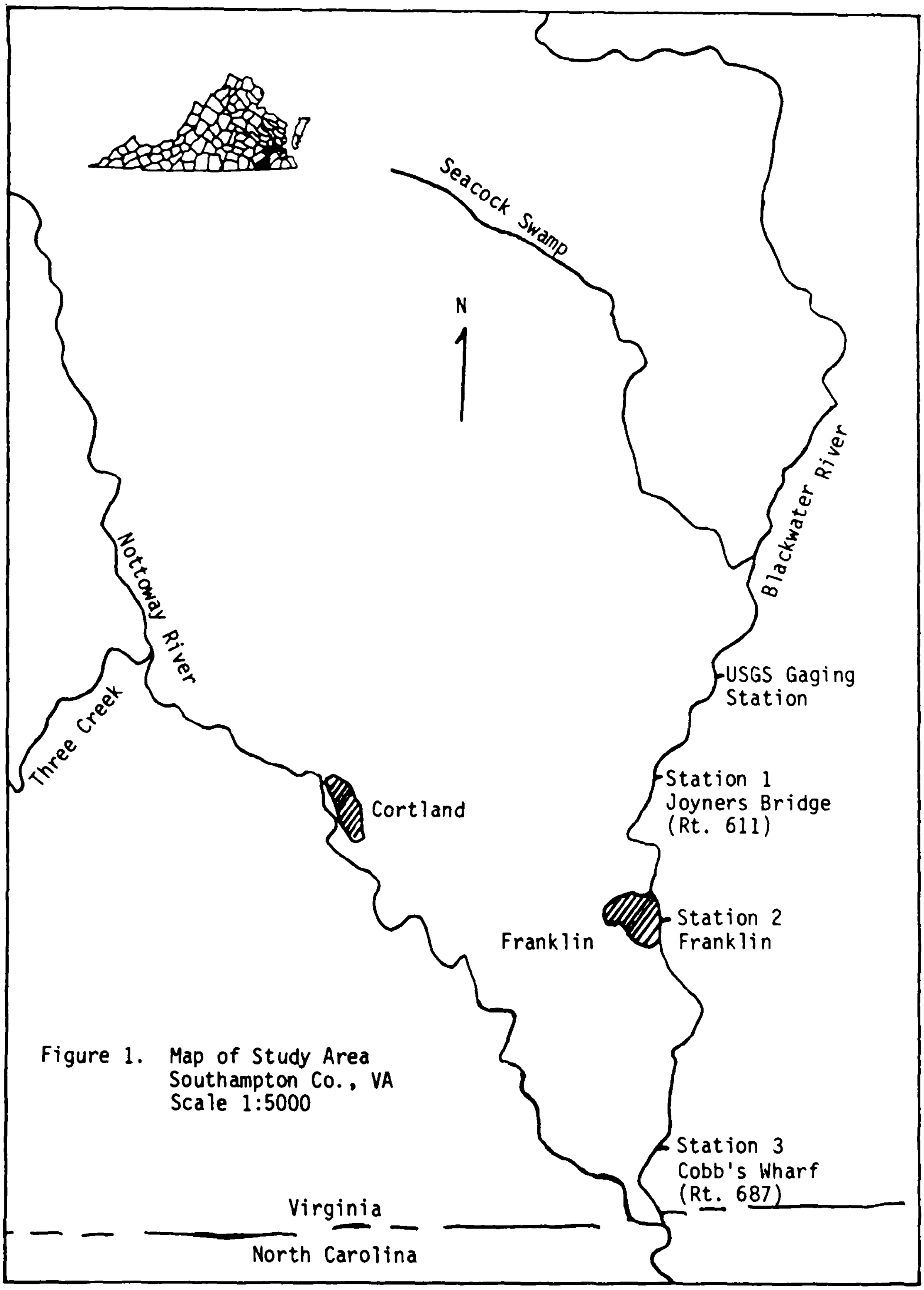

Figure 1. Map of Study Area
Southampton Co., VA
Scale 1:5000

Union Camp discharge point is downstream from Station 3 just above the North Carolina State line at river mile 0.70.

The topography surrounding the Blackwater River is essentially flat and the riparian zone is primarily hardwood wetlands. There is a good surface water supply from several swamps. At the USGS gaging station near Burdette, Virginia, the discharge for calendar year 1980 averaged 430 cfs.

The Blackwater River from Joyner's Bridge (Station 1) to Franklin is classified by the State Water Control Board (SWCB) as a Class III free flowing stream. This classification requires a minimum dissolved oxygen concentration of 4.0 mg/l and a daily average of 5.0 mg/l. Other applicable standards are maintenance of pH from 6.0 to 8.5 and a maximum temperature of 32°C. The riparian zone is heavily wooded wetlands with numerous channel obstructions. Near Franklin the canopy begins to open and there is an increasing presence of lily pads and other macrophytes. The water is dark, as is characteristic of tannic acid water found in swamplands.

Below Franklin the Blackwater River is dredged and channelized to permit barge traffic to reach Union Camp. The channel is approximately 40m wide and from 5m to 8m in depth. This reach of stream is classified by the SWCB as a Class II estuarine system requiring the same dissolved oxygen and pH limitation as in Class III but without a temperature requirement.

B. Problem Definition

The study area on the Blackwater River includes a Class III free-flowing stream and a Class II estuarine river. Part of the Class III section is a freshwater cypress swamp. The water is turbid, nutrient enriched and slightly acidic due to tannins.

In response to the EPA request for Virginia's involvement in the pilot Use Attainability studies, the State Water Control Board chose to examine the Blackwater River in the vicinity of Franklin, Virginia. There were several reasons for this choice. First, the major stress to the system is low dissolved oxygen (DO) concentrations which occur from May through November. Surveys conducted by SWCB staff, and officials from Union Camp in Franklin, found that during certain periods "natural" background concentrations of dissolved oxygen fell below the water quality standard of 4.0 mg/l. This has raised questions as to whether the current standard is appropriate. Virginia's water quality standards contain a swamp water designation which recognizes that DO and pH may be substantially different in some swamp waters and provides for specific standards to be set on a case by case basis. However, no site specific standards have been developed in Virginia to date. One of the goals of this project was to gather information which could lead to possible development of a site specific standard for the Blackwater River. Second, the Franklin STP has applied for a federal grant to provide for improved BOD removals from its effluent.

C. Approach to Use Attainability

On 20 April, 1982, staff of the SWCB met with several EPA officials and their consultant. After visiting the study area on the Blackwater River and reviewing the available information, it was determined that further data should be collected, primarily a description of the aquatic community. The SWCB staff has scheduled four quarterly surveys from June 1982, through March 1983, to collect physical, chemical, and biological information. Interim results are reported herein to summarize data from the first collection. Final conclusions will not be drawn until the data has been compiled for all four quarters.

II. ANALYSES CONDUCTED

A. Physical Analysis

Data on the physical characteristics of the Blackwater River were derived primarily from existing information and from general observations. The entire reach of the Blackwater River from Joyner's Bridge to Cobb's Wharf was traveled by boat to observe channel and riparian characteristics. A sediment sample was collected at each station for partical size analysis.

B. Chemical Analysis

Water samples were collected at Stations 1-3 for analysis of pH, alkalinity, solids, hardness, nutrients, five-day BOD, chemical oxygen demand, total organic carbon, phenols, pesticides, and heavy metals. In addition, previous data on dissolved oxygen concentrations collected by the SWCB and Union Camp were used to examine oxygen profiles in the river. The USGS Water Resources Data for Virginia (1981) provided some chemical data for the Blackwater River near Burdette.

C. Biological Analysis

Periphyton sampling for chlorophyll-a, biomass, and autotrophic index determination was conducted using floating plexiglass samplers anchored by a cement weight. The samplers were placed in the field in triplicate and remained in the river for 14 days. They were located in run areas in the stream. At the end of this two-week period, the samplers were retrieved and the slides removed for biomass determinations and chlorophyll analysis.

Both a cursory and a quantitative survey of macroinvertebrates were conducted at each station. The purpose of the cursory study was to rapidly identify the general water quality of each station by surveying the presence of aquatic insects, molluscs, crustaceans and worms and classifying them according to their pollution tolerance. A record was kept of all organisms found and these were classified to the family level as dominant, abundant, common, few or present. The cursory survey was completed with a qualitative evaluation of the density and diversity of aquatic organisms.

General knowledge of the pollution tolerance of various genera was used to classify the water quality at each station. The benthic macroinvertebrate samples were collected with Hester-Dendy multiplate artificial substrates. The substrates were attached to metal fence posts and held vertically at least 15 cm above the stream bottom. The substrates were left in place for six weeks to allow for colonization by macroinvertebrate organisms. In the laboratory the organisms were identified to the generic level whenever possible. Counts were made of the number of taxa identified and the number of individuals within each taxon.

Fish populations were surveyed at each station by electrofishing. Each station was shocked for 1,000 seconds: 800 seconds at the shoreline and 200 seconds at midstream. Fish collected were identified to species and the total length of each fish was recorded. In addition, general observations were made about the health status of the fish by observing lesions, hemorrhaging, and the presence of external parasites.

Diversity of species was calculated using the Shannon-Weaver index. Additionally, the fish communities were evaluated using an index proposed by Karr (1981) which classifies biotic integrity based on 12 parameters of the fish community.

III. FINDINGS

There are few physical factors which limit aquatic life uses. The habitat is characteristic of a hardwood wetland with few alterations. The major alteration is dredging and channelization below Franklin which eliminates much of the macrophyte community and the habitat it provides for other organisms. The substrate at each station was composed mostly of sand with a high moisture content. This is characteristic of a swamp but is not ideal habitat for colonization by periphyton and macroinvertebrates.

DO concentrations are typically below the Virginia water quality standards during the months of May through November. This is true upstream as well as downstream from the Franklin STP and appears to occur even without the impact of BOD loadings from Franklin. This phenomenon may be typical of enriched freshwater wetlands. However, during the winter months, DO concentrations may exceed 10 mg/l. Another survey conducted by SWCB showed that there were only small changes in DO concentration with depth.

Representatives from 17 families of macroinvertebrates were observed during a cursory investigation. These included mayflies, scuds, midges, operculate and non-operculate snails, crayfish, flatworms, and a freshwater sponge. The majority of these organisms were facultative at Stations 1 and 2. However, there were a few pollution sensitive forms at Station 1, and Station 3 was dominated by pollution sensitive varieties.

Twelve (12) species from seven families of fish were observed during the June 1982 study. Several top predators were present including the bowfin, chain pickerel, largemouth bass and longnose gar. Other fish collected were the American eel, shiners, pirate perch, yellow perch, and five species of sunfish. None of the species are especially pollution sensitive. Results of the fish population survey are presented in Table 2.

TABLE 2

Results of Fish Population Survey in Blackwater River, 9 June 1982

Station	Number Collected	No. of Species	Diversity d	Proportion of Omnivores	Proportion of Carnivores
1. Joyner's Bridge	19	7	2.30	.000	.157
2. Franklin STP	51	6	2.35	.000	.098
3. Cobb's Wharf	44	6	2.35	.000	.114

Based on the EPA 304(a) criteria, low seasonal DO concentrations measured in the river should present a significant stress to the biotic community. Large fish tend to be less resistant to low DO yet large species such as the largemouth bass, American eel and some sunfishes were present in an apparently healthy condition. The explanation for this is unclear. The low dissolved oxygen concentrations are near the physiological limit for many species. Fish may be able to acclimate to low DO to a limited extent if the change in oxygen concentration occurs gradually. The fact that fish are present in a healthy condition suggests that there is a lack of other significant stressors in the system which might interact with low DO stress. It is worth noting that spawning probably occurs in most species before the summer months when dissolved oxygen concentration become critically low.

The autotrophic index determinations show the Joyner's Bridge and Franklin STP stations as having relatively healthy periphyton communities. In each case over 80 percent of the periphytic community was autotrophic in nature. Based on the autotrophic index, both of these stations were in better biological health than the most downstream station, Cobb's Wharf. At Cobb's Wharf the autotrophic index characterized an autotrophic community which was experiencing a slight decline in biological integrity (74 percent autotrophic as compared to greater than 80 percent upstream).

Chemical analyses conducted on water from the Blackwater River did not reveal any alarming concentration of toxicants when compared to EPA Water Quality Criteria Documents, although the zinc concentration at Station 1 was slightly above the 24-hour average recommended by EPA. One sample collected by the USGS had a zinc concentration which was twice this number. The source of this zinc is unknown. Any impact which exists from this problem should be sublethal, affecting growth and reproduction of primarily the most sensitive species. The actual impact of zinc concentrations at Joyner's Bridge is unknown.

Analyses of the periphyton data as well as the water chemistry data indicate that the Blackwater River is nutrient enriched. Some of this nutrient load comes from inadequately protected crop lands and from domestic animal wastes. The Franklin STP also contributes to higher nutrient concentrations. Additionally, an SWCB report estimated that between river mile 20.0 and 6.0, 1,600 lb per day of non-point source carbonaceous BOD_u (ultimate) are added to the river. Consequently, these point and non-point sources appear to be contributing to both organic enrichment and lower dissolved oxygen concentrations.

IV. SUMMARY AND CONCLUSIONS

The Blackwater River from river mile 2.59 to 20.90 has been characterized as a nutrient enriched coastal river much of which is bordered by hardwood wetlands. Periphytic, macroinvertebrate, and fish communities are healthy with fair to good abundance and diversity. The major limitation to aquatic life appears to be low DO concentrations which are enhanced by point and non-point sources of nutrients and BOD. A secondary limitation may be elevated zinc concentrations at Joyner's Bridge.

The primary difficulty in assessing the attainability of aquatic life uses is locating a suitable reference reach to serve as an example of an unaffected aquatic community. Originally, Joyner's Bridge (Station 1) was selected for this purpose, but few major differences occur between populations at all three stations. However, the widespread non-point pollution in Southeastern Virginia makes the location of an undisturbed reference reach impossible. The only alternative, then, is to make the best possible judgment as to what organisms might reasonably be expected to inhabit the Blackwater.

In reference to the Blackwater River, it is probable that most fish species are present that should reasonably be expected to inhabit the river, although possibly in lower numbers. (No attempt has yet been made to assess this with regard to algal and invertebrate communities.) However, based on the 304(a) criteria, the low DO concentrations represent a significant stress of the ecosystem and the introduction of additional stressors could be destructive. It is also probable that higher oxygen concentrations during winter months play a major role in reducing the impact of this stress. Removal of point and non-point source inputs may alleviate some problems. However, DO concentrations may still remain low. The increased effect of oxygen concentrations should be an increase in fish abundance and increased size of individuals. Diversity would probably be unaffected. Nevertheless, no attempt has been made to estimate the magnitude of these changes.

Cairns (1977) has suggested a method for estimating the potential of a body of water to recover from pollutional stress. Although this analysis is only semi-quantitative and subjective, it suggests that the chances of rapid recovery following a disturbance in the Blackwater River are poor.

The absence of an undisturbed reference reach and the difficulty in quantifying changes in dissolved oxygen, population structure, and population abundance make a definite statement regarding attainability of aquatic life uses difficult. However, to summarize, several points stand out. First, the aquatic communities in the Blackwater River are generally healthy with fair to good abundance and distribution. Dissolved oxygen concentrations are low for about half of the year which causes a significant stress to aquatic organisms. Oxygen concentrations are higher during the reproductive periods of many fishes. Because of these stresses and the physical characteristics of the river, the system does not have much resiliency or capacity to withstand additional stress. Although a quantitative statement of changes in the aquatic community with the amelioration of DO stress has not been made, it is probable that additional stresses would degrade the present aquatic community.

The occurrence of low dissolved oxygen concentrations throughout much of the Blackwater is, in part, a "natural" phenomenon and could argue for a reduction in the DO standard. However, if this standard were reduced on a year round basis it is probable that the aquatic community would steadily degrade. This may result in a contravention of the General Standard of Virginia State Law which requires that all waters support the propagation and growth of all aquatic life which can reasonably be expected to inhabit these waters. Because of the lack of resiliency in the system, a year round standards change could irreversibly alter the aquatic community.

WATER BODY SURVEY AND ASSESSMENT
Cuckels Brook
Bridgewater Township, New Jersey

I. INTRODUCTION

A. Site Description

Cuckels Brook, a small tributary of the Raritan River, is located entirely within Bridgewater Township in Somerset County, New Jersey. It is a perennial stream approximately four miles long, having a watershed area of approximately three square miles. The entire brook is classified as FW-2 Nontrout in current New Jersey Department of Environmental Protection (NJDEP) Surface Water Quality Standards.

Decades ago, the downstream section of Cuckels Brook (below the Raritan Valley Line Railroad, Figure 1), was relocated into an artificial channel. This channelized section of Cuckels Brook consists of an upstream subsection approximately 2,000 feet in length and a downstream subsection approximately 6,000 feet in length, with the Somerset-Raritan Valley Sewerage Authority (SRVSA) municipal discharge being the point of demarcation between the two. The downstream channelized subsection (hereinafter referred to as "Lower Cuckels Brook") is used primarily to convey wastewater to the Raritan River from SRVSA and the American Cyanamid Company, which discharges approximately 200 feet downstream of SRVSA. At its confluence with the Raritan River, flow in Lower Cuckels Brook is conveyed into Calco Dam, a dispersion dam which distributes the flow across the Raritan River. Except for railroad and pipeline rights-of-way, all the land along Lower Cuckels Brook is owned by the American Cyanamid Company. Land use in the Cuckels Brook watershed above the SRVSA discharge is primarily suburban but includes major highways.

B. Problem Definition

Lower Cuckels Brook receives two of the major discharges in the Raritan River Basin. SRVSA is a municipal secondary wastewater treatment plant which had an average flow in 1982 of 8.8 mgd (design capacity = 10 mgd). The American Cyanamid wastewater discharge is a mixture of process water from organic chemical manufacturing, cooling water, storm water, and sanitary wastes. This mixed waste receives secondary treatment followed by activated carbon treatment. In 1982 American Cyanamid's average flow was 7.0 mgd (design capacity 20 mgd). These two discharges totally dominate the character of Lower Cuckels Brook.

