United States
Environmental Protection
Agency

Office of Water
Regulations and Standards
Washington, DC 20460

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Water

Water Quality Standards Handbook

FOREWORD

The Water Quality Standards Handbook contains the guidance prepared by EPA to assist States in implementing the revised Water Quality Standards Regulation (48 F.R. 51400, November 8, 1983). Changes in this Handbook may be made from time to time reflecting State/EPA experience in implementing the revised Regulation. The Handbook is organized to provide a general description of the overall standards setting process followed by information on general program administrative policies and procedures, and then a description of the analyses used in determining appropriate uses and criteria.

The Clean Water Act established two types of regulatory requirements to control pollutant discharges: technology-based effluent limitations which reflect the best controls available considering the technical and economic achievability of those controls; and water quality-based effluent limitations which reflect the water quality standards and allowable pollutant loadings set by the States (with EPA oversight).

Technology-based requirements for dischargers are currently being issued. However, in some cases these controls will not be sufficient to eliminate water quality impacts and enable water quality standards to be met. In these cases, water quality-based controls are needed. Two technical approaches are available for developing WQ-based effluent limits, the pollutant-specific approach and the biomonitoring approach. Pollutant-specific techniques are best used where discharges contain a few, well-quantified pollutants and the interactions and effects of the pollutants are known. In addition, pollutant-specific techniques should be used where health hazards are a concern or bioaccumulation is suspected.

It may be difficult, however, in some situations to determine attainment or nonattainment of water quality standards and set appropriate limits because of complex chemical interactions which affect the fate and ultimate impact of toxic substances in the receiving water. In many cases, all potentially toxic pollutants cannot be identified by chemical methods. Also, developing numerical water quality criteria and determining allowable loadings for all of the wide variety of pollutants found in effluents would be very time-consuming and resource intensive. In such situations, it is more feasible to examine overall toxicity and instream impacts using biological methods rather than attempting to identify all toxic pollutants, determining the effects of each pollutant individually, and then attempting to assess their collective effect.

Therefore EPA has developed a two-fold approach to toxics control. In certain situations we must still rely upon the chemical-specific approach, measuring individual toxicants and evaluating their specific toxic properties. In other situations, especially where complex effluents are involved, it is more appropriate to examine the harmful effects of toxicity of the whole effluent rather than attempt to identify individual toxicants and understand their chemical interaction. This second approach relies on newly-developed biological monitoring methods and laboratory testing procedures.

We expect to continue to build the necessary EPA expertise in the area of biomonitoring and work together with other interested groups over the next several years to develop a balanced and integrated biological/chemical-specific approach to developing realistic water quality-based permit limitations.

The purpose of this guidance document is to illustrate the types of scientific and technical data and analyses EPA believes are necessary to be conducted so that the public and EPA can review decisions on water quality standards affecting water quality limited segments, i.e. those water bodies where standards cannot be met even with the implementation of the technology-based controls required by the Act (secondary treatment for municipalities and best available/best conventional treatment for industries).

When a State conducts use attainability analyses or establishes appropriate criteria, EPA is not requiring that specific approaches, methods or procedures be used. Rather, States are encouraged to consult with EPA early in the process to agree on appropriate methods before the analyses are initiated and carried out. States will have the flexibility of tailoring the analyses to the specific water body being examined as long as the methods used are scientifically and technically sound.

State pollution control agencies are encouraged to solicit the assistance of other State agencies, municipalities, industry, environmental groups, and the community-at-large in collecting the data for the analyses. By carefully outlining quality assurance/quality control procedures States can assure the integrity and validity of the data for the analyses, while easing the resource burdens.

A State must conduct and submit to EPA a use attainability analysis where the State designates or has designated uses that do not include the uses specified in Section 101(a)(2) of the Act, or when a State wishes to remove a designated use that is specified in the goals or to adopt subcategories of uses requiring less stringent criteria. A State must adopt criteria sufficient to protect the designated uses. In adopting criteria, States may use Section 304(a) criteria or set site-specific criteria. Analyses conducted in support of revisions to standards are subject to EPA review.

A use attainability analysis is a multi-step scientific assessment of the physical, chemical, biological and economic factors affecting the attainment of a use. In preparing a use attainability analyses, a water body survey and assessment is conducted to examine the physical, chemical and biological characteristics of the water body. This assessment identifies and defines the existing uses of that water body, determines whether the designated uses are impaired, and the reasons for the impairment. By comparing the water body with one that is not impaired by man-induced pollution and with similar physical characteristics, the assessment assists States in projecting the potential uses that the water body could support in the absence of pollution. The next step in a use attainability analysis is a waste load allocation which utilizes mathematical models to predict the amount of reduction in pollutant loadings necessary to achieve the designated use. After determining the technology needed to meet these effluent reductions, an economic assessment may be conducted to determine whether requiring more stringent technology than that mandated by the Act will cause widespread and substantial economic and social impact.

A State may adopt EPA recommended criteria without any analysis or justification. However, EPA's laboratory-derived criteria may not always accurately reflect the toxicity of a pollutant in a particular water body because of differences in temperature, pH, etc. A State may choose to set site-specific criteria based on characteristics of the local water body. Setting site-specific criteria is also appropriate in water bodies with different species than those used in the derivation of the Section 304(a) criteria or where adaptive processes have enabled a viable, balanced community to exist with levels of pollutants that exceed the national criteria.

Any questions on this guidance may be directed to the water quality standards coordinators located in each of the EPA regional offices or to:

David Sabock
U.S. Environmental Protection Agency
Chief, Criteria Branch (WH-585)
401 M Street, S.W.
Washington, D.C. 20460
(Telephone 202-245-3042)

EPA sincerely appreciates the efforts of the many people and organizations who participated in the public review of the water quality standards program regulation and guidance since their proposal in October 1982.

Steven Schatzow

Steven Schatzow, Director
Water Regulations and Standards

WATER QUALITY STANDARDS HANDBOOK

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CHAPTER 1

WATER QUALITY STANDARDS REVIEW AND REVISION PROCESS

Introduction

The Clean Water Act requires that a 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. The Water Quality Standards Regulation also requires that any water body with standards not consistent with the Section 101(a)(2) goals of the Act must be reexamined every three years to determine if new information has become available that would warrant a revision of the standard.

The Regulation allows States to establish procedures for identifying and reviewing the standards on specific water bodies in detail.1/ Water bodies receiving a detailed standards review are most likely to be those where advanced treatment (AT) or combined sewer overflow (CSO) funding decisions are pending, water quality based permits are scheduled to be issued or reissued, or toxics have been identified or are suspected of precluding a use, or may be posing an unreasonable risk to human health. States may have other reasons for wishing to examine a water body in detail.

In selecting specific areas, States should also take into account the "Municipal Wastewater Treatment Construction Grant Amendments of 1981" (P.L. 97-117, December 29, 1981). EPA interprets Section 24 of the Amendments as requiring States to assure that water quality standards influencing construction grant decisions have been reviewed in accordance with Section 303(c) of the Clean Water Act. Section 24 prohibits the issuance of a construction grant after December 1984, unless the State has completed its review of the water quality standard for any segments affected by the project grant (see Construction Grants Program Interim Final Rule 40 CFR 35.2111, 47 FR 20450, May 12, 1982). Additional guidance regarding Section 24 and standards reviews is contained on page 2-3 of this Handbook.

The water quality standards review process described in this Chapter focuses on the analyses used in reviewing standards on water quality limited segments, e.g. those standards which cannot be attained even with the application of the technology-based controls required by the Act.

In reviewing the standards on water quality limited segments, States must perform and submit to EPA a use attainability analysis if the State designates or has designated uses that do not include the uses specified in Section 101(a)(2) of the Act, or the State wishes to remove a designated use that is specified in Section 101(a)(2), or to

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1/ Any procedures States establish to revise standards should be articulated in the Continuing Planning Process document consistent with the Water Quality Management Regulation.

adopt subcategories of uses specified in Section 101(a)(2) which require less stringent criteria than are currently adopted. States may adopt seasonal uses as an alternative to reclassifying a water body or segment thereof to uses requiring less stringent criteria.

States may designate uses which do not reflect the goals of the Act if supported by a use attainability analysis based on one or more of the six factors listed in Section 131.10(g) of the Regulation. In no case can a State downgrade an existing use. No use attainability analysis is required when designating uses which include those specified in Section 101(a)(2) of the Act.1/

States must adopt water quality criteria sufficient to protect the designated use. The criteria adopted must provide sufficient parametric coverage and must be of adequate stringency to protect designated uses. Numerical criteria may be based on criteria recommendations published by EPA or developed by other scientifically defensible methods. States may also modify Section 304(a) criteria and set site-specific criteria where (1) background water quality parameters, such as pH, hardness, temperature, color, etc., appear to differ significantly from the laboratory water used in developing Section 304(a) criteria; or (2) the types of local aquatic organisms in the region differ significantly from those actually tested in the development of the 304(a) criteria or have adapted to higher pollutant levels. EPA believes that setting site-specific criteria will occur on only a limited number of stream segments because of the resources required to conduct the analyses and the basic soundness of the Section 304(a) recommendations. States may also establish narrative criteria based upon biomonitoring methods where numerical criteria cannot be established or to supplement numerical criteria. The revised water quality standards regulation provides increased emphasis on the need for adoption by the States of criteria for toxic pollutants applicable to a water body sufficient to protect designated uses.

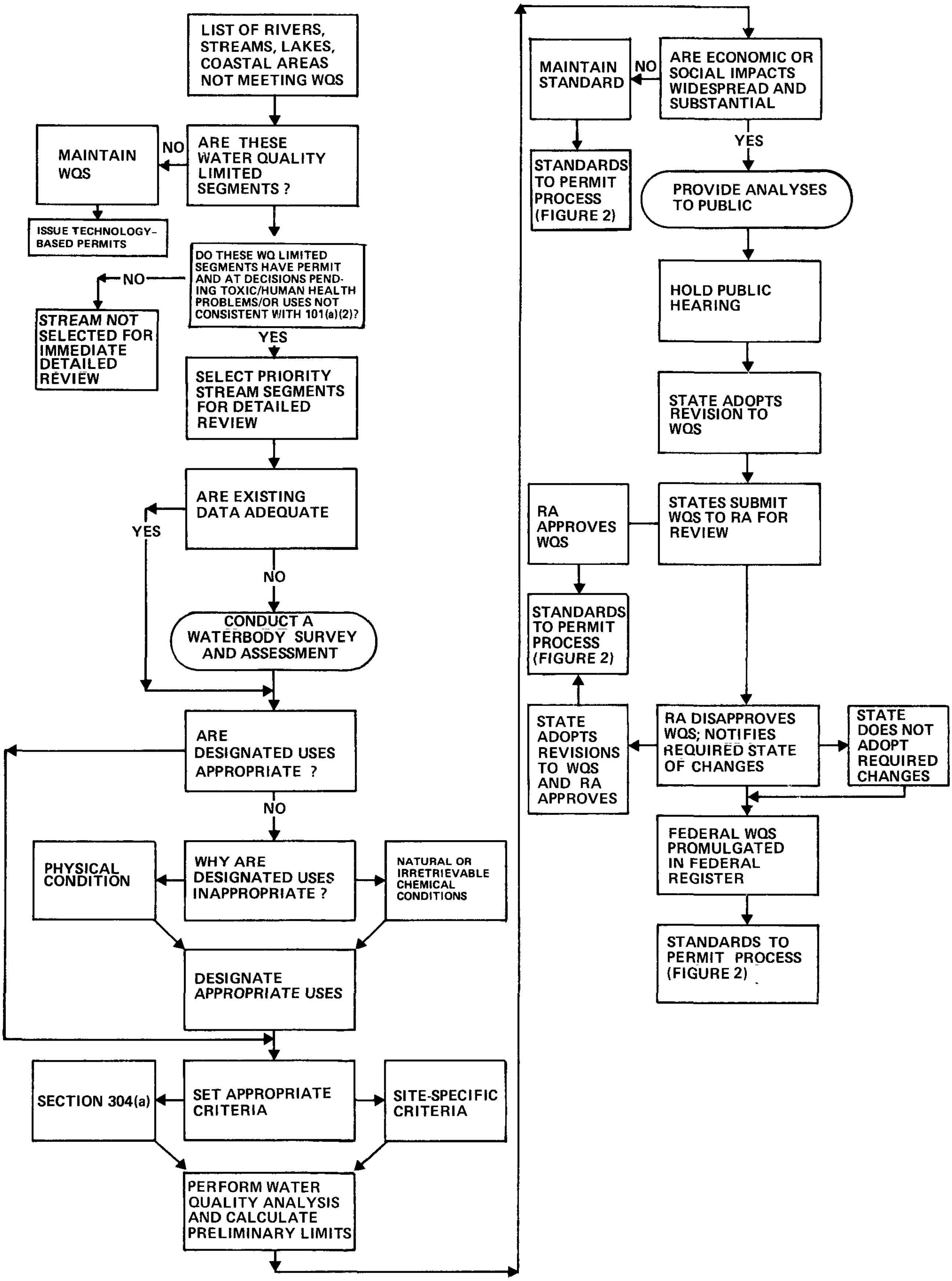
State standards must also contain an antidegradation policy designed to maintain and protect existing uses and water quality, to provide protection for higher quality waters, and to provide protection for outstanding national resource waters.

Before starting the analyses described in this Handbook, the State should agree with EPA on the approach to be used, availability of existing data, scheduling, quality control and assistance procedures, etc. In many instances, EPA may be able to assist. States should also work with municipal and industrial dischargers and other appropriate organizations to enlist their assistance in gathering the data and conducting the analyses to reduce the resource impacts of the analyses on the State.

In the remaining portion of this Chapter, the Steps of the water quality standards review and revision process are described. The steps are outlined in Figure 1.

1/ NOTE: A use attainability analysis may be required to support construction grant funding requests for advanced treatment in publicly owned treatment works.

FIGURE 1
WQS REVIEW AND REVISION PROCESS



List of Rivers, Streams, Lakes, Coastal Areas Not Meeting WQS

States know the location of their water pollution problems and frequently list the segments in order of priority in State water quality reports issued biennially under Section 305(b). Water quality problems are most frequently expressed in terms of impacts on the biota of the water body, restricted beneficial uses, and the extent and frequency of water quality criteria violations.

Select Priority Water Quality Limited Stream Segments for Detailed Water Quality Standards Review

Water quality standards should be revised only where a need exists, given the limited resources available. Section 303(d) of the Act requires States to identify those waters which cannot meet water quality standards with effluent limitations required by Section 301(b)(1) and (2) and to establish a priority ranking for those waters.

EPA recommends that States select for standards review those water quality limited segments on which there are advanced treatment (AT) and combined sewer overflow (CSO) funding decisions pending, major permit revisions are scheduled or toxics have been identified or are suspected of precluding a use. States may select other criteria for determining which segments will be reviewed, such as human health problems, court orders, or costs or economic and social impacts of implementing the existing water quality standards. Any water body with standards not consistent with the Section 101(a)(2) goals of the Act must be reexamined every three years.

States are encouraged to review standards for a large enough area to consider the interaction between both point and nonpoint source discharges. In carrying out standards reviews, the State and EPA should ensure proper coordination of all water quality programs.

Water Body Survey and Assessments

An intensive survey of the water body is not necessary if adequate data are available. The purpose of a survey is to pinpoint problems and to characterize present uses, uses impaired or precluded, and the reasons why uses are impaired or precluded. Information generated in the survey also can be used to establish the basis for seasonal uses and subcategories of uses.

Included in Chapter 3 are examples of a full range of physical, chemical, and biological characteristics of the water body which, depending on the site, may be surveyed when evaluating aquatic protection uses. This information is then used in determining existing species in the water body, the health of those species as well as what species could be in the water body given the physical characteristics of the water body or might be in the water if the quality of the water were improved.

If the results of the survey show that the water body is, in fact, being used for the designated purposes and the biology of the water body is healthy, although monitoring data show criteria continue to be exceeded, EPA recommends that the State adopt appropriate criteria using Section 304(a) criteria, one of the protocols included in Chapter 4, or other scientifically defensible methods.

Review the Cause of Uses Not Being Met

If the survey indicates that designated uses are impaired, the next step is to determine the cause. In many situations, both physical conditions and the presence of water pollutants prevent the water body from meeting its designated use. Physical limitations refer to such factors as depth, flow, habitat, turbulence or structures such as dams which may make a use unsuitable or impossible to achieve regardless of water quality.

If uses are precluded because of physical limitations of the water body, the State may wish to examine modifications which might allow a habitat suitable for a species to thrive where it could not before. Some of the techniques which have been used include: bank stabilization, current deflectors, construction of oxbows or installation of spawning beds. A State might also wish to consider improving the access to the water body or improving facilities nearby so that it can be used for recreational purposes. A State may also consider establishing seasonal uses or subcategories of a use.

If uses are not being met because of water pollution problems, the first step in the process is to determine the cause. If the standards review process is well coordinated with the total maximum daily load process and the permit process, some of the analysis necessary to determine why uses are not attained may be collected by permittees through permit modification or requests for information under section 308. When background levels of pollutants, whether natural or man-induced, are irretrievable and criteria cannot be met, States should evaluate other more appropriate uses for the water body and revise the water quality standards appropriately.

Determine Attainable Uses

Consideration of the suitability of the water body to attain a use is an integral part of the water quality standards review and revision process. The data and information collected from the water body survey provide a firm basis for evaluating whether the water body is suitable for the particular use. Suitability depends on the physical, chemical, and biological characteristics of the water body, its geographic setting and scenic qualities and the socio-economic and cultural characteristics of the surrounding area. It is not envisioned that each water body would necessarily have to have a unique set of uses. Rather the characteristics necessary to support a use can be identified so that water bodies having those characteristics might be grouped together as supporting particular uses.

Suitability, to a great extent, depends on the professional judgment of the evaluators. It is their task to provide sufficient information to the public and the State decision-makers to base a decision. There are instances where non-water quality related factors preclude the attainment of uses regardless of improvements in water quality. This is particularly true for fish and wildlife protection uses where the lack of a proper substrate may preclude certain forms of aquatic life from using the stream for propagation, or the lack of cover, depth, flow, pools, riffles or impacts from channelization, dams, diversions may preclude particular forms of aquatic life from the stream altogether. While physical factors do affect the recreational uses appropriately designated for a water body, States need to give consideration to the incidental uses which may be made of the water body. Even though it may not make sense to encourage use of a stream for swimming because of the flow, depth or the velocity of the water, the States and EPA must recognize that swimming and/or wading may occur anyway. In order to protect public health, States must set criteria to reflect swimming if it appears that primary contact recreation will in fact occur in the stream. While physical factors are important in evaluating whether a use is attainable, physical limitations of the stream may not be an overriding factor. Common sense and good judgment play an important role in setting appropriate uses and criteria. In setting criteria and uses, States must assure the attainment of downstream standards. The downstream uses may not be affected by the same physical limitations as the upstream uses.

Criteria may reflect either primary or secondary contact recreation depending on which is expected to occur, with flow being a consideration in determining which recreational use is protected. Where extremely low flow conditions exist, the State and EPA must be sure that primary contact recreation does not occur in stream pools before adopting the less stringent criteria for protecting secondary contact recreation. (Of, course, if the "existing use" is a recreational use, then both that use and the criteria to protect it must be reflected in the standard. Common sense must be used in deciding whether a use is sufficiently likely to be a "existing use" rather than merely incidental, or, indeed whether it will occur at all.)

The rationale offered by a State for not designating a stream for either primary or secondary contact recreation must be of sufficient detail to indicate that the State has considered the conditions in a particular water body or water bodies rather than a simple blanket Statewide exception. Water bodies, with specific and limited exceptions, should be suitable for human use in recreation activities not involving significant risks of ingestion without reference to official designation of recreation as a water use.

The basis of this policy is that the States and EPA have an obligation to do as much as possible to protect the health of the public even though it may not make sense to encourage use of a stream for swimming or wading because of physical conditions. In certain instances, particulary urban areas, people will use whatever water bodies are available for recreation.

Set Appropriate Criteria

Regardless of whether changes or modifications in uses are made, criteria protective of the use must be adopted. States may use EPA's Section 304(a) criteria or set site-specific criteria. EPA's laboratory-derived criteria may not always accurately reflect the bioavailability and/or toxicity of a pollutant because of the effect of local physical and chemical characteristics or varying sensitivities of local aquatic communities. Similarly, certain compounds may be more or less toxic in some waters because of differences in temperature, hardness, or other conditions. Setting site specific criteria is appropriate where:

- ° background water quality parameters, such as pH, hardness, temperature, color, etc., appear to differ significantly from the laboratory water used in developing the Section 304(a) criteria; or

- ° the types of local aquatic organisms differ significantly from those actually tested in developing the Section 304(a) criteria.

Developing site-specific criteria is a method of taking local conditions into account so that criteria are adequate to protect the designated use without being more or less stringent than needed. A three phase testing program which includes water quality sampling and analysis, a biological survey, and acute bioassays provides an approach for developing site-specific criteria. Much of the data and information for the water quality sampling and analysis and the biological survey can be obtained while conducting the assessment of the water body. Included in Chapter 4 are scientifically acceptable procedures for setting site-specific pollutant concentrations that will protect designated uses.

Perform Water Quality Analysis and Calculate Preliminary Limits

When the technology-based limitations are insufficient to protect the designated uses, the Clean Water Act requires the development of more stringent limitations to maintain the water quality standards (see §301(b)(1)(C)). EPA encourages States to review in detail those segments where more stringent effluent limitations are necessary to meet water quality standards. More stringent limitations are generally developed as part of the total maximum daily load and wasteload allocation processes required under Sections 303(d) and 303(e)(3)(A) of the Act. These sections require States to identify waters requiring more stringent effluent limitations, set priorities for calculating total maximum daily loads and submit the above to the Administrator for approval. Total maximum daily loads of pollutants are calculated so as to meet water quality standards. A wasteload allocation involves: (1) identifying the pollutant sources and their loadings, (2) applying mathematical models and other techniques that predict the amount of load reduction necessary to achieve the water quality standards, and (3) allocating the necessary load reduction among the pollution sources.

Although not included in this document, guidance is available on performing waste load allocations.1/ Again, the water body survey provides much of the data to determine the total maximum daily load and waste load allocation.

In addition to examining more stringent technology-based controls the State should also consider establishing or improving best management practices for the control of pollution from nonpoint sources. Existing BMPs and related control programs should be reviewed to determine if they constitute the most effective way of meeting standards or if revised nonpoint source controls need to be implemented.

Economic Impact Assessment

The Regulation allows States to establish uses that are inconsistent with the Section 101(a)(2) goals of the Act if the more stringent technology to meet the goals will cause substantial and widespread economic and social impact. These are impacts resulting specifically from imposition of the pollution controls and reflect such factors as unemployment, plant closures, changes in the governmental fiscal base, and other factors (see page 2-11 of this Handbook). The analysis should address the incremental effects of water quality standards beyond technology-based or other State requirements. If the requirements are not demonstrated to have an incremental substantial and widespread impact on the affected community, the standard must be maintained or made compatible with the goals of the Act.

Revise Water Quality Standards

If a change in the designated use is warranted based on a use attainability analysis, States may modify the uses now assigned. In doing so, the State should designate uses which can be supported given the physical, chemical, or biological limitations of the water body. Or, a State may designate uses on a seasonal basis. Seasonal use designations may be appropriate for streams that lack adequate water volume to support aquatic life year-round, but can be used for fish spawning, etc. during higher flow periods. In setting seasonal uses, care must be taken not to allow the creation of conditions instream that preclude uses in another season. EPA encourages the designation of seasonal uses as an alternative to completely downgrading the use of a water body.

Change in use designations must also be accompanied by consideration of the need for a change in criteria. If a use is removed, the criteria to protect that use may be deleted or revised to

1/ U.S. Environmental Protection Agency. Technical Guidance Manuals for Performing Wasteload Allocations. Wasteload Allocation Section, (Phone 202-382-7056) Monitoring and Data Support Division (WH-553), 401 M St., S.W., Washington, D.C. 20460, 1983.

assure protection of the remaining uses. If a use is added, there must be adequate water quality criteria to protect the use. Existing criteria may be adequate or new criteria may have to be adopted.

As an alternative to downgrading standards a State may wish to include a variance as part of a water quality standard rather than change the standard across-the-board because the State believes that the standard ultimately can be attained. By maintaining the standard rather than changing it, the State will assure further progress is made in improving water quality and attaining the standard. EPA has approved State-adopted variances in the past and will continue to do so if: the variance is included as part of the water quality standard, it is subjected to the same public review as other changes in water quality standards, and if the variance is granted based on a demonstration that meeting the standard would cause substantial and widespread economic and social impact, the same test as if the State were removing a designated use. A variance may be granted to an individual discharger. However, the determining factor is whether the economic impact on the discharger is sufficient to have a substantial and widespread impact on the affected community and not just on the discharger. Such a variance controls the permit limits for the discharger that received the variance. With the variance provision, NPDES permits may be written such that reasonable progress is made toward attaining the general standard without violating Section 402(a)(1) of the Act which states that NPDES permits must meet the applicable water quality standards. (A word of caution is necessary. The term "variance", if it is used at all in a State's standards, is not always defined consistently from State to State. Therefore, some State "variance" policies and procedures may not be consistent with the standards regulation but, for example, an "exception" policy might be). Office of General Counsel opinion 58, March 29, 1977, provides guidance on the legal basis for granting variances.

Public Hearing

Prior to removing or modifying any use or changing criteria, the Clean Water Act requires the State to hold a public hearing (see Chapter 2). More than one hearing may be required depending on State regulations. It may be appropriate to have EPA review the adequacy of justifications including the data and the suitability and appropriateness of the analyses and how the analyses were applied prior to the public hearing. In cases where the analyses are judged to be inadequate, EPA will identify how the analyses could be improved and suggest the additional types of evaluations or data needed. By consulting with EPA frequently throughout the review process, States can be better assured that EPA will be able to expeditiously review State submissions and make the determination that the standards meet the requirements of the Act.

The analyses and supporting documentation prepared in conjunction with the proposed water quality standards revision should be made available to the interested public prior to the hearing. Open discussion of the scientific evidence and analysis supporting proposed revisions in the water quality standards will assist the State in making its decision.

EPA Review

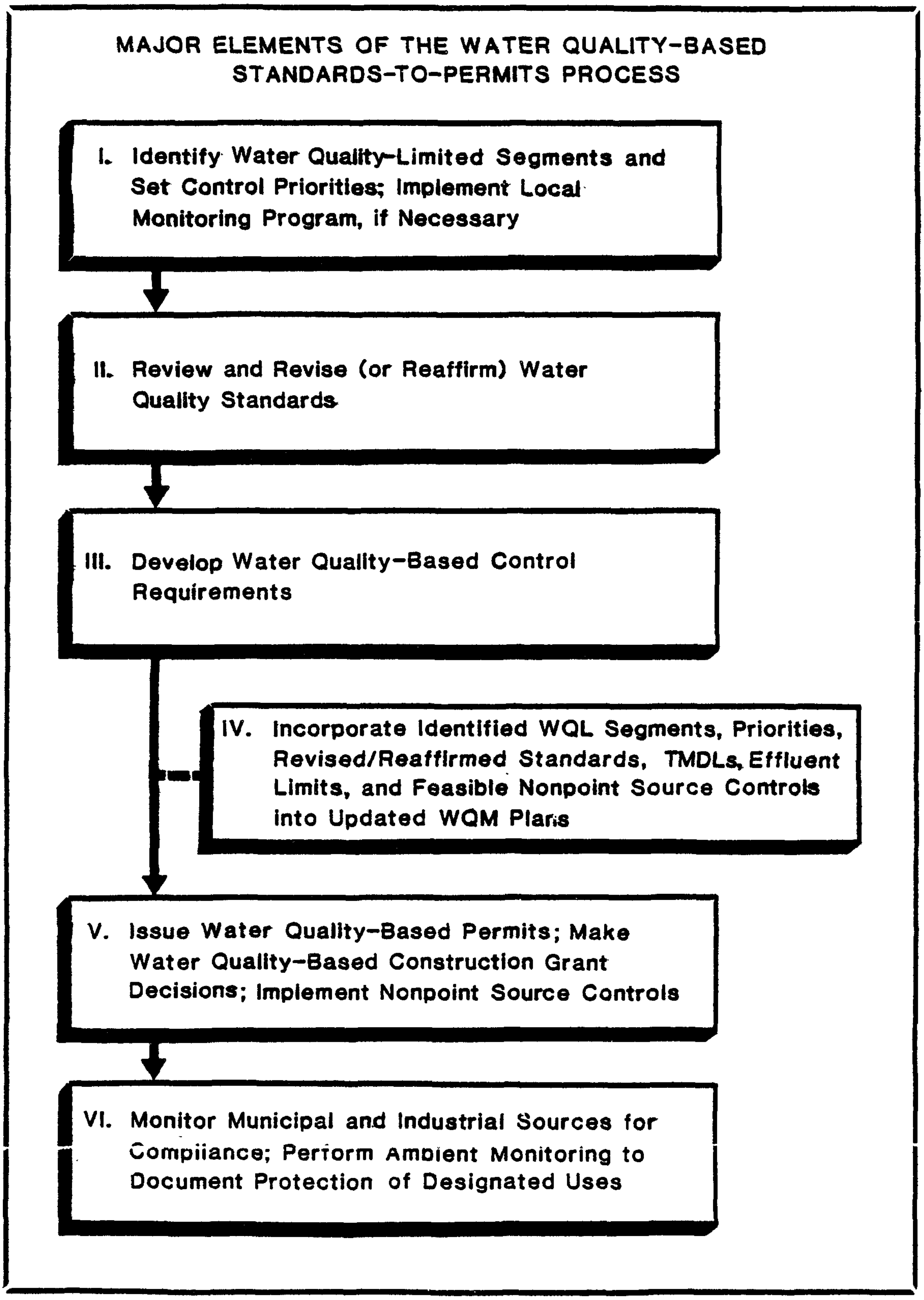
States are to submit their revised water quality standards and supporting analyses to EPA within 30 days of their final administrative action. Final administrative action is meant to be the last action a State must take before its revision becomes a rule under State law and it can officially transmit State-adopted standards to EPA for review. This last action might be a signature, a review by a legislative committee or State Board, or a delay mandated by a State administrative procedures act. In reviewing changes in uses that are inconsistent with the Section 101(a)(2) goals of the Act or changes in criteria, EPA will carefully consider the adequacy of the analyses and the public comments received during the hearing process. Standards are to meet the goals of the Act unless the State can clearly demonstrate that the uses reflected in the goals are unattainable.

Standards to Permit Process

Based on a new or revised water quality standard, a wasteload allocation analysis is conducted, as described earlier, to determine the load reduction necessary to achieve the standard. The results of the wasteload allocation analysis are adopted into the water quality management plan for the stream, and are included as enforceable effluent limits in permits issued to dischargers. These permits are part of the National Pollutant Discharge Elimination System (NPDES) and are the legal basis for requiring dischargers to control the pollutant levels in their effluents.

Figure 2 illustrates the steps involved in moving from a water quality standard to the issuance of a permit reflecting that standard. Details on these activities are beyond the purview of the standards program and this guidance. Permits are issued based on the level of discharge necessary to meet the standard. Dischargers are monitored to determine whether they are meeting their permit conditions and to ensure expected water quality improvements are achieved.

FIGURE 2


MAJOR ELEMENTS OF THE WATER QUALITY-BASED STANDARDS-TO-PERMITS PROCESS
I. Identify Water Quality-Limited Segments and Set Control Priorities; Implement Local Monitoring Program, if Necessary
II. Review and Revise (or Reaffirm) Water Quality Standards
III. Develop Water Quality-Based Control Requirements
IV. Incorporate Identified WQL Segments, Priorities, Revised/Reaffirmed Standards, TMDLs, Effluent Limits, and Feasible Nonpoint Source Controls into Updated WQM Plans
V. Issue Water Quality-Based Permits; Make Water Quality-Based Construction Grant Decisions; Implement Nonpoint Source Controls
VI. Monitor Municipal and Industrial Sources for Compliance; Perform Ambient Monitoring to Document Protection of Designated Uses

CHAPTER 2

GENERAL PROGRAM GUIDANCE

Contents

Chapter 2

GENERAL PROGRAM GUIDANCE

EPA REVIEW, APPROVAL, DISAPPROVAL AND PROMULGATION PROCEDURES

Introduction

Section 303(c) of the Clean Water Act provides the basis for EPA review and approval of State adopted or revised water quality standards. It requires States to hold hearings to review these standards at least once every three years, and to revise standards where necessary; it establishes time limits for various State and federal actions; and it provides a mechanism for Federal promulgation if the State's action is inconsistent with the requirements of the Act.

EPA's revised water quality standards regulation places greater emphasis on the adoption of criteria for toxic pollutants necessary to protect designated uses, requires States to periodically review any standards not consistent with the goals of the Act, allows States to justify standards other than those specified in the goals of the Act through an analysis of the physical, chemical, biological, and economic factors involved, and to develop site-specific water quality criteria.

The revised water quality standards regulation became effective on December 8, 1983. Properly implementing the requirements of the regulation will require extensive cooperation between EPA and the States along with a good deal of common sense. This is because each State administers its standards program differently, therefore, at any point in time each State will be at a different stage in its standards review. Also, it may require several years for the State to develop an adequate response to the requirements of the regulation. EPA and the States should identify areas where changes may be necessary to meet the requirements of the new regulation and establish a schedule for making the changes as soon as possible.

EPA assistance will include meeting with State officials before WQS revisions are initiated to mutually agree upon what standards and water bodies will be reviewed. This agreement will outline the extent and detail of analyses needed to support any changes in the standards, how the analyses will be conducted, who might be participating in the analyses, the sources of existing data and information, and a schedule for completion of the analyses. EPA will assist in the analyses and recommend approaches where needed and requested by the State. The objective is to develop a close working partnership between the States and EPA and to assure the involvement of locally affected parties. Local involvement should assist in developing the acceptance and commitment to achieve the standards. Also, it will assist EPA in its review of State water quality standards and lessen the possibility that EPA will question or disapprove formally adopted standards.

Components of a Water Quality Standards Submission

The Governor, or his designee, should submit the results of the review and any adopted revisions to State water quality standards to the Regional Administrator. The submittal should include the following information:

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

(b) The 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 40 CFR 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 and enforceable pursuant to State law, and

(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.

NOTE: The Administrator or Regional Administrator may also request additional information from the State as an aid in reviewing the adequacy of the State-adopted standards.

EPA's Review of State Water Quality Standards

EPA will review State water quality standards to ensure that the standards meet the requirements of the Act. EPA will review the adequacy of the analyses in support of any changes in the standards. Where the analyses are inadequate, EPA will identify how the analyses need to be improved and will suggest the type of information or analyses needed.

EPA will also be looking at whether uses and/or criteria are consistent throughout the water body and that downstream standards are protected. A review to determine compliance with downstream standards is most likely to involve bodies of water on or crossing interstate and international boundaries.

Timing of State Water Quality Standards Submission

Section 303(c) of the Act requires States to review their standards at least once every three years and modify or adopt standards as appropriate. EPA's regulation interprets that to include the requirement that any water body 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 to cause a revision in the standards. Procedures for identifying and reviewing the standards of specific water bodies in greater detail may be established by the States and identified in the Continuing Planning Process Document.

States may review all or some of their standards more often than once every three years. For purposes of the Act and the regulation, EPA will measure the 3-year review from the date of the letter in which the State informs EPA that revised or new standards have been adopted for the affected waters and are being submitted for EPA review or, if no changes were made in the standards for those waters, from the date of the letter in which the State informs EPA that the standards were reviewed and no changes were made.

Under the "Municipal Wastewater Treatment Construction Grant Amendments of 1981" (§24 of P.L. 97-117 33. U.S.C. 1313a) after 1984, EPA may make a construction grant only where a State has reviewed the water quality standards for the segment affected by the project. Section 24 is no more than a reconfirmation of the requirements of Section 303(c) of the Act and a mechanism to ensure that water quality standards influencing construction grant decisions have in fact been reviewed in accordance with Section 303(c) of the Act. Water quality standards reviews for water body segments involving prospective AT projects should take into account the Agency's policy and technical procedures for review of and funding decisions for such projects.

Policies and Procedures Related to Approvals

Revisions to State water quality standards that meet the requirements of the Act are approved by the appropriate EPA Regional Administrator. The Regional Administrator must within sixty days notify the Governor or his designee by letter of the approval and forward a copy of the letter to the appropriate State agency. The letter should contain any information which may be helpful in understanding the scope of the approval action. If particular events could result in a failure of the approved standards to continue to meet the requirements of the Act, these events should be identified in the approval letter to facilitate future review/revision activities.

When only a portion of the revisions submitted meet the requirements of the Act, the Regional Administrator may only approve that portion. If only a partial approval is made, the Region should, for the revisions which do not meet the requirements of the Act, administer the State notification as a disapproval action.

Policies and Procedures Related to Disapprovals

If the Regional Administrator determines that the revisions submitted are not consistent with or do not meet the requirements of the Act, the Regional Administrator must disapprove such standards within ninety days. Such disapproval is by written notification to the Governor of the State or his designee. The letter must state why the revisions are not consistent with the Act and specify the revisions which must be adopted to obtain full approval. The letter must also notify the Governor that the Administrator will initiate promulgation proceedings if the State fails to adopt and submit the necessary revisions within 90 days after notification.

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.

Policies and Procedures Related to Promulgations

If the State fails to appropriately amend its standards during the 90 day period following the notification of disapproval, the Administrator is required to promptly publish proposed revisions to the State standards in the Federal Register. Generally, a public hearing will be held on the proposed standards. Final standards are promulgated after giving due consideration to written comments received and statements made at any public hearings on the proposed revisions.

Although only the Administrator may promulgate State standards, the Regional Office has a major role in the promulgation process. The Regional Office provides the necessary background information and conducts the public hearings. The Regions are encouraged to prepare drafts of the rationale supporting EPA's action included in the proposed and final rulemakings. The rationale should clearly state the reason for the disapproval of the State standard. The documentation should be forwarded to the Director, Criteria and Standards Division (WH-585).

If a State remedies the deficiencies in its water quality standards prior to promulgation, the Administrator will terminate the rulemaking proceedings. However, if a proposed rulemaking has been published in the Federal Register, then the Regional Administrator must not approve the State's changes until the proposed rulemaking has been withdrawn by the Administrator.

Withdrawal Notices

Proposed Rulemaking

Whenever promulgation proceedings are terminated, a notice of withdrawal of the proposed rulemaking must be published in the Federal Register. The Regional Offices are responsible for initiating such action and furnishing a rationale for use in preparing the notice for the Administrator's signature. These materials should be sent to the Criteria and Standards Division (WH-585).

Promulgation

An EPA-promulgated standard will be withdrawn when revisions to State water quality standards are made which meet the requirements of the Act.

In such a situation, the Regional Office should initiate the withdrawal action by notifying the Criteria and Standards Division (WH-585) that it is requesting the withdrawal and specifying the rationale for the withdrawal. EPA's action to withdraw a Federally promulgated standard requires both a proposed and final Rulemaking.

PUBLIC PARTICIPATION

This guidance includes two objectives that emphasize public participation and intergovernmental coordination. The first is to involve the regulated community (municipalities and industry) in the review and revision of water quality standards. The second objective is to encourage local, State, EPA, Regional and Headquarters personnel to cooperate as partners in the water quality standards review process. This partnership will ensure cross-fertilization of ideas, data and information and will increase the effectiveness of the total water quality management process.

Revisions in the water quality standards regulation were made to foster improved scientific and technical bases of water quality standards decisions. The analyses described in previous sections of this Handbook should assist States in analyzing their standards and in setting appropriate site-specific water quality standards.

An important component of the water quality standards setting process is the meaningful involvement in the process of those affected by standards decisions. At a minimum, States are required by Section 303(c) of the Clean Water Act to hold a public hearing in reviewing and revising their water quality standards. However, States are urged to more actively involve the public in the review process. By opening the water quality standards decision-making process to the public, States can encourage scientific discussion of the analyses and build the consensus necessary for implementing water quality standards decisions. The State may satisfy this public hearing requirement by providing the opportunity for the public to request a hearing. If no such request is forthcoming, the State need not actually conduct a hearing.

There are several points in the water quality standards decision-making process where public (municipal, industrial, environmental, academic, etc.) involvement would be beneficial. Enlisting the support of municipalities, industries, environmentalists and universities in collecting and evaluating data for the recommended analyses is another way States can involve those affected by standards decisions in the review process. The participation of outside groups in data collection and analyses must be based on State guidelines and oversight to ensure the integrity of the analyses. The extra time and effort necessary to organize and coordinate the participation of outside groups is worth the effort, particularly if the standards review is likely to generate widespread interest and/or controversy.

Involving the public in the analysis and interpretation of the data should assist States in improving the scientific basis of the standards decisions and in building support for a standards decision. Scientific discussion of the data can clarify areas of uncertainty, bring in new data, and/or identify areas where new data is necessary. The more people that are involved early in the process of setting appropriate standards, the more support the State will have in implementing the standards.

For the formal public hearings on the reviews and/or revisions of State water quality standards, the following requirements are applicable:

(1) A notice of the public hearing must be published in a newspaper with general circulation in the affected area at least 30 days prior the hearing. The notice should include:

(a) time and location of hearing,
(b) hearing agenda,
(c) notification of the availability of a Fact Sheet (The sheet must outline the major issues to be discussed, relevant State staff reports on the standards, determinations on proposed revisions, and any analyses conducted in support of proposed revisions that the public should be aware of prior to the hearing), and
(d) the location where reports, documents and data to be discussed at the hearing are available for public inspection.

(2) Notice of the public hearing should be mailed at least 30 days prior to the hearing to interested and affected persons and organizations including private and government organizations and individuals who have filed with the State requesting such notices. Notice of hearings should also be mailed to adjoining States and to Federal, interstate, and State agencies which are affected by existing State water quality standards or the proposed revisions.

(3) In addition, any other requirements necesssary to comply with State law for rulemaking hearings.

The hearing notice should solicit comments and provide opportunity for public comment. It is suggested that the hearing be held in the locally affected area. The State should prepare transcripts and summaries of the hearings which would be available for inspection by the public and the Regional Administrator. To facilitate EPA's review of revisions, States should supply the Agency with responses to the public comments related to the revision(s).

As has been indicated, effective public participation in the standards revision process is far more than a public hearing. The interaction of local, State and Federal governments along with the input of industry, municipalities and public interest groups will make the process more effective.

MIXING ZONES

Introduction

The concept of a mixing zone, a limited area or volume of water where initial dilution of a discharge takes place, has been covered by a series of guidance documents issued by EPA and its predecessor agencies. Although mixing zones have been applied in the water quality standards program since its inception, the Water Quality Standards regulation never has had explicit reference to mixing zones. The rule now recognizes that States may adopt mixing zones as a matter of State discretion. Guidance on defining mixing zones has previously been provided in the following documents: the Department of Interior Report, Water Quality Criteria 1968, (Green Book), the National Academy of Science, Water Quality Criteria 1972, (Blue Book), the EPA Quality Criteria for Water 1976 (Red Book), and Chapter 5, "Water Quality Standards, in the Guidelines for State and Area Wide Water Quality Management Program, 1976. The current guidance evolved from and supersedes these sources.

General

A limited mixing zone, serving as a zone of initial dilution in the immediate area of a point source of pollution, may be allowed.* Whether to establish a mixing zone policy is a matter of State discretion. Such a policy, however, must be consistent with the Act and is subject to the approval of the Regional Administrator.

Careful consideration must be given to the appropriateness of a mixing zone where a substance discharged is bioaccumulative, persistent, carcinogenic, mutagenic, or teratogenic. In such cases the State must consider such effects as sediment deposition, bioaccumulation in aquatic biota, bioconcentration in the food chain, and the known or predicted safe exposure levels for the substance. The effects of bioaccumulation will depend on the predicted duration/concentration exposure of the biota; thus, the likelihood that the mixing zone will be inhabited by resident biota for a sufficiently long time to cause adverse effects should be considered. Factors such as size of the zone, concentration gradient within the zone, physical habitat, attraction of aquatic life, etc., are important in this evaluation. In some instances, the ecological and human health effects may be so adverse that a mixing zone is not appropriate.

Definition of Allowable Mixing Zones

Water quality standards should describe the State's methodology for determining the location, size, shape, outfall design and in-zone quality of mixing zones. The methodology should be sufficiently

*In the broadest sense, the zone surrounding, or downstream from, a discharge location is an "allocated impact zone" where numeric water quality criteria can be exceeded as long as acutely toxic conditions are prevented.

precise to support regulatory actions, issuance of permits and determination of BMP's for nonpoint sources. EPA recommends the following:

Location. Biologically important areas are to be identified and protected. Where necessary to preserve a zone of passage for migrating fish or other organisms in a water course, the standards should specifically identify the portion of the waters to be kept free from mixing zones. The zone of passage should be based on the water quality criteria needed to allow migration of fish. This is typically less stringent than water quality criteria needed to maintain good growth and propagation of fish.

Size. Various methods and techniques for defining the surface area and the volume of mixing zones for various types of waters have been formulated. Methods which result in quantitative measures sufficient for permit actions and which protect the designated uses of the water body as a whole are acceptable. The area or volume of an individual zone or group of zones must be limited to an area or volume as small as practicable that will not interfere with the designated uses or with the established community of aquatic life in the segment for which the uses are designated.

Shape. The shape of a mixing zone should be a simple configuration that is easy to locate in the body of water and that avoids impingement on biologically important areas. In lakes, a circle with a specified radius is generally preferable, but other shapes may be specified in the case of unusual site requirements. "Shore-hugging" plumes should be avoided in all water bodies.

Outfall Design. Prior to designating any mixing zone, the State should assure that the best practicable engineering design is used and that the location of the existing or proposed outfall will avoid significant adverse aquatic resource and water quality impacts of the wastewater discharge.

In-zone Quality. Water quality standards should provide that all mixing zones conform with the following requirements. Any mixing zone should be free of point or nonpoint source related:

(a) Materials in concentrations that will cause acute toxicity to aquatic life.*

(b) Materials in concentrations that settle to form objectionable deposits;

* Acute toxicity as used here refers to aquatic life lethality caused by passage through the mixing zone by migrating fish moving up - or downstream, or by less mobile forms drifting through a plume. Requirements for waste water plumes which tend to attract aquatic life should take into account such attraction and reduce toxicity so as not to cause irreversible toxic effects in such attracted aquatic life.

(c) Floating debris, oil, scum and other matter in concentrations that form nuisances;

(d) Substances in concentrations that produce objectionable color, odor, taste or turbidity; and

(e) Substances in concentrations which produce undesirable aquatic life or result in a dominance of nuisance species.

Mixing Zones for the Discharge of Dredged or Fill Material

EPA, in conjunction with the Department of the Army, has developed guidelines to be applied in evaluating the discharge of dredged or fill material in navigable waters. (See 40 CFR Part 230, Federal Register, December 24, 1980). The guidelines include provisions for determining the acceptability of mixing discharge zones (§230.11(f)). The particular pollutants involved should be evaluated carefully in establishing dredging mixing zones. Dredged spoil discharges generally result in a temporary short-term disruption and do not represent a continous discharge of materials that will affect beneficial uses over a long-term. Disruption of beneficial uses should be the primary consideration in establishing mixing zones for dredged and fill activities. State water quality standards should reflect these principles if mixing zones for dredging activities are referenced.

Mixing Zones for Aquaculture Projects

The Administrator is authorized, after public hearings, to permit certain discharges associated with approved aquaculture projects (Section 318 of the Act). The regulations relating to aquaculture (40 CFR §122.56 and §125.11), provide that the aquaculture project must not result in a violation of standards outside of the project area and project approval must not result in the enlargement of any previously approved mixing zone. In addition, the aquaculture regulations provide that designated project areas must not include so large a portion of the body of water that a substantial portion of the indigenous biota will be exposed to the conditions within the designated project area (125.11(d)). Areas designated for approved aquaculture projects should be treated in the same manner as other mixing zones. Special allowances should not be made for these areas.

FLOWS

Water quality standards should protect water quality for designated uses in critical low flow situations. In establishing water quality standards, States may designate a critical low stream flow below which numerical water quality criteria do not apply. However, at all times water shall be free from substances that settle to form objectionable deposits; float as debris, scum, oil, or other matter; produces objectionable color, odor, taste, or turbidity; are acutely toxic, and which produce undesirable or nuisance aquatic life. Additional guidance on flow considerations may be found in Design Conditions, Chapter I, Stream Design Flow (Draft), August 31, 1983. This report is available from the Monitoring and Data Support Division (WH-551).

ECONOMIC GUIDANCE

Part 131.10, paragraph (g)(6) of the Water Quality Standards Regulation allows States, under certain conditions, to change a designated use if attaining that use would result in substantial and widespread economic and social impact. The substantial and widespread criteria should be applied to discrete changes in economic activity due to water quality standards. When considering these changes in economic activity, States should evaluate the incremental effects due to water quality standards; that is, effects due to controls beyond technology-based standards or other State requirements.

For municipalities, States should consider the economic effects associated with controls beyond the technology-based requirements in Section 301(b)(1)(B) of the Clean Water Act. If water quality standards require municipal treatment beyond those levels, EPA believes States should evaluate both the municipality's ability to make the initial pollution control investment and their financial capability over time for continued operation and maintenance. States should also evaluate changes to disposable income resulting from increased user charges or higher taxes. Another effect to consider is a situation where the municipality can make the investment for pollution control only by restricting expenditures for other municipal activities. These types of economic effects are the factors States should consider. States should then determine if the effects on the affected community are substantial and widespread.

When industry is required to install additional controls, the appropriate baseline is the technology-based requirement of Section 301(b)(2). If water quality standards require industrial controls beyond those requirements, States should consider the economic effects associated with the additional level of control. States should consider effects such as plant closure and unemployment, resulting from the inability of the plant to provide the necessary treatment. States should evaluate these effects in light of the level of unemployment in the area. States should also consider the condition where the plant is able to install and operate the treatment, but these expenditures would cancel or delay current plans for plant expansion or modernization. This effect on plant investment could cause reductions in future growth of employment and sales. Other industry effects include shifts in production processes or practices that change the plant's inputs. These shifts could result in changes to local employment, sales, and tax revenues. States should also evaluate effects on profitability and on a firm's competitive position. Further, if the plant's output is used locally, and the plant can pass through the additional costs in the form of higher prices, States should consider the price increase.

The factors listed above are not meant to be all-inclusive of what the States should consider when evaluating economic impact. Other economic effects may be appropriate, depending on the locality. Thus, any evaluation must be site-specific and address specific conditions in the affected community. The appropriate definition for community may vary depending on the type of effect being measured. For example, if unemployment is the effect being considered, the area from which the labor pool is drawn is affected community. After considering the appropriate factors, States should determine whether the effects are substantial and if so, whether they are widespread.

ANTIDEGRADATION POLICY

General

Each State must develop, adopt and retain a Statewide antidegradation policy in the water quality standards and identify methods for its implementation through the State WQM process. At a minimum the policy should contain the following components:

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.

Existing approved antidegradation statements consistent with §131.12 may be retained, but procedures for implementation must be established through the State water quality management (WQM) process. These procedures will enable the State to determine on a case-by-case basis whether, and to what extent, water quality may be lowered.

Public and Intergovernmental Review

The State WQM process must provide that whenever an activity is proposed which may degrade existing high quality waters, the State will assure that there is adequate public and intergovernmental participation.

Where the public and intergovernmental response, taken as a whole, clearly opposes a proposed degradation, the State must give serious consideration to that response and may not allow the proposed degradation activity unless it has a substantial and convincing justification for the activity.

While a State may decide, after satisfying the requirements for coordination and public participation, to allow some degradation of "high quality waters," any such lower water quality must protect existing uses fully and must also reflect the highest statutory and regulatory requirement for all new and existing point sources and all cost-effective and reasonable BMPs for nonpoint source control. "Highest statutory and regulatory requirements" refers to BAT or secondary treatment or new source performance standards (subject to any modifications under 301(g) and 316(a)), and any more stringent requirements imposed under State law or regulation.

Outstanding National Resource Waters

EPA changed the regulatory provision dealing with the degradation of outstanding National resource water (ONRW) to provide a limited exception to the previous absolute "no degradation" requirement. The regulation requires water quality to be maintained and protected in ONRW. EPA interprets this provision such that States may allow some limited activities which result in temporary and short-term changes in the water quality of ONRW. Such activities must not permanently degrade water quality nor result in water quality lower than that necessary to protect the existing uses in the ONRW. It is difficult to give an exact definition of "temporary" and "short-term" because of the variety of activities which might be considered. The intent of EPA's provision clearly is to limit water quality degradation to the shortest possible time. If a construction activity is involved, for example, temporary is defined as the length of time necessary to construct the facility and make it operational. During any period of time when, after opportunity for public participation in the decision, the State allows temporary degradation, all practical means of minimizing such degradation shall be implemented.

This change was made to make the ONRW provision a reasonable one which should encourage more States to make use of this designation. EPA views the effects of the change as minimal and consistent with sound resource management. The change is intended to avoid unreasonable restrictions and provide flexibility within the regulation. Example of situations when flexibility is required follow.

Example 1 - A national park wishes to replace a defective septic tank - drainfield system in a campground. The campground is located immediately adjacent to a small stream with the ONRW use designation.

If the previous regulation were taken literally, no construction would be allowed because if precipitation occurred, sediment would be washed into the stream. Under the new provision, the construction could occur if best management practices were scrupulously followed to minimize any disturbance of water quality or aquatic habitat.

Example 2 - Same situation except the campground is served by a small sewage treatment plant already discharging to the ONRW. It is desired to enlarge the treatment system and provide higher levels of treatment.

Under the previous regulation, since no degradation was permitted, this water-quality-enhancing action would not be permitted because of the temporary increase in sediment and, perhaps, in organic loading which would occur during the actual construction phase. Under the new regulation, it could be allowed.

Example 3 - A National forest with a mature, second growth of trees which are suitable for harvesting, with associated road repair and re-stabilization. Streams in the area designated as ONRW and support trout fishing.

The new regulation intends that best management practices for timber harvesting be followed and might include preventive measures more stringent than for similar logging in less environmentally sensitive areas. Of course, if the lands were being considered for designation as wilderness areas or other similar designations, EPA's regulation should not be construed as encouraging or condoning timbering operations. The regulation only allows temporary and short term water quality degradation while maintaining existing uses or new uses consistent with the purpose of the management of the ONRW area.

Antidegradation and Growth

National antidegradation requirements should not be viewed as a "no growth" rule. Where the State intends to provide for further development, the State WQM process should evaluate the alternative measures which can be taken to preserve water quality, such as requiring land disposal for new projects. The evaluation must take into account the physical, chemical, and biological characteristics of the waterbody and possible widespread economic and social impacts.

Optional State Actions

The State's antidegradation policy is to be used for the protection of existing water quality. Use designations should not be an issue, since the specific water quality standards should always, at a minimum, designate existing beneficial uses. The State's water quality standards for high quality waters may, within the constraints and limitations of monitoring practicability, set forth the existing water quality of a segment. Thus, the State may adopt specific criteria reflecting existing levels measured in the high quality segment, even though such levels may be more stringent than the Section 304(a) criteria minimum levels for given uses. Documentation of existing water quality is essential in the State WQM process as a baseline against which any future degradation could be measured.

Consistency with Section 316

Under Section 316(a) of the Act, if a proper showing is made, NPDES permits may contain thermal effluent limitations which are less stringent than those which might otherwise be required under Section 301(b)(1)(C) to implement State antidegradation requirements. (In this respect, Section 316(a) creates a limited exception to Section 510). Section 131.12(a)(4) of the water quality standard regulation therefore provides that States must ensure that their antidegradation policies are not interpreted or applied to prevent the imposition of modified thermal effluent limitations in NPDES permits under Section 316(a).

Federal Review of Antidegradation Policies and Actions

The State's antidegradation statement and implementing procedures, as a part of its water quality standards and WQM process, are subject to the Regional Administrator's review and approval. EPA encourages submittal of any amendments to this statement and implementing procedures to the Regional Administrator for pre-adoption review so that the State may take EPA comments into account prior to final adoption.

APPLICATION OF NUMERICAL AND NARRATIVE CRITERIA

Introduction

Section 131.11(a)(2) of the Water Quality Standards Regulation provides that the "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." The criteria which are adopted may be numerical or narrative or both. Pollutant-specific numerical criteria may be used when the control of specific pollutants is of concern, and narrative criteria may be applied when the control of either combinations of pollutants together or individual pollutants not contained in State water quality standards is of concern.

When narrative criteria are adopted in lieu of numerical criteria to control toxic pollutants, the Water Quality Standards Regulation requires that "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," and that "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)."

To implement these numerical and narrative criteria, the Environmental Protection Agency encourages the use of an integrated strategy consisting of both biological and chemical methods. Where State water quality standards contain numerical criteria for toxic pollutants, NPDES permits will contain limits as necessary to assure compliance with these standards. In addition to enforcing specific numerical criteria, biological techniques and available data on chemical effects will be used to assess toxicity impacts based on the State's general narrative toxicity standard. The use of such an integrated approach by the EPA and the States has been endorsed in a draft EPA Office of Water "Policy for the Development of Water Quality-Based Permit Limitations for Toxic Pollutants."

The following section discusses how numerical and narrative criteria may be applied for toxic pollutants.

Approaches for Applying Numerical and Narrative Criteria for Toxic Pollutants

All States have a general narrative requirement in their water quality standards that their waters not contain toxic substances in toxic amounts (e.g., the so-called "toxics free from"). This requirement, which focuses on the toxicological properties of either individual substances or mixtures of substances, has most commonly been applied to individual toxic pollutants through the establishment of pollutant-specific water quality based controls, but can also be applied to mixtures of pollutants such as can be found in whole effluents or in receiving waters. The latter application of general narrative toxicity criteria is consistent with §502(13) of the Clean Water Act, which defines "toxic pollutant" as "those pollutants, or combinations of pollutants, . . . which . . . will . . . cause death, disease, behavioral abnormalities, cancer, . . ." (emphasis added).

Narrative toxicity criteria are normally applied to those pollutants identified under §307(a)(1) of the Clean Water Act, but may also be applied to any other individual pollutant or combination of pollutants which fit the definition of §502(13).

The two possible approaches for applying numerical and narrative water quality criteria are discussed further below.

Pollutant-Specific Approach for Applying Toxics Water Quality Criteria

A pollutant-specific approach for controlling toxic pollutants involves the application of numerical water quality criteria which reflect the toxicological properties of individual substances. These numerical criteria express water quality objectives for preventing acute or chronic toxicity or for meeting a defined level of water quality protection that is based on the water body's designated uses.

The pollutant-specific approach is most appropriately used where a few specific pollutants have been identified as the concern, or where human health is the issue. Predictive tools such as water quality fate and transport models are often used to translate specific criteria on a pollutant-by-pollutant basis into a specific water quality based permit limit.

The numerical criteria which are applied in any given case may be based on existing water quality standards or published criteria, or else site-specific numerical criteria may be developed based on the State's general narrative toxicity standard. A recommended procedure for determining appropriate numerical water quality criteria for individual toxic pollutants is outlined below:

1) The designated uses of the receiving water should first be examined to determine whether the protection of aquatic life, human health, or both is of concern. Appropriate criteria protective of aquatic life or human health should then be selected in the steps below for those pollutants present or suspected of being present in the water body.

2) The applicable State water quality standards for the receiving water should be examined to see if a numerical criterion value exists for the parameter of concern, and if it appropriately reflects the aquatic life or human health protection needs of the water body. If so, then this criterion may be applied.

3) If no appropriate criteria appear in the State's water quality standards, then the EPA national criteria for protection of aquatic life and human health (References 1, 2, 3, 4) may be consulted for the pollutant parameters of concern. If a discharge is to a receiving water designated as a domestic water supply, then the finished drinking water health advisories (adjusted for treatment capabilities) should also be consulted. Where a pollutant has both EPA human health water quality criteria and drinking water advisories, the more stringent of the criteria should normally be applied.

4) For those pollutants which have no EPA water quality criteria or drinking water advisories, or the criteria or advisories are inapplicable to the water body of concern, site-specific criteria which are protective of the water body's designated uses should be developed based on the State's general narrative toxicity standard. These site-specific criteria should be developed utilizing toxicity tests, indicator organisms, and application factors which may be contained in the State's water quality standards, or other procedures that are consistent with those outlined in Chapter 4 of this Handbook.

The Pennsylvania Water Quality Standards illustrate how numerical criteria can be developed and applied for pollutants. The Standards list the parameters for which criteria have been established by the State and the values of those criteria; and also acknowledge that the "list of specific water quality criteria does not include all possible substances that could cause pollution," and that "for substances not listed, the general criterion that these substances shall not be inimical or injurious to the designated water use applies." The Pennsylvania standards further define the steps which may be taken when a specific criterion has not been established for a pollutant. They provide that a specific criterion may be determined through establishment of a "safe concentration value," which shall be based upon adequate data obtained from relevant aquatic field studies, available literature, or specific bioassay tests, or, where insufficient data are available to establish a safe concentration value, shall be determined by using specified bioassay testing procedures and by applying appropriate specified application factors to the pollutant's 96-hour (or greater) LC_{50} value.

<u>General Toxicity Approach for Applying Toxics Water Quality Criteria</u>

A general toxicity approach focuses on the overall toxicological properties of mixtures of pollutants in effluents or receiving waters, with the objective of preventing acute and chronic toxicity conditions in the water body and meeting a defined level of water quality protection that is based on the water body's designated uses. With this approach, the State's general narrative toxicity standard is used on a case-by-case basis to ensure that no acute toxicity conditions exist within any State-defined or otherwise identified mixing zone, and no acute or chronic toxicity conditions exist elsewhere in the water body.

The general toxicity approach is most appropriately used where effluent or instream conditions are complex. For example, the toxicity effects of one or several discharges containing many known or unknown constituents can be readily assessed. This approach can also be applied in conjunction with pollutant-specific techniques, especially when residual toxicity or synergistic or other effects are a concern.

The State Water Quality Standards of Maryland and Florida illustrate how narrative criteria can be developed and applied for individual and combinations of pollutants. Maryland's Water Quality Standards contain a general criterion which provides that the "waters of the State at all times shall be free from . . . toxic, corrosive, or other deleterious substances attributable to sewage, industrial waste, or other waste in concentrations or combinations which interfere directly or indirectly with water uses, or which are harmful to human, animal, plant, or aquatic life" (emphasis added). Florida's Water Quality Standards contain a similar general narrative criterion which provides that "all waters . . . shall at all places and at all times be free from . . . components of discharges which, alone or in combination with other substances or in combination with other components of discharges . . . are acutely toxic . . . or . . . carcinogenic, mutagenic, or teratogenic to human beings or . . . aquatic life . . ." (emphasis added). The Florida standards also further specify several of the narrative requirements contained therein: for example, the standards define acute toxicity to mean "the presence of one or more substances or characteristics or components of substances in amounts which . . . are greater than one-third (1/3) of the . . . 96 hr. LC_{50} . . . where the 96 hr. LC_{50} is the lowest value which has been determined for a species significant to the indigenous aquatic community . . . ."

Toxicity tests, including instream or laboratory bioassays and instream biological sampling, may be used to implement this approach. The State should identify, in its water quality standards or a guidance document, the appropriate acute and/or chronic toxicity bioassay tests, number and types of indicator organisms, application factors, water body design conditions, and instream biological sampling procedures to be used. The methods and procedures to be employed should be reflective of the use designations of the water bodies to be protected.

Numerous States already identify various toxicity testing-related requirements in their water quality standards. For example, the West Virginia standards provide that "bioassay testing shall be conducted in accordance with the methodologies outlined in the . . . [EPA publication,] Methods of Acute Toxicity Tests with Fish, Macroinvertebrates, and Amphibians . . .; Standard Methods of [sic] the Examination of Water and Wastewater . . .; . . . Standard Method of Test for ASTM . . .; or . . . [the EPA publication], Methods for Measuring the Acute Toxicity of Effluents to Aquatic Organisms . . .": and Texas' standards state that, "[f]or evaluations of toxicity, bioassay techniques will be selected as suited to the purpose at hand," and "[a]s a general guideline, bioassays will be conducted using fish indigenous to the receiving waters, and water quality conditions . . . which approximate those of the receiving waters."

Reference 5 contains additional information on toxicity testing methods and procedures for setting water quality based controls for toxic pollutants using a general toxicity approach.

References

1. U.S. EPA, Water Quality Criteria Documents (45 FR 79318, November 28, 1980, 46 FR 40919, August 13, 1981).

2. Quality Criteria for Water, U.S. EPA (1976). GPO Stock No. 005-001-01049-4.

3. Water Quality Criteria, U.S. EPA (1972). EPA-R3-73-033.

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RELATIONSHIP OF SECTION 304(a)(1) CRITERIA TO DESIGNATED WATER USES

Introduction

The Section 304(a)(1) criteria published periodically by EPA can be used to support the designated uses which are generally found in State standards. The following sections briefly discuss the relationship between certain criteria and individual use classifications. Additional information on this subject may also be found in the FWPCA report, Water Quality Criteria 1968 ("Green Book"); the National Academy of Science, Water Quality Criteria 1972 ("Blue Book"); the EPA Quality Criteria for Water 1976 ("Red Book"); the EPA Water Quality Criteria Documents(45 FR 79318, November 28, 1980, 46 FR 40919, August 13, 1981); and future EPA section 304(a)(1) water quality criteria publications.

Where a water body is designated for more than one use, criteria necessary to protect the most sensitive use must be applied.

Recreation

Recreational uses of water include activities such as swimming, wading, boating, and fishing. In general, insufficient data exist on the human health effects of physical and chemical pollutants, including most toxics, resulting from exposure through such primary contact as swimming. However, as a general guideline, recreational waters that contain chemicals in such concentrations as to be toxic or otherwise harmful to man if ingested, or to be irritating to the skin or mucous membranes of the human body upon brief immersion should be avoided. The section 304(a)(1) human health effects criteria based on direct human drinking water intake and fish consumption might provide useful guidance in these circumstances. Also, section 304(a)(1) criteria based on human health effects may be used to support this designated use where fishing is included in the State definition of "recreation." In this latter situation, only the portion of the criterion based on fish consumption should be used. Section 304(a)(1) criteria to protect recreational uses are also available for certain physical, microbiological, and qualitative parameters.

The "Green Book" and "Blue Book" provide additional information on protecting recreational uses.

Protection and Propagation of Fish and Other Aquatic Life

The section 304(a)(1) criteria based on toxicity to aquatic life may be used directly to support this designated use. If subcategories of this use are adopted (e.g., to differentiate between cold water and warm water fisheries), then appropriate criteria should be set to reflect the varying needs of such subcategories.

Agricultural and Industrial Uses

The "Green Book" and "Blue Book" provide information for certain parameters on protecting agricultural and industrial uses, although section 304(a)(1) criteria for protecting these uses have not been specifically developed for numerous other parameters, including most toxics.

Where criteria have not been specifically developed for these uses, the criteria developed for human health and aquatic life are usually sufficiently stringent to protect these uses. States may also establish criteria specifically designed to protect these uses.

Public Water Supply

The drinking water exposure component of the section 304(a)(1) criteria based on human health effects can apply directly to this use classification or may be appropriately modified depending upon whether the specific water supply system falls within the auspices of the Safe Drinking Water Act's (SDWA) regulatory control, and the type and level of treatment imposed upon the supply before delivery to the consumer. The SDWA controls the presence of toxic pollutants in finished ("end-of-tap") drinking water. A brief description of relevant sections of this Act is necessary to explain how the SDWA will work in conjunction with section 304(a)(1) criteria in protecting human health from the effects of toxics due to consumption of water.

Pursuant to section 1412 of the SDWA, EPA has promulgated "National Interim Primary Drinking Water Standards" for certain organic and inorganic substances. These standards establish "maximum contaminant levels" ("MCLs") which specify the maximum permissible level of a contaminant in water which may be delivered to a user of a public water system now defined as serving a minimum of 25 people. MCLs are established based on consideration of a range of factors including not only the health effects of the contaminants but also technological and economic feasibility of the contaminants' removal from the supply. EPA is required to establish revised primary drinking water regulations based on the effects of contaminant on human health, and include treatment capability, monitoring availability, and costs. Under Section 1401(1)(D)(i) of the SDWA, EPA is also allowed to establish the minimum quality criteria for water which may be taken into a public water supply system.

Section 304(a)(1) criteria provide estimates of pollutant concentrations protective of human health, but do not consider treatment technology, costs, and other feasibility factors. The section 304(a)(1) criteria also include fish bioaccumulation and consumption factors in addition to direct human drinking water intake. These numbers were not developed to serve as "end of tap" drinking water standards, and they have no regulatory significance under the SDWA. Drinking water standards are established based on considerations, including technological and economic feasibility, not relevant to section 304(a)(1) criteria. Section 304(a)(1) criteria may be analogous to the recommended maximum contaminant levels (RMCLs) under section 1412(b)(1)(B) of the SDWA in which, based upon a report from the National Academy of Sciences, the Administrator should set target levels for contaminants in drinking water at which "no known or anticipated adverse effects occur and which allows an adequate margin of safety." RMCLs do not take treatment, cost, and other feasibility factors into consideration. Section 304(a)(1) criteria are, in concept, related to the health-based goals specified in the RMCLs. Specific mandates of the SDWA such as the consideration of multi-media exposure, as well as different methods for setting maximum contaminant levels under the two Acts, may result in differences between the two numbers.

MCLs of the SDWA, where they exist, control toxic chemicals in finished drinking water. However, because of variations in treatment and the fact that only a relatively small number of MCLs have been developed, ambient water criteria may be used by the States as a supplement to SDWA regulations. States will have the option of applying MCLs, section 304(a)(1) human health effects criteria, modified section 304(a)(1) criteria, or controls more stringent than these three to protect against the effects of toxic pollutants by ingestion from drinking water.

For untreated drinking water supplies, States may control toxics in the ambient water through either use of MCLs (if they exist for the pollutants of concern), section 304(a)(1) human health effects criteria, or a more stringent contaminant level than the former two options.

For treated drinking water supplies serving less than 25 people, States may choose toxics control through application of MCLs (if they exist for the pollutants of concern and are attainable by the type of treatment) in the finished drinking water. States also have the options to control toxics in the ambient water by choosing section 304(a)(1) criteria, adjusted section 304(a)(1) criteria resulting from the reduction of the direct drinking water exposure component in the criteria calculation to the extent that the treatment procedure reduces the level of pollutants, or a more stringent contaminant level than the former three options.

For treated drinking water supplies serving 25 people or greater, States must control toxics down to levels at least as stringent as MCLs (where they exist for the pollutants of concern) in the finished drinking water. However, States also have the options to control toxics in the ambient water by choosing section 304(a)(1) criteria, adjusted section 304(a)(1) criteria resulting from the reduction of the direct drinking water exposure component in the criteria calculation to the extent that the treatment process reduces the level of pollutants, or a more stringent contaminant level than the former three options.

CHAPTER 3

WATER BODY SURVEY AND ASSESSMENT GUIDANCE FOR CONDUCTING USE ATTAINABILITY ANALYSES

Contents

Purpose and Application

The purpose of this guidance is to identify the physical, chemical and biological factors that may be examined to determine if an aquatic life protection use is attainable for a given water body. The use attainability analysis is an important environmental analysis to improve the scientific and technical basis of setting site-specific water quality standards. The specific analyses included in this guidance are optional. However, they 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 may use alternative analyses as long as they are scientifically and technically supportable. This guidance specifically addresses streams and river systems. EPA is presently developing guidance for estuarine and marine systems and plans to issue such guidance in 1984.

Several approaches for analyzing the aquatic life protection uses to determine if such uses are appropriate for a given water body are discussed. States are encouraged to use existing data to perform the physical, chemical, and biological evaluations presented in this guidance document. Not all of these evaluations are necessarily applicable. For example, if a physical assessment reveals that the physical habitat is the limiting factor precluding a use, a chemical evaluation would not be required. In addition wherever possible, States also should consider grouping together water bodies having similar physical, chemical, and biological characteristics to either treat several water bodies or stream segments as a single unit or to establish representative conditions which are applicable to other similar water bodies or stream segments within a river basin. Using existing data and establishing representative conditions applicable to a number of water bodies or segments should conserve the limited resources available to the States.

The evaluations presented in this guidance document should be sufficiently detailed to answer:

- What are the aquatic use(s) currently being achieved in the water body?

- What are the causes of any impairment in the aquatic uses?

- What are the aquatic use(s) which can be attained based on the physical, chemical, and biological characteristics of the water body?

Questions addressing the evaluation of control options are discussed in the Wasteload Allocation Guidance (EPA, 1983).

Table 1 summarizes the types of physical, chemical, and biological evaluations which may be conducted. The guidance document presents

TABLE 2-1: SUMMARY OF TYPICAL USE ATTAINABILITY ANALYSES

PHYSICAL EVALUATIONS	CHEMICAL EVALUATIONS	BIOLOGICAL EVALUATIONS
° Instream Characteristics	° dissolved oxygen	° Biological Inventory (Existing Use Analysis)
- size (mean width/depth)		
- flow/velocity	° toxicants	- fish
- annual hydrograph		- macroinvertebrates
- total volume	° suspended solids	- microinvertebrates
- reaeration rates	° nutrients	- phytoplankton
- gradient/pools/riffles	- nitrogen	- periphyton
- temperature	- phosphorus	- macrophytes
- suspended solids		
- turbidity	° sediment oxygen demand	° Biological Condition/Health Analysis
- sedimentation		- Diversity Indices
- channel modifications	° salinity	- HSI Models
- channel stability		- Tissue Analyses
	° hardness	- Recovery Index
° Substrate composition and characteristics		- Intolerant Species Analysis
	° alkalinity	- Omnivore-Carnivore Analysis
° Channel debris	° pH	° Biological Potential Analysis
		- Reference Reach Comparison
° Sludge deposits	° dissolved solids	
° Riparian characteristics		
° Downstream characteristics		

several approaches for conducting the physical, chemical, and biological evaluations depending on the complexity of the situation. Details on the various evaluations can be found in the Technical Support Manual for Conducting Use Attainability Analyses, available from the person listed in the Foreword of this Handbook. A survey need not consider all of the parameters listed but rather the survey should be designed on the basis of the stream characteristics and other considerations relevant to a particular survey. Case studies showing how the analyses were used in evaluating the attainability of uses and in setting appropriate uses for a site-specific water quality standard are contained in Appendix D.

These approaches may be adapted to the water body being examined. Therefore, a close working relationship between EPA and the States is essential so that EPA can assist States in determining the appropriate analyses to be used in support of any water quality standards revisions or Advanced Treatment Project justifications. These analyses should be made available to all interested parties before any public forums on the water quality standards to allow for full discussion of the data and analyses.

Physical Evaluations

Section 101(a) of the Clean Water Act recognizes the importance of preserving the physical integrity of the Nation's water bodies. Physical habitat plays an important role in the overall aquatic ecosystem and impacts the types and number of species present in a particular body of water. Physical parameters of a water body are examined to identify any non-water quality related factors which impair the propagation and protection of aquatic life and to determine what uses could be obtained in the water body given such limitations. In general, physical parameters such as flow, temperature, water depth, velocity, substrate, reaeration rates and other factors are used to identify any physical limitations that may preclude the attainment of the designated use. Depending on the water body in question any of the following physical parameters may be appropriately examined. A State may utilize any of these parameters for identifying physical limitations and characteristics of a water body. Once a State has identified any physical limitations based on evaluating the parameters listed, careful consideration of "reversibility" or the ability to restore the physical integrity of the water body should be made.

Such considerations may include whether it would cause more environmental damage to correct the problem than to leave the waterbody as is, or whether physical impediments such as dams can be operated or modified in a way that would allow attainment of the use.