Over 90 percent of the flow in Cuckels Brook is wastewater (except after heavy rainfall). The mean depth is estimated to be between 1 and 2 feet, and the channel bottom at observed locations is covered with deposits of

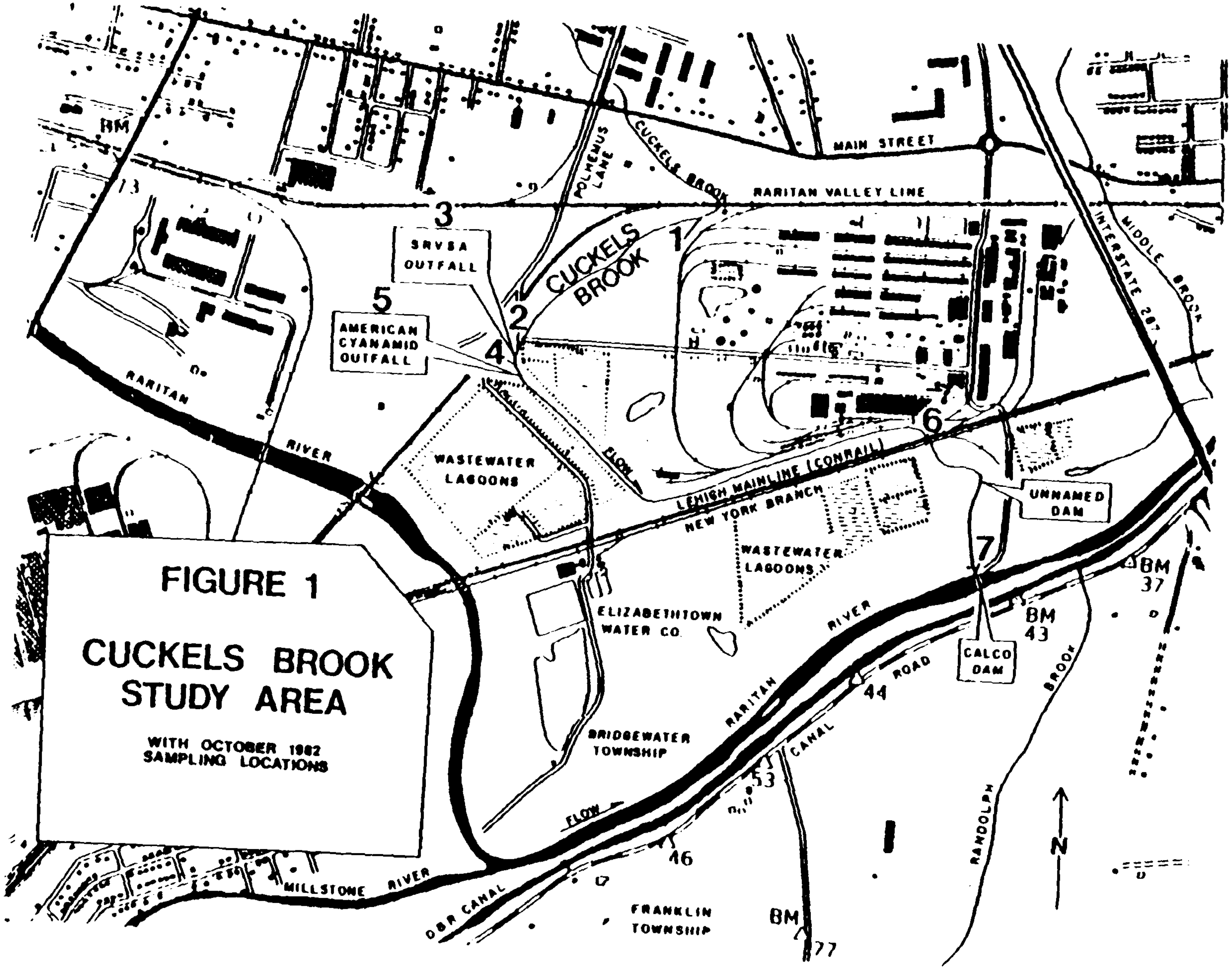
FIGURE 1
CUCKELS BROOK STUDY AREA
WITH OCTOBER 1982 SAMPLING LOCATIONS
MAIN STREET
RARITAN VALLEY LINE
CUCKELS BROOK
POLHEMUS LANE
INTERSTATE 287
MIDDLE BROOK
SRVSA OUTFALL
AMERICAN CYANAMID OUTFALL
WASTEWATER LAGOONS
LEHIGH MAINLINE (CONRAIL)
NEW YORK BRANCH
UNNAMED DAM
CALCO DAM
ELIZABETHTOWN WATER CO.
BRIDGEWATER TOWNSHIP
FRANKLIN TOWNSHIP
RARITAN RIVER
MILLSTONE RIVER
D&R CANAL
CANAL ROAD
RANDOLPH BROOK
FLOW
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black sludge, apparently derived from solids in the SRVSA and Cyanamid discharges (primarily the SRVSA discharge). In contrast, the channelized subsection of Cuckels Brook above the SRVSA discharge is often only inches deep with a bottom of bedrock, rubble, gravel and silt.

Cuckels Brook (including Lower Cuckels Brook) is classified as FW-2 Non-trout in the NJDEP Surface Water Quality Standards. The FW-2 classification provides for the following uses:

1. Potable water supply after such treatment as shall be required by law or regulation;

2. Maintenance, migration, and propagation of natural and established biota (not including trout);

3. Primary contact recreation;

4. Industrial and agricultural water supply; and

5. Any other reasonable uses.

The attainment of these uses is currently prevented by the strength and volume of wastewaters currently discharged to Cuckels Brook. The size of the stream also limits primary contact recreation and other water uses, and physical barriers currently prevent the migration of fish between Cuckels Brook and the Raritan River.

C. Approach to Use Attainability

In response to an inquiry from EPA, Criteria and Standards Division, the State of New Jersey offered to participate in a demonstration Water Body Survey and Assessment. The water body survey of Cuckels Brook was conducted by the New Jersey Department of Environmental Protection, Bureau of Systems Analysis and Wasteload Allocation; with assistance from the EPA Region II Edison Laboratory.

The assessment is based primarily on the results of a field sampling program designed and conducted jointly by NJDEP and EPA-Edison in October 1982. Additional sources of information include self-monitoring reports furnished by the dischargers, and earlier studies conducted by the NJDEP on Cuckels Brook and the Raritan River. Based on this assessment, NJDEP developed a report entitled "Lower Cuckels Brook Water Body Survey and Use Attainability Analysis, 1983."

II. ANALYSES CONDUCTED

A. Chemical Analysis

The major impact of the SRVSA discharge is attributed to un-ionized ammonia and TRC levels, whose concentrations at Station 4, 100 feet below the discharge point were 0.173 and 1.8 mg/l respectively, which are 3.5 and 600 times higher than the State criteria. The un-ionized ammonia concentration of the Cyanamid effluent was low, but stream concentrations at Stations 6 and 7 were relatively high (though below the State criterion of 0.05 mg/l).

The Cyanamid discharge contained 0.8 mg/l TRC. Concentrations at both Stations 6 and 7 were 0.3 mg/l TRC, lower than at Station 4 but still 100 times the State criterion of 0.003 mg/l. The other major impact of the Cyanamid effluent was on instream filterable residue levels. Concentrations at Stations 6 and 7 exceeded 1,100 mg/l, over three times the State criterion (133 percent of background).

The effluents apparently buffered the pH of Lower Cuckels Brook which was approximately pH 7 at Stations 4, 6 and 7, and the pH of the upstream reference stations was markedly alkaline. Dissolved oxygen concentrations decreased in the downstream direction despite low BOD5 concentrations both in the effluents and instream. This suggests an appreciable sediment oxygen demand in Lower Cuckels Brook. Dissolved oxygen levels were greater in the two effluents than in the stream at Stations 6 and 7. The dissolved oxygen concentration at Station 7 of 4.1 mg/l nearly violated the State criterion of 4.0 mg/l; this suggests the potential for unsatisfactory dissolved oxygen conditions during the summer.

The results of the water body survey are generally in good agreement with other available data sources. Recent self-monitoring data for both American Cyanamid and SRVSA agree well with the data collected in this survey. In particular they show consistantly high TRC concentrations in both effluents. High average dissolved solids (filterable residue) concentrations are reported for the Cyanamid effluent. Total ammonia levels as high as 33.5 mg/l NH_3 (27.6 mg/l N) were reported for the SRVSA effluent. The pH of the Cyanamid and SRVSA effluents is sometimes more alkaline than the water body survey values indicating that toxic un-ionized ammonia concentrations may sometimes be higher than measured during the water body survey.

B. Biological Analysis

Fish and macroinvertebrate surveys were conducted in the channelized subsection of Cuckels Brook above the SRVSA discharge. Only three fish species were found: the banded killifish, the creek chub and the blacknose dace. One hundred and eighty-six (186) out of the total 194 specimens collected were banded killifish. Killifish are very hardy and are common in both estuarine and freshwater systems. The largest fish found, a creek chub, was 146 mm long.

The results of the macroinvertebrate survey are discussed in detail in a separate report (NJDEP, 1982). Four replicate surber samples were collected at Stations 1 and 2 above the SRVSA discharge. Diversity indices indicate the presence of similar well-balanced communities at both stations. Species diversity and equitability were 3.9 and 0.7 respectively at Station 1, and 4.3 and 0.7 respectively at Station 2. Productivity at Stations 1 and 2 was low, with mean densities of 59 and 89 individuals per square foot, respectively. The majority of species found at both stations have organic pollution tolerance classifications of tolerant (dominant at Station 1) or facultative (dominant at Station 2).

Overall, the biological data indicate that the upstream channelized subsection of Cuckels Brook supports a limited fish community and a limited macroinvertebrate community of generally tolerant species. The water quality data indicates nothing that would limit the community. One possible limiting factor is that, as a result of channelization, the substrate consists of unconsolidated gravel and rubble on bedrock, which might easily be disturbed by high flow conditions.

Both the chemical data and visual observations at various locations suggest that virtually no aquatic life exists along Lower Cuckels Brook: not even algae were seen. The discharges have seriously degraded water quality. Unionized ammonia concentrations at Station 4 were close to acute lethal levels, while concentrations of TRC were above acute levels at Stations 4, 6 and 7 (EPA, 1976). The sludge deposits which apparently cover most of the bottom of lower Cuckels Brook could exert negative physical (i.e. smothering) and chemical (i.e. possible toxics) effects on any benthic organisms. No biological survey of the lower brook was made because of concern about potential hazards to sampling personnel. Supplemental sampling of the sediments is planned to ascertain levels of toxics accumulation.

As part of their self-monitoring requirements, American Cyanamid performs weekly 96-hour modified flow-through bioassays with fathead minnows using unchlorinated effluent. Of 63 bioassays conducted between 1 May, 1981 and 31 August, 1982, results from eight bioassays had 96-hour LC50 values at concentrations of effluent less than 100 percent (i.e. 26 percent, 58 percent, 77 percent, 83.5 percent, 88 percent, 92 percent, and 95.5 percent). These results suggest that the American Cyanamid effluent would not be extremely toxic if it were reasonably diluted by its receiving waters. Within Lower Cuckels Brook, however, the effluent receives only approximately 50 percent dilution and the potential exists for toxic effects on any aquatic life that may be present. These effects would be in addition to the toxicity anticipated from the TRC concentrations which result from the chlorination of the effluent.

III. FINDINGS

Practically none of the currently designated uses are now being achieved in Lower Cuckels Brook. The principal current use of Lower Cuckels Brook is the conveyance of treated wastewater and upstream runoff to the Raritan River. Judging from the indirect evidence of chemical data and visual observations, virtually no aquatic life is maintained or propagated in Lower Cuckels Brook. It has been well documented that fish avoid chlorinated waters (Cherry and Cairns, 1982; Fava and Tsai, 1976). Any aquatic life that does reside in Lower Cuckels Brook would be sparse and stressed. Migration of aquatic life through Lower Cuckels Brook would probably only occur during periods of high storm water flow when some flow occurs over the un-named dam (Figure 1) which is designed to direct the flow of Cuckels Brook toward Calco Dam. Calco Dam and its associated structures, including the un-named dam, normally prevent the migration of fish between Cuckels Brook and the Raritan River.

Lower Cuckels Brook currently does not support any primary or secondary contact recreation. No water is currently diverted from Lower Cuckels Brook for potable water supply, industrial or agricultural water supply, or any other purpose.

Because Lower Cuckels Brook receives large volumes of wastewater and because there is practically no dilution, water quality in Lower Cuckels Brook has been degraded to the quality of wastewater. Moreover, the bottom of Lower Cuckels Brook has been covered at observed locations with wastewater solids. As a result, Lower Cuckels Brook is currently unfit for aquatic life, recreation, and most other water uses. The technology-based effluent limits required by the Clean Water Act are not adequate to protect the currently designated water uses in Lower Cuckels Brook. SRVSA already provides secondary treatment (except for bypassed flows in wet weather), and American Cyanamid already provides advanced treatment with activated carbon. Because the Raritan River provides far more dilution than does Cuckels Brook, effluent limits which may be developed to protect the Raritan River would not be adequate to protect the currently designated water uses in Lower Cuckels Brook. The only practical way to restore water quality in Lower Cuckels Brook would be to remove the wastewater discharges. However, there are several factors that would limit the achievement of currently designated uses even if the wastewater discharges were completely separated from natural flow.

If it were assumed that the wastewater discharges and sludge were absent, and that the seepage of contaminated groundwater from the American Cyanamid property was insignificant or absent, then the following statements could be made about attainable uses in Lower Cuckels Brook:

> Aquatic Life - The restoration of aquatic life in Lower Cuckels Brook would be limited to some extent by the small size and lower flow of the stream, by channelization, and by contaminants in suburban and highway runoff from the upstream watershed. Lower Cuckels Brook could support a limited macroinvertebrate community of generally tolerant species, and some small fish as were found in the reference channelized subsection above the SRVSA discharge (Stations 1 and 2). Unless it were altered or removed, the Calco Dam complex would continue to prevent fish migration.
>
> Wildlife typical of narrow stream corridors could inhabit the generally narrow strips of land between Lower Cuckels Brook and nearby railroad tracks and waste lagoons. Restoration of aquatic life in Lower Cuckels Brook would be expected to have little impact on aquatic life in the Raritan River.

Recreation - Lower Cuckels Brook would be too shallow for swimming or boating, and its small fish could not support sport fishing. The industrial surroundings of Lower Cuckels Brook, including waste lagoons and active manufacturing facilities and railroads, severely reduces the potential for other recreational activities such as streamside trails and picnic areas, wading, and nature appreciation. As Lower Cuckels Brook is on private industrial property, trespassing along this brook and in the surrounding area is discouraged.

It would appear unlikely that any of the landowners, or any government agency, would develop recreational facilities along lower Cuckels Brook or even remove some of the brush which impairs access to most of the Brook. Recreation along Lower Cuckels Brook would be limited, occasional, and informal.

Other Water Uses - Although water quality in Lower Cuckels Brook would generally meet FW-2 Nontrout criteria, the volume of natural flow in Lower Cuckels Brook would be insufficient for potable water supply or for industrial or agricultural water use.

In general, Lower Cuckels Brook would become a small channelized tributary segment flowing through a heavily industrialized area, free of gross pollution and capable of supporting a modest aquatic community and very limited recreational use.

IV. SUMMARY AND CONCLUSIONS

This use-attainability analysis has discussed the present impairment of the currently designated uses of Lower Cuckels Brook, the role of wastewater discharges in such impairment, and the extent to which currently designated water uses might be achieved if the wastewater discharges were removed. Further analysis, outside the scope of this survey, will be required: to document the costs of removing SRVSA and American Cyanamid effluent from Lower Cuckels Brook, and to evaluate the impact of the SRVSA and American Cyanamid discharges on the Raritan River. These analyses may lead to the development of site-specific water quality standards for Lower Cuckels Brook (designated uses limited to the conveyance of wastewater and the prevention of nuisances), or to the removal of the wastewater discharges from Lower Cuckels Brook. In either case, effluent limits would be established to protect water quality in the Raritan River.

WATER BODY SURVEY AND ASSESSMENT
Deep Creek And Canal Creek
Scotland Neck, North Carolina

I. INTRODUCTION

A. Site Description

The Town of Scotland Neck is located in Halifax County in the lower coastal plain of North Carolina. The Town's wastewater, made up mostly of domestic waste with a small amount of textile waste, is treated in an oxidation ditch of 0.6 mgd design capacity. The treatment plant is located two-tenths of a mile southwest of Scotland Neck off U.S. Highway 258, as seen in Figure 1. The effluent (0.323 mgd average) is discharged to Canal Creek which is a tributary to Deep Creek.

Canal Creek is a channelized stream which passes through an agricultural watershed, but also receives some urban runoff from the western sections of Scotland Neck. It is a Class C stream with a drainage area of 2.4 square miles, an average stream flow of 3.3 cfs, and a 7Q10 of 0.0 cfs. The Creek retains definite banks for about 900 feet below the outfall at which point it splits into numerous shifting channels and flows 800 to 1400 feet through a cypress swamp before reaching Deep Creek. During dry periods the braided channels of Canal Creek can be visually traced to Deep Creek. During wet periods Canal Creek overflows into the surrounding wetland and flow is no longer restricted to the channels.

Deep Creek is a typical tannin colored Inner Coastal Plain stream that has a heavily wooded paludal flood plain. The main channel is not deeply entrenched. In some sections streamflow passes through braided channels, or may be conveyed through the wetland by sheetflow. During dry weather flow periods the main channel is fairly distinct and the adjacent wetland is saturated, but not inundated. During wet weather periods the main channel is less distinct, adjacent areas become flooded and previously dry areas become saturated.

B. Problem Definition

The Town of Scotland Neck is unable to meet its final NPDES Permit limits and is operating with a Special Order by Consent which specifies interim limits. The Town is requesting a 201 Step III grant to upgrade treatment by increasing hydraulic capacity to 0.675 mgd with an additional clarifier, an aerobic digestor, tertiary filters, a chlorine contact chamber, post aeration and additional sludge drying beds. The treated effluent from Scotland Neck is discharged into Canal Creek. The lower reaches of Canal Creek are part of the swamp through which Deep Creek passes.

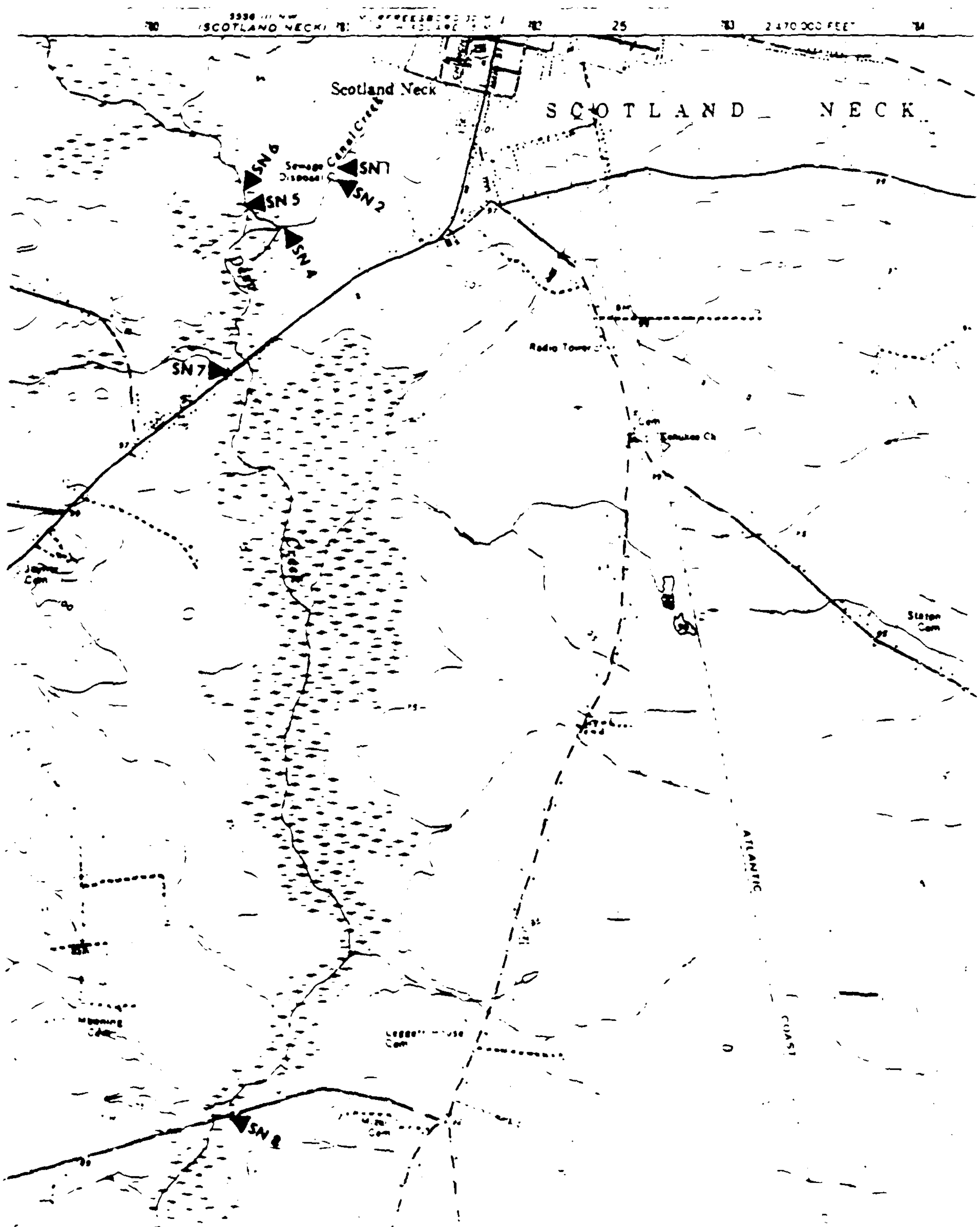


Figure 1. Study Area, Deep Creek
and Canal Creek

Deep Creek carries a "C" classification, but due to naturally low dissolved oxygen and other conditions imposed by the surrounding swamp, it is felt that reclassification to "C-Swamp" should be considered. Deep Creek should be classified C-Swamp because its physical characteristics meet the C-Swamp classification of the North Carolina Administrative Code for Classifications and Water Quality Standards. The Code states: Swamp waters shall mean those waters which are so designated by the Environmental Management Commission and which are topographically located so as to generally have very low velocities and certain other characteristics which are different from adjacent streams draining steeper topograpy. The C-Swamp classification provides for a minimum pH of 4.3 (compared to a range of pH 6.0 to pH 8.5 for C waters), and allows for low (unspecified) DO values if caused by natural conditions. DO concentrations in Deep Creek are usually below 4.0 mg/l.

C. Approach to Use Attainability Analysis

1. Data Available

1. Self Monitoring Reports from Scotland Neck.
2. Plant inspections by the Field Office.
3. Intensive Water Quality Survey of Canal Creek and Deep Creek at Scotland Neck in September, 1979. Study consisted of time-of-travel dye work and water quality sampling.

2. Additional Routine Data Collected

Water quality survey of Canal Creek and Deep Creek at Scotland Neck in June 1982. Water quality data was collected to support a biological survey of these creeks. The study included grab samples and flow measurements.

Benthic macroinvertebrates were collected from sites on Canal Creek and Deep Creek. Qualitative collection methods were used. A two-member team spent one hour per site collecting from as many habitats as possible. It is felt that this collection method is more reliable than quantitative collection methods (kicks, Surbers, ponars, etc.) in this type of habitat. Taxa are recorded as rare, common, and abundant.

II. ANALYSES CONDUCTED

A. Physical Factors

Sampling sites were chosen to correspond with sites previously sampled in a water quality survey of Canal and Deep Creeks. Three stations were selected on Canal Creek. SN-1 is located 40 feet above the Town of Scotland Neck Wastewater Treatment Plant outfall. This site serves as a reference station. The width at SN-1 is 7.0 feet and the average discharge (two flows were recorded in the September 1979 survey and one flow in the June 1982 survey) is 0.65 cubic feet per second. Canal Creek at SN-1 has been channelized and has a substrate composed of sand and silt. SN-4 is located on Canal Creek 900 feet below the discharge point. This section of Canal Creek has an average cross-sectional area of 11.8 feet and an average flow of 1.33 cubic feet per second. The stream in this section is also channelized and also has a substrate composed of sand and silt. There is a canopy of large cypress at SN-4 below the plant, while the canopy above SN-1 is reduced to a narrow buffer zone. The potential uses of Deep Creek are limited by its inaccessability in these areas.

A third station (SN-5) was selected on one of the lower channels of Canal Creek at the confluence with Deep Creek 3200 feet upstream of the U.S. Highway 258 bridge. Discharge measurements could not be accomplished at this site during this survey because of the swampy nature of the stream with many ill-defined, shallow, slow moving courses. Benthic macroinvertebrates were collected from this site.

Three stations were chosen on Deep Creek. SN-6 is approximately 300 feet upstream of SN-5 on Canal Creek at its confluence with Deep Creek and is a reference site. SN-7 is located at the U.S. Highway 258 bridge and SN-8 is located further downstream at the SR 1100 bridge. SN-7 and SN-8 are below Canal Creek. There are some differences in habitat variability among these three sites. The substrate at both SN-6 and SN-7 is composed mostly of a deep layer of fine particulate matter. Usable and productive benthic habitats in this area are reduced because of the fine particulate layer. It is possible that the source of this sediment is from frequent overbank flows and from upstream sources. Productive benthic habitats include areas of macrophyte growth, snags, and submerged tree trunks. Discharge measurements were not taken at any of these three sites during this survey.

B. Chemical Factors

Chemical data from two water quality surveys show that the dissolved oxygen in Canal Creek is depressed while BOD_5, solids and nutrient levels are elevated. The 1982 study indicates, however, that the water quality is better than it was during the 1979 survey. Such water quality improvements may be due to the addition of chlorination equipment and other physical improvements as well as to the efforts of a new plant operator.

Both above and below its confluence with Canal Creek, Deep Creek shows poor water quality which may be attributed to natural conditions, but not to any influence from the waste load carried by Canal Creek. Canal Creek exhibited higher DO levels than Deep Creek.

C. Biological Factors

The impact of the effluent on the fauna of Canal Creek is clear. A 63 percent reduction in taxa richness from 35 at SN-1 to only 13 at SN-4 indicates severe stress as measured against criteria developed by biologists of the Water Quality Section. The overwhelming dominance of *Chironomus* at SN-4 is indicative of a low DO level and high concentrations of organic matter. To what extent this condition is attributable to the effluent or to natural swamp conditions is not clear. No impact to the benthos of Deep Creek was discerned which could be attributed to the effluent.

III. FINDINGS

Deep Creek is currently designated as a class C warm water fishery but due to naturally low dissolved oxygen concentrations may not be able to satisfy the class C dissolved oxygen criteria. The DO criterion for class C waters stipulates a minimum value of 4 ppm, yet the DO in Deep Creek, in both the 1979 and the 1982 studies, was less than 4 ppm. Thus from the standpoint of aquatic life uses, Deep Creek may not be able to support the forms of aquatic life which are intended for protection under the class C standards. Because of prevailing natural conditions, there are no higher potential uses of Deep Creek than now exist; yet because of prevailing natural conditions and in light of the results of this water body assessment, the C-swamp use designation appears to be a more appropriate designation under existing North Carolina Water Quality Standards.

Canal Creek is degraded by the effluent from the Scotland Neck wastewater treatment plant. The BOD_5, fecal coliform, solids and nutrient levels are elevated while the DO concentration is depressed. The reach immediately below the outfall is affected by an accumulation of organic solids, by discoloration and by odors associated with the wastewater.

IV. SUMMARY AND CONCLUSIONS

The water body survey of Deep Creek and Canal Creek included a consideration of physical, chemical and biological factors. The focus of interest was those factors responsible for water quality in Deep Creek, including possible deliterious effects of the Scotland Neck wastewater on this water body. The analyses indicate that the effluent does not appear to affect Deep Creek. Instead, the water quality of Deep Creek reflects natural conditions imposed by seasonal low flow and high temperature, and reflects the nutrient and organic contribution of the surrounding farmland and wetland. It is concluded that the C-Swamp designation more correctly reflects the uses of Deep Creek than does the C designation.

In contrast to Deep Creek, Canal Creek is clearly affected by the treated effluent. Further examination would be required to determine the extent of recovery that might be expected in Canal Creek if the plant were to meet current permit requirements or if the proposed changes to the plant were incorporated into the treatment process.

WATER BODY SURVEY AND ASSESSMENT
Malheur River
Malheur County, Oregon

I INTRODUCTION

A. Site Description

The Malheur River, in southeastern Oregon, flows eastward to the Snake River which separates Oregon from Idaho. Most of Malheur County is under some form of agricultural production. With an average annual precipitation of less than 10 inches, the delivery of irrigation water is essential to maintain the high agricultural productivity of the area.

The Malheur River system serves as a major source of water for the area's irrigation requirements (out of basin transfer of water from Owyhee Reservoir augments the Malheur supply). Reservoirs, dams, and diversions have been built on the Malheur and its tributaries to supply the irrigation network. The first major withdrawal occurs at the Namorf Dam and Diversion, at Malheur River Mile 69. Figure 1 presents a schematic of the study area.

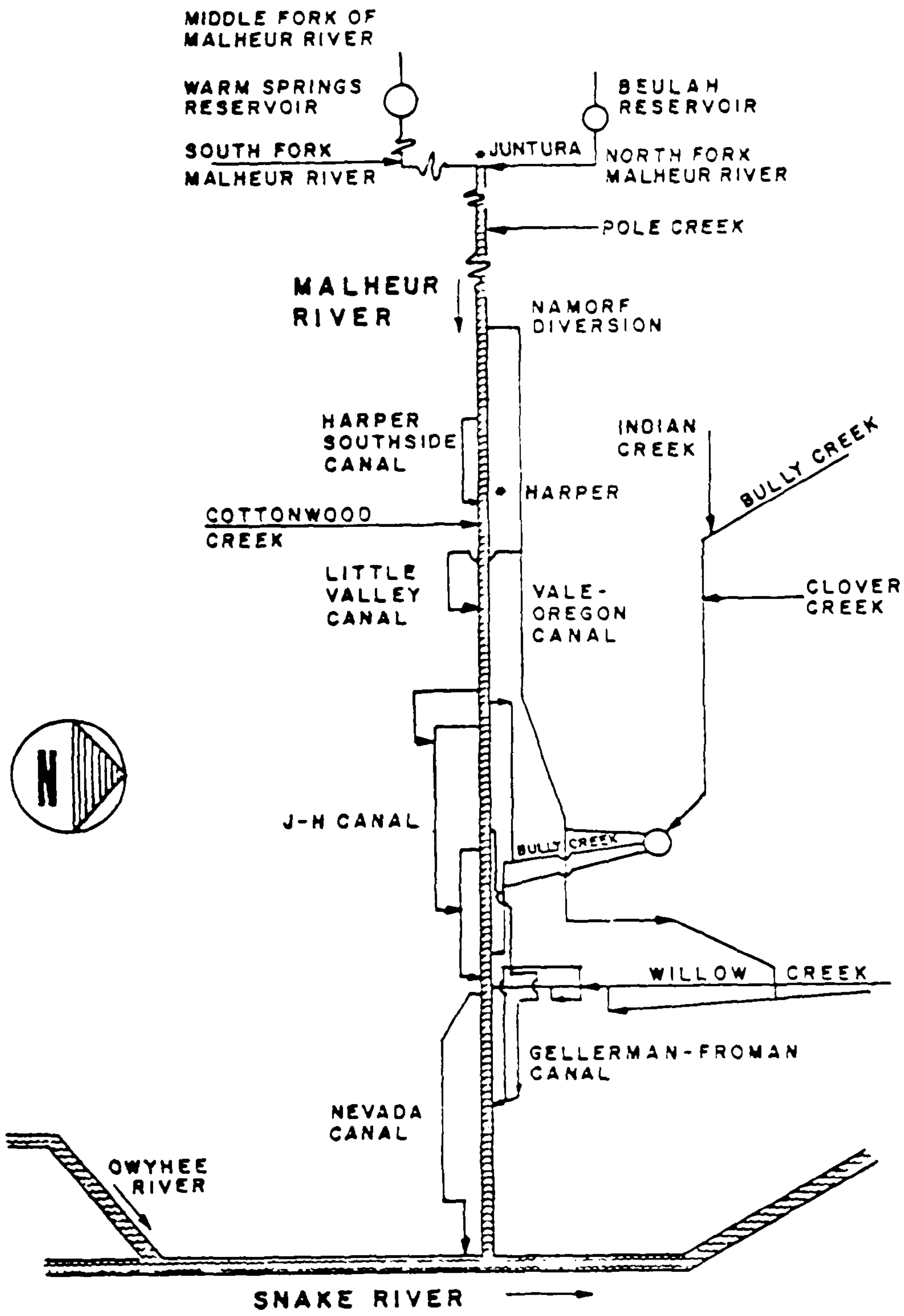
Irrigation water is delivered to individual farms by a complicated system of canals and laterals. Additional water is obtained from drainage canals and groundwater sources. An integral part of the water distribution system is the use and reuse of irrigation return flows five or six times before it is finally discharged to the Snake River.

B. Problem Definition

The Malheur River above Namorf Dam and Diversion is managed primarily as a trout fishery, and from Namorf to the mouth as a warm-water fishery. The upper portion of the river system is appropriately classified. Below Namorf Dam, however, the river is inappropriately classified as supporting a cold-water fishery, and therefore was selected for review. This review was conducted as part of the U.S. Environmental Protection Agency's field test of the draft "Water Body Survey and Assessment Guidance" for conducting a use attainability analysis. The guidance document supports the proposed rule to revise and consolidate the existing regulation governing the development, review, and approval of water quality standards under Section 303 of the Clean Water Act.

C. Approach to Use Attainability Analysis

Assessment of the Malheur River is based on a site visit which included meetings with representatives of the Malheur County Citizen's Water Resources Committee, the USDA-Soil Conservation Service, the Oregon Department of Environmental



SIMPLIFIED FLOW SCHEMATIC
MALHEUR RIVER IRRIGATION SYSTEM

- FIGURE 1 -

Quality (ODEQ), the Oregon Department of Fish and Wildlife (ODFW), and the U.S. Environmental Protection Agency (EPA): and upon the findings reported in two studies:

Final Report, Two Year Sampling Program, Malheur County Water Quality Management Plan, Malheur County Planning Office, Vale, Oregon, 1981.

Bowers, Hosford and Moore, Stream Surveys of the Lower Owyhee and Malheur Rivers, A Report to the Malheur County Water Resources Committee, Oregon Department of Fish and Wildlife, January, 1979.

The first report, prepared under amendments to Section 208 of the Clean Water Act, contains extensive information on the quantity, quality and disposition of the areas' water resources. The second document gives the fish populations found in the lower 69 miles of the Malheur River during June and July, 1978. Information in the ODFW report is incorporated in the 208 report. Additional fisheries information supplied by ODFW was also considered.

A representative of ODEQ, Portland, and the Water Quality Standards Coordinator, EPA Region X, Seattle, Washington, agreed that the data and analyses contained in these two reports were sufficient to re-examine existing designated uses of the Malheur River.

II ANALYSES CONDUCTED

Physical, chemical, and biological data were reviewed to determine: (1) whether the attainment of a salmonid fishery was feasible in the lower Malheur; and (2) whether some other designated use would be more appropriate to this reach. The elements of this review follow:

A. Physical Factors

Historically, salmonid fish probably used the lower Malheur (lower 50 miles) mainly as a migration route, because of the warm water and poor habitat. The first barrier to upstream fish migration was the Nevada Dam near Vale, constructed in 1880. Construction of the Warm Springs Dam in 1918, ended the anadromous fish runs in the Middle Fork Malheur. The construction of Beulah Dam in 1931, befell the remainder of anadromous fish runs on the North Fork Malheur. Finally, the construction of Brownlee Reservoir in 1958 completely blocked salmonid migrants destined for the upper Snake River System.

With the construction of the major irrigation reservoirs on the Malheur River and its tributaries, the natural flow characteristics in the lower river have changed. Instead of high early summer flows, low summer and fall flows and steady winter flow, the peak flows may occur in spring, if and when the upstream reservoirs spill. Also, a high sustained flow exists all summer as water is released from the dams for irrigation. A significant change limiting fish production in the Malheur River below Namorf is the extreme low flow that occurs when the reservoirs store water during the fall and winter for the next irrigation season.