I. Channel and instream characteristics including:
- ° mean stream width and depth
- ° total volume
- ° flow and water velocity
- ° reaeration rates
- ° gradient
- ° pools
- ° riffles
- ° seasonal changes
- ° turbidity
- ° suspended solids
- ° temperature
- ° sedimentation
- ° channel stability
- ° channel obstructions:
  - dams
  - waterfalls, log jams, steep gradient
  - other impoundments and channel obstructions
- ° channel changes:
  - road construction
  - dredging activities
  - clearing areas (culverts, bridges, etc.)
  - channelization
- ° instream cover:
  - undercut banks
  - overhanging brush
- ° snags and woody debris
- ° downstream characteristics

II. Substrate composition and characteristics including:

° organic debris/muck
° clay
° silt
° sand
° gravel
° cobble
° boulder
° bedrock

III. Riparian characteristics including:

° bank cover
  - forested
  - brush
  - grass and herbaceous vegetation
  - non-vegetated areas
° bank stability
° soil composition (percent boulder, gravel, cobble, sand, silt, clay)
° land gradients
° bank width

Several assessment techniques have been developed which correlate physical habitat characteristics to fishery resources (Stalnacker, 1978; Dunham and Cooper, 1975; Collotz and Dunham, 1978). The identification of physical factors limiting a fishery is a critical assessment that provides important data for the management of the water body. The U.S. Fish and Wildlife Service has developed habitat evaluation procedures (HEP) and habitat suitability indices (HSI). Several States have begun developing their own models and procedures for habitat assessments. Parameters generally included in habitat assessment procedures are: temperature, turbidity, velocity, depth, cover, pool and riffle sizes, riparian vegetation, bank stability, siltation, etc. These parameters are correlated to fish species by evaluating the habitat variables important to the life cycle of the species. The value of habitat for other groups of aquatic organisms such as macroinvertebrates and periphyton may also be considered. Continued research and refinement of habitat evaluation procedures reflects the importance of physical habitat.

If physical limitations of a stream restrict the use, there are a variety of habitat modification techniques which might restore a habitat so that a species could thrive where it could not before. Some of the techniques which have been used include: bank stabilization, flow control, current deflectors, check dams, artificial meanders, isolated oxbows, snag clearing when determined not to be detrimental to the life cycle or reproduction of a species, and installation of spawning beds and artificial spawning channels (U.S. Fish and Wildlife Service, 1978). If the habitat is a limiting factor to the propagation and/or survival of aquatic life, the feasibility of modifications might be examined prior to imposing additional controls on dischargers.

Chemical Evaluations

The chemical characteristics of a water body are examined to determine why a designated use is not being met and to determine the potential of a particular species to survive in the water body if the concentration of particular chemicals were modified. The following is a partial list of the parameters that may be evaluated. The State has the discretion to determine the parameters required to perform an adequate water chemistry evaluation.

- toxicants
- nutrients e.g. nitrogen and phosphorus
- sediment oxygen demand
- salinity
- hardness
- pH
- alkalinity
- dissolved solids
- suspended solids

As part of the evaluation of the water chemistry composition, a natural background evaluation is useful in determining the relative contribution of natural background contaminants to the water body as this may be a legitimate factor which effectively prevents a designated use from being met. To determine whether the natural background concentration of a pollutant is adversely impacting the survival of species, the concentration may be compared to one of the following:

- 304(a) criteria guidance documents; or
- site-specific criteria; or
- State-derived criteria.

Another way to get an indication of the potential for the species to survive is to determine if the species are found in other waterways with similar chemical concentrations. However, this is not a precise indicator.

In determining whether man-induced pollution is irreversible, consideration needs to be given to the permanence of the damage, the feasibility of abating the pollution, or the additional environmental damage that may result from removing the pollutants. If nonpoint source pollution cannot be abated with application of best management practices (BMPs) and the activity causing the nonpoint source pollution problem is determined to be essential, States may consider the pollution irreversible. EPA's policy is that feasible BMPs which reduce nonpoint source pollution must be developed in accordance with priorities for developing control programs for all nonpoint sources identified in areawide and State planning areas. Site specific conditions are to be taken into account during BMP design and implementation.

In addition, if instream toxicants cannot be removed by natural processes and cannot be removed by man without severe long-term environmental impacts, the pollution may be considered irretrievable.

In some areas the water's chemical characteristics may have to be calculated, using predictive water quality models, rather than determined empirically. This will be true if the receiving water is to be impacted by new dischargers, changes in land use, or improved treatment facilities. Guidance is available on the selection and use of receiving water models for biochemical oxygen demand, dissolved oxygen and ammonia for instream systems (EPA, 1983) and dissolved oxygen, nitrogen and phosphorus for lake systems, reservoirs and impoundments (EPA, 1981).

Once a State identifies the chemical or water quality characteristics which are limiting the attainment of the use, differing levels of remedial control measures may be explored.

Biological Evaluations

In evaluating what aquatic life protection uses are attainable, the biology of the water body should be evaluated. The interrelationships between the physical, chemical and biological characteristics are complex and alterations in the physical and/or chemical parameters result in biological changes. The biological evaluation described in this section encourages States to (1) provide a more precise statement of which species exist in the water body and should be protected; (2) determine the biological health in the water body and; (3) determine the species that could potentially exist in the water body if the physical and chemical factors impairing a use were corrected. This section of the guidance will present the conceptual framework for making these evaluations. States have the discretion to use other scientifically and technically supportable assessment methodologies deemed appropriate for specific water bodies on a case by case basis. Further details on each of the analyses presented can be found in the Technical Support Manual for Conducting Use Attainability Analyses.

° Biological Inventory (Existing Use Analysis)

The identification of which species are in the water body and should be protected serves several purposes:

(a) By knowing what species are present, the biologist can analyze, in general terms, the health of the water body. For example, if the fish species present are principally carnivores, the quality of the water is generally higher than in a water body dominated by omnivores. It also allows the biologist to assess the presence or absence of intolerant species.

(b) Identification of the species enables the State to develop baseline conditions against which to evaluate any remedial actions. The development of a regional baseline based upon several site-specific species lists increases an understanding of the regional fauna. This allows for easier grouping of water bodies based on the biological regime of the area.

(c) By identifying the species, the decision-maker has the data needed to explain the present condition of the water body to the public and the uses which must be maintained.

The evaluation of the existing biota may be simple or complex depending on the availability of data. As much information as possible should be gathered on the following categories of organisms:

° fish
° macroinvertebrates
° microinvertebrates
° phytoplankton
° periphyton
° macrophytes

It is not necessary to obtain complete data for all six categories. However, it is recommended that whichever combination of categories is chosen, fish should be included. The reasons for this recommendation are: (1) the general public can relate better to statements about the condition of the fish community; (2) fish are typically present even in the smallest streams and in all but the most polluted waters; (3) fish are relatively easy to identify and samples can be sorted and identified at the field site; (4) life-history information is extensive for many fish species so that stress effects can be evaluted (Karr, 1981). Since fish are mobile, States are encouraged to evaluate other categories of organisms also.

Prior to conducting any field work, existing data should be collected. EPA can provide data from intensive monitoring surveys and special studies. Data, especially for fish, may be available from State fish and game departments, recreation agencies, and local governments or through environmental impact statements, permit reviews, surveys, and university and other studies.

° Biological Condition/Biological Health Assessment

The biological inventory can be used to gain insight into the biological health of the water body by evaluating:

(a) species richness or the number of species
(b) presence of intolerant species
(c) proportion of omnivores and carnivores
(d) biomass or production
(e) number of individuals per species

The role of the biologist becomes critical in evaluating the health of the biota as the knowledge of expected richness or expected species comes only from understanding the general biological traits and regimes of the area. Best professional judgments by local biologists are important. These judgments are based on many years of experience and on observations of the physical and chemical changes that have occurred over time.

There are many mechanisms to evaluate biotic communities that have been and are continuing to be developed. The following briefly describes mechanisms that States may want to consider using in their biological evaluations:

- Diversity Indices - Diversity indices permit large amounts of information about the numbers and kinds of organisms to be summarized in a single value. Diversity indices have been applied to ascertain quantitative relationships between the health of the population and waste discharges. However, as summaries, diversity indices lose information concerning the identity of particular species involved and thus may obscure major changes in species composition. These changes are often indicative of changed conditions. The information on species composition can be retained by developing a species list in rank order of abundance such as the biological inventory discussed previously. References on diversity indices may be found in the bibliography of this guidance.

- Habitat Suitability Index (HSI) Models - The U.S. Fish and Wildlife Service Habitat Suitability Index models relate habitat requirements to specific fish species by identifying key habitat variables and the range and optimums for such variables. These index models are hypotheses of species-habitat relationships which may be helpful in identifying the physical habitat characteristics that are crucial to the species and defines the ranges and optimums to allow species survival and propagation.

- Tissue Analyses - Tissue analyses may be conducted to assess the effects of heavy metals and pesticides on the biota present. This chemical analysis of tissue for bioaccumulation is especially important if the water body is used for recreational or commercial fishing as high bioconcentration of metals and pesticides by the organisms may create a human health problem.

- Recovery Index - Estimating the elasticity of an ecosystem, or its ability to recover after displacement of structure and/or function to a steady state closely approximating the original, may be an interesting quantitative evaluation to make to answer the question of what is the potential for recovery in this water body. Cairns et al. (1977) developed an index of elasticity based on the following factors:

(a) existence of nearby epicenters for reinvading organisms
(b) transportability or mobility of disseminules
(c) presence of residual toxicants following pollutional stress
(d) general present condition of habitat following pollutional stress
(e) management or organizational capabilities for immediate or direct control of damaged area.

Stauffer and Hocutt (1980) applied the above index to the Conowingo Creek in Pennsylvania. They believe that this concept may form the foundation for a stream classification system based upon the structure and function of fish communities.

- Intolerant Species Analysis - The evaluation of the presence or absence of intolerant species refers to those species readily identified as declining because of water quality degradation, habitat degradation or a combination of the two. For example in midwestern streams, species such as blacknose shiner, southern redbelly dace, banded darter and others have been found to be intolerant. The application of the intolerant species analysis can be used on macroinvertebrates and periphyton as well as fish to indicate the degree of degradation.

- Omnivore-Carnivore Analysis - The proportion of top carnivores and omnivores may give an indication of the relative health of the community. Karr (1980) found that as a site declines in quality, the proportion of individuals that are omnivores increases. Viable and healthy populations of top carnivore species such as walleye, smallmouth bass, rock bass and others indicate a relatively healthy, diverse community.

A number of other methods have been and are being developed to evaluate the health of biological components of the aquatic ecosystem including short term *in situ* or laboratory bioassays and partial or full life-cycle toxicity tests. These methods are discussed in several EPA publications including: Basic Water Monitoring Programs (1978), Model State Water Monitoring Program (1975) and the Biological Methods Manual (1972). Again, it is not the intent of this document to specify tests to be conducted by the States. This will depend on the information available, the predictive accuracy required, site-specific conditions of the water body being examined, and the cooperation and assistance the State receives from the affected municipalities and industries.

° Biological Potential Analysis

A significant step in the use attainability analysis is the evaluation of what communities could potentially exist in a particular water body if pollution were abated or the physical habitat modified. The approach presented is to compare the water body in question to reference reaches within a region. This approach includes the development of baseline conditions to facilitate the comparison of several water bodies at less cost. As with the other analyses mentioned previously, available data should be used so as to minimize resource impacts.

The biological potential analysis involves:

- ° defining boundaries of fish faunal regions;
- ° selecting control sampling sites in the reference reaches of each area;
- ° sampling fish and recording observations at each reference sampling site;
- ° establishing the community characteristics for the reference reaches of each area; and
- ° comparing the water body in question to the reference reaches.

In establishing faunal regions and sites, it is important to select reference areas for sampling sites that have conditions typical of the region. The establishment of reference areas may be based on physical and hydrological characteristics. The number of reference reaches needed will be determined by the State depending on the variability of the waterways within the State and the number of classes that the State may wish to establish. For example, the State may want to use size, flow and substrate as the defining characteristics and may consequently desire to establish classes such as small, fast running streams with sandy substrate or large, slow rivers with cobble bottom. It is at the option of the State to: (1) choose the parameters to be used in classifying and establishing reference reaches and (2) determine the number of classes (and thus the refinement) within the faunal region. This approach can also be applied to other aquatic organisms such as macroinvertebrates (particularly freshwater mussels) and algae.

Selection of the reference reaches is of critical importance because the characteristics of the aquatic community will be used to establish baseline conditions against which similar reaches (based on physical and hydrological characteristics) are compared. Once the reference reaches are established, the water body in question can be compared to the reference reach. The results of this analysis will reveal if the water body in question has the typical biota for that class or a less desirable community and will provide an indication of what species may potentially exist if pollution were abated or the physical habitat limitations were remedied.

Approaches to Conducting the Physical, Chemical, and Biological Evaluations

Several measurements and experimental techniques have been described for collecting and evaluating the chemical, physical, and biological data to identify and define:

° What aquatic protection uses are currently being achieved in the water body,

° What the causes are of any impairment in the aquatic protection uses, and

° What aquatic protection uses could be attained based on the physical, chemical and biological characteristics of the water body?

States that assess the status of their aquatic resources, in some cases will have relatively simple situations not requiring extensive data collection and evaluation. In other water bodies, however, the complexity resulting from variable environmental conditions and the stress from multiple uses of the resource will require both intensive and extensive studies to produce a sound evaluation of the system. Thus procedures that a State may develop for conducting a water body assessment should be flexible enough to be adaptable to a variety of site-specific conditions.

A common experimental approach used in biological assessments has been a hierarchical approach to the analyses. This can be a rigidly tiered approach. An alternative is presented in Figure 1.

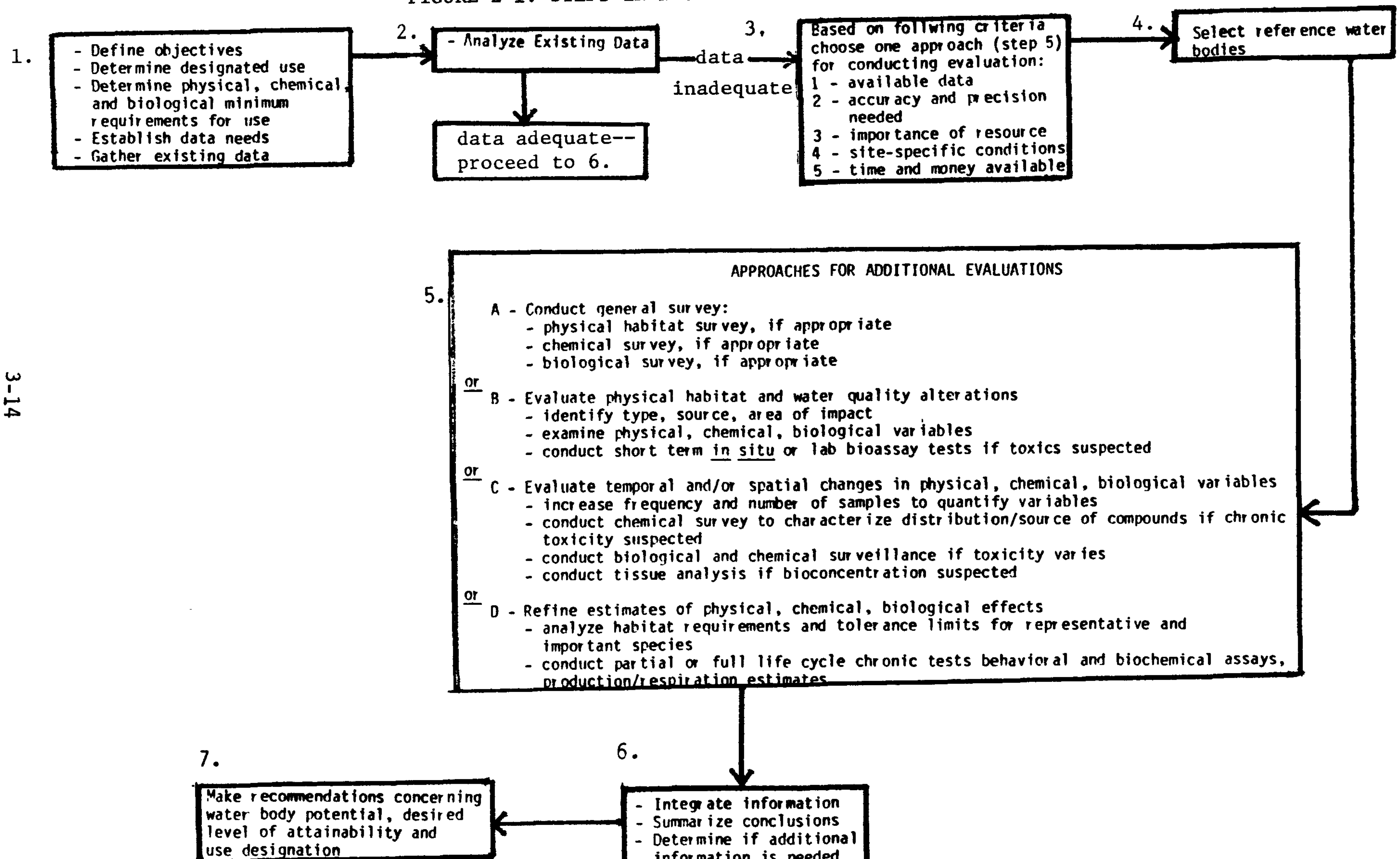
The flowchart is a general illustration of a thought process used to conduct a use attainability analysis. The process illustrates several alternate approaches which can be pursued separately, or to varying degrees, simultaneously depending on:

° the amount of data available on the site;
° the degree of accuracy and precision required;
° the importance of the resource;
° the site-specific conditions of the study area; and
° the controversy associated with the site.

The degree of sophistication is necessarily variable for each approach. Emphasis is placed on evaluating available data first. If this information is found to be lacking or incomplete, then field testing or field surveys should be conducted. A brief description of the major elements of the process is given below.

Steps 1 and 2: These are the basic organizing steps in the evaluation process. By carefully defining the objectives and scope of the evaluation, there will be some indication of the level of sophistication required in subsequent surveys and testing. States and

FIGURE 2-1. STEPS IN A USE ATTAINABILITY ANALYSIS



the regulated community can then adequately plan and allocate resources to the analyses. The designated use of the water body in question should be identified as well as the minimum chemical, physical, and biological requirements for maintaining the use. Minimum requirements may include, for example, dissolved oxygen levels, flow rates, temperature, and other factors. All relevent information on the water body should be collected to determine if the available information is adequate for conducting an appropriate level of analysis. It is assumed that all water body evaluations based on existing data, will either formally or informally be conducted through Steps 1 and 2.

Step 3: If the available information proves inadequate, then decisions regarding the degree of sophistication required in the evaluation process will need to be made. These decisions will, most likely, be based on the 5 criteria listed in Figure 1. Based on these decisions, reference areas should be chosen (Step 4) and one or more of the testing approaches followed.

Steps 5A, B, C, D: These approaches are presented to illustrate, in a general way, several possible ways of analyzing the water body. For example, in some cases chemical data may be readily available for a water body but little or no biological information is known. In this case, extensive chemical sampling may not be required but enough samples should be taken to confirm the accuracy of the available data set. Thus, in order to accurately define the biological condition of the resource, 5C may be chosen, but 5A may be pursued in a less intensive way to supplement the chemical data already available.

Step 5A is a general survey to establish relatively coarse ranges for physical and chemical variables, and the numbers and relative abundances of the biological components (fishes, invertebrates, primary producers) in the water body. Reference areas may or may not need to be evaluated here, depending on the types of questions being asked and the degree of accuracy required.

Step 5B focuses more narrowly on site-specific problem areas with the intent of separating, where possible, biological impacts due to physical habitat alteration versus those due to changes in water quality. These categories are not mutually exclusive but some attempt should be made to define the causal factors in a stressed area so that appropriate control measures can be implemented if necessary.

Step 5C would be conducted to evaluate possibly important trends in the spatial and/or temporal changes associated with the physical, chemical, and biological variables of interest. In general, more rigorous quantification of these variables would be needed to allow for more sophisticated statistical analyses between reference and study areas which would, in turn, increase the degree of accuracy and confidence in the predictions based on this evaluation. Additional laboratory testing may be included, such as tissue analyses, behavioral tests, algal assays, or tests for flesh tainting. Also, high level chemical analyses may be needed, particularly if the presence of toxic compounds is suspected.

Step 5D is, in some respects, the most detailed level of study. Emphasis is placed on refining cause-effect relationships between physical-chemical alterations and the biological responses previously established from available data or steps 5A-5C. In many cases, state-of-the-art techniques will be used. This pathway would only be conducted by the States where it may be necessary to establish, with a high degree of confidence, the cause-effect relationships that are producing the biological community characteristics of those areas. Habitat requirements or tolerance limits for representative or important species may have to be determined for those factors limiting the potential of the ecosystem. For these evaluations, partial or full life-cycle toxicity tests, algal assays, and sediment bioassays may be needed along with the shorter term bioassays designed to elucidate sublethal effects not readily apparent in toxicity tests (e.g., preference-avoidance responses, production-respiration estimates, and bioconcentration estimates).

Steps 6 and 7: After field sampling is completed, all data must be integrated and summarized. If this information is still not adequate, then further testing may be required and a more detailed pathway chosen. With adequate data, States should be able to make reasonably specific recommendations concerning the natural potential of the water body, levels of attainability consistent with this potential, and appropriate use designations.

The evaluation procedure outlined here allows States a significant degree of latitude for designing assessments to meet their specific goals in water quality and water use.

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Appendix A: SAMPLE STATE CLASSIFICATION SYSTEM

States have the responsibility for the development and refinement of use classification systems. The methodology, number of classes and factors to be included in such systems are at the discretion of the States. During the development of this guidance document, several requests were made to include a sample State classification system which is based on a ecosystem evaluation approach. In response to such requests attached is the stream classification guidelines for Wisconsin which includes a stream habitat evaluation. The inclusion of this classification system does not constitute an endorsement or that this system should be adopted in other States. It is provided as information which may be of interest to other States.

STREAM CLASSIFICATION GUIDELINES

FOR WISCONSIN

By

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ABSTRACT

The objective of this classification system is to describe potential stream uses and provide a basis for making and supporting water quality resource management decisions. Only those uses which can be described in terms of biological communities are discussed. "Use" is defined by a class or organisms capable of inhabiting a stream. The "use classes" are: A - cold water sport fish, B - warm water sport fish, C - intolerant forage fish, intolerant macroinvertebrates, or a valuable population of tolerant forage fish, D - tolerant or very tolerant forage or rough fish, or tolerant macroinvertebrates, and E - very tolerant macroinvertebrates or no aquatic life.

The appropriate use class for a stream is determined by comparing the ecological needs of use class organisms with the natural ecological characteristics of a stream system. A set of procedures to evaluate stream system characteristics is presented. Stream system habitat evaluation is stressed. A matrix is used to numerically rank habitat characteristics from excellent to poor. Twelve habitat rating items are listed and include characteristics of the watershed, banks, stream substrate, stream morphology and hydrology, and aesthetics. Other factors used to determine appropriate use classes are background dissolved oxygen, temperature, pH, toxics, and existing biota. A range of values for all of these stream system characteristics is provided which correlated with criteria required to support a specific use class. Although the intent of the system is to provide more objectivity to the classification process, professional judgment of a stream's potential use is still important.

INTRODUCTION

Procedures for classifying Wisconsin streams have been developed to provide a scientific method for designating uses according to a stream's natural ability to support a certain biological community. A specific biological community is termed a "use class." The objective of the classification system is to provide a basis for making and supporting water quality management systems. The need for classifying surface waters is based on the recognition that all surface waters will not support the same level of use, and that different use classes may require different levels of water quality to survive.

To classify streams, and meet both scientific and management objectives, two basic assumptions are necessary: (1) stream systems with similar characteristics will support similar biological communities and can be described as a use class, and (2) if streams within a use class are managed in a similar way they will support a similar use.

Stream classification systems have generally been based on existing conditions; e.g., fish populations, trophic state. The problem with these types of systems is that existing biological communities or trophic state may be a function of controllable pollution, not a function of stream system potential. According to Warren (1979) "classification of stream systems ought not to be based directly on just measurement of stream performance, for then it would have little value for prediction, explanation, understanding and management." He recommended that stream classification systems should be based on "watershed-environment and stream habitat-capacity," not on just biological communities inhabiting a stream when it is classified.

A stream is an ecosystem made up of climate, watershed, banks, bed, water volume, water quality, and biota. A stream's use is dependent upon the natural characteristics of the entire stream ecosystem, and on the cultural alterations or impacts which have occurred or are occurring. Present stream uses are always affected by both natural characteristics and cultural impacts. Potential uses are always affected by natural characteristics, and may be affected by cultural impacts. Since the management goal is to control the cultural impacts affecting stream use, it is logical to base classification on a stream's potential to support a given use in the absence of controllable impacts, not on the present state of the biological community.

To determine the biological community a stream can support, it is necessary to relate the natural characteristics of the whole system to the ecological requirements of use class organisms. A stream classification system structured in this way will predict the potential use of a stream and will also serve to indicate the management necessary to attain the use.

Published stream classification systems based on stream system potential are rare. A few systems include parameters which affect use (Pennak 1971, Platt 1974, Minnesota Pollution Control Agency 1979). However, these systems do not include a method for quantifying data and observations to arrive at an objective classification. Perhaps the reason for this is a lack of information on all the ecological requirements of specific organisms. There is a good data base on how temperature, dissolved oxygen, and other chemical parameters affect aquatic organisms, but not on the influence of habitat. The U.S. Forest Service comes close to providing an adequate stream classification system (U.S. Department of Agriculture 1975). It was developed to quantitatively assess the stability of mountain streams and to identify streams needing intensive management. Some of the parameters in the Forest Service system are not applicable to Wisconsin streams, but the concept is sound, and has been adapted for part of this classification system.

The set of guidelines described in this report is not intended to be a rigid assessment technique. Streams cannot always be realistically classified by a totally objective system. Because of their dynamic nature, biological communities are perhaps the most difficult objects we have chosen to study. Similar stream systems should support similar uses, but each stream is an individual ecosystem and must be classified individually. A stream classification comes down to a final judgment -- a judgment based on measurable factors, and perhaps just as important, on intuition gained from experience and past observation.

FACTORS AFFECTING STREAM USES

A variety of factors affect the ability of a surface water to support certain uses (Table 1). Some are "natural" and are a function of the watershed system in which the stream is embedded. Some are "cultural" and are a function of societal use of the stream system. These natural and cultural factors are characterized as either physical or chemical, and further, they may be controllable or uncontrollable. For the purpose of classification the uncontrollable factors, whether they are natural or cultural, ultimately determine a stream's potential or attainable use. Controllable factors such as point source discharges, which have an impact on stream use, should not influence a stream's classified use. Controllable factors are considered temporary,

TABLE 1. Example of common factors affecting stream uses.

Factor	Comments
Uncontrollable Natural Factors	
1) Flow regime	
2) Habitat structure	Habitat development may be considered in high quality streams
3) Water quality	
Uncontrollable Cultural Factors	
1) Land use	
2) Existing hydrologic modification	
a. Dam	Some management may be possible
b. Straightening	
c. Wetland drainage	
Controllable Cultural Factors	
1) Point sources	These factors are controllable within bounds
a. Municipal	
b. Industrial	
2) Nonpoint sources	
a. Agricultural runoff	
b. Urban runoff	
c. Construction site runoff	
3) Other factors	
a. Water withdrawal	
b. Septic system drainage	
c. Proposed hydrological alterations	

pending implementation of control measures. The effects of some cultural factors may be uncontrollable because they cannot be changed with the application of "reasonable" management. In many cases these cultural factors, and impacts, have become the "natural" characteristics of a stream.

Natural Factors

Since most streams in Wisconsin have been disturbed, it is difficult to define a totally natural factor. For classification, natural factors are defined as the characteristics of a stream system in the absence of direct cultural impacts, such as dams, flow reduction by withdrawal, and point source discharges. Natural factors which affect stream uses are flow, habitat, and "natural" physical or chemical characteristics of water.

Flow Regime

The flow or quantity of water available to support aquatic organisms is of primary importance. It's an obvious fact that large fish species require a higher level of flow than small fish species to survive in a stream. Without adequate flow, large fish would not have room to move, feed or reproduce. Stream flow is directly correlated to the classes of organisms, or uses, a stream is capable of supporting. Flow stability or frequency also becomes an important factor in some streams. Flow stability or frequency also becomes an important factor in some streams. Flow extremes, especially in streams running through altered watershed, can be a major factor in determining appropriate uses.

Habitat Structure

The physical structure and flow of water in a stream interact to create an environment suitable to support various classes of organisms. Substrate, pools and riffles, water depth, erosion and deposition areas, and cover provide necessary habitat. Studies by Gorman and Karr (1978), and Hunt (1971) clearly show that more diverse habitats support more abundant and diverse aquatic communities. A stream with poor habitat structure will support fewer organisms, to the extent that the life support requirements of only very tolerant fish or insects may be met. An analysis of habitat structure is an important factor in the stream classification process.

Water Quality

The natural physico-chemical characteristics of general importance in streams include dissolved oxygen, temperature, suspended solids, and dissolved ions. These parameters are of major concern in determining the ability of a stream to support certain classes of organisms. Water quality extremes are of particular importance. Deviations from water quality criteria levels, even for a short time, may stress aquatic communities beyond recovery.

Natural water quality is influenced by watershed geology, soils, and surface features. Flow regime and instream habitat structure may also have an influence on water quality. To classify a stream into an appropriate use class it's important to determine the natural water quality of a stream system.

Natural factors are generally not controllable. They are the most significant factors in determining the potential uses of a stream.

CULTURAL FACTORS

Culturally induced conditions are those that have been caused by certain actions on the land and in the water. Nearly all waters of the state have been disturbed, in some cases more significantly than others. Cultural factors are broadly defined as point and nonpoint sources of pollution. These factors have an impact on habitat and water quality, and on the uses that may occur in a surface water.

Culturally induced conditions can be further subdivided into controllable and uncontrollable types, or similarly, reversible and irreversible impacts. Theoretically, if cultural impacts are properly managed or removed, an altered environment will revert to its natural state. Grass and trees could be planted instead of corn, and all dams could be dismantled. However, in some cases, actions to control or reserve cultural impacts may not be reasonable.

Uncontrollable Cultural Factors

Uncontrollable cultural factors are those activities over which regulatory agencies have little or no control, or prefer to exercise no control. For purpose of stream classification, two major factors are of concern -- existing land use and hydrologic modifications. These in place activities are generally uncontrollable and may have significant impacts on stream use. When the cause of an impact is uncontrollable, the impact must be considered a normal characteristic of a stream for the purpose of classification.

The present use of land for agriculture and urban development will, in most cases, not change. The impacts of land use on a stream system are not always obvious because they have occurred gradually. For example, removal of native vegetation, destruction of wetlands and paving of streets increases runoff and reduced groundwater recharge. This removal of water may alter the flow regime and water quality of a stream, and affect uses. Such actions may also increase peak flows, resulting in long term and irreversible changes in habitat structure.

A more obvious cultural factor affecting stream use is hydrologic alteration. Existing dams, straightened portions of streams, and wetland drainage are examples of stream alterations which can affect uses and appropriate classifications. The question of controllability of these factors is technically and legally complex, but assuming no regulatory measure can be taken to revert back to an original condition, then these alterations and their impacts must be considered uncontrollable.

Controllable Cultural Factors

Sources of pollution in this category are those that can be controlled by a reasonable level of management. The primary controllable factors are the point sources of wastewater discharge. Programs are in place to regulate what, how, when, and where point sources discharge wastes. Point sources are, within certain bounds, always controllable. The impact of point sources on water quality and stream uses should not be factored into the classification process, assuming the impact can be removed.

Also possibly controllable are activities on the land -- nonpoint sources. Although Wisconsin does not have a program to regulate nonpoint sources* its does have a grant and management program to encourage nonpoint source control. Controllable nonpoint sources, as envisioned here, are those associated with the application of "best management practices" on agricultural and urban lands.

In situations where application of best management practices are likely to result in stream use improvements, the impacts from nonpoint sources should be disregarded in the classification process. However, it may be difficult to show a direct cause and effect relationship between nonpoint sources and water quality. It may be equally difficult to show a direct relationship between nonpoint sources and habitat deterioration except in extreme situations. For instance, even if better land management was applied to a watershed, it may be difficult to predict how long it may take an impacted stream to recover. Classifying a stream to a higher use, based on an anticipated natural improvement, which may or may not take place, may not be logical. In some situations the impact of nonpoint sources on habitat should probably be considered uncontrollable for current actions.

According to Karr and Dudley (1981) nonpoint control efforts that improve water quality may fail to improve the biota of a stream if suitable physical habitats are absent. This does not imply, however, that nonpoint source control efforts are not worthwhile. Over a long time period stream uses will improve, and the effect of nonpoint sources on downstream uses must also be considered.

There are other cultural factors with immediate and direct effects on stream uses which can generally be controlled by regulation. For example, a flow management scheme that results in witholding or diversion of water on a routine basis may preclude certain uses and aquatic populations. Such actions are almost always controllable. Sources of pollution, such as rural septic systems, are controllable. Proposed stream alterations, such as dams and straightening, are

*Wisconsin does have regulatory authority for construction site runoff.

controllable because these are regulated activities. Even an existing dam, already discussed as being uncontrollable, may be managed in certain ways to reduce impacts on stream uses.

Determining the factors affecting stream uses and their status of controllability are the most important parts of this classification procedure. The process of identifying factors and determining controllability serves two important functions: (1) it supplies much of the information required to designate appropriate stream uses, and (2) it identifies the specific management required to achieve designated uses. The most difficult task is determining controllability, especially for nonpoint sources. Another related problem is anticipating the response of a stream to management of pollution sources. To classify streams, subjective judgments regarding the status of these problems will likely have to be made for individual situations.

STREAM USE CLASSES

Stream use classes are listed in Table 2. Stream use is described by the fish species or other aquatic organisms capable of being supported by a natural stream system. Use classes in Table 2 are listed from the most sensitive to the most tolerant use. Common fish species and their representative classification categories are listed in Table 3. The designation of an appropriate use class is based on the ability of a stream to supply habitat and water quality requirements of use class organisms. Sections or "reaches" of a stream may be assigned different use classes, and the same stream or stream reach may be assigned different use classes based on seasonal differences. This concept, termed "seasonal classification," is used to describe variations in stream conditions. For example, a stream may serve as a fish spawning area in the spring, but natural changes in flow or water quality may preclude the existence of fish in other seasons. Following are descriptions of the use classes for classifying Wisconsin streams:

Class A, Cold Water Sport Fish: Streams capable of supporting a cold water sport fishery, or serving as a spawning area for salmonid species. The presence of an occasional salmonid in a stream does not justify a Class A designation (e.g., trout are occassionally taken from the Mississippi River but that fact alone does not justify a cold water sport fish designation).

Class B, Warm Water Sport Fish: Streams capable of supporting a warm water sport fishery, or serving as a spawning area for warm water sport fish.

TABLE 2. Stream use classes for aquatic life

Use Class	Description
A	Capable of supporting cold water sport fish
B	Capable of supporting warm water sport fish
C	Capable of supporting intolerant forage fish*, intolerant macroinvertebrates, or a valuable population of tolerant forage fish
D	Capable of supporting tolerant or very tolerant forage or rough fish*, or tolerant macroinvertebrates
E	Capable of supporting very tolerant macroinvertebrates or no aquatic life

*Refer to Table 3.

Although warm water sport fish are occasionally found in many small streams, a stream should be capable of supporting a "common" designated population to rate a "B" classification.

Class C, Intolerant Forage Fish, Intolerant Macroinvertebrates, or a Valuable Population of Tolerant Forage Fish: Streams capable of supporting an abundant, and usually diverse, population of forage fish or intolerant macroinvertebrates. These streams are generally too small to support cold or warm water sport fish, but have natural water quality and habitat sufficient to support forage fish or macroinvertebrates. Streams capable of supporting valuable populations of tolerant forgage fish should also be included in Class C. This type of stream may provide beneficial uses, such as a food source for a downstream sport fishery, or a sucker fishery.

Class D, Tolerant or Very Tolerant Fish, or Tolerant Macroinvertebrates: Streams capable of supporting only a small population of tolerant forage fish, very tolerant fish or tolerant macroinvertebrates. The aquatic community in such a stream is usually limited due to naturally poor water quality or habitat deficiencies.

Class E, Very Tolerant Macroinvertebrates or No Aquatic Life: Streams only capable at best of supporting very tolerant macroinvertebrates, or an occasional very tolerant fish. Such streams are usually small and severely limited by water quality or habitat. Marshy ditches and intermittent streams are examples of Class E streams.

TABLE 3. Common fish species and classification categories

Sport Fish	Intolerant Forage	Tolerant Forage	Very Tolerant Forage or Rough Fish
Trout (sp)	Stoneroller	Golden Shiner	Carp
Salmon (sp)	Rosyface Shiner	Common Shiner	Goldfish
Northern Pike	Spottail Shiner	Sand Shiner	Goldfish
Muskellunge	Blacknose Shiner	Emerald Shiner	Fathead Minnow
Smallmouth Bass	Blackchin Shiner	Spotfin Shiner	Sheepshead
Largemouth Bass	Dace (sp)	Bluntnose Minnow	Buffalo
Yellow Bass	Hornyhead Chub	Creek Chub	Carp Sucker (sp)
White Bass	Stonecat	Johnny Darter	Gar (sp)
Rock Bass	Tadpole Madtom	Sucker (sp)	Bowfin
Walleye	Redhorse (sp)	Brook Stickleback	Mooneye
Sauger	Darter (sp)-(except Johnny Darter)		
White Crappie	Logperch		
Black Crappie	Sculpin (sp)		
Bluegill			
Sunfish (sp)			
Yellow Perch			
Bullhead (sp)			
Catfish (sp)			
Sturgeon (sp)			

CLASSIFICATION PROCEDURES

The objective of stream classification is to designate logical uses by evaluating and describing stream ecosystems. The classification procedure includes a list of important factors which need to be evaluated, and suggests how to merge data and perceptions into a final decision about appropriate use. Designated uses are based on the relationship and overall quality of all ecosystem components.