Two other physical conditions affect the maintenance of salmonids in the lower Malheur. One is the high suspended solids load carried to the river by irrigation return flows. High suspended solids also occur during wet weather when high flows erode the stream bank and re-suspend bottom sediments. The seasonal range of suspended solids content is pronounced, with the highest concentrations occurring during irrigation season and during periods of wet weather. Observed peaks in lower reaches of the river, measured during the two-year 208 Program, reached 1300 mg/l, while background levels rarely dropped below 50 mg/l. A high suspended solids load in the river adversely affects the ability of sight-feeding salmonids to forage, and may limit the size of macroinvertebrate populations and algae production which are important to the salmonid food chain. A second factor is high summer water temperature which severely stresses salmonids. The high temperatures result from the suspended particles absorbing solar radiation.

B. Biological Factors

The biological profile of the river is mainly based on fisheries information, with some macroinvertebrate samples gathered by the Oregon Department of Fish and Wildlife (ODFW) in 1978. During the site visit, the participants agreed additional information on macroinvertebrates and periphyton would not be needed because the aquatic insect numbers and diversity were significantly greater in the intensively irrigated reach of the river than for the upper river where agricultural activity is sparse.

Although the Malheur River from Namorf to the mouth is managed as a warm water fishery, ODFW has expended little time and few resources on this stretch of the river because it is not a productive fish habitat. Survey results in summer of 1978 showed a low ratio of game fish to rough fish over the lower 69 miles of the Malheur River.

In the section between Namorf and the Gellerman-Froman Diversion Dam there was little change in water quality although water temperatures were elevated. Only three game fish were captured but non-game fish sight-feeders were common. Low winter flows over a streambed having few deep pools for overwinter survival appears to limit fish production in this reach of river.

In the stretch from the Gellerman-Froman Diversion to the mouth, the river flows through a region of intensive cultivation. The river carries a high silt load which affects sight-feeding fish. Low flows immediately below the Gellerman-Froman Dam also limit fish production in this area.

C. Chemical Factors

A considerable amount of chemical data exist on the Malheur River. However, since the existing and potential uses of the river are dictated largely by physical constraints, dissolved oxygen was the only chemical parameter considered in the assessment.

The Dissolved Oxygen Standard established for the Malheur River Basin calls for a minimum of 75 percent of saturation at the seasonal low and 95 percent of saturation in spawning areas or during spawning, hatching, and fry stages of salmonid fishes. One sample collected at Namorf fell below the standard to 73 percent of saturation or 8.3 mg/l in November, 1978. All other samples were above this content, reaching as high as 170 percent of saturation during the summer due to algae. Data collected by the ODEQ from Malheur River near the mouth between 1976 and 1979 showed the dissolved oxygen content ranged from 78 to 174 percent saturation. The dissolved oxygen content in the lower Malheur River is adequate to support a warm-water fishery.

III FINDINGS

A. Existing Uses

The lower Malheur River is currently designated as a salmonid fishery, but it is managed as a warm water fishery. Due to a number of physical constraints on the lower river, conditions are generally unfavorable for game fish, so rough fish predominate. In practice, the lower Malheur River serves as a source and a sink for irrigation water. This type of use contributes to water quality conditions which are unfavorable to salmonids.

B. Potential Uses

Salmonid spawning and rearing areas generally require the highest criteria of all the established beneficial uses. It would be impractical, if not impossible in some areas, to improve water quality to the level required by salmonids. However, even if this could be accomplished, high summer temperatures and seasonal low flows would still prevail. While salmonids historically moved through the Malheur River to spawn in the headwater areas, year-round resident fish populations probably did not exist in some of these areas at the time.

The Malheur River basin can be divided into areas, based upon differing major uses. Suggested divisions are: (1) headwater areas above the reservoirs; (2) reservoirs; (3) reaches below the reservoirs and above the intensively irrigated areas: (4) intensively irrigated areas; and (5) the Snake River.

In intensively irrigated areas, criteria should reflect the primary use of the water. Higher levels of certain parameters (i.e., suspended solids, nutrients, temperature, etc.) should be allowed in these areas since intensively irrigated agriculture, even under ideal conditions, will unavoidably contribute higher levels of these parameters. Criteria, therefore, should be based on the conditions that exist after Best Management Practices have been implemented.

IV SUMMARY AND CONCLUSIONS

Malheur River flows have been extensively altered through the construction of several dams and diversion structures designed to store and distribute water for agricultural uses. These dams, as well as others on the Snake River, to which the Malheur is tributary, block natural fish migrations in the river and, thus, have permanently altered the river's fisheries. In addition, water quality below Namorf Dam has been affected, primarily through agricultural practices, in a way which severely restricts the type of fish that can successfully inhabit the water. One important factor which affects fish populations below Namorf is the high suspended solids loading which effectively selects against sight-feeding species. Other conditions which could affect the types and survival of fish species below Namorf include low flow during the fall and winter when reservoirs are being filled in preparation for the coming irrigation season, as well as high suspended solids, and high temperatures during the summer irrigation season.

Realistically, the Malheur River could not be returned to its natural state unless a large number of hydraulic structures were removed. Removal of these structures would result in the demise of agriculture in the region, which is the mainstay of the county's economy. Furthermore, removal of these structures is out of the question due to the legal water rights which have been established in the region. These water rights can only be satisfied through the system of dams, reservoirs, and diversions which have been constructed in the river system. Thus, the changes in the Malheur River Basin are irrevocable.

Physical barriers to fish migration coupled with the effects of high sediment loads and the hydraulics of the system have for years established the uses of the river. Given the existing conditions and uses of the Malheur River below the Namorf Diversion, classification of this river each should be changed from a salmonid fishery, a use that cannot be achieved, to achievable uses which are based on the existing resident fish populations and aquatic life to reflect the present and highest future uses of the river. Such a change in designated beneficial uses would not further jeopardize existing aquatic life in the river, nor would it result in any degradation in water quality.

WATER BODY SURVEY AND ASSESSMENT
Pecan Bayou
Brownwood, Texas

I. INTRODUCTION

A. Site Description

Segment 1417 of the Colorado River Basin (Pecan Bayou) originates below the Lake Brownwood Dam and extends approximately 57.0 miles to the Colorado River (Figure 1). The Lake Brownwood Dam was completed in 1933. Malfunction of the dam's outlet apparatus led to its permanent closure in 1934. Since that time, discharges from the reservoir occur only infrequently during periods of prolonged high runoff conditions in the watershed. Dam seepage provides the base flow to Pecan Bayou (Segment 1417). The reservoir is operated for flood control and water supply. The Brown County WID transports water from the reservoir via aqueduct to Brownwood for industrial distribution, domestic treated water distribution to the Cities of Brownwood and Bangs and the Brookesmith Water System, and irrigation distribution. Some irrigation water is diverted from the aqueduct before reaching Brownwood.

Pecan Bayou meanders about nine miles from Lake Brownwood to the City of Brownwood. Two small dams impound water within this reach, and Brown County WID operates an auxilliary pumping station in this area to supply their system during periods of high demand.

Two tributaries normally provide inflow to Pecan Bayou. Adams Branch enters Pecan Bayou in Brownwood. The base flow consists of leaks and overflow in the Brown County WID storage reservoir and distribution system. Willis Creek enters Pecan Bayou below Brownwood. The base flow in Willis Creek is usually provided by seepage through a soil conservation dam.

The main Brownwood sewage treatment plant discharges effluent to Willis Creek one mile above its confluence with Pecan Bayou. Sulfur Draw, which carries brine from an artesian salt water well and wastewater from the Atchison, Topeka and Santa Fe Railroad Co., enters Willis Creek about 1,700 feet below the Brownwood sewage treatment plant. Below the Willis Creek confluence, Pecan Bayou meanders about 42.6 miles to the Colorado River, and receives no additional inflow during dry weather conditions. Agricultural water withdrawals for irrigation may significantly reduce the streamflow during the growing season.

The Pecan Bayou drainage basin is composed primarily of range and croplands. The stream banks, however, are densely vegetated with trees, shrubs and grasses. The bayou is typically 10-65 feet wide, 2-3 feet deep, and is generally sluggish in nature with soft organic sediments.

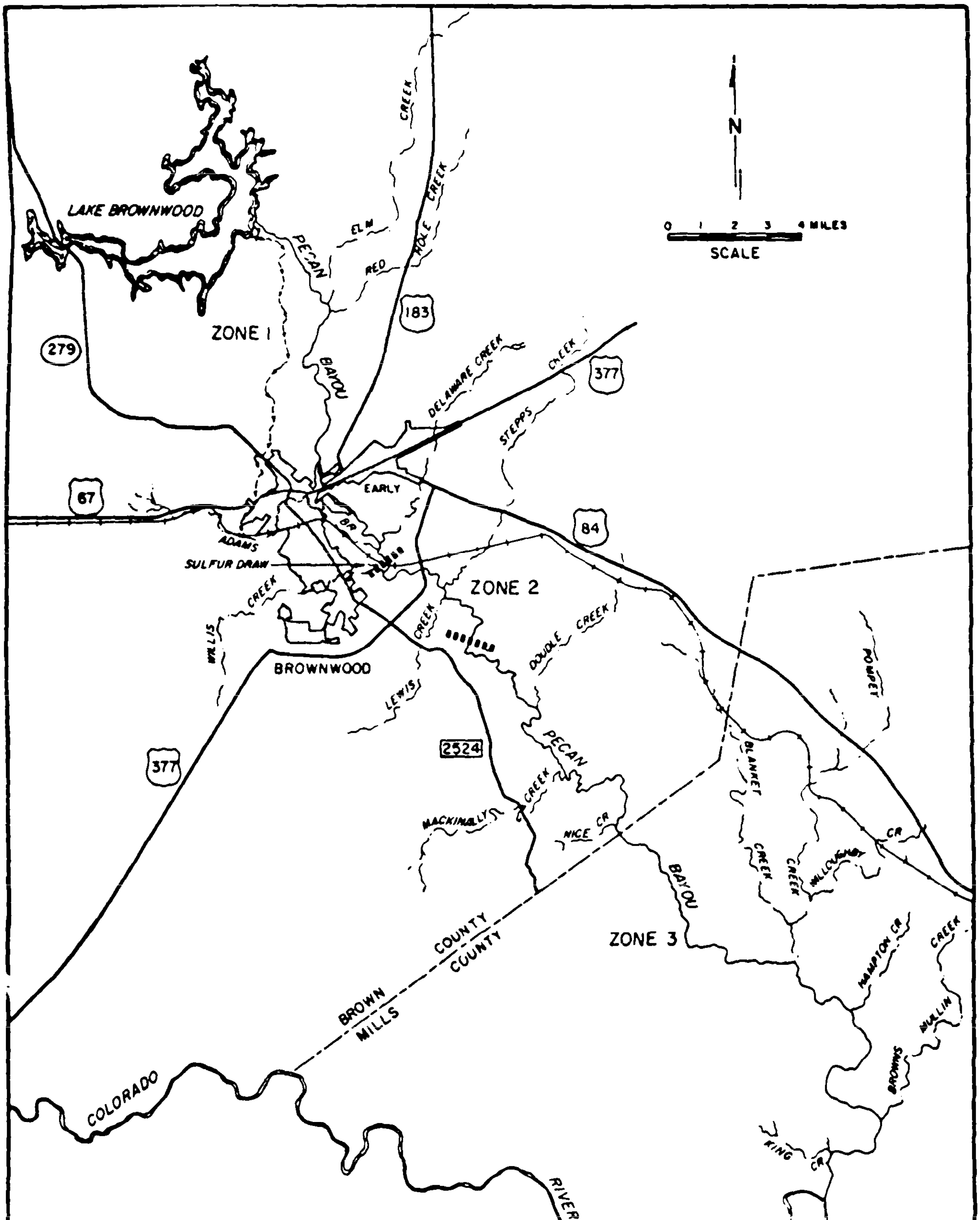


Figure 1

PECAN BAYOU SEGMENT MAP

B. Problem Definition

The designated water uses for Pecan Bayou include noncontact recreation, propagation of fish and wildlife, and domestic raw water supply. Criteria for dissolved oxygen (minimum of 5.0 mg/l), chlorides, sulfates, and total dissolved solids (annual averages not to exeed 250, 200, and 1000 mg/l, respectively), pH (range of 6.5 to 9.0) fecal coliform (log mean not to exceed 1000/100 ml), and temperature (maximum of 90°F) have been established for the segment.

Historically, Pecan Bayou is in generally poor condition during summer periods of low flow, when the Brownwood STP contributes a sizeable portion of the total stream flow. During low flow conditions, the stream is in a highly enriched state below the sewage outfall.

Existing data indicate that instream dissolved oxygen concentrations are frequently less than the criterion, and chloride and total dissolved solids annual average concentrations occasionally exceed the established criteria. The carbonaceous and nitrogenous oxygen deficencies in Pecan Bayou. The major cause of elevated chlorides in Pecan Bayou is the artesian brine discharge in to Sulfur Draw.

Toxic compounds (PCB, DDT, DDD, DDE, Lindane, Heptachlor epoxide, Dieldrin, Endrin, Chlordane, Pentachlorophenol, cadmium, lead, silver, and mercury) have been observed in water, sediment and fish tissues in Pecan Bayou (mainly below the confluence with Willis Creek). It has been determined that the major source was the Brownwood STP, but attempts to specifiy the points of origin further have been unsuccessful. However, recent levels show a declining trend.

C. Approach to Use Attainability

Assessment of Pecan Bayou is based on a site visit which included meetings with representatives of the State of Texas, EPA (Region VI and Headquarters) and Camp Dresser & McKee Inc., and upon information contained in a number of reports, memos and other related materials.

It was agreed by those present during the site visit that the data and analyses contained in these documents were sufficient for an examination of the existing designated uses of Pecan Bayou.

II. ANALYSES CONDUCTED

An extensive amount of physical, chemical, and biological data has been collected on Pecan Bayou since 1973. Most of the information was gathered to assess the impact of the Brownwood STP on the receiving stream. In order to simplify the presentation of these data, Pecan Bayou was divided into three zones (Figure 1): Zone 1 is the control area and extends from the Lake Brownwood Dam (river mile 57.0) to the Willis Creek confluence (river mile 42.6); Zone 2 is the impacted area and extends 9.0 miles below the Willis Creek confluence.

A. Physical Evaluation

With the exception of stream discharge, the physical characteristics of Pecan Bayou are relatively homogeneous by zone. Average width of the stream is about 44-50 feet, and average depth ranges from 2.1 to 3.25 feet. The low gradient (2.8 to 3.9 ft/mile) causes the bayou to be sluggish (average velocity of about 0.1 ft/sec), reaeration rates to be low (K_2 of 0.7 per day at 20°C), and pools to predominate over riffles (96% to 4%). Stream temperature averages about 18°C and ranges from 1-32°C. The substrate is composed primarily of mud (sludge deposits dominate in Zone 2), with small amounts of bedrock, gravel and sand being exposed in riffle areas.

Base flow in Pecan Bayou is provided by dam seepage (Zone 1) and the treated sewage discharge from the City of Brownwood (Zones 2 and 3). Median flow increases in a downstream direction from 2.5 cfs in Zone 1 to 17.4 cfs in Zone 3. Significantly higher mean flows (118 cfs in Zone 1 and 125 cfs in Zone 3) are the result of periodic high rainfall runoff conditions in the watershed.

B. Chemical Evaluation

Existing chemical data of Pecan Bayou characterize the degree of water quality degradation in Zone 2. Average dissolved oxygen levels are about 2.0 mg/l lower in the impact zone, and approximately 50% of the observations have been less than 5.0 mg/l. BOD_5, ammonia, nitrite, nitrate, and phosphorus levels are much higher in the impact zone as compared to the control and recovered zones. Un-ionized ammonia levels are also higher in Zone 2, but most of the concentrations were below the reported chronic levels allowable for warm water fishes. None of the levels exceeded the reported acute levels allowable for warm water fishes, and less than 4% of the levels were between the acute and chronic levels reported. Total dissolved solids, chlorides and sulfates were higher in Zones 2 and 3, mainly as a result of the brine and sewage discharges into Sulfur Draw and Willis Creek.

PCB, DDT, DDD, DDE and Lindane in water, and PCB, DDD, and DDE, Heptachlor epoxide, Dieldrin, Endrin, Chlordane, and Pentachlorophenol in sediment have been detected in Zone 2. PCB, DDT, DDD, and DDE concentrations in water have exceeded the criteria to protect freshwater aquatic life. The Brownwood STP was the suspected major source of these pesticides. Most of the recent levels, however, show a declining trend. PCB was detected also in Zones 1 and 3.

Heavy metals have not been detected in the water. Heavy metals in the sediment have shown the highest levels in Zone 2 for arsenic (3.7 mg/kg), cadmium (1.1 mg/kg), chromium (17.4 mg/kg), copper (9.5 mg/kg), lead (25.1 mg/kg), silver (1.5 mg/kg), zinc (90 mg/kg), and mercury (0.18 mg/kg).

C. Biological Evaluation

Fish samples collected from Zone 1 are representative of a fairly healthy population of game fish, rough fish and forage species. Zone 2 supported a smaller total number of fish which were composed primarily of rough fish and forage species. A relatively healthy balance of game fish, rough fish and forage species reappeared in the recovered zone.

Macrophytes were sparse in Zones 1 and 3. They were most abundant in Zone 2 below the Willis Creek confluence and were composed of vascular plants (pondweed, coontail, false loosestrife and duckweed) and filamentous algae (Cladophora and Hydrodictyon). Macrophyte abundance below Willis Creek is most likely due to nutrient enrichment of the area from the Brownwood STP.

Zone 1 is represented by a fairly diverse macrobenthic community characteristic of a clean-water mesotrophic stream. Nutrient and organic enrichment in Zone 2 has a distinct adverse effect as clean-water organisms are replaced by pollution-tolerant forms. Some clean-water organisms reappeared in Zone 3 and pollution-tolerant forms were not as prevalent; however, recovery to baseline conditions (Zone 1) was not complete.

Net phytoplankton desnities are lowest in Zone 1. Nutrient and organic enrichment in Zone 2 promotes a marked increase in abundance. Peak abundance was observed in the upper part of Zone 3. The decline below this area was probably caused by biotic grazing and/or nutrient deficiencies.

Fish samples for pesticides analyses have revealed detectable levels of PCB, DDE and DDD in Zone 1. Fish collected from zone 2 contained markedly higher amounts of DDE, DDD, DDT, Lindane and Chlordane than Zones 1 or 3. PCB in fish tissue was highest in ZOne 3, and measureable concentrations of DDE and DDD have also been observed. Concentrations of total DDT in whole fish tissues from Zone 2 have exceeded the USFDA Action Level of 5.0 mg/kg for edible fish tissues. Species representing the highest concentrations.

Computer modeling simulation were made to predict the dissolved oxygen profile in the impact zone during the fish spawning season. The results indicate that about three miles of Pecan Bayou in April and May and about 4 1/2 miles in June will be unsuitable for propogation, considering a minimum requirement of 4.0 mg/l. The model predicts a minimum D.O. of 0.8 mg/l in April, 1.2 mg/l in May, and 0 mg/l in June.

D. Institutional Evaluation

Two institutional factors exist which constrain the situation that exists in Pecan Bayou. These are the irrigation water rights and the Brownwood sewage treatment plant discharge permits. Although the sewage treatment plant discharge permits will expire and the problems created by the effluent could be eliminated in the future, there is a need for the flow provided by the discharge to satisfy the downstream water rights used for irrigation. Currently, there are eight water users on Pecan Bayou downstream of the Brownwood STP discharge with water rights permits totaling 2,957 acre-feet/year. Obviously, the 0.1 cfs base flow which exists in Pecan Bayou upstream of the STP discharge is not sufficient to fulfill these downstream demands. Therefore, it appears that the STP flow may be required to supplement the base flow in Pecan Bayou to meet the downstream demands for water unless it could be arranged that water from Lake Brownwood could be released by the Brown Co. WID #1 to meet the actual downstream water needs.

Modeling studies show that although there would be some improvement in water quality as a result of the sewage treatment plant going to advanced waste treatment (AWT), there would still be D.O. violations in a portion of Pecan Bayou in Zone 2. The studies also show that there is minimal additional water quality improvement between secondary and advanced waste treatment, although the costs associated with AWT were significantly higher than the cost for secondary treatment. In this case, the secondary treatment alternative would be the recommended course of action.

III. FINDINGS

A. Existing Uses

Pecan Bayou is currently being used in the following ways:

- Domestic Raw Water Supply
- Propagation of Fish and Wildlife
- Noncontact Recreation
- Irrigation
- City of Brownwood STP discharge (not an acceptable or approved use designation)

Use as a discharge route for the City of Brownwood's sewage treatment plant effluent has contributed to water quality conditions which are unfavorable to the propagation of fish and wildlife in a portion of Pecan Bayou.

B. Potential Uses

The Texas Department of Water Resources has established water uses which are deemed desirable for Pecan Bayou. These uses include: noncontact recreation, propagation of fish and wildlife, and domestic raw water supply.

Of these uses, propagation of fish and wildlife is unattainable in a portion of Pecan Bayou due to the effects of low dissolved oxygen levels in the bayou primarily during the spawning season. If the Brownwood sewage treatment plant effluent could be removed from Pecan Bayou, the persistently low dissolved oxygen conditions which exist and are unfavorable to fish spawning could be alleviated and the propagation of fish and wildlife could be partially restored to Pecan Bayou.

Public hearings held on the proposed expansion of the sewage treatment plant indicate a reluctance from the public and the City to pay for higher treatment levels, since modeling studies show minimal water quality improvement in Pecan Bayou between secondary and advanced waste treatment. In addition, an affordability analysis performed by the Texas Department of Water Resources (Construction Grants) indicates excessive treatment costs per month would result at the AWT level.

It appears that the elimination of the waste discharge from Pecan Bayou is not presently a feasible alternative, since the Brownwood STP currently holds a discharge permit and the water rights issue seems to be the overriding factor. Therefore, in the future, the uses which are most likely to exist are those which exist at present.

IV. SUMMARY AND CONCLUSIONS

A summary of the findings from the use attainability analysis are listed below:

- ° The designated use "propagation of fish and wildlife" is impaired in Zone 2 of Pecan Bayou.

- ° Advanced Treatment will not attain the designated use in Zone 2, partially because of low dilution, naturally sluggish characteristics ($\bar{X}$ velocity 0.1 ft/sec) and as a result, low assimlitive capacity of the bayou (K_2 reaeration rate 0.7 per day at 20°C).

- ° Downstream water rights for agricultural irrigation are significant.

- ° Dissolved oxygen levels are frequently less than the criterion of 5.0 mg/l in Pecan Bayou.

- ° Total DDT in whole fish from Zone 2 exceeded the U.S. Food and Drug Administration's action level of 5.0 mg/kg for edible fish tissues.

- ° Annual average chloride concentrations in Pecan Bayou are occasionally not in compliance with the numerical criteria.

Dissolved oxygen levels less than 5.0 mg/l (about 50% of the measurements) observed in Zone 2 of Pecan Bayou result from the organic and nutrient loading contributed by the Brownwood STP and the corresponding low waste assimilative capacity of the bayou. As previously mentioned, the major source of toxics found in the water, sediment and fish tissues was also determined to be the Brownwood STP. PCB and DDT in water have exceeded the criteria to protect freshwater aquatic life in Zone 2. Although the toxics appear to be declining in the water and sediment, the levels of total DDT found in whole fish exceed the U. S. Food and Drug Administration's action level (5.0 mg/k) for DDT in edible fish tissue. Investigations are underway by the Texas Department of Water Resources to further evaluate the magnitude of this potential problem.

Primarily as a result of the oxygen deficiencies and possibly be cause of the presence of toxic substances, the designated use "propagation of fish and wildlife" is not currently attained in Zone 2 of Pecan Bayou. These problems could be eliminated only if the Brownwood STP ceased to discharges into Pecan Bayou because even with advanced waste treatment the water quality of the receiving stream is not likely to improve sufficiently to support this designated use. Other treatment alternatives such as land treatment or overland flow are not feasible because of the current discharge is necessary to satisfy downstream water rights for agricultural irrigation. If the flow required to meet the water rights could be augmented from other sources, then the sewage treatment plant discharge could be eliminated in the future.

The annual average chloride level in Pecan Bayou are occasionally not in compliance with the established criterion. The primary source has been determined to be a privately owned salt water artesian well. Since efforts to control this discharge have proved futile, some consideration should be given to changing the numerical criterion for chlorides in Pecan Bayou.

In conclusion, it appears that either the Brownwood STP discharge into Pecan Bayou should be eliminated (if an alternative water source could be found to satisy the downstream water rights) or the numerical criterion for dissolved oxygen and the propogation of fish and wildlife use designation should be changed to reflect attainable conditions.

WATER BODY SURVEY AND ASSESSMENT
Salt Creek
Lincoln, Nebraska

I. INTRODUCTION

A. Site Description

The Salt Creek drainage basin is located in east central Nebraska. The mainstem of Salt Creek originates in southern Lancaster County and flows northeast to the Platte River (Figure 1). Ninety percent of the 1,621 square mile basin is devoted to agricultural production with the remaining ten percent primarily urban. The basin is characterized by moderately to steeply rolling uplands and nearly level to slightly undulating alluvial lands adjacent to major streams, primarily Salt Creek. Drainage in the area is usually quite good with the exception of minor problems sometimes associated with alluvial lands adjacent to the larger tributaries. Soils of the basin are of three general categories. Loessial soils are estimated to make up approximately 60 percent of the basin, glacial till soils 20 percent, and terrace and bottomland soils 20 percent.

Frequent high intensity rainfalls and increased runoff from land used for crop production has, in past years, contributed to flood damage in Lincoln and smaller urbanized areas downstream. To help alleviate these problems, flood control practices have been installed in the watershed. These practices, including several impoundments and channel modifications to the mainstream of Salt Creek, were completed during the late 1960's. Channel realignment of the lower two-thirds of Salt Creek has decreased the overall length of Salt Creek by nearly 34 percent (from 66.9 to 44.3 miles) and increased the gradient of the stream from 1.7 feet/mile to 2.7 feet/mile.

Salt Creek is currently divided into three classified segments: (upper reach) LP-4, (middle reach) LP-3a, and (lower reach) LP-3b. (Figure 1). Segments LP-4 and LP-3b are designated as Warmwater Habitats whereas segment LP-3a is designated as a Limited Warmwater Habitat.

B. Problem Definition

"Warmwater Habitat" and "Limited Warmwater Habitat" are two subcategories of the Fish and Wildlife Protection use designation in the Nebraska Water Quality Standards. The only distinction between these two use classes is that for Limited Warmwater Habitat waters, reproducing populations of fish are "...limited by irretrievable man-induced or natural background conditions." Although segment LP-3a is classified Limited Warmwater Habitat and segment LP-3b as Warmwater Habitat, they share similar physical characteristics. Since the existing fisheries of both segments were not thoroughly evaluated when the standard was revised, it is possible that the use designation for one or other segments is incorrect. This study was initiated to determine (1) if the Warmwater Habitat use is attainable for segment LP-3a and (2) what, if any, physical habitat or water quality constraints preclude the attainment of this use.

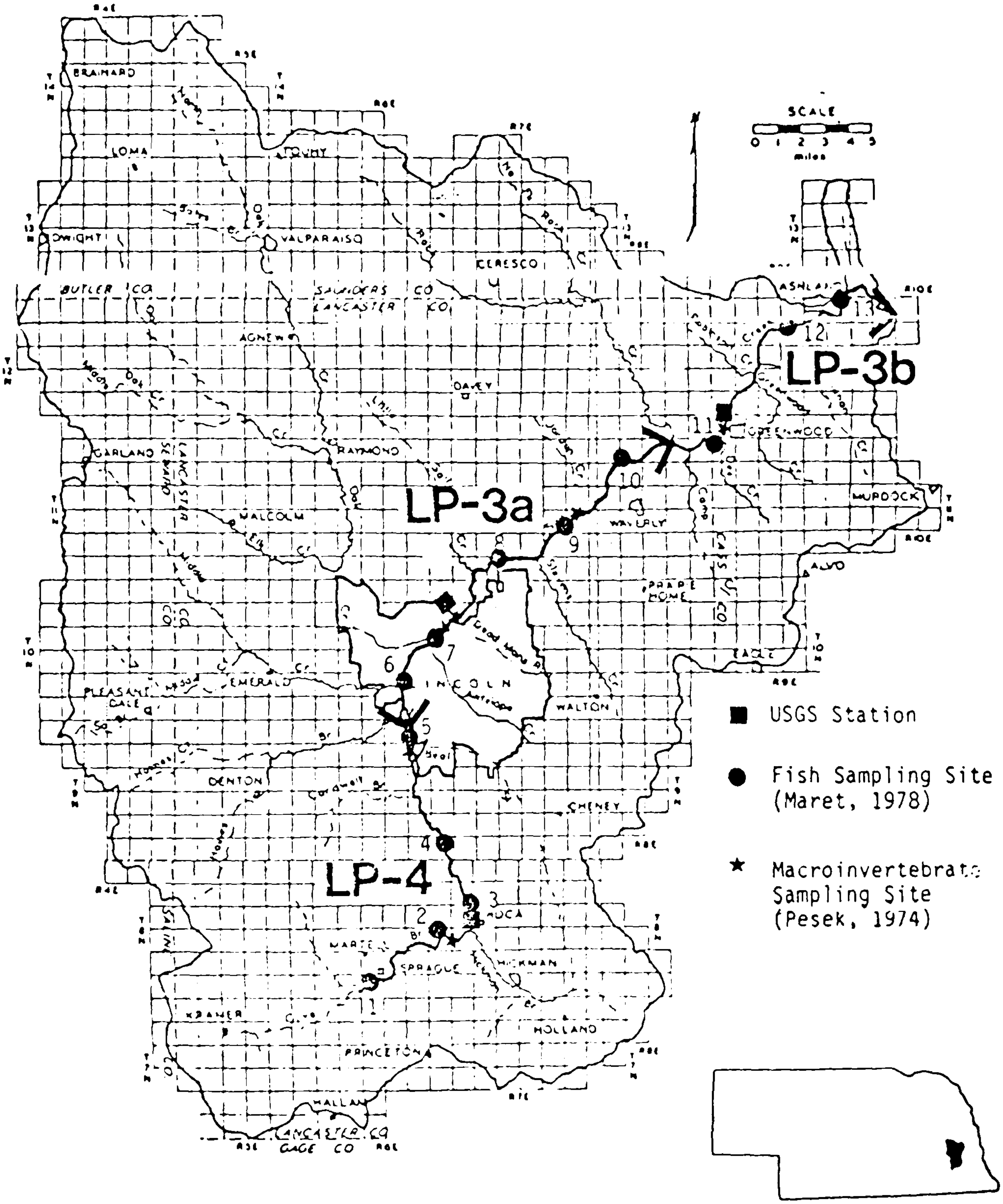


Figure 1. Monitoring sites from which data were used for Salt Creek attainability study.

C. Approach to Use Attainability Analysis

The analytical approach used in this study was a comparison of physical, chemical and biological parameters between the upper, middle, and lower Salt Creek segments with emphasis was on identifying limiting factors in the creek. The uppermost segment (LP-4) was used as the standard for comparison.

The data base used for this study included United States Geological Survey (USGS) and Nebraska Department of Environmental Control (NDEC) water quality data outlined in the US EPA STORET system, two Master of Science theses by Tom Pesek and Terry Maret, publications from the Nebraska Game and Parks Commission and USGS and personal observations by NDEC staff. No new data was collected in the study.

II. ANALYSES CONDUCTED

A review of physical, chemical and biological information was conducted to determine which aquatic life use designations would be appropriate. Physical characteristics for each of the three segments were evaluated and then compared to the physical habitat requirements of important warm water fish species. Characteristics limiting the fishery population were identified and the suitability of the physical habitat for maintaining a valued fishery was evaluated. General water quality comparisons were made between the upper reach of Salt Creek, and the lower reaches to establish water quality differences. A water quality index developed by the NDEC was used in this analysis to compare the relative quality of water in the segments. In addition, some critical chemical constituents required to maintain the important species were reviewed and compared to actual instream data to determine if water quality was stressing or precluding their populations.

The fish data collected by Maret was used to define the existing fishery population and composition of Salt Creek. This data was in turn used to determine the quality of the aquatic biota through the use of six biotic integrity classes of fish communities and the Karr Index tentative numerical index for defining class boundaries.

Macroinvertebrate data based on the study conducted by Pesek was also evaluated for density and diversity.

III. FINDINGS

Chemical data evaluated using the Water Quality Index indicated good water quality above Lincoln and degraded water quality at and below Lincoln. Non-point source contributions were identified as a cause of water quality degradation and have been implicated in fish kills in the stream. Dissolved solids in Salt Creek were found to be considerably higher than in other streams in the State. Natural background contributions are the major source of dissolved solids load to the stream. Water quality criteria violations monitored in Salt Creek during 1980 and 1981 were restricted to unionized ammonia and may have adversely impacted the existing downstream fishery. Toxics which occasionally approach or exceed the EPA criteria are chromium and lindane. Since EPA criteria for both parameters are based on some highly sensitive organisms which are not representative of indigenous populations typically found in Nebraska, the actual impact of these toxics is believed to be minimal.

Channelization was found to be a limiting factor in establishing a fishery in middle and lower Salt Creek. Terry Maret, in his 1977 study, found that substrate changes from silt and clay in the upper non-channelized area to primarily sand in the channelized area causing substantial changes in fish communities. The Habitat Suitability Index (HSI) developed by the Western Energy and Land Use Team of the U.S. Fish and Wildlife Service was used to evaluate physical habitat impacts on one important species (Channel Catfish) of fish in Salt Creek. The results indicated that upper Salt Creek had the best habitat for the fish investigated and middle Salt Creek had the worst. These results support the conclusion that middle Salt Creek lacks the physical habitat to sustain a valued warm water fishery. The Karr numerical index used to evaluate the fish data revealed that none of the stations rated above fair, further indicating the fish community is significantly impacted by surrounding rural and urban land uses.

Analysis of the abundance and diversity of macroinvertebrates indicated that the water quality in Salt Creek became progressively more degraded going downstream. Stations in the upper reaches were relatively unpolluted as characterized by the highest number of taxa, the greatest diversity and the presence of "clean-water" organisms.

IV. SUMMARY AND CONCLUSIONS

Based on the evaluation of the physical, chemical and biological characteristics of Salt Creek, the following conclusions were drawn by the State for the potential uses of the various segments:

1) Current classifications adequately define the attainable uses for upper and middle Salt Creek.

2) The Warmwater Habitat designated use may be unattainable for lower Salt Creek.

3) Channelization has limited existing instream habitat for middle Salt Creek. Instream habitat improvement in middle Salt Creek could increase the fishery but would lessen the effectiveness of flood control measures. Since flood control benefits are greater than any benefits that could be realized by enhancing the fishery, instream physical habitat remained the limiting factor for the fishery.

4) Existing water quality does not affect the limited Warmwater Habitat classification of middle Salt Creek.

5) Uncontrollable background source impacts on existing water quality and the effects of channelization on habitat may preclude attainment of the classified use.

The recommendations of the State drawn from these conclusions are as follows:

1) Keep upper section classification of Warmwater Habitat and middle section classification of Limited Warmwater Habitat.

2) Consider changing the lower section to a Limited Warmwater Habitat because of limited physical habitat and existing water quality.

WATER BODY SURVEY AND ASSESSMENT
South Fork Crow River
Hutchinson, Minnesota

I. INTRODUCTION

A. Site Description

The South Fork Crow River, located in south-central Minnesota, drains a watershed that covers approximately 1250 square miles. This river joins with the North Fork Crow to form the mainstem Crow River which flows to its confluence with the Mississippi River (Figure 1). Within the drainage basin, the predominant land uses are agricultural production and pasture land. The major soil types in the watershed are comprised of dark-colored, medium-to-fine textured silty loams, most of which are medium to well drained in character.

The physical characteristics of the South Fork Crow River are typical of many Minnesota streams flowing through agricultural lands. The upper portions of the river have been extensively channelized and at Hutchinson a forty foot wide, 12 foot high dam forms a reservoir west of the city. Downstream of the dam the river freely meanders through areas with light to moderately wooded banks to its confluence with the North Fork River Crow River. The average stream gradient for this section of the river is approximately two feet per mile and the substrate varies from sand, gravel and rubble in areas with steeper gradients to a silt-sand mixture in areas of slower velocities.