The stream classification procedure combines objective and subjective analysis. Objectivity in the procedure comes from pointing out the major individual factors one needs to evaluate, and by placing bounds on ecological "criteria" which separate streams into use classes. However, because ecosystems are extremely complex, professional judgment must also be part of the classification process. This flexibility is needed to allow for logical decisions about stream use.

The following guidelines do not cover all potential situations and should be viewed as starting points from which experience will dictate the scope of an investigation, including what needs to be added or what can be deleted. The classification process requires five basic steps -- study design, data collection, data evaluation, impact controllability analysis, and appropriate use designation:

Study Design

Because of the management objective of this classification procedure, water quality evaluation staff have major responsibility. However, the process should be a "team" effort and, at minimum, should be a cooperative project with fisheries staff. Staff with expertise in other areas may also be required. The team should determine the detail and scope of analysis required to classify any given stream. In some cases, file information coupled with a desk top evaluation may suffice. In complex situations, detailed studies may be needed to reach a credible decision.

Data Collection

Data located in files, studies, reports, etc. should be reviewed. If sufficient current data exist they may be adequate to form the basis for a classification. However, in all cases, a site visit is necessary to verify the evaluation. If current data are insufficient, a stream evaluation must be conducted.

Stream biota are generally dependent upon extreme conditions which normally occur during periods of low flow. Thus, samples, measurements and observations will give a more reliable indication of appropriate use if taken when the stream is at a low or at least normal flow. In situations where seasonal use changes are possible, additional data at higher flows may be needed.

The following data may be required to determine and justify a use class designation:

1. Stream Flow -- The flow of a stream can vary over a wide range and can be expressed in a number of ways.Stream use is often limited by annual low flow which is expressed here as representative low flow. Flow data for many streams are available from the U.S. Geological Survey (USGS), and can be used as points of reference for determining representative low flow. If flow data are not available, it may be necessary to gauge the present flow and obtain a low flow estimate from USGS.

2. Water Quality -- Natural, or background water quality should generally be used as the basis for classification. Daily, and sometimes seasonal water quality extremes determine the class of organisms a stream is capable of supporting. The most extreme water quality conditions normally occur during low flow periods. Thus, an attempt should be made to collect data at that time.

Water samples and instream data should be collected upstream from controllable sources of pollution. In situations were this is impossible, water quality may be a function of the controllable source and can't generally be used as a basis for classification. Many forms of water quality can have an impact on stream use. However, the parameters most directly related to use include dissolved oxygen, temperature and pH. Toxics and other parameters should be measured if a problem is suspected.

3. Habitat Structure -- Habitat evaluation is considered the most important factor in the stream classification process. In situations where water quality data can't be used, habitat may be the only basis for classification. The habitat rating is based on an evalution of watershed, stream banks, and stream bed characteristics. The habitat evaluation and rating procedure is detailed in a separate section.

4. Stream Biota -- The biological communities presently inhabiting a stream including fish, benthic organisms, rooted vegetation, algae, etc. should be determined. This need not be an exhaustive sample collection effort since designation of attainable use will rarely be based totally on biological data. Knowing what organisms are present in a stream helps determine what the appropriate use class should be. Many biological sampling and analysis methods are available. The methods are left to the discretion of the evaluator, but should be described in the classification report.

Data Evaluation

The use class a stream is capable of attaining is determined by comparing stream system data to the life support needs of use class organisms. Table 4 lists a set of stream system parameters and values for each which correspond to the five use classes. The table is used to estimate appropriate stream use based on the quality of individual

Table 4. Physical and chemical criteria guidelines for aquatic life use classes

Parameter	A	B	C	D	E
		Use Class and Criteria			
Dissoved Oxygen	>4	>3	>3	>1	<1
Temperature	<75	<86	<86	<90	>90
pH	>5,<9.5	>5,<10.5	>5,<10.5	>4,<11	<4,>11
Toxics	<acute	<acute	<acute	acute	>acute
Representative Low Flow	>.5	>3	>.2	>.1	≥0
Habitat Rating	<144	<144	<144	>144	>200

parameters. Parameter values and use classes are listed from high to low quality and are intended to be mutually exclusive. Therefore, the lowest class indicated by the lowest quality parameter is the estimated appropriate use of a stream. The values shown in Table 4 are not water quality standards criteria. Rather, values at the extremes are conditions which the particular biota may be able to tolerate for a short time. Criteria in water quality standards are developed to assure protection for sensitive species throughout their life history of exposure. Table 4 values are guides to determine if tolerable conditions exist in a surface water. Even these values should be used with care because observed conditions outside the noted bounds do not necessarily preclude the existence of a use class. The values in Table 4 should be used to evaluate stream system data and be a major factor in the stream clasification process. Following is a description of the parameters in Table 4, and other stream characteristics used in the evaluation procedure.

1. Flow Characteristics -- In this classification system representative low flow most nearly reflects the long-term ability of a stream to support certain organisms. Representative low flow values in Table 4 are based on a review of fish community data from various Wisconsin streams.

Streams receiving an effluent, or are proposed to receive an effluent, should be evaluated as two representative low flows. One based on natural flow, and one based on natural flow plus design effluent flow adds significantly to a stream base flow. For example, an effluent going to an otherwise dry drainage way creates a stream. This procedure involves interpolation of stream conditions at a higher or lower flow, and relies heavily on professional judgement. The purpose is to provide a more complete evaluation and consideration of alternatives upon which to base a logical designation of appropriate use. The procedures also provides more complete information needed by resource managers to base subsequent decisions regarding effluent limits or other management practices.

2. Water Quality Characteristics -- Criteria in Table 4 are maximum or minimum values at which use class biota may be expected to survive during critical periods. If these extreme values were common in a stream, the corresponding biota would probably not be maintained in a healthy state. However, natural short-term fluctuations in water quality are expected in some streams, and values exceeding "standards" do not necessarily preclude associated uses. If water quality is a use limiting factor due to a controllable impact, and natural water quality cannot be determined, appropriate uses should be based on a flow and habitat.

3. Habitat Rating -- The rating values in Table 4 are a numerical ranking of the overall quality of a stream's watershed, banks and bed characteristics. The rating procedure is described in the final section of the classification guidelines. Rating values can range from 56 to 210 and lower number values indicate higher quality habitat. High quality use usually requires high quality habitat. The range of values within a specific use class also gives an indication of the quality of use. For example, a trout stream with a rating of 60 would be expected to support more fish than a trout stream with a rating of 120.

4. Biological Data Evaluation -- The biological community inhabiting a stream may be used as an indication of attainable use, but should generally not form the only basis for use class designation. Most streams are disturbed in some way, and their present biota may be a function of that impact. Thus, present biological communities may not indicate realistic attainable uses under proper management of the sources of impact. Even in streams with no obvious problems, the present organisms may not reflect what otherwise may be a higher quality use. For example, a stream with trout stream characteristics may not contain trout because they were never introduced. The classification of such a stream, if based only on its present community of organisms, may not indicate its true potential use.

The most important use of a biological evaluation is to determine if a water quality problem exists. For example, a stream with flow and habitat characteristic of a high use class, but not supporting that class of organisms, most likely has a water quality problem. It is then necessary to determine the source to the problem and judge if it is controllable or not. If the problem is controllable the classification should be based on flow and habitat. If the problem is uncontrollable the classification may be based on the biological evaluation.

Impact Controllability Analysis

A major objective of the data evaluation process was to identify the factors limiting stream use. The objective of controllability analysis is to determine if those limiting factors can be managed in some way to improve stream use. That is, are the causes of impacts limiting stream use controllable, and further, are the impacts reversible? Controllability was discussed in the factors affecting stream uses section of these guidelines. Table 1 suggested what may or may not be controllable, but no further guidelines are provided. Determining controllability of sources and impacts can be a complex decision point and it may be necessary to obtain help from other staff with experience in the problem area.

Appropriate Use Designation

The use class designated for a stream should be based on Table 4, any other data which may be available, and the professional judgement of the evaluators. There will always be cases that do not conform to a rigid analysis process, and this system is intended to be flexible enough to account for those situations.

The evaluation of small streams receiving or proposed to receive waste dishcarges may result in two possible use designations. When this occurs it will be necessary to recommend one use class as more appropriate. This is one point where the classification process may, and perhaps should, digress from a purely scientific endeavor. Many factors, such as resource value, downstream uses, effluent characteristics and size, and even economics should be considered before recommending a use class designation. As a final consideration, the biological data can serve as a check on the results of the evaluation as follows:

1. If the biological community conforms to the indicated use class report that classification.

2. If the biological community is better than the indicated use class base the classification on the biological evaluation.

3. If the biological community is lower than the indicated use, determine the factors affecting use and if they are controllable or uncontrollable. If the factors are controllable, base the classification on the use indicated by background water quality, flow, and habitat. If the factors are uncontrollable, the classification can be based on the biological evaluation.

To complete the classification process, the evaluators should file a report which recommends a use class, and outlines why the use class is appropriate. A number of management and administrative decisions may be based on the use class. These decisions may be made by people without first-hand knowledge of the stream. Thus, it is important to document all factors, both objective and subjective, which entered into the classification process. In most situations, there are key factors influencing the use class recommendation, and those should be highlighted in the report.

STREAM SYSTEM HABITAT EVALUATION

Stream system habitat is defined as watershed, stream bank, and instream habitat characteristics. Watershed and stream bank characteristics are included because they directly affect instream characteristics -- e.g., flow, depth, substrate, and pool-to-riffle ratio. Stream system habitat is one of the most important factors determining attainable use, and therefore habitat evalution is stressed in this classification procedure. A detailed discussion of stream system habitat evaluation is presented here to insure that, where practical, uniform evaluation procedures are followed.

The purpose of this evaluation procedure is to integrate and rate stream system habitat characteristics in relation to the various use classifications. The final product is a numerical rank or score of habitat quality which is used to help identify the use (Table 4). The evaluation process used here is similar to one developed by the U.S. Forest Service (1975) to assess the stability of mountain streams. Some of the rating characteristics for stream habitats in that system have been adapted and some new parameters added to fit the character of Wisconsin streams.

Following is a description of stream system habitat characteristics and an excellent-to-poor rating scale for each. The evalution form in Appendix 1 provides a method to integrate data and observations of individual characteristics into an overall habitat rating for a stream.

Watershed - The total area of land above the extreme high water line that contributes runoff to a surface water. The character and condition of a watershed affects the character of a stream and stream bed. The portion of a watershed draining directly to a surface water is usually of greatest concern.

1. Erosion - The existing or potential detachment of soil and movement into a stream. Mass movement of soil into a stream results in destruction of habitat and a reduced potential to suppport aquatic life. This item can be rated by observation of watershed and stream characteristics.

a. Excellent: No evidence of mass erosion that has reached or could reach the stream. The water shed is well managed and usually characterized by mature vegetation. The stream shows no evidence of siltation.

b. Good: May be some erosion evident but few "raw" areas. There may be well-managed agricultural fields in the area. Areas that may have eroded in the past are revegetated and stable. The stream shows little evidence of siltation.

c. Fair: Erosion from fields and some raw areas are evident. Heavy storm events are likely to erode soil resulting in periodic high suspended solids in the stream. Some siltation is evident in the stream, and has resulted in destruction of some habitat. Vegetative cover may be sparse and does not appear stable in all areas. There is moderate potential for mass erosion.

d. Poor: Erosion sources are obvious. Almost any runoff will result in detachment of soil from raw areas and cause suspended solids and siltation problems in the stream. Instream habitat may be poor due to siltation. Stream flow may fluctuate widely ("flashy stream").

2. Nonpoint Source Pollution and Other Compromising Factors - This item refers to problems and potential problems other than siltation. Nonpoint source pollution is defined as diffuse agricultural and urban runoff. Other compromising factors in a watershed which may affect attainable use are feedlots, wetlands, septic systems, dams and impoundments, mine seepage, etc. Nonpoint sources and other compromising factors can be a major source of pollutants, or create problems which affect stream use. Examples of potential problems from these sources include pesticides, heavy metals, nutrients, bacteria, temperature, low dissolved oxygen, etc. If these types of problems are suspected, it may be necessary to conduct an intensive study to determine the problem. It is also important to determine if the problem is controllable or not. If the problem is controllable it should not be factored into the habitat evaluation process.

a. Excellent: No evidence of sources or potential sources.

b. Good: No obvious problems, but there may be potential sources such as agricultural fields, farms, etc. The watershed should be well managed to fit this category.

c. Fair: Potential problems evident. Some runoff from farm fields, watershed intensively cultivated, urban area, small wetland area draining to stream, potential for barnyard runoff, small impoundment, etc.

d. Poor: Sources of pollution which may be affecting stream use are evident. Examples of sources are runoff due to poor land management, high use urban or industrial areas, feed lots, impoundments, drainage from large wetlands, mine seepage, tile field drainage, etc. An absence of intolerant organisms in streams with excellent to good habitat may be an indication of the problems.

Stream Banks - The stream channel is composed of an upper and lower bank, and a bottom (Figure 1). The upper band is the land area from the break in the general slope of the surrounding land to the normal

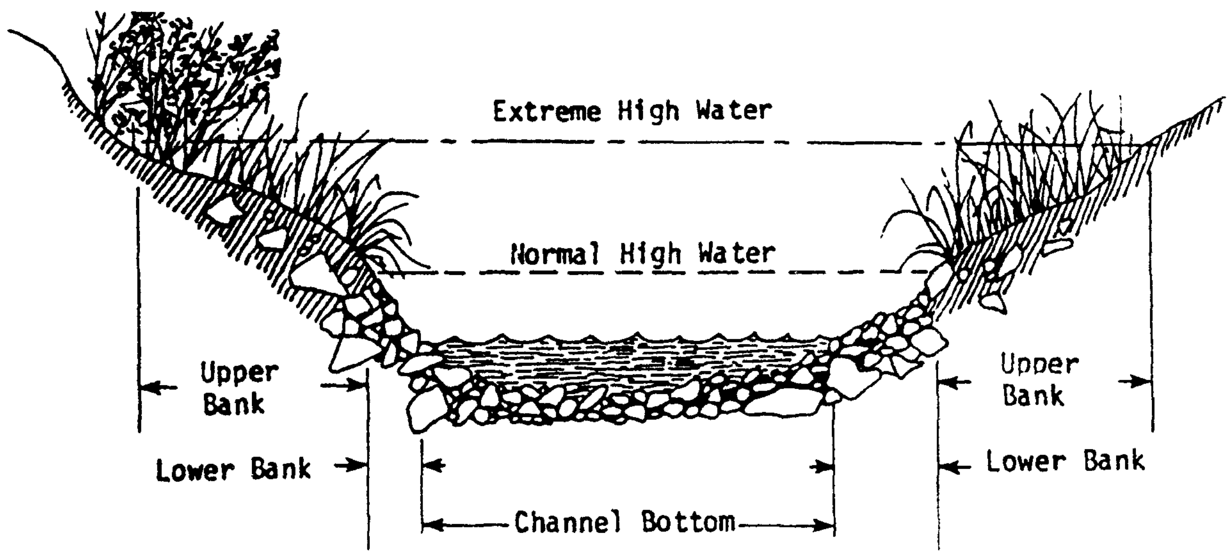

Figure 1. Stream Cross Section

high water line. It is normally vegetated and is covered by water in only extreme high water periods. Land forms vary from wide, flat flood plains to narrow, steep slopes.

The lower bank is the intermittently submerged portion of the stream cross section from the normal high water line to the low water line. The lower channel banks define the stream width. This area varies from bare soil to rock, and the land form may vary from flat to steep.

Stream banks are important in rating stream system habitats because their character and stability directly affect instream characteristics and uses. The evaluation and rating is based on observation of bank characteristics combined with observation of resultant instream characteristics. Habitat rating items 3 and 4 refer to both upper and lower banks because it is sometimes difficult to distinguish a line between the two. Also, the effect on a stream is similar in situations where either bank area is a problem.

3. Bank Erosion, Failure - Existing or potential detachment of soil and movement into a stream. Steeper banks are generally more subject to erosion and failure, and may not support stable vegetation. Streams with poor banks will often have poor instream habitat.

a. Excellent: No evidence of significant erosion or bank failure. Side slopes are generally less than 30% and are stable. Little potential for future problem.

b. Good: Infrequent, small areas of erosion or bank slumping. Most areas are stable with only slight potential for erosion at flood stages. Side slopes up to 40% on one bank. Little potential for major problem.

c. Fair: Frequency and size of raw areas are such that normal high water has eroded some banks. High erosion and failure potential at extreme high stream flows. Side slopes up to 60% on some banks.

d. Poor: Mass erosion and bank failure is evident. Many raw areas are present and are subject to erosion at above normal flow. Erosion and undercutting is evident on bends and some straight channel areas. Side slopes greater than 60% are common and provide large volumes of soil for downstream sedimentation when banks are laterally cut.

4. Bank Vegetative Protection - Bank soil is generally held in place by plant root systems. The density and health of bank vegetation is an indication of bank stability and potential instream sedimentation. Trees and shrubs usually have deeper root systems than grasses and forbs and are, therefore, more efficient in reducing erosion (Khonke and Bertrand 1959). Bank vegetation also helps reduce the velocity of flood flows. Greater density of vegetation is more efficient in reducing lateral cutting and erosion. A variety of vegetation is more desirable than a monotypic plant community.

Vegetative protection is important in evaluating the long term potential for erosion, and stability of the stream system. The evaluation and rating is based on observation of existing vegetation, erosion, and instream conditions.

a. Excellent: A variety of vegetation is present and covers more than 90% of the bank surface. Any bare or sparsely vegetated areas are small and evenly dispersed. Growth is vigorous and reproduction of species is proceeding at a rate to insure continued ground cover. A deep, dense root mat is inferred.

b. Good: A variety of vegetation is present and covers 70-90% of the bank surface. Some open areas with unstable vegetation are evident. Growth vigor is good for all species but reproduction may be sparse. A deep root mass is not continuous and erosion is possible in openings.

c. Fair: Vegetative cover ranges from 50-70% and is composed of scattered shrubs, grasses and forbs. A few bare or sparsely vegetated areas are evident. Lack of vigor and reproduction is evident in some individuals or species. This condition is ranked a fair due to the percent of area not covered by vegetation with a deep root system.

d. Poor: Less than 50% of the banks covered by vegetation. Vegetation is composed of grasses and forbs. Any shrubs or trees exist as individuals or widely scattered clumps. Many bare or sparsely vegetated area are obvious. Growth and reproduction vigor is generally poor. Root mats are discontinuous and shallow.

5. Channel Capacity - Channel width, depth, gradient, and roughness determine the volume of water which can be transmitted. Over time, channel capacity adjusts to the size of watershed, climate, and changes in vegetation (stability). When channel capacity is exceeded, unstable areas are likely to erode resulting in habitat destruction. Indicators of this problem are deposits of soil on the lower banks and organic debris found hung up in bank vegetation. The objective in rating this item is to estimate normal peak flow and if the present lower bank cross section is adequate to carry the load without bank deterioration.

The ability of a stream channel to contain flood flows can be estimated by calculating the width-to-depth ratio (W/D ratio). The W/D ratio is calculated by dividing the the average top width of the lower bank by the height of the lower bank. This item is rated by the W/D ratio, and by observing the condition of banks, position of debris, and instream siltation.

a. Excellent: The stream channel is adequate to contain peak flow volumes plus some additional flow. Overbank floods are rare. W/D ratio less than 7; i.e., 36 ft. wide divided by 6 ft. deep = 6.

b. Good: The stream channel is adequate to contain most peak flows. W/D ratio of 8 - 15.

c. Fair: The channel can barely contain normal peak flows in average years. W/D ratio of 15 - 25.

d. Poor: The channel capacity is obviously inadequate. Overbank flow are common as indicated by condition of banks and accumulation of debris. W/D ratio greater than 25.

6. Bank Deposition - The character of above water deposits is an indication of the severity of watershed and bank erosion, and stability of the stream system. Deposits are generally found on the lee side of rocks and other objects which deflect flow. These deposits tend to be short and narrow. On flat lower banks, deposition during recesssion from peak flows may be quiet large. The growth, or appearance of bars where they did not previously exist is an indication of upstream erosion. These bars tend to grow in depth and length with continued watershed disturbance. Deposition may also occur on the inside of bends, below channel constrictions, and where stream gradient flattens out. This item is evaluated and rated by observation.

a. Excellent: Little or no fresh deposition on point bars or on the lee side of obstructions. Point bars appear stable.

b. Good: Some fresh deposits on old bars and behind obstructions. Sizes tend to be of larger sized coarse gravel and some sand, very little silt.

c. Fair: Deposits of fresh, fine gravel, sand and silt observed on most point bars and behind obstructions. Formation of a few new bars is evident, and old bars are deep and wide. Some pools are partially filled with fine material.

d. Poor: Extensive deposits of fine sand or silt on bars and along banks in straight channels. Accelerated bar development. Most pool areas are filled with silt.

Stream Bottom - The portion of the stream channel cross section which is totally on aquatic environment (Fig. 1). The character and stability of bottom material is important in determining stream use because this area provides habitat necessary to support aquatic life. A variety of stable habitat, which provides area for feeding, resting and reproduction, will generally support a higher class of organisms. Stream bottom characteristics are evaluated and rated by observation. The evaluation should be conducted when the stream is free of suspended material to enhance observation.

7. Scouring and Deposition - This item relates to the destruction of instream habitat resulting from most of the problems defined under 1 through 6 above. Deposition material comes from watershed and bank erosion. Scouring results from high velocity flows and is a function of watershed characteristics, stream hydrology, and stream morphology. Characteristics to look for are stable habitat and degree of siltation in pools and riffles. Shallow, uniform stream stetches ("flat areas") may be considered either scoured or silted, depending on stream velocity. The rating is based on an estimate of the percent of an evaluated reach that is scoured or silted; i.e., 50 ft. silted in a 100 ft. stream length equals 50%.

a. Excellent: No significant scouring or deposition is evident. Up to 5% of the stream reach evaluated may be scoured or silted; i.e., 0-5 ft. in a 100 ft. stream reach.

b. Good: Some scouring or deposition is evident but a variety of good habitat is still present. Scouring is evident at channel constriction or where the gradient steepens. Deposition is in pools and backwater areas. Sediment in pools tend to move on through so pools change only slightly in depth. The affected area ranges from 5 to 30% of the evaluated reach.

c. Fair: Scoured or silted area covers 30 to 50% of the evaluated stream reach. Scouring is evident below obstructions, at constrictions, and on steep grades. Deposits tend to fill and decrease the size of some pools. Riffles areas are not significantly silted.

d. Poor: Scouring or deposition is common. More than 50% of evaluated stream reach is affected. Few deep pools are present due to siltation. Only the larger rocks in riffle areas remain exposed. Bottom silt may move with almost any flow above normal.

8. Bottom Substrate - This item refers to the availability of habitat for support of aquatic organisms. A variety of substrate material and habitat types is desirable. Different organisms are adapted to different habitats; thus, a variety of habitat is necessary for development of a diverse community. The presence of rock and gravel in flowing streams is generally considered more desirable habitat, However, other forms of habitat may provide the niches required for community support. For example, trees, tree roots, vegetation, undercut banks, etc., may provide excellent habitat for a variety of organisms. This item is evaluated and rated by observation. The evaluation should be conducted when stream flow is at a normal or lower stage to enhance observation.

a. Excellent: Greater than 50% stable habitat. Rocks, logs, etc. provide shelter. Gravel, debris, riffle areas provide habitat for insects and feeding areas for fish.

b. Good: Stable habitat in 30 to 50% of the stream reach evaluated. Habitat is adequate for development and maintenance of fish and insects communities.

c. Fair: 10-30% stable habitat. Habitat is approaching a monotypic type and may have a limiting effect on fish and insect populations. Habitat is less than desirable.

d. Poor: Less than 10% stable habitat. Almost no habitat available for shelter or development of a desirable insect or fish community. Lack of habitat is obvious.

Stream Morphology and Flow - The rating items in this category include depth, flow, and run-to-riffle or pool-to-bend ratio. These stream characterisitics are closely related to previous rating items. Stream depth, morphology and flow are a function of watershed characteristics and climate. They may be the most important evaluation parameters because they relate to the volume of water and habitat available to provide life support requirements i.e., shelter, food and reproduction needs. Low stream flow and shallow depth can be major limiting factors preventing a certain use. Stream morphology relates to habitat and can also become a limiting factor.

In situations where effluent flow significantly adds to or subtracts from natural stream flow, the stream should be evaluated under both flow conditions. This procedure applies to the Average Depth and Stream Flow rating items.

9. Average Depth at Representative Low Flow - Average stream depth is estimated by measuring the maximum depth in riffles and pools, adding those depths and dividing by the total number of riffles and pools. This rough estimate should be adequate because it relates to the ability of a stream to provide a medium for shelter and movement. It may not be practical to measure depth at a representative low flow. However, if a stream is evaluated at average or lower flow, a

representative low flow depth can be reasonable estimated. The representative low flow depth is rated because it is a better expression of prevailing conditions and the uses possible in a stream most of the time. The following rating depths are based on depths of streams in southern Wisconsin known to support various communities. The rating depths are general guidelines only. For example, a cold water stream with an average depth less than 24 inches may deserve an excellent rating if otherwise excellent habitat is available.

a. Excellent: Average depth greater than 24 inches. Riffle depths allow for free passage of fish and shelter when feeding. Pool depths provide security and ample space for several fish, even at a very low flow.

b. Good: Average depth 12-24 inches. Most riffles allow free passage and shelter at normal flow conditions. Most pools provide adequate shelter under all but very low flow conditions.

c. Fair: Average depth 6-12 inches. Many riffles are too shallow for free passage of fish at normal flow. Some habitat is provided by pools but only at normal or higher flow. Depth may be sufficient to support forage species and macroinvertebrates.

d. Poor: Average depth less than 6 inches. Riffles are shallow, even at normal flow. Pools and flat area are shallow and uniform in depth. Little cover available for any fish species. Stream may cease to flow in very dry periods.

10. Stream Flow, at a Representative Low Flow - Stream flow relates to the ability of a stream to provide and maintain a stable aquatic environment. The rating flows are based on a review of Surface Water Resources of Wisconsin Counties publications, Wisconsin Department of Natural Resources. Flows were compared to species of fish known to inhabit streams.

a. Excellent: Stream flow greater than 5 cfs for warm water streams, and greater than 2 cfs for cold water streams. These values are based on the potential of a stream to support warm or cold water sport fish.

b. Good: Stream flow 2 to 5 cfs for warm water streams, and 1 to 2 cfs for cold water streams. Surface water resources data for Wisconsin indicates many warm water streams, with good habitat, in this flow range support sport fish. Other streams, with good water quality, support diverse forage fish populations. Many cold water streams in this flow range will support trout, if habitat is good.

c. Fair: Stream flow 0.5 to 2 cfs for warm water streams, and 0.5 to 1 cfs for cold water streams. These stream flows are sufficient to support forage species in warm water. Cold water streams in this flow range may support a few trout. Streams with exceptional habitat may support a fishable trout population. Many cold water streams in this range will support diverse forage fish and macroinvertebrate populations.

d. Poor: Stream flow less than 0.5 cfs for both warm and cold water streams. Streams in this category may become intermittent in dry periods. Streams with exceptional water quality and habitat may support forage fish, or even serve as spawning or nursery areas for trout.

11. Pool/Riffle or Run/Bend Ratio - This rating item assumes a stream with a mixture of riffles or bends contains better habitat for community development than a straight or uniform depth stream. "Bends" refer to a meandering stream. Bends are included because some low gradient streams may not have riffle ares, but excellent habitat can be provided by the cutting action of water at bends. The ratio is calculated by dividing the average distance between riffles or bends by the averge stream width. If a stream contains both riffles and bends, the most dominant feature which provides the best habitat should be used.

a. Excellent: Pool-to-riffle or run-to-bend ratio to 5-7. Pools are deep and provde good habitat. Riffles are deep enough for free passage of fish.

b. Good: Pool-to-riffle or run-to-bend ratio of 7-15. Adequate depth in pools and riffles.

c. Fair: Pool-to-riffle- or run-to-bend ratio of 15-25. Occasional riffle or bend. Variable bottom contours may provide some habitat.

d. Poor: Pool-to-riffle or run-to-bend ration greater than 25. Essentially a straight and uniform depth stream. Little habitat of any kind.

12. Aesthetics - This rating item does not necessarily relate to the ability of a stream to support aquatic life. However, people's perception of what constitutes a desirable surface water is important. Even though a stream may not be capable of supporting high-use-class orgnaisma, it may have desirable aesthetic qualities which deserve protection. It is not possible to guide everyone to a uniform aesthetic rating decision. However, various studies have been conducted on what most people consider as aesthetics when viewing a a setting. The various factors important in this evaluation include:

1. Visual pattern quality
2. Land husbandry
3. Degree of change
4. Recovery potential
5. Naturalness
6. Geological values
7. Historical values
8. Flora and fauna diversity

a. Excellent: The stream or stream section has wilderness characteristics, outstanding natural beauty, or flows through a wooded or unpastured corridor.

b. Good: High natural beauty -- trees, historic site. Some watershed development may be visible such as agricultural fields, pastures, some dwellings. Land in use is well managed.

c. Fair: Common setting, but not offensive. May be a developed but uncluttered area.

d. Poor: Stream does not enhance aesthetics. Condition of stream is offensive, and recovery without extensive renovation of watershed and stream is unlikely.

Habitat Rating Procedure - The habitat characteristics described are rated from excellent to poor on the form provided at the end of this section. The habitat score obtained from the rating form is used in Table 4 to assist in determining attainable stream use. The rating numbers are relative to one another from excellent to poor, and number values are weighted to give more important rating items (depth, flow, substrate) more significance in the total score. It is the proportion of the rating values to one another that is important, not the actual number value.

The rating form is completed using field measurements, observations, maps, aerial photos, etc. If a stream is divided into segments, a separate form is used for each one. One of the numbers best describing the condition of the rating item is circled. If the actual conditions fall somewhere between the conditions described, the number is crossed out and an intermediate number that better describes the situation is written in. When all items have been rated the total score in each column is added up and the column scores totalled for a final ranking score.

The rating items are interrelated so do not dwell on any one item for long. Avoid keying in on a single indicator unless it has significant impact on the stream's potential to support aquatic life. The weight given to more important items is intended to account for this. In this system a stream with excellent characteristics will receive a lower number score than one with poor characteristics, i.e., the lower the score, the better the stream system habitat.

The rating form should be completed in the field to insure all items are rated at the site. The descriptions are intended to stimulate mental images of indicator conditions which lead to consistent, reproducible habitat ratings by different evaluators.

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STREAM SYSTEM HABITAT RATING FORM

Stream ________ Reach Location ________________ Reach Score/Rating ________

County ________ Date ________ Evaluator ________________ Classification ________

Rating Item	Category: Excellent		Good		Fair		Poor	
1. Watershed Erosion	No evidence of significant erosion. Stable forest or grass land. Little potential for future erosion.	8	Some erosion evident. No significant "raw" areas. Good land mgnt. practices in area. Low potential for significant erosion.	10	Moderate erosion evident Erosion from heavy storm events obvious. Some "raw" areas. Potential for significant erosion.	14	Heavy erosion evident Probable erosion from any runoff.	16
2. Watershed Nonpoint Source	No evidence of significant source. Little potential for future problem.	4	Some potential sources (roads, urban area, farm fields).	8	Moderate sources. (Small wetlands, tile fields, urban area. Intense agriculture).	16	Obvious sources. (Major wetland drainage, high use urban or industrial area, feed lots, impoundment).	20
3. Bank Erosion Failure	No evidence of significant erosion or bank failure. Little potential for future problem.	6	Infrequent, small areas, mostly healed over. Some potential in extreme floods.	9	Moderate frequency and size. Some "raw" spots. Erosion potential during high flow.	15	Many eroded areas. "Raw" areas frequent along straight sections and bends.	18
4. Bank Vegetative Protection	90% plant density. Dirverse trees, shrubs, grass. Plants healthy with apparently good root system.	6	70-90% density. Fewer plant species. A few barren or thin areas. Vegetation appears generally healthy.	9	50-70% density. Dominated by grass, sparse trees and shrubs. Plant types and conditions suggest poorer soil binding.	15	<50% density. Many raw areas. Thin grass, few if any trees and shrubs	18
5. Lower Bank Channel Capacity	Ample for present peak flow plus some increase. Peak flows contained. W/D ratio ≤7.	8	Adequate. Overbank flows rare. W/D ratio 0-15.	10	Barely contains present peaks. Occasional overbank flow. W/D ratio 15 to 25.	14	Inadequate, overbank flow common. W/D ratio >25.	16
6. Lower Bank Deposition	Little or no enlargement of channel or point bars.	6	Some new increase in bar formation, mostly from course gravel.	9	Moderate deposition of new gravel and course sand on old and some new bars.	15	Heavy deposits of fine material. Increased bar development.	18
7. Bottom Scouring and Deposition	Less than 5% of the bottom affected by scouring and deposition.	4	5 to 30% affected. Scour at constrictions and where grades steepen. Some deposition in pools.	8	30 to 50% affected. Deposits and scour at, obstructions, constrictions and bends. Some filling of pools.	16	More than 50% of the bottom changing nearly year long. Pools almost absent due to deposition.	20
8. Bottom Substrate	Greater than 50% rubble, gravel or other stable habitat.	2	30 to 50% rubble, gravel or other stable habitat. Adequate habitat.	7	10 to 30% rubble, gravel or other stable habitat. Habitat availability less than desirable.	17	Less than 10% rubble, gravel or other stable habitat. Lack of habitat is obvious.	22
9. Average Depth at Rep. Low Flow	Greater than 24".	0	12" to 24".	6	6" to 12".	18	Less than 6".	21
10. Flow, at Rep. Low Flow	Warm water >5 cfs. Cold water, >2 cfs.	0	Warm water, 2 to 5 cfs. Cold water, 1 to 2 cfs.	6	Warm water, .5 to 2 cfs. Cold water, .5 to 1 cfs. Continuous flow.	18	Less than .5 cfs. Stream may cease to flow in very dry years.	24

Rating Item	Category							
	Excellent		Good		Fair		Poor	
11. Pool/Riffle, Run/Bend Ratio	5 to 7. Variety of habitat. Deep riffles and pools.	4	7 to 15. Adequate depth in pools and riffles. Bends provide habitat.	9	15 to 25. Occassional riffle or bend. Bottom contours provide some habitat.	16	Greater than 25. Essentially a straight stream. Generally all "flat water" or shallow riffle. Poor habitat.	28
12. Aesthetics	Wilderness characteristics, outstanding natural beauty. Usually wooded or unpastured corridor.	8	High natural beauty. Trees, historic site. Some development may be visible.	10	Common setting, not offensive. Developed but uncluttered area.	14	Stream does not inhance aesthetics. Condition of stream is offensive.	16

Column Total Without Effluent --

Column Total With Effluent --

Add Column Scores Without Effluent, E___ + G___ + F___ + P___ = Reach Score

Add Column Scores with Effluent, E___ + G___ + F___ + P___ = Reach Score

≤70 = Excellent, 71-129 = Good, 130-200 = Fair, >200 = Poor

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Appendix B: FISH TAXONOMIC REFERENCES

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Appendix D: 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.

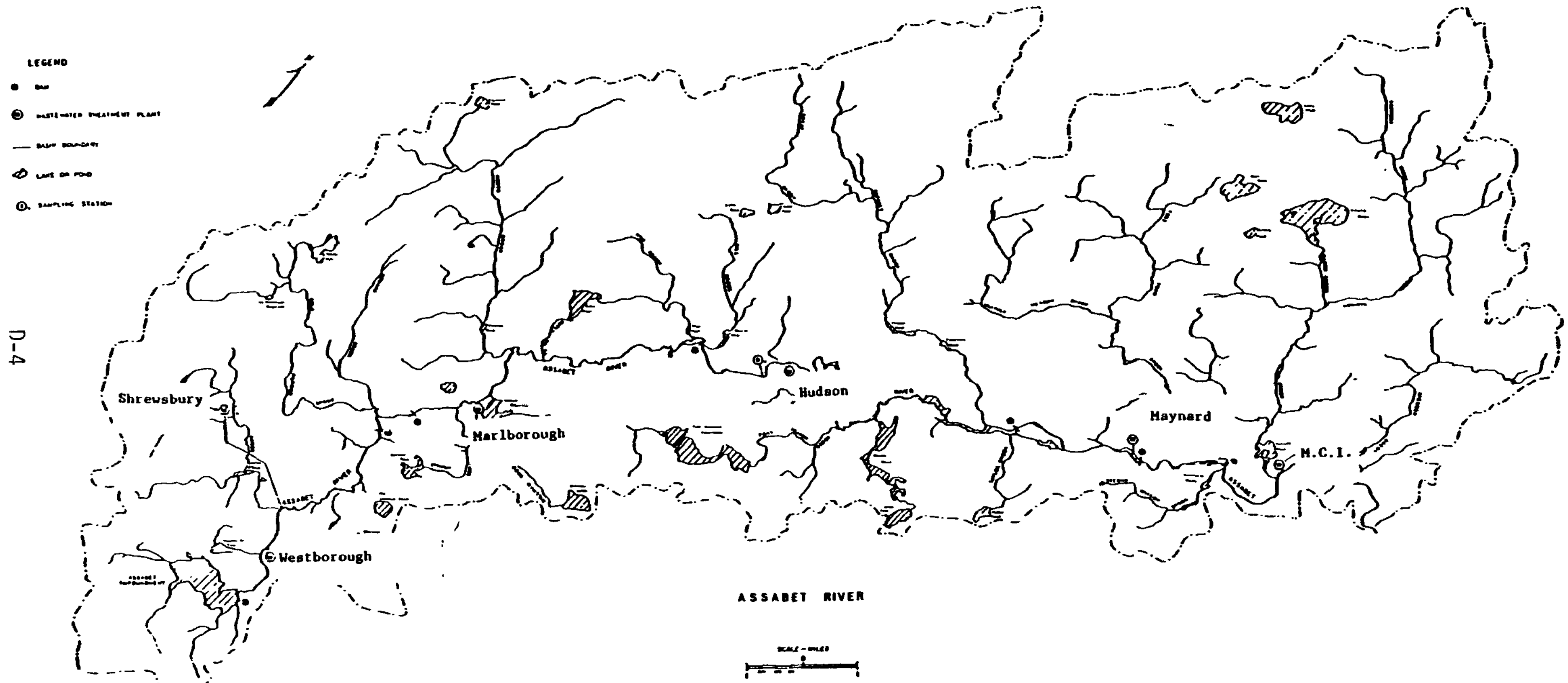


Figure 1 ASSABET RIVER DRAINAGE SYSTEM

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

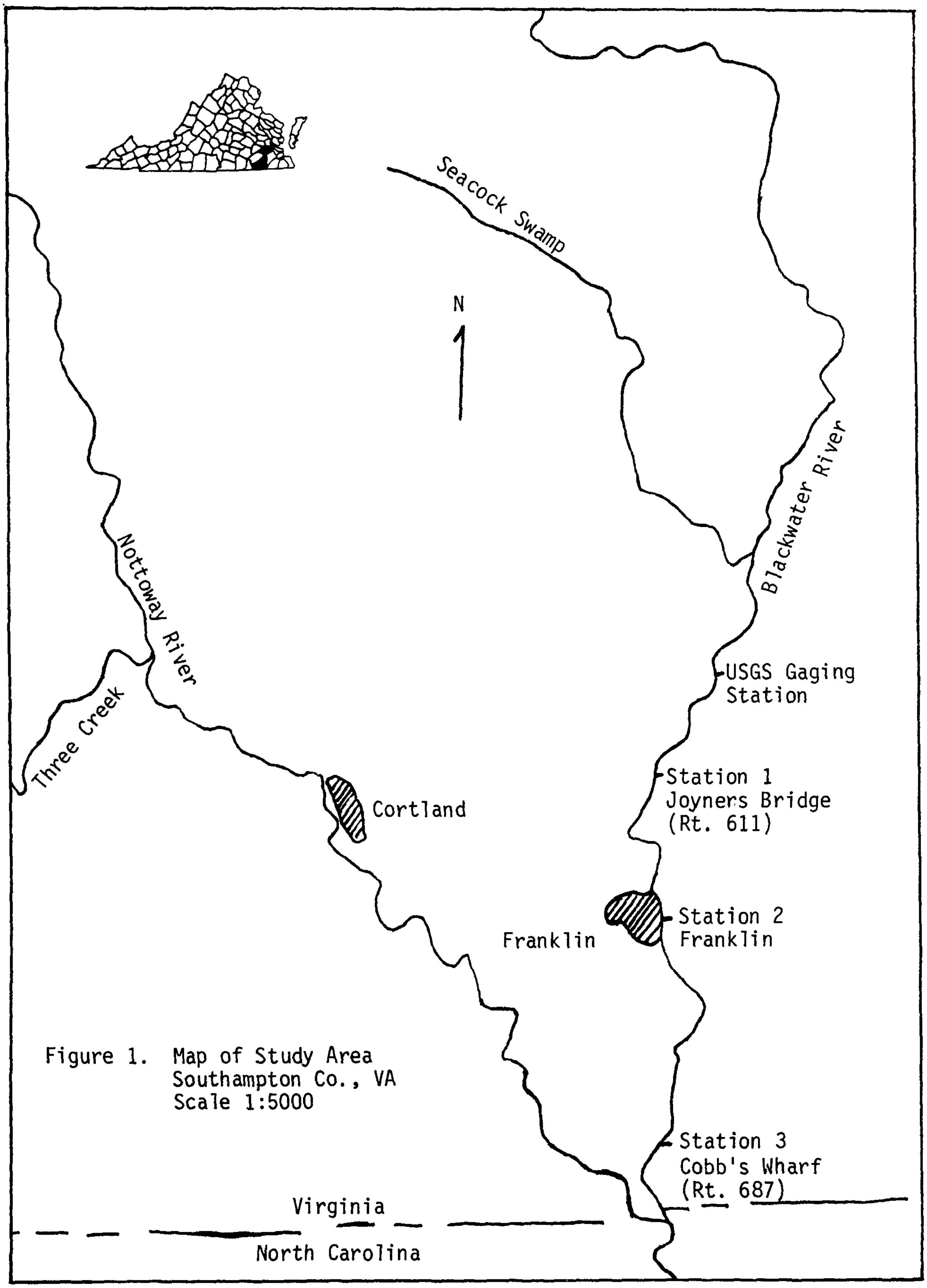


Figure 1. Map of Study Area
Southampton Co., VA
Scale 1:5000

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

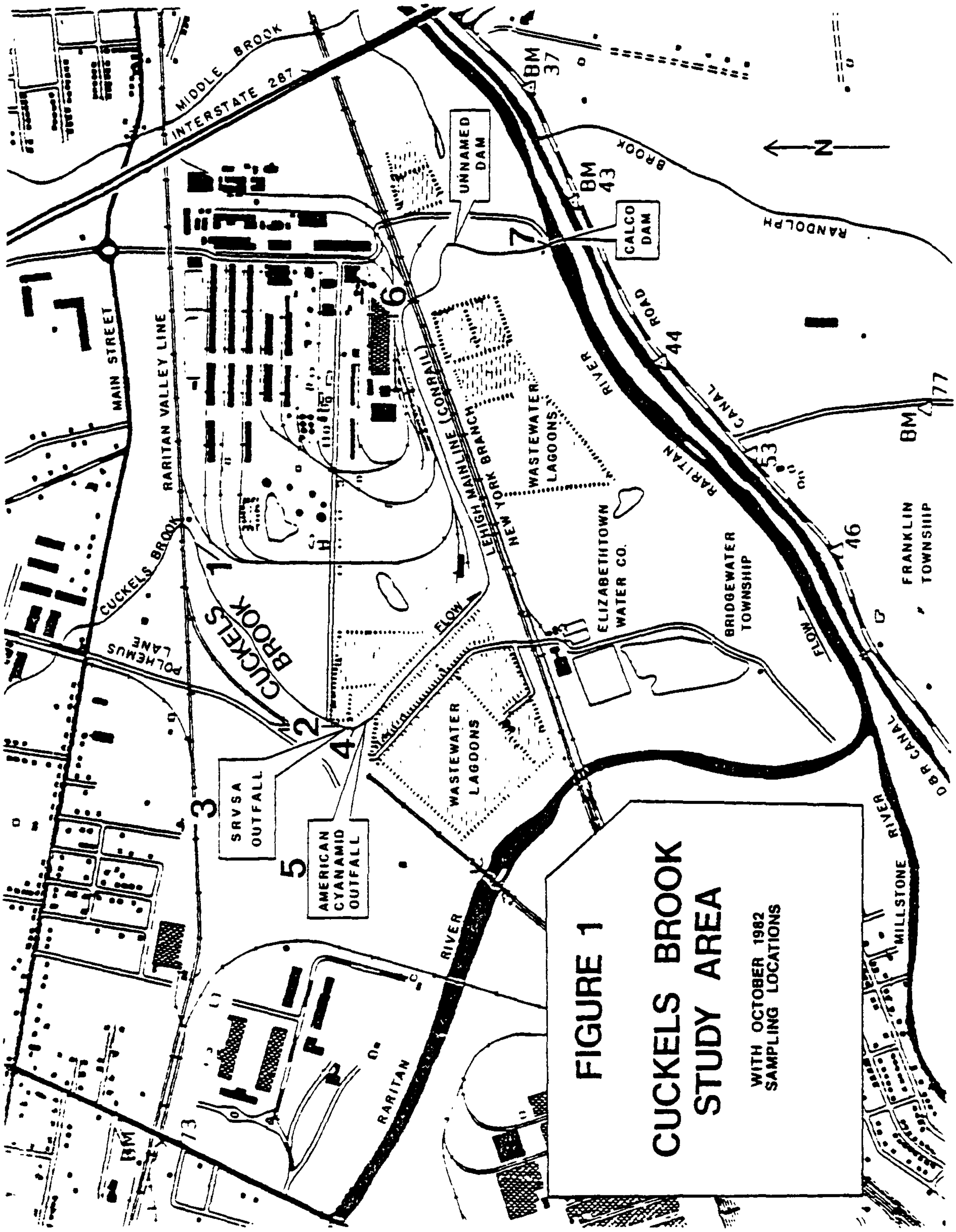


FIGURE 1
CUCKELS BROOK
STUDY AREA
WITH OCTOBER 1982
SAMPLING LOCATIONS
MIDDLE BROOK
INTERSTATE 287
MAIN STREET
RARITAN VALLEY LINE
CUCKELS BROOK
CUCKELS BROOK
POLHEMUS LANE
1
2
3
4
5
6
7
SRVSA
OUTFALL
AMERICAN
CYANAMID
OUTFALL
FLOW
WASTEWATER
LAGOONS
WASTEWATER
LAGOONS
LEHIGH MAINLINE (CONRAIL)
NEW YORK BRANCH
UNNAMED
DAM
CALCO
DAM
RANDOLPH BROOK
RIVER ROAD
RARITAN RIVER
CANAL
ELIZABETHTOWN
WATER CO.
BRIDGEWATER
TOWNSHIP
FRANKLIN
TOWNSHIP
FLOW
RARITAN
RIVER
MILLSTONE RIVER
D&R CANAL
BM
37
BM
43
44
53
46
BM
77
N

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 Nontrout 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.

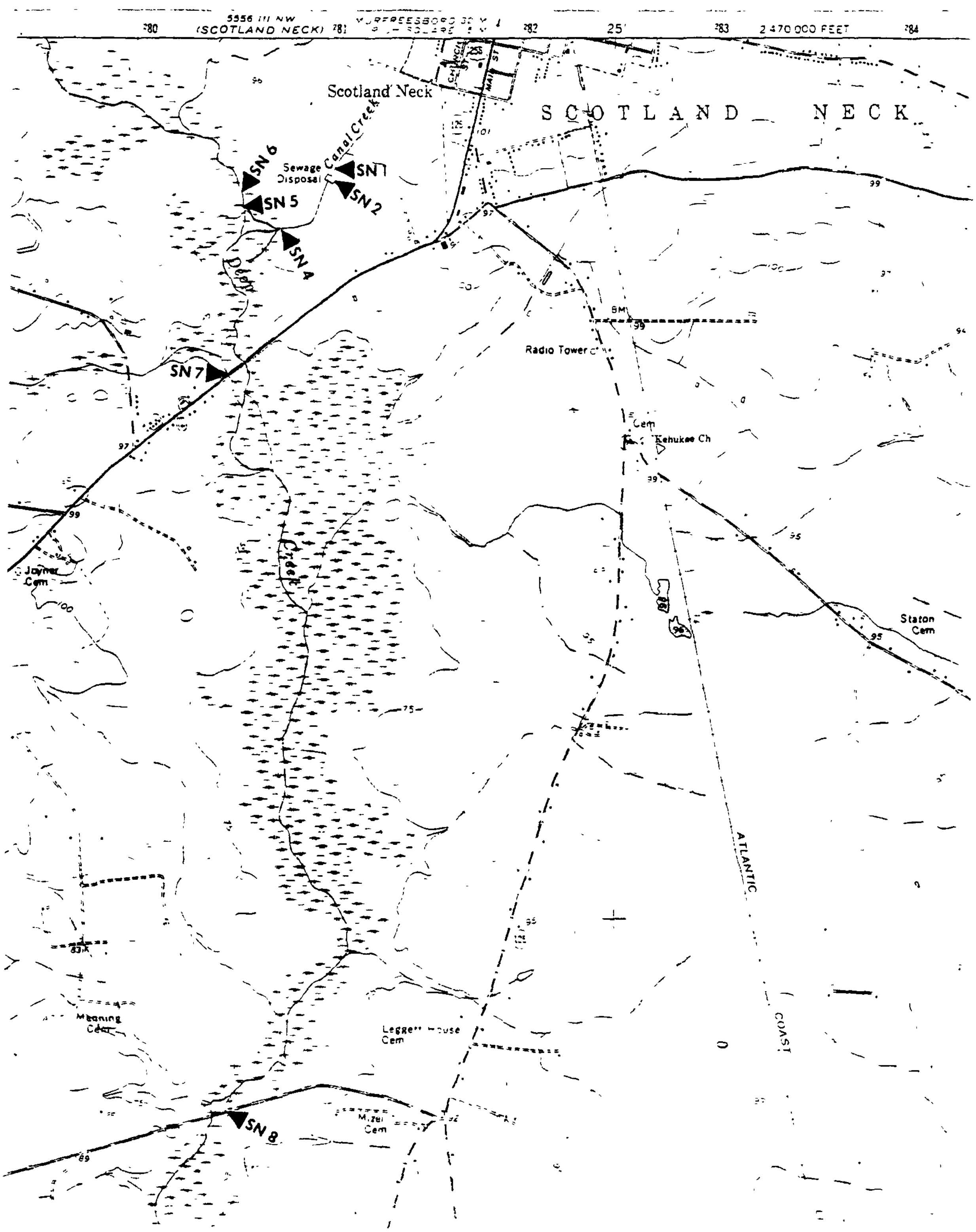


Figure 1. Study Area, Deep Creek and Canal Creek

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.

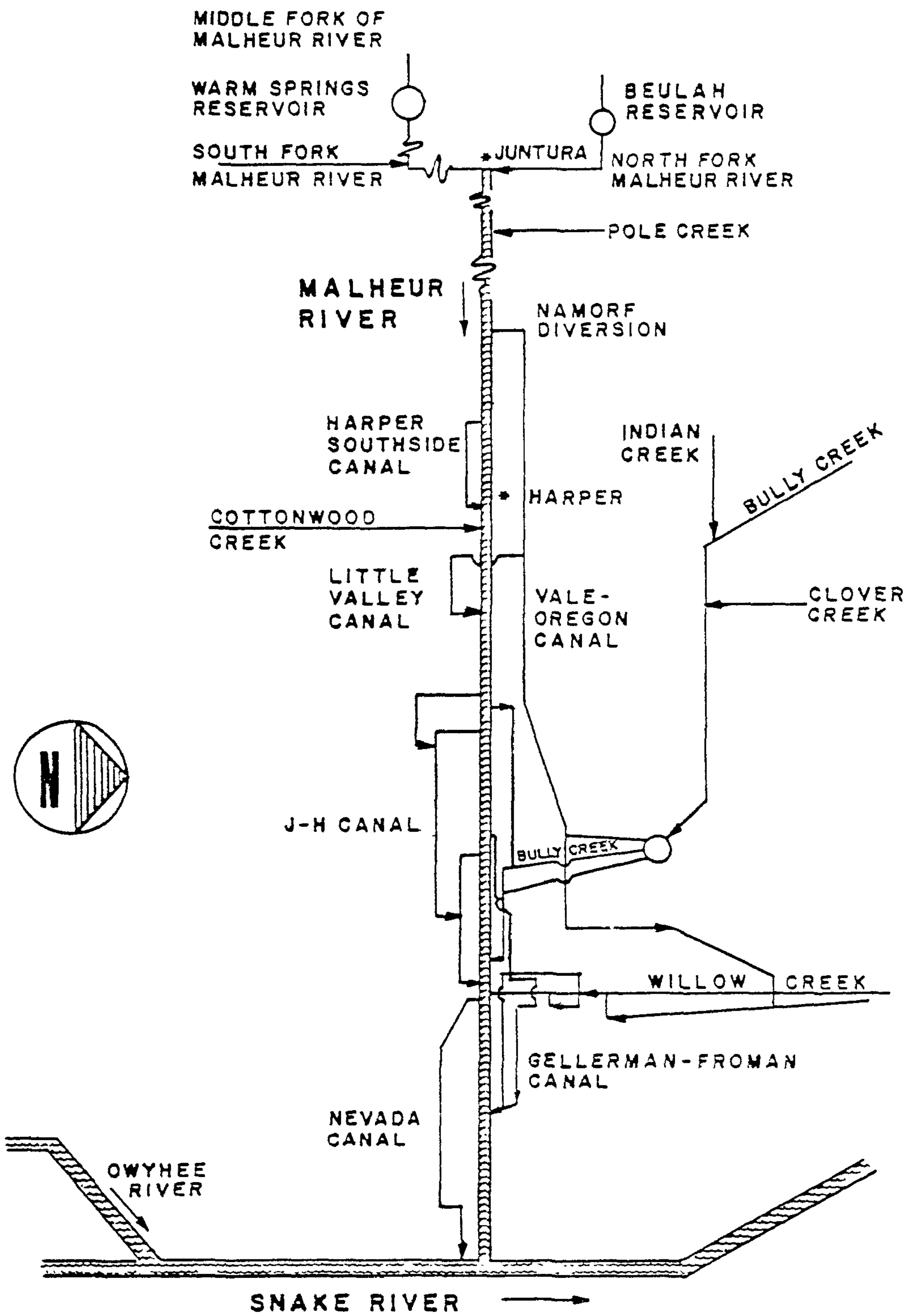
Irrigation water is delivered to individual farms by a complicated system of canals and laterals. Additional water is obtained from drainage canals and groundwater sources. An integral part of the water distribution system is the use and reuse of irrigation return flows five or six times before it is 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.

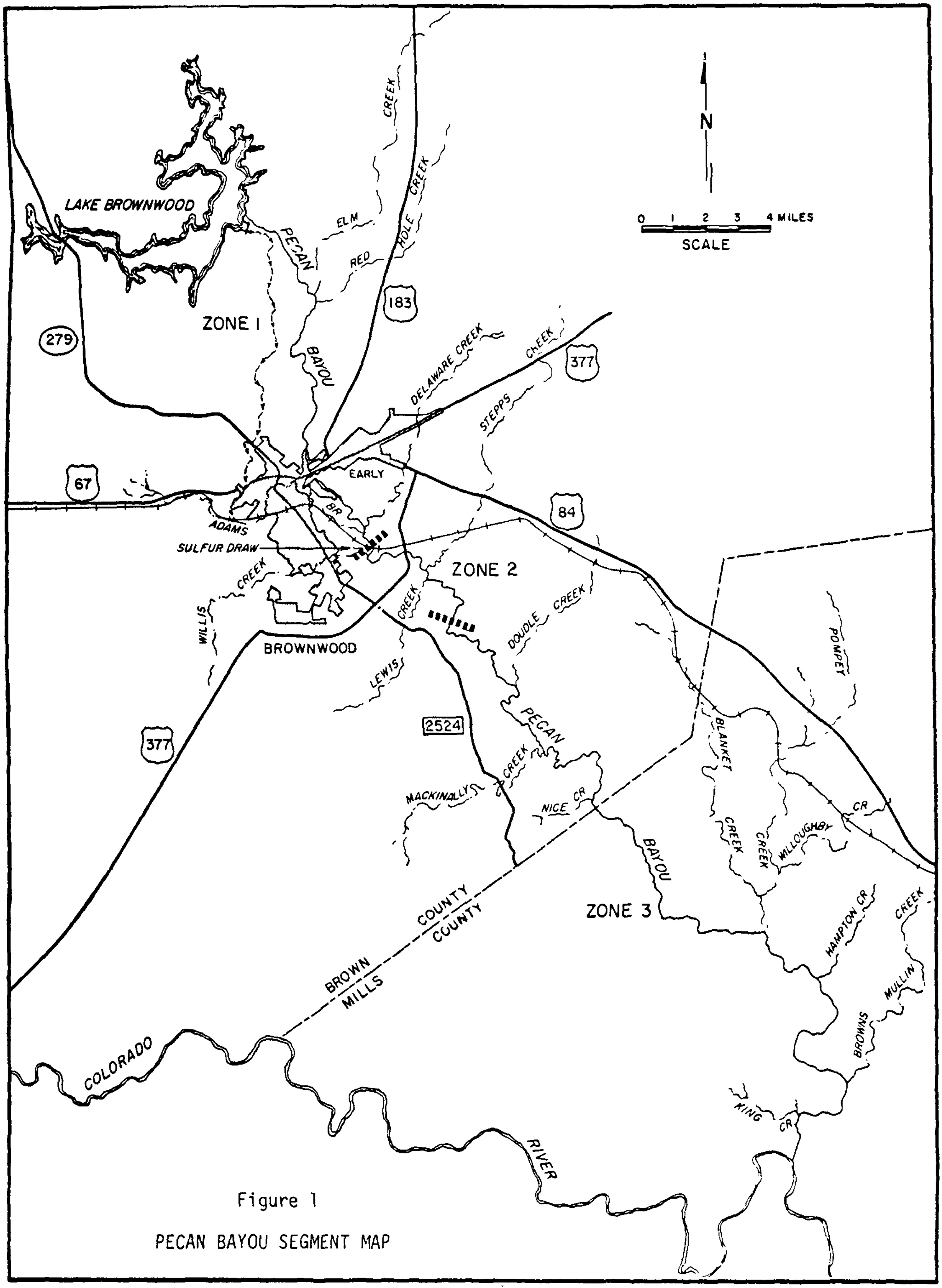


Figure 1

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. <u>ANALYSES CONDUCTED</u>

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 (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.

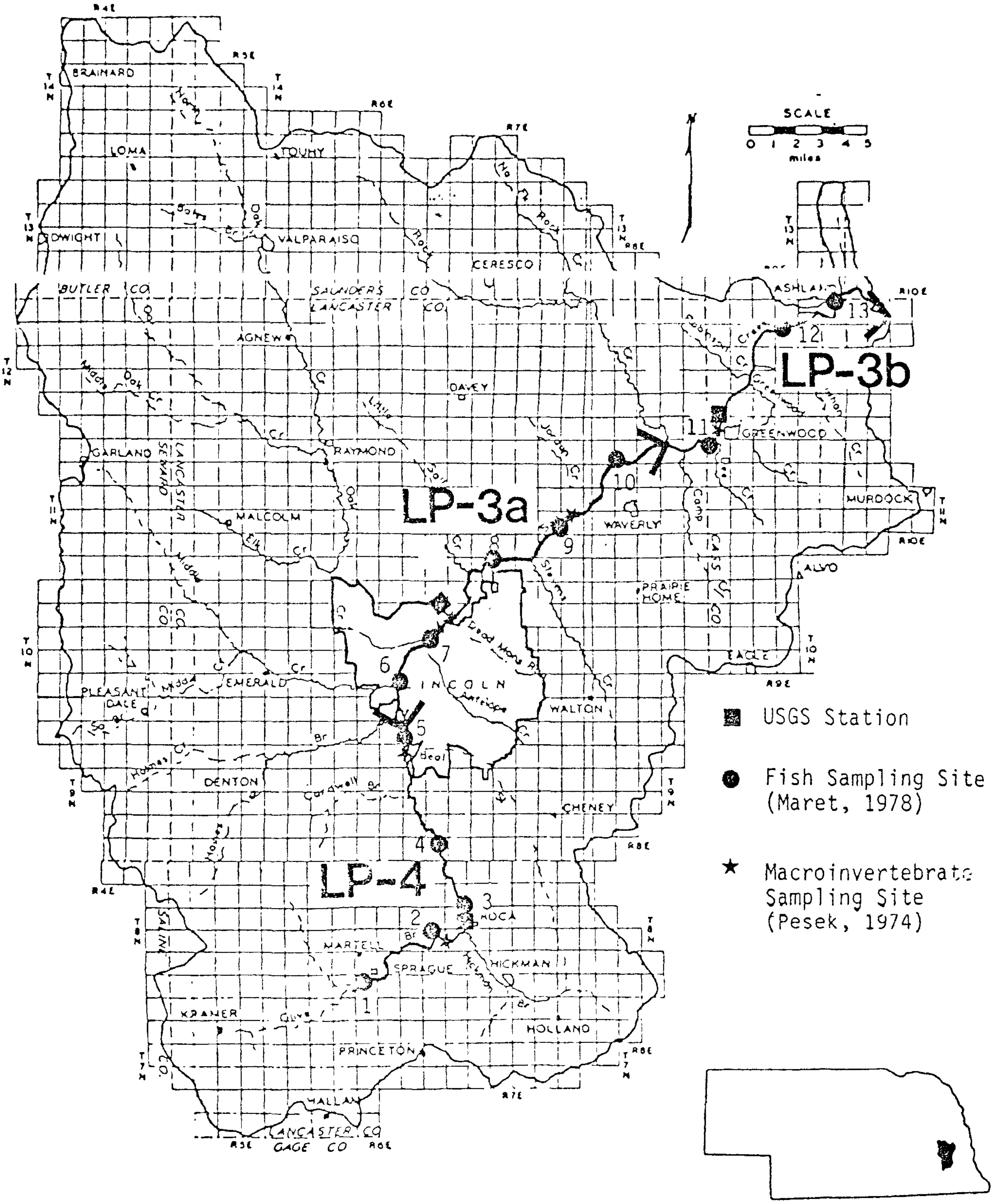


Figure 1. Monitoring sites from which data were used for Salt Creek attainability study.

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.

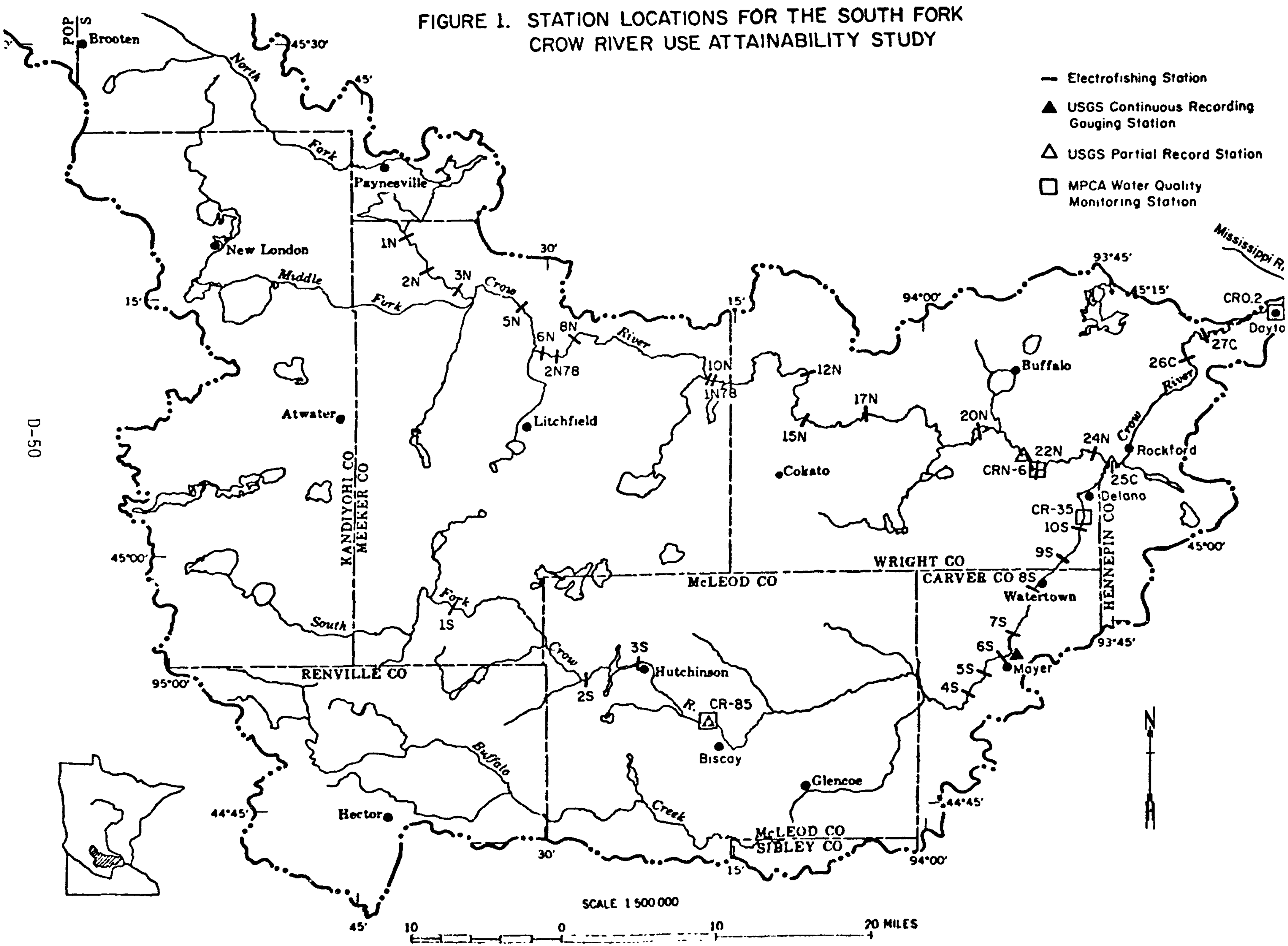
B. Problem Definition

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.

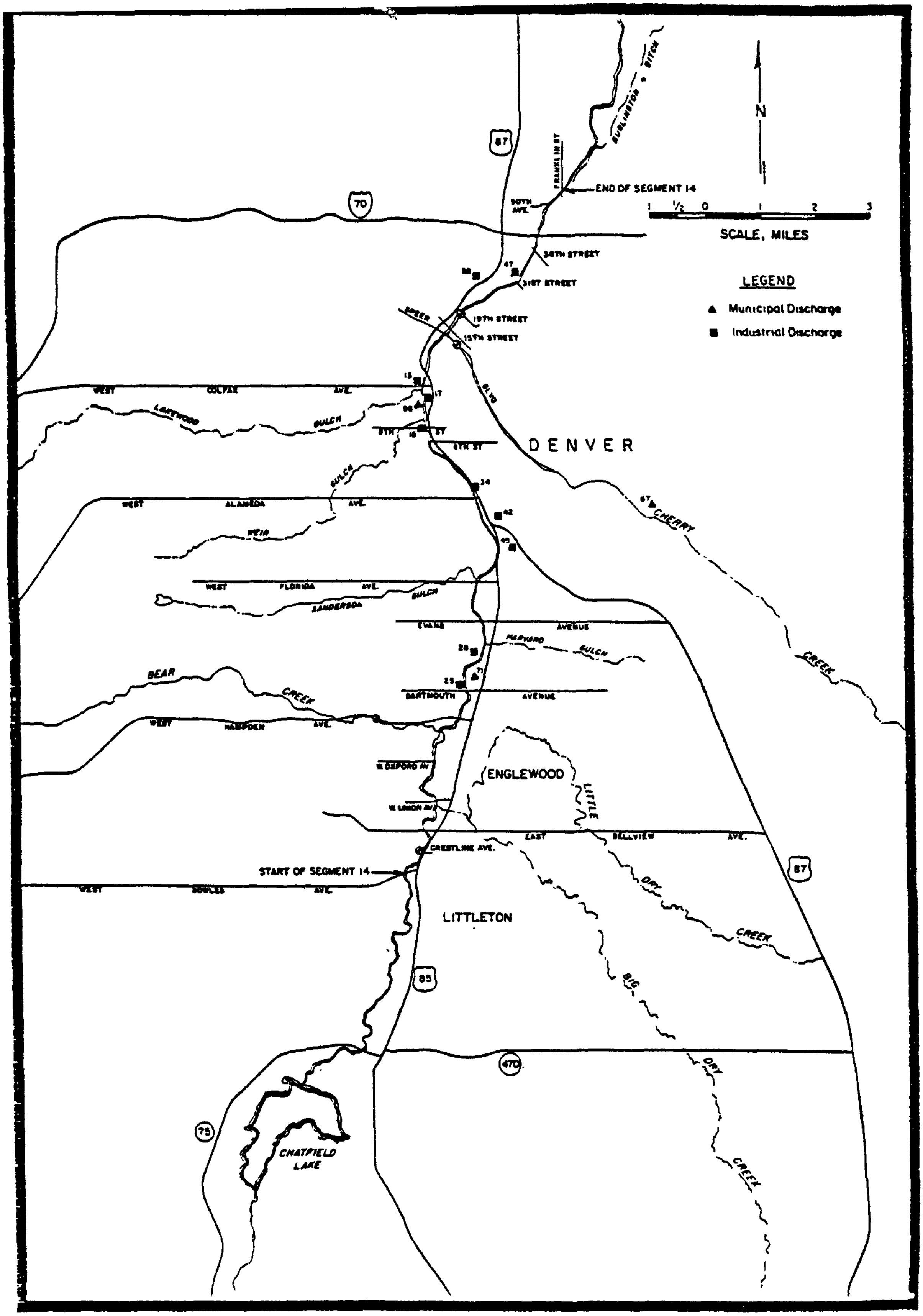


Figure 1

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.

CHAPTER 4

GUIDELINES FOR DERIVING SITE-SPECIFIC WATER QUALITY CRITERIA

Contents

Purpose and Application

The purpose of these guidelines is to provide guidance for the development of water quality criteria which reflect local environmental conditions. These site-specific criteria may be utilized as a basis for establishing water quality standards to protect the uses of a specific water body.

Water quality criteria must be based on a sound scientific rationale in order to protect a designated use. EPA is not advocating that States use site-specific criteria development procedures for setting all criteria as opposed to using the national Section 304(a) criteria recommendations.[1/] Site-specific criteria are not needed in all situations. When a State considers the possibility of developing site-specific criteria, it is essential to involve the appropriate EPA office at the start of the project so that agreement can be reached concerning data currently available, additional data needs, the best source for generating the new data, the best testing procedure to be used, the schedule to be followed, and quality control and quality assurance provisions. This early planning is also essential if it appears that the data generation and testing may be conducted by a party other than the State or EPA. The State and EPA need to apply the procedures judiciously and must consider the complexity of the problem and the extent of knowledge available concerning the fate and effect of the pollutant under consideration. If site-specific criteria are developed without early involvements of EPA in the planning and design of the task, the State may expect EPA to closely scrutinize the results before granting any approval to the formally adopted standards.

The procedures described in this chapter represent the first attempts at describing acceptable methods for developing site-specific criteria. EPA will be monitoring their implementation and developing additional procedures in the future. These procedures periodically will be revised to reflect field experiences and additional research.

1/ National water quality criteria for toxic pollutants were published as guidance under Section 304(a) of the Clean Water Act, Nov. 28, 1980, (45 FR 79318). Site-specific criteria are criteria that are intended to be applicable to a given localized site.

Rationale

National water quality criteria for aquatic life may be underprotective or unnecessarily stringent if: (1) the species at the site are more or less sensitive than those included in the national criteria data set or (2) physical and/or chemical characteristics of the site alter the biological availability and/or toxicity of the chemical. Therefore, it is appropriate that the individual Site-Specific Guidelines procedures address each of these conditions separately, as well as the combination of the two. Table 1 lists the chemicals for which national criteria are presently available.

Site-specific critera development may be justified because species at a given site may be more or less sensitive than those represented in the national criteria document. For example, the national criteria data set contains data for trout, salmon, penaeid shrimp, and other aquatic species that have been shown to be especially sensitive to some materials. Because these or other sensitive species may not occur at a particular site, they may not be representative of those species that do occur there. Conversely, there may exist at a site, untested sensitive species that are ecologically important and would need to be protected.

In addition, differences in physical and chemical characteristics of water have been demonstrated to ameliorate or enhance the biological availability and/or toxicity of chemicals in freshwater and saltwater environments. Alkalinity, hardness, pH, suspended solids and salinity influence the concentration(s) of the toxic form(s) of some heavy metals, ammonia, and other chemicals. For some chemicals, hardness or pH-dependent national criteria are available for freshwater. No salinity-dependent criteria have been derived because most of the saltwater data for heavy metals has been developed in high salinity waters. However, in some estuarine sites where salinity may vary significantly, the development of salinity-dependent site-specific criteria may be appropriate.

The effect of seasonality on the physical and chemical characteristics of water and subsequent effects on biological availability and/or toxicity of a chemical may also justify seasonally dependent site-specific criteria. The major implication of seasonally dependent criteria is whether or not the "most sensitive" time of the year coincides with that time for which the flow is the basis for waste treatment facilities design or NPDES permits. That is, if the physical and chemical characteristics of the water during low flow seasons increases the biological availability and/or toxicity of the chemical of concern, the permit limitations may be more restrictive than if the converse relationship were to apply.

TABLE 1

FRESHWATER AND SALTWATER NATIONAL CRITERIA LIST

(x = criteria are available)

(0 = criteria will be available in 1984)

Chemical	Freshwater	Saltwater
Aldrin	x	x
Ammonia	x	-
Dieldrin	x	x
Chlordane	x	x
DDT	x	x
Endosulfan	x	x
Endrin	x	x
Heptachlor	x	x
Lindane	x	x
Toxaphene	x	x
Arsenic(III)	x	0
Cadmium	x	x
Chlorine	x	x
Chromium(VI)	x	x
Chromium(III)	x	-
Copper	x	x
Cyanide	x	0
Lead	x	0
Mercury	x	x
Nickel	x	x
Selenium(IV)	x	x
Silver	x	x
Zinc	x	x

Definition of Site

Since the rationale for the Site-Specific Guidelines is usually based on potential differences in species sensitivity, physical and chemical characteristics of the water, or a combination of the two, the concept of site must be consistent with this rationale.

A site may be limited to that area affected by a single point source discharge or can be quite large. If water quality effects on toxicity are not a consideration, the site will be as large as a generally consistent biogeographic zone permits. In this case, for example, large portions of the Chesapeake Bay, Lake Michigan, or the Ohio River may each be considered as one site because their respective aquatic communities may not vary substantially. Unique populations or less sensitive uses within sites may justify a designation as a distinct site (subsite). When sites are large, the necessary data generation can be more economically supportable.

If the selected species of a site are toxicologically comparable to those species in the national criteria data set for a material of interest, and physical and/or chemical water characteristics are the only factors supporting modification of the national criteria, then the site would be defined on the basis of expected changes in the material's biological availability and/or toxicity due to physical and chemical variability of the site water.