The average annual precipitation in the watershed is 27.6 inches. The runoff is greatest during the spring and early summer, after snowmelt, when the soils are generally saturated. Stream flow decreases during late summer and fall and is lowest in late winter. Small tributary streams in the watershed often go dry in the fall and winter because they have little natural storage and receive little ground water contribution. The seven-day ten year low flow condition for the South Fork below the dam at Hutchinson is approximately 0.7 cubic feet per second.

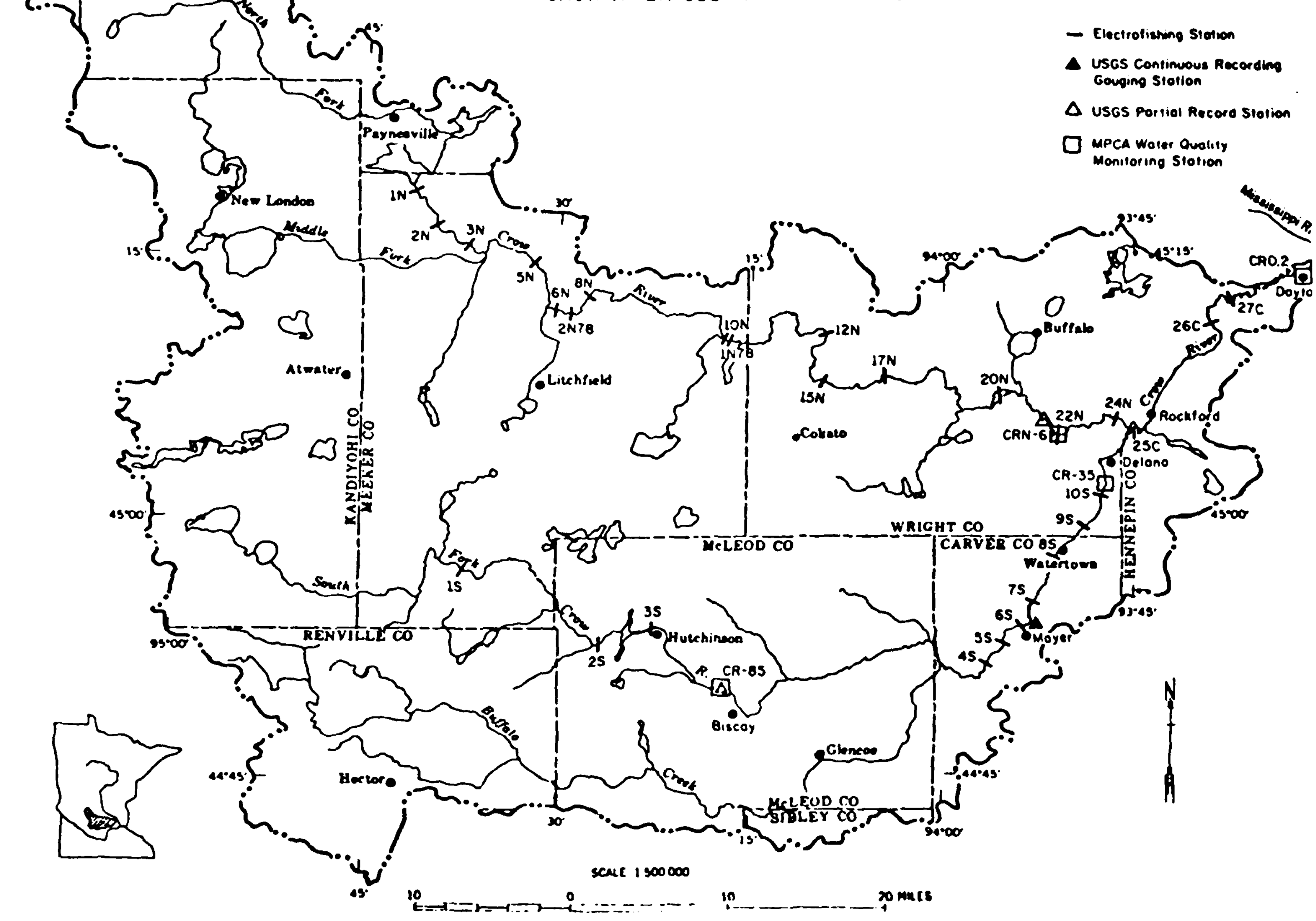
B. Problem Definition

The study on the South Fork Crow River was conducted in order to evaluate the existing fish community and to determine if the use designations are appropriate. At issue is the 2B fisheries and recreational use classification at Hutchinson. Is the water use classification appropriate for this segment?

C. Approach to Use Attainability

The analysis utilized an extensive data base compiled from data collected by the Minnesota Pollution Control Agency (MPCA), Minnesota Department of Natural Resources (MDNR) and United States Geological Survey (USGS). No new data was collected as part of the study. The USGS maintains partial or continuous flow record stations on both forks

FIGURE 1. STATION LOCATIONS FOR THE SOUTH FORK CROW RIVER USE ATTAINABILITY STUDY



and the mainstem Crow River with a data base of physical and chemical parameters available on STORET. The USGS data was used in the physical evaluation of the river. MPCA has a water quality monitoring data base on STORET for five stations in the Crow River watershed. The MPCA data plus analytical data from a waste load allocation study on the South Fork below Hutchinson was used in the chemical evaluation of the river. MDNR fisheries and stream survey data, a MDNR report on the analysis of the composition of fish populations in Minnesota rivers, and personal observations of MDNR personnel was used to evaluate the biological characteristics of the river.

The analytical approach used by the MPCA sought to 1) compare instream fish community health of the South Fork to that of the North Fork, the mainstem Crow River, and other warm water rivers in the State and 2) evaluate physical and chemical factors affecting fisheries and recreational uses. The North Fork of the Crow River was used for comparison because of sufficient fisheries data, similar land uses and morphologies, similar non-point source impacts and the lack of any significant point source dischargers.

II. ANALYSES CONDUCTED

Physical, chemical and biological factors were considered in this use attainability analysis to determine the biological health of the South Fork and to define the physical and chemical factors which may be limiting. A general assessment of the habitat potentials of the South Fork Crow River was performed using a habitat evaluation rating system developed by the Wisconsin Department of Natural Resources. In addition, the Tennant method for determining instream flow requirements was also employed in this study.

Fish species diversity, equitability and composition were used to define the biological health of the South Fork relative to that of the North Fork, the mainstem Crow and other warmwater rivers in Minnesota. Water quality monitoring data from stations above and below the point source discharges at Hutchinson were used to compare beneficial use impairment values pertaining to the designated fisheries and recreational uses of the South Fork Crow River. A computer data analysis program developed by EPA Region VIII was used to compute these values.

III. FINDINGS

The comparison of species diversity values for the North Fork and mainstem Crow River to the South Fork showed higher values for the North Fork and mainstem Crow. On the other hand, the South Fork had higher species equitability values. The percent species composition compared favorably to Peterson's (1975) estimates for median species diversity for a larger Minnesota river. Recruitment from tributaries, marshes, lakes and downstream rivers has given the South Fork a relatively balanced community which compares well to other warmwater rivers in the State. The calculated species diversity and equitability indices coupled with the analysis of species composition indicated that the South Fork of the Crow River does support a warmwater fishery with evidence of some degree of environmental stress.

The MPCA employed the Wisconsin habitat rating system and the Tennant method designated to quantify minimum instream fisheries flow requirements to identify any physical limiting factors. Based on the Wisconsin habitat evaluation assessment, habitat rating score were fair. The limiting factors identified via this assessment were: 1) lack of diverse streambed habitat suitable for reproduction, food production and cover and 2) instream water fluctuations (low flow may be a major controlling factor).

The State utilized EPA Region VIII's data analysis program to express stream water quality as a function of beneficial use. The closest downstream station to Hutchinson had the highest warmwater aquatic life use impairment values. Warmwater aquatic life use impairment values declined further downstream indicating that the point source dischargers were major contributors to this use impairment. However, primary contact recreational use impairment values were high throughout the stream. This led the State to believe that the impairment of primary contact recreational use is attributable to non-point sources.

IV. SUMMARY AND CONCLUSION

The State concluded from the study that: 1) the South Fork of the Crow River has a definite fisheries value although the use impairment values indicate some stress at Hutchinson on an already limited resource and 2) although the South Fork of the Crow River has a dominant rough fish population, game and sport fish present are important component species of this rivers' overall community structure.

From these conclusions the State recommended that the South Fork of the Crow River retain its present 2B fisheries and recreational use classification. Furthermore, efforts should continue to mitigate controllable factors that contribute to impairment of use. The effort should entail a reduction of marsh tilling and drainage, acceptance and implementation of agricultural BMP's and an upgrade of point source dischargers in Hutchinson.

WATER BODY SURVEY AND ASSESSMENT

South Platte River
Denver, Colorado

I. INTRODUCTION

A. Site Description

Segment 14 of the South Platte River originates north of the Chatfield Lake at Bowles Avenue in Arapahoe County and extends approximately 16 miles, through metro Denver, in a northerly direction to the Burlington ditch diversion near the Denver County-Adams County line. A map of the study area is presented in Figure 1. Chatfield Lake was originally constructed for the purposes of Flood control and recreation. The reservoir is owned by the U.S. Army Corps of Engineers and is essentially operated such that outflow equals inflow, up to a maximum of 5,000 cfs. In addition, water is released to satisfy irrigation demands as authorized by the State Engineers Office. There is also an informal agreement between the State Engineers Office and the Platte River Greenway Foundation for timing releases of water to increase flows during periods of high recreational use. The Greenway Foundation has played an important role in the significant improvement of water quality in the South Platte River.

There are several obstructions throughout Segment 14 including low head dams, kayak chutes (at Confluence Park and 13th Avenue), docking platforms, and weir diversion structures which alter the flow in the South Platte River. There are four major weir diversion structures in this area which divert flows for irrigation; one is located adjacent to the Columbine Country Club, a second near Union Avenue, a third upstream from Oxford Avenue, and a fourth at the Burlington Ditch near Franklin Street.

Significant dewatering of the South Platte River can occur due to instream diversions for irrigation and water supply and pumping from the numerous ground water dwells along the river.

Eight tributaries normally provide inflow to the South Platte River in Segment 14. These include Big Dry Creek, Little Dry Creek, Bear Creek, Harvard Gulch, Sanderson Gulch, Weir Gulch, Lakewood Gulch, and Cherry Creek.

There are several municipal and industrial facilities which discharge either directly to or into tributaries of the South Platte River in this reach. The major active discharges into the segment are the Littleton-Englewood wastewater treatment plant (WWTP), the Glendale WWTP, the City Ice Company, two Public Service company power plants (Zuni and Arapahoe), and Gates Rubber.

The South Platte River drainage basin in this area (approximately 120,000 acres) is composed primarily of extensively developed urban area (residential, industrial, commercial, services, roads), parks and recreational areas, gravel mining areas, and rural areas south of the urban centers for farming and grazing.

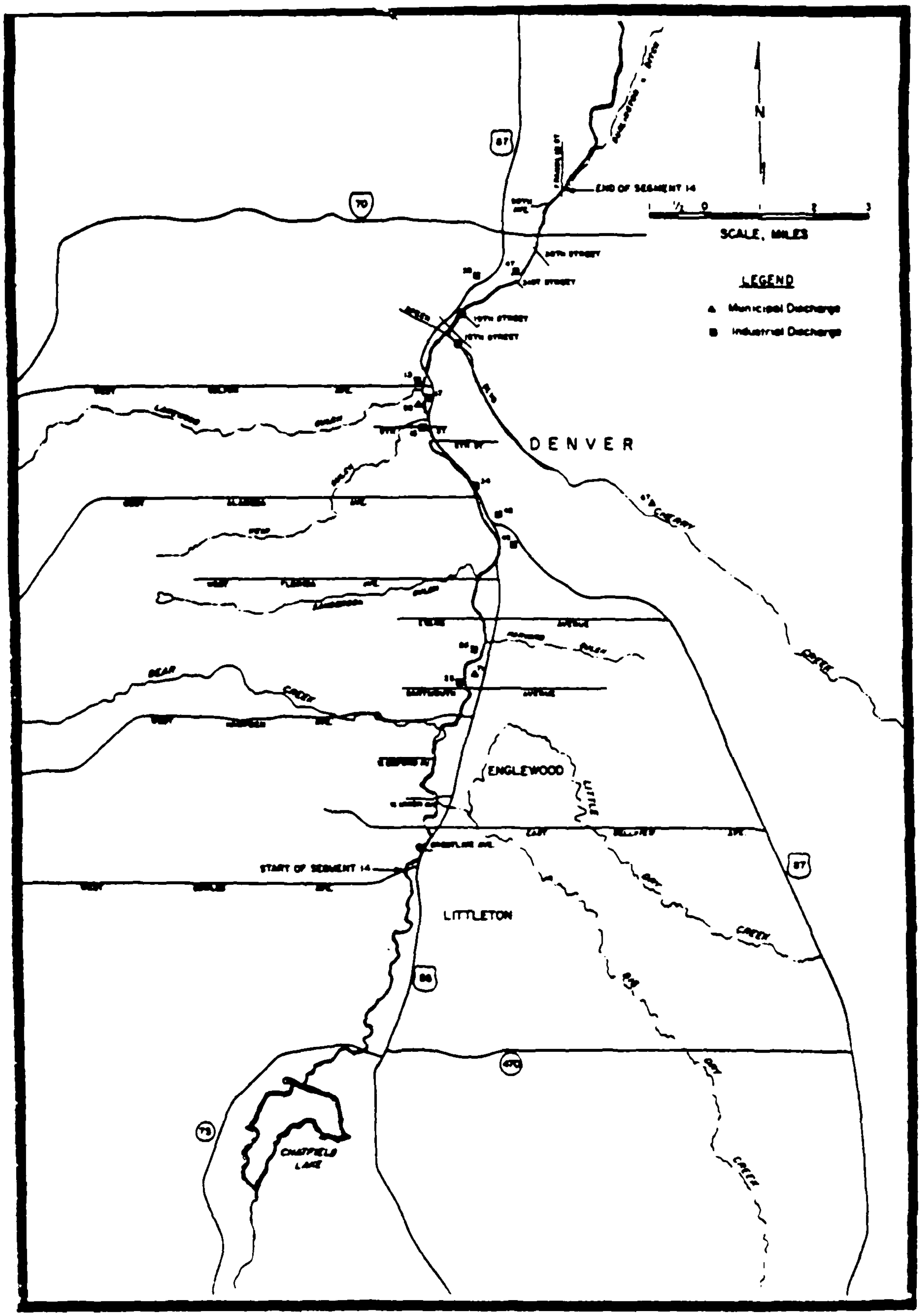


Figure 1

SOUTH PLATTE RIVER STUDY AREA MAP

In the study area, the South Platte River is typically 50-150 feet wide and 1-16 feet deep (typically 1-2 feet) and has an average channel bed slope of 12.67 feet per mile, with alternating riffle and pool reaches. The channel banks are composed essentially of sandy-gravelly materials that erode easily when exposed to high-flow conditions. The stream banks are generally sparsely vegetated with trees, shrubs, and grasses (or paving in the urban centers.)

B. Problem Definition

The following use classifications have been designated for Segment 14 of the South Platte River:

- Recreation - Class 2 - secondary contact
- Aquatic Life - Class 1 - warm water aquatic life
- Agriculture
- Domestic Water Supply

Following a review of the water quality studies and data available for Segment 14 of the South Platte River, several observations and trends in the data have been noted, including:

- Fecal coliform values exceeded the recommended limits for recreational uses in the lower portion of Segment 14.

- Un-ionized ammonia levels exceeded the water quality criterion for the protection of aquatic life in the lower portion of the segment.

- Levels of total recoverable metals (lead, zinc, cadmium, total iron, total manganese, and total copper) have been measured which exceed the water quality criteria for the protection of aquatic life.

Although the exact points of origin have not been specified, it is generally felt that the source of the ammonia is municipal point sources, and the sources of the metals are industrial point sources.

In addition, the cities of Littleton and Englewood have challenged the Class I warm water aquatic life use on the basis that the flow and habitat are unsuitable to warrant the Class I designation, and they have also challenged the apporopriateness of the 0.06 mg/l un-ionized ammonia criteria on the basis of new toxicity data. The Colorado Water Quality Control Commission in November, 1982 approved the Class I aquatic life classification and the 0.06 mg/l un-ionized ammonia criteria.

C. Approach to Use Attainability

Assessment of Segment 14 of the South Platte River was based on a site visit (May 3-4, 1982) which included meetings with representatives of the Colorado Department of Health, EPA (Region VIII and Headquarters) and Camp Dresser & McKee Inc., and upon information contained in a number of reports, hearing transcripts and the other related materials. Most of the physical, chemical and biological data was obtained from the USGS, EPA (STORET), DRURP, and from studies. It was agreed that there was sufficient chemical, physical and biological data to proceed with the assessment, even though physical data on the aquatic habitat was limited.

II. ANALYSES CONDUCTED

A. Physical Factors

Streamflow in the South Platte River (Segment 14) is affected by several factors including releases from Chatfield Dam, diversions for irrigation and domestic water supply, irrigation return flows, wastewater discharges, tributary inflows, pumping from ground water wells in the river basin, evaporation from once-through cooling at the two power plants in Segment 14, and natural surface water evaporation. Since some of these factors (particularly ground water pumping, evaporation and irrigation diversions) are variable, flow in the South Platte River is used extensively for irrigation and during the irrigation season diversions and return flows may cause major changes in streamflow within relatively short reaches. During the summer, low-water conditions prevail because of increased evaporation, lack of rainfall, and the various uses made of the river water (e.g. irrigation diversions). Municipal, industrial, and storm-water discharges also contributes to the streamflow in the South Platte River.

Natural pools in the South Platte River are scarce and the shifting nature of the channel bed results in temporary pools, a feature which has a tendency to greatly limit the capacity for bottom food production. There are approximately 3-4 pools per river mile with the majority being backwater pools upstream of diversion structures, bridge crossings, low head dams, docking platforms, drop-off structures usually downstream of wastewater treatment plant outfalls, kayak chutes, and debris. The hydraulic effect of each obstruction is generally to cause a backwater condition immediately upstream from the structure, scouring immediately downstream, and sandbar development below that. These pools act as settling basins for silt and debris which no longer get flushed during the high springs flows once Chatfield Lake was completed.

In the plains, channels of the South Platte River and lower reaches of tributaries cut through deep alluvial gravel and soil deposits. Sparse vegetation does not hold the soils, so stream bank erosion and channel bed degredation is common during periods of high flow, particularly during the spring snowmelt season. The high intensity - low duration rainstorms which occur during the summer (May, June, and July) also temporarily muddy the streams.

An evaluation of the physical streambed characteristics of Segment 14 to determine the potential of the Segment to maintain and attract warm water aquatic life was conducted by Keeton Fisheries Consultants, Inc. The study concluded that the sediment loads in this reach of the South Platte River could pose a severe problem to the aquatic life forms present, however, further study needs to be conducted to substantiate this conclusion. Furthermore, some gravel mining operations have recently been discontinued thus the sediment problem may have been reduced.

The temperature in the South Platte River is primarily a function of releases from the bottom of Chatfield Lake, the degree of warming that takes place in the shallow mainstream and isolated pools, and the warming that occurs through the mixing of power plant cooling water with the South Platte River.

B. Chemical Factors

Water quality conditions in the South Platte River are substantially affected by municipal and industrial wastewater discharges, irrigation return flows and other agricultural activities, and non-point sources of pollution (primarily during rainfall-runoff events). Irrigation and water supply diversions also exert a major influence on water quality by reducing the stream flow, and thereby reducing the dilution assimilative capacity of the river.

- Dissolved oxygen levels were above the 5.0 mg/l criteria acceptable for the maintenance of aquatic life.
- Average concentrations of un-ionized ammonia exceeded the State water quality criteria of 0.06 mg/l NH_3-N only in the lower portion of Segment 14 (north of Speer Blvd.)
- Average total lead concentrations exceeded the water quality criteria of 25 ug/l in Big Dry Creek, Cherry Creek, and the South Platte River north of Cherry Creek, ranging from 30-72 ug/l.
- Average total zinc concentrations exceeded the criteria of 11 ug/l at all the DRURP sampling stations, ranging from 19-179 ug/l.
- Average total cadmium concentrations exceeded the criteria of 1 ug/l in Beer Creek, Cherry Creek and several sites in the South Platte, ranging from 2.2-3.6 ug/l.
- Average total iron concentrations exceeded the criteria of 1,000 ug/l in Cherry Creek and several locations on the South Platte River, ranging from 1129-9820 ug/l.
- Average soluble manganese concentrations exceeded the criteria of 50 ug/l in the South Platte River north of (and including) 19th Street and in Cherry Creek, ranging from 51-166 ug/l.
- Average total copper concentrations equalled or exceeded the criteria of 25 ug/l at all but two of the DRURP sampling sites, ranging from 25-83 ug/l.

C. Biological Factors

Several electrofishing studies have been conducted on the South Platte River in recent years. Most of the sampling took place in the fall with the exception of the study in the spring (1979). The data was reviewed by Colorado Department of Health personnel and it was generally agreed that the overall health of the existing warm water fishery is restricted by temperature extremes (very cold and shallow during the winter and low flow and high temperatures during the summer), the lack of sufficient physical habitat (i.e. structures for cover including rocks and dams, and deep pools) and the potentially stressful conditions created by the wastewater discharges (i.e. silt and organic and inorganic enrichment).

Following a review of the physical, chemical, and biological data available on the South Platte River, it was concluded that a fair warm water fishery could exist with only modest habitat improvements and maintenance of the existing ambient water quality and strict regulation prevent overfishing. With large habitat and water quality improvements, brown trout could potentially become a part of the fishery in Segment 14 of the South Platte River.

III. FINDINGS

A. Existing Uses

Segment 14 of the South Platte River is currently being used in the following ways:

- Irrigation Diversions and Return Flows
- Municipal and Industrial Water Supply
- Ground Water Recharge
- Once-through Cooling
- Municipal, Industrial, and Stormwater Discharges
- Recreation
- Warm Water Fishery

The irrigation diversions, water supply, ground water recharge, and cooling uses have primarily affected the flow in the South Platte River, resulting in significant dewatering at times. Irrigation return flows and wastewater dishcharges, on the other hand, exert their effects on the ambient and storm water quality in the River. These previous uses ultimately affect the existing warm water fishery and how the public perceives the river for recreation purposes.

B. Potential Uses

With the exception of a potential for increased recreation and the improvement of a limited warm water fishery, it is anticipated that the existing uses are likely to exist in the future. The increased recreational use will result from future Platte River Greenway Foundation projects. The improvement of a limited warm water fishery may come about in the future as the result of habitat improvements (pools, cover) control of toxic materials (un-ionized ammonia, heavy metals, cynanide), and the prevention of extensive sedimentation. However, the success of the fishery would rely on strict fishery regulations to prevent overfishing.

IV. SUMMARY AND CONCLUSIONS

A summary of the findings from the use attainability analysis are listed below:

- There is evidence to indicate that a warm water aquatic life community does exist and the potential for an improved fishery could be attained with slight habitat modifications (i.e. cover, pool).

- Elevated un-ionized ammonia levels were exhibited in the lower portion of Segment 14, although this cannot be attributed to the Littleton-Englewood WWTP discharge upstream. However, at the present time there is no basis for a change in the existing un-ionized ammonia criterion, particularly if EPA's methodology for determining site specific criteria becomes widely accepted.

- Increased turbidity exists in the South Platte River during a good portion of the fish spawning season, which represents a potential for problems associated with fish spawning.

- Increased sedimentation and siltation in the South Platte River could pose a potential threat to the aquatic life present; however, this condition might be reduced if Chatfield Lake could be operated to provide periodic flushing of the river.

- Elevated levels of heavy metals were observed in water and sediment samples, which could potentially affect the existing aquatic life.

- Insufficient data existed to determine the possible effects of chlorine and cyanide on the aquatic life present.

- Fecal coliform levels were extremely high in the lower portion of the South Platte River and Cherry Creek during periods of both low and high flow. The source in the South Platte River is apparently Cherry Creek, but the origin in Cherry Creek is unknown at this time.

On the basis of the preceding conclusions and recommendations, the warmwater fishery use classification and the un-ionized ammonia criterion (0.06 mg/l) recommended for Segment 14 of the South Platte should remain unchanged until there is further evidence to support making those changes.

APPENDIX U

List of EPA Regional Water Quality Standards Coordinators

WATER QUALITY STANDARDS COORDINATORS

Eric Hall, WQS Coordinator
EPA Region 1
Water Division
JFK Federal Building
Boston, MA 02203
617-565-3533

Wayne Jackson, WQS Coordinator
EPA Region 2
Water Division
26 Federal Plaza
New York, NY 10278
212-264-5685

Evelyn MacKnight, WQS Coordinator
EPA Region 3
Water Division
841 Chestnut Street
Philadelphia, PA 19107
215-597-4491

Fritz Wagener, WQS Coordinator
EPA Region 4
Water Division
345 Courtland Street, N.E.
Atlanta, GA 30365
404-347-3555x6633

David Pfeifer, WQS Coordinator
EPA Region 5
Water Division
77 West Jackson Boulevard
Chicago, IL 60604-3507
312-353-9024

Cheryl Overstreet, WQS Coordinator
EPA Region 6
Water Division
1445 Ross Avenue
First Interstate Bank Tower
Dallas, TX 75202
214-655-6643

Larry Shepard, WQS Coordinator
EPA Region 7
Water Complainance Branch
726 Minnesota Avenue
Kansas City, KS 66101
913-551-7441

Bill Wuertherle, WQS Coordinator
EPA Region 8
Water Division
999 18th Street
Denver, CO 80202-2405
303-293-1586

Phil Woods, WQS Coordinator
EPA Region 9
Water Division
75 Hawthorne Street
San Francisco, CA 94105
415-744-1997

Marcia Lagerloef, WQS Coordinator
EPA Region 10
Water Division (WS-139)
1200 Sixth Avenue
Seattle, WA 98101
206-553-0176

-or-

Sally Brough, WQS Coordinator
EPA Region 10
Water Division (WS-139)
1200 Sixth Avenue
Seattle, WA 98101
206-553-1754

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APPENDIX V

Water Quality Standards Program Document Request Forms

REV 02/07

WATER RESOURCE CENTER
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COMPLETE REQUESTOR PROFILE BELOW:

STANDARDS & APPLIED SCIENCE DIVISION/WATER QUALITY STANDARDS BRANCH

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TELEPHONE NUMBER	DATE REQUEST RECEIVED

DUE TO RESOURCE LIMITATIONS, ONLY ONE (1) COPY OF EACH DOCUMENT CAN BE PROVIDED TO A REQUESTOR.

TITLE	CHECK DOCUMENT REQUESTED
1. **Water Quality Standards Regulation, Part II, Environmental Protection Agency, Federal Register, November 8, 1983** *Regulations that govern the development, review, revision and approval of water quality standards under Section 303 of the Clean Water Act.*	
2. **Water Quality Standards Handbook, Second Edition, September 1993** *Contains guidance issued to date in support of the Water Quality Standards Regulation.*	
• **Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria, EPA 822/F-93-009, October 1993** *This memorandum transmits Office of Water policy and guidance on the interpretation and implementation of aquatic life metals criteria. It covers aquatic life criteria, total maximum daily loads permits, effluent monitoring, compliance and ambient monitoring.*	
3. **Water Quality Standards for the 21st Century, 1989** *Summary of the proceedings from the first National Conference on water quality standards held in Dallas, Texas, March 1-3, 1989.*	
4. **Water Quality Standards for the 21st Century, 1991** *Summary of the proceedings from the second National Conference on water quality standards held in Arlington, Virginia, December 10-12, 1990.*	
5. **Compilation of Water Quality Standards for Marine Waters, November 1982** *Consists of marine water quality standards required by Section 304(a)(6) of the Clean Water Act. The document identifies marine water quality standards, the specific pollutants associated with such water quality standards and the particular waters to which such water quality standards apply. The compilation should not in any way be construed as Agency opinion as to whether the waters listed are marine waters within the meaning of Section 301(h) of the Clean Water Act or whether discharges to such waters are qualified for a Section 301(h) modification.*	

STANDARDS & APPLIED SCIENCE DIVISION/WATER QUALITY STANDARDS BRANCH	
TITLE	CHECK DOCUMENT REUQESTED
6. **Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, November 1983** *Contains technical guidance to assist States in implementing the revised water quality standards regulation (48 FR 51400, November 8, 1983). The guidance assists States in answering three key questions:* *a. What are the aquatic protection uses currently being achieved in the waterbody?* *b. What are the potential uses that can be attained based on the physical, chemical and biological characteristics of the waterbody?* *c. What are the causes of any impairment of the uses?*	
7. **Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Volume II: Estuarine Systems** *Contains technical guidance to assist States in implementing the revised water quality standards regulation (48 FR 51400, November 8, 1983). This document addresses the unique characteristics of estuarine systems and supplements the Technical Support Manual: Waterbody Summary and Assessments for Conducting Use Attainability Analyses (EPA, November 1983).*	
8. **Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Volume III: Lake Systems, November 1984** *Contains technical guidance to assist States in implementing the revised water quality standards regulation (48 FR 51400 November 8, 1983). The document addresses the unique characteristics of lake systems and supplements two additional guidance documents: Technical Support Manual: Waterbody Survey and Assessments for Conducting Use Attainability Analyses EPA, (November 1983) and Technical Support Manual: Waterbody Surveys and Assessments for Conducting Use Attainability Analyses, Vol II: Estuarine Systems.*	
9. **Health Effects Criteria for Marine Recreational Waters, EPA 600/1-80-031, August 1983** *This report presents health effects quality criteria for marine recreational waters and a recommendation for a specific criterion. The criteria were among those developed using data collected from an extensive in-house extramural microbiological research program conducted by the U.S. EPA over the years 1972-1979.*	
10. **Health Effects Criteria for Fresh Recreational Waters, EPA 660/1-84-004, August 1984** *This report presents health effects criteria for fresh recreational waters and a criterion for the quality of the bathing water based upon swimming - associated gastrointestinal illness. The criterion was developed from data obtained during a multi-year freshwater epidemiological-microbiological research program conducted at bathing beaches near Erie, Pennsylvania and Tulsa, Oklahoma. Three bacterial indications of fecal pollution were used to measure the water quality: E. Coli, enterococci and fecal coliforms.*	
11. **Introduction to Water Quality Standards, EPA 440/5-88-089, September 1988** *A primer on the water quality standards program written in question and answer format. The publication provides general information about various elements of the water quality standards program.*	
12. **Ambient Water Quality Criteria for Bacteria - 1986 EPA 440/5-84-002** *This document contains bacteriological water quality criteria. The recommended criteria are based on an estimate of bacterial indicator counts and gastro-intestinal illness rates.*	

STANDARDS & APPLIED SCIENCE DIVISION/WATER QUALITY STANDARDS BRANCH	
TITLE	**CHECK DOCUMENT REQUESTED**
13. Test Methods for Escherichia Coil and Enterococci; In Water by the Membrane Filter Procedure, EPA 600/4-85/076, 1985 *Contains methods used to measure the bacteriological densities of E. coli and enterococci in ambient waters. A direct relationship between the density of enterococci and E. coli in water and the occurrence of swimming - associated gastroenteritis has been established through epidemiological studies of marine and fresh water bathing beaches. These studies have led to the development of criteria which can be used to establish recreational water standards based on recognized health effects-water quality relationships.*	
14. Twenty-Six Water Quality Standards Criteria Summaries, September 1988 *These documents contain twenty-six summaries of State/Federal criteria. Twenty-six summaries have been compiled which contain information extracted from State water quality standards. Titles of the twenty-six documents are: Acidity-Alkalinity, Antidegradation, Arsenic, Bacteria, Cadmium, Chromium, Copper, Cyanide, Definitions, Designated Uses, Dissolved Oxygen, Dissolved Solids, General Provisions, Intermittent Streams, Iron, Lead, Mercury, Mixing Zones, Nitrogen-Ammonia/Nitrate/Nitrite, Organics, Other Elements, Pesticides, Phosphorus, Temperature, Turbidity, and Zinc.*	
15. Fifty-Seven State Water Quality Standards Summaries, September 1988 *Contains fifty-seven individual summaries of State water quality standards. Included in each summary is the name of a contact person, use classifications of water bodies, mixing zones, antidegradation policies and other pertinent information.*	
16. State Water Quality Standards Summaries, September 1988 (Composite document) *This document contains composite summaries of State water quality standards. The document contains information about use classifications, antidegradation policies and other information applicable to a States' water quality standards.*	
17. Transmittal of Final "Guidance for State Implementation of Water Quality Standards for CWA Section 303(c)(2)(B)", December 12, 1988 *Guidance on State adoption of criteria for priority toxic pollutants. The guidance is designed to help States comply with the 1987 Amendments to the Clean Water Act which requires States to control toxics in water quality standards.*	
18. Chronological Summary of Federal Water Quality Standards Promulgation Actions, January 1993 *This document contains the date, type of action and Federal Register citation for State water quality standards promulgated by EPA. The publication also contains information on Federally promulgated water quality standards which have been withdrawn and replaced with State approved standards.*	
19. Status Report: State Compliance with CWA Section 303(c)(2)(b) as of February 4, 1990 *Contains information on State efforts to comply with Section 303(c)(2)(B) of the Clean Water Act which requires adoption of water quality standards for priority pollutants. The report identifies the States that are compliant as of February 4, 1990, summarizes the status of State actions to adopt priority pollutants and briefly outlines EPA's plan to federally promulgate standards for noncompliant States.*	
20. Water Quality Standards for Wetlands: National Guidance, July 1990 *Provides guidance for meeting the priority established in the FY 1991 Agency Operating Guidance to develop water quality standards for wetlands during the FY 1991-1993 triennium. By the end of FY 1993, States are required as a minimum to include wetlands in the definition of "State waters," establish beneficial uses for wetlands, adopt existing narrative and numeric criteria for wetlands, adopt narrative biological criteria for wetlands and apply antidegradation policies to wetlands.*	

STANDARDS & APPLIED SCIENCE DIVISION/WATER QUALITY STANDARDS BRANCH	
21. Reference Guide for Water Quality Standards for Indian Tribes, January 1990 *Booklet provides an overview of the water quality standards program. Publication is designed primarily for Indian Tribes that wish to qualify as States for the water quality standards program. The booklet contains program requirements and a list of reference sources.*	
22. Developing Criteria to Protect Our Nation's Waters, EPA, September 1990 (Pamphlet) *Pamphlet which briefly describes the water quality standards program and its relationship to water quality criteria, sediment criteria and biological criteria.*	
23. Water Quality Standards for the 21st Century, EPA 823-R-92-009, December 1992 *Summary of the proceedings from the Third National Conference on Water Quality Standards held in Las Vegas, Nevada, August 31-September 3, 1992*	
24. Biological Criteria: National Program Guidance for Surface Waters, EPA-440/5-90-004, April 1990 *This document provides guidance for development and implementation of narrative biological criteria.*	
25. Amendments to the Water Quality Standards Regulation that Pertain to Standards on Indian Reservations - Final Rule. Environmental Protection Agency, Federal Register, December 12, 1991 *This final rule amends the water quality standards regulation by adding: 1) procedures by which an Indian Tribe may qualify for treatment as a State for purposes of the water quality standards and 401 certification programs and 2) a mechanism to resolve unreasonable consequences that may arise when an Indian Tribe and a State adopt different water quality standards on a common body of water.*	
26. Guidance on Water Quality Standards and 401 Certification Programs Administered by Indian Tribes, December 31, 1991 *This guidance provides procedures for determining Tribal eligibility and supplements the final rule "Amendments to the Water Quality Standards Regulation that Pertain to Standards on Indian Reservations".*	
27. Water Quality Standards; Establishment of Numeric Criteria for Priority Toxic Pollutants; State's Compliance - Final Rule, Environmental Protection Agency, Federal Register, December 22, 1992 *This regulation promulgates for 14 States, the chemical specific, numeric criteria for priority toxic pollutants necessary to bring all States into compliance with the requirements of Section 303(c)(2)(B) of the Clean Water Act. Staates determined by EPA to fully comply with Section 303(c)(2)(B) requirements are not affected by this rule.*	
28. Interim Guidance on Determinations and Use of Water-Effect Ratios for Metals, EPA 823-B-94-001, February 1994 *This guidance contains specific information on procedures for developing water-effect ratios.*	