Two additional considerations in defining a site are: 1) viable communities must occur, or be historically documented, in order to select resident species for use in deriving site-specific criteria, and 2) the site must contain acceptable quality dilution water if site water will be required for testing (to be discussed later in these Guidelines).

For the purpose of the Site-Specific Guidelines, the term "selected resident species" is defined as those species that commonly occur in a site including those that occur only seasonally (migration) or intermittently (periodically returns or extends its range into the site). It is not intended to include species that were once present in that site and cannot return due to physical habitat alterations.

Selection of a resident species should be designed to account for differences between the sensitivities of the selected resident species and those in the national data set. There are several possible reasons for this potential difference. The principal reason is that the resident communities at a site may represent a more or less narrow mix of species due to a limited range of natural environmental conditions (e.g., temperature, salinity, habitat, or other factors affecting the spatial distribution of aquatic species). The number of resident species will generally decrease as the size of the site decreases.

A second potential reason for a real difference in sensitivity could be the absence of most of the species or groups of species (e.g., families) that are traditionally considered to be sensitive to certain, but not all, chemicals (e.g., trout, salmon, saltwater penaeid shrimp, and Daphnia magna). Predictive relative species sensitivity does not apply to all materials, and the assumption that sensitive species are unique rather than representative of equally sensitive untested species is tenuous. A final reason could be that the resident species may have evolved a genetically based greater resistance to high concentrations of a material, but no data have been presented to demonstrate such a genetic difference. A few instances of increased resistance have been suggested but may be due to an acclimation of individual organisms to a stream. However, such an acclimation, should it occur, would be transitory.

Assumptions

There are numerous assumptions associated with the Site-Specific Guidelines which also apply to and have been discussed in the National Guidelines. A few need to be emphasized. The principal assumption is that the species sensitivity ranking and toxicological effect end points (e.g., death, growth, or reproduction) derived from appropriate laboratory tests will be similar to those in site situations. Another assumption is that the protection of all of the site species all of the time is not necessary because aquatic life can tolerate some stress and occasional adverse effects.

Another assumption of the Site-Specific Guidelines which follows directly from the National Guidelines is that criteria should be developed to protect the use of aquatic organisms, as well as the organisms themselves. For example, some of the national criteria were developed specifically to protect aquatic organisms from accumulating tissue residue levels of toxics which would harm wildlife predators or exceed FDA action levels. The Site-Specific Guidelines have provided procedures which enable such criteria to be adjusted to reflect local considerations, such as the fat content of resident species.

It is assumed that the Site-Specific Guidelines are an attempt to more correctly protect the resident aquatic life by accounting for toxicological differences in species sensitivity and/or water quality at the sites. Modification of the national biological data base and use of bioassay data obtained on resident species in either laboratory or site water must always be scientifically justifiable and consistent with the assumptions, rationale, and spirit of the National Guidelines.

Site-specific and national criteria are not intended or assumed to be enforceable numbers. The criteria may be used by the States to develop enforceable water quality standards and/or water quality based effluent limits. The development of standards or limits should also take into account additional factors such as the use of the site, as well as social, legal, economic, and institutional considerations. Many factors may impact the site, the environmental and analytical chemistry of the chemical, the extrapolation from laboratory data to field situations, and the relationship between the species for which data are available and the species in the body of water which is to be protected.

Procedures

° Summary

There are three procedures described in these Site-Specific Guidelines for developing site-specific criteria. The procedures for the derivation of a site-specific criterion are:

A. The recalculation procedure to account for differences in resident species sensitivity to a chemical.

B. The indicator species procedure to account for differences in biological availability and/or toxicity of a chemical caused by physical and/or chemical characteristics of a site water.

C. The resident species procedure to account for differences in resident species sensitivity and differences in the biological availability and/or toxicity of a chemical due to physical and/or chemical characteristics of a site water.

The following is the sequence of decisions to be made before any of the above procedures is initiated:

1) Define the site boundaries.

2) Determine from the national criterion document and other sources if physical and/or chemical characteristics are known to affect the biological availability and/or toxicity of the material of interest.

3) If data in the national criterion document and/or from other sources indicate that the range of sensitivity of the selected resident species to the material of interest is different from that range for the species in the national criterion document and variation in physical and/or chemical characteristics of the site water is not expected to be a factor, use the recalculation procedure (A).

4) If data in the national criterion document and/or from other sources indicate that physical and/or chemical characteristics of the site water may affect the biological availability and/or toxicity of the material of interest, and the resident species range of sensitivity is similar to that for the species in the national criterion document, use the indicator species procedure (B).

5) If data in the national criterion document and/or from other sources indicate that physical and/or chemical characteristics of the site water may affect the biological availability and/or toxicity of the material of interest, and the resident species range of sensitivity is different from that for the species in the national criterion document, use the resident species procedure.

Recalculation Procedure

° Definition

The recalculation procedure allows modifications in the national acute toxicity data set on the basis of eliminating data for species that are not resident at that site. When the elimination of data for this recalculation procedure for the site-specific Final Acute Value results in not meeting the national minimum data set requirements, additional resident species acute testing in laboratory water is required before this procedure can be used.

° Rationale

This procedure is designed to compensate for any real difference between the sensitivity range of species represented in the national data set and species found at a site.

° Conditions

- If acute toxicity data for resident species are insufficient to meet the minimum data set requirements of the National Guidelines, additional acute toxicity data in laboratory water for untested resident species would be needed before a calculation of the site-specific criterion could be made.

- Certain families or organisms have been specified to be represented in the National Guidelines acute toxicity minimum data set (e.g., Salmonidae in freshwater and Penaeidae or Mysidae in saltwater). If this or any other requirement cannot be met because the family or other group (e.g., insect or benthic crustacean in freshwater) is not represented by resident species, select a substitute(s) from a sensitive family represented by one or more resident species and meet the 8 family minimum data set requirement. If all the families at the site have been tested and the minimum data set requirements have not been met use the most sensitive resident family mean acute value as the site-specific Final Acute Value.

- Due to the emphasis this procedure places on resident species testing when the minimum data set has not been met, there may be difficulty in selecting resident species compatible to laboratory testing. Some culture and/or handling techniques may need to be developed.

- No chronic testing is required by this procedure since the national acute-chronic ratio will be used with the site-specific Final Acute Value to obtain the site-specific Final Chronic Value.

- For the lipid soluble chemicals whose national Final Residue Values are based on Food and Drug Administration (FDA) action levels, adjustments in those values based on the percent lipid content of resident aquatic species is appropriate for the derivation of site-specific Final Residue Values.

- For lipid soluble chemicals, the national Final Residue Value is based on an average 11 percent lipid content for edible portions for the freshwater chinook salmon and lake trout and an average of 10 percent lipids for the edible portion for saltwater Atlantic herring. Resident species of concern may have higher (e.g., Lake Superior siscowet, a race of lake trout) or lower (e.g., many sport fish) percent lipid content than used for the national Final Residue Value. An adjustment for these differences may be necessary.

- For some lipid soluble chemicals such as polychlorinated biphenyls (PCB) and DDT, the national Final Residue Value is based on wildlife consumers of fish and aquatic invertebrate species rather than an FDA action level because the former provides a more stringent residue level (see National Guidelines for details). Since the data base on the effects of ingested aquatic organisms on wildlife species is extremely limited, it would be inappropriate to base a site-specific Final Residue Value on resident wildlife species. Consequently, site-specific modifications for those chemicals is based on percent lipid content of resident species consumed by humans.

- For the lipid soluble chemicals whose national Final Residue Values are based on wildlife effects, the limiting wildlife species (mink for PCB and brown pelican for DDT) are considered acceptable surrogates for resident avian and mammalian species (e.g., herons, gulls, terns, otter, etc.). Conservatism is appropriate for those two chemicals, and no less restrictive modification of the national Final Residue Value is appropriate. The site-specific Final Residue Value would be the same as the national value.

° Details of Procedure

- If the minimum data set requirements are met as defined in the National Guidelines or through substitution of one or more sensitive resident family(ies) for non-resident family(ies) or group(s) required in the National Guidelines, calculate a site-specific Final Acute Value using all available resident species data in the national document and/or from other sources. If all the families at the site have been tested and the minimum data set requirements have not been met, use the most sensitive resident family mean acute value as the site-specific Final Acute Value.

- If the minimum data set requirements are not met, satisfy those requirements with additional testing of resident species in laboratory water.

- If all species in a family at the site have been tested, then their Species Mean Acute Values should be used to calculate the site-specific Family Mean Acute Value and data for non-resident species in that family should be deleted from the calculation. If all resident species in that family have not been tested, the site-specific Family Mean Acute Value would be the same as the national Family Mean Acute Value.

- To derive the site-specific maximum concentration divide the site-specific Final Acute Value by 2, as prescribed in the National Guidelines.

- Divide the site-specific Final Acute Value by the national Final Acute-Chronic Ratio to obtain the site-specific Final Chronic Value.

- When a site-specific Final Residue Value can be derived for lipid soluble chemicals controlled by FDA action levels, the following recalculation equation would be used:

  site-specific Final Residue Value =

$$\frac{\text{FDA action level}}{(\text{mean normalized BCF from criterion document})(\text{appropriate \% lipids})}$$

  where the appropriate percent lipid content is based on consumed resident species. A recommended method to determine the lipid content of tissues is given in Appendix A.

- For PCB and DDT whose national Final Residue Values are based on wildlife consumers of aquatic organisms, no site-specific modification procedure is appropriate.

- In the case of mercury (a non-lipid soluble material), a site-specific Final Residue Value can be derived by conducting acceptable bioconcentration tests with edible aquatic resident species using accepted test methods given in Appendix A. For a saltwater residue value, a bivalve species is required, (the oyster is preferred) and for a freshwater value, a fish species is required. These taxa yield the highest known bioconcentration factors for metals. The following recalculation equation would be used:

site-specific Final Residue Value =

$$\frac{\text{FDA action level}}{\text{site-specific BCF}}$$

- The lower of either the site-specific Final Chronic Value and the site-specific Final Residue Value becomes the site-specific maximum 30-day average concentration unless plant or other data indicate that a lower value is appropriate.

° Limitations

- Whatever the results of this recalculation procedure may be, a decision should be made as to whether the numerical differences, if any, are sufficient to warrant changes in the national criterion.

- The number of families used to calculate any Final Acute Value significantly affects that value. Even though the four lowest Family Mean Acute Values (most sensitive families) are most important in that calculation, the smaller N (total number of families) is, the lower the Final Acute Value. Consequently, if none of the four most sensitive families are changed or deleted, any reduction in N will result in a lower Final Acute Value. Changes in or deletions of any of the four lowest values, regardless of whether N is changed, may result in a higher or lower Final Acute Value.

- Site-specific or national Final Residue Values based on FDA action levels may not precisely protect aquatic life, since the FDA action levels are adverse (i.e., loss of marketability).

- Bioaccumulation, except in field studies, does not add to the laboratory-derived bioconcentration factors because the laboratory procedures preclude food chain uptake. Consequently, some residue levels obtained by laboratory studies of bioconcentration (direct uptake of the chemical from water) may underestimate potential effects encountered at a site. The magnitude of site-specific bioconcentration factors obtained in the laboratory, therefore, may be insufficient to protect the public from the effects of the ingested chemical of concern.

Indicator Species Procedure

° Definition

This procedure is based on the assumption that physical and/or chemical characteristics of water at a site may influence biological availability and/or toxicity of a chemical. Acute toxicity in site water and laboratory water is determined using species resident to the site, or acceptable non-resident species, as indicators or surrogates for species found at the site. The difference in toxicity values, expressed as a water effect ratio, is used to convert the national maximum concentration for a chemical to a site-specific maximum concentration from which a site-specific Final Acute Value is derived.

This procedure also provides three ways to obtain a site-specific Final Chronic Value. It may be (1) calculated (no testing required) if a Final Acute-Chronic Ratio for a given chemical is available in the national criteria document. This ratio is simply divided into the site-specific Final Acute Value to obtain the site-specific Final Chronic Value; (2) obtained by performing two acute and chronic toxicity tests which include both a fish and invertebrate species (resident or non-resident) in site water. Acute-chronic ratios are calculated for each species, and the geometric mean of these ratios is then divided into the site-specific Final Acute Value to obtain the site-specific Final Chronic Value; and (3) obtained by performing chronic toxicity tests with at least one fish and one invertebrate (resident or non-resident) in both laboratory water and site water and calculating a geometric mean chronic water effect ratio which is used to modify the national Final Chronic Value.

° Rationale

This procedure is designed to compensate for site water which may affect the biological availability and/or toxicity of a chemical. Major factors affecting aquatic toxicity values of many chemicals, especially the heavy metals, have been identified. For example, the carbonate system of natural waters (pH, hardness, alkalinity, and carbon dioxide relationships) has been the most studied and quantified with respect to effects on heavy metal biological availability and/or toxicity in freshwater. The literature indicates that in natural systems organic solutes, inorganic and organic colloids, salinity, and suspended particles also play an important but less quantifiable role in the biological availability and/or toxicity of heavy metals to aquatic life. This procedure provides a means of obtaining a site-specific Final Chronic Value for a chemical when the acute-chronic ratios in the national criteria document are thought to be inapplicable to site-specific situations.

° Conditions

- There is no reason to suspect that the resident species sensitivity is different from those species in the national data set.

- The toxic response seen in the tests used in the development of the national water quality criterion would be essentially the same if laboratory test water required in this procedure had been used instead.

- Differences in the toxicity values of a specific chemical determined in laboratory water and site water may be attributed to chemical (e.g., complexing ligands) and/or physical (e.g., adsorption) factors that alter the biological availability and/or toxicity of the chemical.

- Selected indicator species directly integrate differences in the biological availability qnd/or toxicity of a chemical. They provide a direct measure of the capacity of a site water to increase or decrease toxicity values relative to values obtained in laboratory water.

- National Final Acute-Chronic Ratios for certain chemicals can be used to establish site-specific Final Chronic Values.

- A site-specific acute-chronic ratio, obtained in site water testing, reflects the integrated effects of the physical and/or chemical characteristics of water on toxicity values.

- The water effect ratio concept used in this procedure for modifying national Final Acute Values to site-specific situations is also applicable to modifying national Final Chronic Values to site-specific situations.

° Details of Procedure

- Test at least two indicator species, a fish and an invertebrate, using laboratory dilution water and site dilution water according to acute toxicity test procedures recommended in Appendix A. Test organisms must be drawn from the same population and be tested at the same time and most importantly, except for the water source, be tested under identical conditions (i.e., temperature, lighting, etc.). The concentration of the chemical in the acute toxicity tests must be measured and be within the solubility limits of the chemical. Therefore, species selected for testing should be among the more sensitive to the chemical of interest.

- Compare the laboratory and site water LC50 values for each indicator species to determine if they are significantly different ($P\leq 0.05$) (see statistical procedure in Appendix B). If the LC50 values are not different, then the national maximum concentration is the site-specific maximum concentration. If the LC50 values are different, calculate the water effect ratio for each species according to the following equation:

  $$\text{Water Effect Ratio} = \frac{\text{Site Water LC50 Value}}{\text{Laboratory Water LC50 Value}}$$

  Determine if the two ratios are statistically different ($P\leq 0.05$) (see Appendix B).

  If the two ratios are not statistically different calculate the geometric mean of the water effect ratios. The site-specific maximum concentration can be calculated by using this geometric mean water effect ratio in the following equation: site- specific maximum concentration = water effect ratio x the national maximum concentration (or x the national maximum concentration adjusted to a water characteristic of the laboratory water when appropriate).

  If the two ratios are different, additional tests may have to be conducted to confirm or refute the data. In such cases professional judgment is appropriate in determining if some or none of the ratio data can be used to modify the national maximum concentration.

  The site-specific maximum concentration is multiplied by the 2 to obtain the site-specific Final Acute Value which is used to calculate the site-specific Final Chronic Value.

- If the national Final Acute-Chronic Ratio for the chemical of interest was used to establish a national Final Chronic Value, the site-specific Final Chronic Value may be calculated using the acute-chronic ratio in the following equation:

  Site-Specific Chronic Value =

  $$\frac{\text{Site-Specific Acute Value}}{\text{Final Acute/Chronic Ratio}}$$

- If the national Final Acute-Chronic Ratio was not used to establish a national Final Chronic Value, the national Final Chronic Value may be used as the site-specific Final Chronic Value, or it may be measured by performing 2 acute and 2 chronic tests, (Appendix A) using site water. Test at least one fish and one invertebrate species, and conduct using site water of similar quality. These data are used to calculate an acute-chronic ratio for each species. If these ratios are within a factor of 10, the geometric mean of the 2 acute-chronic ratios (the site-specific Final Acute-Chronic Ratio) is used to calculate the site-specific Final Chronic Value using the following equation:

$$\text{Site-Specific Final Chronic Value} = \frac{\text{Site-Specific Final Acute Value}}{\text{Site-Specific Final Acute-Chronic Ratio}}$$

After an acute-chronic ratio is determined for one species and if that ratio is within the range of the values used to establish the national acute-chronic ratio, it is recommended that the site-specific ratio be used in recalculating the national ratio. This recalculated ratio would then be used as the site-specific Final Acute-Chronic Ratio in the above equation.

- A site-specific Final Chronic Value can be obtained by testing indicator species for chronic toxicity. Test at least two indicator species, a fish and an invertebrate, using laboratory dilution water and site dilution water according to chronic toxicity test procedures recommended in Appendix A. For each species, use organisms from the same population, conduct tests at the same time and most importantly (except for the water source) under similar conditions (e.g., temperature, lighting). The concentration of the chemical in the toxicity tests must be within the solubility limits of the chemical. To avoid solubility problems, species selected for testing should be among the most sensitive to the chemical of interest (screening tests may be necessary).

  Compare the laboratory and site water chronic values for each of the indicator species to determine if they are significantly different (limits of chronic values do not overlap).

  If for a species the chronic values are not different, the water effect ratio = 1.0.

  If the chronic values are different, calculate the water effect ratio for each species according to the following equation:

$$\text{Chronic Water Effect Ratio} = \frac{\text{Chronic Value in Site Water}}{\text{Chronic Value in Laboratory Water}}$$

  Calculate the geometric mean of the water effect ratios for the species tested.

If the mean water effect ratio is not different from 1.0, the national Final Chronic Value is the site-specific Final Chronic Value.

If the water effect ratio is different from 1.0, the site-specific Final Chronic Value can be calculated by using the following equation: site-specific Final Chronic Value = Chronic Water Effect Ratio x the national Final Chronic Value (or the national Final Chronic Value adjusted to a quality characteristic of the laboratory water when appropriate).

The site-specific Final Chronic Value is used in the determination of the site-specific 30-day average concentration. The lower of the site-specific Final Chronic Value and the recalculated site-specific Final Residue Value (as described in the Recalculation Procedure) becomes the site-specific 30-day average concentration unless plant or other data (including data obtained from the site-specific tests) indicates a lower value is appropriate. If a problem is identified, judgment should be used in establishing the site-specific criterion.

° Limitations

- If filter feeding organisms are determined to be among the most sensitive to the chemical of interest from the national criteria document and/or other sources, and members of the same group are important components of the site food web, a member of that group, preferably a resident species, should be tested in order to discern differences in the biological availability and/or toxicity of the chemical of interest due to ingested particulates.

- Site water for testing purposes should be obtained under typical conditions and can be obtained at any time of the day or season. Storm or flood impacted water is unacceptable as test water in the acute tests used to calculate water effect ratios and acute-chronic ratios but is acceptable test water for short periods of time in long-term chronic tests used to calculate these ratios. There are some special cases when storm impacted water is acceptable in acute toxicity testing for use in criteria development. For example, an effluent discharge may be allowed only during high water periods, or a non-point source of a chemical pesticide may be of most concern during storm-related runoff events.

- Site water must not be influenced by effluents containing the chemical of interest or effluents that may impact the material's biological availability and/or toxicity. The site water should be used as soon as possible after collection in order to avoid significant water quality changes. If diurnal water quality cycles (e.g., carbonate systems, salinity, dissolved oxygen) are known to markedly affect a chemical's toxicity, use of on-site flow-through testing is suggested; otherwise transport of water to off-site locations is acceptable. During transport and storage, great care should be taken to maintain the original quality of the water; however, certain conditions of the water may change and the degree of these changes should be measured and reported.

- Seasonal site-specific criteria can be derived if monitoring data are available to delineate seasonal periods corresponding to significant differences in water characteristics (e.g., carbonate systems, salinity, turbidity).

- The frequency of testing (e.g. the need for seasonal testing) will be related to the variability of the physical and chemical characteristics of site water as it is expected to affect the biological availability and/or toxicity of the material of interest. As the variability increases, the frequency of testing will increase.

- With the exception that storm or flood impacted water may be used in chronic toxicity tests, the limitations on the use of indicator species to derive a site-specific Final Chronic Value are the same as those for site-specific modifications of a national Final Acute Value.

Resident Species Procedure

° Definition

Derivation of the site-specific maximum concentration and site-specific 30-day average concentration would be accomplished after the complete acute toxicity minimum data set requirements have been met by conducting tests with resident species in site water. Chronic tests may also be necessary.

° Rationale

This procedure is designed to compensate concurrently for any real differences between the sensitivity range of species represented in the national data set and for site water which may markedly affect the biological availability and/or toxicity of the material of interest.

° Conditions

Develop the complete acute toxicity minimum data set using site water and resident species.

° Details of Procedure

- Complete the acute toxicity minimum data set test requirements by testing resident species in site water and derive a site-specific Final Acute Value.

- The guidance for site water testing has been discussed in the indicator species procedure.

- Certain families of organisms have been specified in the National Guidelines acute toxicity minimum data set (e.g., Salmonidae in fresh water and Penaeidae or Mysidae in salt water); if this or any other requirement cannot be met because the family or other group (e.g., insect or benthic crustacean) in fresh water is not represented by resident species, select a substitute(s) from a senstive family represented by one or more resident species and meet the 8 family minimum data set requirement. If all the families at the site have been tested and the minimum data set requirements have not been met, use the most sensitive resident family mean acute value as the site-specific Final Acute Value.

- To derive the site-specific maximum concentration divide the site-specific Final Acute Value by two.

- The site-specific Final Chronic Value can be obtained as described in the indicator species procedure. An exception is that a chronic water effect ratio should not be used to calculate a Final Chronic Value.

- The lower of the site-specific Final Chronic Value and the recalculated site-specific Final Residue Value (as described in the Recalculation Procedure) becomes the site-specific 30-day average concentration unless plant or other data (including data obtained from the site-specific tests) indicates a lower value is appropriate. If a problem is identified, judgment should be used in establishing the site-specific criterion.

° Limitations

- The frequency of testing (e.g., the need for seasonal testing) will be related to the variability of the physical and chemical characteristics of site water as it is expected to affect the biological availability and/or toxicity of the material of interest. As the variability increases, the frequency of testing will increase.

- Many of the limitations discussed for the Recalculation and Indicator Species procedures would also apply to this procedure.

Heavy Metal Speciation

The national criteria for metals are established primarily using laboratory data in which reported effect concentrations have been analyzed primarily as total, total recoverable, or acid extractable metal concentrations. Metals exist in a variety of chemical forms in water. Toxicological data have demonstrated that some forms are much more toxic than others. Most of the toxicity appears to reside in the soluble fraction and, potentially, in the easily labile, nonsoluble fraction. The national criteria values may be unnecessarily stringent if applied to total metal measurements in waters where total metal concentrations include a preponderance of metal forms which are highly insoluble or strongly complexed. Derivation of criteria based on metal forms is not possible at this time because adequate laboratory or field data bases do not exist in which metal toxicity is partitioned among the various metal forms. Analysis of total and soluble metal concentrations when soluble metal is added to site water may indicate that the metal is rapidly converted to insoluble forms or to other forms with presumed low biological availability. Under these circumstances, derivation of a site-specific criterion based on site-water effect in either the indicator or resident species procedures will probably result in less stringent criteria values.

Use of the indicator species or resident species procedures is encouraged for derivation of site-specific criteria for those metals whose biological availability and/or toxicity is significantly affected by variation in physical and/or chemical characteristics of water. Measurement of both total recoverable and soluble metal concentrations during toxicity testing is recommended.

Plant and Other Data

In the published criteria documents, no national criterion is based on plant data or "Other Data" (e.g., flavor impairment, behavioral, etc.). For some chemicals, observed effects on plants occurred at concentrations near the criterion. The Site-Specific Guidelines procedures do not contain techniques for handling such data, but if a less stringent site-specific criterion is derived, those data may need to be considered.

Appendix A: <u>TEST METHODS</u>

The following procedures are recommended for conducting tests with aquatic organisms, including fishes, invertebrates, and plants. These procedures are the state-of-the-art based on currently available information. Because all details are not covered in the following procedures, experience in aquatic toxicology, as well as familiarity with the pertinent references listed, is needed for conducting these tests satisfactorily.

In all site-specific criteria determinations, proper Quality Assurance/Quality Control procedures should be planned and followed. EPA has published guidance in this area in Guidance for Preparation of Combined Work/Quality Assurance Project Plan for Water Monitoring (OWRS QA-1) May 27, 1983.

Requirements concerning tests to determine the toxicity and bioconcentration of a chemical in aquatic organisms are given in the National Criteria Document Guidelines.

A. ACUTE TESTS:

American Public Health Association, American Water Works Association, and Water Pollution Control Federation. 1980. Standard methods for the examination of water and wastewater. 15th ed. American Public Health Association, Washington, D.C. 1134 p.

American Society for Testing and Materials. 1980. Standard practice for conducting acute toxicity tests with fishes, macroinvertebrates, and amphibians. Standard E 729-80, American Society for Testing and Materials, Philadelphia, Penn. 25 p.

American Society for Testing Materials. 1980. Standard practice for conducting static acute toxicity tests with larvae of four species of bivalve molluscs. Standard E 724-80, American Society for Testing and Materials, Philadelphia, Penn. 17 p.

B. PLANT TESTS:

American Public Health Association, American Water Works Association, and Water Pollution Control Federation. 1980. Standard methods for the examination of water and wastewater. 15th ed. American Public Health Association, Washington, D.C. 1134 p.

Lockhart, W. L. and A. P. Blouw. 1979. Phytotoxicity tests using the duckweek *Lemna minor*. pp. 112-118, IN: Toxicity tests for freshwater organisms. E. Scherer (ed.), Can. Spec. Publ. Fish. Aquat. Sci. 44. (Canadian Government Publishing Centre, Supply and Services Canada, Hull, Quebec, Canada K1A 059.)

Joubert, G. 1980. A bioassay application for quantitative toxicity measurements, using the green algae Selenastrum capricornutum. Water Res. 14: 1759-1763.

Miller, W. E., J. C. Greene, and T. Shiroyama. 1978. The Selenastrum capricornutum Printz algal assay bottle test - Experimental design, application, and data interpretation protocol. EPA-600/9-78-018, Environmental Research Laboratory-Corvallis, Corvallis, Oreg. 125 p.

Steele, R. L., and G. B. Thursby. A toxicity test using life stages of Champia parvulas [Rhodophyta]. Presented at the Sixth Symposium on Aquatic Toxicology. Sponsored by the American Society for Testing and Materials Committee E-47 on Biological Effects and Environmetal Fate. 13-14 October 1981. American Society for Testing and Materials, Philadelphia, Penn.

U.S. Environmental Protection Agency. 1974. Marine algal assay procedure; bottle test. Eutrophication and Lake Restoration Branch, National Environmental Research Center, Corvallis, Ore. 43 p.

C. FISH LIPID ANALYSIS PROCEDURE:

Approximately 10 g tissue is homogenized with 40 g anhydrous sodium sulfate in a Waring blender. The mixture is transferred to a Soxhlet extraction thimble and extracted with a 1:1 mixture of hexane and methylene chloride for 3-4 hours. The extract volume is reduced to approximately 50 ml and washed into a tared beaker, being careful not to transfer any particles of sodium sulfate which may be present in the extract. The solvent is removed in an air stream and the sample is heated to 100° C for 15 minutes before weighing the sample.

The lipid content is calculated as follows:

$$\% \text{ lipid} = \frac{\text{total residue} - \text{tare weight}}{\text{tissue weight}} \times 100$$

U.S. Environmental Protection Agency, Environmental Research Laboratory-Duluth, Duluth, MN 55804.

D. BIOCONCENTRATION FACTOR (BCF) TEST:

American Society for Testing and Materials. Proposed standard practice for conducting bioconcentration tests with fishes and saltwater bivalve molluscs. J. L. Hamelink and J. G. Eaton (Task Group Co-chairmen). American Society for Testing and Materials, Philadelphia, Penn. (latest draft.)

Veith, G. D., D. L. DeFoe, and B. V. Bergstedt. 1979. Measuring and estimating the bioconcentration factor of chemicals in fish. J. Fish. Res. Board Can. 36: 1040-1048.

E. CHRONIC TESTS:

American Public Health Association, American Water Works Association, and Water Pollution Control Federation. 1980. Standard methods for the examination of water and wastewater. 15th ed. American Public Health Association, Washington, D.C. 1134 p.

American Society for Testing and Materials. Proposed standard practice for conducting toxicity tests with early life stages of fishes. S. C. Schimmel (Task Group Chairman). American Society for Testing and Materials, Philadelphia, Penn. (latest draft).

American Society for Testing and Materials. Proposed standard practice for conducting Daphnia magna renewal chronic toxicity tests. R. M. Comotto (Task Group Chairman). American Society for Testing and Materials, Philadelphia, Penn. (latest draft).

American Society for Testing and Materials. Proposed standard practice for conducting Daphnia magna chronic toxicity tests in a flow-through system. W. J. Adams (Task Group Co-chairman). American Society for Testing and Materials, Philadelphia, Penn. (latest draft.)

American Society for Testing and Materials. Proposed standard practice for conducting life cycle toxicity tests with saltwater mysid shrimp. Susan Gentile and Charles McKenny (Task Group Co-chairman). American Society for Testing and Materials, Philadelphia, Penn. (latest draft.)

Appendix B: DETERMINATION OF STATISTICALLY SIGNIFICANTLY DIFFERENT LC50 VALUES

The following problems are addressed and examples are given:

(1) how to determine if two LC50 values are statistically significantly different, and

(2) how to determine if the difference between two pairs of LC50 values is statistically significant.

To determine if two LC50 values are statistically significantly different (at $p \leq 0.05$):

(a) Obtain the 95% confidence limits for both LC50 values.

(b) If the confidence intervals do not overlap the two values are different.

(c) If one confidence interval encompasses the other the values are not different.

(d) If the confidence intervals partly overlap the values may be different. To ascertain if they are different, further statistical analysis must be done.

If the above procedure does not indicate whether or not the LC50 values are statistically significantly different, examine the confidence interval of either the ratio or the difference of the two values. If the confidence interval of the ratio brackets one, the two LC50 values are not statistically significantly different; if the confidence interval does not bracket one, then there is a statistical difference. The difference between two LC50 values is not statistically significant if the confidence interval of the difference includes zero; if the confidence interval does not cover zero, then the difference is statistically non-zero.

The following example demonstrates how the ratio of the LC50 values can be compared when the estimated LC50 values are obtained by the Trimmed Spearman-Karber method. (See Hamilton, et al. 1977, for a discussion of the Trimmed Spearman-Karber method, including calculation of the variance.) The example presents a difference between laboratory and site LC50 values that is statistically significant.

Table 1a gives the estimated LC50 values with 95% confidence intervals for both the lab and site measurements. The LC50 values are obtained by using the Trimmed Spearman-Karber method on the natural logarithm of the concentrations.

To determine if there is a statistically significant difference, it is essential to work with the metric in which the analysis was performed. In the example the metric is the natural logarithm of the concentration. The LC50 values in Table 1 were obtained from the results in Table 1b, which gives $\log_e$ LC50 values and variances.

The calculations for the ratio and its 95% confidence interval are given in Table 1c. Since the confidence interval does not cover one, the laboratory and site LC50 values are statistically significantly different.

To compare two pairs of LC50 values several different procedures are possible. The procedure that follows shows one way to compare the ratios of the LC50 values. Specifically, the variable that is examined is the difference of the ratio of LC50 values:

$$\frac{\log_e LC50_{site\ 1}}{\log_e LC50_{lab\ 1}} - \frac{\log_e LC50_{site\ 2}}{\log_e LC50_{lab\ 2}}.$$

(As stated before, it is necessary to work in the metric in which the analysis was performed. Since the Trimmed Spearman-Karber estimate is usually obtained from an analysis of the logarithm of the dose, the ratio above should be of the logarithms of the LC50 values.)

The following four steps may indicate whether or not the difference is significant (at $p \leq .05$) without calculating the confidence interval of the difference:

(1) Obtain the 95% confidence limits for both LC50 values.

(2) If the confidence intervals do not overlap the two values are different.

(3) If one confidence interval encompasses the other the values are not different.

(4) If the confidence intervals partly overlap the values may be different. To ascertain if they are different further statistical analysis must be done.

If the above four steps do not indicate whether or not the difference of the ratios is statistically significant, the confidence interval of the difference should be examined. If the confidence interval of the difference brackets zero, the difference is not statistically significant; if the confidence interval does not cover zero, the difference is statistically significant.

An example is given in Tables 2a-2c. Table 2a gives the estimated LC50 values with 95% confidence intervals for two sets of site and lab measurements. These results were obtained from Table 2b which gives the results in natural log units based on the Trimmed Spearman-Karber Method of estimation.

Table 2c demonstrates how to determine if the difference is statistically significant. In this example, the difference is not significant. Note that this result means that there is no evidence that there is a difference; it does not mean that two ratios are necessarily identical.

References:

Hamilton, M.A., R.C. Russo, and R.V. Thurston. 1977. "Trimmed Spearman-Karber Method for Estimating Median Lethal Concentrations in Toxicity Bioassays". Environ. Sci. Technol. 11(7): 714-719. Correction 12(4): 417 (1978).

Ku, H.H. 1966. "Notes on the Use of Propagation of Error Formulas". J. of Research of the National Bureau of Standards - C. Engineering and Instrument 70C: 331-263--341-273.

Tables 1a-c Analysis of Lab and Site LC50 Values

Table 1a LC50 Values

Source	Estimated LC50	95% Confidence Interval
Lab	75	(55,104)
Site	130	(100,169)

Table 1B Log_e LC50 Value

Source	Log_eLC50	Variance
Lab	4.32	.0256
Site	4.87	.0169

Table 1c Calculation of Ratio of Site to Laboratory LC50 Values* and 95% Confidence Intervals

(i) Ratio = $\log_e$ LC50 site/$\log_e$ LC50 lab = 4.87/4.32 = 1.13

(ii) Variance of ratio =

$$\left(\frac{\log_e LC50_{site}}{\log_e LC50_{lab}}\right)^2 \quad \frac{\text{variance } \log_e LC50_{site}}{(\log_e LC50_{site})^2} + \frac{\text{variance } \log_e LC50_{lab}}{(\log_e LC50_{lab})^2}$$

$$= \left(\frac{4.87}{4.32}\right)^2 \quad \frac{.0169}{(4.87)^2} + \frac{.0256}{(4.32)^2}$$

$$= .0026$$

(iii) Confidence limit = 2 x (variance of difference)$^{1/2}$

$$= 2 \times (.0026)^{1/2} = .10$$

(iv) Confidence interval = ratio $\pm$ confidence limit

$$= 1.13 \pm .10 = (1.03, 1.23)$$

(v) Since the confidence interval does not bracket one, the ratio of site to laboratory LC50 values is statistically significant at $\alpha \leq .05$.

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* Note that in this example the ratios are of $\log_e$ LC50 values since the Trimmed Spearman-Karber Method of estimating LC50 values was used. This method estimates the LC50 based on the logarithm of the concentration, so the logarithm of the LC50 should be used here.

Tables 2a-c Analysis of the Lab and Site LC50 Values for Two Species

Table 2a LC50 Values

	Source	Estimated LC50	95% Confidence Interval
Species 1	Lab	75	(55,104)
	Site	130	(100,169)
Species 2	Lab	60	(48, 75)
	Site	90	(67,122)

Table 2b Log_e LC50 Values

	Source	$Log_e LC50$	Variance
Species 1	Lab	4.32	.0256
	Site	4.87	.0169
Species 2	Lab	4.10	.0121
	Site	4.50	.0225

Table 2c Calculation of Difference of Ratios Between Field and Site LC50 Values* and 95% Confidence Intervals

(i) Difference =

$$\frac{\log_e LC50_{site\ 1}}{\log_e LC\ 50_{lab\ 1}} - \frac{\log_e LC50_{site\ 2}}{\log_e LC50_{lab\ 2}}$$

$$= \frac{4.87}{4.32} - \frac{4.50}{4.10} = 1.13 - 1.10 = .03$$

* Note that in this example the ratios are of $\log_e$ LC50 values since the Trimmed Spearman-Karber Method of estimating LC50 values was used. This method estimates the LC50 based on the logarithm of the concentration, so the logarithm of the LC50 should be used here.

(ii) Variance of difference =

$$\text{variance}\left(\frac{\log_e LC50_{site\ 1}}{\log_e LC50_{lab\ 1}}\right) + \text{variance}\left(\frac{\log_e LC50_{site\ 2}}{\log_e LC50_{lab\ 2}}\right)$$

(where variance $\frac{\log_e LC50_{site}}{\log_e LC50_{lab}}$ is found as in Table 1c (ii)).

= .0026 + .0022 = .0049

(iii) Confidence limit = 2 x (variance of difference)$^{1/2}$

$$= 2 \times (.0049)^{1/2} \approx .14$$

(iv) Confidence interval = difference $\pm$ confidence limit

$$= .03 \pm .14\ (-.11,\ .17)$$

(v) Since the confidence interval does bracket zero, there is not enough evidence to reject the hypothesis that the ratios are different.

Appendix C: CASE STUDIES

Background

The Site-Specific Criteria Guidance describes protocols for developing site-specific water quality criteria, an activity which EPA expects will be done by the States in only a limited number of instances based on need and resource constraints. These protocols are designed to take into account the sensitivity of local aquatic life as well as local environmental effects on pollutant toxicity. EPA wanted to evaluate the utility of these procedures, to develop field experience with the new techniques, and to introduce States to the concept of setting appropriate site-specific water quality criteria.1/

The proposed protocols were field-tested at numerous sites located throughout the United States. EPA initially solicited the nomination of candidate sites from all ten EPA Regions to apply the site-specific criteria development protocols. In turn, the EPA Regional offices, in cooperation with their respective States, jointly selected candidate sites. The sites selected for field testing the protocols appear in Table 1. Participation in the demonstration project was entirely voluntary on the part of the States and was not designed to require any changes in State water quality standards or individual permits.

Findings

The protocol field-tested at most of the sites was the Indicator Species Procedure. This procedure entails a three phase testing program which includes water quality sampling and analysis, a biological survey, and conducting paired acute toxicity tests in both site and laboratory dilution water.

EPA developed an ambitious schedule to conduct these field tests because of the many candidate sites that were selected and a desire to have some results available for discussion at the series of public meetings held to discuss the proposed revisions to the water quality standards regulation. The desire to generate as much data as possible at each site to field-test the protocol was compromised with both time and cost restrictions, factors which normally will be considered and planned for by any State wishing to develop site-specific criteria for use in their standards.

1/ The basic concept of site-specific criteria development is only one of several means by which a State may adopt water quality criteria as a part of State water quality standards. In most instances, States will adopt the EPA recommendations for water quality criteria issued periodically under Section 304(a) of the Clean Water Act.

A wide range of sites were chosen to obtain practical experience on the feasibility, resources, and technical merit for implementing site-specific studies.

One of the first sites where the Indicator Species Procedure was field tested was at the Norwalk River, near Georgetown, Connecticut. This pilot study was conducted by the Connecticut Department of Environmental Protection in cooperation with the USEPA. For details of the methodology used and a site description, see Page C-8. This was an attempt by the State of Connecticut to derive site-specific water quality criteria. It allowed the State an opportunity to evaluate the potential for using the site-specific protocols in establishing site-specific criteria for its waters.

The State of Connecticut felt that this site-specific criteria development study was successful in that it resulted in systematically derived, site-specific criteria for this particular segment of the Norwalk River. This project was to evaluate and adjust water quality criteria for which a substantial data base already existed. The data gathered in this exercise could also be incorporated into the State's ambient monitoring network data base used in establishing final effluent limitations, (Dunbar and Pizzuto, 1982). Participation in this project also allowed the State of Connecticut an opportunity to better evaluate the relative merits of a comparative toxicity testing approach.

The California State Water Resources Control Board also participated in a site-specific case study. It attempted to develop site-specific water quality criteria for the BEE ester of 2,4-D in the North Coast Region of California. This project was to develop a site-specific water quality criterion for a pollutant where there was no National criterion and a limited data base. For details of the methodology used and a site description, see Page C-16. This site-specific criteria development study provided the State of California with a full aquatic life toxicity data base for 2,4-D esters. This project also gave the State of California much experience in working with EPA's criteria development protocols.

Seasonal variations in water quality criteria are also of concern and a study to investigate this concern was incorporated in one of the pilot studies. During the winter of 1981 and summer of 1982, with the cooperation of the Iowa Department of Environmental Quality, EPA initiated another site-specific criteria development project on the Iowa River in Marshalltown, Iowa. This study was an attempt to evaluate ammonia toxicity during summer and winter conditions. (The Iowa DEQ maintains a summer and winter ammonia standard). For details of the methodology used and a site description, see Page C-21. The site-specific study will eventually lead to the incorporation of the site-specific protocol into Iowa's water quality standards program.

The State of North Carolina also tested the site-specific criteria development concept. For details of the methodology used and a site description, see Page C-31. Their Department of Natural Resources and Community Development committed substantial resources to the collection and interpretation of data for the case study in Mulberry Creek. North Carolina not only explored the site-specific protocol but did a comparative analysis of the results with other existing information and guidance on the applicability of the water quality criterion. Based upon this experience, they have determined that the promulgation of site-specific criteria should become an integral part of the State's water quality standards program.

Important limitations of the protocols which should be considered when conducting these procedures were identified in the case studies.

1. In many cases, only two species were tested in both site and laboratory waters. The number of different species necessary for testing to establish a true water effects ratio may exceed this minimal requirement in many situations. Based upon this and other scientific analysis, the minimum data base has been changed in the final protocol.

2. A major assumption in the protocol was that acute toxicity effects observed could be extrapolated to predict concentrations of pollutants associated with chronic toxicity. After field-testing these protocols, EPA has determined that there may be a need for some chronic toxicity testing at each site to verify these extrapolated chronic toxicity concentrations. A new short term chronic toxicity test to be conducted in a reasonable time period and with reasonable resources is under development by EPA.

3. The proposed protocol was not specific enough in providing guidance on measuring certain factors which can have an impact on the bioavailability and toxicity of the pollutants. While water hardness was measured in most cases, other parameters such as pH, D.O., salinity and temperature were not uniformly measured.

4. The protocol did not account for seasonal differences and due to time and resource constraints, the protocol was field tested only one time at each site except in Marshalltown, Iowa. Seasonal changes may influence the persistence, fate, and bioavailability of toxicants as well as the presence or absence of sensitive life stages. The final protocol will encourage seasonal testing.

5. While the protocol did provide general guidance for the location of sampling stations, in many cases sampling sites were not comparable in slope, habitat characteristics, and other parameters. A program to assist EPA Regional Staffs and State officials is being developed to improve guidance in this area.

Recommendations

The case studies published in this document constitute a variety of examples available from the series of field tests conducted using the site-specific guidelines. Several of the case studies are not included here. After scientific review they were found to have several technical shortcomings including the need for additional field work, significant deviations from the site-specific guidelines, or the results were too ambiguous to allow proper interpretations. However, complete reports, for all site-specific case studies, are available from EPA upon request from the name and address listed in the introduction of this Handbook, with the exception of the California study. More details on the California project are available from the California State Water Resources Control Board, Toxics Special Project, P.O. Box 100, Sacramento, CA 95801. Each case study report contains a section with specific recommendations as to how the individual study could have been improved. These recommendations will also benefit the design of future site-specific studies.

Future Activities

EPA continues to investigate alternative protocols for establishing site-specific criteria.

One such procedure is a metal detoxification mechanism which EPA, assisted by experts in the field, hopes to develop into a protocol and eventually field test. EPA plans to investigate this procedure at sites which have previously been studied as well as new candidate sites. This will provide additional information which should allow EPA to evaluate the suitability of this technique for site-specific criteria development.

Another protocol EPA plans to investigate is the "chemical model" for use in establishing site-specific water quality criteria. This procedure would help to derive site-specific criteria for metals from estimates and/or measurements of the chemical speciation of metals in site water. The effect of speciation on metal toxicity would be quantified without need of actual on-site bioassay data.

EPA will monitor the use of all protocols and revise them from time to time to reflect State/EPA experiences in their application.

Conclusion

Problems were encountered with these field studies due to the large number of studies conducted under time and resource constraints. The overall project was generally considered successful in meeting the primary objectives which were: (1) a field-test of the proposed site-specific protocols and, (2) a learning experience for EPA and the States. The final protocol incorporates the new scientific information which resulted from the field studies and therefore, makes the protocol more practical and useful. One State has already formally incorporated the Indicator Species Procedure into its water quality standards program. Others have indicated that they intend to use the procedure on a case-by-case basis in setting permit limits. The consensus is that with additional data development and more explicit guidance, the Indicator Species Procedure provides a realistic mechanism for developing site-specific water quality criteria.

References

Dunbar, L.E. and E. Pizzuto Jr. 1982. Derivation of Site-Specific Water Quality Criteria - Norwalk River at Georgetown, Ct. State of Connecticut Department of Environmental Protection

Table 1: Site-Specific Criteria Development Case Studies

Region	State	Site	Source	Pollutants
I	Connecticut	Norwalk River	wire manufacturer	Lead, Zinc
II	New Jersey	Walkill River	metal finisher	Nickel, Chromium
III	Maryland	Piney Run	POTW	Ammonia, Chlorine
IV	North Carolina	Mulberry Creek	mirror finishing	Copper
	Georgia	Suwannee Creek	tannery	Ammonia
V	Michigan	Flint River	POTW	Cadmium, Copper
	Minnesota	Crow River	POTW	Cyanide,Copper
VI	Texas	Leon River	POTW	Cadmium, Chromium
	Louisiana	Selzer Creek	battery processing plant	Lead
	Oklahoma	Mingo Creek	airplane parts manufacturer	Zinc, Chromium
		Skeleton Creek	Oil refinery, POTW, Fertilizer manufacturer	Zinc, Chromium
VII	Iowa	Marshaltown	POTW	Ammonia
		Mill Creek	Machine tools manufacturer	Cyanide
	Nebraska	Salt Creek	suspected POTW	Lindane
VIII	Montana	Prickly Pear Creek	mine drainage	Copper, Zinc
IX	California	North Coast Region	silvaculture	2,4D esters
X	Washington	Spokane River	mining, smelters	Zinc

SITE SPECIFIC CRITERIA MODIFICATION

Norwalk River

Georgetown, Connecticut

I. INTRODUCTION

A. Site Description

The Norwalk River Basin encompasses 64.2 square miles of southwestern Connecticut and includes a small area of Westchester County, New York. The Upper Norwalk watershed, where this study was conducted, covers an area of 18.5 square miles and includes the region extending from the headwaters of the Norwalk River to its confluence with Comstock Brook.

There are two point source discharges of sewage upstream of the study area. Prior to discharge, this waste undergoes secondary treatment. The POTW of the town of Ridgefield discharges roughly 400,000 GPD of treated sewage near the headwaters of the Norwalk, 13.5 stream miles upstream of the study site. A second POTW discharges (35,000 GPD) to the Norwalk River 9 miles upstream of the study area. An area of failed septic systems near the smaller sewage discharge also contributes to the pollutant loading of the river.

Although water quality is degraded somewhat in the immediate vicinity of these pollutant sources, as the river flows southward towards Long Island Sound it recovers to support a valuable recreational trout fishery. There are no industrial point source discharges of metals upstream of the study area.

Within the study area itself, the Gilbert and Bennet Manufacturing Company discharges treated process water to the Norwalk River at a point below Factory Pond in Georgetown, Connecticut (Figure 1). Gilbert and Benentt is a wire drawing operation (cleaning, drawing, and coating of metal wire). Wastewater is primarily generated during the wire cleaning process. The NPDES permit for the company specifies an allowable daily discharge of up to 1.96 kg of lead, 2.78 kg of zinc, and 3.68 kg of iron. The wastewater treatment system of the Gilbert and Bennett Company consists of pH neutralization and equalization followed by precipitation and clarification of the effluent before discharge to the river. The treated wastewater is discharged intermittently to the river.

B. Problem Definition

The Connecticut Department of Environmental Protection (DEP) nominated the Norwalk River site because of high metal loading to the river (attributed to the Gilbert and Bennett Manufacturing Co.) and occasional violation of national water quality criteria. The Gilbert and Bennett NPDES permit was also due for renewal. A "desk-top" evaluation by DEP indicated that the aquatic community would show evidence of impact downstream of the point of release. In this evaluation, acute and chronic national criteria for lead and zinc were compared with calculated instream concentrations of the same metals. Calculations were made at seven-day, ten-year low flow (1.34 cfs) and at average annual flow (22.5 cfs). In order to evaluate the effect of site water

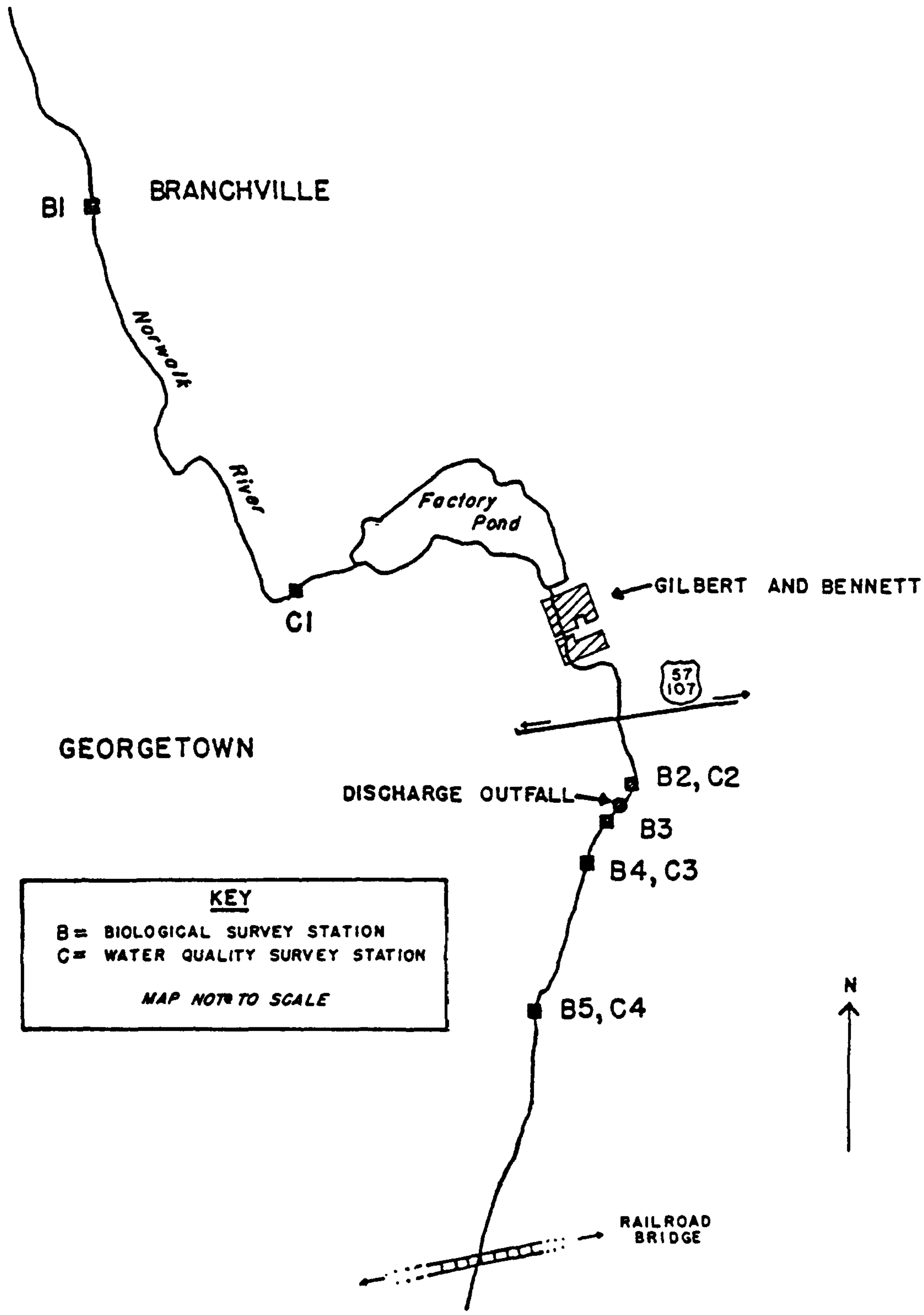

FIGURE 1 STUDY AREA: NORWALK RIVER

(Dunbar and Pizzuto, 1982)

on the toxicity of lead and zinc, EPA and State water quality officials decided to use a site-specific criteria modification protocol.

C. Approach to Criteria Modification

The decision to use a site-specific criteria modification procedure is usually made (1) after analyzing data obtained from a water body survey and assessment conducted in conjunction with a use attainability analysis (USEPA, 1982) or (2) after examining data available to state or local water quality management officials. In this study on the Norwalk River, macroinvertebrate surveys and water chemistry analyses were performed in conjunction with bioassay experiments.

The indicator species approach was chosen for this study. This procedure accounts for differences in bioavailability of a compound and therefore the effective toxicity of a chemical as a function of site water quality parameters (e.g., pH, hardness, alkalinity, presence of other contaminants, etc.). This approach requires testing of a sensitive invertebrate and fish in both site and reconstituted laboratory dilution water.

Acute toxicity tests were conducted with laboratory reared *Daphnia magna* and rainbow trout (as surrogates for sensitive organisms found at the site). These organisms were exposed to lead and zinc in Norwalk River water and a laboratory prepared reference water. The difference in measured toxicity with laboratory and site water, expressed as a water effect ratio, can then be used to modify the national criteria document Final Acute Value; to obtain a site-specific Final Acute Value. In addition to the tests required by the indicator species procedure, the toxicity of the Gilbert and Bennett wastewater effluent as a whole was evaluated.

II. ANALYSES CONDUCTED

Analysis of Water Chemistry

Based on a preliminary, qualitative, biological survey, the stream was divided into control, impact, and recovery zones and four chemical sampling stations (C_1, C_2, C_3, and C_4) were identified. C_1 and C_2 were in the control zone. C_1 is the upstream control station. C_2 is the downstream control station. C_3 was located in the impact zone and C_4 was in the recovery zone (Figure 1).

ISCO® automatic water samplers were placed at each station and used to sample ambient levels of toxic metals. Samples were taken every hour for a period of four days. Three consecutive samples were combined to form three-hour composites. All samples were analyzed for cadmium, chromium, copper, nickel, iron, lead, and zinc. Grab samples of effluent were taken at random intervals during periods of active discharge of wastewater from the Gilbert and Bennett facility. These samples were analyzed in the same manner as the composite samples.

Analysis of Biota

Benthic populations were sampled at five locations (B_1, B_2, B_3, B_4, B_5) to assess the impact of the discharge on the stream community. Four Surber samples were collected at each of the five locations (Figure 1). B_1 is the upstream control zone or reference station. B_2 is the downstream control station and was the primary reference point for the purpose of impact evaluation. B_3 and B_4 are in the impact zone, and B_5 is in the recovery zone. Physical substrate, stream velocity, and water depth were similar at each location. Organisms were sorted in the field, preserved in 70% ethanol and returned to the laboratory for identification and enumeration.

Toxicity Testing

Ninety-six hour acute toxicity tests (static with measured concentrations of toxicant) were conducted with laboratory reared rainbow trout (*Salmo gairdneri*) and 48-hour acute toxicity tests (static with measured concentrations) were conducted with laboratory reared *Daphnia magna*. Lead and zinc concentrations were measured in the test waters at the beginning of the test, after 48 hours, and at 96 hours (in the study with rainbow trout). Measured LC_{50} values were calculated based on concentrations at test termination.

Toxicity tests were conducted in Norwalk River water and reconstituted water using lead, zinc, and Gilbert and Bennett effluent as the toxicants. Norwalk River water was withdrawn from station C_1 and was transported along with the effluent back to the laboratory.

III. FINDINGS

Water Chemistry

Analysis of effluent samples from the Gilbert and Bennett waste treatment system indicates that lead, cadmium, and copper are present at levels which could exceed the EPA acute and chronic water quality criteria under low flow conditions. Lead concentrations averaged twice the maximum limit allowed by DEP in their technology based permits. Zinc concentrations were only 20% of the limit specified in the Gilbert and Bennett NPDES permit. Cadmium is not currently listed in the discharge permit.

Mean instream concentrations of lead, zinc, and cadmium were lower in the control zone than in the other sampling locations. Levels of cadmium and copper exceeded the acute criteria at all sampling locations, including the control zone. Note, however, that a diverse, stable biological community was observed to exist in the control zone. The highest levels of lead were detected just below the discharge. Maximum zinc and iron concentrations were monitored just above the outfall. These levels above the outfall are not natural, but were later found to be due to an undetected discharge from the Gilbert and Bennett Manufacturing Co.

Biota

Forty-four taxa were collected at the Branchville location (B_1). Most of the species collected can be classified as sensitive or facultative with respect to pollution tolerance (Weber, 1973; Gaufin, 1973; Roback, 1974). Species diversity was also high (a Shannon Diversity index of 3.4) indicating acceptable water quality and aquatic habitat.

At station B_2, there was a dramatic reduction in the number of taxa and individuals. Total number of organisms decreased from 889 at B_1 to 415 at B_2. The number of taxa decreased from 44 to 15, and the Shannon Diversity index fell to 1.0. This impact may be associated, in part, with the impoundment located a short distance upstream. Impoundment effects might include elevation in water temperature, reduction in downstream drift of organisms and detritus, or an increase in suspended algae.

Impacts of the effluent were observed at stations B_3 and B_4. Samples from station B_3 were collected within the discharge plume 15 m below the point of discharge. The total number of organisms present was less than half that collected at B_2, yet the number of taxa and overall community composition remained unchanged. The total number of organisms collected at B_4 just below the mixing zone was lower than that at station B_3. This difference was probably not significant, however. The number of taxa present was higher at B_4, but the community structure remained essentially unchanged.

At station B_5 (500 m downstream from the discharge) a dramatic increase was observed in the total number of organisms present. A greater number of organisms were found here than at any of the other four stations sampled. Community composition and total number of taxa remained unchanged from station B_4. The increased abundance of organisms at station B_5 indicated a reduction in the effects of the discharge from that observed at stations B_3 and B_4. While the benthic community at B_5 did not return to (recover to) conditions present in the control zone B_1, it was comparable to the downstream control station at B_2 which was the primary reference control station used for impact evaluation.

Toxicity Testing

Static bioassays were conducted exposing *Daphnia magna* to zinc. Based upon measured concentrations, 48-hour LC_{50} values and 95% confidence intervals (in parentheses) were determined: 0.90 (0.74-1.1) mg/l for river water and 0.40 (0.38-0.48) mg/l for laboratory reconstituted water. *Salmo gairdneri* exposed to zinc resulted in 96-hour LC_{50} values and 95% confidence intervals of 1.5 (1.2-1.5) mg/l for river water and 1.0 (0.85-1.2) mg/l for laboratory water. From these data it appears that zinc is less toxic in Norwalk River water than in laboratory water.

Static bioassays conducted with lead (based on measured concentrations) yielded results similar to that of the zinc test. Forty-eight hour LC_{50} values for *D. magna* (exposed to lead) were 1.3 (0.95-1.9) mg/l in river water and 0.32 (0.29-0.36) mg/l for laboratory water. Bioassays with *S. gairdneri* yielded LC_{50} values less than 9.6 mg/l for river water and 2.6 (1.9-3.6) mg/l for laboratory water. The effective toxicity of lead is thus reduced in river water.

Total lead measurements taken after bioassays were terminated (96 hours for trout and 48 hours for *Daphnia*) indicated a large difference between nominal and measured concentrations, particularly at high dose levels. It appears that lead solubility was reduced at the pH and hardness of test waters. In addition, the solubility of lead seemed more greatly reduced in laboratory water than in Norwalk River water. Measured concentrations at test termination averaged 7.2% and 49% of nominal concentrations in laboratory and river waters, respectively. LC_{50} values for lead that are based on these measured concentrations are a conservative estimate of toxicity.

In static bioassays in which S. gairdneri were exposed to effluent from the Gilbert and Bennett Manufacturing Co., the following LC_{50} values (based on measured concentrations) were determined: 60% (46-77) in river water and 68% (60-77) in laboratory water. These tests indicate that there is no significant difference in toxicity of the effluent in river water and site water. However, the no discernable effect concentration for trout was found to be slightly lower in laboratory water (22%) than in river water (36%). This does suggest a possible water effect, i.e., the river water may mitigate the toxicity of the effluent to a small degree.

Effluent from the Gilbert and Bennett plant was not sufficiently toxic to D. magna to allow calculation of an LC_{50} value. The no discernable effect concentration for Daphnia was slightly lower in the laboratory water (13%) than in the river water (36%) indicating that the effluent may be less toxic in Norwalk River than in laboratory water.

Calculations of the Water Effect Ratio

The indicator species approach to developing site-specific criteria is based on the calculation of a water effect ratio (below). The ratio accounts for the difference in the apparent toxicity of a toxicant between site water and laboratory or reference water. The total water effect ratio for a given toxicant is defined as the geometric mean of the water effect ratios for all species tested (USEPA, 1982).

$$\text{Water Effect Ratio} = \frac{\text{Site Water } LC_{50}}{\text{Lab Water } LC_{50}}$$

Measured LC_{50} values for a toxicant must be significantly different in the dilution waters to calculate a water effect ratio. Statistical significance is assumed when the 95% confidence intervals for the LC_{50} values do not overlap.

The State decided to calculate a conservative water effect ratio for zinc. That is, the ratio was based only on data for S. gairdneri, the species with the smaller water effect ratio, rather than on the geometric mean of the ratios for both S. gairdneri and D. magna.

$$\text{Zinc Water Effect Ratio} = \frac{1.50 \text{ mg/l}}{1.00 \text{ mg/l}} = 1.50$$

D. magna data were used to calculate a water effect ratio for lead.

$$\text{Lead Water Effect Ratio} = \frac{1.3 \text{ mg/l}}{0.32 \text{ mg/l}} = 4.06$$

IV. SUMMARY AND CONCLUSIONS

The Gilbert and Bennett Manufacturing Company, a wire drawing operation, discharges lead, zinc, and other metals to the Norwalk River. Ambient instream levels of the contaminants are occasionally in excess of national water quality criteria. The result of both a "desk-top" evaluation of metal loadings to the river and a preliminary biological survey indicated deterioration of water quality and adverse impact to the biota in the vicinity of the discharge. As a result, EPA and State water quality officials decided to conduct a study based upon EPA site-specific criteria modification procedures. The purpose of the study was to determine the effect of Norwalk River water on the apparent toxicity of lead and zinc. Both of these metals are present in the effluent of the Gilbert and Bennett wastewater plant and are specified in this company's NPDES permit. Macroinvertebrate surveys and water chemistry analyses were conducted in conjunction with laboratory bioassay experiments.

Analysis of the results of the biological survey indicated that the reach of stream above the discharge and Factory Pond is able to support a diverse, stable, aquatic community. Examination of the downstream stations revealed that a change in the aquatic community occurred downstream from Factory Pond from unknown causes, and further changes attributed to the discharge occurred downstream from the reference point. Impact was primarily measured in terms of organism abundance.

The results of the chemical survey parallel that of the biological assessment. Waters of the control zone contained the lowest metal concentrations and exhibited the best overall water quality of all the sampling stations. The national acute water quality criteria for copper and cadmium were exceeded in the control zone. However, a diverse, stable biological community was present. Cadmium concentrations also exceeded the national criterion at the remaining stations. Zinc, copper, lead, and iron concentrations were also elevated at the remaining stations. The impact of these metals at sampling stations B_2-B_5 was demonstrated in the biological survey.

Analysis of the toxicity tests indicate that Norwalk River water reduces the effective toxicity of lead and zinc. The extent to which the river water reduces toxicity may be examined by calculating a water effect ratio. A water effect ratio of 1.50 was calculated for zinc and a ratio of 4.06 was calculated for lead.

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SITE-SPECIFIC CRITERIA MODIFICATION
North Coast Region of California

I. <u>INTRODUCTION</u>

A. Site Description

The forested regions of Northern California particularly the northwest corner of the Klamath River basin comprise the site for this study.

B. Problem Definition

The timber industry in the North Coast Region of California uses 2,4-D in aerial spraying. In January 1982, the North Coast Regional Water Quality Control Board adopted Basin Plan amendments to control the discharge of 2,4-D esters that result from spraying in the North Coast Region. The State Water Resource Control Board reviewed these amendments and recommended discharge limits. A two number limit for 2,4-D esters was developed according to the EPA methodology for deriving water quality criteria. The two number limit, based on toxicological information on the propylene glycol butyl ether ester (PGBEE, the predominant form used in the North Coast Region) consisted of a 40 ppb instantaneous maximum limit and a 24-hour average not to exceed a 2 ppb (total acid concentration).

Recently the manufacturer of PGBE announced they will no longer be producing or marketing this product. Industry representatives have indicated that the new product of choice is the butoxy ethyl ester (BEE). Representatives of the timber industry subsequently petitioned the State Water Resources Control Board to have new water quality criteria developed for BEE. The State Water Resources Control Board reviewed the available toxicity data for BEE, but did not have sufficient information to calculate new water quality criteria. The purpose of this study was to derive acute and chronic toxicity data for 2,4-D BEE, using resident North Coast organisms, for the development of site-specific water quality criteria.

C. Approach to Criteria Modification

The resident species approach was chosen for this study. In this procedure a new minimum data base of acute and chronic toxicity values is derived in site water. This procedure is designed to adjust for any differences between the sensitivity range of species in the national data set and species resident to the site, as well as any differences in site water which may affect the toxicity of a chemical (USEPA 1982).

Acute toxicity tests were conducted under flow-through conditions with juvenile chinook salmon (<u>Oncorhyncus tshawytscha</u>), steelheads (<u>Salmo gairdneri</u>) and rainbow trout (<u>Salmo gairdneri</u>). These organisms were exposed to PGBEE and BEE individually in filtered American River water. Static toxicity tests were also conducted with steelheads exposed to BEE to evaluate differences which might result from flow-through and static tests (All existing toxicity tests for BEE except one were conducted under static conditions). In addition, a 90 day chronic embryo larval study was conducted with the chinook salmon. Results of this test are still being analyzed and will not be discussed as part of this summary.

In conjunction with the toxicity tests, the California State Water Resources Control Board will be conducting an intensive field sampling survey in the spring and fall of 1983. The intent of the survey is to characterize the discharge of 2,4-D and break down products during spraying and the first rainstorm following the spray period.

II. ANALYSES CONDUCTED

Bioassays were conducted with three juvenile salmonid species important to the North Coastal area of California, the chinook salmon (Oncorhyncus tshawytscha), steelheads (Salmo gairdneri) and rainbow trout (Salmo gairdneri). Chinook salmon and steelhead smolts were obtained 4 days prior to testing from stock at the California Department of Fish and Game's Nimbus Hatchery. Rainbow trout fry were obtained one week prior to testing from stock at the California Department of Fish and Game's Hot Creek Hatchery. Fish were maintained at the test lab in 1,000 liter circular tanks and fed up to 96 hours before testing. Sand filtered American River water was used in all of the toxicity tests.

Ninety-six hour flow-through tests were conducted with each of the organisms exposed to PGBEE and BEE individually. Ten to 25 fish were placed in each test chamber and 2 test chambers per concentration were used. Fish loading factors were within the recommended limits for flow-through tests (ASTM 1980). Grab samples were withdrawn from each test chamber for analysis of total 2,4-D acid at 48 hours. Water samples were analyzed for concentrations of 2,4-D esters at 0, 48, and 96 hours. Dissolved oxygen and temperature were also measured daily. Hardness and alkalinity were measured once during the tests and pH was measured twice.

Total 2,4-D acid concentrations were determined by esteration of the acid with gas chromatography and a Ni^{63} electron capture detector (Olson et. al. 1978). The detection limit was 5 ug/liter total 2,4-D acid. BEEE and PGBEE ester concentrations of 2,4-D were determined by repeating the hexane extraction and then combining the extracts. The extracts were concentrated with granular Na_2SO_4. The concentrated extract was analyzed using gas chromatography and a Ni^{63} electron capture detector. Detection limits were 10 ug/l for both esters.

Static tests were conducted with steelheads in 20-liter glass aquaria. Tests were conducted for 96 hours and chemical analyses were performed as for the flow-through tests.

LC50 values were calculated with the binomial test. In flow-through tests these were based on measured concentrations. Static LC50 values were based on initial 2,4-D ester concentrations. Ester concentrations decreased below detection limits (10 ug/l) with 24 to 48 hours after the tests were begun.

III. FINDINGS

Chinook salmon LC50 values and 95 percent confidence intervals in BEE tests were 1375(1306 - 1444) and 481(456 - 506) for total acid and ester respectively. Steelhead BEE LC50 values and 95 percent confidence intervals were 1400(914 - 1816) and 489(343 - 635) for total acid and ester respectively. Rainbow trout BEE LC50 values and 95 percent confidence intervals were 575(561 - 585) and 465(451 - 479) for total acid and ester respectively.

Chinook salmon PGBEE LC50 values and 95 percent confidence intervals were 1180(72 - 2288) and 318(18 - 618) for total acid and ester respectively. Steelhead LC50 values and 95 percent confidence intervals were 1610(1305 - 1915) and 434(352 - 516) for total acid and esters respectively. Rainbow trout LC50 values and 95 percent confidence intervals were 565(551 - 579) and 355(258 - 452) for total acid and ester respectively.

LC50 values for static tests with BEE conducted under two different loading factors with steelheads were 2200 ug/l for total acid and 1800 ug/l as BEE (loading factor 4.2 g/l). Tests with a higher loading factor (8.8 g/l) were 3850 ug/l as total acid and 3150 ug/l as BEE.

Analysis of the LC50 values indicate that PGBEE may be 23% slightly more toxic than BEE. In addition, it was determined that static toxicity tests grossly underestimate BEE toxicity. This is due to the hydrolysis of BEE to a less toxic form by the fish. Hydrolysis was influenced by the fish loading factor.

IV. SUMMARY AND CONCLUSIONS

The California State Water Resources Control Board is attempting to set site-specific water quality criteria for the BEE ester of 2,4-D which is used by the timber industry as an herbicide in their aerial spraying program in the North Coast Region of California. The criteria modification study was designed to provide a substantial toxicity data base using resident North Coast species.

Acute toxicity tests were conducted with juvenile chinook salmon, steelheads and rainbow trout. These organisms were exposed to PGBEE and BEE individually in filtered American River water. Static toxicity tests were also conducted with steelheads exposed to BEE to evaluate differences which might result from flow-through and static tests. In addition a 90 day chronic embryo larval study was conducted with the chinook salmon. Results of this test are still being analyzed and will not be discussed as part of this summary.

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More details on this project are available from John Norton, California State Water Resources Control Board, Toxics Special Project, P.O. Box 100, Sacramento, California, 95801 (916) 322-4506.

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SITE-SPECIFIC CRITERIA MODIFICATION
Iowa River
Marshalltown, Iowa

I. INTRODUCTION

A. Site Description

The Iowa River is a typical slow moving midwestern stream located in central Iowa (Figure 1). It meanders in an easterly direction through the northern part of Marshalltown, Iowa. The stream channel ranges from 30 - 40 m in width and stream velocity ranges from 0.1 - 0.75 m/sec.

The substrate in the Iowa River consists of shifting sand with small patches of gravel. Adjacent land use consists of agricultural development. Riparian vegetation offers considerable cover to much of the stream reach.

The Marshalltown POTW is an activated sludge plant which discharges its treated effluent to the Iowa River. The POTW is the only major point source discharge to the Iowa River in the vicinity of Marshalltown. The influent to the plant is a mixture of domestic, pretreated industrial, and untreated municipal wastewater. The average discharge from the POTW is 0.25 m^3/sec. (7.5 cfs) and remains fairly constant 24 hours per day, 7 days per week. Ammonia is a constituent routinely identified in the effluent and is of particular concern in this study.

B. Problem Definition

The Marshalltown POTW currently exceeds the state ammonia standard (2.0 mg/l total ammonia-summer 5.0 mg/l total ammonia-winter) and EPA national criterion for unionized ammonia under certain environmental conditions (low flow, high temperatures). It has been estimated that the number and severity of the violations will increase as the city grows. The Marshalltown POTW is thus one of a number of Iowa wastewater plants that has been identified for the installation of advanced treatment facilities for ammonia removal. Concurrently, the State of Iowa is evaluating its ammonia standard to determine if it is adequate or overly stringent for the protection of aquatic life. As a result, state and EPA water quality officials decided to apply site-specific criteria modification procedures to the Iowa River to evaluate seasonal influences and the effect of site water quality on the toxicity of ammonia as well as the applicability of the national ambient water quality criteria for ammonia on the Iowa River.

C. Approach to Criteria Modification

The decision to use a site-specific criteria modification procedure is usually made after analyzing (1) data obtained from a water body survey and assessment conducted in conjunction with a use attainability analysis (USEPA 1982), or (2) data available to state or local water quality management officials. In this study on the Iowa River, complete biological surveys and water chemistry analyses were conducted in conjunction with field bioassay experiments.

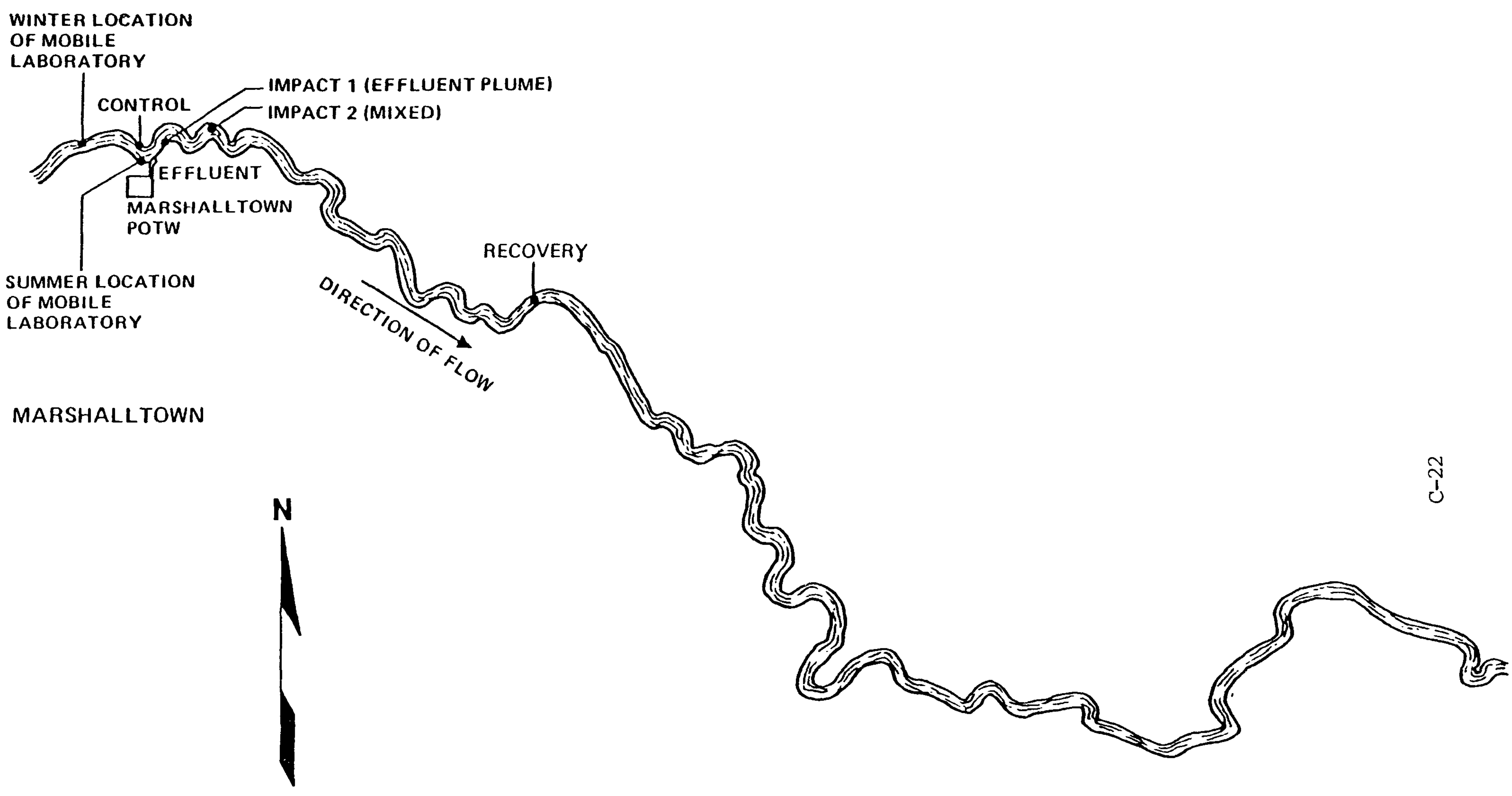

FIGURE 1. IOWA RIVER STUDY SITE. (JRB Associates 1983)

The indicator species approach was chosen for this study. This procedure accounts for differences in bioavailability of a compound in different waters. Therefore, the effective toxicity of a chemical as a function of site water quality parameters (e.g., pH, hardness, alkalinity, presence of other contaminants, etc) is examined. The approach requires testing of a sensitive invertebrate and fish in both site water and reconstituted laboratory dilution water.