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WATERSHED MODELING SECTION TITLE	CHECK DOCUMENT REQUESTED
1. **Guidance for Water Quality-based Decisions: The TMDL Process, EPA 440/4-91-001, April 1991** *This document defines and clarifies the requirements under Section 303(d) of the Clean Water Act. Its purpose is to help State water quality program managers understand the application of total maximum daily loads (TMDLs) through an integrated, basin-wide approach to controlling point and nonpoint source pollution. The document describes the steps that are involved in identifying and prioritizing impaired waters and developing and implementing TMDLs for waters listed under Section 303(d).* **Contact: Don Brady (202) 260-5368**	
2. **Technical Guidance Manual for Performing Waste Load Allocations - Book II Streams and Rivers - Chapter 1 Biochemical Oxygen Demand/Dissolved Oxygen, EPA 440/4-84-020, September 1983** *This chapter presents the underlying technical basis for performing WLA and analysis of BOD/DO impacts. Mathematical models to calculate water quality impacts are discussed, along with data needs and data quality.* **Contact: Bryan Goodwin (202) 260-1308**	
3. **Technical Guidance Manual for Performing Waste Load Allocations - Book II Streams and Rivers - Chapter 2 Nutrient/Eutrophication Impacts, EPA 440/4-84-021, November 1983** *This chapter emphasizes the effect of photosynthetic activity stimulated by nutrient discharges on the DO of a stream or river. It is principally directed at calculating DO concentrations using simplified estimating techniques.* **Contact: Bryan Goodwin (202) 260-1308**	
4. **Technical Guidance Manual for Performing Waste Load Allocations - Book II Streams and Rivers - Chapter 3 Toxic Substances, EPA 440/4-84-022, June 1984** *This chapter describes mathematical models for predicting toxicant concentrations in rivers. It covers a range of complexities, from dilution calculations to complex, multi-dimensional, time-varying computer models. The guidance includes discussion of background information and assumptions for specifying values.* **Contact: Bryan Goodwin (202) 260-1308**	

STANDARDS & APPLIED SCIENCE DIVISION/EXPOSURE ASSESSMENT BRANCH	
WATERSHED MODELING SECTION TITLE	CHECK DOCUMENT REQUESTED
5. **Technical Guidance Manual for Performing Waste Load Allocations - Simplified Analytical Method for Determining NPDES Effluent Limitations for POTWs Discharging into Low-Flow Streams** *This document describes methods primarily intended for "desk top" WLA investigations or screening studies that use available data for streamflow, effluent flow, and water quality. It is intended for circumstances where resources for analysis and data acquisition are relatively limited.* **Contact: King Boynton (202) 260-7013**	
6. **Technical Guidance Manual for Performing Waste Load Allocations - Book IV Lakes and Impoundments - Chapter 2 Nutrient/Eutrophication Impacts, EPA 440/4-84-019, August 1983** *This chapter discusses lake eutrophication processes and some factors that influence the performance of WLA analysis and the interpretation of results. Three classes of models are discussed, along with the application of models and interpretation of resulting calculations. Finally, the document provides guidance on monitoring programs and simple statistical procedures.* **Contact: Bryan Goodwin (202) 260-1308**	
7. **Technical Guidance Manual for Performing Waste Load Allocations - Book IV Lakes, Reservoirs and Impoundments - Chapter 3 Toxic Substances Impact, EPA 440/4-87-002, December 1986** *This chapter reviews the basic principles of chemical water quality modeling frameworks. The guidance includes discussion of assumptions and limitations of such modeling frameworks, as well as the type of information required for model application. Different levels of model complexity are illustrated in step-by-step examples.* **Contact: Bryan Goodwin (202) 260-1308**	
8. **Technical Guidance Manual for Performing Waste Load Allocations - Book VI Design Conditions - Chapter 1 Stream Design Flow for Steady-State Modeling, EPA 440/4-87-004, September 1986** *Many state water quality standards (WQS) specify specific design flows. Where such design flows are not specified in WQS, this document provides a method to assist in establishing a maximum design flow for the final chronic value (FCV) of any pollutant.* **Contact: Bryan Goodwin (202) 260-1308**	
9. **Final Technical Guidance on Supplementary Stream Design Conditions for Steady State Modeling, December 1988** *WQS for many pollutants are written as a function of ambient environmental conditions, such as temperature, pH or hardness. This document provides guidance on selecting values for these parameters when performing steady-state WLAs.* **Contact: Bryan Goodwin (202) 260-1308**	
10. **Technical Guidance Manual for Performing Waste Load Allocations - Book VII: Permit Averaging, EPA 440/4-84-023, July 1984** *This document provides an innovative approach to determining which types of permit limits (daily maximum, weekly, or monthly averages) should be specified for the steady-state model output, based on the frequency of acute criteria violations.* **Contact: Bryan Goodwin (202) 260-1308**	
11. **Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water - Part I - EPA 600/6-85-022a, September 1985** *This document provides a range of analyses to be used for water quality assessment. Chapters include consideration of aquatic fate of toxic organic substances, waste loading calculations, rivers and streams, impoundments, estuaries, and groundwater.* **Contact: Bryan Goodwin (202) 260-1308**	

STANDARDS & APPLIED SCIENCE DIVISION/EXPOSURE ASSESSMENT BRANCH	
WATERSHED MODELING SECTION TITLE	CHECK DOCUMENT REQUESTED
12. **Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water - Part II - EPA 600/6-85-022b, September 1985** *This document provides a range of analyses to be used for water quality assessment. Chapters include consideration of aquatic fate of toxic organic substances, waste loading calculations, rivers and streams, impoundments, estuaries, and ground water.* **Contact: Bryan Goodwin (202) 260-1308**	
13. **Handbook - Stream Sampling for Waste Load Allocation Applications, EPA 625/6-86/013, September 1986** *This handbook provides guidance in designing stream surveys to support modeling applications for waste load allocations. It describes the data collection process for model support, and it shows how models can be used to help design stream surveys. In general, the handbook is intended to educate field personnel on the relationship between sampling and modeling requirements.* **Contact: Bryan Goodwin (202) 260-1308**	
14. **EPA's Review and Approval Procedure for State Submitted TMDLs/WLAs, March 1986** *The step-by-step procedure outlined in this guidance addresses the administrative (i.e., non-technical) aspects of developing TMDLs/WLAs and submitting them to EPA for review and approval. It includes questions and answers to focus on key issues, pertinent sections of WQM regulations and the CWA, and examples of correspondence.* **Contact: Bryan Goodwin (202) 260-1308**	
15. **Guidance for State Water Monitoring and Wasteload Allocation Programs, EPA 440/4-85-031, October 1985** *This guidance is for use by States and EPA Regions in developing annual section 106 and 205(j) work programs. The first part of the document outlines the objectives of the water monitoring program to conduct assessments and make necessary control decisions. The second part describes the process of identifying and calculating total maximum daily loads and waste load allocations for point and nonpoint sources of pollution.* **Contact: King Boynton (202) 260-7013**	
16. **Technical Guidance Manual for Performing Waste Load Allocations Book III Estuaries - Part 1 - Estuaries and Waste Load Allocation Models, EPA 823-R-92-002, May 1990** *This document provides technical information and policy guidance for preparing estuarine WLA. It summarizes the important water quality problems, estuarine characteristics, and the simulation models available for addressing these problems.* **Contact: Bryan Goodwin (202) 260-1308**	
17. **Technical Guidance Manual for Performing Waste Load Allocations Book III Estuaries - Part 2 - Application of Estuarine Waste Load Allocation Models, EPA 823-R-92-003, May 1990** *This document provides a guide to monitoring and model calibration and testing, and a case study tutorial on simulation of WLA problems in simplified estuarine systems.* **Contact: Bryan Goodwin (202) 260-1308**	

STANDARDS & APPLIED SCIENCE DIVISION/EXPOSURE ASSESSMENT BRANCH	
WATERSHED MODELING SECTION TITLE	CHECK DOCUMENT REQUESTED
18. **Technical Guidance Manual for Performing Wasteload Allocations-Book III: Estuaries - Part 3 - Use of Mixing Zone Models in Estuarine Wasteload Allocations, EPA 823-R-92-004** *This technical guidance manual describes the initial mixing wastewater in estuarine and coastal environments and mixing zone requirements. The important physical processess that govern the hydrodynamic mixing of aqueous discharges are described, followed by application of available EPA supported mixing zone models to four case study situations.* **Contact: Bryan Goodwin (202) 260-1308**	
19. **Technical Guidance Manual for Performing Wasteload Allocations - Book III - Estuaries - Part 4 - Critical Review of Coastal Embayment and Estuarine Wasteload Allocation Modeling, EPA 823-R-92-005, August 1992** *This document summarizes several historical case studies of model use in one freshwater coastal embayment and a number of estuarine discharge situations.* **Contact: Bryan Goodwin (202) 260-1308**	
20. **Technical Support Document for Water Quality-based Toxics Control, EPA 505/2-90-001, March, 1991** *This document discusses assessment approaches, water quality standards, derivation of ambient criteria, effluent characterization, human health hazard assessment, exposure assessment, permit requirements, and compliance monitoring. An example is used to illustrate the recommended procedures.* **Contact: King Boynton (202) 260-7013**	
21. **Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling (Second Edition), U.S. EPA 600/3-85/040, June 1985** *This manual serves as a reference on modeling formulations, constants and rates commonly used in surface water quality simulations. This manual also provides a range of coefficient values that can be used to perform sensitivity analyses.* **Contact: Bryan Goodwin (202) 260-1308**	
22. **Dynamic Toxics Waste Load Allocation Model (DYNTOX), User's Manual, September 13, 1985** *A user's manual which explains how to use the DYNTOX model. It is designed for use in wasteload allocation of toxic substances.* **Contact: Bryan Goodwin (202) 260-1308**	
23. **Windows Front-End to SWMM (Storm Water Management Model), EPA 823-C-94-001, February 1994** *A user interface (front-end) to the Storm Water Management Model (SWMM) and supporting documentation is avaiable on diskette. Operating in the Microsoft Windows Environment, this interface simplifies data entry and model set-up.* **Contact: Jerry LaVeck (202) 260-7771**	
24. **Windows Front-End to SWRRBWQ (Simulator for Water Resources in Rural Basins-Water Quality), EPA 823-C-94-002, February 1994** *A user interface (front-end) to the Simulator for Water Resource in Rural Basins-Water Quality model and supporting documentation is available on diskette. Operating in the Microsoft Windows environment, this interface simplifies data entry and model set-up.* Contact: Jerry LaVeck (202) 260-7771	

STANDARDS & APPLIED SCIENCE DIVISION/EXPOSURE ASSESSMENT BRANCH	
ENVIRONMENTAL ASSESSMENT SECTION TITLE	CHECK DOCUMENT REQUESTED
25. De Minimis Discharges Study: Report to Congress, U.S. EPA 440/4-91-002, November 1991 *This report to Congress addresses the requirements of Section 516 by identifying potential de minimis discharges and recommends effective and appropriate methods of regulating those discharges.* **Contact: Rich Healy (202) 260-7812**	
26. National Study of Chemical Residues in Fish. Volume I, U.S. EPA 823-R-92-008 a, September 1992 *This report contains results of a screening study of chemical residues in fish taken from polluted waters.* **Contact: Richard Healy (202) 260-7812**	
27. National Study of Chemical Residues in Fish. Volume II. U.S. EPA 823-R-92-008 b, September 1992 *This report contains results of a screening study of chemical residues in fish taken from polluted waters.* **Contact: Richard Healy (202) 260-7812**	

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DUE TO RESOURCE LIMITATIONS, ONLY ONE (1) COPY OF EACH DOCUMENT CAN BE PROVIDED TO A REQUESTOR.

SEDIMENT CONTAMINATION SECTION TITLE	CHECK DOCUMENT REQUESTED
1. **Sediment Classification Methods Compendium, U.S. EPA, EPA 823-R-92-006, September 1992** *This compendium is an "encyclopedia" of methods that are used to assess chemically contaminated sediments. It contains a description of each method, associated advantages and limitations and existing applications.* **Contact: Beverly Baker (202) 260-7037**	
2. **Managing Contaminated Sediments: EPA Decision-Making Processes, Sediment Oversight Technical Committee, U.S. EPA Report - 506/6-90/002, December, 1990** *This document identifies EPA's current decision-making process (across relevant statutes and programs) for assessing and managing contaminated sediments. Management activities relating to contaminated sediments are divided into the following six categories: finding contaminated sediments, assessment of contaminated sediments, prevention and source controls, remediation, treatment of removed sediments, and disposal of removed sediments.* **Contact: Mike Kravitz (202) 260-7049**	
3. **Contaminated Sediments: Relevant Statutes and EPA Program Activities, Sediment Oversight Technical Committee, U.S. EPA Report - 506/6-90/003, December, 1990** *This document provides information on program office activities relating to contaminated sediment issues, and the specific statutes under which these activities fall. A table containing major laws or agreements relevant to sediment quality issues is included.* **Contact: Mike Kravitz (202) 260-7049**	

STANDARDS & APPLIED SCIENCE DIVISION/RISK ASSESSMENT AND MANAGEMENT BRANCH	
SEDIMENT CONTAMINATION SECTION TITLE	**CHECK DOCUMENT REQUESTED**
4. **Contaminated Sediments News, U.S. EPA 823-N92-001** *This newsletter, issued periodically, contains information about contaminated sediment issues. Back issues of the newsletter are available.* • **Contact: Beverly Baker (202) 260-7037**	
• **Contaminated Sediments News, Number 1, August 1989**	
• **Contaminated Sediments News, Number 2, April 1990**	
• **Contaminated Sediments News, Number 3, April 1991**	
• **Contaminated Sediments News, Number 4, February 1992**	
• **Contaminated Sediments News, Number 5, April 1992**	
• **Contaminated Sediments News, Number 6, August 1992**	
• **Contaminated Sediments News, Number 7, December 1992**	
• **Contaminated Sediments News, Number 8, May 1993**	
• **Contaminated Sediment News, Number 9, August 1993**	
• **Contaminated Sediment News, Number 10, December 1993**	
5. **Proceedings of the EPA's Contaminated Sediment Management Forum, U.S. EPA, Report 823-R-92-007, September 1992** *This report summarizes the proceedings of three EPA sponsored forums designed to obtain input on EPA's Contaminated Sediment Management Strategy.* **Contact: Beverly Baker (202) 260-7037**	
6. **Selecting Remediation Techniques for Contaminated Sediment, U.S. EPA 823-B93-001, June 1993** *This planning guide assists federal-State remedial managers, local agencies, private cleanup companies and supporting contractors in remedial decision-making process at contaminated sediment sites.* **Contact: Beverly Baker (202) 260-7037**	
7. **Questions and Answers About Contaminated Sediments, U.S. EPA 823-F-93-009, May 1993** *This general pamphlet highlights what sediments are, how they are contaminated and what can be done.* **Contact: Beverly Baker (202) 260-7037**	
8. **Tiered Testing Issues for Freshwater and Marine Sediments, U.S. EPA 823-R93-001, February 1993, Proceedings of A Workshop Held in Washington, DC, September 16-18, 1992.** *This report summarizes the proceedings of the workshop sponsored by the Office of Water and Office of Research and Development. The workshop was held to provide an opportunity for experts in sediment toxicology and EPA to discuss the development of standard freshwater and marine sediment bioassay procedures.* **Contact: Thomas Armitage (202) 260-5388**	

STANDARDS & APPLIED SCIENCE DIVISION/RISK ASSESSMENT AND MANAGEMENT BRANCH	
FISH CONTAMINATION SECTION TITLE	**CHECK DOCUMENT REQUESTED**
9. Special Interest Group (SIG) Forum for Fish Consumption, User's Manual, V.1.0., U.S. EPA 822/8-91/001, February 1992 *This user's manual describes various features of the Special Interest Group (SIG) Forum for fish consumption advisotries, bans and risk management. The manual explains how to access the SIG and use its data bases, messags, bulletins and other computer files.* **Contact: Jeff Bigler (202) 260-1305**	
10. Consumption Surveys for Fish and Shellfish, A Review and Analysis of Survey Methods, U.S. EPA-822/R-92-001, February 1992. *This document contains a critical analysis of methods used to determine fish consumption rates of recreational and subsistence fisherment, groups that have the greates potential for exposure to contaminants in fish tissues.* **Contact: Jeff Bigler (202) 260-1305**	
11. Proceedings of the U.S. Environmental Protection Agency's National Technical Workshop "PCBs in Fish Tissue", U.S. EPA/823-R-93-003, September 1993 *This documents summarizes the proceedings of the EPA sponsored workshop held on May 10-11, 1993 in Washington, DC.* **Contact: Rick Hoffman (202) 260-0642**	
12. Guidance for Assessing Chemical Contaminant Data for Use in Risk Advisories, Volume 1: Fish Sampling and Analysis, EPA 823-R-93-002, August 1993 *This document provides detailed technical guidance on methods for sampling and analyzing chemical contaminants in fish and shellfish tissues. It addresses monitoring strategies, selection of fish species and chemical analytes, field and laboratory procedures and data analyses.* **Contact: Jeff Bigler (202) 260-1305**	
13. National Fish Tissue Data Repository User Manual, Version 1.0, EPA 823-B-903-003, November 1993 *The U.S. EPA has developed the National Fish Tissue Data Repository (NFTDR) for collection and storage of fish and shellfish contaminants data. The data repository is part of a large EPA data base system called the Ocean Data Evaluation System (ODES). This manual explains how to access information from the ODES database.* **Contact: Rick Hoffman (202) 260-0642**	
14. National Fish Tissue Data Repository: Data Entry Guide, Version 1.0, EPA 823-B-93-006, November 1993 *The U.S. EPA has developed the National Fish Tissue Data Repository (NFTDR) for collection and storage of fish and shellfish contaminants data. The data repository is part of a larger EPA data base system known as the Ocean Data Evaluation System (ODES). This manual assists State and Federal Agencies in submitting data to the NFTDR.* **Contact: Rick Hoffman (202) 260-0642**	

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APPENDIX W

Update Request Form for
Water Quality Standards Handbook
Second Edition

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APPENDIX X

Summary of Updates

APPENDIX X

WATER QUALITY STANDARDS HANDBOOK

SECOND EDITION