Acute toxicity tests were conducted during the winter portion of this study with the channel catfish (Ictalurus punctatus). Channel catfish were exposed to ammonia in site water taken from the Iowa River (this test was conducted by the field crew and repeated by state personnel), and in a 3:1 mixture of river water to nonchlorinated effluent. The purpose of the 3:1 mixture was to simulate the instream conditions at low flow. Acute toxicity tests were conducted during the late summer with channel catfish, (Ictalurus punctatus) bluegills (Lepomis macrochirus) and a mayfly (Stenonema terminatum). These organisms were exposed to ammonia in Iowa River water, a laboratory prepared reference water, 3:1 mixture of river water to nonchlorinated effluent and a 3:1 mixture of river water to chlorinated effluent. The difference in measured toxicity with laboratory water and site water is expressed as a water effect ratio. This ratio can be used to modify the national ambient water quality criteria document Final Acute Value and to obtain a Site-Specific Final Acute Value for ammonia in the Iowa River.

II. ANALYSIS CONDUCTED

A. Analysis of Water Chemistry

Based on an inspection of the study area, the river was divided into a control, two impact zones and a recovery zone. Sampling stations were identified in each of the zones. The Control Zone Station (Station 1) was located approximately 50 meters upstream from the confluence with the POTW outfall. The first Impact Zone Station (Station 2) was located in the effluent plume approximately 50 meters downstream from the outfall. The second Impact Zone Station (Station 3) was located approximately 800 meters downstream from the confluence of the POTW discharge with the river and immediately downstream from the area of complete mixing. The Recovery Zone Station (Station 4) was located approximately 3.2 kilometers downstream from the discharge.

Due to the freezing temperatures and icy conditions only a limited chemical survey was conducted as part of the winter study. A series of grab samples were taken above and below the POTW discharge in order to characterize the POTW plume. Samples were analyzed for total ammonia, nitrates, nitrites, Kjeldahl nitrogen, and filterable and nonfilterable residues.

During the later summer phase, field samples were collected at each station and analyzed for nitrite, ammonia, Kjeldahl nitrogen, total and filterable residue, biochemical and chemical oxygen demand, cyanide, and total and dissolved organic carbon. Depth, velocity, temperature, specific conductance, dissolved oxygen and pH were also measured at each station.

Grab samples were taken to measure variations in ammonia concentrations instream and in the POTW effluent. Samples were collected weekly from August 19 - October 13, 1982 while the periphyton and macroinvertebrate samplers were allowed to colonize.

B. Analysis of Biota

Fish, periphyton, and invertebrates were sampled as part of the biological survey. No attempts to collect organisms were made during the winter. Due to the shifting sand substrate in the Iowa River, artificial substrates were used to sample the invertebrate populations. Ten modified Hester-Dendy Multiplate Samplers were placed at sampling Stations 1 - 4 and allowed to incubate for five weeks. During this period of time the POTW was not chlorinating its effluent. After five weeks one-half of the substrates were removed and these substrates represent nonchlorinated effluent samples. The remaining substrates were allowed to incubate for an additional 19 days during which time the POTW resumed chlorination. These substrates represent the chlorinated samples.

The organisms collected were preserved and returned to the laboratory for identification. All organisms were identified to the lowest possible taxon. Because of the shifting sand substrate and flow variations, several substrates became partially or totally buried in the sand, limiting the habitat available for colonization. Unfortunately many of these buried samplers were in the Control Zone. As a result, the comparison of diversity and equitability between zones was more meaningful than a comparison of total numbers.

Artificial substrates were also placed in the Iowa River to sample the periphyton community. The samplers consisted of six, glass microscope slides secured in a plastic frame. The substrates were suspended from floats at a uniform depth at each sampling station. The substrates were left in the stream for a period of 17 days during which time the POTW was not chlorinating its effluent. When chlorination resumed fresh substrates were placed in the river as in the nonchlorinated phase. Samples were preserved in Lugols solution and analyzed according to Weber (1973). All algal types present were counted, but only diatoms were identified to species. Slides were also analyzed for chlorophyll content and ash free dry weight. Shannon-Weaver diversity indices and equitability values for the nonchlorinated and chlorinated portions of the study were calculated.

Fish collections were conducted by the Iowa Conservation Commission. The fish were collected using a 230 volt boatmounted electroshocker and a thirty foot (1/4 inch mesh) minnow seine. Three individual runs of approximately 100 meters were taken with the electroshocker and one pull with the seine was taken in each sampling zone. All fish were counted and identified to the species level.

C. Toxicity Testing

Winter bioassays were conducted with the channel catfish while late summer tests were conducted with channel catfish, bluegills, and mayflies. Juvenile catfish were obtained from the Lake Rathbun Fish Hatchery Rathbun, Iowa. Bluegills each weighing 0.5 - 2 gm were obtained from the Fairport Fish

Hatchery. Mayflies were collected from the Iowa River approximately 12 km downstream from Marshalltown.

Ninety-six hour flow-through tests were conducted with the fish and the mayflies in site water from the Control Zone and in a 3:1 mixture of river water to effluent water (nonchlorinated and chlorinated effluent). Ammonia concentrations were measured every 12 hours for the duration of the test. Temperature, pH, and dissolved oxygen concentrations were measured in conjunction with each ammonia analysis.

Ninety-six hour static renewal tests were conducted with the fish and the mayflies in a laboratory reference water. Test solutions were renewed every 12 hours due to the volatility of ammonia. Ammonia, temperature, pH, and dissolved oxygen concentrations were measured at the beginning and end of the 12-hour volume replacement period. Throughout the tests, ammonia concentrations never fell below 80 percent of initial concentrations.

Field analysis of ammonia concentrations in the test chambers was conducted using an Orion Specific Ion Electrode. A new standard curve was prepared prior to each analysis. In addition, split lab and field samples were collected in triplicate at 0 hours, 48 hours and 96 hours during the tests and analyzed by the University of Iowa Hygienic Laboratory. Ammonia concentrations were measured within 24 hours after the laboratory received the samples.

III. FINDINGS

A. Water Chemistry

Results of the physical and chemical measurements indicate that the study reach was characterized by generally uniform habitat and moderate riparian canopy. Stream velocity averaged 0.75 m/sec at all stations and depth averaged 60 cm. The stream substrate was dominated by unstable sandy conditions.

Analyses of water quality (grab samples) indicate that most chemical parameters were stable and within normal expected ranges throughout the study reach. Dissolved oxygen concentrations remained at or above saturation although there was a significant increase in biological oxygen demand downstream from the POTW when the effluent was bypassed following primary clarification. The stream was generally turbid however. When bypassing occurred, nonfilterable solids increased. Except for ammonia, all toxics were below detection limits or below their respective water quality criteria values.

Winter grab samples taken in the vicinity of the discharge plume indicate that ammonia concentrations rapidly attenuate within the effluent plume. By the time complete mixing of effluent and river water had occurred, all measured nitrogen compounds had fallen to near Control Zone concentrations.

Analysis of weekly grab samples revealed that unionized ammonia concentrations were occassionally in excess of 0.2 mg/l in the effluent plume. At the point of complete mixing concentrations were generally below 0.02 mg/l.

B. Biota

Analysis of the invertebrate samples from nonchlorinated and chlorinated study phases indicated that Impact and Recovery zones could be defined, but too few samples were recovered to quantify the Control Zone. Total number of organisms did not differ significantly in either of the Impact or Recovery Zones, but diversity and equitability values were lower at Impact 1 (nonchlorinated and chlorinated).

Mayfly percent relative abundance (PRA) demonstrated a difference between nonchlorinated and chlorinated conditions. The PRA in Impact 1, Impact 2, and the Recovery Zone decreased dramatically from the nonchlorinated to the chlorinated samples. This is thought to be an avoidance reaction to residual chlorine, but cannot be confirmed since residual chlorine was not measured.

Periphyton diversity and equitability values for nonchlorinated and chlorinated samples do not decline in the Impact Zones. However in both sets of samples a shift in species dominance can be observed in the Impact Zones. In the nonchlorinated study, Gomphonema olivaceum was the dominant species in the Control, Impact Zone 2, and Recovery zones. This species is characteristic of sites that have experienced inorganic nutrient enrichment. However, it normally occurs where biodegradation is complete. In the Impact Zone 1, an area of high biodegradation, G. olivaceum numbers are sharply reduced. Nitzchia palea, a good indicator of organic pollution and Cyclotella striata, which is stimulated by slight increases in salts, are the dominant taxa at this station (USEPA 1974).

In the chlorination study the diatom Nitzchia dissipata is the dominant diatom in the Control Zone. This species is common to water with high dissolved oxygen (USEPA 1974). This species is not as common in the Impact and Recovery Zones. The dominant species at the Impact 1 Station (Nitzchia palea) is common to zones of organic degradation and low dissolved oxygen (USEPA 1974).

Analysis of chlorophyll concentrations, ash free dry weight and autotrophic indices indicate that the Iowa River is affected by organic enrichment throughout the study reach especially at the Impact 1 Station. The acidification ratios (chlorophyll a to pheophytin a) in the nonchlorinated and chlorinated studies were the lowest at the Impact 1 Station. Ash free dry weights were highest at the Impact 1 Station. The autotrophic index at all stations in both studies was greater than 100 which is indicative of an area affected by organic pollution (Weber 1973).

Fish collected in the Control Zone were diverse in number of species as well as trophic position in the community. There were a relatively high proportion of carnivores (i.e., centrarchids and ictalurids) and planktivores (i.e., clupeides). At Impact 1 the number of planktivores and carnivores is as reduced from Control populations. The reduction or absence of carnivores in the fish community is an indication of a system degraded by poor habitat or water quality (Karr 1982). The failure of these organisms to also successfully inhabit Impact 2 and the Recovery Zone suggests chronic water quality degradation or a general shift in the habitat or trophic structure of the Iowa River.

C. Toxicity Testing

LC50 values and 95 percent confidence intervals were estimated by the binomial, probit, and moving average methods. Mean ammonia concentrations, based on all field measurements taken during each test were used in the LC50 calculations. Determination of unionized ammonia concentrations were based on the average temperature and pH measured during each test.

Winter total ammonia LC50 values and 95 percent confidence intervals (binomial method mg/l) for catfish were 40.99 (38.8 - 47.6), 41.3 (36.1 - 45.1), and 43.0 (37.0 - 72.1) for Site Water Test 1, Site Water Test 2, and 3:1 river water to nonchlorinated effluent tests respectively. Winter unionized ammonia LC50 values and 95 percent confidence intervals were 0.49 (0.38 - 0.70), 0.49 (0.31 - 0.66), and 0.43 (0.23 - 0.83) for Site Water Test 1, Site Water Test 2, and 3:1 river water to nonchlorinated effluent tests respectively. The LC50 values did not vary significantly in these tests.

Late summer total ammonia LC50 values and 95 percent confidence intervals (binomial method in mg/l) for the channel catfish were 27.3 (21.4 - 35.9), 18.5 (7.4 - 27.4), 27.7 (13.9 - 32.9), 25.0 (13.7 - 32.6) for the lab water, site water, chlorinated effluent and nonchlorinated effluent tests respectively. Late summer unionized ammonia LC50 values and 95 percent confidence intervals were 0.61 (0.56 - 0.75), 0.69 (0.36 - 0.84), 1.4 (0.68 - 1.6), 1.2 (0.63 - 1.5) for the lab water, site water, chlorinated effluent, and nonchlorinated effluent tests respectively. The LC50 values did not vary significantly in these tests, although LC50 values from the effluent tests appear to be somewhat higher than the site water and lab water tests.

It was not possible to determine LC50 values for all of the mayfly tests. Total ammonia LC50 values and 95 percent confidence intervals (probit method in mg/l) were 7.2 (0 - 20.0) and 79.8 (25.9 -∞) for the site water and nonchlorinated effluent tests respectively. Unionized ammonia LC50 values and 95 percent confidence intervals for these same tests were 0.35 (0 - 0.72) and 3 (1.19 - ∞). These tests indicate that mayflies were as sensitive or less sensitive to ammonia than catfish.

Forty-eight hour bluegill LC50 values for total ammonia and 95 percent confidence intervals (probit method mg/l) were 20.6 (16.7 - 25.2) and 8.7 (4.3 - 12.3) for laboratory and site water respectively. Corresponding forty eight hour LC50 values and 95 percent confidence intervals for unionized ammonia were 0.48 (0.41 - 0.56), and 0.45 (0.27 - 0.57) for lab water and site water respectively. Although total ammonia values appear to differ significantly in these tests, unionized ammonia LC50 values (the most toxic fraction) do not vary significantly.

Ninety-six hour bluegill LC50 values for total ammonia and 95 percent confidence intervals (probit method mg/l) were 16.1 (13.0 - 19.4), 13.0 (10.1-15.6), and 16.7 (14.8 - 18.9) for laboratory water, chlorinated effluent, and nonchlorinated effluent respectively. Corresponding 96 hour LC50 values and 95 percent confidence intervals for unionized ammonia are 0.40 (0 - ∞), 0.63 (0.48 - 0.75), and 0.77 (0.68 - 0.87) for laboratory water, chlorinated effluent and nonchlorinated effluent respectively. These LC50 values do not vary significantly.

D. Calculation of the Water Effect Ratio

The indicator species approach to deriving site-specific criteria is based upon the calculation of a water effect ratio (below). This ratio accounts for the difference in the apparent toxicity of a contaminant in site water and a laboratory or reference water. The total water effect ratio for a given toxicant is defined as the geometric mean of the water effect ratios for all species tested.

$$\text{Water Effect Ratio} = \frac{\text{Site Water LC50}}{\text{Lab Water LC50}}$$

Measured LC50 values for a toxicant must be significantly different in the dilution waters to calculate a water effect ratio. Statistical significance is assumed when the 95 percent confidence intervals for the LC50 values do not overlap. When the confidence intervals do overlap, the water effect ratio is equal to one.

On the basis of these tests, the confidence intervals of the dilution waters overlap, therefore the water effect ratio is, in effect, equal to one. A water effect ratio equal to one would not result in any modification of the national criteria values.

IV. SUMMARY AND CONCLUSIONS

A Water Quality Criteria Modification demonstration project was conducted to evaluate the appropriateness of the acute criterion for ammonia in the Iowa River at Marshalltown, Iowa. On-site bioassays were conducted during winter and late summer in a mobile laboratory positioned upstream from the Marshalltown POTW which discharges to the Iowa River. A chemical survey of the Iowa River was conducted to determine instream concentrations of ammonia and other potential pollutants. In addition, a biological survey was conducted to evaluate periphyton, macroinvertebrate and fish community structure upstream and downstream from the confluence with the discharge canal.

Results of this investigation indicated that there were some trends in the number of species and individuals in the fish, invertebrate and periphyton communities downstream from the POTW outfall. However, the only obvious differences occurred in the samples collected from Impact Zone 1. At this station there was a substantial shift in relative abundance in the invertebrate community as compared to uptream and downstream from the outfall. However, whether this was the result of physical habitat or water quality limitations remains unclear.

On-site bioassays were designed to test the toxicity of ammonia to indigenous fish and invertebrate species in upstream (Control Zone) water, 1/4 non-chlorinated effluent and 3/4 Control Zone water, 1/4 chlorinated effluent and 3/4 Control Zone water and a standard reconstituted laboratory water. Tests were also conducted during winter and late summer to evaluate the influence of seasonal temperature differences on ammonia toxicity.

Results of these tests indicated no significant difference between laboratory water and site water. However, significant differences occurred between the winter and late summer tests, and between tests with Control Zone water and 1/4 effluent: 3/4 control zone water tests.

These differences were attributed to differences in test temperature and pH which occurred between the two testing regimes. Although the EPA draft water quality criteria document (USEPA 1983) incorporates a correction factor for pH differences, evidence exists here that various temperatures may also cause significant difference in test results.

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SITE SPECIFIC CRITERIA MODIFICATION

Un-Named Tributary to Mulbery Creek
North Wilkesboro, North Carolina

I. INTRODUCTION

A. Site Description

Site specific work was conducted on a small un-named tributary (UT) which flows into Mulberry Creek near North Wilkesboro. Two mirror plating plants--Carolina Mirror and Gardner Mirror--discharge effluents containing copper and possibly silver into UT about two miles above its confluence with Mulberry Creek.

UT begins, as a small spring, about nine-tenths of a mile north of Carolina Mirror. It is characterized by a series of riffles and pools, and falls about 70 feet in elevation before reaching the north edge of Carolina Mirror. The bottom is rocky, with occasional sediment deposits. The water is clear and colorless and becomes well aerated as it flows through the riffles.

The flow in UT is carried in a natural channel about 1.9 miles further to its confluence with Mulberry Creek. The lower reaches of UT are shallow, but relatively wider than near the discharge points. The channel bed in this section is covered with small stones and leaf packs which provide a more suitable habitat for benthic macroinvertebrates than the sediment layer observed in the vicinity of Carolina Mirror.

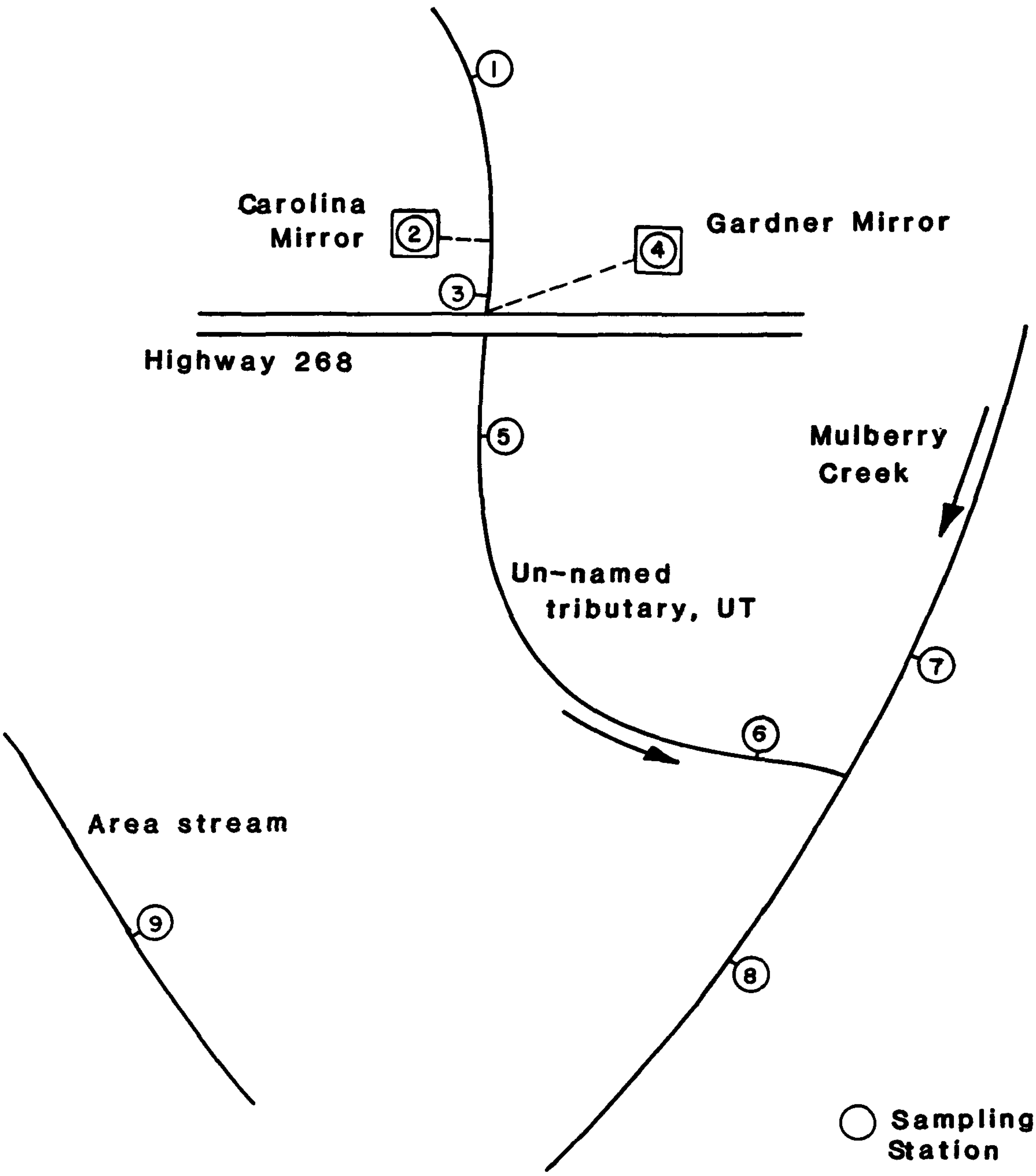
Mulberry Creek is considerably larger than UT, being some twenty-five feet wide and one to one-and-a-half feet deep. Its flow passes over a large riffle just before the confluence, so would be well aerated at this point.

B. Discharge Treatment

Carolina Mirror and Gardner Mirror treat their process wastewaters to remove both copper and silver. Silver which is recovered in the treatment process is recycled through the mirror plating line. Any silver in the effluent occurs at concentrations below the 50 ug/l detection limit of the analytical procedures used by the State. Copper is less successfully removed, and may often be found in the final effluent at concentrations greater than the 40 ug/l detection limit.

C. Approach to Criteria Modification

In response to inquiries from EPA, the State of North Carolina nominated several sites that it thought would be suitable for a test of the criteria modification protocol. The North Wilkesboro site was selected and an initial site visit conducted in September, 1981. A schematic diagram of the study area is presented in Figure 1.



SCHEMATIC DIAGRAM OF DEMONSTRATION SITE

FIGURE 1

The site visit was attended by representatives of the State of North Carolina, Department of Natural Resources and Community Development; EPA, Criteria and Standards Division; EPA, Athens laboratory; Camp Dresser & McKee; and Harbridge House. A cursory biological survey was conducted of UT and of Mulberry Creek. Subsequent discussion of the site enabled development of a Work Plan by the State.

Based on the Work Plan, the State invested considerable effort in characterizing the water chemistry of the Carolina Mirror effluent and the receiving waters (UT and Mulberry Creek) and in conducting bioassays and a biological survey of the receiving water.

II. ANALYSES CONDUCTED

A. Water Chemistry Analysis

A summary analysis of the mirror-plating effluents was provided by the State. While silver concentrations in grab samples were below detection levels, copper levels were often high in the vicinity of the discharges. Measured copper concentrations, which ranged as high as 140 ug/l are quite a bit higher than the allowable instantaneous value of 6.3 ug/l, at a hardness of 26 mg/l, which would be calculated according to the national criteria document for copper.

High levels for conductivity, suspended solids, phenol, and MBAS (methylene-blue active substances, i.e., detergents) were also detected in the vicinity of the discharges. As would be expected, measured concentrations drop appreciably after UT joins Mulberry Creek, and, in general Mulberry Creek does not appear to be affected (from the standpoint of water chemistry) by the effluents carried by UT.

B. Biological Monitoring

State biologists visited the demonstration site a number of times in order to: collect water samples for chemical analysis; sample the biota to determine species diversity, evenness and richness; identify resident fish; collect resident fish and macroinvertebrates for toxicity testing; obtain site water for the toxicity tests; and to collect fish for tissue analysis. Based on a qualitative survey, rosyside dace and creek chub were selected for toxicity testing, and a sufficient number of fish collected to perform these tests.

Duplicate kick samples of benthic macroinvertebrates were collected from six stations. The macroinvertebrates were identified, and this information analyzed by the State. According to criteria developed by biologists with the Division of Environmental Management, the 65 percent reduction in taxa richness seen below the discharge point is an indication of severe stress on the benthos. A biotic index of 4.4 below the discharges as well as the reduction in number of intolerant organisms (Ephemeroptera and Trichoptera) from 12 to 1 also indicates poor conditions. However, biologic conditions had improved by the confluence, and there is no apparent adverse effect in Mulberry Creek below the confluence.

C. Toxicity Tests

Effluent samples from both Carolina Mirror and Gardner Mirror were collected for toxicity testing. The results indicated to the State that there was little acute toxicity to Daphnia pulex during the 48-hour test period. The test was repeated, with similar results. An in-situ "bioassay" was also conducted using common shiner collected from a nearby stream. Fish cages were placed in UT and checked after a six day period. All the fish were still alive and apparently healthy in the cage placed just above the confluence. The results of these tests suggest that there is little acute toxicity associated with the mirror plant effluents and their presence in UT water. However, the limited array of macroinvertebrates found above the confluence suggests that there may be a toxic fraction in the mirror plating effluents which becomes concentrated in UT sediment.

Static, 48-hour, acute toxicity tests were performed on five species of aquatic fauna considered members of the upper piedmont biota. Three of these were vertebrate species (fathead minnow, rosyside dace and creek chub) and two were invertebrates (Daphnia pulex and Ephemera simulans). All test organisms were acclimated in site water for at least four days. The site water had been collected from UT above Carolina Mirror and transported to the laboratory for use both in acclimation and as bioassay dilution water.

Replicate tests were performed on all test species except the mayfly (Ephemera) which could not be found in adequate numbers. A probit analysis using the Statistical Analysis System (SAS) was performed to determine LC50 values.

III. FINDINGS

To some extent, whether or not the mirror plating discharges have a significant impact on UT becomes a value judgment. While the benthic survey shows a significant change in macroinvertebrate populations immediately below the discharges, there appears to be some recovery by the time flow reaches the confluence of UT and Mulberry Creek. There is no discernable adverse affect on Mulberry Creek due to UT. Based on the in-situ "bioassay" conducted near the confluence, it would appear that fish too are not adversely affected by the mirror plating effluents.

It is assumed in this demonstration that the only pollutant of consequence being released to UT is copper. The national criteria value for copper at a hardness of 26 mg/l (as measured in UT) is 6.3 ug/l, a value which is considerably less than concentrations measured in the vicinity of the discharges. High copper levels are probably not unusual. If the aquatic life of UT has not been severely affected by the frequent occasions when in-stream copper concentrations exceed the national criteria (as suggested by the benthos above the confluence, and the results of the in-situ bioassay), we may then consider modifying the copper criteria, for UT specifically, to a value which reflects site water effects on toxicity.

The national data base used to develop individual water quality criteria comprises the results of 96-hour vertebrate bioassays and 48-hour invertebrate bioassays. Unfortunately, the bioassays conducted as part of this demonstration are all 48-hour tests and thus should not be compared with the 96-hour results in the national data base. Nevertheless, it will be instructive to develop a modified copper criterion for UT, while emphasizing that this is done for the purpose of illustration only.

A. Water Effect Ratio Method

A site specific criterion for UT may be derived by adjusting the national criterion value by the ratio of site water bioassay results to laboratory water results. The geometric mean of the ratios is 2.5, as seen in Table 1.

For total recoverable copper, the criterion to protect freshwater aquatic life, as derived using the Guidelines, should not exceed the numerical value in micrograms per liter given by:

$$\exp[0.94 \times \ln(\text{Hardness}) - 1.23] \qquad (1)$$

The hardness of UT water after receiving the two mirror plating discharges is 26 mg/l. The instantaneous maximum copper concentration calculated by Equation 1 is 6.3 ug/l. Adjustment of the national criterion value (6.3 ug/l) by the ratio of site water to laboratory water results yields a site specific copper criterion of:

$$(6.3 \text{ ug/l})(2.5) = 15.8 \text{ ug/l} \qquad (2)$$

B. Resident Species Calculation

The national criterion for a given pollutant is determined analytically or graphically according to a procedure prescribed by EPA's Office of Research and Development. The procedure is described in detail in the Federal Register of November 28, 1980. The minimum data base discussed in this early presentation of the criteria calculation method has been revised such that the definition of minimum data base will be left to the discretion of the states.

The information required for a site specific calculation is displayed in Table 2 and in Figure 2. In the procedure, ln LC50 values are grouped into intervals defined by the LC50 for the most sensitive species in the national data base for copper. The natural log of the LC50 of the most sensitive species (Daphnia pulicaria, LC50 of 0.23 ug/l) is -1.47. Each interval has a width of 0.25 log units, thus the boundaries of these intervals become -1.47, -1.22, -0.97, 4.03, 4.28.

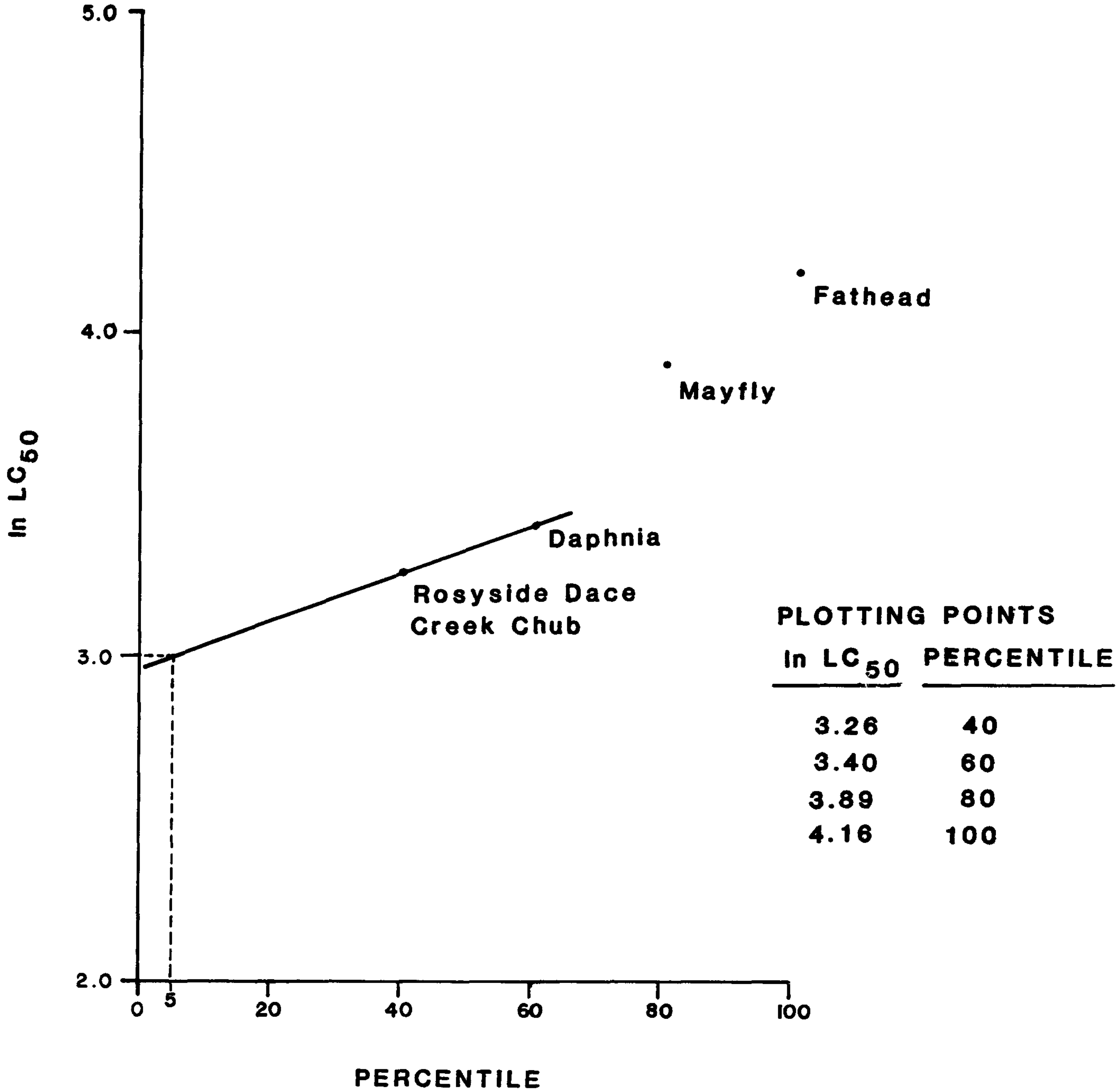
TABLE 1. ACUTE TOXICITY TESTS
48-HOUR LC_{50} FOR COPPER
(ug/l)

Test Organism	Site Water	Lab Water	Ratio
Fathead*	64	21	3.0
Mayfly	49	-	-
Daphnia*	30	14	2.1
Creek Chub*	26	-	-
Rosy Side Dace	26	-	-
Geometric Mean	36.4	17.1	2.5
Arithmetic Mean	39.0	17.5	2.6

* National Copper Data Base

TABLE 2. INFORMATION REQUIRED FOR RESIDENT SPECIES CALCULATION

Organism	Site Water LC_{50}	Site Water ln LC_{50}	Percentile
Rosyside Dace	26	3.26	20
Creek Chub	26	3.26	40
Daphnia	30	3.40	60
Mayfly	49	3.89	80
Fathead	64	4.16	100



PLOTTING POINTS

ln LC_{50}	PERCENTILE
3.26	40
3.40	60
3.89	80
4.16	100

GRAPHICAL DERIVATION OF COPPER CRITERIA
BASED ON RESIDENT SPECIES TOXICITY TESTS

FIGURE 2

The lowest two ln LC50 values fall in the interval 3.03 to 3.28. The geometric mean ln LC50 of 3.26 represents the fortieth percentile. The next interval, 3.28 to 3.53, includes the ln LC50 for Daphnia only. The table of plotting points shown in Figure 1 is generated in this manner.

Once the points are plotted, a straight line is drawn through the two points representing the most sensitive species in the array. Since the derived criterion is intended to protect all but the most sensitive 5 percent of resident aquatic life, a line drawn at the fifth percentile will indicate the concentration which should not be exceeded in order to protect 95 percent of aquatic life in the receiving water. In this illustration the fifth percentile corresponds to an ln LC50 of 3.0, or a copper concentration of 20.1 ug/l.

C. Comparison of Criteria

The criteria developed in this investigation are compared in Table 3.

TABLE 3. ALTERNATIVE COPPER CRITERIA, ug/l

National Criteria	6.3
Ratio Method Modification	15.8
Resident Species Modification	20.1

It is interesting to note that the site water LC50 values indicated by the bioassays for copper, which cover a range from 26 ug/l to 64 ug/l (see Table 1), are higher (less restrictive) than the modified criteria values developed by the ratio method and the resident species method. While the modified values may appear rather high at first--especially in contrast to the highly conservative values that the States have become accustomed to, which reflect an application factor of 0.1--it must be remembered that they reflect both the mitigating effect of site water on a toxic pollutant, and they reflect a procedure which is designed to protect 95 percent of the aquatic life in a stream. On the other hand, the method in which an arbitrary application factor is used to adjust laboratory water bioassay results does not take site specific factors into consideration, and may be so conservative that a criterion cannot realistically be met.

The resident species recalculation procedure is labor intensive and could require a considerable effort to collect a sufficient number of fish or other organisms for the required site water toxicity tests. While the results of the resident species recalculation method might be felt to carry more weight than a ratio method modification performed with hatchery fish, the ratio method requires less manpower and less time to complete.

IV. SUMMARY AND CONCLUSIONS

Toxicity test results are the most important element in the site specific criteria modification protocol investigated in North Wilkesboro. While the benthic survey and the water chemistry analysis provide necessary insight into the biological health of the receiving water, and may point out problems that would not be reflected in the bioassays, it is the bioassay results which provide a basis for modifying a national criterion number to reflect local conditions.

The national criteria numbers are based on toxicity tests run in laboratory water and, thus, may not adequately represent site water effects. A site specific criterion is most easily developed by adjusting the national criterion number to reflect the differences observed in parallel sets of toxicity tests, one set using site water as dilution water, the parallel set run in conventional fashion using laboratory water as dilution water.

There is a pronounced site water effect seen in the bioassays which points to the conclusions that: 1) the toxicity of copper to aquatic life (fathead minnow and Daphnia) is mitigated by water from UT, and 2) that an adjustment to a less stringent copper criterion may therefore be justified.

Whether such an adjustment should be based directly on the analyses presented in this report would be controversial since the procedures followed were not strictly in accord with the criteria modification protocol under investigation. Whether or not to base an adjustment on the findings of this study would fall on the judgement of the State.

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